ENDORSED FOR PUBLIC CONSULTATION

DRAFT SCIENTIFIC OPINION

DRAFT Scientific Opinion on the risks to public health related to the presence of bisphenol A (BPA) in foodstuffs – Part: exposure assessment[1]

EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF)[2, 3]

European Food Safety Authority (EFSA), Parma, Italy

ABSTRACT

The EFSA asked its Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF) to provide a scientific opinion on bisphenol A (BPA). As important toxicological studies on BPA are to be published shortly, and hazard identification/characterisation requires further discussions, a two-step approach for public consultation on the draft opinion on BPA has been taken. The current draft thus addresses only the assessment of exposure to BPA. Total exposure to BPA was estimated by two different procedures, one involving exposure modelling and the other urinary biomonitoring data. Exposure modelling involved the assessment of exposure to BPA through different sources (food and non-food) and routes of exposure (oral, inhalation and dermal) in the EU population. Data on BPA concentrations in food were combined with food consumption data to estimate dietary exposure and concentration data in/from non-food sources were combined with behaviour patterns to estimate non-dietary exposure. Diet was found to be the main source of exposure to BPA in all population groups, but modelled estimates were much lower than the estimates reported by EFSA in 2006. In the previous assessment, high exposure was up to 5 300 ng/kg bw/day in toddlers and up to 11 000 ng/kg bw/day in infants aged 3 months, compared with the current estimates of up to 857 ng/kg bw/day for toddlers and up to 495 ng/kg bw/day for infants of 1-5 days. Thermal paper was the second source of exposure in all population groups above 3 years of age. The uncertainty around the estimate of exposure to BPA from thermal paper was considerably higher than that around dietary exposure. Biomonitoring estimates based on urinary BPA concentrations are in good agreement with modelled BPA exposures from all sources, suggesting that no major exposure sources have been missed for the modelled exposure assessment.

KEY WORDS

Bisphenol A, exposure assessment, food and non-food sources

[1] On request from EFSA, Question No EFSA-Q-2012-00423.

[2] Panel members: Ulla Beckman Sundh, Mona-Lise Binderup, Claudia Bolognesi, Leon Brimer, Laurence Castle, Alessandro Di Domenico, Karl-Heinz Engel, Roland Franz, Nathalie Gontard, Rainer Gürtler, Trine Husøy, Klaus-Dieter Jany, Martine Kolf-Clauw, Catherine Leclercq, Wim Mennes, Maria Rosaria Milana, Maria de Fátima Tavares-Poças, Iona Pratt, Kettil Svensson, Fidel Toldrá and Detlef Wölfle. Correspondence: cef@efsa.europa.eu

[3] Acknowledgement: The Panel wishes to thank the members of the BPA Exposure WG: Emma Bradley, Catherine Leclercq, Inger Therese Laugsand Lillegaard, Ralph Pirow, Iona Pratt, Catherine Simoneau, Maria de Fátima Tavares-Poças, Jacqueline Van Engelen, Natalie von Götz and EFSA staff: Davide Arcella and Francesco Pomilio for the support provided to this scientific opinion.

SUMMARY

The European Food Safety Authority (EFSA) asked the Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF) to provide a scientific opinion on the risks for public health related to the presence of bisphenol A (BPA) in foodstuffs. In particular, the opinion should:

(i) evaluate the toxicity of BPA for humans, including for specific (vulnerable) groups of the population (e.g. pregnant women, infants and children, etc.) and considering all relevant toxicological information available;

(ii) carry out an exposure assessment on the basis of the occurrence data available in the public domain and other occurrence data that may be available, and quantify as far as possible not only dietary exposure but also exposure from non-dietary sources;

(iii) consider specifically the exposure situation for the supposedly most vulnerable groups of the population (e.g. pregnant women, infants and children, etc.) and take into account, if available, biomonitoring data when assessing the exposure and compare the results with the calculated exposure; and

(iv) characterise the human health risks taking into account specific groups of the population.

Taking into account that important toxicological studies on BPA are to be published shortly, and acknowledging that the hazard identification and characterisation of BPA requires further discussions before endorsement, a two-step approach for public consultation on the draft opinion on BPA was proposed by the CEF Panel. The current draft document thus addresses the 2nd and 3rd part of the terms of reference only i.e. the assessment of exposure to BPA. The full draft opinion on BPA is intended to be released for public consultation at a later stage.

The previous exposure assessment of BPA by EFSA from 2006 did not consider non-dietary sources of exposure and was based on basic conservative assumptions in relation to BPA occurrence in food. In the present opinion, a detailed analysis of data becoming available since 2006 on food consumption and BPA occurrence in food was performed. Furthermore, in the present opinion non-food sources of exposure to BPA have also been addressed.

BPA uses

BPA is used in the manufacture of polycarbonate (PC) plastics, epoxy resins and other polymeric materials, and also for certain paper products (e.g. thermal paper). PC is used for food and liquid containers such as tableware (plates and mugs), microwave ovenware, cookware, reservoirs for water dispensers and non-food applications such as toys and pacifiers with PC shields. BPA-based epoxyphenolic resins are used as protective linings for food and beverage cans and as a coating on residential drinking water storage tanks. BPA is also used in a number of non-food-related applications, e.g. epoxy resin based paints, medical devices, surface coatings, printing inks and flame retardants.

General approach taken for the assessment

Average and high total chronic BPA exposure was assessed in the different age classes, considering the supposedly vulnerable groups: infants, children and women of childbearing age (in order to address potential exposure in the fetus and in breastfed infants). For food the average exposure was assessed based on average concentration and average consumption data, while high exposure was based on average concentration and high consumption. In the present opinion BPA concentrations have been assigned to more detailed food categories than in the earlier EFSA opinion on BPA. For non-food sources, to estimate average exposure the average values for all parameters were chosen. To estimate the high exposure from non-food sources, the same average parameters were used for absorption rates and occurrence data but in line with the methodology used to assess exposure from

food, the frequency of use parameters was modified to account approximately for the highest 95th percentile among all EU countries.

Total exposure to BPA was estimated by two different procedures independent of each other: one was based on exposure modelling calculations and the other on urinary biomonitoring data. Exposure modelling involved the assessment of chronic exposure (absorbed dose) to BPA through different sources (diet, thermal paper, air, dust, toys, cosmetics, dental sealants) and routes of exposure (oral, inhalation and dermal) in the EU population. Analytical/experimental BPA concentrations were combined with food consumption (including human milk) to estimate dietary exposure and concentration data in and from non-food sources with behaviour patterns to estimate non-dietary exposure. Then, total average exposure was calculated by adding up average exposure from all dietary and non-dietary sources. Total high exposure was calculated by adding up high levels of exposure from the two highest sources and average exposure levels from all other sources.

These modelled calculations aimed to assess the total daily amount of BPA absorbed by the body by any route. The absorption factors considered in these calculations were 1 for oral, 1 for inhalation and 0.3 for dermal. The results provide an estimate comparable to that obtained by assessing total daily urinary excretion of BPA. However, while urinary biomonitoring provides estimates of total exposure only, modelling allows estimation of exposure from all the sources of exposure which could be identified and quantified individually. In order to quantify the relative impact of each source, the assumptions made in the exposure estimates were aimed at obtaining a similar degree of conservativeness among the different sources.

The current draft opinion is thus focused on the modelled exposure (absorbed dose) of consumers to BPA (through different routes), taking into account the different absorption factors for the different routes of exposure, and the comparison of these exposure estimates with the total daily urinary excretion of BPA, assessed by urinary biomonitoring. The uncertainty in the exposure estimates was assessed systematically both for the modelling and the biomonitoring approach. The estimates do not reflect the proportion of the BPA dose bioavailable (unconjugated BPA) after absorption by the body and subsequent metabolism. The conversion of the exposure estimates from each source into internal (bioavailable) doses of BPA has not yet been considered. This conversion into internal doses needs to be considered in the subsequent step of risk characterisation of BPA. Uncertainties affecting the parameters that will be used for this conversion are not considered in the present document but will be taken into consideration in later steps of the risk assessment of BPA.

All data on BPA occurrence in food and non-food sources and all biomonitoring data have undergone a thorough quality check before being considered in the assessments. Whenever available data from Europe were considered for the quantitative assessment, while non-European data related to BPA have been used for comparison purposes.

Assessments for BPA exposure in specific disease states, occupational exposure of workers handling BPA containing products, or acute exposure (with the exception of dental materials) to BPA were not developed in this opinion.

Dietary exposure

Dietary exposure to BPA has been estimated in different population groups by combining information on the levels of BPA in food with the corresponding consumption levels.

Information on BPA occurrence in food has been derived from EFSA's call for data together with a systematic review of scientific literature covering the period 2006 until December 2012. For biomonitoring data literature published before 2006 was also included in order to increase the information for certain countries or matrices, e.g. human milk.

A total of 2 521 samples of food and beverages were selected as the basis to assess BPA concentrations in the different food categories for the scope of the present opinion. Data from the literature and from the call for data did not show major differences in BPA concentrations and so have been merged for each food category. These merged BPA concentrations have been used in the exposure calculations.

Left-censored data, i.e. from samples with concentrations below the limit of detection (LOD) or quantification (LOQ), were handled through the substitution method. The lower bound (LB) was obtained by assigning a value of zero to all the samples reported as less than the left-censoring limit, the middle bound (MB) by assigning half of the left-censoring limit, and the upper bound (UB) by assigning the left-censored limit (LOD or LOQ) as the sample result.

Systematic differences in BPA concentration between canned and non-canned food were observed in a large majority of food categories, with higher BPA concentrations in the canned food. Seven out of 17 canned food categories presented an average (MB) BPA concentration above 30 µg/kg ("Grain and grain-based products", "Legumes, nuts and oilseeds", "Meat and meat products", "Fish and other seafood", "Herbs, spices and condiments", "Composite food", and "Snacks, desserts, and other foods"). Lower levels were found in other categories and in particular average (MB) BPA concentration was lower than 3 µg/kg in canned beverages (water, alcoholic and non alcoholic beverages, fruit and vegetables juices). Among the 19 non-canned food categories, the highest levels of BPA were found in the categories "Meat and meat products" and "Fish and other seafood" with average (MB) BPA concentrations of 9.4 and 7.4 µg/kg, respectively. When comparing European with non-European concentration data for food, average BPA levels were mostly in the same range.

In residential buildings where water pipes had been repaired with a two-components technique the average and high BPA concentrations in cold water were 0.1 and 1.1 µg/l, respectively. These values have been considered when calculating exposure through drinking water in specific population groups.

Biomonitoring studies suggested relatively high levels of BPA in the initial human milk (colostrum), which is produced during the first to approximately 5th day after delivery, compared with mature human milk. The CEF Panel noted that only very few data from Europe and/or obtained by a reliable analytical method were available and therefore decided to take into account data from Japan, reporting an average BPA concentration of 3 µg/l and a modelled high concentration estimate of 6.6 µg/l in initial human milk. However, these data from Japan were obtained using ELISA methodology and the samples dated back to 2000. These limitations were addressed in the uncertainty analysis. Based on different studies, the average and high concentrations of total BPA in mature human milk were found to be 1.2 µg/l and 2.6 µg/l, respectively.

BPA migration data from food packaging materials into food simulants, retrieved from the literature and EFSA's call for data, were used to assess the exposure of specific groups of consumers. In particular, average BPA migration levels were derived for the following PC articles: water coolers with PC reservoirs (0.81 µg/l in water), PC water kettles (0.11 µg/l in warm water), PC filters (0.04 µg/l in water), PC tableware and cookware (0.09 and 0.29 µg/l, respectively, in foods that can be consumed hot).

Data from the EFSA Comprehensive European Food Consumption Database were used to assess dietary exposure to BPA in all age groups, from infants (6-12 months) to the elderly and very elderly (older than 65 years), excluding infants aged 0 to 6 months. Consumption data observed in toddlers were used as an estimate of consumption in infants aged 6 to12 months since no data were available in the latter age class. Consumption data from a total of 32 different dietary surveys carried out in 22 different Member States covering more than 67 000 individuals are included in the Comprehensive Database. In order to consider separately women of childbearing age, in the present assessment the adult age group has been broken down in three subgroups, comprising women from 18 to 45 years, men from 18 to 45 years and other adults from 45 to 65 years. Only a limited number of dietary

surveys in the Comprehensive Database have information on the type of packaging (canned or non-canned, in particular).

Two scenarios were therefore developed to consider the higher levels of BPA in canned foods. In scenario 1 only foods specifically codified as canned in the dietary survey are assigned the corresponding occurrence level for BPA. In scenario 2 any food category (at the lowest available level of food category classification) which has been codified as canned in at least one survey is always considered to be consumed as canned in all dietary surveys included in the Comprehensive Database.

In the case of infants a consumption of 150 g/kg bw/day was used as a standard for both human milk and infant formula with the exception of breastfed infants over their first five days of life for whom the consumption was assumed to be 75 g/kg bw/day.

Due to a very low percentage of left censored samples, in particular among canned foods, the techniques used to model data below the LOD or LOQ had a very small impact on the average concentration in the different food categories and, consequently, on the exposure. Therefore, middle bound average BPA concentration values were used in the final exposure assessment.

Dietary exposure for the population older than 6 months

The modelled dietary exposure (MB) obtained by scenario 2, for infants (6 to 12 months), toddlers (12 to 36 months) and other children (3 to 10 years) ranged from 290 to 375 ng/kg bw/day for the average exposure and from 813 to 857 ng/kg bw/day for the high exposure, respectively. Additional dietary exposure from a number of food contact articles was also assessed in specific population groups. The highest estimated high exposure from PC tableware and cookware was observed for infants and toddlers (14 ng/kg bw/day for PC tableware and 46 ng/kg bw/day for cookware). This age class is also the one in which regular use of tableware (made of PC but also other materials) is most likely to occur since `unbreakable` plastic mugs and beakers are often used for toddlers. The highest estimated exposures to BPA migrating from water coolers with PC reservoirs and PC filters into drinking water were also observed in infants and toddlers (22 ng/kg bw/day for water coolers and 3.8 ng/kg bw/day for PC filters). High estimated exposure in residents of buildings with old water pipes repaired with epoxy resins was up to 29 ng/kg bw/day in infants and toddlers.

