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Review

Occurrence of Bisphenol A and its analogues in some foodstuff marketed in Europe

Giacomo Russo[a,b], Francesco Barbato[a,b], Damiano Gustavo Mita[b,c], Lucia Grumetto[a,b,*]

[a] Pharm-Analysis & Bio-Pharm Laboratory, Department of Pharmacy, School of Medicine and Surgery, University of Naples Federico II, Via D. Montesano, 49, I-80131, Naples, Italy

[b] Consorzio Interuniversitario INBB, Viale Medaglie d'Oro, 305, I-00136, Rome, Italy

[c] National Laboratory on Endocrine Disruptors, National Institute of Biostructures and Biosystems (INBB), Institute of Genetics and Biophysics of CNR, Via Pietro Castellino, 111, 80131, Naples, Italy

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ABSTRACT

Bisphenol A and its analogues belong to the class of endocrine disrupting chemicals, massively employed by industries to produce polycarbonate and epoxy resins, designed to be in direct contact with foodstuffs. Their leaching from the canned packaging into its content results in food contamination. This review aims at offering a country-specific overview of the occurrence of bisphenols in six main categories of foodstuff marketed in the EU, based on monitoring studies performed in the 27 EU countries for which data are available and prevalently published in the last five years.

The general overview of the literature data shows that concentration values of BPs detected into foodstuff is lower in Northern Europe than Southern Europe. A probable daily intake was hypothesized for some countries to provide an EU population exposure assessment. The consumption of canned meat and vegetables is responsible of PDI values higher than those of other food categories. These data emphasize that food and beverage monitoring should deserve greater attention especially by European countries for which no studies are available and especially with regards to bisphenols other than BPA whose limits are not set by the European regulations and whose toxicity has not been fully established.

1. Introduction

Exposure to environmental pollutants is one of the risk factors that poses greatest concern for human health. Bisphenol A (BPA) (2,2-Bis (4-hydroxyphenyl)propane) is an environmental contaminant usually found in soil and water environments (Fromme et al., 2002; Kang et al., 2007; Staples et al., 1998) as well as in food, drinking water, dental sealants and indoor dusts (Calafat et al., 2008; Geens et al., 2009, 2011; Rudel et al., 2003; Vandenberg et al., 2007; Wilson et al., 2001). It is commonly used in the production of polycarbonates and epoxy resins intended as food contact materials such as plastic packaging, internal lining of cans and bottle caps, pipes and tanks for drinking water (Berge et al., 2017; Russo et al., 2017a; Younglai et al., 2002). BPA, along its analogous bisphenol S (BPS) (4,4′-sulfonyldiphenol), is also employed as colour trigger in the production of thermal paper (Russo et al., 2017b) and, due to the use of this paper in the recycling streams, even recycled carton and paper food packaging may contain bisphenols (BPs) as well (Liao and Kannan, 2011; Mendum et al., 2011; Russo et al., 2017b; Vinggaard et al., 2000). Nowadays, BPA represents one of the most abundant xenoestrogen investigated in food matrices, although other analogues were sometimes detected at even higher concentrations indicating that industries are progressively switching from Bisphenol A to other bisphenols in food packaging. Indeed, to minimize the BPA toxic effects, other BPs, *e.g.* BPS, Bisphenol B (BPB) (2,2-Bis(4-hydroxyphenyl)butane), and Bisphenol A diglycidyl ether (BADGE) (2-[[4-[2-[4-(Oxiran-2-ylmethoxy)phenyl]propan-2-yl]phenoxy]methyl]oxirane), are being used by industries (Chen et al., 2016).

Human exposure to BPs, which are ubiquitous in the environment, is virtually unavoidable and can occur through both dietary and non-dietary sources. However, the former is prevalent since BPs, BPA included, may contaminate food due to direct bioaccumulation in food matrices caused by environmental pollution or to their possible leaching from packaging (Repossi et al., 2016). Migration from packaging into food is modulated by various factors, including *i*) food

* Corresponding author. Pharm-Analysis & Bio-Pharm Laboratory Department of Pharmacy, School of Medicine and Surgery, University of Naples Federico II, Via D. Montesano, 49, I-80131, Naples, Italy.
E-mail address: grumetto@unina.it (L. Grumetto).

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Abbreviation list

BADGE	Bisphenol A diglycidyl ether
BADGE·H_2O	Bisphenol A (2,3-dihydroxypropyl) glycidyl ether
BADGE·$2H_2O$	Bisphenol A bis(2,3-dihydroxypropyl) ether
BADGE·HCl	Bisphenol A (3-chloro-2-hydroxypropyl) glycidyl ether
BADGE·2HCl	Bisphenol A bis(3-chloro-2-hydroxypropyl) ether
BADGE·HCl·H_2O	Bisphenol A (3-chloro-2-hydroxypropyl) (2,3-dihydroxypropyl) ether
BFDGE	Bisphenol F diglycidyl ether
BP	Bisphenol
BPA	Bisphenol A
BPB	Bisphenol B
BPM	Bisphenol M
BPS	Bisphenol S, Bisphenol Z, 4,4′-cyclohexylidenebisphenol
bw	Body weight
EC	European Community
ECHA	European Chemicals Agency
EDCs	Endocrine disrupting chemicals
EFSA	European Food Safety Authority
EPA	Environmental Protection Agency
ER	Estrogen Receptor
EU	European Union
FAO	Food and Agriculture organization of the United Nations
LOD	Limit of detection
LOQ	Limit of quantification
PDI	Probable Daily Intake
PE	Polyethylene
PEHD	High-density polyethylene
PET	Polyethylene terephthalate
PVC	Polyvinyl chloride
TDI	Total Daily Intake
WHO	World Health Organization
WWTPs	Wastewater Treatment Plants

composition (*e.g.* fat content); *ii*) whether or not the contact with food is direct; *iii*) contact time (the concentration of the migrating molecules in food is directly proportional to the square root of the contact time); *iv*) contact temperature and pH values (higher temperatures and lower pH induce higher migration rates); *v*) thickness of packaging material (thinner packages are linked with higher migration rates); *vi*) chemical nature and *vii*) amount of migrant compound (Repossi et al., 2016).

As can be seen in Fig. 1, all BP feature a scaffold consisting of two phenol rings, linked by a variously substituted carbon atom and the two phenol moieties are functionalized with various groups. Only Bisphenol M (BPM) supports three phenyl groups, instead. However, it is the core scaffold all BPs share that is believed to be directly responsible for the endocrine activity (Kitamura et al., 2005).

BPA belongs to the class of endocrine disrupting chemicals (EDCs) since it exerts estrogenic activity, even at concentrations below $1\ ng\ L^{-1}$. BPA and its analogues were demonstrated to interact with a variety of endogenous receptors, *e.g.* receptors of thyroid and glucocorticoid hormones, androgenic and estrogenic receptors (ER). In addition, BPA was demonstrated to modulate several serotonin- and dopamine-associated genes (Castro et al., 2015). Interactions with the ER occur via ER dependent pathways and BPs can act as either weak agonist or antagonist (MacKay and Abizaid, 2017; Molina-Molina et al., 2013; Watson et al., 2011). The widespread use of BPA is confirmed by the circumstance that more than 90% of world population, including infants, have its traces in blood and urine at concentration values ranging from 0.32 to $2.5\ ng\ mL^{-1}$ and from 0.11 to $946\ ng\ mL^{-1}$, respectively (Hanaoka et al., 2002; Liao and Kannan, 2012; Vandenberg et al., 2010). The increased urinary BPA concentration in adults was found linked to hypertension associated to obesity (Bae et al., 2012; Wang et al., 2015). Indeed, in a study carried out by Bae et al. 521 participants were recruited and the exposure to BPA was demonstrated to be associated with *a)* decreased heart rate variability (HRV), *b)* increased blood pressure and *c)* increased cardiovascular diseases (CVDs) (Bae et al., 2012; Wang et al., 2015). Extensive discussions of the effects of BPA exposure and its relashionship with several metabolic and cardiovascular diseases are reported in the literature (Rancière et al., 2015).

Upon ingestion by humans, BPA is metabolized by the liver into bisphenol A-glucuronide, a highly water-soluble compound, whose half-life is less than 6 h and whose complete elimination requires about 24 h (Volkel et al., 2002).

European Food Safety Authority (EFSA) established the quantity of a chemical substance that can be ingested daily without posing significant risks to health, namely the Tolerably Daily Intake (TDI). In the light of extensive studies published by the U.S. Environmental Protection Agency (EPA), in 2015 EFSA re-examined BPA exposure and toxicity issues, reducing the BPA TDI, previously set at $50\ \mu g\ kg^{-1}$ body weight (bw) day^{-1} to $4\ \mu g\ kg^{-1}$ bw day^{-1} (EFSA Scientific Opinion, 2015). The TDI for the sum of BADGE and its metabolites (BADGE·H_2O and BADGE·$2H_2O$) was set at $0.15\ mg\ kg^{-1}$ bw per day. For BADGE chlorohydrins (*i.e.* BADGE·2HCl, BADGE·HCl, and BADGE·HCl·H_2O), EFSA considered that the current restriction of $1\ mg\ kg^{-1}$ of food is still appropriate. No specific limits were indicated for BPs other than BPA, BADGE and its chlorohydrins. However, according to EFSA, there is a considerable uncertainty in the exposure estimate for the non-dietary sources (EFSA Scientific Opinion, 2013).

This review is aimed at offering a country-specific overview of the occurrence of BPA and its structural analogues in six main categories of foodstuff marketed in the EU, *i.e.* beverages, milk and dairy products, fish, meat, vegetables and fruits, pasta and cereals, and composite food.

This work should be viewed as an update of the last EFSA review, dated back to 2013 since it provides an extension of the EDCs examined, because not only BPA but also its analogues are here considered. This aspect represents a reason of novelty of the present paper. This review is based on peer-reviewed studies reporting the concentrations of BPs measured in food commodities, predominantly published in the last five years, in the nine of twenty-seven EU countries

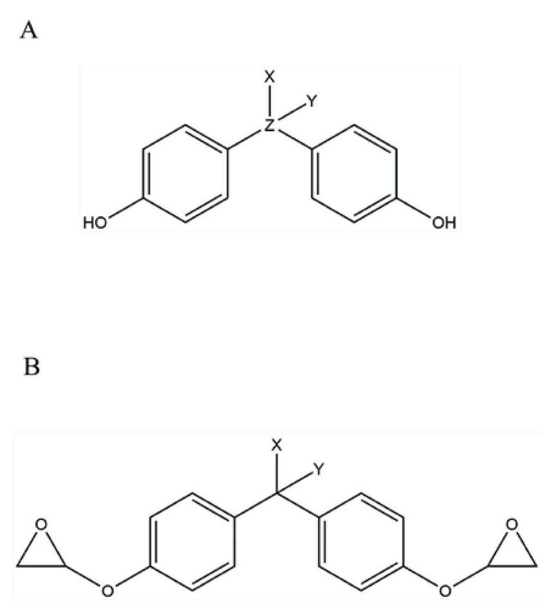

Fig. 1. Generic structures of (A) a bisphenol and (B) a bisphenol diglycidyl ether.

for which data are available. Data are crossed and grouped by investigated bisphenols, country of origin or area of marketing, kind of packaging, food subcategory, retrieved concentration range (minimum-maximum value), limit of detection (LOD) and limit of quantitation (LOQ) values and references. Unfortunately, monitoring studies have been conducted as patchwork in the various EU countries and no data are available for some of them. Therefore, the aim is neither to draw any conclusive evidence, but rather to summarize monitoring studies from the last years to possibly evaluate the trend of BP contamination of foodstuff marketed in EU.

