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Dioxin-like polybrominated biphenyls (PBBs) and *ortho*-substituted PBBs in edible cod (*Gadus morhua*) liver oils and canned cod livers

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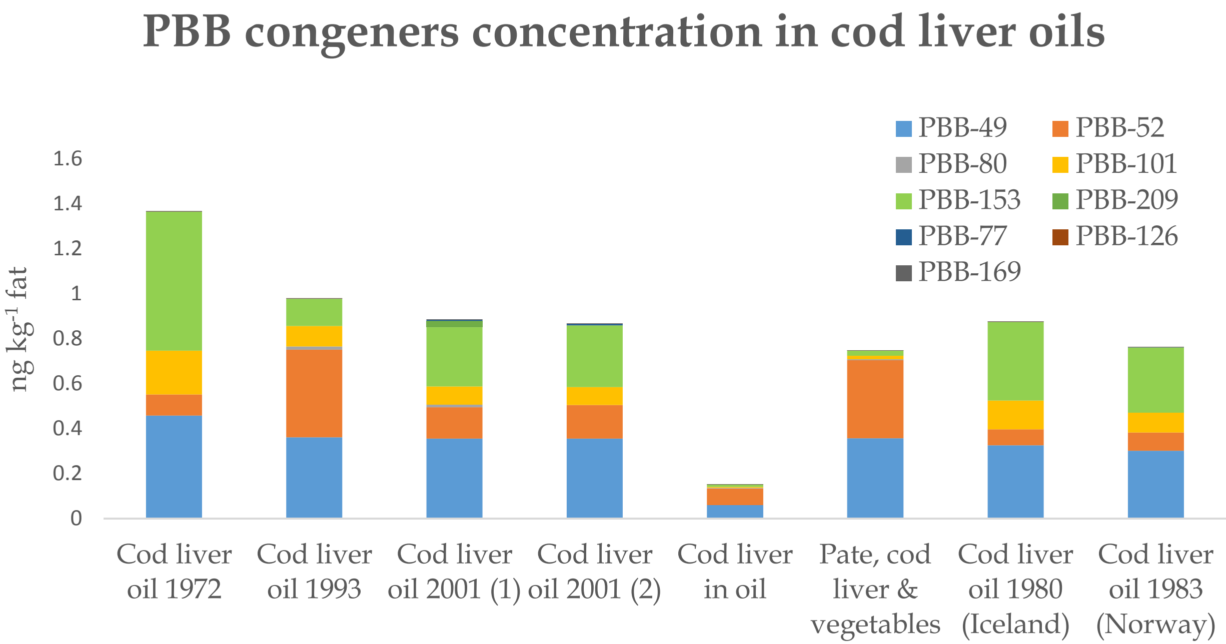
PBB congeners concentration in cod liver oils
PBB-49
PBB-52
PBB-80
PBB-101
PBB-153
PBB-209
PBB-77
PBB-126
PBB-169
ng kg-1 fat
0
0.2
0.4
0.6
0.8
1
1.2
1.4
1.6
Cod liver oil 1972
Cod liver oil 1993
Cod liver oil 2001 (1)
Cod liver oil 2001 (2)
Cod liver in oil
Pate, cod liver & vegetables
Cod liver oil 1980 (Iceland)
Cod liver oil 1983 (Norway)

Dioxin-like polybrominated biphenyls (PBBs) and *ortho*-substituted PBBs in edible cod (*Gadus morhua*) liver oils and canned cod livers

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Highlights

- PBBs were detected in all cod liver oils and canned liver products from 1972-2017
- *Ortho*-PBBs 49, 52, 101, 153 and non-*ortho* PBB 77 occurred in all samples
- PBB levels in cod liver oils peaked at the turn of the century
- During 1972-1993, contamination levels for Baltic Sea and North Atlantic were similar

Abstract

This study investigates the occurrence of polybrominated biphenyls (PBBs), a legacy flame retardant, in fishery products such as medicinal grade cod liver oils and canned liver products, sourced from the North Atlantic during 1972-2017. It also assesses the dietary and supplementary (the oils were commonly administered as dietary supplements to children and youth) intake of PBBs from these products. Summed *ortho*-PBB concentrations ranged from 770 to 1400 pg g^{-1} fat in the oils and from 99 to 240 pg g^{-1} whole weight in canned livers, with PBB-49, 52, 101 and 153 accounting for most of these levels. Among the more toxic non-*ortho*-PBBs, PBB-126 and PBB-169 were not detected, but PBB-77 concentrations ranged from 0.6 to 5.78 pg g^{-1} fat in the oils and 0.06 to 0.126 pg g^{-1} whole weight in canned livers. During 1972-1993, PBB contamination levels were similar for cod liver oils from the Baltic Sea and other North Atlantic regions, but over the timescale of the study, Baltic Sea products appear to show a decline in PBB concentrations. As PBB-77 was the only dioxin-like PBB detected in the samples, the corresponding supplementary (oils, 1972-2001) and dietary (cod liver from 2017) intakes were very low, at < 0.001 pg TEQ kg^{-1} bm d^{-1} (or < 0.01 pg TEQ kg^{-1} bm d^{-1} upper bound) for the sum of all the measured dioxin-like PBBs –four to six orders of magnitude lower than that arising from other dioxin-like contaminants that were shown to occur in these products, from earlier studies.