The modelled dietary exposure (MB) obtained by scenario 2, for teenagers, adults (including women of childbearing age) and elderly/very elderly, ranged from 116 to 159 ng/kg bw/day for the average exposure and from 335 to 388 ng/kg bw/day for the high exposure, respectively. Additional dietary exposure from a number of food contact articles was also assessed in specific population groups within this population. Estimated exposure from PC kettles ranged from 2 to 3.2 ng/kg bw/day with the highest values being observed in adults and the elderly due to their higher consumption of coffee and tea.

The ratio between the modelled exposures derived from one or other of the two scenarios related to the food categories consumed as canned was lowest in countries where many food codes were available for canned products and/or where canned products are largely consumed. This was the case for UK men and women 18 to 45 years where the ratio was 1.9 and 2.2 at the average, respectively and 1.7 and 2.1 at the high exposure level, respectively. The highest difference was noted in Belgian toddlers with a ratio equal to 5.0 and 6.8 for the average and the high exposure level, respectively.

Under scenario 1, canned foods contributed always with less than 50 % to the average exposure for all age classes with the exemption of one survey related to men 18 to 45 years old where it was 50– 75 %. Under scenario 2, canned products dominated in all surveys, with the percentage contribution to BPA from non-canned foods mainly ranging between 10-25 %. Under scenario 1, non-canned “meat and meat products” turned out to be a major contributor to BPA average exposure in the large majority of countries and age classes. “Vegetables and vegetable products” was the only canned food category that contributed up to 25-50 % in some of the population groups under this scenario. “Meat and meat

products" was the major contributor among the non-canned food categories also under scenario 2 but never exceeded 10-25 % of total exposure. On the other hand, the canned versions of "vegetables and vegetable products", "meat and meat products" and "composite food" were the major sources of average BPA exposure under scenario 2.

Overall, among the population older than 6 months, infants and toddlers presented the highest estimated average (375 ng/kg bw/day) and high (857 ng/kg bw/day) dietary exposure. The CEF Panel considered that this was mainly due to their higher consumption of foods and beverages per kg bw.

Compared with the current assessment, dietary exposure to BPA estimated by EFSA in 2006 for the population older than 6 months was far higher (up to 5 300 ng/kg bw/day in toddlers), due to the lack of data at that time which led to the use of very conservative assumptions in relation to both the level of consumption of canned food and the estimated BPA concentration in these foods.

Dietary Exposure for infants aged 0-6 months

For breastfed infants, the estimated average dietary exposure was 225, 135 and 119 ng/kg bw/day for infants in the first five days of life, infants from 6 days up to 3 months and infants 4-6 months, respectively. The estimated high dietary exposure was 495, 390 and 343 ng/kg bw/day, respectively. The CEF Panel noted that, due to the lack of recent European data related to initial human milk, the estimated dietary exposure in the first five days of life was based on BPA concentration in samples collected in Japan in 2000 and generated using ELISA methodology. The Panel noted these limitations in the data and the consequent uncertainties in the estimates for this age group.

Average and high additional exposure to infants that would derive from the consumption of herbal tea prepared with water heated in a PC kettle would be as low as 2 and 4 ng/kg bw/day, respectively.

In the case of formula-fed infants (0-6 months), the estimated average and high exposure were 30 and 80 ng/kg bw/day, respectively. These estimates are based on the most common situation i.e. the use of non-PC baby bottles and the use of water containing low BPA levels to reconstitute the infant formula. Additional dietary exposure may occur in specific population groups due to i) the use of tap water in buildings where old water pipes have been relined with epoxy resins releasing BPA (estimated high exposure: 165 ng/kg bw/day) and ii) the use of old PC bottles bought before the 2011 ban (estimated high exposure: 684 ng/kg bw/day). The percentage of infants to which these cases would apply is unknown. If this percentage was higher than 5 % in some countries, it would lead to a high dietary exposure which is significantly higher than 80 ng/kg bw/day.

Dietary exposure from further sources in other specific population groups of infants was assessed: average exposure in infants fed powdered formula reconstituted with water heated in PC kettles or with water from PC filters were 16.5 ng/kg bw/day and 6 ng/kg bw/day, respectively. The assumptions used to estimate these average exposure values were conservative and would also cover high exposure.

Compared with the current assessment dietary exposure to BPA estimated by EFSA in 2006 in the population 0 to 6 months was far higher (up to 11 000 ng/kg bw/day in infants aged 3 months in one of the scenarios considered), due to the lack of data at that time, which led to very conservative assumptions in relation to BPA concentration in infant formula and to BPA migration from PC bottles.

Non-dietary exposure

Exposure to BPA was estimated from the non-food sources of thermal paper, indoor air (including air-borne dust), dust, dental materials, toys and articles intended to be mouthed and cosmetics. The CEF Panel noted that outdoor air and surface water are also sources of BPA. However, data on BPA concentrations in outdoor air vary widely and depend on regional factors. Reported concentrations of BPA in surface water are very low and, together with contact to surface water, e.g. swimming in lakes and rivers, will constitute only negligible exposure to BPA. Therefore these sources were not included

in the current exposure assessment. Medical devices other than dental materials were also not considered. Since the BPA levels in saliva after dental treatment are reported to be very low (the BPA level before treatment is the same as about 24h after treatment), it could be argued whether this really represents exposure to dental materials. Therefore, exposure to dental materials was not included in the total exposure calculation.

Data on occurrence, migration and transfer of BPA from non-food sources are scarce. The following concentration data were selected from the scientific literature and other risk assessment reports to calculate exposure in the EU: for indoor air 1 ng/m^3; for dust 1 460 µg/kg, and for cosmetics (such as body wash, and body lotions, etc.) 31 µg/kg. A migration of 0.14 µg/toys and 0.32 µg/pacifiers with PC shield into saliva over a 24 h period was assumed. The transfer of BPA from thermal paper to fingers was estimated to be 1.4 µg/finger considering 10 s of contact with paper. Handling events were assumed as 1 per day for teenagers and adults to assess average exposure and as 4.6 per day to assess high exposure. For children the handling events were assumed as 0.5 time per day for average exposure and 2 times per day for high exposure. The thermal paper was assumed to be handled mainly by the finger tips of three fingers each of one (average exposure) or two hands (high exposure).

For the calculation of total exposure the contributions of dust, toys, indoor air, thermal paper and cosmetics were summed up for the respective age groups.

The contribution of the different non-dietary sources to average exposure was similar in infants aged 6 days to 3 years. The sources of BPA were identified and distinguished between infants (6 days to 12 months) and toddlers. The obtained values, given in brackets for infants and toddlers, respectively, show that the main non-food source is cosmetics (e.g. body lotions, etc., 2.9 and 1.7 ng/kg bw/day), followed by dust (2.6 and 1.1 ng/kg bw/day), indoor air (2.4 and 1.4 ng/kg bw/day) and toys (0.3 and 0.02 ng/kg bw/day). When considering the high exposure, the main source was dust (31 and 12.9 ng/kg bw/day), followed by indoor air (5.8 and 3.4 ng/kg bw/day), cosmetics (5.6 and 3.3 ng/kg bw/day), and toys (1.2 and 0.5 ng/kg bw/day). Infants and toddlers using pacifiers with PC shields were considered as a specific group. The exposure estimates from this source were 7.6 and 9.8 ng/kg bw/day for infants with average and high exposure. For toddlers the exposure estimate was 6.6 ng/kg bw/day.

For the rest of the population (children above 3 years, teenagers and adults) handling of thermal paper was considered as a source and changes this pattern. When considering the average exposure, thermal paper became the main non-food source (21, 28 and 18 ng/kg bw/day), followed by cosmetics (1.3, 1.5 and 1.2 ng/kg bw/day), indoor air (0.7, 1.1 and 0.7 ng/kg bw/day) and dust (1.3, 0.2 and 0.1 ng/kg bw/day). When considering the high exposure, thermal paper was still the major source of exposure (165, 259 and 163 ng/kg bw/day), but then exposure to dust (4.6, 4.6 and 2.9 ng/kg bw/day) becomes higher than that of cosmetics (2.5, 2.9 and 2.4 ng/kg bw/day) and was followed by indoor air (1.8, 2.1 and 1.3 ng/kg bw/day) as the lowest contributor. The CEF Panel noted that the average values for dust and thermal paper differed by a factor 10 from the respective high values. This is due to highly conservative assumption for dust ingestion and frequency of and number of fingers handling thermal paper when assessing high exposure.

Total exposure

The modelled average total exposure for the populations older than 6 months ranged from 314 to 383 ng/kg bw/day in infants, toddlers and children aged 3 to 10 years of age and from 136 to 190 ng/kg bw/day in teenagers, adults and elderly/very elderly.

The modelled high total exposure for population older than 6 months ranged from 873 to 981 ng/kg bw/day in infants, toddlers and children aged 3 to 10 years and from 500 to 642 ng/kg bw/day in teenagers, adults and elderly/very elderly.

In formula-fed infants, the modelled average and high total exposure for infants 0-6 months were 38 and 117 ng/kg bw/day, respectively.

In breastfed infants, the modelled average total exposure was 228, 143 and 127 ng/kg bw/day for infants in the first five days of life, infants from day 6 to 3 months and infants 4-6 months, respectively. The modelled high total exposure was 501, 427 and 380 ng/kg bw/day, respectively.

Biomonitoring studies have been used to assess how much total BPA is excreted in urine, allowing for an estimation of exposure from all sources to total BPA. A relatively large amount of information on urinary BPA concentration is available for Europe. All age classes are covered in the different studies available: children (except 1–3 years old toddlers), 14–15 years old teenagers, pregnant women, and 20–74 year old adults.

The distributional characteristics of the total BPA concentrations in urine in terms of shape and spread are generally quite homogeneous across the different studies. Total BPA concentrations (GM) were, with some exceptions, in the range of 1.1–3.6 µg/l. Estimates for the average and high levels of daily BPA exposure were calculated by using the geometric mean (GM), the median (P50) and the 95th percentile (P95) of the urinary BPA. The following average exposure estimates were derived: 20 ng/kg bw/day (for 7–44 days old newborns) and <10 ng/kg bw/day (for 1–2 month old infants), 107 ng/kg bw/day (for the children 3–5 years old) and 58 ng/kg bw/day (for children 5–10 years old), 49 ng/kg bw/day (for teenagers and adults), and 40–73 ng/kg bw/day (for the elderly). The estimates for high BPA exposure were 136 ng/kg bw/day (for infants), 676 ng/kg bw/day (for 3–5 years old children), 311 ng/kg bw/day (for 5–10 years old children), 225 ng/kg bw/day (for the teenagers), 234 ng/kg bw/day (for the adults), and 203 ng/kg bw/day (for the very elderly).

The estimates for the average and high total exposure to BPA in the general population, as obtained by the modelling approach, were compared with the biomonitoring estimates. The modelling approach gave estimates which were approximately 4-fold higher (38–383 ng/kg bw/day vs. <10–107 ng/kg bw/day) than those obtained by the biomonitoring approach for average exposure, and 3-fold higher for high exposure. The different statistical procedures used to derive central tendency and the scenarios for modelling the dietary and non-dietary exposure are important contributions to these discrepancies. These comparative results show however that the existence of unrecognised sources of exposure is unlikely.

Diet was the main source of total exposure in all population groups (from 78-99%). Dietary exposure in women of childbearing age was slightly higher (132 and 388 ng/kg bw/day for average and high exposure, respectively) than that for men of the same age (126 and 355 ng/kg bw/day for average and high exposure, respectively). This may be due to different food items consumed by women as reported in the individual surveys. The uncertainty around the estimates of dietary exposure based on the EFSA comprehensive database was judged as relatively low.

Thermal paper was the second source of total exposure in all population groups above 3 years of age whereas exposure to BPA from thermal paper was considered to be negligible under the age of 3. The contribution to the total average exposure ranged between 7 and 15 %, taking into account all population groups above 3 years of age. The uncertainty around the estimate of exposure to BPA from thermal paper was judged to be considerably higher than that around dietary exposure. The CEF Panel is aware of an ongoing study on BPA pharmacokinetic and dermal exposure in cashiers sponsored by the National Institute of Environmental Health Sciences (NIEHS) under the National Toxicology Program (NTP). The results of this study will be considered by the CEF Panel as they will be an additional source of information regarding the absorption of BPA from thermal paper.

Dust was the second source of exposure for children under the age of 3 years (except infants in the first few days of life). However, dust contributed comparatively little (2.1 %) to the average total exposure with the exception of formula-fed infants 0-6 months for which it was up to 6.9 %.

354 Average exposure to BPA from other sources such as toys and cosmetics was estimated to be less than
355 0.3 ng/kg bw/day and 2.9 ng/kg bw/day, respectively in all population groups.

356 Overall, the CEF Panel concluded that diet is the major source of exposure to BPA in the EU
357 population. Another important source for BPA exposure could be thermal paper in all population
358 groups above 3 years. Due to the relatively large uncertainty around the estimate of exposure for this
359 source, the CEF Panel considered that more data would be needed in relation to BPA absorption
360 through the skin and to patterns of thermal paper handling by the general population in order to
361 provide a refined estimate of exposure from this source.

362 **TABLE OF CONTENTS**

BACKGROUND AS PROVIDED BY EFSA

Bisphenol A (BPA) is used as a monomer in the manufacture of polycarbonates and epoxy resins and as an additive in plastics. Polycarbonates are used in food contact materials such as reusable beverage bottles, infant feeding bottles, tableware (plates and mugs) and storage containers. Epoxy resins are used in protective linings for food and beverage cans and vats.

EFSA issued scientific opinions on BPA in 2006, 2008 and in 2010 (EFSA 2006a, 2008; EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2010).

In its opinion of 2006, EFSA performed a risk characterisation for BPA, including a dietary exposure assessment and a hazard characterisation. In this opinion, EFSA established a tolerable daily intake (TDI) for BPA of 0.05 milligram per kilogram (mg/kg) body weight, based on the no adverse effect level of 5 mg/kg body weight in multi-generation rodent studies and applying an uncertainty factor of 100.