Furthermore, this review provides a rough estimate of dietary BP exposure in the various EU countries based on a probabilistic approach. This was calculated by multiplying the average intake of a food by the maximum concentration found for each food category in that country (*i.e.* the worst scenario of contamination). Therefore, when monitoring studies are available, the Probable Daily Intake (PDI) of BPs is estimated by assuming it as an index of population contamination from the diet. The PDI computation was based on the food consumption data surveyed by EFSA in 2011, the most complete and detailed source currently available for EU. The EFSA survey is based on different methodologies used to collect the data and refers to different years of survey for the various countries. Unfortunately, no EFSA data were provided for Greece, Spain and Portugal and the survey refers to data provided in 2007 by KNOEMA, a US-based company of open statistic data platform (Knoema Inc., 2017).

2. Foodstuff

2.1. Beverages and drinking water

The possible occurrence of BPs in beverages and drinking water is justified by their presence in the coating materials of cans, bottle tops and water supply pipes (Amiridou and Voutsa, 2011). However, concentration values of BPs found in beverages are the lowest among other food categories, as shown in Table 1, and in many cases, BPs are not even found. This could most probably be due to *i*) the relatively high LODs of several performed analytical methods, *ii*) the different chemistry of the coating used in the two-piece easy open cans for soft drinks, and *iii*) the amount of the coatings applied in cans for soft drinks that is lower than that applied in cans designed for other foodstuff (Goodson et al., 2002; Horie et al., 1999; Kawamura et al., 2001; Thomson and Grounds, 2005). BPs were never found in glass bottled beverages but detected in all other canned and plastic packed products, suggesting that all types of packaging, except glass, can release BPs. BADGE is frequently used in the internal coating for beverage cans, because it entries in the production of epoxy phenolic resins, but BADGE is also an additive for the elimination of the excess of hydrochloric acid in the production of polyvinyl chloride (PVC) organosols (emulsion paints).

Many monitoring studies regarding the BP contamination of commonly consumed beverages were performed in EU. In Belgium, BPA was detected at concentrations ranging from 0.02 to 8.1 $\mu g\,L^{-1}$ in 75% of 45 analysed canned beverages, among which soft and energy drink, cola, beer and juices (Geens et al., 2010). BPA was found at detectable levels (LOD equal to 0.18 μg L-1 and LOQ equal to 0.54 μg L-1, respectively) in 57% of the carbonated beverages, in 50% of the non-carbonated beverages and in 100% of the milk based beverages in 71 Italian variously packaged beverages, including milk-based ones (Fasano et al., 2015). The monitoring study performed in Austria by Braunrath et al. in 2005 reported very low BPA levels ranging from "not-detectable" (0.2–0.8 $ng\,mL^{-1}$) to 3.4 $\mu g\,L^{-1}$ (Braunrath et al., 2005). Energy drink consumption has rapidly grown in recent years with a global consumption nearly doubling between 2006 and 2012 (Nowak and Jasionowski, 2015) with a marketing often youth-oriented (Harris and Munsell, 2015). In fact, in the last 20 years, the energy drink consumption has increased among children and teenagers with the young adults, aged 18–34, being the major demographic target

Table 1
BPs in beverages and drinking water differently packaged.

Bisphenol	Country	Year (ref)	Packaging	Subcategory	LOD $ng\,mL^{-1}$	LOQ[a] $ng\,mL^{-1}$	Min-Max $ng\,mL^{-1}$
BPA	Austria	2005 (Braunrath et al., 2005)	canned	soft/energy/beer	0.10–9.30	nr	*0.1–3.4*
BPA	Belgium	2010 (Geens et al., 2010)	canned, PET, Tetra Pak®,	soft and energy drinks, cola, beer, juice		0.02	*0.02–8.1*
BPA, BPB	Portugal	2011 (Cunha et al., 2011)	canned	soft drinks and beers	0.002–0.005	nr	*0.03–4.70*
BADGE, BFDGE and their derivatives	Spain	2011 (Gallart-Ayala et al., 2011b)	canned	soft drinks, cola, beer, tea, orange		130–4000	*2.3–5.1*
BPA + BPs	Spain	2011 (Gallart-Ayala et al., 2011a)	canned	soft drinks	0.025–0.050	0.084–0.167	*0.044–0.607*
BPA + BPs	Spain	2012 (Cacho et al., 2012)	canned	Sport drinks, tonic water, beer, soda, tea	0.0009–0.0025	0.0031–0.0084	*0.08–0.68*
BPA	Italy	2013 (Maggioni et al., 2013)	fountain, PET	water		0.00075	*0.00073–0.102*
BPA	Norway	2014 (Sakhi et al., 2014)	plastic, canned, cardboard box with plastic cap and metal foil	soft drinks, bottle water, juice		0.10	*0.02–0.37*
BPA	France	2014 (Bemrah et al., 2014)	not canned	drinking water, soft drinks, alcoholics	0.09		*0.04–5.960*
BPA + BPs	Italy	2014 (Errico et al., 2014)	canned	Energy drinks			*0.50–19.4*
BPA	Italy	2015 (Fasano et al., 2015)	canned, PET, aluminum, Tetra Pack®	Soft drinks	0.18	0.54	*0.54–4.98*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	plastic	soft drinks	0.10	0.40	*0.4–10.2*
BPA + BPS, BADGE, BFDGE	Italy	2017(Gallo et al., 2017)	canned	Energy drinks	0.15	0.50	*0.50–3.3*

[a] Nr = not reported.

(Pomeranz et al., 2013; Reid et al., 2017). In Italy, 40.6% in 2008 and 48.3% in 2010 of 8-9-year aged children consumed such drinks at least once a day (Nardone et al., 2014), due to the often-high content of caffeine, sugar, and other energizing ingredients (Harris and Munsell, 2015; Pomeranz et al., 2013; Reissig et al., 2009). A recent study performed by Bae and Hong (Bae and Hong (2015) demonstrated that the habitual consumption of canned drinks led to increased BPA blood levels, with effects on blood pressure. They indeed demonstrated that BPA levels in urine increased in patients consuming canned beverages by > 1600% as compared to those patients having glass bottled beverages. The same authors clinically observed an increase in the systolic blood pressure by ≈ 4.5 mm Hg after the consumption of only 2 canned beverages in comparison with patients having 2 glasses of bottled beverages (Bae et al., 2017).

A study commissioned by EFSA established that the highest prevalence of energy drink consumption in the EU population was represented by teenager consumers (68%) 56% of which Italians. About 12% (7% from Italy) of them were identified as "high chronic" consumers, *i.e.* consuming energy drinks 4–5 times *per* week or more (Zucconi et al., 2013). Therefore, despite the low concentration levels retrieved, BP contamination may be relevant because the amount of consumed liquids, *e.g.* beverages and drinking water, is much higher than that of the solids. Moreover, this represents a serious health issue for the chronic consumers, *i.e.* the youngest. Gallo et al. (Gallo et al. (2017) found that out of forty energy drink samples analysed, twenty-two samples (55.0%) were contaminated above the LOQs (0.50 $\mu g\ L^{-1}$) by one, two or three bisphenols. BPA was the BP more frequently detected, *i.e.* in 17 energy drinks (42.5% of samples) at contamination levels ranging from 0.50 to 3.3 $ng\ mL^{-1}$. Interestingly, these values are comparable to the levels in this foodstuff reported in the three studies performed in Spain in 2011 and 2012 (Cacho et al., 2012; Gallart-Ayala et al., 2011a, b).

No data are available on BPA levels in beverages marketed in France after the ban on BPA use in 2015, since the last study in this country was performed in 2014. Therefore, the source of BP contamination in this foodstuff cannot be demonstrated at this stage, although it seems to be strongly dependent upon packaging features and not upon the food matrix. However, this hypothesis is partly in contrast with the research of Maggioni et al. (2013) performed on water samples collected from 35 major Italian cities and on 5 Italian popular brands of polyethylene terephthalate (PET) bottled mineral water, to assess the quality of Italian drinking water. The highest BPA concentration was found in the water sampled from pipelines (0.102 $\mu g\ L^{-1}$) in Ferrara, a city located in Northern Italy, while in two brands of PET bottled mineral water BPA was detected at concentrations as low as 0.83–1.13 $ng\ L^{-1}$. Indeed, PET containers are not expected to release BPA, suggesting that the source of BPA in PET bottled water may be related to bottle closures, pollution of water prior to bottling, or the use of recycled PET (Bach et al., 2012). Table 1 lists the monitoring studies on beverages and drinking water marketed in EU.

2.1.1. Milk and dairy products

Milk is by far one of the most consumed food and, along with its derivatives, provides almost one third of the daily intake of proteins, being suitable for human consumption at all ages. Although in the last five years its consumption has been reduced on behalf of vegetable–based beverages, for the European population it amounts to about 65.85 Kg of milk *per* person *per* year (CLA Website, 2018). The growing demand of transport and the high costs of milk production within the boundaries led to the development of various containers in plastic materials, or in combination with paper, termed cartons, suitable for its safe carriage over long distances from "farm to fork". Although such containers are not believed a potential source of BP contamination in milk, being made of material such as PET that does not allow the release of bisphenols, recent studies revealed the occurrence of BPA in milk, even when the chemical nature of its packaging would not allow its release (Grumetto et al., 2013). This suggests that the presence of BPA into milk can also occur during its production throughout the supply chain, through the contact with utensils and equipment during the food processing. Another possible route for milk contamination from BPs is through animal feeding. BPA was determined either as single analyte or simultaneously with other BPs or alkylphenols, such as nonylphenol and octylphenol (Ferrer et al., 2011). The occurrence of BPs in milk and dairy products is presented in Table 2. For BPA, the lowest milk concentration found was 0.02 $\mu g\ kg^{-1}$ in Norwegian cardboard packaged milk (Sakhi et al., 2014), while the highest one was 800 $\mu g\ kg^{-1}$ (Ferrer et al., 2011) in canned skimmed milk from Spanish market. However, BPA concentrations as high as 521 $\mu g\ kg^{-1}$ and 169 $\mu g\ kg^{-1}$ are also reported in other two studies performed on milk marketed in Italy (Cirillo et al., 2015; Grumetto et al., 2013), in spite of the fact that milk was packaged in materials that do not allow any BPA release, such as PET, polyethylene (PE), high-density polyethylene (PEHD), and aluminium. Conversely, the other studies in Table 2, concerning the French, Swedish and Greek markets (Bemrah et al., 2014; Gyllenhammar et al., 2012; Maragou et al., 2006), reported very low concentrations of BPs, although the milk collected from the Swedish and Greek markets were both canned. Few information can be achieved from these studies about the possible sources of BP contamination.

Table 2
BPs in milk differently packaged.

Bisphenol	Country	Year (ref)	Packaging	Subcategory	LOQ $ng\ mL^{-1}$	LOD $ng\ mL^{-1}$	Min-Max $\mu g\ Kg^{-1}$
BPA, BADGE	Spain	2004 (Casajuana and Lacorte, 2004)	Tetra Brik®, HDPE, canned	Milk	0.15		*0.99–2.64*
BPA	Greece	2006 (Maragou et al., 2006)	canned	Milk	1.70	5.10	*1.7–15.2*
BPA	Spain	2011 (Ferrer et al., 2011)	canned	skimmed milk	5.00	16.00	*800*
BPA	Spain	2012 (Molina-Garcia et al., 2012)	canned	milk, powdered milk,	0.06	0.19	*0.38–5.47*
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	canned	milk, sour milk, yogurt, cream		2.0	*< LOQ –2.4*
BPA + BPs, BADGE, BFDGE	Italy	2013 (Grumetto et al., 2013)	Tetra Pak®, Tetra Brik®, PEHD, PET, PE	milk	0.30–4.20	1.00–14.00	*< 1 – 521*
BPA	Norway	2014 (Sakhi et al., 2014)	cardboard box	milk		0.10	*0.02*
			plastic	hard cheese			*0.72*
			plastic and metal foil	cheese spreads			*< 0.10*
			plastic, aluminium	brown cheese			*< 0.10*
BPA	France	2014 (Bemrah et al., 2014)	HDPE	milk	0.09		*0.045–1.654*
				ultrafresh dairy products			*0.045–2.018*
				cheese			*0.105–6.103*
BPA	Italy	2015 (Cirillo et al., 2015)	PET, Tetra Pak®, _aluminium	milk	3.00	9.00	*3–169*

Table 3
BPs in canned and not canned vegetables and fruits.

