Bisphenol A in edible part of seafood

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Abstract

Bisphenol A (BPA) is a man-made compound, mainly used as a monomer to produce polycarbonate (PC), epoxy resins, non-polymer additives to other plastics, which have many food related applications, such as food storage containers, tableware and internal coating of cans, as well as non-food applications such as electronic equipment, construction materials and medical devices. BPA exposure can occur when the residual monomer migrates into packaged food and beverages. Moreover, due to the ubiquitous presence of this compound, the general population can be exposed to environmental sources such as water, air and soil. Many studies have investigated the potential health hazards associated with BPA, which can elicit toxic and cancerogenic effects on humans. According to the European Food Safety Authority opinion, diet is considered to be the main source of exposure, especially canned food; moreover, among non-canned food, meat and fish products have the highest levels of BPA contamination. This review focuses on BPA contamination in seafood, analysing worldwide literature (from January 2010 to October 2015) on BPA contamination of edible parts. The authors try to identify differences between canned and non-canned seafood in literature, and gaps in the state of art. The data evaluated underline that all concentrations for both canned and non-canned seafood were below the specific migration limit set by the European Community Directive for BPA in food. Moreover, the canned seafood is more contaminated than the non-canned one.

Introduction

Brief history and chemical properties

Bisphenol A (BPA) [4,40-dihydroxy-2,2-diphenylpropane] is a man-made compound, produced from the condensation of two moles of phenol and one mole of acetone (Geens *et al*., 2012). This monomer was first synthesized by A.P. Dianin in 1891 (Michalowicz, 2014).

BPA has a molecular weight of 228.29 and its physical state is represented by white to light brown flakes or powder (PubChem, 2015).

In 1957 the commercial production of BPA began in the United States and only one year later in Europe (Corrales *et al.,* 2015). Since then, it has been used as a monomer to produce polycarbonate (PC), epoxy resins for coating, the flame retardant tetrabromobisphenol A, PVC stabilizers and antioxidant in the production plastics (Groshart *et al.,* 2001).

It has a high water solubility (120 mg/L at 25°C), and according to its chemical-physical properties, it (log K_{OW} is about 3.4) shows a moderate bioaccumulative potential (Corrales *et al.,* 2015; Groshart *et al.,* 2001). Nevertheless, recent studies focused on the bioaccumulation and toxicological properties of moderately lipophilic substances (Corrales *et al.,* 2015).

BPA is also characterised by rapid photo-oxidation in air, which explains its short half-life in the atmosphere (0.2 days). BPA has been found in many environmental matrices (water, soil, and air), but also in humans and wildlife despite the above-mentioned half-life and the moderate potential for bioaccumulation (Corrales *et al.,* 2015).

In 1993 a research team accidentally discovered a substance, with estrogenic properties in the yeast culture media, leached out of PC flasks during the autoclaving procedure.

After many investigations, this substance was identified as BPA (Krishnan *et al.,* 1993), whose estrogenic properties had been acknowledged since 1930s (Dodds and Lawson, 1936). In 1995 it was reported, for the first time, that some canned food and its preserving liquid can acquire estrogenic activity: this phenomenon was reasonably attributed to BPA leached from the coating lacquer (Brotons *et al.,* 1995). Since that moment there was an increasingly interest by scientists all over the world to further investigate the migration of this compound from packaging to canned food.

Applications and exposure sources

BPA is one of the highest volume chemicals manufactured globally. The worldwide production is expected to exceed 6.3 million metric tons in 2015 (Cantonwine *et al.,* 2013). The majority of BPA is used for the production of polycarbonate plastics: due to its thermoresistance and transparency, it is used for the production of optical media, construction materials, electronic equipments, food and storage containers, tableware, reusable water bottles (Arnich *et al.,* 2011; Geens *et al.,* 2011; EFSA, 2015d). The second largest application is represented by epoxy resins, which perform well in terms of corrosion protection, mechanical and thermal stability. They are used in several consumer and industry applications: internal coating of metallic food and beverage cans, pipes for drinking water and food industry, construction panels and automotive parts (Vanderberg *et al.,* 2010; Arnich *et al.,* 2011; Geens *et al.*, 2011). Other polymers include for example polysulfone, which have several medical applications, and the flame retardant tetrabromobisphenol A (Geens *et al.,* 2011).

BPA is also utilised in stabilisers and antioxidants products such as vinyl chloride, and it is also employed in thermal paper production for tickets, newspapers and food packaging (Geens *et al.,* 2011; Michalowitz, 2014).