A new opinion on the toxicokinetics of BPA was adopted by EFSA in 2008. Here, EFSA reaffirmed the TDI established in 2006, concluding that age-dependent toxicokinetics differences of BPA in animals and humans would have no implication for the assessment of BPA previously carried out by EFSA.

In 2010, the CEF Panel performed a new hazard characterisation of BPA, based on a comprehensive evaluation of recent toxicity data. The Panel concluded that no new scientific evidence had been published since the EFSA opinions of 2006 and 2008 that would call for a revision of the current TDI. However, it emphasised that there were uncertainties concerning some BPA-related effects of possible toxicological relevance, in particular biochemical changes in brain, immune-modulatory effects and enhanced susceptibility to breast tumours emerging from studies on developing animals. Given several methodological shortcomings in the studies showing these effects, the Panel concluded that the relevance of these findings for human health could not be assessed, but that it would reconsider its opinion should any new relevant data became available. A Panel member expressed a minority opinion based on those uncertainties.

In 2011, EFSA was asked to provide scientific advice in relation to possible divergences between the conclusions of the EFSA Scientific Opinion on BPA of September 2010 and those in the reports on BPA published in September 2011 by the French Agency for Food, Environmental and Occupational Health and Safety (ANSES). On 1 December 2011 EFSA published a Panel statement (EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2011a) on BPA in which the information in the ANSES report was considered not to change the views that the Panel expressed in 2010. However, concerning additional data in recent literature, the Panel stated that it would need further time to review more in depth the new studies. The Panel also underlined that there are ongoing low dose studies at the National Center for Toxicological Research/FDA and at the National Toxicological Program/National Institute of Environmental Health Sciences which aim to address, at least in part, the current uncertainties regarding the potential health effects of BPA.

The ANSES risk assessment of BPA (including exposure assessment from the diet as well as from other routes) was finalised during the preparation of this scientific opinion and was published in April, 2013 (ANSES, 2013).

After its 2011 scientific advice on BPA, EFSA noted that its latest exposure assessment to BPA through dietary sources dates back to 2006, and needed to be updated in the light of the data since then available. The relevance of a dietary exposure assessment versus a more general exposure assessment via various routes of exposure should also be explored. Also, in line with the 2011 conclusions of the CEF Panel, it is advisable for EFSA to undertake a full re-evaluation of the safety of BPA, based on all the most recent experimental evidence.

TERMS OF REFERENCE AS PROVIDED BY EFSA

In accordance with Article 29 (1) of Regulation (EC) No 178/2002, the European Food Safety Authority asks its scientific Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF) to provide by November 2013 a scientific opinion on the risks for public health related to the presence of bisphenol A in foodstuffs.

In particular, the opinion should:

- evaluate the toxicity of BPA for humans, including for specific (vulnerable) groups of the population (e.g. pregnant women, infants and children, etc.) and considering all relevant toxicological information available;

- carry out an exposure assessment on the basis of the occurrence data available in the public domain and other occurrence data that may be available, and quantify as far as possible not only dietary exposure but also exposure from non-dietary sources;

- consider specifically the exposure situation for the supposedly most vulnerable groups of the population (e.g. pregnant women, infants and children, etc.) and take into account, if available, biomonitoring data when assessing the exposure and compare the results with the calculated exposure;

- characterise the human health risks taking into account specific groups of the population.

INTERPRETATION OF THE TERMS OF REFERENCE AS PROVIDED BY EFSA

The Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF) received the request from the European Food Safety Authority, proposing an endorsement of the full draft opinion on Bisphenol A (BPA) for public consultation by July 2013, and to provide by November 2013 a final scientific opinion on the risks for public health related to the presence of bisphenol A in foodstuffs. Taking into account that important toxicological studies on BPA are to be published shortly, and acknowledging that the hazard identification and characterisation of BPA requires further discussions before endorsement, a two-step approach for consultation on the draft opinion on BPA was proposed by the CEF Panel. The current draft document thus addresses the 2nd and 3rd part of the terms of reference only. The full draft opinion on BPA will be released for public consultation by the end of 2013.

ASSESSMENT

1. Introduction

Bisphenol A (BPA) is an industrial chemical that is widely used as a monomer or additive for the manufacture of polycarbonate (PC) plastics and epoxy resins and other polymeric materials, and also certain paper products (e.g. thermal paper). The properties of PC, e.g. rigidity, transparency and resistance, make these plastics particularly suitable for many technical applications. PC is used for food and liquid containers, such as tableware (plates and mugs), microwave ovenware, reservoirs for water dispensers and non-food applications such as toys and pacifiers with PC shields. BPA-based epoxyphenolic resins are used as protective linings for food and beverage cans and as a coating on residential drinking water storage tanks. BPA is also used in a number of non-food-related applications, e.g. epoxy-resin based paints, medical devices, surface coatings, printing inks, thermal paper, and flame retardants.

1.1. EU and national provisions regarding BPA

BPA was authorised in Europe by the Commission Directive 2002/72/EC[4] of 6 August 2002, to be used as monomer and additive for the manufacture of plastic materials and articles intended to come in contact with foodstuffs together with a specific migration limit of 0.6 mg per kilogram food (SML (T) = 0.6 mg/kg). This Directive was amended by the Commission Directive 2011/8/EU of 28 January 2011[5], placing a temporary ban on the use in the manufacture of polycarbonate infant feeding bottles as from 1 March 2011 and the placing on the market of these feeding bottles as from 1 June 2011. The definition of 'infant' in Directive 2006/141EC[6], namely children under the age of 12 months, applies.

Since May 2011 Directive 2002/72/EC is replaced by Regulation (EU) No 10/2011[7], which has 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 its authorisation as an additive in plastic food contact materials and articles.

Bans on the use of BPA for food packaging intended for young children (0-3 years old) have been proposed by several EU Member States.

In May 2010, Denmark banned the use of BPA in infant feeding bottles and all food contact materials of foods particularly intended for children between 0 and 3 years of age and it is now included in the Bekendtgørelse om fødevarekontaktmaterialer 579/2011 [8].

Sweden has decided to ban the use of BPA or compounds containing BPA in varnishes or coatings of packaging for food intended for children between the age of 0 and 3 years (Regulation SFS 2012:991[9]). The ban entered into force 1 July 2013.

France adopted on 24 December 2012 a law suspending the manufacturing, import, export and putting on the market of all food contact materials containing BPA. This law will apply gradually with an application date of 1 January 2013 for food contact materials coming into contact with food intended for children between 0 and 3 years of age and an application date of 1 January 2015 for all food

[4] Commission Directive 2002/72/EC of 6 August 2002 relating to plastic materials and articles intended to come into contact with foodstuffs, OJ L 220, 15.8.2002, p.18-58.
[5] Commission Directive 2011/8/EU of 28 January 2011 amending Directive 2002/72/EC as regards the restriction of use of Bisphenol A in plastic infant feeding bottles, OJ L 26, 29.1.2011, p.11-14.
[6] Commission Directive 2006/141/EC of 22 December 2006 on infant formulae and follow-on formulae and amending Directive 1999/21/EC. OJ L 401, 30.12.2006, p.1-33.
[7] Commission Regulation (EU) No 10/2011 of 14 January 2011 on plastic materials and articles intended to come into contact with food. OJ L 12, 15.1.2011, p.1-89.
[8] Bekendtgørelse om fødevarekontaktmaterialer 579/2011 (§ 8, stk. 2); https://www.retsinformation.dk/Forms/R0710.aspx?id=136917&exp=1
[9] Regulation No 991/2012 of 20 December 2012 amending the Food Regulation No 813/2006, Svensk författningssamling (SFS), 4.1.2013, p.1.

contact materials. In the meantime, once a decree with specifications is adopted, labelling requirements for pregnant women, breastfeeding women and small children will apply[10].

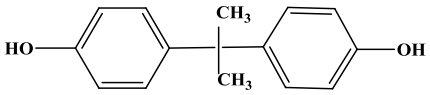
In September 2012, Belgium published an amendment of the national law concerning the protection of consumer health, regarding food commodities and other products, banning the marketing or putting on the market and manufacture of containers for food commodities, containing BPA, particularly intended for children between 0 and 3 years of age[11]. This amendment was based on the opinion of the Belgium Superior Health Council, issued on 3 November 2012. The law entered into force on 1 January 2013.

BPA is listed as entry 1 176 in Annex II (list of substances prohibited in cosmetic products) of Regulation (EC) No 1223/2009 of the European Parliament and of the Council of 30 November 2009 on cosmetic products[12].

2. Physical and chemical characterisation

BPA is an organic chemical synthesised by condensation of 2 mol phenol with 1 mol acetone in the presence of an acid catalyst. It has the chemical formula $C_{15}H_{16}O_2$, with a molecular weight (MW) of 228.29 g/mol. It has the CAS No 80-05-7 and EC-No 201-245-8 (EINECS number).

Chemical structure:



IUPAC Name:
4,4-Dihydroxy-2,2- diphenylpropane
2,2-bis(4-Hydroxyphenyl)propane
4-[2-(4-Hydroxyphenyl)propan-2-yl]phenol

EINECS name:
4,4′ -Isopropylidenediphenol

CAS name:
Phenol, 4,4'-(1-methylethylidene)bis-

Other names:
Bisphenol A
Bis(4-hydroxyphenyl)dimethyl methane
4,4'-Dihydroxydiphenyl propane
Diphenylolpropane

BPA is a white solid available as crystals or flakes (O'Neil 2006; Lewis, 2001). It crystallises as prisms from dilute acetic acid and as needles from water (Lide, 1994) and has a mild phenolic odour under ambient conditions (O'Neil 2006). It has a melting point of 150-158 °C, a boiling point of 360-398 °C (at 101.33 kPa, (IUCLID, 2000; Cousins et al., 2002) and a density of 1.195 kg/dm^3 at 25 °C (IUCLID, 2000; Lewis, 2001). The vapour pressure is 5.3x10^{-6} Pa at 25 °C (Cousins et al., 2002).

BPA is a moderately hydrophobic compound with an octanol–water partition coefficient (log Pow) of 3.32 (Hansch et al., 1995), with a slight polarity due to the two hydroxyl groups. It is soluble in acetic acid (Lide, 1994) and soluble in aqueous alkaline solution, alcohol, acetone (O'Neil, 2006), benzene

[10] Regulation No 1442/2012 of 24 December 2012 aiming at banning the manufacture, import, export and commercialisation of all forms of food packaging containing bisphenol A. OJ of the French Republic (OJFR), 26.12.2012, text 2 of 154.
[11] Loi du 4 septembre 2012 modifiant la loi du 24 janvier 1977 relative à la protection de la santé des consommateurs en ce qui concerne les denrées alimentaires et les autres produits, visant à interdire le bisphénol A dans les contenants de denrées alimentaires publiée au Moniteur Belge le 24 septembre 2012
[12] Regulation (EC) No 1223/2009 of the European Parliament and of the Council of 30 November 2009 on cosmetic products, OJ L342, 22.12.2009, p.59-209.

and diethyl ether (Lide, 2004). It is has a fairly low solubility of 120-300 mg/l in water at 25 °C (Dorn et al., 1987, Cousins et al., 2002).

The pKa value of BPA is between 9.59 and 11.30 (Cousins et al., 2002); thus BPA will be present mainly in its molecular form in liquid media with pH lower than 7. The BPA molecule has a fairly strong fluorophore and it can be detected by its fluorescence. Its chromophore is relatively weak, and the sensitivity of ultraviolet (UV) detection is much lower than that of fluorescence detection.

The Cousins report cited above also summarised environmental information as follows: BPA does not persist in the environment, although it is fairly stable in its solid form. Aerobic biodegradation is the dominant loss process for BPA in river water and soil, with a degradation half-life is about 4.5 days (Cousins et al., 2002). Its loss process in the atmosphere is due to the rapid reaction with hydroxyl radicals, and the photo-oxidation half-life for BPA in air is about 4 h (Cousins et al., 2002).

Chlorinated BPA can be found in both wastewater and drinking water, as BPA can be chlorinated by sodium hypochlorite, a bleaching agent in paper factories and a disinfection agent in sewage treatment plants (Fukazawa et al., 2001; Yamamoto and Yasuhara, 2002), and by chlorine, a chemical used in the disinfection of drinking water (Gallard et al., 2004). The present assessment does not deal with chlorinated BPA.

Food production animals may be exposed to BPA which is then present in their tissues as glucuronated (conjugated) BPA. When total BPA is measured in animal products (e.g. meat, milk, eggs) this may therefore include conjugated BPA, deriving from exposure of the animal, in addition to any unconjugated BPA deriving from contamination and/or migration from food contact materials. Dietary exposure to total BPA is indeed of interest since part of the glucuronated BPA will be deconjugated to release unconjugated BPA (see Chapter 4.3.5).

3. Potential sources of exposure

3.1. Materials and uses

Polycarbonate plastics

Polycarbonates (PC) are a group of thermoplastic polymers produced by the condensation polymerisation reaction of BPA and carbonyl chloride or by melt-transesterification reaction between BPA and diphenylcarbonate. The production of PC is the main use for BPA. PC plastics are amorphous, transparent polymers with high levels of impact strength and ductility, stability, heat resistance and useful engineering properties over a wide temperature range, as well as good resistance to UV light. (CEH, 2008; IHS, 2013). Because of these properties PC plastics and PC blends with, for example, polybutylene terephthalate and acrylonitrile-butadiene-styrene (ABS) polymers are used in numerous applications (BPF, 2013). PC and PC blends may be used in the manufacture of consumer products such as CDs and DVDs, jars/containers, identity cards and toys. PC plastics are also used in the automotive industry, in glazing (e.g. greenhouses), in optical media including lenses for glasses, as well as in food contact materials and articles and in medical devices.