Keywords: Baltic Sea, dioxin-like, medicinal grade, fish oil, dietary intake, dietary supplements

1. Introduction

Polybrominated biphenyls (PBBs) are the brominated analogues of the more widely known polychlorinated biphenyls (PCBs). They were introduced as brominated flame retardants (BFRs) in the 1950s and commercially produced in the United States and later in Europe (UK, Germany and France). Like their chlorinated counterparts, the PBBs, they were also mass produced. These products had relatively high bromination levels, with approximately 76% for hexabromobiphenyls and 81–85% for octabromobiphenyl (OBB) and decabromobiphenyl products (chlorination levels in commercial PCB mixtures could be as low as 10%). Some OBB formulations were composed of 1.8% heptabromobiphenyl, 45.2% octabromobiphenyl, 47.4% nonabromobiphenyl, and 5.7% decabromobiphenyl, or 1.0% heptabromobiphenyl (HBB), 33.0% octabromobiphenyl, 60.0% nonabromobiphenyl and 6.0% decabromobiphenyl (DBB), while the commercial DBB was composed of 96.8% decabromobiphenyl, 2.9% nonabromobiphenyl, and 0.3% octabromobiphenyl (Di Carlo et al., 1978). It has been assumed that > 19 environmentally stable atropisomeric PBBs exist and some have been enantioselectively separated and identified (Berger et al., 2000).

In an unfortunate accident in Michigan, USA, in 1973, FireMaster BP-6 was mistakenly added to animal feed resulting in the contamination of thousands of cattle, pigs, sheep and up to 1.5 million chickens, as well as a huge amount of food that subsequently needed disposal. Some of the PBB contaminated foods entered the human food chain before the accident was identified. The investigations that followed from this incident revealed occupational exposure to volatilized or dust-borne PBBs to the personnel involved in the production of PBBs or the manufacture of PBB-containing products. PBB emissions from manufacturing sites into the surrounding air, water and sediments via sewers, and disposal as solid waste to landfills and soils near the plants has led to pollution of the environment and local areas (Di Carlo et al., 1978). Large scale manufacture has long since ceased and PBBs are now recognized as largely legacy contaminants in foods and the environment.

PBBs are also found as a byproduct in other BFRs such as polybrominated diphenyl ethers (PBDEs) (Hanari et al., 2006). Thermodynamically, they can be formed from precursors by *de novo* synthesis during the combustion of bromobenzenes (Saeed et al., 2015), and possibly also from the combustion of other products or waste that contain BFRs. The reported concentration of PBBs in air in the atmosphere near a municipal solid waste incinerator was 341 fg Nm^3 (149 - 556 fg Nm^3) Taiwan (Wang et al., 2010). However, as they were not manufactured or used in many countries, their sources at a regional level originate from relatively small emissions or leaching, etc. resulting from the international trade in goods containing PBBs and other BFRs, and combustion/disposal of such products. Additionally, as they are relatively stable chemicals, they can undergo long range atmospheric or marine transport. Reflecting this mobility, PBB congeners (IUPAC numbers: 15, 52, 153, 180 and 194) were recorded (Chao et al., 2014) at greater concentration (0.144 pg m^3; n = 2) in the atmosphere over a rural region in Taiwan than in the offshore oceanic atmosphere in early November 2012 (0.0265 pg m^3 with 100% contribution from PBB-15; n = 6). Some PBBs (PBB-15 and PBB-153, but not PBB-52, PBB-180 and PBB-194) were found in the range from 0.079±0.049 to 2.52±1.20 pg g^{-1} in soils (n=36) and from 1.65±0.99 to 1.74±1.79 pg g^{-1} in ornithogenic soils (with layer of indurated guano crust; n=18), and from 0.186 to 0.477 pg g^{-1} (n=4) in lichen sampled from the Ardley Island in Antarctica in 2010 (Mwangi et al., 2016). PBB-101 was found to occur in the eggs of the Ivory Gull (*Pagophila eburnea*) collected from the Canadian Arctic at concentrations: 5.6±0.5 ng g^{-1} fat in 1976, 9.3±1.0 ng g^{-1} fat in 1987 and 5.6 ng g^{-1} fat in 2004 (Braune et al., 2004). When compared to PCBs and similar compounds, the scale of diffusion into the environment via the atmosphere is lower for various reasons, e.g. the larger size of the molecules, significant photo lability and degradation to lower brominated congeners when in the gaseous phase.