Bisphenol	Country	Year (ref)	Packaging	Subcategory	LOD ng mL^{-1} for liquid; ng g^{-1} for solid	LOQ ng mL^{-1} for liquid; ng g^{-1} for solid	Min- Max μg Kg^{-1}
BPA	Austria	2005 (Braunrath et al., 2005)	canned	haricots, beans, young peas, red kidney beans, lentils and crushed tomatoes.	1.10–7.40		*8.5–35*
BPA, BPB	Italy	2008 (Grumetto et al., 2008)	canned	tomato	15.4–20.0	51.3–66.9	*20.5–115.3*
BPA	Belgium	2010 (Geens et al., 2010)	canned, Tetra Pack®, glass, plastic	black olives, carrots, corn, carrots and peas, peeled tomatoes, bamboo, mushrooms, sieved tomato, red cabbage, pickles, green olives		0.10	*< LOQ -116.3*
BPA	Spain	2010 (Viñas et al., 2010)	canned	peas, carrot, artichoke, bean shoot, mixed vegetables	0.016	0.55	*8.90 –365*[a]
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	not canned	vegetables, potatoes		2	*< LOQ*
BPA + BPs	Spain	2012 (Cacho et al., 2012)	canned	Mushroom, asparagus, artichoke, olive, peas, pepper	0.0009–0.0025	0.0031–0.0084	*0.76–13.98*[b]
BPA, BPB	Portugal	2013 (Cunha and Fernandes, 2013)	canned	maize, mushroom, peas, sliced peeled tomato, soybean sprouts, spinach, green beans, carrots	0.60		*6.5–265.6*
BPA	France	2014 (Bemrah et al., 2014)	not canned	vegetables, pulses	0.21		*0.105–82.736*
BPA				potatoes			*0.105–2.556*
BPA	Norway	2014 (Sakhi et al., 2014)	canned, plastic	tomato, frozen vegetables		0.10	*0.10–5.4*
BPA + BPs	Spain	2014 (Alabi et al., 2014)	canned	red pepper, green beans chickpeas, lentils mushroom, asparagus	0.3–1.1	0.9–3.5	*7.1–959*
BPA	Italy	2014 (Errico et al., 2014)	canned	tomato	0.09	0.26	*0.31–235.88*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	canned	mushrooms, tomato, tomato paste	0.60	1.90	*4.9–66.0*
BPA	Austria	2005 (Braunrath et al., 2005)	canned	pineapple, peach in syrup lychees and mango	1.20–5.40		*5.0–24*
BPA	Belgium	2010 (Geens et al., 2010)	canned, Tetra Pack®, glass, plastic	peaches, pears, pineapple		0.10	*0.11–20.0*
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	not canned	fruit		2.00	*< LOQ*
BPA, BPB	Portugal	2013 (Cunha and Fernandes, 2013)	canned	pineapple in syrup, peach in syrup, passion fruit pulp, mango, pears in syrup and fruit cocktail	0.60		3.4–10.2
BPA	France	2014 (Bemrah et al., 2014)	not canned	fruits, dried fruits, nuts and seeds	0.21		*0.105–2.130*
BPA + BPs	Spain	2014 (Alabi et al., 2014)	canned	peaches, pineapple	0.30–1.10	0.90–3.50	*6.1–13*

[a] Data expressed as μg L^{-1}.
[b] Data expressed as μg L^{-1}.

Grumetto et al. found that the concentrations of BPA and its analogues in milk were not related to the fat content, but were probably dependent on other factors, such as food contact materials during milk processing and/or temperature and time of manufacturing (Grumetto et al., 2013). Furthermore, Grumetto and co-workers were the only authors that investigated the presence in this matrix of BPs other than BPA, namely BPB, BPF, BADGE, and BFDGE. They retrieved BPB concentrations as high as 67 μg kg^{-1}, which emphasizes the need to detect not only BPA, but also its structural congeners. However, even if the recent data are not complete enough for drawing some solid conclusions, the huge differences of BPA levels in milk found in the more recent studies regarding the Italian market compared to French market in the time frame 2013–2015 catches attention, particularly in light of the fact that France banned BPA in 2015. Therefore, it is reasonable to assume that packaging plays only a minor role in milk contamination, which probably depends on different thermic treatments and/or different utensils used for food processing. It is important to point out that BPs, secreted in cow milk, could be concentrated at higher levels particularly in fat dairy products. Unfortunately, most information gathered from the scientific literature regards BP levels in milk and only few information is provided about the contamination along the supply chain of dairy products, whose production could concentrate BPs.

2.2. Vegetables and fruits

The analyses of BPA and its analogues in non-canned vegetables and fruits have been rarely performed in EU countries, complicating an accurate estimate of their content. On the contrary, a reasonable estimate of the occurrence in canned vegetables and fruits is possible, because of several studies reporting the BP concentration levels in this foodstuff (Alabi et al., 2014; Bemrah et al., 2014; Braunrath et al., 2005; Cunha and Fernandes, 2013; Errico et al., 2014; Geens et al., 2010; Grumetto et al., 2008; Gyllenhammar et al., 2012; Sakhi et al., 2014; Tzatzarakis et al., 2017). Table 3 shows the results of monitoring studies on vegetables marketed in EU. As can be seen, from an analytical point of view, vegetables packed in cans, are an extremely heterogeneous subcategory because they show huge differences not only as matrix (mushrooms, tomatoes, lentils, etc.), but also in preservative medium employed for their conservation such as water, oil or not present at all. These differences apparently reflect the noticeable variability of the concentration levels of BPs found in this commodity. To date, literature reports the results of analyses on at least 18 different vegetable matrices. The highest BPA concentration was found by Alabi et al. (2014) in canned asparagus (959 μg kg^{-1}), occurring in samples collected in 2014 from Spanish market. Values significantly lower, but of the same order of magnitude (265.6, 235.88, 116.3, and 115.3 μg kg^{-1}), are reported in four studies, two performed in Italy on canned tomato (Errico et al., 2014; Grumetto et al., 2008), and two in Portugal and Belgium on various canned vegetables including tomato (Cunha and Fernandes, 2013; Geens et al., 2010). Geens et al. found a significant difference in concentration levels of BPs, between vegetables packed in metallic cans and in glass jars with a ratio of 100 between the levels found in these two types of packaging. For instance, Geens et al. (2010), when comparing BPA content of vegetable in these two different packaging, found higher contamination levels in metallic cans, with BPA concentrations in canned carrots and corn 60 and 25 folds higher than those in glass containers, respectively. Actually, BPA levels found in unpacked vegetable samples are usually lower than those found in the canned ones. Furthermore, a monitoring study performed in Portugal (Cunha and Fernandes, 2013) demonstrated that 87% of tested canned vegetables and fruit samples showed detectable levels of BPA, ranging from 3.4 to 265.6 μg kg^{-1}, with all the canned vegetable samples (100%) positive to BPA, while in canned fruits the percentage of occurrence was 72%. Tomato is the most investigated matrix (mentioned in seven works out of twelve). An extensive investigation of many canned tomato samples (42 samples) was reported by Grumetto et al. (2008). The authors found levels ranging from 20.5 to 115.3 μg kg^{-1}. This study on products marketed in Italy, also found BPB in nine tomato samples at levels ranging from 27.1 μg kg^{-1} to 85.7 μg kg^{-1}. A wider range of values (from 0.31 to 235.88 μg kg^{-1}) was reported in another monitoring study performed in Italy by Errico et al. (2014). The authors considered both intact and dented cans, and three different tomato types: peeled tomatoes, cherry tomatoes and concentrated tomato sauce. BPA was found in all types of investigated tomatoes, and its levels from the undamaged cans were smaller than those measured in the dented cans, confirming the hypothesis that damage to the cans, in the form of denting, has an appreciable effect on the BPA migration, leading to an increase of more than 30%. Even though BPs were found also in fresh vegetables, this data supports the hypothesis that the migration from packaging contributes significantly to the overall BP content in this foodstuff. This is in agreement with the results by Viñas et al. (2010) and Cacho et al. (2012), who analysed the suspension liquid of canned vegetables and inferred that bisphenol contamination occurred from the packaging. Regarding the research paper authored by Viñas and co-workers (Viñas et al., 2010), Table 3 summarizes only the concentration values of BPs other than BPS, since Cao (2019) casted some doubts regarding the identity of the peak originally attributed by the authors to BPS, that was indeed attributed by this author to an impurity. The extensive presence of BPA in canned vegetables had an indirect confirmation in a work of Braun et al. (2011), who found a positive relationship between their consumption and urinary BPA concentrations in humans. In fact, they demonstrated that the consuming of canned vegetables at least once *per* day was associated with BPA urine concentrations of about 2.3 μg g^{-1} as compared with those consuming no canned vegetables of about 1.6 μg g^{-1}.

BPA levels in fruits, mainly fruit in syrup, are also reported in Table 3. As can be seen, BP values are quite low, as compared to other food categories. The highest number of 78 investigated non-canned fruits is reported in a study on French market (Bemrah et al., 2014), where BPA levels ranged from 0.105 to 2.130 μg kg^{-1}. Another study by Cunha et al. (Cunha and Fernandes, 2013) on 20 canned fruits from Portuguese market, reported BPA levels ranging from 3.4 to 10.2 μg kg^{-1} with the highest level observed in peach in syrup. The highest BPA concentration in fruit was reported by Braunhart et al. (Braunrath et al., 2005), who found 24 μg kg^{-1} in canned mango in syrup marketed in Austria. A possible explanation to the relatively low levels of BPs found in fruit was given by Oldring and Nehring (Oldring PK, 2007) since usually food industry uses electrolytic tinplate instead of epoxy phenolic films in canned fruit containers. Due to the alleged inability of packaging to release BPs other than BPA, most of the studies did not consider their monitoring. However, a study performed on canned vegetables and fruit marketed in Portugal, detected BPB in two canned foodstuffs, *i.e.* fruit and vegetables, at levels as high as 3.0 μg kg^{-1} and 3.4 μg kg^{-1}, respectively (Cunha and Fernandes, 2013).