Until 2011 PC plastics were used in the manufacture of infant feeding bottles. However, this application was withdrawn in the European Union (EU) following the introduction of the Commission Directive 2011/8/EU of 28 January 2011, which restricts the use of BPA in these articles[13]. Other PC food contact applications include water coolers with refillable PC reservoirs (PC coolers), tableware, chocolate moulds, kettles and kitchen utensils. PC plastics may also be used for water pipes in public water distribution networks. The migration of residual BPA in the polymer, present due to incomplete polymerisation, or the hydrolysis of the polymer and migration of the BPA released from these PC

[13] Commission Directive 2011/8/EU of 28 January 2011 amending Directive 2002/72/EC as regards the restriction of use of Bisphenol A in plastic infant feeding bottles, OJ L 26, 29.1.2011, p.11-14.

materials into the foods and beverages with which they come into contact, has the potential to provide a source of dietary exposure to BPA.

Some toys may be made with PC plastics (KEMI, 2012). Mouthing of the toys by children may result in exposure to any BPA leaching from these articles into the saliva (KEMI, 2012). For baby pacifiers a large Danish retailer of pacifiers estimated that for 10-20 % on the Danish market in 2010 the shield and ring were made of PC plastics (Lassen et al., 2011). Since the saliva of a baby is spread around the mouth during sucking and may then be ingested, the shield may represent a source of oral exposure to BPA.

About 3 % of total polycarbonate production is reported to be used for the manufacture of medical devices (Beronius and Hanberg, 2011). Some BPA-containing medical devices may have direct and/or indirect contact with the patients (e.g. autotransfusion apparatus, filters, bypasses, tubing, pumps, instruments, surgical equipment, blood pathway circuits and respiratory tubing circuits, dialysis equipment). It has also been reported that breast milk pumps are made from PC plastics (Beronius and Hanberg, 2011). The transfer of BPA from these PC plastics into the biological human matrices with which they come into contact or the migration of BPA into human milk to be consumed by an infant can result in exposure to BPA.

Epoxy resins

Epoxy resins are thermosetting polymers that have good mechanical properties, as well as high temperature and chemical resistance. As such, these resins have a wide range of applications, including use as coatings applied to metal substrates in food contact materials, in dental fillings, in electronics/electrical components, in high tension electrical insulators, in fibre-reinforced plastic materials, in structural adhesives and in the relining of aged water pipes.

Epoxy resins may be produced by the reaction of BPA with epichlorohydrin forming BPA diglycidyl ethers (commonly abbreviated to BADGE), which is the primary chemical building block for the broad spectrum of materials referred to generally as epoxy resins. Alkoxylated BPA may also be used to prepare epoxy resins.

Epoxy resins represent the second largest use for BPA. Epoxy resins may be cross-linked with phenolic resins, amino resins, acrylic resins or anhydride resins producing epoxy phenolic, epoxy amino, epoxy acrylic and epoxy anhydride can coatings. Following a request from EFSA, industry noted that "the content of the statement on epoxy phenolic resins in the EFSA opinion of 2006 is still correct, but that BPA based phenolics stopped being used in Europe a few years ago." (email from PlasticsEurope to EFSA on 5 February 2013). As well as canned food and beverages, epoxy based coatings have been reported to be used in other food contact applications including re-usable drinks bottles and wine vats. They may also be used in construction products such as drinking water pipes and storage tanks.

Epoxy resins may also be used as stabilisers (hydrochloric acid scavengers) and as plasticisers in PVC organosol coatings that may be used as base coatings for metal lids applied to glass jars. Any residual BPA in the cured coating has the potential to migrate into the food or beverage with which it comes into contact, thereby providing a potential source of dietary exposure.As for plastic food contact materials and articles, the extent of the migration from the coating, and hence the potential exposure, is dependent on contact surface, time and temperature. With the high temperature processing conditions and the long shelf-life of canned foods, as long as the BPA is soluble in the foodstuff, the migration of any residual BPA will occur, resulting in dietary exposure.

Epoxy resins may also be reacted with ethylenically unsaturated monocarboxylic acids to form vinyl esters, and it has been stated that these too may be used in food contact applications (email from PlasticsEurope to EFSA on 5 February 2013).

Epoxy resins may further be used in non-food contact applications including flooring and non-food tanks and pipes. The cross-linking of epoxy resins with phenol gives rise to a higher molecular weight solid epoxy resin known as a phenoplast (WUR, 2001). These resins are used as materials in the construction sector and as such are considered to constitute a source of exposure through indoor air and dust (see Chapter 4.3.6).

Thermal paper

Thermal paper consists of a smooth paper to which a coating is applied. This coating is made from a leuco dye and a phenol developer such as BPA. The leuco dye exists in two forms, one of which is colourless. On printing, a thermal head causes the coating components to melt and react with each other, causing the dye to become dark (Biedermann et al., 2010; Mendum et al., 2011). Exposure from this source can occur via dermal contact, in particular for cashiers handling receipts as BPA can be transferred from the paper surface to the skin (Biedermann et al., 2010), but also for consumers. Thermal papers are used in different areas, such as bus tickets, airline tickets, cash receipts and papers for laboratory use (Liao and Kannan, 2011a, b). According to the European Thermal Paper Association BPA is still used in thermal paper and in 2012, 80 % of thermal paper is used for POS (Point of Sales) grades which are mainly used for supermarkets and shop tickets and not for tickets for transport (bus/boarding passes) and tickets for lotteries (email from European Thermal Paper Association to EFSA from 17 June 2013).

Recycled paper

Recycled paper and board may contain BPA if paper products that contain BPA (e.g. thermal papers) are included in the recycling feedstock and if the BPA is not completely removed during the recycling decontamination process. Thermal paper was estimated to be a major source for the contamination of recycled paper with BPA (Gehring et al., 2004). BPA is listed as an evaluated monomer permitted for use in printing inks in the Swiss Ordinance of the FDHA on articles and materials (RS 817.023.21[14]). The use of BPA as an ingredient in inks is no longer widespread, but its presence as an impurity in ink formulations cannot be excluded (email from PlasticsEurope to EFSA on 5 February 2013). Food contact papers and cartons include fast-food and snack wrappers and boxes, paper cups, paper plates and food cartons, such as pizza boxes. These may include a recycled component within the food packaging material and so may provide a source of exposure to BPA. BPA was detected in 45 % of the take-away food cartons tested with higher levels in cardboard than in paper (Lopez-Espinosa et al., 2007). In this study all but one of the 40 samples tested contained recycled fibres. Any migration from the recycled paper or board into food will result in dietary exposure to BPA. BPA was also detected in toilet paper (Gehring et al., 2004) and in kitchen towels (Ozaki et al., 2004) made from recycled paper.

Polyvinyl chloride

PVC is the third-most widely produced plastic, after polyethylene and polypropylene. PVC is produced by polymerisation of the monomer vinyl chloride. BPA has been used historically as (i) a production aid to stabilise vinyl chloride monomer; (ii) in the polymerisation of PVC plastics; (iii) as an antioxidant in plasticisers used in PVC. According to the European Council of Vinyl Manufacturers, the use of BPA for polymerisation and as a stabiliser for storage of vinyl chloride monomer was discontinued in Europe from December 2001 (email from PlasticsEurope to EFSA on 5 February 2013). Additionally, the use of BPA as an additive for food contact plastics, including PVC, is not permitted in the EU according to Regulation (EU) No 10/2011.

However, BPA may still be used in the production of PVC e.g. for toys and therefore, exposure may occur by the transfer of BPA through the saliva. Also, the use of BPA as a production aid in PVC

[14] Ordinance No 817.023.21 of 25 November 2005 on materials and articles. Swiss Federal Department of Home Affairs (FDHA), 1.4.2013, p.1-96

cannot be excluded, since such use as a polymer production aid is outside the scope of Regulation (EU) No 10/2011.

BPA methacrylate containing resins

BPA containing resins may be used in dental sealants. BPA is not used directly in dental materials, but BPA glycidyl methacrylate (bis-GMA) and other acrylate-based derivatives (BPA dimethacrylate) of BPA are used. Any BPA that is present as an impurity in the used methacrylate derivative or is released from the dental sealant by degradation of the polymer has the potential to contribute to oral exposure to BPA (Van Landuyt et al., 2011).

Polyetherimides

Polyetherimides (PEIs) are synthesised by the melt condensation of BPA dianhydride with a diamine, usually m-phenylenediamine. PEIs find use in food contact applications, e.g. microwave cookware in blends with PC (FAO/WHO, 2011) as a consequence of their high heat stability, and migration of any residual BPA may occur. PEIs may also be used in medical applications, in electronic components and in aircraft interiors.

Polysulfone resins

Polysulfone resins are made by condensation of the disodium salt of BPA with 4,4-dichlorodiphenyl sulfone. They exhibit thermal stability, toughness, transparency and resistance to degradation by moisture (FAO/WHO, 2011). They are used in electrical components, appliances, transportation, medical equipment (Geens et al., 2011), pumps, valves and pipes.

Polyarylates

Polyarylates are amorphous polymers that may be formed by co-polymerisation of BPA with aromatic dicarboxylic acids (mainly terephthalic and isophthalic acids). Polyarylates have excellent thermal resistance, toughness in combination with clarity and ultraviolet stability, and compete with traditionally less expensive engineering plastics for applications in the automotive, electronics, aircraft and packaging industries. If used in food packaging applications, the migration of BPA from these into food or beverage provides a potential source of exposure. However according to the FAO/WHO report, high cost, poor chemical resistance and a tendency to yellow have prevented polyarylates from gaining wider acceptance and so exposure from these materials is not considered likely (FAO/WHO, 2011).

Flame retardants

BPA may be used in the production of two flame retardants, tetrabromobisphenol A (TBBPA) and BPA bis(diphenyl phosphate) (CEH, 2010). TBBPA is used to impart flame resistance to epoxy resins used in printed circuit boards, to PC, to ABS resins and, to a lesser extent, to unsaturated polyester resins and other engineering thermoplastics. TBBPA is also used as an intermediate in the production of other flame retardants, such as brominated epoxy oligomers and brominated carbonate oligomers. BPA bis(diphenyl phosphate) is used as a flame retardant in polyphenylene oxide and PC/ABS blends. The latter are not used in food contact applications and so any exposure to BPA from this source will occur through dermal contact, indoor air or dust (see Chapter 3.2).

Other uses

The presence of BPA has also been reported in table cloths and mittens (VKM, 2008). However, the material type (other than plastic) was not specified in the report. BPA was also detected in low amounts in cosmetics on the European market (Cacho et al., 2013). BPA is not permitted for use in

cosmetics in the EU[15], however migration of BPA from packaging materials into the cosmetics or as an impurity in the cosmetic ingredients may constitute a source of exposure through dermal contact (see Chapter 4.3.6).

Also other uses have been reported, such as the use of BPA in polyester resins such as bisphenol fumarates formed by reacting BPA with propylene oxide to form a glycol, which is then reacted with fumaric acid to produce a resin mainly used for its exceptional corrosion resistance to caustic environment (e.g. AOC, 2013). Typical applications of bisphenol fumarate resins are fiber-reinforced tanks and piping. BPA may also be used as an additive in polyamide materials used mainly in electrotechnical applications (ECB, 2010).

The use of BPA as a monomer in plastic food contact materials other than PC cannot be excluded. BPA is subjected to a specific migration limit of 0.6 mg/kg food (Regulation (EU) No 10/2011). BPA was detected in PA baby bottles collected from the EU market in 2010 (Simoneau et al., 2012). This use is not expected in baby bottles made of PA, other plastic materials or silicone. The high migration levels suggest rather the illegal use as additive in PA or a contamination with material not intended for food contact.

3.2. Environmental sources

The general population can be exposed to BPA via food or via the use of non-food consumer products such as thermal paper, toys, etc (see Chapter 3.1). The general population can also be exposed to BPA from environmental sources such as surface water (during swimming) and outdoor air (inhalation of aerosols). In addition, the release of BPA from epoxy-based floorings, adhesives, paints, electronic equipment, and printed circuit boards is reported to be a source of contamination of indoor air (including air-borne dust) and dust (Loganathan and Kannan, 2011). Environmental sources therefore can potentially contribute to oral, inhalation and dermal exposure to BPA (see Chapter 4.3.6).

4. Exposure assessment

4.1. Scope of the exposure assessment

The scope of this opinion is to assess average and high chronic exposure to BPA through different sources and routes of exposure in the EU population. For this purpose the exposure concentrations through the different routes (oral, dermal, inhalation) are added up. Specific scenarios were developed to cover the exposure patterns in the different age classes and vulnerable groups (infants and young children, pregnant and breast-feeding women). Scenarios to assess acute exposure to BPA (with the exception of dental materials) or BPA exposure in specific disease states or occupational exposure of workers handling BPA containing products were not developed in this opinion.

4.2. Sampling and methods of analysis

When considering the inclusion of occurrence and migration data in the assessment of the exposure to BPA it is essential that the methodology used to derive the data is of an appropriate quality. Only those data that met the criteria described in Appendix I were included in the exposure assessment.

4.3. Occurrence data

4.3.1. General introduction

Screening of scientific publications

One aspect of the terms of reference related to the exposure assessment was to especially take into account occurrence data available in the public domain. To address this point EFSA performed a systematic review of scientific literature on occurrence and exposure data for BPA covering the period

[15] Regulation (EC) No 1223/2009 of the European Parliament and of the Council of 30 November 2009 on cosmetic products, OJ L342, 22.12.2009, p.59-209.

2006 until December 2012. The review was continued after December 2012 and publications were monitored but not considered in this opinion except for special cases.

As a general rule, for BPA occurrence in food, only data published from 2006 onwards were retrieved. The reason for this is that data published before 2006 have already been reviewed in 2006 when EFSA assessed the dietary exposure to BPA within its safety evaluation. The pattern of use of BPA in food packaging may have changed in the meanwhile and there is a need to provide an up-to-date description of the occurrence of BPA in food in order to estimate current dietary exposure. Moreover in the last years a lot of effort has been made to increase the performance of analytical determinations of BPA in terms of increased sensitivity and reduction of BPA contamination; more recent data should therefore be of better quality than older data.

