Research Article

Dietary exposure to brominated flame retardants correlates with male blood levels in a selected group of Norwegians with a wide range of seafood consumption

Helle K. Knutsen, Helen E. Kvalem, Cathrine Thomsen, May Frøshaug, Margaretha Haugen, Georg Becher, Jan Alexander and Helle M. Meltzer

Norwegian Institute of Public Health, Oslo, Norway

This study investigates dietary exposure and serum levels of polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecane (HBCD) in a group of Norwegians (n = 184) with a wide range of seafood consumption (4 – 455 g/day). Mean dietary exposure to Sum 5 PBDEs (1.5 ng/kg body weight/day) is among the highest reported. Since concentrations in foods were similar to those found elsewhere in Europe, this may be explained by high seafood consumption among Norwegians. Oily fish was the main dietary contributor both to Sum PBDEs and to the considerably lower HBCD intake (0.3 ng/kg body weight/day). Milk products appeared to contribute most to the BDE-209 intake (1.4 ng/kg body weight/day). BDE-209 and HBCD exposures are based on few food samples and need to be confirmed. Serum levels (mean Sum 7 PBDEs = 5.2 ng/g lipid) and congener patterns (BDE-47 > BDE-153 > BDE-99) were comparable with other European reports. Correlations between individual congeners were higher for the calculated dietary exposure than for serum levels. Further, significant but weak correlations were found between dietary exposure and serum levels for Sum PBDEs, BDE-47, and BDE-28 in males. This indicates that other sources in addition to diet need to be addressed.

Keywords: Blood / Exposure / Food / Hexabromocyclododecane / Polybrominated diphenyl ether

Received: March 12, 2007; revised: June 20, 2007; accepted: June 22, 2007

1 Introduction

Brominated flame retardants (BFRs), such as polybrominated diphenyl ethers (PBDEs) and hexabromocyclododecane (HBCD), have been used extensively in consumer products such as foam cushions, carpets, and electronic equipment in Norway as in the rest of the western world. Several of the BFRs are lipophilic, persist in the environment over long periods of time, and have the potential to bioaccumulate, and may thereby reach high levels in the food chains, particularly in the aquatic food chain.

Human exposure routes to BFRs have not been established, but both diet and house dust are suspected routes of exposure [1, 2]. The relative impact of these exposure routes may differ in different parts of the world, depending on usage of commercial mixtures of BFRs and different dietary habits. PBDEs have been found at considerably lower levels in human blood and breast milk from Europe compared to those found in North America. Little is known about the dietary exposure in the general Norwegian population or in subgroups with special dietary habits.

Fish generally contains higher levels of PBDEs and HBCD than other food groups. Norwegians on average have a high fish consumption; the consumption is among the highest reported in Europe [3]. This study presents dietary exposure to PBDEs and HBCD in a group of Norwegians with a wide range of fish consumption. Furthermore, correlations between dietary exposures and blood levels are studied in the same group. Some of the food items with the highest content of persistent organic pollutants are eaten on a seasonal basis and often only by a small part of the population. The food frequency questionnaire (FFQ) used in the present study aimed to register food consumption during the last 12 months, including food that people gathered and caught themselves [4]. In order to investigate the range of

Correspondence: Dr. Helle K. Knutsen, Division for Environmental Medicine, Department of Food Toxicology, POB 4404 Nydalen, 0403 Oslo, Norway
E-mail: helle.knutsen@fhi.no
Fax: +47-22-042243

Abbreviations: BFRs, brominated flame retardants; **bw**, body weight; **FD**, food diary; **FFQ**, food frequency questionnaire; **HBCD**, hexabromocyclododecane; **LB**, lower bound; **MB**, medium bound; **PBDE**, polybrominated diphenyl ether; **UB**, upper bound

possible dietary PBDE and HBCD exposures, we selected a group of participants with a wide range of consumption of food known to contain relatively high background levels of pollutants, and included a reference group covering the general population.

The purpose of the study was (i) to calculate dietary exposure to PBDEs and HBCD in a group of Norwegians with a wide range of seafood consumption and in a reference subgroup with average seafood consumption, (ii) to find the relative contribution to exposure from different food groups, (iii) to characterize the serum levels, and (iv) to investigate correlations between dietary exposure and PBDE levels in serum.

2 Materials and methods

2.1 Study group

Participants were recruited from "The Norwegian Fish and Game Study." This study, with 5499 participants from coastal and inland areas in Norway, included a semi-quantitative FFQ with questions on consumption of seafood (different fish species living in saltwater and freshwater, fish liver, crustaceans, seagull eggs) and game. None of the participants was living in areas with known contamination of persistent organic pollutants or heavy metals above a level that can be considered as background. Based on concentrations of different contaminants in the actual foods, a rough estimate of the individual intake of PCBs, dioxins, mercury, and cadmium was established for the participants in this study. Based on their estimated high exposure to dioxins and PCBs, mercury or cadmium, 420 subjects were invited to participate. In order to constitute a reference group, 282 subjects were drawn randomly from the remaining population and invited to participate. Of a total of 702 subjects invited, 193 gave informed consent, of which 78 belonged to the reference group. All participants answered a 12-page semi-quantitative FFQ. The logistics of the study, carried out in the spring of 2003 and including the gathering of blood and urine samples, is described in more detail elsewhere (H.M. Meltzer *et al.*, manuscript in preparation). The reference group ($n = 78$) did not have significantly different means from all participants in the Norwegian Fish and Game Study ($n = 5499$) with respect to place of living (coast/inland), age, gender, body weight (bw), body mass index (BMI), smoking, and total fish consumption. The reference group can be considered representative for the Fish and Game Study, which again was country representative.

2.2 Dietary exposure

Dietary exposure was assessed using a 12-page semi-quantitative FFQ consisting of 340 questions organized into 40 groups according to the Norwegian meal pattern. Three of the groups had questions about dietary patterns and 23 about the use of 255 specific food items, with the goal of monitoring energy intake, nutrients, non-nutrients, foods, and food groups. The frequency of consumption was given *per* day, *per* week, and/or *per* month depending on the food item and covered consumption over the last 12 months. The FFQ was originally developed for use in the Norwegian Mother and Child Cohort Study and is described in detail in a report by Meltzer *et al*. [4]. The questionnaires were optically read and food frequencies were converted into consumption (grams/day) by multiplying with standard gender-specific portion sizes. For contaminant calculations FoodCalc was used (available at http://www.ibt.ku.dk/jesper/foodcalc). Nine of the 193 participants were excluded from analysis of dietary exposure on the basis of unlikely energy intakes (less than 1000 or more than 4000 kcal/day), leaving 184 participants for the final dietary exposure analyses. Of these, 73 belonged to the reference group.

The FFQ has been subject to a thorough validation in a pregnancy subcohort, where the reference method was a 4-day weighed food diary (FD) in addition to blood and urine biomarker [5]. The average correlation coefficient between the FFQ and FD for daily intake was 0.48 for foods and 0.36 for nutrients, which is comparable to other pregnancy validation studies. The FFQ was able to distinguish between high and low consumers (Q1 *vs*. Q5) for all the FD estimates as well as for urinary nitrogen, urinary iodine, plasma 25(OH)D, and serum folate. The degree of misclassification was small, while around two-thirds of the subjects were classified into the same or adjacent quintile according to the FFQ and reference measures. Classification into the same or adjacent quintile according to the two dietary methods was similar to that reported for a questionnaire used for assessment of diet in pregnant women in Finland [6] and for the questionnaire used in the Danish National Birth Cohort [7]. In general, correlation coefficients between a test method and reference method tend to be lower in pregnancy cohorts than in non-pregnant, adult populations, because of the large intra-individual variations due to pregnancy complications that may influence eating habits, *e.g.*, nausea, vomiting, constipation, and bed rest.