2.3. Seafood

In 2015, EU consumers spent 54 billion euros for buying fisheries and aquaculture products, reaching the highest amount ever recorded (ECHA, 2016). In that year, the EU seafood consumption *per capita* increased from 24.6 g/day/person, as surveyed in 2013, to 25.6 g/day/person, with the lowest value for Slovakia (9.4 g/day/person) and the highest for Norway (62.7 g/day/person). Due to sea pollution, fish contain significant levels of EDCs, among which BPA and its analogues as well as other environmental contaminants. In the years 2005–2006 the annual production volume of BPA in the EU increased to 1,150,000 metric tons (JRC Scientific and Technical Reports European Union Risk Assessment Report, 2008) and the high production volume of this plasticizer reflects in its constant detection in aquatic ecosystems. Indeed, environmental pollutants, including EDCs, are transferred via municipal and industrial wastewater to the sea because conventional wastewater treatment plants (WWTPs), based on activated sludge, often fail to remove them completely (Ting and Praveena, 2017). Therefore, sea and river organisms are impacted by WWTP effluents and continuously exposed to low doses of EDCs. Although these chemicals are usually present at low levels in oceans, they bio-magnificate and bio-accumulate in the aquatic food chain, such that their levels are generally higher in older and larger predatory fish and marine mammals. Although consumers can reduce their exposure by removing skin and fat from fish before cooking as filets, this practice does not allow eliminating EDCs because they are also distributed throughout the muscle, skinning and trimming. For instance, BPA concentrations in fresh fish were determined both in muscle and liver, since it is well known that exposure to contaminants will first result in a passage through the liver, after which the compound reaches the muscle (Ferrara et al., 2008; Mita et al., 2011). Furthermore, EDCs bio accessibility, *i.e.* the fraction released from the food matrix in the gastro-intestinal tract and available for absorption (Heaney, 2001) is higher in canned seafood than in other food matrices. Indeed, a study on BPA bioaccessibility in canned seafood revealed that it ranged from 80 to 99% (Cunha et al., 2017). This probably occurs because the migration from packaging adds to the levels already present in fresh fish tissues and seafood due to the release from plastic debris in oceans (Gatidou et al., 2010; Liu et al., 2012; Yang et al., 2014).

Even if the general assumption is that "fresh is best", for geographic and economic reasons the consumption of canned seafood is very popular in EU. Tuna (2.6 Kg *per capita* in 2014) and cod (2.4 Kg *per capita*), followed by salmon and Alaska Pollock, are the top consumed canned products.

Their consumption increased by 22% from 2012 to 2014 (Knoema Inc., 2007). Many studies investigated the concentration levels of BPs in canned fish and seafood as reported in Table 4, regarding tuna, sardines, cod, salmon, but also mussels and others. The only two monitoring studies concerning unpackaged river fish were performed in France (20–79 $\mu g\, kg^{-1}$) and Spain (223.91 $\mu g\, kg^{-1}$) (Jakimska et al., 2013; Miege et al., 2012). This contamination may be generated by the pollution of rivers and lakes since they collect industrial and domestic waste water from surrounding lands. However, few data are available to relate EDC levels in fishes with water pollution. An interesting monitoring study was performed in 2013 on Danube river pollution. Indeed, The Joint Danube Survey 3 of the International Commission for the Protection of the Danube River ICPDR 2013 (Liš;ka et al., 2013), reported that Danube, one of the five primary rivers in Europe, reaches BPA levels of a maximum concentration of 1.94 $\mu g\, L^{-1}$.

Filtering organisms, such as mussels, are chosen in monitoring studies as bio-indicators of pollution due to their capability of accumulating contaminants at higher concentrations and the limited ability to metabolize them (Jakimska et al., 2013; Miege et al., 2012). The EDCs in bivalve molluscs can affect human health by means of direct ingestion or because their bioaccumulation propagates along the food chain. A study by Gatidou et al. (2010) on various species of bivalves indicated that BPA concentrations were similar in Mediterranean mussels from various geographical areas (0.16–626.3 $\mu g\, kg^{-1}$). Furthermore, the study demonstrated that if BPA average concentration is 0.300 $\mu g\, L^{-1}$ in unfiltered seawater, Mediterranean mussels significantly bio-accumulate it and after 28 days the concentration levels reaches 1159 $\mu g\, kg^{-1}$ (dried weight) (Gatidou et al., 2010), a value of about 3000 times that at the beginning of the experiment. In contrast, Staniszewska et al. (2014) performed an EDCs monitoring study on various marine organisms, including mussels, of the Gulf of Gdańsk (Baltic Sea) and, based on the very low levels found, concluded that this coastal area was not contaminated from BPA and other EDCs and consequently, rather safe for humans.

As to the studies concerning river fish (Jakimska et al., 2013; Miege et al., 2012), the authors reported BP concentrations of about 200 $\mu g\, kg^{-1}$.

A special regard concerns tuna that canned or not, is one of the most consumed fish. Various studies on canned tuna samples found lower BP concentrations in liquid portions than in the respective solid part, with BPA concentrations generally 10-times higher in the solid food (Fattore et al., 2015) and with a higher presence in oily rather than aqueous preservation medium. As can be seen in Table 4, considering the research works regarding only canned tuna, the highest concentration

Table 4
BPs in canned and not canned seafood.

Bisphenols	Country	Year (ref)	Packaging	Subcategory	LOD ng g^{-1} for solid	LOQ ng g^{-1} for solid	Min-Max $\mu g\, Kg^{-1}$
BPA	Austria	2005 (Braunrath et al., 2005)	canned	tuna, sardines in oil	0.4		*2.1–43*
BPA	Belgium	2010 (Geens et al., 2010)	canned	tuna, salmon, anchovies,		0.10	*0.9–169.3*
BPA	Greece	2010 (Gatidou et al., 2010)	not canned	mussels	115[a]		*< LOD- 626.3*
BADGE, BFDGE	Spain	2012 (Miguez et al., 2012)	canned	mussel, anchovy, tuna pilchard, mackerel	0.50–3.10	1.80–10.30	*18–625*
BPA	France	2012 (Miege et al., 2012)	not canned	river fish		10.00–15.00	*20–79*
BPA	Spain	2012 (Salgueiro-Gonzalez et al., 2012)	not canned	Mussels	0.90	3.30	*11.2*
BPA BPB	Portugal	2012 (Cunha et al., 2012)	canned	tuna, sardines, mackerels, squid, octopuses, mussels, eels, anchovies	0.20–0.40		*1–99.9*
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	canned/not canned	not specified	2.00–4.00		*2.5–29*
BPA	Spain	2013 (Jakimska et al., 2013)	unpackaged	different fish from four Mediterranean rivers"	0.01[b]	0.04[c]	*223.91*
BPA + BPs	Spain	2014 (Alabi et al., 2014)	canned	cockles, mackerels, tuna	0.30–1.10	0.90–3.50	*7.2–662*
BPA	France	2014 (Bemrah et al., 2014)	not canned	fish, crustaceans and mollusks	0.21		*0.105–97.934*
BPA	Norway	2014 (Sakhi et al., 2014)	plastic	fish ball, frozen fish, caviar spread, codroe		0.10	*0.10–7.3*
BPA + BPs	Italy	2015 (Fattore et al., 2015)	canned	tuna	0.90–3.70	3.00–12.30	*6.3–187.0*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	canned	fish, seafood	0.60	1.90	*17.2–30.7*
BPA	Portugal	2017 (Cunha et al., 2017)	canned	tuna, sardines	0.01		*1–62*
BPA	France	2017 (Gorecki et al., 2017)	not canned	fish, crustaceans and mollusks	0.10 μg kg	0.30 μg/kg	*0.09–35.58*

[a] Dry weight.
[b] Dry weight.
[c] Dry weight.

levels of BPs (187 μg kg^{-1}) were found in a study conducted on 33 samples of canned tuna marketed in Italy in 2015 (Fattore et al., 2015). Alabi et al. (2014) also reported concentration values reaching values as high as 662 μg kg^{-1}, but the study considered not only tuna but also mussels, cockles and other canned seafood. Forty-one canned seafood from Spanish origin including mussels, anchovy, pilchards, mackerels and tuna were analysed by Miguez et al. (2012), but in 37 samples no BADGEs or BFDGEs were detected; only in the four tuna samples concentration values of BADGE derivatives from 18 to 625 Kg^{-1} were found. Overall, these data suggest that the seafood level of BPs arise from the polluted marine environment, while the migration from packaging contributes to a much lesser extent.

2.4. Meat

In 2015, EFSA published a scientific opinion underlining that canned and not canned meat were the two main BPA dietary contributors in most countries (EFSA; Gorecki et al., 2017). The contribution of canned and not canned meat products to the total dietary exposure to BPs ranged between 10% and 50% depending on the EU country and exposure scenario considered. As can be seen in Table 5, monitoring studies regarding meat products showed low values of BPA in various packaged meat from Norwegian (3.2 μg kg^{-1}) and Greek (0.6 μg kg^{-1}) markets (Sakhi et al., 2014; Tzatzarakis et al., 2017). On the other hand, the study performed in Spain by Alabi et al. in 2014 on the content of BPA and 11 BP derivatives in canned meat, reported BP values up to 630.0 μg kg^{-1} (Alabi et al., 2014). BPA was found in canned and not canned meat in the studies performed in Belgium and Sweden, reporting values of 26.7 and 13 μg kg^{-1}, respectively (Geens et al., 2010; Gyllenhammar et al., 2012). Two French studies by Bemrah et al. (2014) and Gorecki et al. (2017) deserve special merits as they dealt with the time evolution of bisphenol contamination in food. In such works, BPA content in various food commodities, gathered in the French market between 2007 and 2009, was compared with the values found in the same typology of food collected in 2015. Rather high BPA levels were found in the study carried out in 2014 in 173 meat-based products, *i.e.* in pooled liver (394.757 μg kg^{-1}) and cooked veal (224 μg kg^{-1}), whilst pork meat contained BPA concentrations higher than 20 μg kg^{-1}. Globally, the research work demonstrated the ubiquitous presence of BPA in this food with a background level of contamination of less than 5 μg kg^{-1} in 85% of the 1498 analysed samples, belonging to various foodstuff. On the other hand, the study published in 2017 showed a noticeable decrease in the BPA mean contaminations as compared to the previous study. Indeed, the mean concentration in meat was found to be 9.71 μg kg^{-1} in the samples collected between 2007 and 2009 against 2.78 μg kg^{-1} in the samples gathered in 2015 (Gorecki et al., 2017), with a concentration range spanning from 0.09 to 60.19 μg kg^{-1}. These studies allowed the assessment of both the BPA basic content in this food category, *i.e.* not arising from packaging migration, and the trend of BPA levels during the years. It should be underlined that French authorities banned BPA in the manufacture of food contact materials in 2015 (French legislation, 2010). However, among the not canned food categories, when all EU data for not canned meat and meat products surveyed by EFSA were pooled, average BPA concentration (middle bound) was 9.4 μg kg^{-1}. In comparison to that, fish and other seafood have an average concentration of 7.4 μg kg^{-1} (EFSA panel on food contact materials, 2013). As in all foods, the source of BPA in meat is due to migration from packaging and the contact with items during the transformation processes as well as to its previous presence in animal tissues because of the animal feeding. Overall, the studies summarized in Table 5 report concentration values ranging from 0.09 to 630 μg kg-1s in canned meat with an average concentration above 30 μg kg^{-1}.

2.5. Pasta and cereals

Cereals are an important source of energy for humans. Wheat production, in twenty-seven selected EU countries, is now estimated to be 134.4 million tons (Eurostat, 2017). Thus, BPs contamination in cereals might be a great source of dietary intake. Unfortunately, to date only six studies were conducted on BPA contamination and, among these, only the three studies from the Spanish market, considered also BPA derivatives (Alabi et al., 2014; Cacho et al., 2012; Viñas et al., 2010). The monitoring studies concerning pasta and cereals are listed in Table 6. Three typical varieties of cereals were tested for bisphenol contamination: rice, maize and wheat, packaged either in cans or in plastics. As can be seen, few studies are performed on this foodstuff. This is probably because the matrix is (*a*) solid, (*b*) not usually marketed in cans and (*c*) freshly consumed in most of the cases. The first study considered in this review was performed in 2010 by Geens et *al.* on samples of "*ravioli*", a typical meat filled Italian pasta, and corn (canned or in glass) collected from Belgium market. BPA levels found were up to 73.1 μg kg^{-1}. Levels of the same order of magnitude were reported for canned sweet corn in 2010 by Viñas et al. (2010) on the Spanish market and more recently (2017) by Tzatzarakis et al. (2017) for canned corn from Greek market. Much lower levels were found in three studies performed in 2012 on a large variety of cereals, in Norway, Sweden and Spain. The first study (Sakhi et al., 2014) did not consider canned cereals and reported BPA concentrations in the range 0.10–0.24 μg kg^{-1}. Cacho et al. (2012) in Spain, and Gyllenhammar et al. (2012) in Sweden found BPA levels not higher than 2.45 μg L^{-1} and 2.0 μg kg^{-1} respectively. It's important unlied that Cacho et al. expressed their results in μg L^{-1} since the analysis was conducted on filling liquids. However, the highest values are reported by Alabi et al. in 2014 on canned sweet corn from Spanish market (142 μg kg^{-1}) in the only study that considers as many as eleven BPA derivatives.