The following bibliographic databases were searched for the term "Bisphenol A" and/or "BPA": ISI Web of Knowledge - Web of Science (WoS), CAB Abstracts, American Chemical Society (ACS), EBSCOhost, Elsevier Science Direct, InformaWorld, SpringerLink. Combination with other search terms, e.g. "food" or "food contact material" were not performed in order not to miss important publications. The search was done independently by two experts who compared the results and discussed possible discrepancies. All publications were screened for relevance. Emphasis was put on migration studies on BPA, occurrence and intake levels of BPA from various dietary sources for the general population and for specific subgroups of the population (e.g. infants, young children, etc), occurrence and human exposure to BPA from non-dietary sources via inhalation or dermal contact and human internal exposure to BPA (biomonitoring) and physiologically based pharmacokinetic (PBPK) modelling studies. Different sources of information were considered: journals and books recorded in electronic bibliographic databases, full text journals, journal tables of content and grey literature, e.g. conference proceedings, annual reports and poster abstracts. The former and reference lists of previous risk assessments e.g. by FAO/WHO 2011, ANSES 2013 and review articles were screened as cross-checking quality assurance measures to ensure that no publications were missed in the bibliographic database searches.

EFSA call for data

In July 2012, Member States, research institutions, academia, food business operators (e.g. food packaging manufacturers and food industries) and other stakeholders were invited by EFSA to submit single analytical data on 1) occurrence of BPA in food and beverages intended for human consumption, 2) BPA migration from food contact materials and 3) BPA occurrence in food contact materials.

In total 3 609 results were submitted to EFSA, 2 076 results for BPA occurrence in food, 988 results for BPA migration from food contact materials and 545 results for BPA occurrence in food contact materials. These data were obtained on samples collected in the European Economic Area (EEA) countries and Switzerland, the vast majority of the samples were collected from 2006 to 2012.

Data were sent by Governmental institutions (3 115 results), Academia (417 results), food manufacturers and 2 associations (Fédération romande des consommateurs (FRC) and PlasticsEurope) (77 results).

4.3.2. Summary from EFSA's call for data

Food and beverages intended for human consumption

EEA countries and Switzerland submitted BPA occurrence data from different kinds of food, 2 076 results were reported from 2004 to 2012.

Data on BPA occurrence in food and beverages intended for human consumption were provided by 8 countries, most of the information coming from France (75.5 %), Germany (10.1 %), Ireland (6.6 %), United Kingdom (2.6 %), Norway (1.8 %), Switzerland (1.3 %), Finland (1.2 %), Spain (0.8 %).

The large majority of the 2 076 submitted results on food (95 %) originated from accredited laboratories and 5 % of results submitted from non accredited laboratories.

Migration data from food contact materials

EEA countries and Switzerland submitted BPA migration data from different kind of materials, 988 results were reported from 2004 to 2012, the large majority (93 %) originated from accredited laboratories.

The packaging samples analysed classified according to the EFSA's standard sample description system were: polycarbonate 82.8 %, polypropylene 3.9 %, aluminium foil/aluminium sheet 2.4 %, packed (no additional information provided) 2.2 %, metal 2.1 %, plastic/plastic film 1.4 %, combined aluminium and film packaging 1 %, tinplate and varnished/partly varnished 1 %, polyamide 0.8 %, combined material 0.4 %, PET polyethylene terephthalate (1 sample). No information was sent for 1.8 % of the samples including the variables "No information" and "Not packed (loose; open)".

Occurrence data in food contact materials

Germany submitted BPA occurrence data for different kinds of food contact materials (plastic, paper and board, others, aluminium, glass). 545 results were reported from 2001 to 2012, the large majority (98 %) originated from accredited laboratories. The packaging samples, classified according to EFSA's standard sample description system and taking into account the information provided in the data element "Packaging" and "Product comment", were: paper and board (39.1 %), plastic (38.2 %), plastic/plastic film and combined paper and film packaging (2.8 %), tinplate aluminium (2.2 %), glass (0.2 %), no information and not packed (loose; open) (17.5 %). In the standard sample description system it was not always possible to give detailed information, so for glass most likely the twist-off lid of a glass jar was analysed and in case of tinplate aluminium the coating was most likely analysed.

More details on the quality of data received are given in Appendix II.

4.3.3. Handling of data

Left-censored data, i.e. samples with concentrations below the limit of detection (LOD) or quantification (LOQ) were handled as recommended in the 'Principles and Methods for the Risk Assessment of Chemicals in Food' (WHO, 2009) and in the EFSA scientific report 'Management of left-censored data in dietary exposure assessment of chemical substances' (EFSA, 2010) through the substitution method. The lower bound (LB) was obtained by assigning a value of zero to all the samples reported as less than the left-censoring limit, the middle bound (MB) by assigning half of the left-censoring limit and the upper bound (UB) by assigning the left-censored limit as the sample result.

4.3.4. Data on occurrence in and migration from food contact materials into food simulants

Values for BPA occurrence in different food contact materials and for BPA migration into food simulants reported in the scientific literature and obtained through EFSA's call for data were screened. Only studies focusing on samples collected in Europe were considered. The quality of data was assessed according to criteria defined in Appendix I. The outcome of the assessment of the scientific literature is reported in Table 65 and Table 66 in Appendix IX.

Occurrence data in food contact materials

The majority of the studies involved the determination of the residual level of the BPA monomer in PC plastics and in particular in baby bottles (Ehlert et al., 2008; Mercea, 2009; Alin and Hakkarainen, 2012;). Values of residual BPA in PC containers, water coolers with PC reservoirs, bottles, baby bottles, trays, etc. reported in the literature ranged from 400 to 70 000 µg/kg. Values specific for PC baby bottles averaged 9 422 µg/kg with a maximum of 35 300 µg/kg. Average values for other PC bottles and water coolers with PC reservoirs were 10 224 and 18 763 µg/kg, respectively.

BPA content in cookware coatings was detected in 7 out of 26 samples with values ranging from 0.5 to 18 µg/dm^2, with an average value of 3.2 µg/dm^2 (or 10 224 µg/kg for an average coating weight of 313 mg/dm^2) (Bradley et al., 2007).

BPA content in a small number of recycled paper and board food contact samples were reported (Bradley et al., 2008a; Pérez-Palacios et al., 2012). The following average values were found: paper cloth – 25 400 µg/kg, paperboard box – 7 390 µg/kg, paper bag – 500 µg/kg and kitchen paper – 330 µg/kg (Pérez-Palacios et al., 2012). Lopez-Espinosa et al. (2007) investigated the BPA content in 40 paper and paperboard containers used for take-away food. BPA was detected in 47 % of the samples and concentrations ranged from 0.05 to 1 817 µg/kg in paperboard products and from 0.08 to 188 µg/kg in paper products. All but one of the 40 samples tested contained recycled fibres

Residual BPA was detected in metal closure coatings (epoxy phenolic basecoat plus organosol topcoat) in the range of 2-16 µg/dm^2 (Oldring et al., 2013). The authors report a ratio of surface area to food weight for metal closures ranging from 0.2 to 2.2 dm^2/kg. If a total migration of residual BPA is assumed, an average migration value of 12.5 µg/kg would be obtained. These estimates were not used in the present exposure assessment because this is a unlikely worst-case scenario.

Migration data from food contact materials

BPA can migrate from PC into foods by diffusion of residual BPA present in the polymer after the manufacturing process, and after hydrolysis of ester bonds of the polymer, a reaction that is catalysed by hydroxide when the polymer is in contact with aqueous food and simulants (Mountfort et al., 1997; Hoekstra and Simoneau, 2013). Some studies indicate that diffusion-controlled migration of the residual monomer has a minor contribution to the release of BPA from polycarbonate articles, and that hydrolysis of the polycarbonate polymer chains at the interface with the aqueous media is the main process that results in higher levels of migration (Biedermann-Brem et al., 2008; Biedermann-Brem and Grob, 2009; Mercea, 2009). In fact, BPA migration from PC plastics into aqueous media was found to be essentially independent of the residual concentration (Mercea, 2009), indicating that transfer mechanisms other than diffusion take place. The migration experiments used conditions as foreseen in the applicable European legislation (Council Directive 82/711/EEC) by that time[16].

Many of the published studies have investigated the effect of a number of factors on BPA migration from PC plastics. These include the effect of temperature and normal repeated use (De Coensel et al., 2009; Kubwabo et al., 2009; Mercea, 2009), the effect of water pH, which can be related to the nature of the water used and also to alkali washing detergents (Biedermann-Brem et al., 2008; Maragou et al., 2008; Biedermann-Brem and Grob, 2009; Kubwabo et al., 2009; Maia et al., 2009; Mercea, 2009), and the effect of PC aging (Le et al., 2008; Kubwabo et al., 2009; Mercea, 2009;). Hoekstra and Simoneau (2013) have reviewed the studies on the release of BPA from PC.

Temperature has a major impact on the BPA migration level into water. An increase from 40 ºC to 60 ºC can lead to a 6–10-fold increase in the migration level (De Coensel et al., 2009; Kubwabo et al., 2009; Mercea, 2009). Although temperature has a major effect on migration, no significant difference was noted between water bath and microwave heating (Ehlert et al., 2008; De Coensel et al., 2009).

The majority of the reported BPA migration studies involve PC plastics, particularly baby bottles. Results from Simoneau et al. (2011) showed BPA < LOD (0.1 µg/kg) in 32 out of 40 PC baby bottles analysed in the European market, when tested with 50 % ethanol for 2 h at 70 ºC after boiling for 5 min. The highest migration value was 1.83 µg/kg and most of the bottles did not release detectable levels of BPA in the 2nd or 3rd migration test carried out.

[16] Council Directive 82/711/EEC of 18 October laying down the basic rules necessary for testing migration of the constituents of plastic materials and articles intended to come into contact with foodstuffs OJ L 297, 23.10.1982, p. 26–30

Samples of PC baby bottles (72) from 12 different brands collected in the Spanish market were tested for BPA migration into 50 % ethanol and 3 % acetic acid, for 2 h at 70 ºC followed by 24 h at 40 ºC. Results were below the LOD (5 µg/kg) in the 3rd migration test in most cases. The highest value found in the 3rd migration test was 18 µg/kg into 3 % acetic acid, migrating from one of the bottles tested (Santillana et al., 2011).

Since hydrolysis of the PC is catalysed by hydroxide, raising the pH of water leads to an increased migration (Mercea, 2009). Studies show evidence of increased BPA migration into water due to the effect of residual alkaline detergent remaining on the surface of the baby bottle after dishwashing (Biedermann-Brem et al., 2008; Maragou et al., 2008; Biedermann-Brem and Grob, 2009; Maia et al., 2009). Results highlighted the importance of good practices of rinsing and drying PC baby bottles after washing in order to reduce the migration of BPA. Degassing (including loss of carbon dioxide) of tap water during boiling can also cause a water pH increase and consequently can lead to higher migration values as compared to fresh water (Biedermann-Brem and Grob, 2009).

At constant temperature conditions, migration was found to increase over time following a quadratic equation law (Cao and Corriveau, 2008a). However, repeated use simulated by sequential migration experiments has shown that migration levels had a tendency to decrease with use (when contact conditions do not promote hydrolysis of PC).

Kubwabo et al. (2009) carried out a study on the migration from PC and other plastic baby bottles, PC reusable drinking bottles and baby bottle liners. 24 baby bottles (PES, PP, PC), 10 baby bottle liners (LDPE, HDPE, vinyl acetate, "BPA-free"), 5 new re-usable PC bottles and five old bottles (6 months to 10 years) were tested for BPA migration into water. A range of migration test conditions were investigated. After 10 days at 40 ºC migration of BPA from PC baby bottles reached a concentration of 1.88 µg/kg into water and 2.39 µg/kg into 50 % ethanol.

Significant differences between BPA migration from new and used PC drinking bottles of 0.01 and 0.2 µg/kg, respectively, were found (Kubwabo et al., 2009). However, different results were reported by Le et al. (2008) that indicated that at room temperature the migration of BPA is independent of whether or not the PC bottle has been previously used. After 7 days of contact at room temperature, the migration values from new (1.0 µg/kg) and used (1 to 9 years) PC bottles (0.7 µg/kg) were not significantly different.

Migration of BPA from 31 PC baby bottles into aqueous food simulants was studied under real repetitive use (effect of cleaning in a dishwasher or with a brush, sterilisation with boiling water and the temperature). Brushing did not seem to have an impact whereas temperature was found to be the crucial factor, in line with the findings of other studies. All samples released BPA in the concentration range of 2.4–14.3 µg/kg when filled with boiled water and left at ambient temperature for 45 min. Normal repeated use was simulated over 12 cycles, and migration values showed a decrease of BPA release in the sterilisation water and in the food simulant (Maragou et al., 2008).

Migration of BPA from PC baby bottles into water after microwave heating to 100 ºC ranged from 0.1 to 0.7 µg/kg. No correlation was found between the residual content of BPA in the bottles and the migration of BPA into water or between the amounts of BPA in consecutive migration extracts (Ehlert et al., 2008).

Migration of BPA from epoxy coated food cans was higher into 3 % acetic acid than into water and higher results were obtained for higher temperature contact conditions as expected (Viñas et al., 2010).

Migration values from cooking ware coatings were found to be lower than 6 µg/kg after the 3rd reuse with olive oil at 175 ºC for 30 min and with a tendency to decline in sequential contact periods (Bradley et al., 2007).

The migration of BPA into food simulants from 11 common food packaging materials was assessed by Fasano et al. (2012). The packages comprised cans intended for tuna (both natural and packed in oil) and caps for marmalade jars, all coated with epoxy resins and well as several plastic packages/materials such as HDPE yogurt packaging, PS dish, teat, bread bag, LDPE film, PC baby bottle, aseptic plastic laminated paperboard carton and 2 synthetic plastic wine tops.

Used PC moulds (5 years old) for chocolate pralines were tested for migration into the simulant olive oil. Results after the 3rd test at 70 ºC and 2 h were < 0.2 mg/kg (email from PlasticsEurope to EFSA on 13 June 2013).

The results for BPA migration from food packaging materials retrieved from the literature are summarised in Table 1.