A database was built comprising available concentrations of PBDEs and HBCD in Norwegian foods. For the majority of food items, data on levels of the three major stereoisomers of HBCD, namely, α-, β-, and γ-HBCD, were available. For the rest, only total HBCD was determined by GC-MS. The samples collected were from the period 2002 to 2006 with a few older samples dating back to 1995 for seagull eggs and wild trout in order to make the database more complete for all congeners. Data were obtained from the Norwegian Food Safety Authority, the National Institute of Nutrition and Seafood Research, the Norwegian Institute for Water Research, the Norwegian Institute of Public Health, the Norwegian Pollution Control Authority, and the Norwegian Veterinary Institute as well as from a conference

proceeding [8]. Most analytical data are on composite samples except for fish and seagull eggs.

Dietary exposure to BFRs was calculated using mean and median levels of the individual congeners calculated as lower bound (LB; analyzed concentrations below LOQ are set to 0), medium bound (MB; analyzed concentrations below LOQ are set to ½ LOQ), and upper bound (UB; analyzed concentrations below LOQ are set equal to LOQ). For food items for which no analytical data were available, contamination levels were estimated from similar foods adjusted by the fat%.

2.3 Demographic information

Of the 184 remaining participants (83 male, 101 female) after exclusion of those with unlikely energy intakes, the average age was 54 years (range 21 – 80 years) and the average bw was 74 kg (range 36 – 115 kg). Among the 73 participants remaining in the reference group (32 male, 40 female), average age was 51 years (range 21 – 75 years), and average bw was 76 kg (range 53 – 115 kg).

2.4 Serum analyses

Concentrations of PBDEs were determined in 126 participants of which 44 belonged to the reference group. Serum sample volume was the selection criterion for serum analyses. All participants having serum sample volumes above 4.5 mL were analyzed, except for the eight with highest sample volume below 4.5 mL.

The individual PBDE congeners BDE-28, -37, -47, -85, -99, -119, -153, -154, -181, and -183, as well as HBCD, were from Cambridge Isotope Laboratories (Andover, MA, USA). BDE-18, -51, -103, and -138 were obtained from AccuStandard Inc. (New Haven, CT, USA), while BDE-100 and -156 was from Wellington Laboratories (Guelph, Ontario, Canada). All solvents used were of pesticide grade from SDS (Peypin, France), and sulfuric acid, silica gel, and sodium sulfate were from Merck (Darmstadt, Germany). The polychlorinated biphenyl CB-207 (AccuStandard Inc.) was used for calculation of the absolute recovery.

The serum samples were extracted using manual solid phase extraction on columns of Isolute 101, a highly cross-linked polystyrene-divinylbenzene polymer (200 mg from International Sorbent Technology, Mid Glamorgan, UK) after minor modifications of a previously described method [9]. An additional clean-up on sulfuric acid-silica columns was performed. The extracts were analyzed using GC-MS (6890/5973, Agilent, Avondale, PA, USA) equipped with a 7683 series autosampler according to a previously described method [9]. In brief, the separation was performed on a DB-5MS column (30 m length, 0.25 mm id, 0.25 µm film thickness, Agilent Technologies Inc., CA, USA) connected to a deactivated retention gap of 1.5 m length and 0.32 mm id fused silica (Agilent). The MS was operated in electron capture negative ionization mode (ECNI) using methane (99.99%, Aga, Norway) as buffer gas. The PBDEs were monitored on *m/z* 79 and *m/z* 81 and the recovery standard (CB-207) on *m/z* 464. All measured ions were used for identification and quantification. The calibration solutions covered the concentration range 0.6 – 120 pg PBDEs/g serum and 2.4 – 6000 pg HBCD/g serum. Internal standard calibration was used for quantification as described elsewhere [10]. BDE-156 was used as internal standard for semi-quantitative determination of HBCD. The LOD was about 0.6 pg/g serum (~0.1 ng/g lipids) for the PBDEs, based on the lowest level in the calibration curve. Twenty procedural blank samples were included in the analysis series, BDE-47 and BDE-99 were found in almost all, at levels in accordance with those found in procedural blanks in other series of serum analysis performed at our laboratory. Due to lower absolute recovery and less GC-MS response, the analysis uncertainty of the procedural blanks (water) was higher compared to the serum samples. The mean blank level exceeded the level found in about one-fourth of the samples, indicating that the actual blank level was lower. The concentrations of BDE-47 and BDE-99 were thus corrected by subtracting half of the blank level from all the individual measurements. The LOQ was set to a level equal to LOD for the other PBDEs. Compounds not detected were assigned the value 0 ("LB" approach). The uncertainty of the analysis was found to be about 25%.

The lipids were determined enzymatically at the National Hospital of Norway (Oslo, Norway) and the total lipid content of the samples calculated according to the method described by Grimvall *et al.* [11].

2.5 Statistical analysis

Dietary intake and serum concentrations of PBDEs and HBCD were not normally distributed. Hence these data are presented as medians, min, and max in addition to the 95th percentile. To be able to weigh our dietary intake and serum concentrations data against data from other countries, mean values of intake and serum concentrations are also presented. Spearman's *rho* was used to calculate correlations and the Mann – Whitney *U*-test was used to test for differences between groups. All p values below 0.05 were considered statistically significant. Statistical analyses were performed using SPSS, version 14.0.2.

3 Results and discussion

3.1 Levels in food

Figure 1 shows average LB levels of Sum 5 PBDEs, BDE-209, and HBCD in Norwegian foods, whereas data in Table 1 present congener-specific average LB and UB levels and the number of analyses on which exposure calculations are based. The congeners comprising Sum 5 PBDEs

Table 1. Concentrations of PBDEs and HBCD in foods used in exposure calculations. Data are shown as ng/g fresh weight; *n* = number of analyses