Due to the massive presence of its products in so many applications that can potentially release BPA, it is considered a ubiquitous environmental contaminant (Flint *et al.,* 2012). Therefore, the environment (air, soil and water) can be a human source of exposure but the most important is the diet (Kang *et al.,* 2006; Geens *et al.,* 2012; EFSA, 2013). In particular, canned food is considered the major contributor for all age groups (EFSA, 2015d).

In fact, consumers may be exposed to BPA when the residual monomer in the polymer migrates from cans into food and beverages, or if the polymer itself undergoes to hydrolysis (Hoekstra and Simoneau, 2013).

In general, the migration of substances from packages into food and beverages is a complex phenomenon, influenced by different factors

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such as (Arvanitoyannis and Kotsanopoulos, 2014): food composition (foodstuff with high fat content were reported to show high migration levels); type of contact [the migration is influenced by the specific type of contact between food and packaging (direct and indirect contact)]; time of contact (the concentration of the migrating molecule in food is directly proportional to the square root of the contact time); temperature of contact (higher temperature seems to be related to an high migration rate); type of packaging material (thinner packages are linked with higher migration rates); nature and amount of migrant compound.

Dental fissure sealants, printing inks, adhesives, thermal paper, medical devices and children's toys and articles intended to be mouthed, represent other sources of consumer exposure, (Hoekstra and Simoneau, 2013; EFSA, 2015d).

Bisphenol A in seafood

According to EFSA report (EFSA, 2013) fish and seafood, a relevant part of human diet, present one of the highest BPA contamination. While canned products can be contaminated through the mechanism described above, BPA may leach from plastic in oceans, causing a direct contamination of fish. Plastic debris may be the major source of BPA in seafood: this material can contain BPA at concentration up to 300 ppb. Bioaccumulation rates in fish tissues and seafood were already measured (Gatidou *et al.,* 2010; Liu *et al.,* 2012; Yang *et al.,* 2014). Some researchers argue that the BPA environmental contamination in fish can be higher than in canned foods (Engler, 2012).

Health effects

BPA is considered an endocrine disrupting compound (EDC) (Rubin and Soto, 2009; Rochester, 2013), which is defined by the European Community as *an exogenous substance or mixture that alters function(s) of the endocrine system*.

BPA can stimulate the transcriptional activity of estrogen receptor and (ER and ER , respectively) at a concentration of 100-1000 nM. Moreover, it shows an affinity 10.000-fold lower than estradiol (E2) for both ER and ER (Kuiper *et al.,* 1998).

BPA completes its disrupting activity mimicking, enhancing or inhibiting the activity of endogenous estrogens, disrupting estrogen nuclear hormone receptor action (Wetherill *et al.,* 2007). This molecule is defined as a selective estrogen receptor modulator (SERM) because it can exert estrogenic or anti-estrogenic activity in different tissues (Richter *et al.,* 2007; Wetherill *et al.,* 2007).

BPA can elicit adverse health effects both in humans, animals and in wildlife species (Oehlmann *et al.,* 2009; Flint *et al.,* 2012; Rochester, 2013); human exposure can be related to alterations in fertility, development, cardiovascular apparatus, metabolic functions and immune system (Rochester, 2013; Mileva *et al.,* 2014). Recent studies investigated the epigenetic effects linked to this contaminant (Susiarjo *et al.,* 2007; Singh and Li, 2012; Mileva *et al.,* 2014). BPA exposure can alter selected gene expression through histone methylation and induce DNA methylation changes, which are altered in cancer cells (Susjario *et al.,* 2007; Romani *et al.,* 2015). BPA exposure studies, in developmental *in vivo* models, show increase susceptibility to prostate, hepatic and breast carcinogenesis (Ferreira *et al.,* 2015; Romani *et al.,* 2015).

BPA is considered to follow the *low-dose* and non-monotonic nature of hormones and endocrine disruptors: non-monotonic dose-response curves often show a *U* or *inverted U* shape in a dose-response curve. The low-dose hypothesis is based on the idea that *low-doses* can act in a non-linear manner, and exhibit significant effects at low levels, which could not happen at higher doses (Vandenberg *et al.,* 2012; Rochester, 2013).

This behaviour has been observed in animals, but further studies are needed to confirm it in humans.

In 2014 EFSA published *Public consultation on the draft opinion on bisphenol A (BPA) – Assessment of human health risks*. ANSES (*Agence Nationale de Sécurité Sanitaire de l'alimentation, de l'environnement et du travail*) highlighted the extensive work conducted by EFSA, but identified certain differences in the evaluation and interpretation of the available studies (ANSES, 2015).