Structurally, the planar configured PBBs (those with a lateral bromine substitution) have a common mechanism of toxic action as chlorinated dioxins and dioxin-like PCBs, via binding to the aryl hydrocarbon receptor (AhR) (van den Berg et al., 2013). Recent studies during the last 10 years shows that some PBB congeners are among the most active inducers of AhR in aquatic food chains, foods and humans, and contribute, albeit at a lower level, to the cumulative burden of dioxin-like toxicity (Bramwell et al., 2017; Fernandes et al., 2008; 2009b; 2018; 2019; Gieroń et al., 2010; Rose et al., 2015; Watanabe et al., 2003). As a result of their persistence, bioaccumulative potential, ability to survive long range environmental transport and toxicity, PBBs (in particular HxBBs) were listed in Annex A of the Stockholm convention since 2009 (Stockholm Convention, 2019).

Cod (*Gadus morhua*) liver and cod liver products in the past were contaminated with a range of different halogenated compounds (Falandysz et al., 1993, 1994a, 1994b). In a recent retrospective study, medicinal grade cod liver oil and canned liver products showed substantial contamination with dioxin-like contaminants including polychlorinated dibenzo-*p*-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), PCBs and polychlorinated naphthalenes (PCNs), and PBDEs (Falandysz et al., 2019a, 2019b). This study quantifies concentrations of individual PBB congeners in cod liver-derived products and investigates the variation in concentrations during the last decades. It also assesses the exposure to PBBs from these products which include medicinal grade cod liver oils that were popularly administered to children and youth as dietary supplements. It also briefly reviews the very small volume of published data on PBBs in fish from the North Atlantic and from inland waters in Europe.

2. Materials and methods

2.1. Cod-derived product samples

Cod liver oil products that originated from the Baltic Sea or the North Atlantic cod were obtained as retail products or from production plants. The canned cod livers were purchased from a retail store in Gdańsk, Poland in early 2017. The more historic samples (collected between 1972 and 2001) were stored in the original colored glass or metal containers at 4 °C until analysed (Falandysz et al., 2019a). Details of the samples are as follows: Poland, 1972: Medicinal grade cod liver oil Production plant, Gdynia, Poland, 1993 and 2001: Two samples of cod liver oil Iceland 1980 (Red Cross donation): Medicinal grade cod liver oil Norway 1982: Retail medicinal grade cod liver oil e) Łeba, Poland 2017: Two cod liver products, with Polish label translations, "cod liver in its oil" and "cod liver and vegetable pate".

2.2. Analysis

The analytical methodology has been presented in detail before (Fernandes et al., 2004, 2008) but a summarized description is given below. Analysis was based on internal standardization with isotopically labelled PBBs and analysis by high resolution gas chromatography coupled to high resolution mass spectrometry (HRGC-HRMS).

Analytes: The following PBB congeners were analysed (IUPAC numbers): 49, 52, 77, 80, 101, 126, 153, 169 and 209. Reagents and standards: $^{13}C_{12}$-labelled PBB standards were purchased from Wellington Laboratories Inc. (Ontario, Canada) and Cambridge Isotope Laboratory (Andover, MA, USA) and used as internal standards for the analysis. All other reagents were as described before (Fernandes et al., 2004, 2008).

Extraction, cleanup and fractionation: Three to five gram sample aliquots, fortified with internal standards were extracted using a dichloromethane:hexane (40:60) mixture. The *ortho*-substituted PBBs were separated from non-*ortho* PBBs and other contaminants using an activated carbon column. These two fractions were purified using an activated alumina column Instrumental analysis: HRGC-HRMS measurements were carried out using a Waters Autospec Ultima mass spectrometer coupled to a Hewlett Packard 6890N gas chromatograph fitted with a 60 meters DB-5MS column (or 30 meters RTX-1614 column for non-*ortho* PBBs) and a programmable temperature vaporization injector. Mass spectrometric measurements were carried out in positive electron ionization mode as a mass resolution of 10,000.

Quality assurance and quality control: Samples were analysed with a procedural blank and a reference material as described before (Li et al., 2019; Fernandes et al., 2016a and 2019), and these were evaluated for quality prior to quantitation and reporting. The analytical recoveries and precision (RSD) of the method (lipid weight) based on the use of $^{13}C_{12}$ labelled surrogates were typically in the ranges: 50–110%, and 10% respectively. Detection limits (LOD) were 0.002 ng g^{-1} for *ortho*-PBBs and 0.1–6.0 pg g^{-1} (non-*ortho*-PBBs). Although there is no performance testing (PT) for PBBs, the methodology involves simultaneous analysis of PBDEs for which excellent scores have been recorded for PTs carried out during the course of the work (e.g. Dioxins in Food, 2018). Measurement uncertainty (an expanded uncertainty with a coverage factor of 2) estimates range from around 20% (at ≥ 10x the LOD) to around 200% at the LOD.