Food	fat%	PBDE-28			PBDE-47			PBDE-99			PBDE-100			PBDE-153			PBDE-154			PBDE-183			PBDE-209			α-HBCD[a)]			β-HBCD			γ-HBCD		
	mean	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*	LB	UB	*n*
Fish, fish products, shell fish and marine oils																																		
Farmed trout	19.5	0.073	0.073	3	1.130	1.130	3	0.273	0.273	3	0.217	0.217	3	0.060	0.060	3	0.120	0.120	3	0.007	0.013	3	0.303	0.303	3	0.653	0.653	3	0.020	0.020	3	0.053	0.053	3
Wild trout	2.1	0.020	0.020	1	0.298	0.298	13	0.389	0.389	14	0.160	0.160	1	0.049	0.050	14	0.080	0.080	1	0.010	0.010	1	0.080	0.080	1									
Herring	11.4	0.043	0.043	5	1.009	1.009	5	0.070	0.070	5	0.143	0.143	5	0.016	0.016	5	0.048	0.048	5	0.000	0.003	4				0.334	0.584	4						
Sardines	29.1	0.038	0.038	4	0.755	0.755	4	0.219	0.219	4	0.149	0.149	4	0.032	0.032	4	0.042	0.042	4	0.005	0.013	4	0.423	0.473	4	0.622	0.622	3	0.000	0.129	3	0.011	0.206	3
Farmed salmon	14.8	0.102	0.102	26	1.458	1.458	26	0.239	0.239	26	0.262	0.262	26	0.042	0.044	26	0.100	0.100	26	0.000	0.003	6				0.128	0.545	6						
Halibut	11.0	0.210	0.210	1	4.680	4.680	1	0.150	0.150	1	0.440	0.440	1	0.070	0.070	1	0.190	0.190	1															
Mackerel	25.7	0.059	0.059	7	0.723	0.723	7	0.262	0.262	7	0.174	0.174	7	0.057	0.057	7	0.078	0.078	7	0.028	0.032	6	0.958	0.965	4	0.493	0.827	6	0.033	0.078	4	0.040	0.135	4
Pike	0.12	0.000	0.010	1	0.120	0.120	1	0.090	0.090	1	0.030	0.030	1	0.010	0.010	1	0.010	0.010	1	0.000	0.010	1	0.000	0.120	1									
Perch	0.43	0.005	0.013	4	0.495	0.495	4	0.188	0.188	4	0.138	0.138	4	0.043	0.043	4	0.063	0.063	4	0.000	0.010	4	0.010	0.045	4									
Cod roe	6.8	0.010	0.014	4	0.210	0.210	4	0.013	0.013	4	0.034	0.034	4	0.000	0.019	4	0.010	0.015	4	0.000	0.011	4	0.354	0.354	4	0.000	0.049	3	0.000	0.027	3	0.000	0.044	3
Cod liver	55.7	0.341	0.341	2	6.909	6.909	2	0.096	0.096	2	0.769	0.769	2	0.019	0.019	2	0.397	0.397	2	0.000	0.006	1				3.609	3.609	1						
Sardines	29.1	0.038	0.038	4	0.755	0.755	4	0.219	0.219	4	0.149	0.149	4	0.032	0.032	4	0.042	0.042	4	0.005	0.013	4	0.423	0.473	4									
Caviar	36.4	0.000	0.010	2	0.070	0.070	2	0.006	0.011	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2	0.117	0.117	2	0.000	0.015	2	0.000	0.100	2	0.000	0.140	2
Smoked salmon	11.1	0.033	0.033	5	0.560	0.560	5	0.117	0.117	5	0.114	0.114	5	0.025	0.025	5	0.057	0.057	5	0.000	0.010	5	0.164	0.164	5	0.241	0.241	5	0.000	0.042	5	0.007	0.076	5
Mackerel in tomato sauce	19.5	0.023	0.029	4	0.339	0.339	4	0.177	0.177	4	0.079	0.082	4	0.019	0.036	4	0.028	0.038	4	0.000	0.012	4	1.064	1.108	4	0.534	0.534	3	0.000	0.094	3	0.009	0.153	3
Cod liver roe pâté, old recipe	26.9				5.321	5.321	2																											
Cod liver roe pâté, new recipe	31.2	0.139	0.139	6	2.203	2.203	6	0.049	0.049	6	0.307	0.307	6	0.004	0.014	6	0.096	0.096	6	0.000	0.010	6	1.397	1.434	6	0.483	0.483	5	0.063	0.109	5	0.000	0.157	5
Shrimp	1.2	0.000	0.011	3	0.016	0.016	3	0.000	0.011	3	0.000	0.010	3	0.000	0.014	3	0.000	0.011	3	0.000	0.011	3	0.034	0.079	3	0.005	0.012	2	0.000	0.009	2	0.000	0.011	2
Crab	9.2	0.017	0.020	3	0.360	0.360	3	0.237	0.237	3	0.087	0.090	3	0.177	0.187	3	0.043	0.050	3	0.000	0.010	1	0.120	0.120	1	0.000	0.080	1	0.000	0.070	1	0.000	0.130	1
Blue mussel	2.1				0.080	0.080	1	0.040	0.040	1	0.020	0.020	1	0.020	0.020	1	0.020	0.020	1															
Cod liver oil	100.0	0.225	0.284	9	2.925	2.927	22	0.455	0.506	22	0.302	0.333	9	0.076	0.098	9	0.271	0.296	9	0.000	0.082	9				0.000	5.000	5						
Fish oil	100.0	0.206	0.259	10	3.635	3.636	25	0.752	0.820	25	0.304	0.332	10	0.090	0.110	10	0.253	0.276	10	0.000	0.077	10				0.000	5.000	6						
Meat, dairy products, eggs																																		
Lamb liver	6.1							0.006	0.006	1	0.005	0.005	1	0.002	0.002	1	0.007	0.007	1															
Pork liver	3.4				0.014	0.014	1	0.016	0.016	1	0.003	0.003	1	0.003	0.003	1	0.002	0.002	1	0.001	0.001	1												
Bovine liver	2.6	0.000	0.000	1	0.008	0.008	1	0.008	0.008	1	0.004	0.004	1	0.000	0.000	1	0.001	0.001	1															
Bovine meat	17.8	0.000	0.010	2	0.019	0.026	3	0.014	0.016	5	0.000	0.010	2	0.003	0.004	6	0.002	0.007	4	0.004	0.011	3	0.074	0.074	2	0.000	0.020	2	0.000	0.015	2	0.000	0.025	2
Sheep meat	19.4	0.002	0.007	2	0.047	0.052	2	0.063	0.065	4	0.039	0.041	4	0.007	0.009	4	0.005	0.007	4	0.007	0.012	2	0.092	0.092	1	0.000	0.050	1	0.000	0.040	1	0.000	0.060	1
Pork meat	24.8	0.001	0.004	3	0.050	0.050	5	0.044	0.044	5	0.008	0.010	5	0.012	0.014	5	0.005	0.007	5	0.026	0.029	4	0.091	0.091	1	0.021	0.021	1	0.000	0.010	1	0.007	0.007	1
Sausage	16.4	0.000	0.010	2	0.034	0.034	2	0.047	0.047	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2	0.070	0.076	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2
Chicken meat	3.3				0.010	0.010	3	0.010	0.010	3	0.003	0.003	3	0.002	0.002	3	0.002	0.002	3	0.001	0.001	3												
Liver pâté	21.1	0.000	0.010	2	0.092	0.092	3	0.117	0.117	3	0.020	0.020	3	0.021	0.021	3	0.010	0.013	3	0.009	0.013	3	0.082	0.082	2	0.019	0.019	2	0.003	0.011	2	0.030	0.041	2
Dairy products	52.1	0.004	0.010	8	0.058	0.059	12	0.058	0.058	13	0.012	0.015	11	0.006	0.013	13	0.003	0.008	13	0.005	0.010	13	0.974	1.029	7	0.018	0.223	8	0.016	0.171	8	0.015	0.268	8
Hen egg	9.8	0.000	0.010	6	0.038	0.038	11	0.038	0.038	11	0.018	0.019	11	0.007	0.012	11	0.006	0.011	11	0.002	0.008	11	0.183	0.183	6	0.095	0.116	6	0.011	0.040	6	0.008	0.053	6
Seagull egg	9.4	1.079	1.079	22	41.518	41.518	26	7.427	7.427	26	6.417	6.417	22	1.727	1.727	22	1.429	1.429	22	3.099	3.099	13	1.964	1.964	16	6.620	6.620	20						
Reindeer meat	2.3							0.002	0.002	4				0.001	0.001	5	0.000	0.000	5	0.001	0.001	2												
Various																																		
Vegetable oil	100.0	0.000	0.010	2	0.010	0.015	2	0.034	0.037	3	0.000	0.010	2	0.006	0.010	3	0.001	0.007	3	0.004	0.010	3	0.225	0.225	2	0.000	0.125	2	0.000	0.110	2	0.000	0.145	2
Ice cream	10.3	0.000	0.010	2	0.014	0.014	2	0.015	0.015	2	0.000	0.010	2	0.005	0.010	2	0.000	0.010	2	0.010	0.015	2	0.304	0.304	2	0.000	0.010	2	0.000	0.010	2	0.000	0.010	2
Biscuit	14.0	0.000	0.010	7	0.007	0.013	7	0.007	0.012	7	0.000	0.010	7	0.002	0.013	7	0.000	0.010	7	0.001	0.010	7	0.444	0.444	7	0.009	0.071	7	0.008	0.052	7	0.015	0.090	7
Banana	0.1	0.000	0.010	1	0.000	0.010	1	0.000	0.010	1	0.000	0.010	1	0.000	0.010	1	0.000	0.010	1	0.000	0.010	1	0.000	0.034	1	0.000	0.010	1	0.000	0.010	1	0.000	0.020	1

a) If HBCD stereoisomers were not determined separately, sum HBCD was set as α-HBCD