In 2015 report EFSA concluded that BPA does not present any risk for consumer health at the current exposure levels, but nonetheless updated the tolerable daily intake (TDI) from 50 to 4 g/kg body weight per day; this value is twelve and a half times lower than the previous level (EFSA, 2015b).

This TDI is temporary (t-TDI), pending the outcome of an on-going long-term study in rats involving prenatal and postnatal exposure to BPA (EFSA, 2015c).

However, according to EFSA latest Risk Assessment (EFSA, 2015a), based on animal studies, BPA at high doses [more than 100 times the tolerable daily intake (TDI)] may cause adverse effects in liver and kidney. Moreover, possible adverse effects on nervous, immune, reproductive, cardiovascular, metabolic systems, and cancerogenicity are not considered likely at present, but they could not be excluded.

In conclusion, EFSA experts affirmed that, on the basis of the available data, there is no evidence on BPA non-monotonic dose-response relationships for the health effects considered: *BPA poses no health risk to consumers of any age group (including unborn children, infants and adolescents) at current exposure levels, from diet, dust, cosmetics and thermal papers* (EFSA, 2015a).

Nevertheless, the French Agency ANSES established that BPA is not harmless. In this sense France Government decided to ban BPA in food packaging produced or imported into its territory as from 1 January 2015 (European Parliament, 2015).

In addition, the Danish National Food Institute's scientists evaluate that EFSA's new TDI does not give protection against endocrine disrupting effects: it is proposed a new TDI of 0.7 mg/kg body weight per day or lower. The institute's assessment is based on the same studies as those in the EFSA report (DTU Food, 2015).

Other non European Authorities expressed their opinion on this issue. In 2005, the Japanese Institute of Advanced Industrial Science and Technology reported BPA was unlikely to represent unacceptable risks to human health and this opinion was confirmed in 2011 (EFSA, 2015e).

In 2008 the US National Toxicology Program (NTP) expressed some concerns about BPA effects on the infants and children behaviour, on the development of prostate and brain after pre-natal and post-natal exposure at current levels of human exposure; in addition NTP assessed minimal concerns about mammary gland development, acceleration in puberty and fetal and neonatal mortality (EFSA, 2015e).

In 2014 the US Food and Drug Administration (FDA) announced the safety of BPA at the current levels occurring in foods: in addition, together with Health Canada and Food Standard Australia New Zealand, established a TDI of 50 mg/kg body weight per day (EFSA, 2015e).

Legislation

In Europe, the Commission Directive 2002/72/EC (European Commission, 2002) authorised the use of BPA as monomer and additive for the manufacture of plastic materials and articles intended to come in contact with foodstuffs, and set a specific migration limit (SML) of 0.6 mg per kilogram food. In 2011 this Directive was amended by the Commission Directive 2011/8/EU, which posed a temporary ban on its use in manufacture of infant feeding bottles.

Then, Directive 2002/72/EC was replaced by Regulation (EU) No 10/2011 (European Commission, 2011), which maintained the ban of BPA in polycarbonate infant feeding bottles and kept the current restriction for BPA as a monomer with a specific migration limit (SML)=0.6 mg/kg food, but removed it from the authorisation list as an additive in plastic food contact materials and articles.

In 2008 Canada established a ban on the use

of BPA in polycarbonate baby bottles in order to reduce infants exposure to this contaminant (Carwile *et al.,* 2009).

In 2010 Australian Government established a voluntary phase-out of the use of PC plastic baby bottles (Food Standards Australia New Zealand, 2015).

Analytical methods in foodstuffs

Highly selective and sensitive techniques are required for BPA determination in foodstuffs, due to the low levels at which it can be found and the different composition of food matrices. Although the European Commission SML is set at 0.6 mg/kg, analytical methods have been developed with low detection limits to better assess the human exposure (Ballesteros-Gómez *et al.,* 2009).

Table 1. Bisphenol A mean values (expressed in ppb) of canned seafood products.