Table 5
Bisphenol in canned and not canned meat.

Bisphenol	Country	Year (ref)	Packaging	Subcategory	LOD ng g^{-1}	LOQ ng g^{-1}	Min-Max μg Kg^{-1}
BPA	France	2014 (Bemrah et al., 2014)	not canned	meat, poultry and game, offals, delicatessen meats	*0.21*		*0.105–394.757* *0.105–16.351*
BPA	Belgium	2010 (Geens et al., 2010)	Canned, glass	Meat		*0.10*	*0.86–26.7*
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	canned/not canned	Meat	2.00–4.00		*6.9–13*
BPA	Norway	2014 (Sakhi et al., 2014)	plastic, metal foil	minced meat, chicken fillet, sausages, hamburger, sliced salami, liver pate, sliced ham, sliced turkey		0.10	*0.10–3.2*
BPA + BPs	Spain	2014 (Alabi et al., 2014)	canned	Tripe meat balls	0.30–1.10	0.90–3.50	*39–630* *9.2–341*
BPA	France	2017 (Gorecki et al., 2017)	not canned	meat, poultry, game and offals, delicatessen meats	*0.02*	*1.2*	*0.09–60.19* *0.09–18.69*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	metal/Plastic	meat products			*0.6*

Table 6
Bisphenols in canned and not canned pasta and cereals.

Bisphenols	Country	Year (ref)	Packaging	Products	LOQ ng g^{-1}	Min-Max µg Kg^{-1}
BPA	Belgium	2010 (Geens et al., 2010)	canned canned glass	*ravioli*, corn	*0.10*	*0.94–73.1*
BPA, BP	Spain	2010 (Viñas et al., 2010)	canned	sweet corn		*11.7 – 31.5*[a]
BPA + BPs	Spain	2012 (Cacho et al., 2012)	canned	corn		*2.45*[b]
BPA	Sweden	2012 (Gyllenhammar et al., 2012)	canned/not canned	bread, flour, pasta rice, corn flakes, grain		*2 .0*
BPA + BPs	Spain	2014 (Alabi et al., 2014)	canned	sweet corn		*13–142*
BPA	Norway	2014 (Sakhi et al., 2014)	plastic, paper bag	bread, pasta, burns, breakfast cereals, flour		*0.10–0.24*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	canned	corn		*48.3*

[a] Data expressed as µg L^{-1}.
[b] Data expressed as µg L^{-1}.

Furthermore, high levels were also found in canned corn from Greek market (48.3 µg kg^{-1}) (Tzatzarakis et al., 2017).

2.6. Composite food and others

The category of composite food is reported in this review just for the sake of completeness, because any evaluation of BP content due to heterogeneity of the considered matrices is not feasible. A composite food is defined in European Union legislation as "*a foodstuff intended for human consumption that contains both processed products of animal origin and products of plant origin*". Due to the heterogeneity of the considered matrices, it is impossible any evaluation about the different studies. As evident from the subcategories listed in Table 7, the composition of these foods hugely varies, including both composite foods and single ingredients such as eggs, honey and oil, preventing a straightforward comparison among the different studies. As example, the monitoring study carried out in France by Bemrah et al. in 2014, analysed as many as 176 various composite food (Bemrah et al., 2014), and these food categories had a background BPA contamination spanning from 0.105 to 28.370 µg kg^{-1}. These categories, generally marketed as canned food, appear noticeably affected by packaging in BP content. In fact, the study by Carwile et al. reported that eating canned soup for five consecutive days increased urinary BPA concentration by > 1000% as compared to eating soup cooked with fresh ingredients (Carwile et al., 2011). This increase in urinary BPA is likely a transient peak, whose duration is still uncertain. The effect of such intermittent elevations in BPA urinary concentration is unknown. The only study found in literature before 2014 was by Petersen et al. (2003) which analysed four production lots of lightweight containers filled with goulash and potatoes. The sum of BADGE derivatives (BADGE·HCl, BADGE·2HCl, BADGE·HCl·H_2O, BADGE·H_2O, and BADGE·$2H_2O$) in all investigated lots spanned from 2139 to 2925 µg kg^{-1}, while the minimum value for each single derivative was of 25 µg kg^{-1} and the maximum of 1085 µg kg^{-1}. The average amount of BADGE and its derivatives of all 4 investigated lots (1680–2340 µg/kg) exceeded the regulatory limits, that should not be higher than 1000 µg kg^{-1} of food (Commission Regulation No, 1895/2005, 2005). A monitoring study was performed on honey by Česen et al. (2016) aimed at dosing BPA, BPAF, BPE, BPF, BPS and Bisphenol Z (BPZ, 4,4′-cyclohexylidenebisphenol) in thirty-six honey lots. The authors found BP concentration values spanning from 0.364 to 302 µg kg^{-1}. Anyway, concerning the composite food category, the concentration levels, reported in Table 7, would only indicate that various bisphenol analogues are currently used in manufacturing even if, in some cases, they probably derive from sources other than the final packaging. An interesting case of BP contamination from source different from packaging migrations is the natural occurrence of BPF documented in mild mustard made of the seeds of *Sinapis Alba L.* BPF forms from the breakdown of the glucosinolate glucosinalbin with 4-hydrobybenzyl alcohol. In other words, BPF forms during mustard production from a natural ingredient of mustard grains at so high concentrations that it may pose a risk to specific groups of the human population that consume mustard. Indeed, the consumption of 20 g of mustard can lead to an intake of 100–200 µg of BPF, an analogue structurally very similar to BPA and with similar biological effects (Rochester and Bolden, 2015). A relatively high value of BPF was also found in mustard seeds (8350 µg kg^{-1}) by Zoller et al. (2016).

3. Dietary intake

In 2006, the EU worst-case scenario for BPA daily intake was estimated to be up to 1.5 µg kg^{-1} bw for adults (EFSA panel on food contact materials, 2013). In 2011, the World Health Organization (WHO) estimated the mean dietary daily intake of BPA for adults to be 0.4–1.4 µg kg^{-1} bw, with a worst-case exposure scenario of 4.2 µg kg^{-1} bw for adults consuming 100% of coffee, tea and alcoholic drinks and 100% of solid food as packaged foods and beverages (Report of Joint FAO/WHO Expert Meeting, 2010; Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids and Materials in Contact with Food, 2006). Literature reports various studies regarding BPA intake from food, mainly canned, calculated considering the food consumption for each category (Bemrah et al., 2014; Kawamura et al., 2001; Lorber et al., 2015). Assessments of BPA intake are currently done either by both direct measurements in food items and considering the *per capita* consumption for each food category, or, indirectly, by back calculation from urinary BPA concentrations (Braun et al., 2011; Stacy et al., 2016). For instance, Kawamura et al. (2001) estimated a daily intake of BPA by multiplying the average concentration of BPA in canned food for each food category by the daily intake of the canned foods in each food group. The total BPA intake was then calculated by summing up the intakes from each food group. According to this procedure we calculate the maximum Probably Daily Intake (PDI) for each category; Fig. 2 reports the hypothesized BP intake for each foodstuff category in each European country, where monitoring studies are available. In other words, the PDI was obtained by multiplying the average intake of each main food group, without subdivisions into subcategories, by the maximum concentration of BPs found for each main food category in that country (*i.e.* the worst scenario). For the PDI assessment, we did not take into account the study by Cacho et al. (2012) and that by Viñas et al. (2010) because of BP levels for "pasta and cereals" and "vegetables" foodstuff categories were assessed in the preservative medium but not in the food itself. The *per capita* food consumption for each of the principal six categories is reported in Table S1. It should be noted that the intake data we report should be cautiously regarded because its assessment is rather difficult to achieve in an accurate way. This occurs since: *i*) complete data for nine EU countries are only available for seafood and vegetables; *ii*) the *per capita*

Table 7
Bisphenols in canned and not canned composite food and others.

Bisphenol	Country	Year (ref)	Packaging	Subcategory	LOD ng g^{-1}	LOQ ng g^{-1}	Min-Max µg Kg^{-1}
BPA + BADGE and its derivatives	Germany	2003 (Petersen et al., 2003)	canned	liver sausage, sliced cold meat, processed cheese with pepper, pork with lard, semolina pudding, mixed fruits, sweet rice dessert with fruits, hamburger with tomato sauce, lentil stew, vegetable pan, noodles with meat and sauce, cevapcici, goulash with potatoes	*7.80–23.70*		*25–1085*
BPA	France	2014 (Bemrah et al., 2014)	not-canned, PET	pizza, quiches, savory pastries, cakes, sandwiches snacks, soup and broths, mixed dishes, desserts and dairy-based desserts, compotes and cooked fruit, seasoning and sauces, food for particular uses, edible ices, chocolate and sugar derivatives	*0.21*		*0.105–28.370*
BPA	France	2014 (Bemrah et al., 2014)	canned, PET	eggs and egg products, butter, oil, margarine	*0.21*		*0.045–4.510*
BPA + BPs	European and non-European countries	2016 (Chen et al., 2016)	PET, plastic	honey	*0.000597–0.147*	*0.00199–0.489*	*0.364–302*
BPA	Italy	2016 (Lo Turco et al., 2016)	not canned	honey	*270.00*	*800.00*	*n.d*
BPF	Switzerland	2016 (Zoller et al., 2016)	not canned	mustard seeds	10.00	30.00	*10- 8350*
BPA	Greece	2017 (Tzatzarakis et al., 2017)	canned, Tetra Pak®	sauces	0.60	1.90	*n.d.*

food consumption of the various countries are gathered during a nine-year period from EFSA; *iii*) the consumption data for Spain, Greece and Portugal, for which no EFSA data are available, are gathered only in 2007 and taken from an alternative source; *iv*) different methodologies were used in the surveys both from EFSA and other sources to collect the data; *v*) most of monitoring studies considered only BPA and not its analogues. However, looking Fig. 2 some general considerations can still be drawn: *i*) the exposure to BPs from food sources in Europe is generally lower in the Northern countries than in the Southern ones; *ii*) the highest PDI values concern canned meat and vegetables, while the lowest ones are for canned fruits; *iii*) migration from packaging into beverages and drinking waters is as important as migration to solid food. Overall, these data may suggest that the use of BPA-free plastic materials predominates in the countries with lower PDIs (*e.g.* France and Scandinavia). Fig. 2H depicts the number of foodstuff monitoring studies for each EU country. As can be seen, only two countries, *i.e.* Spain and Italy, have 12 and 8 studies in the last ten years, respectively. On the contrary, no studies or reports are available for as many as twenty EU countries. EFSA recently issued a comprehensive report concerning the concentration of BPA in foodstuff in EU. The document considered both canned and non-canned foods (EFSA panel on food contact materials, 2013). The ratio in BP concentrations between canned and non-canned foods ranged from 3 to 500 times for the different food categories suggesting that the contamination could also realize during food processing and production (Pjanic, 2017). The European Chemicals Agency (ECHA) officially declared that the exposure to BPA poses serious human health concerns (ECHA CLA Website, 2018), following a report provided by the French government, that in 2015 banned BPA from any food packaging (ECHA). Therefore, ECHA revised the list of dangerous substances and included BPA. On the other hand, in 2015 EFSA declared that the BPA has no role in increasing the risk concerning the human health at the current exposure levels. The Commission Directive 2004/19/EC defined a specific migration limit in foods and beverages as high as 0.6 mg kg^{-1} and 9.0 mg kg^{-1} of food for BPA and BADGE, respectively. On 23rd August 2017, the European Commission (EC) published the draft regulation “Bisphenol A in varnishes and coatings and plastics intended to come into contact with food”. This document sets a new specific migration limit for BPA amounting to 0.05 mg kg^{-1} food from varnishes or coatings applied to materials or articles intended for food contact. Furthermore, this draft establishes that BPA migration is not allowed at any extent from plastic materials and articles designed to be in direct contact with infants' and toddlers’ food. Even if most BP concentrations reported in the collected research papers are too low to produce PDI exceeding the levels considered safe by EFSA, thus reasonably excluding any risk of acute toxicity, the debate whether or not low doses of BPs can have adverse health effects is still controversial (Wang et al., 2017). However, the PDI values in Fig. 2 consider only one food category at once, neglecting the sums of the PDIs arising from a complete dietary regimen based on the combination of various foodstuffs (ECHA, 2016; EFSA panel on food contact materials, 2013; Pjanic, 2017) while toxic effects on the human health can arise both from the chronic toxicity due to bioaccumulation and from the simultaneous exposition (and consequent intake) to EDCs (“cocktail effect”).