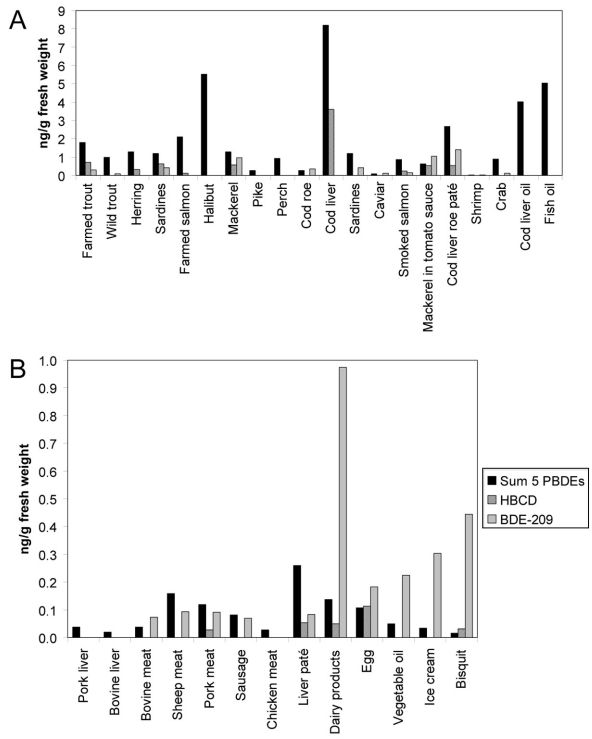

Figure 1. Occurrence of brominated flame retardants in Norwegian food. Please note different scaling of the axes in the two panels. A: Levels in fish and other seafood. B: Levels in meat, dairy products (milk, yoghurt, cheese, butter), eggs, and vegetable fat.

(47, 99, 100, 153, and 154) were determined in practically all samples, and the highest concentrations were found in sea gull eggs (not included in Fig. 1 due to scaling, see Table 1), fish liver, and fish fillet. BDE-47 was the most abundant congener in fish (Table 1). In hen eggs and meat, BDE-99 was equally abundant as BDE-47, although at a substantially lower level than in fish. Of samples analyzed for BDE-209, the highest levels were found among the above-mentioned food items and in addition in dairy products, which include milk, cheese, and butter. Among the HBCD stereoisomers, α-HBCD was definitely the most abundant one, and again the highest concentrations were found in seagull eggs, cod liver, and fish.

Sum 5 PBDE levels measured in Norwegian foods were by and large within the range as those reported from Finland [12], Sweden [13], Spain [14, 15], Belgium [16], Japan [1] and lower than those reported from the USA [17, 18]. Very few studies have been published on BDE-209 levels in foods, but based on the limited analyses in the present study, the levels seem higher in Norway than in the USA and Spain [15, 17]. Knowledge of HBCD levels in foods is very scarce, but levels in fish in the present study seem to be within the range found in various environmental studies in Europe [19].

3.2 Calculated dietary exposures

Among all the participants, including the reference group, the mean and median exposure to Sum 5 PBDEs (sum of BDE-47, -99, -100, -153, and -154) based on LB occurrence levels in food were 1.4 and 1.1 ng/kg bw/day, respectively (Table 2). The mean dietary exposure of Sum 5 PBDEs consisted of 69% BDE-47, 14% BDE-99, 11% BDE-100, 4% BDE-154, and 2% BDE-153. The contribution from BDE-47 to Sum 5 PBDE intake in the present study is higher than reported in the total diet baskets from Finland (47% BDE-47) and Sweden (52% BDE-47), perhaps due to different diet compositions in the Nordic countries [12, 13]. The calculated exposure to BDE-209 was 1.5 (mean) and 1.4 (median) ng/kg bw *per* day, which is higher than the exposure both to Sum 5 PBDEs and Sum 7 PBDEs (Sum 5 PBDEs + BDE-28 and BDE-183). The range of exposures was wider for Sum 5 PBDEs than for BDE-209. Dietary exposure to HBCD, with a mean of approximately 0.3 ng/kg bw *per* day, was considerably lower than that to the Sum 5 PBDEs. Dietary exposures to PBDEs and HBCD were skewed, with a long "tail" on the high end intake as reflected in the fact that the mean levels were 1.1- to 1.3-fold higher than the median levels for Sum 5 PBDEs, BDE-209, and HBCD.

The dietary intake assessments for the 184 participants constituting the study group are not representative for the general population in Norway, but show the range of possible dietary exposures among Norwegians with both average and high consumption of fish and other seafood, including cod liver, cod liver oil, and seagull eggs. In this group, the mean dietary exposure to Sum 5 PBDEs (108 ng/day; LB) was higher than those previously reported from Sweden (51 ng/day and 27 ng/day; MB) [13, 20], Finland (44 ng/day; MB) [12], Spain (82 ng/day; LB, 38.5 ng/day; UB) [14, 15], UK (107 ng/day; LB) [21] and Belgium (23 ng/day; LB) [16]. In the USA dietary PBDE exposure, including BDE-209, was recently found to be 84 ng/day in males and 63 ng/day in females [17]. In spite of an approximately ten times higher body burden of PBDEs in North America, it was surprising that the US dietary exposure levels do not exceed intakes of Sum 5 PBDEs in Europeans [22]. It has recently been indicated that higher concentrations of PBDEs in indoor dust from North America *vs*. Europe provide a likely rationale for the order-of-magnitude higher body burdens in North Americans than in Europeans [23]. A report from the UK Committee on Toxicology (COT) indicated that mean UB HBCD exposure in the UK is four to fivefold higher than in the present study [24]. To the best of our knowledge there are no other reports on dietary exposure to HBCD.

In the reference group, the mean dietary exposures to Sum 5 PBDEs, BDE209, and HBCD were quite similar to the median exposures among all the participants. However, the ranges of intakes were narrower in the reference group

Table 2. Dietary PBDE (ng/kg bw/day) exposure based on mean LB concentrations in food

	All participants, including reference group, $n = 184$					Reference group, $n = 73$				
	Mean	Median	95-perc	Min	Max	Mean	Median	95-perc	Min	Max
BDE-28	0.05	0.04	0.13	0	0.35	0.04	0.02	0.10	0	0.16
BDE-47	0.97	0.68	2.63	0.05	5.29	0.69	0.44	1.97	0.05	2.51
BDE-99	0.19	0.16	0.47	0.04	0.76	0.16	0.13	0.35	0.04	0.42
BDE-100	0.15	0.11	0.41	0.01	0.85	0.11	0.08	0.29	0.01	0.41
BDE-153	0.03	0.02	0.09	0	0.14	0.03	0.02	0.07	0	0.09
BDE-154	0.06	0.05	0.15	0	0.30	0.04	0.03	0.12	0	0.15
BDE-183	0.02	0.01	0.10	0	0.15	0.01	0.01	0.07	0	0.12
BDE-209	1.52	1.38	2.91	0.50	4.62	1.39	1.22	3.04	0.50	3.86
Sum 5 PBDE[a)]	1.40	1.06	3.66	0.14	7.00	1.03	0.74	2.78	0.14	3.45
Sum 7 PBDE[b)]	1.47	1.11	3.84	0.14	7.36	1.08	0.77	2.94	0.14	3.63
HBCD[c)]	0.33	0.27	0.83	0.06	1.35	0.27	0.23	0.60	0.06	0.87

a) Sum BDE-47, BDE-99, BDE-100, BDE-153, BDE-154
b) Sum 5 PBDEs + BDE-28 and BDE-183
c) Sum α-HBCD, β-HBCD, γ-HBCD