Continent	Seafood type	% Pos	Total samples	Mean (ppb)	Range (ppb)	Origin/ commercial area	Method	Authors
America	Tuna	-	15	137.0	9.0-534.0	Canada*	GC-MS	Cao *et al.* (2010)
	Tuna	-	5	78.6	51.0-109.0	Canada°	GC-MS	Cao *et al.* (2015)
	Tuna	100.0	1	45.6	-	Ecuador	LC-MS/MS	Fattore *et al.* (2015)
	Tuna	-	4	11.4	5.8-17.0	USA°	LC-MS/MS	Noonan *et al.* (2011)
	Tuna in oil	-	2	4.5	-	USA°	LC-MS/MS	Noonan *et al.* (2011)
	Tuna albacore	-	4	11.5	11.0-13.0	USA°	LC-MS/MS	Noonan *et al.* (2011)
	Mackerel	-	3	22.0	-	USA°	LC-MS/MS	Noonan *et al.* (2011)
	Boiled salmon	0.0	1	0.0	-	USA*	GC-MS	Kawamura *et al.* (2014)
	Canned fish	33.3	3	1.1	-	USA°	HRGC/LRMS	Lorber *et al.* (2015)
	Canned fish	100.0	52	28.0	1.0-265.0	Canada°	GC-MS	Cao and Popovic (2015)
Europe	Tuna in oil	100.0	1	169.3	-	Belgium*	GC-MS	Geens *et al.* (2010)
	Tuna in water	100.0	1	126.4	-	Belgium*	GC-MS	Geens *et al.* (2010)
	Tuna in vegetable oil	92.3	13	21.2	0.2-99.9	Portugal	GC-MS	Cuhna *et al.* (2012)
	Tuna in olive oil	71.4	7	5.2	0.2-18.9	Portugal	GC-MS	Cuhna *et al.* (2012)
	Tuna in olive oil	80.0	12	62.3	1.3-132.9	Italy°	LC-MS/MS	Fattore *et al.* (2015)
	Tuna in olive oil	100.0	1	30.4	-	Italy	LC-MS/MS	Fattore *et al.* (2015)
	Tuna in olive oil	66.7	3	29.3	1.3-13.0	Spain	LC-MS/MS	Fattore *et al.* (2015)
	Natural tuna	100.0	2	17.5	2.2-32.8	Portugal	GC-MS	Cuhna *et al.* (2012)
	Natural tuna	66.7	6	35.9	1.3-60.9	Italy°	LC-MS/MS	Fattore *et al.* (2015)
	Salmon	100.0	1	3.4	-	Belgium*	GC-MS	Geens *et al.* (2010)
	Anchovy	100.0	1	0.9	-	Belgium*	GC-MS	Geens *et al.* (2010)
	Anchovy fillets in vegetable oil	0.0	1	0.0	-	Portugal	GC-MS	Cuhna *et al.* (2012)
	Mackerel fillets in vegetable oil	100.0	2	21.7	-	Portugal	GC-MS	Cuhna *et al.* (2012)
	Mackerel	0.0	1	0.0	-	Spain°	LC-FL	Alabi *et al.* (2014)
	Mussels in pickled sauce	100.0	1	1.4	-	Portugal	GC-MS	Cuhna *et al.* (2012)
	Mussels	100.0	1	117.0	-	Spain°	LC-FL	Alabi *et al.* (2014)
	Cockles	100.0	1	182.0	-	Spain°	LC-FL	Alabi *et al.* (2014)
	Sardines in vegetable oil	60.0	5	3.7	0.2-8.7	Portugal	GC-MS	Cuhna *et al.* (2012)
	Sardines in oil	100.0	1	150.0	-	Spain*	GC-MS	Kawamura *et al.* (2014)
	Fish	100.0	2	1.1	0.9-1.3	Spain°	GC-MS	Fasano *et al.* (2015)
Africa	Tuna in oil	0.0	1	0.0	-	Ivory Coast	LC-MS/MS	Fattore *et al.* (2015)
Asia	Tuna in oil	-	3	4.3	0.5 -13.0	Japan	GC-MS	Kawamura *et al.* (2014)
	Tuna in oil	100.0	1	36.0	-	Thailand*	GC-MS	Kawamura *et al.* (2014)
	Tuna in oil	100.0	1	120.0	-	Vietnam*	GC-MS	Kawamura *et al.* (2014)
	Tuna	100.0	1	55.8	-	Indian Ocean	LC-MS/MS	Fattore *et al.* (2015)
	Tuna	100.0	1	81.1	-	Thailand	LC-MS/MS	Fattore *et al.* (2015)
	Boiled tuna	100.0	1	6.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Tuna and bonito in oil	100.0	1	56.0	-	Thailand*	GC-MS	Kawamura *et al.* (2014)
	Bonito in oil	100.0	1	5.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Sardine in oil	100.0	1	0.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Boiled salmon	100.0	1	12.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Boiled saury	100.0	1	0.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Boiled mackerel	100.0	1	7.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Boiled crab	100.0	1	7.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Blue crab	100.0	1	320.0	-	Thailand*	GC-MS	Kawamura *et al.* (2014)
	Boiled scalloper	100.0	1	21.0	-	Japan	GC-MS	Kawamura *et al.* (2014)
	Oyster in oil	100.0	1	10.0	-	Korea*	GC-MS	Kawamura *et al.* (2014)
Various origin	Natural tuna	100.0	1	187.0	-	Pacific Ocean	LC-MS/MS	Fattore *et al.* (2015)
	Natural tuna	0.0	1	0.0	-	FAO 34, 47, 51, 57, 71	LC-MS/MS	Fattore *et al.* (2015)
	Natural tuna	100.0	1	38.5	-	FAO 67, 71, 77, 87	LC-MS/MS	Fattore *et al.* (2015)
	Tuna in olive oil	100.0	1	25.4	-	Atlantic Ocean	LC-MS/MS	Fattore *et al.* (2015)