Exposure estimation through dietary or supplementary intake was based on suggested daily doses for medicinal grade cod liver oil and on the typical weekly amount of canned cod liver products consumed by some individuals, as detailed in an earlier study (Falandysz et al., 2019a). Suggested doses for cod liver oil were 1 to 4 tablespoons (adult; 70 kg body mass), 1 to 2 tablespoons (teenager; 56 kg body mass) and 1 to 2 teaspoons (child; 26 kg body mass), On a mass basis, these volumes correspond to 12 g per tablespoon and 4 g per teaspoon for the oils, and a weekly intake of 105 - 150 g (adult), 52-75 g (teenager) and 26-37 g (child) for the canned liver products. The daily intake of six *ortho* PBB congeners (PBB-49, 52, 80, 101, 153 and 209) and one non-*ortho* PBB congener (PBB-77 were selected based on occurrence), was estimated using individual congener concentrations and the quantity of product consumed, divided by the appropriate body mass. The toxic equivalence intake was estimated using the maximum concentration of PBB-77, or the LOQ values for PBBs 126 and 169. Toxic equivalence factors corresponding to the analogous PCBs 77, 126 and 169 (0.0001, 0.1 and 0.03 respectively) were used as relative potency values to estimate the TEQ.

3. Results and discussion

3.1. Concentrations

Individual concentrations of the measured PBB congeners detected in cod liver oils and cod liver products from this study are listed in Table 1. Data are provided on a lipid basis (ng g^{-1} fat) in order to allow comparison with other data, but in order to estimate the exposure rates, the concentrations for the canned products are also given in ng g^{-1} whole weight (ww). In a manner similar to PCBs, non-*ortho* PBBs generally occur at much lower levels, so the concentrations for these are expressed in pg g^{-1}.

All samples were contaminated with PBBs (Fig. 1), but concentrations varied depending on the sample and the PBB congener. *Ortho*-PBBs were detected in all cod liver oils sampled, with concentration ranges in the sub part per billion levels i.e.: PBB-49, 301 – 457 pg g^{-1}; PBB-52 71 – 390 pg g^{-1}; PBB-80, < 2.0 – 13 pg g^{-1}; PBB-101, 79 - 194 pg g^{-1}; PBB-153, 122 – 0.618 pg g^{-1} fat. The canned cod liver products on a whole weight basis, also contained these contaminants in the ranges: PBB-49, 37 - 116 pg g^{-1}; PBB-52, 46 - 112 pg g^{-1}; PBB-80, < 2 – 2 pg g^{-1}; PBB-101, 3 – 5 pg g^{-1}; and PBB-153, 6 – 7 pg g^{-1}. PBB-209 was only detected in one oil sample at 28 pg g^{-1}. Among the non-*ortho* PBBs, PBB-126 and PBB-169 were not detected and PBB-77 was in the ranges 0.6 to 5.78 pg g^{-1} fat in oils and 0.06 to 0.126 pg g^{-1} whole weight in canned livers.

This data shows that in common with PCBs, non-*ortho* PBB concentrations in cod liver oils are considerably lower than the *ortho*-substituted congeners. The occurrence of non-*ortho* PBBs is very rarely reported, most likely because concentrations are very low in fish (Fernandes et al., 2018) and other foods (Fernandes et al., 2008; 2019). In general, the low occurrence levels reflects the lower and earlier utilization of PBBs in some parts of the world (Fernandes et al., 2016b; Gieroń et al., 2010) including the Baltic Sea relative to other BFRs such as PBDEs.

Figure 1

In a recent evaluation (IARC, 2016) by the European Food Safety Agency (EFSA), fish and seafood were considered as a significant source of PBBs in the European diet, with limited number of data, largely from the UK (Bramwell et al., 2017). An earlier study (Gieroń et al., 2010) on retail fish from Polish and French markets, included several species of fish from the North Sea such as the North Sea Atlantic salmon (*Salmo salar*); herring (*Clupea harengus*), scarp (*Psetta maxima*), gilthead seabream (*Sparus aurata*), grey gurnard (*Eutrigla gurnardus*) and Baltic Sea species such as salmon and Baltic herring (*Clupea harengus membras*). Some differences in homologue groups patterns were observed for PBB levels in fish from the two countries with tetrabrominated biphenyls dominating in both locations. Individual congeners such as PBB-29, 49, 52, 101 and 153 were noted together with some other unidentified *ortho*-PBBs.

Coincidentally, the study by Gieroń et al. (2010) also included two cans of cod liver products ("cod liver in cod liver oil") collected in 2007. Currently these canned products are commonly retailed and consumed all over the Baltic and wider North Atlantic region as a food commodity and have been shown to be contaminated with a number of other halogenated POPs such as PCBs, PCDDs, PCDFs, PCNs and PBDEs (Falandysz et al., 2019a, 2019b). In terms of PCBs, the concentrations in canned cod liver in 2017 (Falandysz et al., 2019a) were around threefold lower than those sampled in 1990 (Falandysz et al., 1992, 1993).