Table 3. Spearman rank correlation coefficients between dietary exposures to BFRs, $n = 184$. Calculated exposures are based on mean LB levels in food. All correlations are significant at the 0.01 level (two-tailed)

	BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	BDE-209	Sum 5 PBDE[a)]	Sum 7 PBDE[b)]
BDE-47	0.98									
BDE-99	0.78	0.79								
BDE-100	0.97	0.99	0.84							
BDE-153	0.80	0.81	0.95	0.86						
BDE-154	0.99	0.98	0.81	0.98	0.83					
BDE-183	0.42	0.48	0.63	0.50	0.69	0.45				
BDE-209	0.50	0.46	0.70	0.50	0.58	0.51	0.41			
Sum 5 PBDE[a)]	0.95	0.97	0.79	0.97	0.82	0.96	0.49	0.43		
Sum 7 PBDE[b)]	0.98	0.99	0.84	0.99	0.86	0.98	0.51	0.51	0.97	
HBCD[c)]	0.80	0.81	0.79	0.86	0.78	0.81	0.41	0.53	0.87	0.83

a) Sum BDE-47, BDE-99, BDE-100, BDE-153, BDE-154
b) Sum 5 PBDEs + BDE-28 and BDE-183
c) Sum α-, β-, γ-HBCD

(Table 2). The calculated average dietary exposure among those in the smaller reference group (81 ng/day; LB) is also in the upper range among calculated exposures in other European countries. This is probably caused by the generally high fish consumption among Norwegians; the mean consumption of fish, fish, products and shellfish being 79 g/day in the whole group and 62 g/day in the reference group in the present study. The FFQ used in the present study is the same as the one used in the on-going Norwegian Mother and Child Cohort Study, except for omission of questions about dietary changes after pregnancy and inclusion of detailed questions on fish and game consumption [4]. The birth cohort questionnaire has recently been validated although not yet with respect to fish consumption [5]. It is well known that an over-representation of questions about one food item can lead to over-reporting on consumption of that food item. However, the high Norwegian fish consumption has been confirmed by other national food consumption surveys [25].

There were generally very good inter-congener correlations between dietary exposures to the different PBDEs, all being significant with $p < 0.01$ (Table 3). Because of the relative high abundance of BDE-47, both the Sum 5 PBDEs and Sum 7 PBDEs were highly dependent on the BDE-47 level. BDE-47 showed correlation coefficients between 0.79 and 0.99 with the other PBDEs included in Sum 5 PBDEs, but the correlation coefficients were only 0.48 and 0.46 with BDE-183 and 209, respectively. The lower correlation between BDE-47 and BDE-183 may be due to low levels of BDE-183, being detectable in only a few food items. The lower correlation with the very abundant congener BDE-209 may indicate other dietary sources for BDE-209 than for BDE-47 and the other congeners in Sum 5 PBDEs. The intake level of HBCD showed higher correlation with Sum 5 PBDEs and Sum 7 PBDEs exposure than with BDE-209 exposure.

Due to the low number of analyses for several food items it was not possible to determine whether the levels were

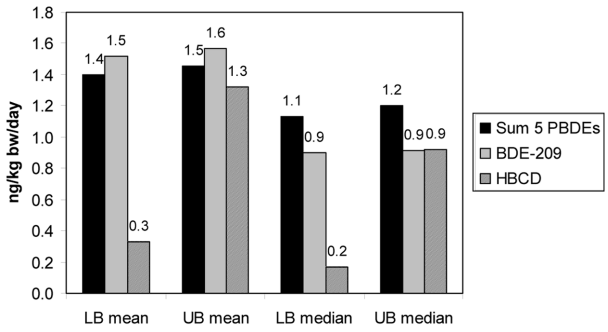

Figure 2. Mean dietary intake of Sum 5 PBDEs, BDE-209, and HBCD based on mean and median lower bound (LB) and upper bound (UB) levels in food.

normally distributed, and there were sometimes substantial differences between the LB and UB concentrations. In order to get an impression of the variability in exposure calculations that this could cause, we compared the mean and median exposures calculated on the basis of UB and LB median and mean levels in food (Fig. 2). There were almost no differences between LB and UB dietary exposures for Sum 5 PBDEs or for BDE-209, whereas for HBCD the calculated UB intake level was about fourfold higher than the LB intake. The differences between exposures based on mean and median levels in food were larger for BDE-209 than for HBCD and Sum 5 PBDEs. These results indicate a higher uncertainty for calculated HBCD exposure than for PBDE exposure, which is explained by a lower number of analyzed foods for HBCD and BDE-209. Hence, these calculations need to be confirmed in further studies.

3.3 Contribution from different food groups

Oily fish species were the dominating dietary source of Sum 5 PBDE exposure (Fig. 3). Dairy products contributed somewhat more than meat and lean fish. Oily fish species were also the main dietary source of HBCD exposure, followed by meat, hen eggs, and dairy products.

Fish liver, fish oil, and seagull eggs were eaten by 33%, 38%, and 13% of the 184 participants, respectively. These food items were important contributors to the intake of both Sum 5 PBDEs and HBCD among consumers of such food, and this affected the average exposure of the whole group. However, these foods were not important for exposure among participants with median dietary exposure (not shown). The main food groups contributing to exposure were similar in the reference group, but this group had lower average exposure from fish, fish liver, and seagull eggs (not shown).

Fish was also the main dietary source of PBDE in Sweden, Finland, and Belgium, even though their fish consumption is lower [12, 13, 16, 20]. In contrast, meat was the main dietary source of PBDE in the USA [17].

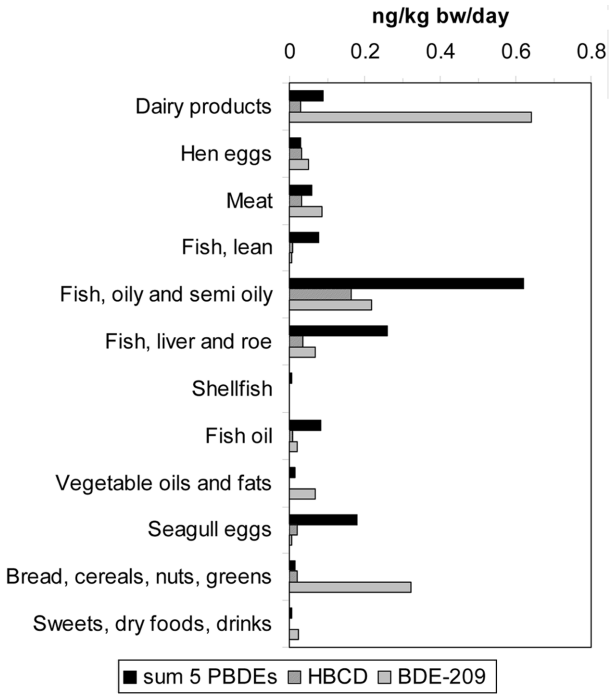

Figure 3. Contribution from different food groups to mean PBDE and HBCD exposure. Data are mean contribution for all participants in the study ($n = 184$). Dairy products include milk, cheese, yoghurts, and butter. Lean fish has lipid <2% and oily and semi oily fish has lipid >2%.

Confirming the suspicion that intake of BDE-209 and Sum 5 PBDEs may have different dietary sources, dairy products proved to be the most important dietary source of BDE-209 exposure, followed by bread, cereals, and fish, both among all participants (Fig. 3) and among the reference group (not shown). The calculated high contribution from bread and cereals was probably due to the relative high abundance in biscuits (Table 1). Since we lacked analytical data for bread, it was assumed that the lipid-based level in biscuits could be used as the corresponding level in bread. Bread is an important part of the Norwegian diet, and is consumed by many both for breakfast and lunch. BDE-209 levels measured in vegetable oil were also high (Table 1), but the amounts consumed as dressing, *etc*. did not add much to the total BDE-209 exposure.