% Pos, percentage of samples above LOD/ quantification; GC-MS, gas chromatography coupled with mass spectrometry; LC-MS/MS, liquid chromatography coupled with tandem mass spectrometry; HRGC-LRMS, high resolution gas chromatography/low resolution mass spectrometry; LC-FL, liquid chromatography coupled with fluorescence detection. *Domestic or imported product commercialised in the area; °area of commercialisation. Mean values represent the average value for every seafood product in each article.

The analytical methods for the determination of BPA in seafood, include liquid chromatography (LC) and gas chromatography (GC) methods coupled with various detectors.

LC offers the advantage to analyse BPA directly, while GC needs a derivatisation step, but it is characterised by higher peak resolution (Ballesteros-Gómez *et al.,* 2009).

Fluorescence detection (FL) can be coupled with liquid chromatography, because BPA shows native fluorescence with excitation and emission wavelengths at 275 and 305 nm, respectively (Ballesteros-Gómez *et al.* 2009). The technique is utilised for BPA determination in different food matrices such as beverages, vegetables, fruit and fish (Geens *et al.,* 2010; Bemrah *et al.,* 2014; Liao and Kannan, 2013). LC coupled with fluorescence detection (LC-FL) can be followed by confirmatory methods such as liquid chromatography coupled with mass spectrometry (LC-MS).

LC-MS and GC coupled with mass spectrometry (GC-MS) offer more attractive and more reliable methods. Sample treatment can be reduced by the use of mass spectrometry, although a good sample preparation is still necessary: LC-MS and GC-MS methods provide higher confidence in BPA identification than other methods.

GC-MS is characterised by higher resolution and lower detection limits than LC-MS for the determination of BPA in food. However, this technique is considered quite complex, due to time-consuming derivatisation step, which can introduce new sources of errors (Ballesteros-Gómez *et al.,* 2009).

In addition, the analytical performance of GC methods can be reduced by the presence of lipids: extensive clean-up step is required for fatty food, such as fish (Ballesteros-Gómez *et al.,* 2009).

Aim

The purpose of this review is to investigate seafood contaminated by BPA, a relevant component of human diet, using worldwide data. This is done in order to: i) summarise the recent literature (from 2010 to present) regarding BPA contamination of seafood (canned and non-canned); ii) determine the different factors that can affect BPA seafood contamination; iii) underline the gaps and research needs about the current state of art (*e.g.* limited data about BPA contamination of a variety of seafood products, different geographical areas).

Literature search

A literature search was conducted using different electronic bibliographic databases (Scopus, ProQuest Biological Science Collection, PubMed, Google Scholar): the research was performed using different key words such as *bisphenol A AND fish*, and further refined adding *muscle AND/OR edible* in case of a large number of items. The publications analysed in this review were chosen according to the following criteria: language (only scientific studies written in English); publication period (from January 2010 to October 2015); topic (were considered scientific studies investigating BPA contamination in every seafood products); sample type (only data obtained from edible part of seafood were considered). Studies that performed BPA determination in other parts of fish, such as brain, gonads and liver, were excluded. Data for composite samples, such as *flavoured, cooked* samples, and fish products containing everything different from water (oil or pickled sauce) were not included; geographic origin of the samples (all papers from any geographic origin, in order to have a worldwide view of BPA contamination in seafood, all papers regarding samples from any geographic origin were considered). For canned seafood, if the geographical origin was not specified, it was considered the geographical area of commercialisation.

The publications that met these criteria were further analysed to ensure that the analytical method used to obtain the data was reliable. The articles have to specify the limit of detection (LOD) and/or limit of quantification (LOQ). In addition, they have at least to declare the accuracy and/or precision. The method should be validated in seafood matrix, or at least in other food matrices. Articles that used a method validated on other matrices such as plasma, wastewater, sediments, were not included.

In order to obtain the average value for every seafood product in each article, data reported as not detected were considered equal to zero. The range was defined between LOD and the maximum level measured.