4. Conclusion

The main purpose of this paper was the analysis of the various monitoring studies in which BPA and its analogues were found in foodstuff marketed in EU, mainly in the last five years. Unfortunately, such information is decidedly lacking with regards to twenty EU countries, for which monitoring studies regarding BPA were conducted but no data are available on the contamination from BPs other than BPA. The results of this review suggest a progressive shift by industries of metal cans from BPA to other BP monomers in manufacturing of food packaging and allowing their consequent possible migration into food.

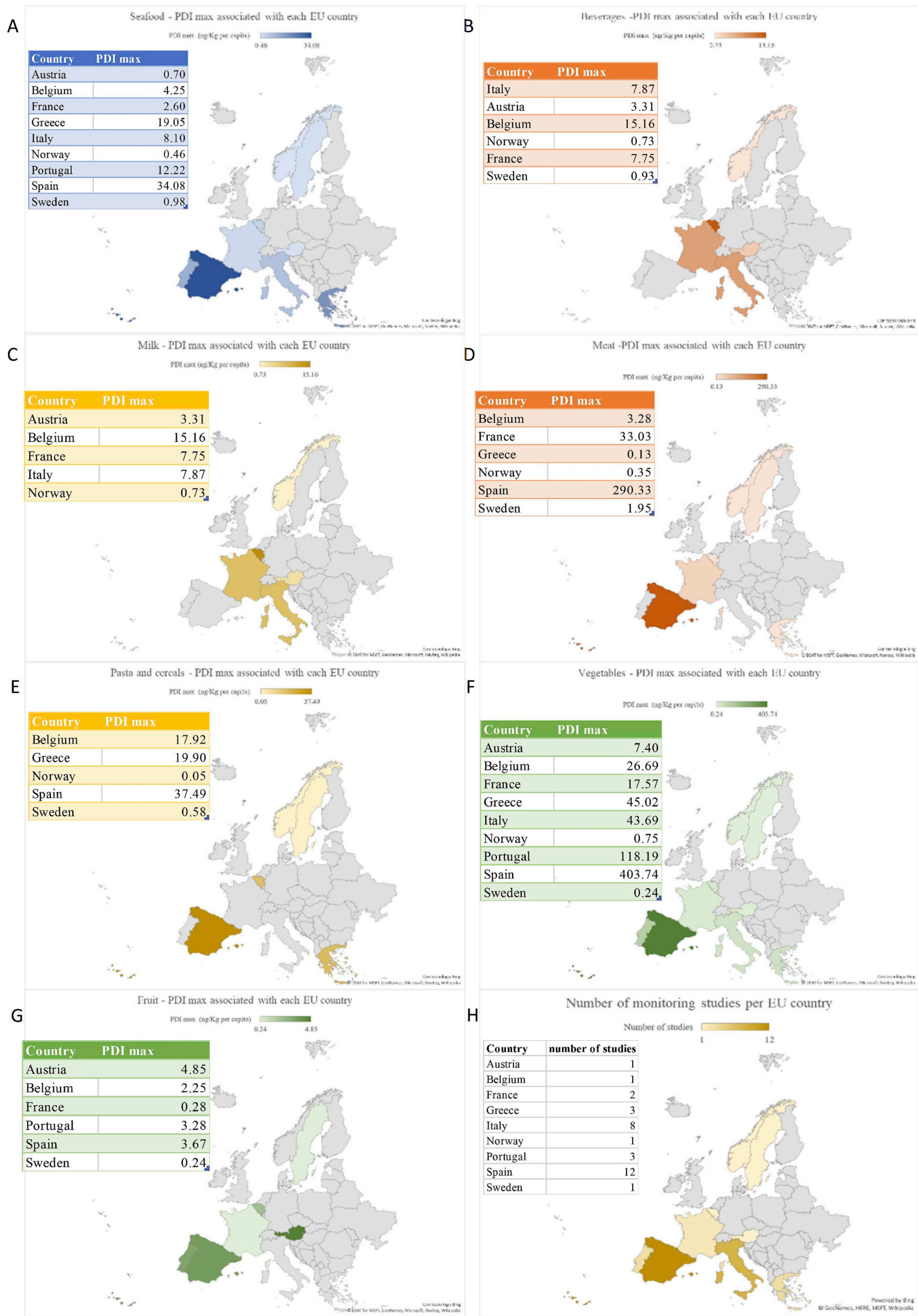

Fig. 2. Country-specific probable daily intake based on the monitoring studies reported in the literature and human consumption data surveys. *A*, seafood; *B*, beverages; *C*, milk; *D*, meat; *E*, pasta and cereals; *F*, vegetable; *G*, fruit; *H*, number of monitoring studies for each EU country.

Anyhow, it is necessary to keep in mind that BPA analogues are steadily being used for various industrial applications and their safety as compared to BPA is still far to be demonstrated. In the light of these findings, a constant and geographically-complete monitoring of BP presence in foodstuffs on the markets is strongly desirable to increase risk awareness regarding this issue. Furthermore, a precautionary principle must be applied even if detected levels of BPs other than BPA are below the legal limits.

Conflicts of interest

The authors do not declare any competing financial interest.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.fct.2019.110575.

Transparency document

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References

Česen, M., Lambropoulou, D., Laimou-Geraniou, M., Kosjek, T., Blaznik, U., Heath, D., Heath, E., 2016. Determination of bisphenols and related compounds in honey and their migration from selected food contact materials. J. Agric. Food Chem. 64, 8866–8875.

Alabi, A., Caballero-Casero, N., Rubio, S., 2014. Quick and simple sample treatment for multiresidue analysis of bisphenols, bisphenol diglycidyl ethers and their derivatives in canned food prior to liquid chromatography and fluorescence detection. J. Chromatogr. A 1336, 23–33.

Amiridou, D., Voutsa, D., 2011. Alkylphenols and phthalates in bottled waters. J. Hazard Mater. 185, 281–286.

Bach, C., Dauchy, X., Chagnon, M.C., Etienne, S., 2012. Chemical compounds and toxicological assessments of drinking water stored in polyethylene terephthalate (PET) bottles: a source of controversy reviewed. Water Res. 46, 571–583.

Bae, S., Hong, Y.C., 2015. Exposure to bisphenol A from drinking canned beverages increases blood pressure: randomized crossover trial. Hypertension 65, 313–319.

Bae, S., Kim, J.H., Lim, Y.H., Park, H.Y., Hong, Y.C., 2012. Associations of bisphenol A exposure with heart rate variability and blood pressure. Hypertension 60, 786–793.

Bae, S., Lim, Y.H., Lee, Y.A., Shin, C.H., Oh, S.Y., Hong, Y.C., 2017. Maternal urinary bisphenol a concentration during midterm pregnancy and children's blood pressure at age 4. Hypertension 69, 367–374.

Bemrah, N., Jean, J., Riviere, G., Sanaa, M., Leconte, S., Bachelot, M., Deceuninck, Y., Bizec, B.L., Dauchy, X., Roudot, A.C., Camel, V., Grob, K., Feidt, C., Picard-Hagen, N., Badot, P.M., Foures, F., Leblanc, J.C., 2014. Assessment of dietary exposure to bisphenol A in the French population with a special focus on risk characterisation for pregnant French women. Food Chem. Toxicol. 72, 90–97.

Berge, T.L., Lygre, G.B., Jonsson, B.A., Lindh, C.H., Bjorkman, L., 2017 Nov. Bisphenol A concentration in human saliva related to dental polymer-based fillings. Clin. Oral Investig. 21 (8), 2561–2568.

Braun, J.M., Kalkbrenner, A.E., Calafat, A.M., Bernert, J.T., Ye, X., Silva, M.J., Barr, D.B., Sathyanarayana, S., Lanphear, B.P., 2011. Variability and predictors of urinary bisphenol A concentrations during pregnancy. Environ. Health Perspect. 119, 131–137.

Braunrath, R., Podlipna, D., Padlesak, S., Cichna-Markl, M., 2005. Determination of bisphenol A in canned foods by immunoaffinity chromatography, HPLC, and fluorescence detection. J. Agric. Food Chem. 53, 8911–8917.

Cacho, J.I., Campillo, N., Viñas, P., Hernandez-Cordoba, M., 2012. Stir bar sorptive extraction coupled to gas chromatography-mass spectrometry for the determination of bisphenols in canned beverages and filling liquids of canned vegetables. J. Chromatogr. A 1247, 146–153.

Calafat, A.M., Ye, X., Wong, L.Y., Reidy, J.A., Needham, L.L., 2008. Exposure of the U.S. population to bisphenol A and 4-tertiary-octylphenol: 2003-2004. Environ. Health Perspect. 116, 39–44.

Cao, X.L., 2019. Letter to the Editor regarding "Comparison of two derivatization-based methods for solid-phase microextraction-gas chromatography-mass spectrometric determination of bisphenol A, bisphenol S, and biphenol migrated from food cans. Anal. Bioanal. Chem. 411, 287–288.

Carwile, J.L., Ye, X.Y., Zhou, X.L., Calafat, A.M., Michels, K.B., 2011. Canned soup consumption and urinary bisphenol a: a randomized crossover trial. JAMA, J. Am. Med. Assoc. 306, 2218–2220.

Casajuana, N., Lacorte, S., 2004. New methodology for the determination of phthalate esters, bisphenol A, bisphenol A diglycidyl ether, and nonylphenol in commercial whole milk samples. J. Agric. Food Chem. 52, 3702–3707.

Castro, B., Sanchez, P., Torres, J.M., Ortega, E., 2015. Bisphenol A, bisphenol F and bisphenol S affect differently 5alpha-reductase expression and dopamine-serotonin systems in the prefrontal cortex of juvenile female rats. Environ. Res. 142, 281–287.

Chen, D., Kannan, K., Tan, H., Zheng, Z., Feng, Y.L., Wu, Y., Widelka, M., 2016. Bisphenol analogues other than BPA: environmental occurrence, human exposure, and toxicity-A review. Environ. Sci. Technol. 50, 5438–5453.

Cirillo, T., Latini, G., Castaldi, M.A., Dipaola, L., Fasano, E., Esposito, F., Scognamiglio, G., Francesco, F.D., Cobellis, L., 2015. Exposure to di-2-ethylhexyl phthalate, di-n-butyl phthalate and bisphenol A through infant formulas. J. Agric. Food Chem. 63, 3303–3310.