3.4 Serum levels

The PBDE congeners BDE-28, BDE-37, BDE-47, BDE-85, BDE-99, BDE-100, BDE-119, BDE-153, BDE-154, and BDE-183 were determined in the serum of 125 participants. BDE-47 showed the highest mean serum levels, followed by BDE-153, BDE-99, BDE-154, and BDE-100, both in all samples analyzed and among samples from the smaller reference group (Table 4). The mean concentration of Sum 5 PBDEs was 4.96 ng/g lipid for all samples ana-

Table 4. Serum PBDE concentrations (LB, ng/g lipid)

PBDE	28	37	47	85	99	100	119	153	154[a]	183	Sum5[b]	Sum7[c]
All analyzed $n = 125$												
n detected	50	1	124	13	123	115	2	124	118	37		
Mean	0.11	0	2.00	0.02	0.65	0.43	0	1.36	0.52	0.09	4.96	5.16
% of total[d]	2	0	39	0	12	8	0	26	10	2	96	100
Median	0	0	1.44	0	0.43	0.34	0	1.10	0.39	0	3.96	4.01
95-perc.	0.40	0	6.94	0.15	2.18	1.14	0	3.95	1.47	0.51	13.7	13.7
Min	0	0	0	0	0	0	0	0	0	0	0.78	0.78
Max	1.09	0.09	13.2	0.24	5.20	2.47	0.17	9.56	2.90	0.72	19.1	20.2
Outlier[e]	4.56	0	87.7	1.40	10.3	11.2	1.43	6.31	1.08	0	116	121
Reference group $n = 44$												
Mean	0.12	0	2.38	0.04	0.92	0.47	0	0.97	0.38	0.12	5.13	5.37
% of total[d]	2	0	44	1	17	9	0	18	7	2	95	99
Median	0	0	1.51	0	0.54	0.35	0	0.93	0.31	0	3.62	3.74
95-perc.	0.57	0	8.96	0.21	3.84	1.16	0	1.82	1.22	0.54	14.5	15.3
Min	0	0	0.20	0	0	0	0	0	0	0	0.78	0.78
Max	0.95	0.09	13.2	0.24	5.20	1.23	0.10	1.95	1.50	0.59	19.1	20.2

a) BDE-154 co-eluted with BB-153
b) BDE-47, 99, 100, 153, 154
c) Sum 5 PBDEs + BDE-28 and BDE- 183
d) Total = Sum 7 PBDE + BDE-37, BDE-85 and BDE-119
e) One extremely high outlier (not in the reference group) was excluded from the summary statistics based on Dixon's Q test

Table 5. Spearman rank correlation coefficients between PBDE congeners in serum, $n = 125$

	BDE-28	BDE-47	BDE-85	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	Sum 5 PBDE
BDE-47	0.63[a]								
BDE-85	0.34[a]	0.40[a]							
BDE-99	0.22[b]	0.64[a]	0.44[a]						
BDE-100	0.54[a]	0.87[a]	0.40[a]	0.65[a]					
BDE-153	0.22[b]	0.32[a]	0.07	0.15	0.41[a]				
BDE-154	0.27[a]	0.23[a]	−0.05	0.02	0.26[a]	0.56[a]			
BDE-183	0.16	0.17	0.20[b]	0.14	0.18[b]	0.19[b]	0.17		
Sum 5 PBDE[c]	0.57[a]	0.88[a]	0.38[a]	0.65[a]	0.88[a]	0.63[a]	0.46[a]	0.18[b]	
Sum 7 PBDE[d]	0.60[a]	0.88[a]	0.38[a]	0.63[a]	0.87[a]	0.62[a]	0.45[a]	0.24[a]	0.99[a]

a), b) Correlations are significant at the 0.01 level[a] or 0.05 level[b] (two-tailed)
c) Sum BDE-47, BDE-99, BDE 100, BDE 153, BDE 154
d) Sum 5 PBDEs + BDE-28 and BDE-183

lyzed (with one outlier excluded) and 5.13 ng/g lipid in the reference group. These five congeners were detected in the serum of the majority of the participants and represented on average 96% of the total amount of PBDEs analyzed in serum. BDE-37, BDE-85, and BDE-119 were detected in only a few of the samples. The mean PBDE levels were in the same range as those reported previously in Norway [10] and elsewhere in Europe [20, 26–28], as well as in Japan [29, 30] and New Zealand [31], which is approximately ten-fold lower than in the North American population[22, 32].

In serum, the highest inter-congener correlations were seen between BDE-47, BDE-100, and BDE-99, with correlation coefficients between 0.87 and 0.64 (Table 5). The inter-congener correlations were quite similar for males and females (data not shown). BDE-47, being the most abundant congener, correlated significantly with all other PBDEs except for BDE-183. All congeners correlated with Sum 5 PBDEs and Sum 7 PBDEs. Generally, correlations between PBDEs in serum were lower than correlations between dietary exposures to PBDEs. This may be explained by differences in the kinetics for PBDE congeners, *e.g.*, congeners with longer half-lives showing higher concentrations. The blood levels of PBDEs may also be influenced by sources other than dietary ones.

Table 6. Spearman rank correlation coefficients between dietary exposures and serum PBDE levels. $n = 67$ for females (F) and 54 for males (M)

		Diet (ng/kg bw/day)[a]								
		BDE-28	BDE-47	BDE-99	BDE-100	BDE-153	BDE-154	BDE-183	Sum 5 PBDEs	Sum 7 PBDEs
Serum (ng/g lipid)										
BDE-28	F	0.00	0.01	−0.07	0.03	−0.08	−0.02	−0.04	0.01	0.00
	M	**0.35**[b]	**0.42**[b]	0.18	**0.39**[b]	0.18	**0.35**[b]	0.24	**0.41**[b]	**0.41**[b]
BDE-47	F	−0.06	−0.07	−0.16	−0.06	−0.14	−0.10	−0.07	−0.08	−0.08
	M	0.22	**0.30**[c]	0.21	**0.29**[c]	0.21	0.24	**0.30**[c]	**0.32**[c]	**0.31**[c]
BDE-99	F	**−0.34**[b]	**−0.34**[b]	**−0.33**[b]	**−0.32**[b]	**−0.31**[c]	**−0.37**[b]	**−0.26**[c]	**−0.36**[b]	**−0.36**[b]
	M	**−0.15**	**−0.15**	**−0.15**	**−0.14**	**−0.16**	**−0.15**	**−0.03**	**−0.14**	**−0.14**
BDE-100	F	−0.06	−0.06	−0.20	−0.07	−0.19	−0.09	−0.19	−0.09	−0.09
	M	0.14	0.19	0.16	0.19	0.15	0.17	0.20	0.21	0.21
BDE-153	F	0.06	0.08	0.09	0.06	0.10	0.07	0.12	0.07	0.07
	M	0.24	0.25	0.18	0.26	0.12	0.25	0.12	0.26	0.26
BDE-154	F	0.17	0.16	0.02	0.13	0.04	0.16	0.09	0.12	0.13
	M	**0.38**[b]	**0.39**[b]	**0.36**[b]	**0.41**[b]	**0.38**[b]	**0.39**[b]	0.13	**0.41**[b]	**0.40**[b]
BDE-183	F	−0.15	−0.18	−0.03	−0.17	−0.10	−0.14	−0.07	−0.15	−0.15
	M	−0.14	−0.12	−0.09	−0.10	−0.15	−0.14	−0.08	−0.10	−0.10
Sum 5 PBDEs	F	−0.08	−0.08	−0.17	−0.07	−0.16	−0.11	−0.10	−0.09	−0.09
	M	0.24	**0.27**[c]	0.20	**0.28**[c]	0.17	0.25	0.18	**0.29**[c]	**0.29**[c]
Sum 7 PBDEs	F	−0.06	−0.07	−0.16	−0.06	−0.16	−0.10	−0.10	−0.09	−0.08
	M	0.24	**0.29**[c]	0.20	**0.29**[c]	0.18	0.26	0.19	**0.30**[c]	**0.30**[c]

a) Based on mean LB concentrations in food. The outlier (male) was excluded from the correlations, but inclusion of this participant did not affect the outcome

b), c) Correlation is significant at the 0.01 level[b] or at the 0.05 level[c] (two-tailed)