Average values reported in Tables 1 and 2 are taken from the articles, or calculated: values are expressed using one decimal figure to uniform the data. In order to calculate BPA mean value in seafood products from each continent, concentrations expressed in dry weight or lipid weight were converted in wet weight, to make data comparable.

Discussion

In Tables 1 and 2 BPA mean values (expressed in ppb) of canned and non-canned seafood products are reported. Data are grouped by continents, seafood product type, percentage of positive samples (values>LOD), number of samples analysed, range (minimum-maximum value), country of origin or area of commercialisation, method utilised for analysis and references.

The means of BPA concentration levels reported for canned and non-canned seafood highlight a different contamination of these products: the canned food BPA levels (46.2 ppb) are higher than those non-canned (14.9 ppb). These data seem to be in line with those reported in the EFSA report, related to European data published for the period between January 2006 and December 2012.

About the geographical distribution, the data are limited to Europe, North America and Asia: in fact there are only few data in South America (Fattore *et al.,* 2015; Munaretto *et al.,* 2013) and Africa (Fattore *et al.,* 2015), and there are no data from Oceania (Tables 1 and 2). Moreover, in canned seafood products there are no remarkable differences between the BPA mean values, calculated for the different continents: 34.0 ppb for America, 48.9 ppb for Europe, 46.3 ppb for Asia, for Africa there is only one data available equal to zero. Instead, BPA mean values for non-canned seafood products show bigger differences: mean value of data from America (2.2 ppb) is lower than other continents, Europe and Asia with respectively 21.5 and 14.4 ppb.

The majority of canned seafood data are derived from tuna (in oil, water, vegetable oil, natural tuna, albacore) (Table 1). It is interesting to underline that a sample of blue crab from Thailand (Kawamura *et al.,* 2014) shows the highest value (320 ppb) of BPA concentration: this is the only data found for this product. BPA levels in canned seafood, especially the canned tuna, have been investigated, evaluating the different migration from the packaging to the solid and liquid food portions. Different authors underline that BPA would preferably be present in the solid portion of food with respect to the liquid phase (Geens *et al.,* 2010; Fattore *et al.,* 2015; Noonan *et al.,* 2011).

Furthermore, BPA migration from can is related to the preservation medium: BPA levels were higher in canned tuna preserved in oil that in aqueous medium, suggesting that oil may promote BPA migration from the can lining into the food (Fattore *et al.,* 2015). In addition, the BPA presence in aqueous medium is not relevant, while the oil show BPA contamination, even if lower than the solid food (Fattore *et al.,* 2015).

Storage time does not show any effect in BPA migration, while can coating type, together with sterilisation conditions, seem to be the major factors (Geens *et al.,* 2010; Fattore *et al.,* 2015).

Very few works have studied the influence of coating type on BPA migration in canned seafood and from their results there is no evidence of clear relationships between the coating type and the level of migration (Munguia-

Table 2. Bisphenol A mean values (expressed in ppb) of non-canned seafood products.