Table 1: BPA migration into food simulants

	Average migration, (µg/l)			Max	Non detects/ N	Reference
FCM	**LB**	**MB**	**UB**			
Can epoxy	1.26	1.26	1.27	16.00	8/23	Fasano et al.,2012; Cooper et al., 2011; Viñas et al., 2010
Can polyester	0.00	0.03	0.05	0.05	4/4	Cooper et al., 2011
Cookware coating	0.60	0.68	0.76	5.80	21/26	Bradley et al., 2007
Copolyester bottle	0.00	0.04	0.09	0.09	10/10	Cooper et al., 2011; Simoneau et al., 2012
HDPE cup	0.00	0.02	0.03	0.03	3/3	Fasano et al., 2012
LDPE film	0.09	0.10	0.11	0.19	3/6	Fasano et al., 2012
PA baby bottle(2)	25	25	25	329	8/28	Simoneau et al., 2012
PC baby bottle	0.30	0.89	1.48	5.00	74/100	[1]Fasano et al., 2012; [1, 2]Simoneau et al., 2011; [1]Santillana et al., 2011; Kubwabo et al., 2009; Ehlert et al., 2008; [1]Cao and Corriveau, 2008a; Cao et al., 2008; Biedermann-Brem et al., 2008
PC bottle	0.92	0.92	0.92	7.67	4/44	[1]Cooper et al., 2011; [1]Kubwabo et al., 2009; [1]Cao and Corriveau, 2008a; Cao et al., 2008; [1]Le et al., 2008
PC container	2.64	2.64	2.64	2.64	0/10	[1]Guart et al., 2011
PC tableware	0.95	0.95	0.95	1.27	0/4	[1]Oca et al., 2013
PE/board	0.00	0.02	0.03	0.03	3/3	Fasano et al., 2012
PS cup	0.00	0.02	0.03	0.03	3/3	Fasano et al., 2012
Silicone teat	0.00	0.02	0.03	0.03	3/3	Fasano et al., 2012
PP baby bottle	0.00	0.05	0.10	0.10	149/149	Simoneau et al., 2012
PES baby bottle	0.00	0.05	0.10	0.10	30/30	Simoneau et al., 2012
Silicone baby bottle	0.00	0.05	0.10	0.10	5/5	Simoneau et al., 2012

MB: average (middle bound) BPA concentration (assigning the value for LOD/2 or LOQ/2 when LOD or LOQ is reported)
UB: average (upper bound) BPA concentration (assigning the value for LOD or LOQ when LOD or LOQ is reported);
LB: average (lower bound) BPA concentration (assigning the value 0 when LOD or LOQ is reported); Max: maximum value reported (assigning LOD or LOQ when LOD or LOQ is reported); N: total number of samples
[1] Studies used to retrieve data to estimate exposure in Chapter 4.6.2
[2] Migration values in PA bottles refer to a contamination during production

The values for migration of BPA from food packaging materials into food simulants retrieved from the literature and from the call for data were not used in the exposure assessment of the general population. Instead, occurrence values in foods, presented in the following Chapter were used for the general population. However, selected data on migration into simulants from published studies were used to assess the exposure of specific groups of consumers: those consuming water from water coolers with PC reservoirs and users of PC tableware, PC water kettles, PC filters and cookware. Those studies from which data were retrieved are marked in Table 1.

Consumers tend to be loyal to the type of water they consume, and will either consume bottled water or tap water (either as such or filtered). Water from water coolers with PC reservoirs would mainly be consumed away from home (usually at working places) and also in this case consumers might be loyal consumers.

To determine a BPA concentration value for the estimation of exposure from water coolers with PC reservoirs, data were retrieved from published literature and were combined with data provided to EFSA by PlasticsEurope (email from PlasticsEurope to EFSA on 29 November 2012).

The data from the literature were from migration experiments conducted at moderate temperature (typically 20 – 40 ºC) from all PC products into water for all migration times. Concentration data in 10 samples of water stored in water coolers with PC reservoirs were available from the literature in Spain (Guart et al., 2011). BPA concentrations ranged from 1.6 µg/kg to 4.44 µg/kg. Average BPA concentration was 2.64 µg/kg.

Data from PlasticsEurope (email from PlasticsEurope on 29 November 2012) on migration of BPA from 41 samples of water coolers with PC reservoirs (both new and used), collected for different periods of use at temperatures from 5 to 36 ºC were also provided through the EFSA call of data. BPA concentrations ranged from 0.001 µg/kg to 4.05 µg/kg. Average BPA concentration was 0.50 µg/kg.

When all data for water coolers with PC reservoirs were pooled (from literature and the call), the average BPA concentration of 0.81 µg/l was derived (see Table 2) and this value was used to estimate the exposure of this specific group of consumers.

The concentration values in water stored in water coolers with PC reservoirs in China (Chen et al. 2011) and in most samples in Canada (Cao et al., 2008) were in the same range as in the European samples. However, the water in two PC carboys in Canada had BPA concentrations of 6.5 µg/kg and 8.8 µg/kg. The authors suggest that the carboys had been exposed to high temperature for extended periods of time during storage or transport.

Several earlier opinions have not considered a specific BPA value for water stored in water coolers with PC reservoirs (EFSA, 2006a; FAO/WHO, 2011). In the ANSES report (2013) water from water coolers with PC reservoirs were found to have an average concentration of 1 µg/l and a 95th percentile of 4 µg/l.

Migration data into water from PC products, tested at temperatures in the range of 70 to 100 ºC for 24 h and data obtained from the scientific literature, were considered to derive a migration value associated with the use of PC kettles. A PC kettle is typically used to warm/boil water to prepare hot beverages such as tea and coffee, foods such as soups, and other dehydrated products such as infant formula. The average migration value for the 24h contact time derived from the literature (2.55 µg/l) was divided by 24 to reflect the migration occurring during a cycle of 1 h of contact during which the water is boiled, allowed to cool and fresh water may be added to the water remaining in the kettle and a new boiling cycle started. This is considered the typical behavior of a user of such kettles. An average value of 0.11 µg/l was derived (see Table 2).

For PC tableware, migration data from all PC products, into water, 3 % acetic acid and 50 % ethanol, obtained under testing conditions of 2 h at 70ºC from the literature, was considered. These data were

combined with data from the EFSA call for data obtained under the same testing conditions. The average values ranged from 0.18 µg/l (LB) to 1.31 µg/l (UB). The average values from the 2h contact time were divided by 8 to reflect a single use of ca 15 min use (5 min of heating in a microwave + 10 min of additional contact during consumption). Average migration values of 0.02 µg/l (LB), 0.09 µg/l (MB) and 0.16 µg/l (UB) were derived from experiments using any simulants (see Table 2).

PC filters are most likely used in shorter periods of contact time as compared to water coolers with PC reservoirs. The migration was estimated considering the same data as for water coolers with PC reservoirs but only for periods of time up to 24 h. It is reasonable to assume that this condition of contact (1 h at room temperature) also covers the potential migration for longer periods of contact at the refrigerator temperature. An average value of 0.96 µg/l was derived from the data and divided by 24 to simulate a maximum 1 h of contact time for this application, assuming a constant BPA transfer rate. An average value of 0.04 µg/l was used to estimate exposure.

For cooking ware coatings an average value of 0.29 µg/kg (MB) was derived to be used in estimating exposure, taking into consideration the decrease in migration observed after the 3rd reuse with olive oil at 175 ºC for 30 min (Bradley et al., 2007) and extrapolating it over a set of 100 uses.

A survey on potential migrants, including BPA, from non-PC baby bottles was performed by Simoneau et al. (2012). BPA was not detected in baby bottles made of PP, PES or silicone but was detected in some samples of two models of polyamide baby bottles of one single brand found in Switzerland and The Netherlands. Levels ranged from 1 to 329 µg/kg, with average value of all data (including non-detects) of 25 µg/kg in the 3rd migration test. In the 1st migration test a high migration value of 1 005 µg/kg was found for one bottle. A follow up investigation indicated an incidental illegal presence of BPA. The follow up given by local authorities and industry professional associations established that the incident was limited and under control (email from PlasticsEurope and World Association of the Manufacturers of Bottles and Teats to the European Commission from 30 May 2013 provided to EFSA on 31 May 2013).

A survey on potential migrants, including BPA, from non-PC baby bottles was performed by A potential exposure was calculated based on a hypothetical group consuming 6 times per day for 3 months (90 days) from these bottles with initial detectable BPA. Data showed that migration decreased by 80 % from 1st to 3rd migration. A linear decrease was assumed, which meant falling below the LOD (0.1 µg/kg) between the 3rd to the 6th use (i.e. day 1). The simulation was based on the experimental value from migration into 50 % ethanol as simulant, i.e. a worst case compared to milk/infant formula. It led to an average of 0.45 µg/kg and the 95th percentile was 1.24 µg/kg (middle bound).

Table 2: Estimated migration values for specific PC food contact materials used in the exposure assessment

	Average BPA migration (µg/l)			Max	Non detects/N
	LB	MB[(a)]	UB		
Water cooler with PC reservoirs	0.81	0.81	0.81	4.10	4/100
PC tableware	0.02	0.09	0.16	0.63	217/232
PC kettle	0.11	0.11	0.11	0.32	0/6
PC filter	0.04	0.04	0.04	0.17	2/17
Cookware	0.20	0.29	0.39	7.60	21/26

MB: average (middle bound) BPA concentration (assigning the value for LOD/2 or LOQ/2 when LOD or LOQ is reported); UB: average (upper bound) BPA concentration (assigning the value for LOD or LOQ when LOD or LOQ is reported); LB: average (lower bound) BPA concentration (assigning the value 0 when LOD or LOQ is reported); Max: maximum value reported (assigning LOD or LOQ when LOD or LOQ is reported)
N: total number of samples both from literature and EFSA's call for data
[(a)]MB values were used for exposure estimate

4.3.5. Occurrence data in food

Data on occurrence of BPA in food were retrieved from scientific journals and through EFSA's call for data. European data published from 2006 onwards have been used. Quality criteria for occurrence data in food was assessed (see Chapter 4.2 and Appendix I) and a specific table was developed for data retrieved from the literature (see Table 63 and 64 in Appendix IX).

A total of 2 521 samples of food and beverages were selected as the basis to assess BPA concentrations in the different food categories for the scope of the present opinion. Data from the literature and from the call for data did not show major differences in BPA concentrations and so have been merged to provide one BPA concentration for each food category. These merged BPA concentrations have been used in the exposure calculations.

A specific inclusion criterion for data on occurrence in food reported in the scientific literature is that only foods purchased in the European region (EU and non EU), or purchased in another region of the world but produced in the European region, would be included in the exposure assessment. The reason for this is that data on BPA occurrence in food are collected in order to assess dietary exposure to BPA in Europe. Data from a market basket survey recently conducted in Sweden (Gyllenhammar et al., 2012) were not considered in the exposure assessment since analytical determinations were performed on composite samples of non-canned and some canned products. These values could therefore not be assigned to either canned or non-canned products and the proportion of canned/not canned products in each category could not be considered representative of other European countries. They have however been used for comparison of BPA levels between the market baskets and the occurrence data used in this opinion. Also non-European data are summarised in relation to the descriptions of the food categories (Appendix III - Food categories). These data have been used for comparison with European data as a check of the BPA concentration levels.

The present opinion has assigned BPA concentrations to more specific food categories than the earlier EFSA opinion on BPA (EFSA, 2006a), and the FAO/WHO opinion (2011). In the present opinion all foods were categorised and were assigned a BPA concentration. This approach differs from some earlier opinions where for instance non-canned foods were not assigned a BPA concentration

The large majority of information on the occurrence of BPA in food and beverages were available at the level of individual samples, both from literature and from EFSA's call for data. In the case of aggregated results, average results have been weighed for the number of samples in order to calculate the overall average for the food category. When only a median value was available for aggregated results it was considered as a proxy for the average.

Where available the information on the type of packaging (not packaged, canned, glass jar with metal lid, etc.) was reported and codified. When this information was not available, but assumptions could be made that the food was most likely non-canned (e.g. pizza, coffee), it was assigned to the non-canned food category. Otherwise the information was not used in the calculation.

Analytical data were grouped according to the type of packaging and to the food category, with the use of EFSA's food classification and description system FoodEx system. The assumption is that a large portion of the variability observed in BPA concentration between samples of the same food category is related to the packaging. Thus, in the study by Grumetto et al. (2008) on peeled tomatoes, no BPA could be detected in products packaged in glass whereas BPA could be detected in more than half of canned products. Analytical data were grouped by food category, since it was observed that BPA concentration in food with the same type of packaging could vary according to the type of food, i.e. lower BPA concentrations were observed in canned beverages compared to solid foods (Geens et al., 2012a).

Systematic differences in BPA concentration between canned and non-canned food were observed in the large majority of food categories, with higher BPA concentrations in the canned food. However, noteworthy differences in BPA levels can also be observed within the canned and the non-canned food

categories as illustrated in Table 3 (see column "All – Average BPA"). Seven out of 17 canned food categories present have an average (MB) BPA concentration above 30 µg/kg ("Grain and grain-based products", "Legumes, nuts and oilseeds", "Meat and meat products", "Fish and other seafood", "Herbs, spices and condiments", "Composite food", and "Snacks, desserts, and other foods"). Four of the canned food categories have average BPA concentrations (MB) between 2.7 and 23.5 µg/kg ("Vegetables and vegetable products", "Fruit and fruit products", "Fruit and vegetable juices", and "Milk and dairy products"), while the remaining 6 categories have average BPA concentrations (MB) below 1.2 µg/kg.

Among the 19 non-canned food categories, the highest levels of BPA were found in the categories "Meat and meat products" and "Fish and other seafood" with average BPA concentrations (MB) of 9.4 and 7.4 µg/kg, respectively (Table 3, column "All – average BPA").

Any BPA to which food production animals are exposed is likely to be present in their tissues as glucuronated BPA (ANSES, 2013). When BPA is measured in food of animal origin (e.g. meat, milk, eggs), it is possible that deconjugation occurs. Another potential source of unconjugated BPA in meat products is its migration from any food contact materials or from articles used in the processing of the product. With the exception of the data submitted by France through EFSA's call for data, none of the methods, published in the scientific literature or obtained through the EFSA's call, described deconjugation steps and so it was assumed that the BPA concentrations reported were for unconjugated BPA only. The levels of total and unconjugated BPA in foods of animal origin were reported by ANSES to be virtually the same (ANSES, 2013). Therefore the data on total BPA reported by France were merged with the other data from EFSA's call for data.