3.5 Correlation between dietary exposure and serum levels

Among males, dietary intakes of BDE-28, BDE-47, BDE-154, and Sum 5 PBDEs and 7 PBDEs showed significant correlations with serum levels (Table 6). Levels of BDE 154 were biased by co-elution of the hexa-brominated biphenyl BB-153 in serum, and probably also in several food samples, and cannot be interpreted further at this point. The correlation coefficients for Sum 5 PBDEs and for BDE-47, the major component with respect to both dietary exposure and serum concentration, indicate that dietary exposures are important determinants for the serum concentrations. BDE-28, on the other hand, is a minor component in both blood (detected in one-third of the serum samples) and diet. The significant correlation coefficients were weak, pointing toward sources other than diet being important contributors to the exposure. In females, correlation coefficients were close to zero, with the exception of a negative correlation between dietary intake and the concentration of BDE-99 in serum. The striking and substantial differences between males and females may be explained by women excreting PBDEs in milk during the nursing period. The breastfeeding prevalence in Norway is among the highest in the world [33]. However, no connection between numbers of births, duration of breastfeeding, and serum PBDEs could be found among the women in the present study (data not shown).

HBCD levels have been determined semi-quantitatively in the serum samples, but did not correlate with dietary intake or with serum PBDEs (data not shown). Dietary exposure to BDE-209 did not correlate with serum levels of the analyzed PBDE congeners (data not shown). Serum levels of BDE-209 have not been analyzed in these samples.

Interestingly, the discrepancy between contribution of BDE-153 to Sum 5 PBDE dietary intakes (2%) and the mean BDE-153 contribution to Sum 5 PBDE in blood (26%) is striking. The higher percentage of BDE-153 in serum may be connected to dust exposure and/or accumulation of BDE-153 in blood due to longer half-life [34]. However, very little is known about half-lives of hexa-BDEs in humans, but they seem to be longer than those of the higher brominated congeners [35].

The significant, but weak, correlations between dietary PBDE exposure and serum levels indicate that other sources may contribute significantly to exposure in the Norwegian population. Due to the climate, much time is spent in-doors, and exposure from in-door air and dust may be important. However, the data on levels in food collected so far are limited and the calculated exposures have uncertainties attached. Interestingly, the same food intake data as used in the present study could very well predict PCB levels in blood [36]. In another study we investigated serum concentrations of PBDEs among consumers of fish caught in a Norwegian lake contaminated with PBDE and found that the serum concentrations correlate nicely with consumption

of fish [37]. Furthermore, a recent study in the Boston Massachusetts area reported positive association between PBDE concentration in breast milk and level in house dust as well as with reported dietary habits [38].

4 Concluding remarks

This study on dietary exposures to BFRs in Norway demonstrates weak, but significant, correlations between dietary exposure and serum concentrations of BDE-28, BDE-47, Sum 5 PBDEs, and Sum 7 PBDEs in men but not in women. BDE-47 constituted approximately two-thirds of the Sum 5 PBDE intake, and fish and other seafood were the main dietary sources. The study group included participants with a wide consumption range of seafood, and this probably explains why the calculated mean intake of Sum 5 PBDEs (1.4 ng/kg bw/day) was among the highest reported so far, keeping in mind that the contamination level in food was similar to the rest of Europe. The mean intake in the smaller reference group was lower, as expected (1.0 ng/kg bw/day), but still among the highest reported. This could probably be attributed to the high fish consumption among Norwegians. Interestingly, the mean dietary intake of BDE-209 (approximately 1.5 ng/kg bw/day) was higher than that of Sum 5 PBDEs, and milk products seemed to be the dominating source. Dietary exposure to HBCD was considerably lower, approximately 0.3 ng/kg bw/day, and came mainly from seafood. However, the exposure calculations on BDE 209 and HBCD were based on few analyses, and should be interpreted with care.

Levels and compositions of PBDEs in serum were in the same range as in the rest of Europe. Interestingly, inter-congener correlations were higher in diet than in serum. Combined with the generally weak correlations between dietary exposure and levels in serum, this clearly shows the need for further studies in order to clarify the impact of other sources of PBDE exposures, *e. g.*, in-door air and dust.

We express our thanks to the participants for answering extensive questionnaires and donating biological material. The study was supported financially by the Norwegian Food Safety Authority. We want to thank the Norwegian Food Safety Authority, the National Institute of Nutrition and Seafood Research, the Norwegian Institute for Water Research, the Norwegian Institute of Public Health, the Norwegian Pollution Control Authority, and the Norwegian Veterinary Institute for providing data on occurrence of BFRs in food.

The authors have declared no conflict of interest.

5 References

[1] Ohta, S., Ishizuka, D., Nishimura, H., Nakao, T. *et al.,* Comparison of polybrominated diphenyl ethers in fish, vegetables, and meats and levels in human milk of nursing women in Japan. *Chemosphere* 2002, *46*, 689 – 696.

[2] Harrad, S., Hazrati, S., Ibarra, C., Concentrations of polychlorinated biphenyls in indoor air and polybrominated diphenyl ethers in indoor air and dust in Birmingham, United Kingdom: implications for human exposure. *Environ. Sci. Technol.* 2006, *40*, 4633 – 4638.

[3] EFSA (European Food Safety Authority). Opinion of the Scientific Panel on Contaminants in the Food Chain on a request from the Commission to assess the health risks to consumers associated with exposure to organotins in foodstuffs. *EFSA* 2004.

[4] Meltzer, H. M., Brandsæter, A. L., Ydersbotn, T. A., Alexander, J. *et al.*, the Moba Dietary Support Group. Methodological challenges when monitoring the diet of pregnant women in a large cohort study; experiences from The Norwegian Mother and Child Cohort Study. *Matern. Child Nutr.* 2008, *1*, 14 – 27.

[5] Brandsæter, A. L., Haugen, M., Alexander, J., Meltzer, H. M., Validity of a new Food Frequency Questionnaire for pregnant women in the Norwegian Mother and Child Cohort Study (MoBa). *Matern. Child Nutr.* 2008, *1*, 28 – 43.

[6] Erkkola, M., Karppinen, M., Javanainen, J., Rasanen, L. *et al.*, Validity and reproducibility of a food frequency questionnaire for pregnant Finnish women. *Am. J. Epidemiol.* 2001, *154*, 466 – 476.

[7] Mikkelsen, T. B., Osler, M., Olsen, S. F., Validity of protein, retinol, folic acid and n-3 fatty acid intakes estimated from the food-frequency questionnaire used in the Danish National Birth Cohort. *Public Health Nutr.* 2006, *9*, 771 – 778.

[8] Betthune, C., Nielsen, J., Julshamn, K., Current levels of primary polybrominated diphenyl ethers (PBDEs) in Norwegian seafood. *Organohalogen Compd.* 2004, *66*, 3814 – 3819.