Continent	Seafood type	% Pos	Total samples	Mean (ppb)	Range (ppb)	Origin/ commercial area	Method	Authors
America	Prawn	83.3	6	1.8	0.0-5.5	USA	GC-MS	Zuo and Zhu (2014)
	Fish	0.0	3	0.0	-	USA*	GC-MS	Lorber *et al.* (2015)
	Fish muscle	100.0	52	7.3°	5.1-8.9°	USA	GC-MS	Yu and Wu (2014)
	Fish marine	100.0	5	2.0	0.5-4.5	Canada*	GC-MS	Cao *et al.* (2015)
	Shellfish	40.0	5	0.4	0.4-1.2	Canada*	GC-MS	Cao *et al.* (2015)
	Freshwater	60.0	5	1.1	0.4-3.4	Canada*	GC-MS	Cao *et al.* (2015)
	Tilapia	100.0	1	2.7	-	Brazil	GC-MS	Munaretto *et al.* (2013)
	Striped catfish	0.0	2	0.0	-	Brazil	GC-MS	Munaretto *et al.* (2013)
	Catfish	100.0	2	10.4	6.2-14.5	Brazil	GC-MS	Munaretto *et al.* (2013)
Europe	Anchovy	100.0	1	0.7	-	Belgium*	GC-MS	Geens *et al.* (2010)
	Mediterranean mussel	100.0	7	453.6°	342.8-611.9°	Greece	GC-MS	Gatidou *et al.* (2010)
	Stripped venus	75.0	4	330.8°	115.0-626.3°	Greece	GC-MS	Gatidou *et al.* (2010)
	Bearded horse mussel	100.0	7	388.5°	209.2-515.2 °	Greece	GC-MS	Gatidou *et al.* (2010)
	Mussel	-	10	43.3°	6.8-197.2°	Poland-Russia	HPLC-FL	Staniszewska *et al.* (2014)
	Bivalves	14.3	7	1.6°	0.9-11.2°	Spain	LC-MS/MS	Salgueiro-Gonzalez *et al.* (2012)
	Barbel	12.5	8	3.2°	-	France	LC-MS/MS	Miège *et al.* (2012)
	Common bream	55.6	9	19.8°	-	France	LC-MS/MS	Miège *et al.* (2012)
	White bream	40.0	5	9.6°	-	France	LC-MS/MS	Miège *et al.* (2012)
	Club	40.0	10	18.6°	-	France	LC-MS/MS	Miège *et al.* (2012)
	Fish	-	36	11.9	0.1-97.9	France*	GC-MS/MS	Bemrah *et al.* (2014)
	Shellfish	-	33	6.7	1.4-26.2	France*	GC-MS/MS	Bemrah *et al.* (2014)
	Herring	-	10	98.6°	-	Poland-Russia	HPLC-FL	Staniszewska *et al.* (2014)
	Flounder	-	6	430.4°	98.3-755.7°	Poland-Russia	HPLC-FL	Staniszewska *et al.* (2014)
	Cod	-	6	236.3°	25.4-798.4°	Poland-Russia	HPLC-FL	Staniszewska *et al.* (2014)
	Barbus graellsii	0.0	9	0.0°	-	Spain	LC-MS/MS	Jakimska *et al.* (2013)
	Cyprinus carpio	20.0	15	37.3°	0.0-223.9°	Spain	LC-MS/MS	Jakimska *et al.* (2013)
	Luciobarbus sclateri	20.0	15	11.8°	0.0-59.1°	Spain	LC-MS/MS	Jakimska *et al.* (2013)[#]
	Lepomis gibbosus	0.0	3	0.0°	-	Spain	LC-MS/MS	Jakimska *et al.* (2013)
	Frozen fish[§]	100.0	2	11.5	10.0-13.0	Norway*	LC-MS/MS	Sakhi *et al.* (2014)
	Caviar spread, cod roe^	-	2	20.0	-	Norway*	LC-MS/MS	Sakhi *et al.* (2014)
Asia	Big head carp	-	6	1.9	-	China	LC-MS/MS	Wei *et al.* (2011)
	Catfish	-	21	2.0	-	China	LC-MS/MS	Wei *et al.* (2011)
	Grass carp	-	6	1.3	-	China	LC-MS/MS	Wei *et al.* (2011)
	Grey mullet	-	18	0.6	-	China	LC-MS/MS	Wei *et al.* (2011)
	Mandarin fish	-	3	1.9	-	China	LC-MS/MS	Wei *et al.* (2011)
	Mud fish	-	15	2.0	-	China	LC-MS/MS	Wei *et al.* (2011)
	Rice field eel	-	14	0.5	-	China	LC-MS/MS	Wei *et al.* (2011)
	Snakehead	-	10	0.6	-	China	LC-MS/MS	Wei *et al.* (2011)
	Spotted snakehead	-	10	1.3	-	China	LC-MS/MS	Wei *et al.* (2011)
	Bartail flathead	-	6	0.6	-	China	LC-MS/MS	Wei *et al.* (2011)
	Bigeye	-	33	0.7	-	China	LC-MS/MS	Wei *et al.* (2011)
	Bleeker's gruper	-	10	0.0	-	China	LC-MS/MS	Wei *et al.* (2011)
	Goldspotted rabbitfish	-	15	0.7	-	China	LC-MS/MS	Wei *et al.* (2011)
	Golden threafin bream	-	36	0.8	-	China	LC-MS/MS	Wei *et al.* (2011)
	Orange-spotted grouper	-	9	0.7	-	China	LC-MS/MS	Wei *et al.* (2011)
	Snubose pompano	-	9	1.0	-	China	LC-MS/MS	Wei *et al.* (2011)
	Tongue sole	-	27	1.1	-	China	LC-MS/MS	Wei *et al.* (2011)
	Yellow croaker	-	15	0.9	-	China	LC-MS/MS	Wei *et al.* (2011)
	Yellow seafin	-	9	0.5	-	China	LC-MS/MS	Wei *et al.* (2011)
	Oreochromis mossanbicus and *niloticus*	100.0	6	30.8	9.4-51.8	Taiwan	LC-MS/MS	Chen *et al.* (2012)
	Fish, shrimp, squid	100.0	11	14.1	0.3-42.1	China	LC-MS/MS	Liao and Kannan (2013)
	Tilapia	-	10	1.4	-	China	LC-MS/MS	Wei *et al.* (2011)
	Tilapia	-	114	33.6	0.1-102.1	Taiwan	LC-MS/MS	Chen *et al.* 2014
	Tilapia	-	1	54.2[$]	-	China	GC-MS	Zhang *et al.* (2011)
	Carp	-	380	23.5	-	China	GC-MS	Zheng *et al.* (2015)
	Short neckled clam	-	1	181.3[$]	-	China	GC-MS	Zhang *et al.* (2011)
	Black seabream	-	1	177.4[$]	-	China	GC-MS	Zhang *et al.* (2011)
	Yellow fin seabream	-	1	93.0[$]	-	China	GC-MS	Zhang *et al.* (2011)
	Crucian	0.0	1	0.0	-	China	HPLC-FL	Wei *et al.* (2013)
	Common carp	-	30	1.6°	1.3-1.8°	Iran	GC-MS	Mortazavi *et al.* (2013)
	Weaver	0.0	1	0.0	-	China	HPLC-FL	Wei *et al.* 2013
	Bream	-	1	4.0	-	China	HPLC-FL	Wei *et al.* (2013)
	Nile tilapia	-	24	0.5	0.3-12.3	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Springer	-	2	0.7	0.3-1.4	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Black mullet	-	3	0.2	0.4-0.5	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Oxeye	-	4	0.4	0.3-2.7	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Grass carp	0.0	1	0.0	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Taiwan torrent carp	-	3	0.0	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Minnow	0.0	1	0.0	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Holland's crap	100.0	1	2.3	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Common carp	50.0	2	0.7	0.3-1.5	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Spotted catfish	-	4	6.3	0.3-25.2	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Striped snakehead	100.0	1	2.0	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Taiwan shoveljaw carp	0.0	1	0.0	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Milkfish	100.0	1	0.7	-	Taiwan	LC-MS/MS	Lee *et al.* (2015)
	Fish, river snail, clam	100.0	23	115.2°	37.3-475.0°	China	LC-MS/MS	Wang *et al.* (2015)