CLA Website, 2018. https://www.clal.it/?section=tabs_consumi_procapite.

, Commission Directive 2004/19 EC Office Journal of European Union.

Commission regulation (EC) No 1895/2005 of 18 November 2005 on the restriction of use of certain epoxy derivatives in materials and articles intended to come into contact with food. Off. J. Eur. Union L 302/28.

Cunha, S.C., Fernandes, J.O., 2013. Assessment of bisphenol A and bisphenol B in canned vegetables and fruits by gas chromatography-mass spectrometry after QuEChERS and dispersive liquid -liquid microextraction. Food Control 33, 549–555.

Cunha, S.C., Almeida, C., Mendes, E., Fernandes, J.O., 2011. Simultaneous determination of bisphenol A and bisphenol B in beverages and powdered infant formula by dispersive liquid-liquid micro-extraction and heart-cutting multidimensional gas chromatography-mass spectrometry. Food Addit. Contam. Part A Chem Anal Control Expo Risk Assess 28, 513–526.

Cunha, S.C., Cunha, C., Ferreira, A.R., Fernandes, J.O., 2012. Determination of bisphenol A and bisphenol B in canned seafood combining QuEChERS extraction with dispersive liquid-liquid microextraction followed by gas chromatography-mass spectrometry. Anal. Bioanal. Chem. 404, 2453–2463.

Cunha, S.C., Alves, R.N., Fernandes, J.O., Casal, S., Marques, A., 2017. First approach to assess the bioaccessibility of bisphenol A in canned seafood. Food Chem. 232, 501–507.

ECHA, 2016. https://echa.europa.eu/chemicals-in-our-life/hot-topics/bisphenol-a. edition. https://ec.europa.eu/fisheries/eu-fish-market-2016-edition_en.

EFSA, 2011. Use of the EFSA comprehensive european food consumption database in exposure assessment. EFSA J. 9 (3), 2097. 2011. http://onlinelibrary.wiley.com/doi/10.2903/j.efsa.2011.2097.

EFSA, 2013. panel on food contact materials, e., flavourings and processing Aids. DRAFT Scientific opinion on the risks to public health related to the presence of bisphenol A (BPA) in foodstuffs - Part: Exposure assessment.

EFSA, 2015. Scientific Opinion on the Risks to Public Health Related to the Presence of Bisphenol A (BPA) in Foodstuffs.

Errico, S., Bianco, M., Mita, L., Migliaccio, M., Rossi, S., Nicolucci, C., Menale, C., Portaccio, M., Gallo, P., Mita, D.G., Diano, N., 2014. Migration of bisphenol A into canned tomatoes produced in Italy: dependence on temperature and storage conditions. Food Chem. 160, 157–164.

Eurostat, 2017. http://ec.europa.eu/eurostat/statistics-explained/index.php/Agricultural_production_-_crops#Cereals.

Fasano, E., Esposito, F., Scognamiglio, G., Di Francesco, F., Montuori, P., Amodio Cocchieri, R., Cirillo, T., 2015. Bisphenol A contamination in soft drinks as a risk for children's health in Italy. Food Addit. Contam. Part A Chem Anal Control Expo Risk Assess 32, 1207–1214.

Fattore, M., Russo, G., Barbato, F., Grumetto, L., Albrizio, S., 2015. Monitoring of bisphenols in canned tuna from Italian markets. Food Chem. Toxicol. 83, 68–75.

Ferrara, F., Ademollo, N., Delise, M., Fabietti, F., Funari, E., 2008. Alkylphenols and their ethoxylates in seafood from the Tyrrhenian Sea. Chemosphere 72, 1279–1285.

Ferrer, E., Santoni, E., Vittori, S., Font, G., Mañes, J., Sagratini, G., 2011. Simultaneous determination of bisphenol A, octylphenol, and nonylphenol by pressurised liquid extraction and liquid chromatography–tandem mass spectrometry in powdered milk and infant formulas. Food Chem. 126, 360–367.

French legislation tendant à suspendre la commercialisation de tout conditionnement comportant du bisphénol A et destiné à recevoir des produits alimentaires in: N. SASX1008554L (2010).

Fromme, H., Kuchler, T., Otto, T., Pilz, K., Muller, J., Wenzel, A., 2002. Occurrence of phthalates and bisphenol A and F in the environment. Water Res. 36, 1429–1438.

Gallart-Ayala, H., Moyano, E., Galceran, M.T., 2011a. Analysis of bisphenols in soft drinks by on-line solid phase extraction fast liquid chromatography-tandem mass spectrometry. Anal. Chim. Acta 683, 227–233.

Gallart-Ayala, H., Moyano, E., Galceran, M.T., 2011b. Fast liquid chromatography-tandem mass spectrometry for the analysis of bisphenol A-diglycidyl ether, bisphenol F-diglycidyl ether and their derivatives in canned food and beverages. J. Chromatogr. A 1218, 1603–1610.

Gallo, P., Di Marco Pisciottano, I., Esposito, F., Fasano, E., Scognamiglio, G., Mita, G.D., Cirillo, T., 2017. Determination of BPA, BPB, BPF, BADGE and BFDGE in canned energy drinks by molecularly imprinted polymer cleaning up and UPLC with fluorescence detection. Food Chem. 220, 406–412.

Gatidou, G., Vassalou, E., Thomaidis, N.S., 2010. Bioconcentration of selected endocrine disrupting compounds in the Mediterranean mussel, Mytilus galloprovincialis. Mar. Pollut. Bull. 60, 2111–2116.

Geens, T., Roosens, L., Neels, H., Covaci, A., 2009. Assessment of human exposure to Bisphenol-A, Triclosan and Tetrabromobisphenol-A through indoor dust intake in Belgium. Chemosphere 76, 755–760.

Geens, T., Apelbaum, T.Z., Goeyens, L., Neels, H., Covaci, A., 2010. Intake of bisphenol A from canned beverages and foods on the Belgian market. Food Addit. Contam. Part A Chem Anal Control Expo Risk Assess 27, 1627–1637.

Geens, T., Goeyens, L., Covaci, A., 2011. Are potential sources for human exposure to bisphenol-A overlooked? Int. J. Hyg Environ. Health 214, 339–347.

Goodson, A., Summerfield, W., Cooper, I., 2002. Survey of bisphenol A and bisphenol F in canned foods. Food Addit. Contam. 19, 796–802.