[9] Thomsen, C., Lundanes, E., Becher, G., A simplified method for determination of tetrabromobisphenol A and polybrominated diphenyl ethers in human plasma and serum. *J. Sep. Sci.* 2001, *24*, 282 – 290.

[10] Thomsen, C., Liane, V. H., Becher, G., Automated solid-phase extraction for the determination of polybrominated diphenyl ethers and polychlorinated biphenyls in serum-application on archived Norwegian samples from 1977 to 2003. *J. Chromatogr. B* 2007, *846*, 252 – 263.

[11] Grimvall, E., Rylander, L., Nilsson-Ehle, P., Nilsson, U. *et al.*, Monitoring of polychlorinated biphenyls in human blood plasma: methodological developments and influence of age, lactation, and fish consumption. *Arch. Environ. Contam. Toxicol.* 1997, *32*, 329 – 336.

[12] Kiviranta, H., Ovaskainen, M. L., Vartiainen, T., Market basket study on dietary intake of PCDD/Fs, PCBs, and PBDEs in Finland. *Environ. Int.* 2004, *30*, 923 – 932.

[13] Darnerud, P. O., Atuma, S., Aune, M., Bjerselius, R. *et al.*, Dietary intake estimations of organohalogen contaminants (dioxins, PCB, PBDE and chlorinated pesticides, e. g. DDT) based on Swedish market basket data. *Food Chem. Toxicol.* 2006, *44*, 1597 – 1606.

[14] Bocio, A., Llobet, J. M., Domingo, J. L., Corbella, J. *et al.*, Polybrominated diphenyl ethers (PBDEs) in foodstuffs: human exposure through the diet. *J. Agric. Food Chem.* 2003, *51*, 3191 – 3195.

[15] Gomara, B., Herrero, L., Gonzalez, M. J., Survey of polybrominated diphenyl ether levels in Spanish commercial foodstuffs. *Environ. Sci. Technol.* 2006, *40*, 7541 – 7547.

[16] Voorspoels, S., Covaci, A., Neels, H., Schepens, P., Dietary PBDE intake: a market-basket study in Belgium. *Environ. Int.* 2007, *33*, 93 – 97.

[17] Schecter, A., Papke, O., Harris, T. R., Tung, K. C. *et al.*, Polybrominated diphenyl ether (PBDE) levels in an expanded market basket survey of U.S. food and estimated PBDE dietary intake by age and sex. *Environ. Health Perspect.* 2006, *114*, 1515 – 1520.

[18] Huwe, J. K., Larsen, G. L., Polychlorinated dioxins, furans, and biphenyls, and polybrominated diphenyl ethers in a U.S. meat market basket and estimates of dietary intake. *Environ. Sci. Technol.* 2005, *39*, 5606 – 5611.

[19] Law, R. J., Allchin, C. R., De Boer, J., Covaci, A. *et al.*, Levels and trends of brominated flame retardants in the European environment. *Chemosphere* 2006, *64*, 187 – 208.

[20] Lind, Y., Darnerud, P. O., Atuma, S., Aune, M. *et al.*, Polybrominated diphenyl ethers in breast milk from Uppsala County, Sweden. *Environ. Res.* 2003, *93*, 186 – 194.

[21] Harrad, S., Wijesekera, R., Hunter, S., Halliwell, C. *et al.*, Preliminary assessment of U.K. human dietary and inhalation exposure to polybrominated diphenyl ethers. *Environ. Sci. Technol.* 2004, *38*, 2345 – 2350.

[22] Schecter, A., Papke, O., Tung, K. C., Joseph, J. *et al.*, Polybrominated diphenyl ether flame retardants in the U.S. population: current levels, temporal trends, and comparison with dioxins, dibenzofurans, and polychlorinated biphenyls. *J. Occup. Environ. Med.* 2005, *47*, 199 – 211.

[23] Harrad, S., Diamond, M., New directions: Exposure to polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyls (PCBs): Current and future scenarios. *Atmos. Environ.* 2006, *40*, 1187 – 1188.

[24] Committee on Toxicity (COT). Brominated Chemicals: UK Dietary Intakes. *Report* 2006, http://www.food.gov.uk/multimedia/pdfs/fsis1006.pdf.

[25] Norwegian Scientific Committee for Food Safety, A comprehensive assessment of fish and other seafood in the Norwegian diet. 2007.

[26] Ingelido, A. M., Ballard, T., Dellatte, E., di Domenico, A. *et al.*, Polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) in milk from Italian women living in Rome and Venice. *Chemosphere* 2007, in press.

[27] Naert, C., Piette, M., Bruneel, N., Van Peteghem, C., Occurrence of polychlorinated biphenyls and polybrominated diphenyl ethers in Belgian human adipose tissue samples. *Arch. Environ. Contam. Toxicol.* 2006, *50*, 290 – 296.

[28] Kalantzi, O. I., Martin, F. L., Thomas, G. O., Alcock, R. E. *et al.*, Different levels of polybrominated diphenyl ethers (PBDEs) and chlorinated compounds in breast milk from two U.K. Regions. *Environ. Health Perspect.* 2004, *112*, 1085 – 1091.

[29] Inoue, K., Harada, K., Takenaka, K., Uehara, S. *et al.*, Levels and concentration ratios of polychlorinated biphenyls and polybrominated diphenyl ethers in serum and breast milk in Japanese mothers. *Environ. Health Perspect.* 2006, *114*, 1179 – 1185.

[30] Eslami, B., Koizumi, A., Ohta, S., Inoue, K. *et al.*, Large-scale evaluation of the current level of polybrominated diphenyl ethers (PBDEs) in breast milk from 13 regions of Japan. *Chemosphere* 2006, *63*, 554 – 561.

[31] Harrad, S., Porter, L., Concentrations of polybrominated diphenyl ethers in blood serum from New Zealand. *Chemosphere* 2007, *66*, 2019 – 2023.

[32] She, J., Holden, A., Sharp, M., Tanner, M. *et al.*, Polybrominated diphenyl ethers (PBDEs) and polychlorinated biphenyls (PCBs) in breast milk from the Pacific Northwest. *Chemosphere* 2007, *67*, S307 – S317.

[33] Lande, B., Andersen, L. F., Baerug, A., Trygg, K. U. *et al.*, Infant feeding practices and associated factors in the first six months of life: the Norwegian infant nutrition survey. *Acta Paediatr.* 2003, *92*, 152 – 161.

[34] Staskal, D. F., Hakk, H., Bauer, D., Diliberto, J. J. *et al.*, Toxicokinetics of polybrominated diphenyl ether congeners 47, 99, 100, and 153 in mice. *Toxicol. Sci.* 2006, *94*, 28 – 37.

[35] Thuresson, K., Hoglund, P., Hagmar, L., Sjodin, A. *et al.*, Apparent half-lives of hepta- to decabrominated diphenyl ethers in human serum as determined in occupationally exposed workers. *Environ. Health Perspect.* 2006, *114*, 176 – 181.

[36] Kvalem, H. E., Knutsen, H. K., Thomsen, C., Haugen, M. *et al.*, The predictive value of dietary intake estimations on serum levels of some non-dioxin like PCBs. *Organohalogen Compd.* 2006, *68*, 1517 – 1520.

[37] Thomsen, C., Knutsen, H. K., Liane, V. H., Frøshaug, M. *et al.*, Consumption of fish from a contaminated lake strongly affects the concentrations of polybrominated diphenyl ethers and hexabromocyclododecane in serum. *Mol. Nutr. Food Res.* 2008, *52*, 228 – 237.

[38] Wu, N., Herrmann, T., Paepke, O., Tickner, T. *et al.*, Human exposure to PBDEs: Associations of PBDE body burdens with food consumption and house dust concentrations. *Environ. Sci. Technol.* 2007, *41*, 1584 – 1589.