For the remaining 17 non-canned food categories the average BPA concentrations (MB) were all equal to or below 1.2 µg/kg, with the exception of "Composite foods", which includes fish and meat based products and had a BPA average equal to 2.4 µg/kg.

When comparing the European with non-European concentration data, average BPA levels of concentration resulted mostly in the same range as the samples from Europe. However, there were single non-European foods that were reported to have higher BPA concentrations than found in Europe. For instance some canned beans and peas from the United States of America (USA) had a concentration four times above the highest European value, and a canned mango from Singapore with ten times higher values. It seems however that these very high values may be outliers and not representative for the non-European BPA concentrations. Data presented at the national meeting of the American Chemical Society meeting in April 2013 indicated that BPA concentrations in foods which are produced and canned in Japan have dropped considerably since 2000. In comparison to imported canned food from other countries the decrease has been of the order of a factor of 10-20. Concentration values for Japanese canned food are in the range of some tens µg/kg. (summary provided to EFSA by K. Kawamura by email on 23 May 2013).

A comprehensive description of data from the EFSA's call for data can be found in Appendix II.

1145 **Table 3:** Summary of average BPA concentrations (μg/kg) from the literature and EFSA's call for
1146 data

Food category and type of packaging (canned or non-canned)	Literature				Call for data				All					
	N [(a)]	MB [(b)]	% <LOD/LOQ [(e)]	Max [(f)]	N [(a)]	MB [(b)]	% <LOD/LOQ [(e)]	Max [(f)]	N [(a)]	LB [(d)]	MB [(b)]	UB [(c)]	% <LOD/LOQ [(e)]	Max [(f)]
Canned														
Grains and grain-based products	1	67.4	0	67.4	18	34.9	0	47.5	19	36.6	36.6	36.6	0	67.4
Vegetables and vegetable products	50	26.0	40	116	73	21.7	18	100	123	22.9	23.5	24.0	27	116
Legumes, nuts and oilseeds	2	121	0	103	18	28.8	33	137	20	32.6	34.6	36.6	30	137
Fruit and fruit products	7	15.9	0	24.4	14	12.2	21	107	21	13.1	13.4	13.7	14	107
Meat and meat products	31	14.7	39	51.1	16	64.2	38	203	47	27.7	31.5	35.4	45	203
Fish and other seafood	107	39.5	20	169	67	33.0	33	198	174	34.7	37.0	39.2	27	198
Milk and dairy products	19	2.6	63	15.2	3	19.8	0	35.9	22	4.4	4.9	5.5	55	35.9
Sugar and confectionary	1	0.2	0	0.2	-	-	-	-	1	0.2	0.2	0.2	0	0.2
Fruit and vegetable juices	5	2.7	0	4.7	-	-	-	-	5	2.7	2.7	2.7	0	4.7
Non alcoholic beverages	54	0.5	26	8.1	11	0.5	27	1.5	65	0.5	0.5	0.5	26	8.1
Alcoholic beverages	18	0.9	17	4.7	49	0.8	35	4.5	67	0.7	0.8	0.8	30	4.7
Drinking water	11	0	100	0	-	-	-	-	11	0.0	0.0	0.0	0	0
Herbs, spices and condiments	-	-	-	-	2	41.4	0	82.1	2	41.4	41.4	41.4	0	82.1
Food for infants and small children	10	0.3	70	2.2	-	-	-	-	10	0.3	0.3	0.3	70	2.2
Products for special nutritional use	14	1.2	36	4.8	-	-	-	-	14	1.2	1.2	1.2	36	4.8
Composite food	6	25.9	17	73.1	25	39.6	20	110	31	34.6	37.0	39.3	19	110
Snacks, desserts, and other foods	1	52.0	0	52.0	-	-	-	-	1	52.0	52.0	52.0	0	52.0
Non-canned														
Grains and grain-based products	1	0.9	0	0.9	95	1.0	43	11.9	96	0.8	1.0	1.1	43	11.9

Food category and type of packaging (canned or non-canned)	Literature				Call for data				All					
	N [a]	MB [b]	% <LOD/LOQ [e]	Max [f]	N [a]	MB [b]	% <LOD/LOQ [e]	Max [f]	N [a]	LB [d]	MB [b]	UB [c]	% <LOD/LOQ [e]	Max [f]
Vegetables and vegetable products	4	0.4	0	1.0	201	1.2	34	5.3	205	1.2	1.2	1.3	33	5.3
Starchy roots and tubers	-	-	-	-	45	0.7	16	2.6	45	0.6	0.7	0.7	16	2.6
Legumes, nuts and tubers	-	-	-	-	5	0.2	60	0.5	5	0.1	0.2	0.3	60	0.5
Fruit and fruit products	3	0.5	0	1.3	85	0.3	73	2.1	88	0.2	0.3	0.4	71	2.1
Meat and meat products	1	0.9	0	0.9	191	9.5	5	395	192	9.4	9.4	9.5	5	395
Fish and other seafood	8	1.9	75	11.2	68	8.1	3	97.9	76	7.4	7.4	7.4	11	97.9
Milk and dairy products	1	2.6	100	-	151	0.3	52	6.1	152	0.2	0.3	0.4	52	6.1
Eggs and egg products	-	-	-	-	15	0.9	20	4.5	15	0.8	0.9	0.9	20	4.5
Sugar and confectionary	1	0.3	0	0.3	19	0.5	42	2.6	20	0.5	0.5	0.6	40	2.6
Animal and vegetable fats and oils	-	-	-	-	26	0.5	46	1.4	26	0.3	0.5	0.7	46	1.4
Fruit and vegetable juices	2	0.01	100	-	14	0.8	71	6.0	16	0.4	0.7	0.9	75	6.0
Non alcoholic beverages	1	0.01	100	-	72	0.2	64	1.7	73	0.1	0.2	0.2	64	1.7
Alcoholic beverages	59	0.5	22	2.1	35	0.5	71	1.6	94	0.4	0.5	0.6	40	2.1
Drinking water	159	0.2	90	4.4	460	0.2	84	4.5	619	0.2	0.2	0.2	84	4.5
Herbs, spices and condiments	2	0.3	0	0.3	17	1.3	71	2.5	19	0.2	1.2	2.2	63	2.5
Food for infants and small children	1	0.9	100	-	-	-	-	-	1	0.0	0.9	1.7	100	-
Composite food	3	0.3	0	0.4	107	2.4	13	25.8	110	2.3	2.4	2.4	13	25.8
Snacks, desserts, and other foods	-	-	-	-	31	0.4	68	0.4	31	0.1	0.4	0.7	68	0.4

[a] N: number of samples
[b] MB: average (middle bound) BPA concentration (assigning the value for LOD/2 or LOQ/2 when LOD or LOQ is reported)
[c] UB: average (upper bound) BPA concentration (assigning the value for LOD or LOQ when LOD or LOQ is reported)
[d] LB: average (lower bound) BPA concentration (assigning the value 0 when LOD or LOQ is reported)
[e] % <LOD/LOQ: percentage of samples below limit of detection/limit of quantification
[f] Max: highest BPA concentration

4.3.6. Occurrence, migration and transfer data from non-dietary sources

Occurrence, migration and transfer data for BPA from non-food sources were retrieved from scientific journals and risk assessment reports (FAO/WHO, 2011; ANSES, 2013); an overview of the literature concerning non-food sources considered is given in Appendix IV. The quality of each study was assessed on the basis of the criteria in Chapter 4.2 and Appendix I. All available information was collected, with a focus on environmental matrices sampled in Europe or consumer articles sold in Europe. The term "non-food sources" summarises all sources that contribute to exposure via pathways other than the food pathway (food pathway: food itself, migration from food contact materials, migration from the lining of water supply pipes).

Environmental media can be inhaled (air-born dust, vapours) or ingested (water, dust) directly, so that occurrence can be directly linked to exposure. Drinking water is not considered as an environmental medium, since it is classified as food (see Table 3, Chapter 4.3.5), but untreated surface water may be ingested occasionally during e.g. swimming in a lake. Consumer products and articles are included as non-food sources in the present assessment only if they are potentially in close contact with the consumer (e.g. dermal exposure, mouthing, hand-to-mouth contact possible) and if migration and/or transfer rates have been reported. This is e.g. the case for children's toys (KEMI, 2012) and indicatively for thermal paper. Consequently, for consumer products, in addition to occurrence data also data for migration into saliva and transfer to skin are summarised in this Chapter.

The pathway of exposure via medical devices and medical materials is currently under review by the Scientific Committee on emerging and newly identified health risks (SCENIHR) of DG SANCO. Therefore, occurrence data are summarised below, but only relevant dental materials are included in the exposure assessment as these are medical treatments applied on a regular basis for a large proportion of the population.

The known sources of exposure that presumably are the most relevant for the consumers by magnitude of exposure and prevalence of sources are discussed below.

Environmental sources (air, dust and surface water)

Outdoor air

Data for outdoor air in Europe are only available from two studies in Greece and in France. In Greece the presence of BPA was determined in outdoor air in the city of Thessaloniki (Salapasidou et al., 2011). From January to February 2007, ambient PM10 (particle matter < 10 µm) was sampled from an urban traffic site and an industrial site. BPA in the particulate phase was collected using a low flow air sampler over 24 h and analysed by GC-MS. BPA concentrations measured in the particulate phase ranged between 0.06 and 47.3 ng/m^3. At the urban traffic site, the BPA concentrations in the particulate phase ranged from 0.06-18.6 ng/m^3 (average 6.78 ng/m^3); at the industrial site the BPA concentrations ranged from LOD-47.3 ng/m^3 (average 13.2 ng/m^3). It was estimated that 99 % of the BPA is present in the particulate phase and only a small fraction is present in the gaseous phase of the air.

The first results from a French study show that BPA was detected in the outdoor air in the gaseous phase and particulate phase in an urban setting in Paris and in the forest in Fontainebleau at concentrations varying from 1 to a few ng/m^3 (ANSES, 2013).

Further data for outdoor air are available from the USA. Wilson et al. (2007) collected outdoor air samples in children's homes and day care centres in two states in the USA (North Carolina and Ohio). Outdoor air concentrations (75th percentiles) ranged between 1.0 and 1.5 ng/m^3 in North Carolina and between 0.7 and 0.9 ng/m^3 in Ohio. The 50th percentile values were below the method detection limit (not fully specified, around 0.9 ng/m^3). These levels were confirmed by Rudel et al. (2010), who measured BPA in outdoor air in Richmond and Bolinas (California, USA). Median levels were around 0.5 ng/m^3, the highest level was below 2 ng/m^3. For Osaka, Japan, Matsumoto et al. (2005) measured BPA in urban ambient outdoor air during six months. Samples were collected using a high volume air sampler situated on a roof top and analysed with GC/MS. BPA concentrations ranged from 0.02 to 1.92 ng/m^3, with an average of 0.51 ng/m^3. The highest and lowest average concentrations were reported for February and October, respectively. Fu and Kawamura (2010) reported that the concentrations of BPA in outdoor air ranged over four orders of magnitude in the world (0.001-17.4 ng/m^3, aerosol sampling) with a declining trend from the Continents to remote sites. The highest concentrations were measured in the rural areas (mainly in Asia, no data for Europe were reported). The two US studies show that the concentration levels in indoor air are higher than those in outdoor air, suggesting that the indoor air in the house contributes more than the outside air to exposure to BPA through inhalation in the general population. For this reason and because of the high variations in the data for outdoor air for Europe (only from one Member State) this source was not considered in the exposure assessment.

Indoor air

Volatilisation and/or abrasion of very small particles from epoxy-based floorings, adhesives, paints, electronic equipments, and printed circuit boards are a source of contamination of indoor air and dust (Loganathan and Kannan, 2011).

Since BPA has a comparatively low vapour pressure, from indoor air it is deposited onto surfaces or dust. As a result of the low vapour pressure, concentrations of BPA in air can be expected to be low and it will be present mainly in the particulate phase, adsorbed to dust. European data are only available from one recent report by ANSES (2013). BPA levels were measured in indoor air of 30 French homes with an average of 1.0 ng/m^3 (median: 0.6 ng/m^3) in the particulate phase of the air. The highest level was 5.3 ng/m^3.

US data are in the same range. Wilson et al. (2007) measured indoor air concentrations in 257 US homes with an LOD around 0.9 ng/m^3 (LOD deduced by Beronius and Hanberg, 2011). Concentrations in indoor air from homes and daycare centers ranged from < LOD to 193 and 8.99

ng/m^3, respectively, with a median and 95th percentile for homes of 1.82 and 11.1 ng/m^3, respectively. A second study from the USA (Rudel et al., 2010) determined BPA in indoor air of 50 non smoking Californian households. BPA was only found in 5 samples with concentrations of 0.5 to 20 ng/m^3, the median for all samples was given as 0.5 ng/m^3 (which was also the LOD).

For the exposure calculation the average level of 1 ng/m^3 reported by ANSES (2013) was used as this is the only study available for indoor air in Europe.

Dust

Ingestion of house dust was reported to be an exposure pathway of BPA in young children due to the use in a variety of indoor applications and consumer products, and due to children's more frequent hand-to-mouth contact and larger intake of dust compared to adults (Jones-Otazo et al., 2005; Calafat et al., 2008). BPA was observed in dust from homes, laboratories (Loganathan and Kannan, 2011) and offices (Geens et al., 2009a). Data for Europe are available from three studies conducted in Germany (Völkel et al., 2008), Belgium (Geens et al., 2009a) and France (ANSES, 2013). They are in the same order of magnitude as data from private homes in the USA (Rudel et al., 2003; Loganathan and Kannan, 2011).

Völkel et al. (2008) measured BPA in dust from 12 homes in Germany to investigate potential sources of contamination of urine samples in a biomonitoring study. Samples were collected by residents in homes using regular vacuum cleaners. BPA concentrations in dust ranged from 117 to 1 486 µg/kg with a median of 553 µg/kg.