Gorecki, S., Bemrah, N., Roudot, A.C., Marchioni, E., Le Bizec, B., Faivre, F.,

Kadawathagedara, M., Botton, J., Riviere, G., group, E.m.-c.c.s, 2017. Human health risks related to the consumption of foodstuffs of animal origin contaminated by bisphenol A. Food Chem. Toxicol. 110, 333–339.
Grumetto, L., Montesano, D., Seccia, S., Albrizio, S., Barbato, F., 2008. Determination of bisphenol a and bisphenol B residues in canned peeled tomatoes by reversed-phase liquid chromatography. J. Agric. Food Chem. 56, 10633–10637.
Grumetto, L., Gennari, O., Montesano, D., Ferracane, R., Ritieni, A., Albrizio, S., Barbato, F., 2013. Determination of five bisphenols in commercial milk samples by liquid chromatography coupled to fluorescence detection. J. Food Prot. 76, 1590–1596.
Gyllenhammar, I., Glynn, A., Darnerud, P.O., Lignell, S., van Delft, R., Aune, M., 2012. 4-Nonylphenol and bisphenol A in Swedish food and exposure in Swedish nursing women. Environ. Int. 43, 21–28.
Hanaoka, T., Kawamura, N., Hara, K., Tsugane, S., 2002. Urinary bisphenol A and plasma hormone concentrations in male workers exposed to bisphenol A diglycidyl ether and mixed organic solvents. Occup. Environ. Med. 59, 625–628.
Harris, J.L., Munsell, C.R., 2015. Energy drinks and adolescents: what's the harm? Nutr. Rev. 73, 247–257.
Heaney, R.P., 2001. Factors influencing the measurement of bioavailability, taking calcium as a model. J. Nutr. 131, 1344S–1348S.
Horie, M., Yoshida, T., Ishii, R., Kobayashi, S., Nakazawa, H., 1999. Determination of bisphenol A in canned drinks by LC MS. Bunseki Kagaku 48, 579–587.
Jakimska, A., Huerta, B., Barganska, Z., Kot-Wasik, A., Rodriguez-Mozaz, S., Barcelo, D., 2013. Development of a liquid chromatography-tandem mass spectrometry procedure for determination of endocrine disrupting compounds in fish from Mediterranean rivers. J. Chromatogr. A 1306, 44–58.
JRC Scientific and Technical Reports European Union Risk Assessment Report, 2008. http://publications.jrc.ec.europa.eu/repository/bitstream/JRC59988/lbna24589enn.pdf.
Kang, J.H., Asai, D., Katayama, Y., 2007. Bisphenol A in the aquatic environment and its endocrine-disruptive effects on aquatic organisms. Crit. Rev. Toxicol. 37, 607–625.
Kawamura, Y., Inoue, K., Nakazawa, H., Yamada, T., Maitani, T., 2001. [Cause of bisphenol A migration from cans for drinks and assessment of improved cans]. Shokuhin Eiseigaku Zasshi 42, 13–17.
Kitamura, S., Suzuki, T., Sanoh, S., Kohta, R., Jinno, N., Sugihara, K., Yoshihara, S., Fujimoto, N., Watanabe, H., Ohta, S., 2005. Comparative study of the endocrine-disrupting activity of bisphenol A and 19 related compounds. Toxicol. Sci. 84, 249–259.
Knoema Inc, 2007. https://knoema.com/atlas/topics/Food-Security.
Liao, C., Kannan, K., 2011. Widespread occurrence of bisphenol A in paper and paper products: implications for human exposure. Environ. Sci. Technol. 45, 9372–9379.
Liao, C., Kannan, K., 2012. Determination of free and conjugated forms of bisphenol A in human urine and serum by liquid chromatography-tandem mass spectrometry. Environ. Sci. Technol. 46, 5003–5009.
Liška, I., Wagner, F., Sengl, M., Deutsch, K., Slobodník, J., 2013. Joint Danube Survey 3 of the International Commission for the Protection of the Danube River ICPDR - A Comprehensive Analysis of Danube Water Quality.
Liu, J., Wang, R., Huang, B., Lin, C., Zhou, J., Pan, X., 2012. Biological effects and bioaccumulation of steroidal and phenolic endocrine disrupting chemicals in high-back crucian carp exposed to wastewater treatment plant effluents. Environ. Pollut. 162, 325–331.
Lo Turco, V., Di Bella, G., Potorti, A.G., Tropea, A., Casale, E.K., Fede, M.R., Dugo, G., 2016. Determination of plasticisers and BPA in Sicilian and Calabrian nectar honeys by selected ion monitoring GC/MS. Food Addit. Contam. Part A Chem Anal Control Expo Risk Assess 33, 1693–1699.
Lorber, M., Schecter, A., Paepke, O., Shropshire, W., Christensen, K., Birnbaum, L., 2015. Exposure assessment of adult intake of bisphenol A (BPA) with emphasis on canned food dietary exposures. Environ. Int. 77, 55–62.
MacKay, H., Abizaid, A., 2017. A plurality of molecular targets: the receptor ecosystem for bisphenol-A (BPA). Horm. Behav.
Maggioni, S., Balaguer, P., Chiozzotto, C., Benfenati, E., 2013. Screening of endocrine-disrupting phenols, herbicides, steroid estrogens, and estrogenicity in drinking water from the waterworks of 35 Italian cities and from PET-bottled mineral water. Environ. Sci. Pollut. Res. Int. 20, 1649–1660.
Maragou, N.C., Lampi, E.N., Thomaidis, N.S., Koupparis, M.A., 2006. Determination of bisphenol A in milk by solid phase extraction and liquid chromatography-mass spectrometry. J. Chromatogr. A 1129, 165–173.
Mendum, T., Stoler, E., VanBenschoten, H., Warner, J.C., 2011. Concentration of bisphenol A in thermal paper. Green Chem. Lett. Rev. 4, 81–86.
Miege, C., Peretti, A., Labadie, P., Budzinski, H., Le Bizec, B., Vorkamp, K., Tronczynski, J., Persat, H., Coquery, M., Babut, M., 2012. Occurrence of priority and emerging organic compounds in fishes from the Rhone River (France). Anal. Bioanal. Chem. 404, 2721–2735.
Miguez, J., Herrero, C., Quintas, I., Rodriguez, C., Gigosos, P.G., Mariz, O.C., 2012. A LC-MS/MS method for the determination of BADGE-related and BFDGE-related compounds in canned fish food samples based on the formation of [M + NH(4)](+) aducts. Food Chem. 135, 1310–1315.
Mita, L., Bianco, M., Viggiano, E., Zollo, F., Bencivenga, U., Sica, V., Monaco, G., Portaccio, M., Diano, N., Colonna, A., Lepore, M., Canciglia, P., Mita, D.G., 2011. Bisphenol A content in fish caught in two different sites of the Tyrrhenian Sea (Italy). Chemosphere 82, 405–410.
Molina-Garcia, L., Fernandez-de Cordova, M.L., Ruiz-Medina, A., 2012. Analysis of Bisphenol A in milk by using a multicommuted fluorimetric sensor. Talanta 96, 195–201.
Molina-Molina, J.M., Amaya, E., Grimaldi, M., Saenz, J.M., Real, M., Fernandez, M.F., Balaguer, P., Olea, N., 2013. In vitro study on the agonistic and antagonistic activities of bisphenol-S and other bisphenol-A congeners and derivatives via nuclear receptors. Toxicol. Appl. Pharmacol. 272, 127–136.
Nardone, P., Spinelli, A., Buoncristiano, M., Lauria, L., Pizzi, E., Andreozzi, S., Galeone, D., 2014. Il Sistema di sorveglianza OKkio alla SALUTE: risultati. available. http://old.iss.it/binary/publ/cont/ONLINE_Okkio.pdf.
Nowak, D., Jasionowski, A., 2015. Analysis of the consumption of caffeinated energy drinks among polish adolescents. Int. J. Environ. Res. Public Health 12, 7910–7921.
Oldring PK, N.U., 2007. Packaging Materials 7: Metal Packaging for Foodstuffs. ILSI Europe.
Opinion of the scientific panel on food additives, Flavourings, processing Aids and materials in contact with food on a request from the commission related to 2,2-BIS(4-ydroxyphenyl)Propane (bisphenol A). EFSA J. 428 1 of 75. http://www.efsa.europa.eu/en/science/afc/afc_opinions.html.
Petersen, H., Schaefer, A., Buckow, C.A., Simat, T.J., Steinhart, H., 2003. Determination of bisphenol A diglycidyl ether (BADGE) and its derivatives in food: identification and quantification by internal standard. Eur. Food Res. Technol. 216, 355–364.
Pjanic, M., 2017. The role of polycarbonate monomer bisphenol-A in insulin resistance. PeerJ 5, e3809.
Pomeranz, J.L., Munsell, C.R., Harris, J.L., 2013. Energy drinks: an emerging public health hazard for youth. J. Public Health Policy 34, 254–271.
Rancière, F., Lyons, J.G., Loh, V.H., Botton, J., Galloway, T., Wang, T., Shaw, J.E., Magliano, D.J., 2015. Bisphenol A and the risk of cardiometabolic disorders: a systematic review with meta-analysis of the epidemiological evidence. Environ. Health. May 31, 14–46.
Reid, J.L., McCrory, C., White, C.M., Martineau, C., Vanderkooy, P., Fenton, N., Hammond, D., 2017. Consumption of caffeinated energy drinks among youth and young adults in Canada. Prev. Med. Rep. 5, 65–70.
Reissig, C.J., Strain, E.C., Griffiths, R.R., 2009. Caffeinated energy drinks–a growing problem. Drug Alcohol Depend. 99, 1–10.
Report of Joint FAO/WHO Expert Meeting 2–5 November 2010 and Report of Stakeholder Meeting on Bisphenol A 1 November 2010 Ottawa, Canada. http://apps.who.int/iris/bitstream/10665/44624/1/97892141564274_eng.pdf.
Repossi, A., Farabegoli, F., Gazzotti, T., Zironi, E., Pagliuca, G., 2016. Bisphenol a in edible part of seafood. Ital. J. Food. Saf. 5, 5666.
Rochester, J.R., Bolden, A.L., 2015. Bisphenol S and F: a systematic review and comparison of the hormonal activity of bisphenol a substitutes. Environ. Health Perspect. 123, 643–650.
Rudel, R.A., Camann, D.E., Spengler, J.D., Korn, L.R., Brody, J.G., 2003. Phthalates, alkylphenols, pesticides, polybrominated diphenyl ethers, and other endocrine-disrupting compounds in indoor air and dust. Environ. Sci. Technol. 37, 4543–4553.
Russo, G., Barbato, F., Cardone, E., Fattore, M., Albrizio, S., Grumetto, L., 2017a. Bisphenol A and Bisphenol S release in milk under household conditions from baby bottles marketed in Italy. J. Environ. Sci. Health B 1–5.
Russo, G., Barbato, F., Grumetto, L., 2017b. Monitoring of bisphenol A and bisphenol S in thermal paper receipts from the Italian market and estimated transdermal human intake: a pilot study. Sci. Total Environ. 599–600, 68–75.
Sakhi, A.K., Lillegaard, I.T., Voorspoels, S., Carlsen, M.H., Loken, E.B., Brantsaeter, A.L., Haugen, M., Meltzer, H.M., Thomsen, C., 2014. Concentrations of phthalates and bisphenol A in Norwegian foods and beverages and estimated dietary exposure in adults. Environ. Int. 73, 259–269.
Salgueiro-Gonzalez, N., Turnes-Carou, I., Muniategui-Lorenzoa, S., Lopez-Mahia, P., Prada-Rodriguez, D., 2012. Fast and selective pressurized liquid extraction with simultaneous in cell clean up for the analysis of alkylphenols and bisphenol A in bivalve molluscs. J. Chromatogr. A 1270, 80–87.
Stacy, S.L., Eliot, M., Calafat, A.M., Chen, A., Lanphear, B.P., Hauser, R., Papandonatos, G.D., Sathyanarayana, S., Ye, X., Yolton, K., Braun, J.M., 2016. Patterns, variability, and predictors of urinary bisphenol a concentrations during childhood. Environ. Sci. Technol. 50, 5981–5990.
Staniszewska, M., Falkowska, L., Grabowski, P., Kwasniak, J., Mudrak-Cegiolka, S., Reindl, A.R., Sokolowski, A., Szumilo, E., Zgrundo, A., 2014. Bisphenol A, 4-tert-octylphenol, and 4-nonylphenol in the Gulf of gdansk (southern Baltic). Arch. Environ. Contam. Toxicol. 67, 335–347.
Staples, C.A., Dorn, P.B., Klecka, G.M., O'Block, S.T., Harris, L.R., 1998. A review of the environmental fate, effects, and exposures of bisphenol A. Chemosphere 36, 2149–2173.
Thomson, B.M., Grounds, P.R., 2005. Bisphenol A in canned foods in New Zealand: an exposure assessment. Food Addit. Contam. 22, 65–72.
Ting, Y.F., Praveena, S.M., 2017. Sources, mechanisms, and fate of steroid estrogens in wastewater treatment plants: a mini review. Environ. Monit. Assess. 189, 178.
Tzatzarakis, M.N., Karzi, V., Vakonaki, E., Goumenou, M., Kavvalakis, M., Stivaktakis, P., Tsitsimpikou, C., Tsakiris, I., Rizos, A.K., Tsatsakis, A.M., 2017. Bisphenol A in soft drinks and canned foods and data evaluation. Food Addit. Contam. Part B Surveill 10, 85–90.
Vandenberg, L.N., Hauser, R., Marcus, M., Olea, N., Welshons, W.V., 2007. Human exposure to bisphenol A (BPA). Reprod. Toxicol. 24, 139–177.
Vandenberg, L.N., Chahoud, I., Heindel, J.J., Padmanabhan, V., Paumgartten, F.J., Schoenfelder, G., 2010. Urinary, circulating, and tissue biomonitoring studies indicate widespread exposure to bisphenol A. Environ. Health Perspect. 118, 1055–1070.
Viñas, P., Campillo, N., Martinez-Castillo, N., Hernandez-Cordoba, M., 2010. Comparison of two derivatization-based methods for solid-phase microextraction-gas chromatography-mass spectrometric determination of bisphenol A, bisphenol S and biphenol migrated from food cans. Anal. Bioanal. Chem. 397, 115–125.
Vinggaard, A.M., Korner, W., Lund, K.H., Bolz, U., Petersen, J.H., 2000. Identification and quantification of estrogenic compounds in recycled and virgin paper for household use as determined by an in vitro yeast estrogen screen and chemical analysis. Chem. Res. Toxicol. 13, 1214–1222.

Volkel, W., Colnot, T., Csanady, G.A., Filser, J.G., Dekant, W., 2002. Metabolism and kinetics of bisphenol a in humans at low doses following oral administration. Chem. Res. Toxicol. 15, 1281–1287.
Wang, T., Xu, M., Xu, Y., Lu, J., Li, M., Chen, Y., Wang, W., Lai, S., Bi, Y., Ning, G., 2015. Association of bisphenol a exposure with hypertension and early macrovascular diseases in Chinese adults: a cross-sectional study. Medicine (Baltim.) 94, e1814.
Wang, Z., Liu, H., Liu, S., 2017. Low-dose bisphenol a exposure: a seemingly instigating carcinogenic effect on breast cancer. Adv. Sci. 4, 1600248.
Watson, C.S., Jeng, Y.J., Guptarak, J., 2011. Endocrine disruption via estrogen receptors that participate in nongenomic signaling pathways. J. Steroid Biochem. Mol. Biol. 127, 44–50.
Wilson, N.K., Chuang, J.C., Lyu, C., 2001. Levels of persistent organic pollutants in several child day care centers. J. Expo. Anal. Environ. Epidemiol. 11, 449–458.
Yang, J., Li, H., Ran, Y., Chan, K., 2014. Distribution and bioconcentration of endocrine disrupting chemicals in surface water and fish bile of the Pearl River Delta, South China. Chemosphere 107, 439–446.
Younglai, E.V., Foster, W.G., Hughes, E.G., Trim, K., Jarrell, J.F., 2002. Levels of environmental contaminants in human follicular fluid, serum, and seminal plasma of couples undergoing in vitro fertilization. Arch. Environ. Contam. Toxicol. 43, 121–126.
Zoller, O., Bruschweiler, B.J., Magnin, R., Reinhard, H., Rhyn, P., Rupp, H., Zeltner, S., Felleisen, R., 2016. Natural occurrence of bisphenol F in mustard. Food Addit. Contam. Part A Chem Anal Control Expo Risk Assess 33, 137–146.
Zucconi, I., Volpato, C., Adinolfi, F., Gandini, E., Gentile, E., Loi, A., Fioriti, L., March 2013. Gathering consumption data on specific consumer groups of energy drinks. EFSA Support. Publ. 10 (3) 394E.