Edible muscle meat of both marine fish sourced in the past from the regions of the North Atlantic and fish sampled from the inland waters in Europe usually showed contamination with PBBs, even if the reported data shows small difference in the number of minor compounds quantified or the non-reporting of some minor congeners (Gieroń et al., 2010; Rose at al., 2015). Herring from the Baltic Sea contained ΣiPBBs in concentration from 0.50 ± 0.09 to 14 ± 5 pg g^{-1} ww and salmon from 17 ± 9 to 40 ± 3 pg g^{-1} ww, and in species from the North Sea was 22 ± 5 to 270 ± 72 pg g^{-1} ww) with herring showing the lowest concentration at 22 ± 5 pg g^{-1} ww (Gieroń et al., 2010). In a later study (Fernandes et al., 2018), herring (n=19) from the wider North-East Atlantic (coastal areas from Norway to Portugal) region showed PBB concentrations in the range of 5.1 to 34 pg g^{-1} ww (average 17 pg g^{-1} ww), in good agreement with the study from 2010 (Gieroń et al., 2010), although as noted, the level of contamination varied depending on location, with most of the higher PBB concentrations being observed for fish sampled off the southern coast of England and northern France. Similarly, other seafood such as shellfish also showed PBB contamination but at low concentrations (Fernandes et al., 2008; 2009b). Data on PBBs in fishery products available prior to 2010 has been compiled and reviewed (EFSA, 2010).

Freshwater species appear to be less contaminated with PBBs than marine fish as described in two studies from Poland and the UK. Carp and trout from Poland, showed ΣPBBs in muscle tissue at a concentration range of 0.57 ± 0.25 to 6.1 ± 1.1 pg g^{-1} ww (Gieroń et al., 2010). In the study from the UK, (Rose et al., 2015) which examined a range of different freshwater fish species, *ortho* PPBs were not detected in any of the samples examined (LOD - 10 pg g^{-1} ww). However, non-*ortho*-substituted PBBs were measured at a much lower LOD of 0.01 pg $g\text{-}^{1}$ ww and PBB-77 was detected between the range of 0.01 to 0.09 pg g^{-1} ww, while PBB-126 and PBB-169 were not detected. This is likely due to the lower detection limits, but it is interesting to note that in the present study, PBB-77, was detected in all samples in the range 0.06 to 0.126 pg g^{-1} whole weight, in canned liver products and in the range 0.6 to 5.8 pg g^{-1} in cod liver oils (Table 1).

A duplicate total diet study in the UK showed low levels of food contamination with PBBs (Bramwell et al., 2017). Interestingly, some non-*ortho*-PBBs including PBB-126 and PBB-169 have been found in pooled Irish mothers milk samples in concentrations: 0.12 pg g^{-1} fat weight (< 0.11 to 0.14 pg g^{-1} fw) for PBB-77, 0.26 (0.22 to 0.40 pg g^{-1} fw) for PBB-126, 0.05 (< 0.02 to 0.06 pg g^{-1} fw) for PBB-169, and 0.13 pg g^{-1} fw for PBB-153 (Pratt et al., 2013).

3.2. Trend

As mentioned earlier, in the Polish study by Gieroń et al. (2010), five PBB congeners (PBB-29, 49, 52, 101 and 153) were identified and measured in the study samples which included canned cod livers (2 samples). The average concentrations for the PBBs that were common to both studies (PBBs: PBB-49, 52, 101 and 153) in pg g^{-1} whole weight were: < 0.45 to 99, 130 to 440, 93 to 310 and < 0.90 to 93, respectively (Gieroń et al., 2010). The concentrations were similar for PBB-49 in the canned cod liver but other PBBs were higher than those reported in the present study (Table 1). This comparative outcome is at best, indicative, because of the very small number of samples and additionally, there is a difference of 10 years between sampling of the canned cod livers, with the samples in the present study being collected 10 years later than the earlier study. Nonetheless, the results of both studies underline the relatively low concentrations of PBBs in comparison to other mass produced halogenated POPs such as PCBs, PCNs, PBDE, DDT etc. (Falandysz et al., 1992, 1993, 2019a, 2019b).

Although a trend to lower concentrations of POPs have been reported in recent years, there is continuing debate about the rate of decline for some contaminants such as PCNs and PCBs (Haglund et al., 2010, Karl et al., 2010) in Baltic Sea fish. However, the PBB data for the samples sourced from the Baltic Sea in this study do appear to show a decline, as seen in Fig. 2. As mentioned earlier, the observation is indicative, because of the small number of samples. The smaller reduction in PBB concentrations in the later samples may also be due to improving purification techniques for fish oils. Although some cod liver oil producers in Poland were obliged to process fresh fish livers within 24 h, using cold filtration through a diatomaceous earth to obtain a high quality clear oil with a delicate hint of fish as required by the Pharmacopoeia. This processing is unlikely to have been as effective as current techniques that use molecular distillation and/or activated adsorbents (charcoal) to remove halogenated contaminants. However, it is consistent with the general decline observed for some other contaminants, and also with the lack of PBB manufacture in the Baltic region. This excludes Western Germany (EFSA, 2010), and the import of PBB formulations and products, particularly by former Eastern bloc states before transition.

Figure 2.