% Pos, percentage of samples above limit of detection/limit of quantification; GC-MS, gas chromatography coupled with mass spectrometry; LC-MS/MS, liquid chromatography coupled with tandem mass spectrometry; HPLC-FL, high-performance liquid chromatography coupled with fluorescence detection; HRGC-LRMS, high resolution gas chromatography/low resolution mass spectrometry; LC-FL, liquid chromatography coupled with fluorescence detection. *Area of commercialisation; °mean value and range expressed in dry weight; [#]measured limit of detection was 0.01 ppb; [§]product in contact with plastic; ^product in contact with metal tube; [$]mean value expressed in lipid weight. Mean values represent the average value for every seafood product in each article and are expressed in wet weight.

Lopez, 2005; Geens *et al.,* 2010). Moreover, the leaching, within the same coating type, shows great variability (Takao *et al.*, 2002; Munguia-Lopez *et al.*, 2005).

Wei *et al.* (2011), on the basis of their data, reported that a relatively great BPA concentration was observed in carnivorous fish (rice field eel and yellow croaker), both of freshwater and seawater. They suggest that carnivorous fish species can accumulate more pollutants since they are at the top of the food chain.

According to Staniszewska *et al.* (2014), diet and habitat are the most influential factors on BPA contamination in fish tissues, together with the lipid content: the higher BPA concentrations were found in herring muscle, followed by flounder and cod.

Data collected by Staniszewska *et al.* (2014) showed the possible bioaccumulation BPA in the food web, specifically on lower levels of the trophic chain (zooplankton and mussels).

Conclusions

BPA contamination levels, from all data considered, for both canned and non-canned seafood, were below the specific migration limit of 0.6 mg/kg set by the EC Directive for BPA in food or food simulants. Moreover, the canned seafood is more contaminated than non-canned one. There are no differences in BPA levels found in canned food in the considered continents, while non-canned seafood products show bigger differences.

Finally, the positions of authorities on food supply chain BPA contamination are sometimes conflicting. ANSES established that BPA is not harmless, despite the reassuring EFSA report, in which a series of recommendations are pointed out in order to better characterise the risk assessment. EFSA considers that further research on the potential adverse health effects of BPA will be useful, using validated and robust methodologies (European Parliament, 2015; EFSA, 2015c).

In spite of divergences, many studies agree that there are numerous scientific uncertainties regarding assessment of BPA. The latest recommendation of 2015 EFSA report, on which ANSES agrees, claims the need to *clarify the extent and the sources of unconjugated BPA in meat and fish* (EFSA, 2015c).

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