3.3. Intake and TEQ

As mentioned earlier, the relatively more toxic non-*ortho*-PBB-77 (AhR mediated toxicity) occurred in cod liver oils at a concentration ranging from 0.6 pg g^{-1} fat (Iceland) to 5.686 pg g^{-1} fat (mean, Baltic Sea, 2001) and at 0.06 to 0.126 pg g^{-1} whole weight in canned cod livers. The other non-*ortho*-PBB congeners (PBB-126 and PBB-169) were not detected even at the low detection limits achieved in this study (typically around 0.05 pg g^{-1}). Estimated daily intakes for the PBB-77 contained in cod liver oils were in the range from 0.10 to 3.9 pg kg^{-1} bm for adult, from 0.13 to 2.4 pg kg^{-1} bm for a teenager and from 0.092 to 1.7 pg kg^{-1} bm for a child. Estimated weekly intakes of PBB-77 from canned cod liver were highest for adults, ranging from 0.091 to 0.27 pg kg^{-1} bm, and for teenagers were 0.056 to 0.17 pg kg^{-1} bm and for childrens from 0.060 to 0.18 pg kg^{-1} bm (Table 2).

In term of dioxin-like toxic equivalence (TEQ), PCB-77 has a relative low toxic equivalency factor of 0.0001 (EC, 2011). If we assume the same potency for the brominated analogue (PBB-77), and apply this value for estimating TEQ (van den Berg et al., 2013), the resulting TEQ intake is negligible at < 0.001 pg kg^{-1} bm day^{-1} for all populations (Table 2). This is in marked contrast to the estimated TEQ intake arising from PCDD/F, PCB and PCN intake from the same samples (Falandysz et al., 2019a) and reflects the combination of the relatively low occurrence levels of PBB-77 and the applied relative potency value. The other contributors to TEQ, PBBs-126 and 129, were below the LOQs (< 0.02 to 0.08 pg g^{-1}), so the overall contribution to the dioxin-like TEQ arising from the measured PBBs in these samples can also be assumed to be low (< 0.01 pg kg^{-1} bm day^{-1}, measured using the upper bound concentrations). This level is considerably lower (around three to four orders of magnitude) than the daily intake of TEQ arising from other dioxin-like contaminants that were present in these oils. In an earlier study, Falandysz et al., 2019a, reported daily intakes of PCDD/Fs + dl-

PCBs + dl-PCNs from the same oils at 15 to 293 pg TEQ kg^{-1} bm (Baltic Sea), 23 to 101 pg TEQ kg^{-1} bm (Norway) and 11 to 48 pg TEQ kg^{-1} bm (Iceland). Corresponding intakes from consumption of canned liver over a week were in the range 32 to 99 pg TEQ kg^{-1} bm $week^{-1}$. As far as dioxin-like effects were considered, the PBB concentrations in these oils would imply a relatively low level of health concern.

Conclusions

Cod liver products, both oils and the canned livers, produced in Poland and other North Atlantic regions over the previous 40 to 50 years covered by this study, were found to show contamination with PBBs. The *ortho*-substituted PBBs (49, 52, 101, 153) occurred to a greater extent than the non-*ortho*-substituted PBBs, of which PBBs 126 and 169 were not detected. In comparison to other BFRs such as PBDEs, the occurrence levels are considerably lower and probably reflect the lower levels of usage in this region. During the period (1972-1993) for which samples from both the studies areas were available, PBB contamination levels for the Baltic Sea and the North Atlantic were similar. Dietary and supplementary intakes of the more toxic PBBs from the consumption of the studied products is relatively lower than the corresponding toxic equivalent (TEQ) intakes arising from the presence of other similar contaminants that were found in these sample in earlier studies.

Disclaimer

The authors assert no conflict of interest.

Credit authorship contribution statement

Jerzy Falandysz: Conceptualization, Resources, Methodology, Funding acquisition, Formal analysis, Data curation, Writing - original draft, Writing - review & editing. **Frankie Smith**: Resources, Analysis, Data curation, Investigation. **Alwyn R. Fernandes**: Conceptualization, Resources, Methodology, Formal analysis, Data curation, Writing - original draft, Writing - review & editing.

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FIGURE LEGENDS

Figure 1. The distribution of bromobiphenyls (PBB 49, 52, 77, 80, 101, 126, 153, 169 and 209) concentrations in cod liver products from the Baltic Sea and North Atlantic.

Figure 2. Trend in PBB concentrations in Baltic Cod liver and cod liver oil.

Table 1. PBBs: *ortho*- and non-*ortho* PBBs (pg g^{-1} fat) in cod liver oils and canned liver products (fat weight // whole weight) produced from cod liver sourced from the North Atlantic in 1972 – 2017

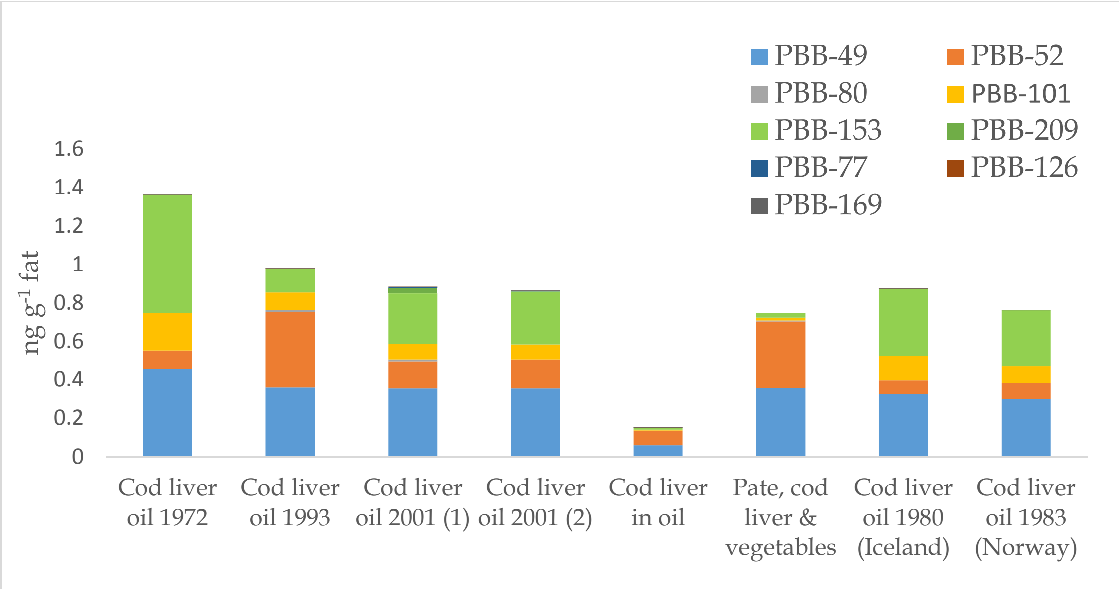
PBB Congeners	Region of the North Atlantic							
	Baltic Sea						North Atlantic-Iceland	North Atlantic-Norway
	Cod liver oil				Canned cod livers Fat weight // Whole weight		Cod liver oil	
	1972	1993	2001	2001	2017[A]	2017[B]	1980	1982
***Ortho*-PBBs**								
PBB 49	457	361	355	356	59 // 37	357 // 116	326	301
PBB 52	95	390	140	149	74 // 46	347 // 112	71	81
PBB 80	< 2	13	11	< 2	3 // 2	4 // < 2	< 2	< 2
PBB 101	194	91	81	79	5 // 3	16 // 5	127	89
PBB 153	618	122	263	275	9 // 6	22 // 7	350	290
PBB 209	< 24	< 21	28	< 21	< 15 // < 10	< 28 // < 9	< 25	< 21
Non-*ortho*-PBBs								
PBB 77	0.726	0.922	5.588	5.784	0.2 // 0.126	0.185 // 0.06	0.6	0.93
PBB 126	< 0.045	< 0.04	< 0.04	< 0.039	< 0.029 // < 0.018	< 0.052 // < 0.017	< 0.046	< 0.039
PBB 169	< 0.083	< 0.074	< 0.073	< 0.071	< 0.052 // < 0.033	< 0.095 // < 0.031	< 0.085	< 0.072
Σ_6*ortho*-PBBs[#]	1390	998	878	882	165 // 104	774 // 251	901	784
$\Sigma_3$non-*ortho*-PBBs[#]	0.854	1.036	5.7	5.89	0.281 // 0.177	0.332 // 0.108	0.731	1.04
TEQs pg g^{-1}								
ΣNon-*ortho*-PBBs[#]	0.007	0.006	0.007	0.007	0.004 // 0.003	0.008 // 0.003	0.007	0.006

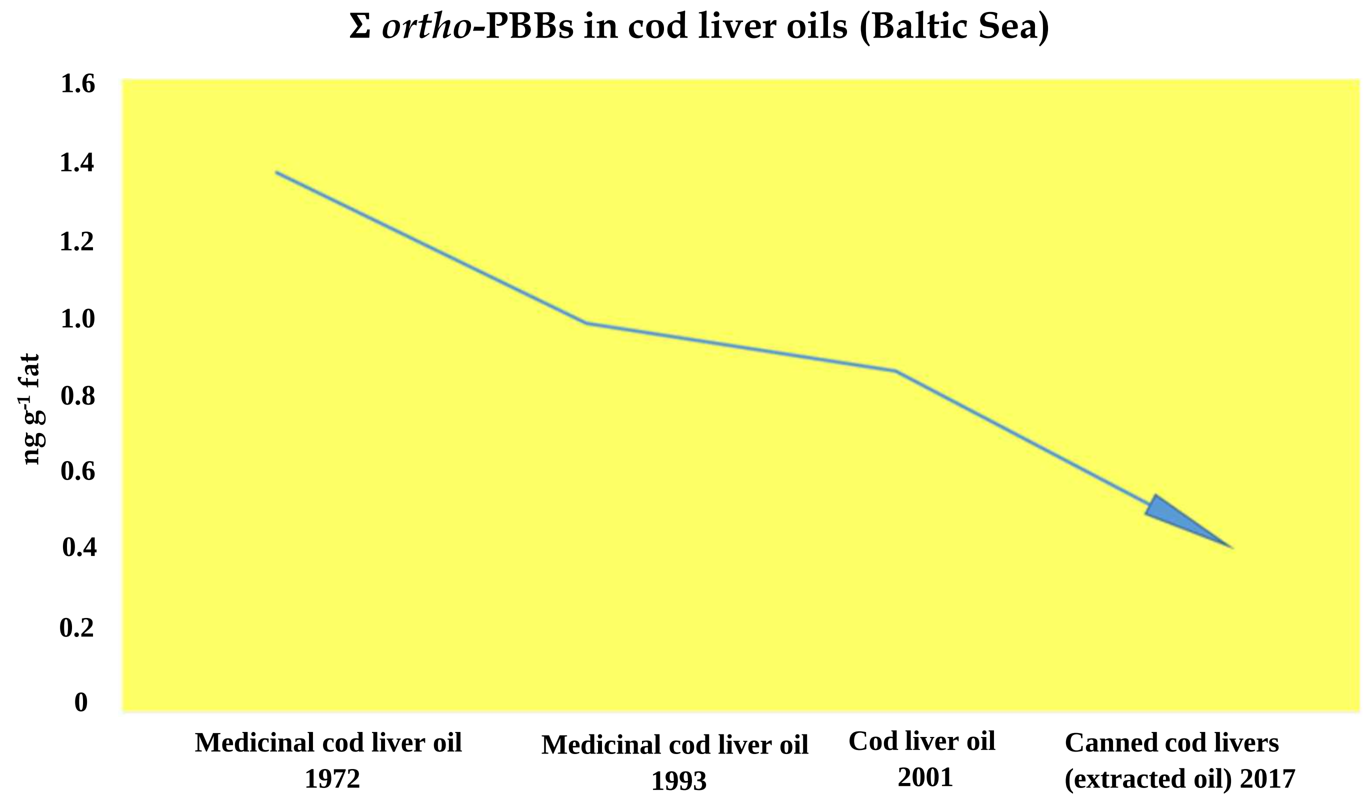
[A] and [B]Two types of canned cod liver products: "cod livers in own juice" (fat content - 62.8%)[A] and "pate, cod liver & vegetables" (fat content 32.3%)[B] produced in Łeba (Poland) in February 2017; [#]sum of upper bound values

Table 2. Contaminant intake through the diet (canned products) or through supplements (oils)

Parameters	Product intake (g)	Contaminant intake	
		PBB 77	TEQ
Cod liver oil		(pg kg^{-1} bm day^{-1})	
Baltic Sea – Poland; 1972-2001			< 0.001 pg^{-1} bm day^{-1}
Adult (70 kg bm)	12	0.12 – 0.97	
	24	0.25 – 1.9	
	48	0.50 – 3.9	
Teen age 14 (56 kg bm)	12	0.16– 1.2	
	24	0.31 – 2.4	
Child age 7 (26 kg bm)	4	0.11 – 0.87	
	8	0.22 – 1.7	
Atlantic – Norway; 1982			< 0.001 pg^{-1} bm day^{-1}
Adult (70 kg bm)	12	0.16	
	24	0.32	
	48	0.64	
Teen age 14 (56 kg bm)	12	0.20	
	24	0.40	
Child age 7 (26 kg bm)	4	0.14	
	8	0.29	
Atlantic – Iceland; 1980			< 0.001 pg^{-1} bm day^{-1}
Adult (70 kg bm)	12	0.10	
	24	0.21	
	48	0.41	
Teen age 14 (56 kg bm)	12	0.13	
	24	0.26	
Child age 7 (26 kg bm)	4	0.092	
	8	0.18	
Canned cod livers (w/w); 2017		(pg kg^{-1} bm week^{-1})	< 0.001 pg^{-1} bm week^{-1}
Adult (70 kg bm)	105	0.091 - 0.19	
	150	0.13 - 0.27	
Teenager 14 (56 kg bm)	52[A]	0.056 - 0.12	
	75[A]	0.080 - 0.17	
Child age 7 (26 kg bm)	26[B]	0.060 - 0.13	
	37[B]	0.085 – 0.18	

Notes: A (a half of a package); B (a quarter of a package)


PBB-49
PBB-52
PBB-80
PBB-101
PBB-153
PBB-209
PBB-77
PBB-126
PBB-169
ng g-1 fat
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
Cod liver oil 1972
Cod liver oil 1993
Cod liver oil 2001 (1)
Cod liver oil 2001 (2)
Cod liver in oil
Pate, cod liver & vegetables
Cod liver oil 1980 (Iceland)
Cod liver oil 1983 (Norway)


Σ *ortho*-PBBs in cod liver oils (Baltic Sea)
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0
ng g-1 fat
Medicinal cod liver oil 1972
Medicinal cod liver oil 1993
Cod liver oil 2001
Canned cod livers (extracted oil) 2017

Highlights

- PBBs were detected in all cod liver oils and canned liver products from 1972-2017
- *Ortho*-PBBs 49, 52, 101, 153 and non-*ortho* PBB 77 occurred in all samples
- PBB levels in cod liver oils peaked at the turn of the century
- During 1972-1993, contamination levels for Baltic Sea and North Atlantic were similar