Geens et al. (2009a) measured concentrations of BPA in indoor dust from 18 homes and 2 offices in Belgium. Samples were collected using a vacuum cleaner. BPA concentrations measured in dust from homes ranged from 535 to 9 729 µg/kg with a median of 1 460 µg/kg. The concentrations of BPA in dust from the two offices were 4 685 and 8 380 µg/kg. The reason for the higher concentrations of BPA in offices was not explained by the authors.

ANSES (2013) measured settled dust in 25 houses in France. The average, median and maximum concentrations of BPA were 5.8, 4.7 and 20 mg/kg, respectively.

For the exposure calculation, the median dust concentration of 1 460 µg/kg was taken from Geens et al. (2009a). This value was chosen for the exposure assessment, because the author reported the average median concentrations among the recent dust studies available for Europe.

Surface water

In a recent study, the concentrations of BPA in North American and European aquatic environments were critically reviewed and statistically characterised (Klecka et al., 2007). A total of 100 papers or reports, published between 1991 and 2007, were identified that contained environmental monitoring data for BPA in European and North American surface water and sediment. Median BPA concentrations in freshwater in Europe were lower than those for North America (0.01 and 0.08 µg/l, respectively), although the 95th percentile concentrations were similar (0.35 and 0.47 µg/l, respectively).

Deblonde et al. (2011) reported concentrations of BPA in wastewater treatment plants to range from 0.088 to 11.8 µg/l in the influent and from 0.006 to 4.09 µg/l in the effluent. This is in agreement with the levels reported by Klecka et al. (2007).

Data on BPA from surface water were not included in the exposure assessment as this source contributes very little to the overall dermal exposure as confirmed by ANSES (2013).

Paper products

BPA is present in thermal papers that are used as cash receipts, airline tickets, bus tickets and papers for laboratory use (Liao and Kannan, 2011a). BPA is loosely bound to the paper surface. It has been reported that in Europe, thermal paper containing BPA amounts to 72 (ANSES, 2013) or 80 % (Lassen et al., 2011) of total thermal paper. According to the European Thermal Paper Association BPA is still used in thermal paper and in 2012, 80 % of thermal paper was used for POS (Point of Sales) grades which are mainly used for supermarkets and shop tickets and not for tickets for transport (bus/boarding passes) and tickets for lotteries. (email from European Thermal Paper Association to EFSA from 17 June 2013). In Switzerland 11 samples out of 13 investigated thermal papers contained BPA (Biedermann et al., 2010). Reported values ranged from 8 to 17 g/kg, with a average of 13.3 g/kg. In Sweden, receipt and receipt-like papers contained on average 14 and 16 g/kg, respectively (Östberg and Noaksson, 2010). The highest levels in this study were found in car park tickets and bus tickets with an average concentration of 32 and 23 g/kg, respectively. In Belgium 73 % of collected thermal paper samples had BPA concentrations between 9 and 21 g/kg, the remaining 27 % were <0.1g/kg (Geens et al., 2012a). Similar values have been reported for the USA. 94 % of all thermal receipt papers contained BPA and ranges were from below the LOQ of 1 µg/kg up to 13.9 g/kg (Liao and Kannan, 2011a).

Receipts and bus tickets are commonly stored in wallets in close contact with paper currency. BPA has been shown to be transferred from thermal paper to paper currencies with levels ranging from 0.001 to 82.7 mg/kg for currencies worldwide (Liao and Kannan, 2011b). These levels are considerably lower than levels of BPA in thermal paper. Levels in other paper products are e.g. 3.2-46.1 mg/kg dry matter for recycled toilet paper (Gehring et al., 2004) with BPA originating from the waste paper used in the recycling process. In this case, BPA is included in the bulk of the paper and not readily available from the surface.

BPA may also be present in some cigarette filters (Jackson and Darnell, 1985). However, no analytical data are available for BPA in cigarette filters.

Consequently, consumers are predominantly exposed to BPA in thermal papers by handling cash receipts, tickets etc. Biedermann et al. (2010) determined the amount of BPA transferred to the finger tips of one volunteer by touching thermal paper. Different scenarios were tested with regard to the moisture and grease content of the finger tips. BPA transfer increased with wetness and greasiness. For what the authors called “standard skin” (slightly greasy skin) 5 different thermal papers were touched for 30 seconds. The average transferred amount by one handling was found to be 1.1 µg BPA per finger. In another study, migration from paper receipts from Denmark was investigated (Lassen et al., 2011). 8 fingers touched 5 different receipts for 10 seconds. Migration to dry fingers on average was 11 µg, i.e. 1.4 µg/finger, which is similar to the value derived by Biedermann et al. (2010). In order to create a conservative average value, the latter value was used in this assessment.

Children’s toys and articles intended to be mouthed

Information on the potential exposure to BPA from toys in children is rather limited. A recent study (Viñas et al., 2012) investigated migration of BPA into artificial saliva from articles purchased in Spanish supermarkets. Migration from 2 toys and 3 pacifiers tested by 1 min immersion without stirring in 100 ml of artificial saliva was in the range of 0.2-0.3 µg/l, while the migration from a teether was 5.9 µg/l. The contact time of 1 min used by Viñas et al. (2012) was considered too short to account for real migration, and therefore the data from this study are not used.

In another migration study, toys and pacifiers from the Swedish market were put into contact with artificial saliva at 24 °C for 24 h (KEMI, 2012) by submersing the toys in the smallest volume of artificial saliva needed to completely cover the toys, which was between 100-700 ml (pers comm. KEMI, 2013). Migration of <0.1 µg/l (LOQ) up to 2.1 µg/l was reported with 8 of 14 toys/pacifiers below LOQ. The maximum levels of 2.1 µg/l were reported for a rattle (0.63 µg BPA migration per

product) and a pacifier (0.21 µg BPA migration per product). The average values in this study were 0.14 µg/product for rattles and 0.11 µg/product for pacifiers. The authors of the study state that it had been difficult to find children's products made of polycarbonate. In order to find 14 products that contained BPA they had to buy altogether 80 products.

Migration from pacifiers into artificial saliva was also determined by Lassen et al. (2011). BPA was detected in 6 out of 8 migration experiments (LOD: 0.1 µg/kg saliva). The maximal amount detected was 1.36 µg migration after 7.75 h at 37 °C. Average amounts were from 0.28 to 0.36 µg/product (lower to higher bound), and the average middle bound was 0.32 µg/product.

Exposure was calculated from rattles as a surrogate for any PC toy that can be mouthed (general population - children) and pacifiers with PC shields (specific population groups). Migration data for rattles from KEMI (2012) were used in the exposure calculation: the average migration (middle bound) was 0.14 µg/product. For pacifiers the average middle bound found by Lassen et al. (2011) was used (0.32 µg/product).

Cosmetics

In Europe, BPA is not permitted as an ingredient in cosmetics (Appendix II: list of substances prohibited in cosmetic products of Regulation (EC) no 1223/2009 of the European Parliament and of the Council of 30 November 2009 on cosmetic products[17]). However, if BPA was present in the packaging (e.g. PC packaging), it could migrate into the cosmetic products.

European data on BPA in cosmetics are very scarce. A recent study (Cacho et al., 2013) reports levels of <LOQ to 88 µg/kg for different cosmetics (shower gel, hair gel, face lotion, make-up remover and mouthwash) bought in Spain. Also world-wide data are scarce. Another recent study reports BPA concentrations, banded in the crude range of 1-100 mg/kg in a number of personal care products bought in the USA such as bar soap, body lotion, shampoo, conditioner, shaving cream, face lotion, facial cleanser, body wash and nail polish (Dodson et al., 2012). No reasoning was given by the authors as to why BPA was present in these products.

As shown by Cacho et al. (2013) BPA can be present in trace amounts in cosmetics. The source could be migration from cosmetic packaging or alternatively BPA may be present as an impurity in the cosmetic ingredients. The European cosmetics legislation allows impurities to be present in "small quantity" (Cosmetics Directive Article 17) as long as it is "safe for human health" (Article 3). Cosmetics could therefore contain trace amounts of BPA as impurity. The most important contribution to exposure will be from body lotion, because of the large body surface that is treated and since this product is nearly entirely taken up by the skin (Lorenz et al., 2011). The concentration of 31 µg/kg found in facial lotion by Cacho et al. (2013) was chosen for exposure calculation from e.g. the use of body lotion.

Medical devices

Medical devices are a particular product category in which BPA is found. Examples of these products are implants, catheters, and dental devices. BPA-containing medical devices may have direct and/or indirect contact with the patients (e.g. autotransfusion apparatus, filters, bypasses, tubing, pumps, instruments, surgical equipment, blood pathway circuits and respiratory tubing circuits). The pathway of exposure via medical devices is currently under review by SCENIHR of DG SANCO. In the present assessment, where the risk of BPA for the general public is assessed, the exposure to these medical devices will not be included, since they are used in specific sub populations only. However, dental materials are used in the general population, so the exposure to BPA via this application is considered here.

[17] Regulation (EC) No 1223/2009 of the European Parliament and of the Council of 30 November 2009 on cosmetic products, OJ L342, 22.12.2009, p.59-209

Dental materials

Dental sealants and composite filling materials containing BPA are used in dentistry, especially in children (Fleisch et al., 2010). The most commonly used BPA-derived material is BPA glycidyl methacrylate (bis-GMA). BPA dimethacrylate (bis-DMA), BADGE and BPA ethoxylate dimethacrylate (bis-EMA) are also used. The resins are polymerised *in situ* during placement of dental sealants and unpolymerised material may be released into saliva directly after treatment. The release of BPA over time due to hydrolysis of the resin (Pulgar et al., 2000) was reported. However, other studies describe BPA exposure after dental sealant placement as an acute event (Fleisch et al., 2010; Kang et al., 2011). Variability between brands and analytical method sensitivity and uncertainty make it difficult to draw conclusions regarding exposure from this source (Beronius and Hanberg, 2011). Polydorou et al. (2009a) demonstrated that bleaching did not increase the release of BPA from composite materials.

Van Landuyt et al. (2011) reviewed the release of substances from dental materials into water-based solutions and the highest individual value for BPA was 67 nmol/mm^2 surface area of material. According to Van Landuyt, the value corresponds to a worst case release of 132 µmol after 24 h on one full crown restoration of a molar.

Zimmerman-Downs et al. (2010) studied the effect of dental sealants on the BPA concentration in saliva in 30 volunteers (with no history with dental sealants or composite material treatment). One group of 15 volunteers received one occlusal sealant, the other group received 4 sealants. One h before treatment, the mean baseline value was around 1 µg/l saliva. In the high dose group, the mean peak value was 6 µg/l (measured one hour after treatment) whereas in the low dose group this mean peak value was around 2 µg/l. Sasaki et al. (2005) measured BPA levels in saliva in 21 volunteers after restoration with composite resins (from 9 different companies). BPA levels in saliva ranged from several tens to 100 µg/l but sufficient gargling could remove it from the oral cavity. Both studies indicate that BPA levels in saliva return to baseline (1 µg/l saliva) after 24 h.

A few studies have also investigated systemic absorption of BPA after placement of dental sealants. Measured levels in blood up to five days after sealant placement could not detect any BPA (Fung et al., 2000; Zimmerman-Downs et al., 2010). Median urinary levels of BPA increased from 2.4 µg/l (pretreatment) to 12.9 µg/l 1 h after treatment with one type of sealant but treatment with another brand did not result in the same increase in urinary concentrations (Joskow et al., 2006). Urinary concentrations of BPA had decreased significantly after 24h but were not completely back to baseline within this time.

Kang et al. (2011) reported BPA levels in saliva and urine samples collected from 22 volunteers who received a lingual bonded retainer on their mandibular dentition. Samples were collected immediately before placement and 30 minutes, 1 day, 1 week, and 1 month after placement. The only significantly high level of BPA was observed in the saliva collected just after placement of the lingual bonded retainer (average 5 µg/l; max value 21 µg/l). One day after placement, the level decreased to the background level again (average value: 0.5 µg/l saliva). No statistically significant increase of BPA in the urine samples at any time point was observed.

For the exposure assessment the value was used that occurs on a chronic basis, which is the background level of 0.5 µg/l (Kang et al., 2011). However, it can be argued whether this background level relates to lingual bonded retainer or is the consequence of the exposure to other sources.

Based on the assessment of occurrence, migration and transfer data presented above, the data presented in Table 4 have been selected for use in the exposure calculation for non-food (see chapter 4.6.3).

Table 4: Overview of BPA concentrations and sources considered for the present exposure assessment

Source	Pathway	Type of study (direct/migration/transfer)	BPA concentration	Unit	Reference	Reasoning
Air	Inhalation	direct	1.0	ng/m^3	Anses, 2013	Single data source for indoor air in Europe
Dust	Inhalation/Ingestion	direct	1 460	µg/kg dust	Geens et al., 2009a	Middle median from three European studies
Thermal paper	Dermal	transfer to finger	1.4	µg/finger	Lassen et al., 2011	Most extensive study available
Toys (rattle)	Ingestion	migration into saliva	0.14	µg/product	KEMI, 2012	Most reliable study conditions
Pacifiers with PC shields	Ingestion	migration into saliva	0.32	µg/product	Lassen et al., 2011	Most reliable study conditions
Cosmetics	Dermal	direct	31	µg/kg	Cacho et al., 2013	Single data source for cosmetics in Europe, value for face lotion used
Dental materials	Ingestion	migration into saliva	0.5	µg/l	Kang et al., 2011	Most extensive study available

4.4. Food consumption

Data from the EFSA Comprehensive European Food Consumption Database (hereafter called Comprehensive Database) were used to assess dietary exposure to BPA in all age groups excluding infants aged 0 to 6 months. The Comprehensive Database was built in 2010 from existing national information on food consumption at a detailed level. Competent organisations in the European Union Member States provided EFSA with data from the most recent national dietary survey in their country at the level of consumption by the individual consumer. Survey results for children were mainly obtained through the EFSA Article 36 project "Individual food consumption data and exposure assessment studies for children" through the EXPOCHI consortium (EFSA, 2011). Results from a total of 32 different dietary surveys carried out in 22 different Member States covering more than 67 000 individuals are included in the Comprehensive Database version 1 as published (EFSA, 2011; Merten et al., 2011).

There are two surveys available for infants, nine surveys available for toddlers, 17 surveys available for other children, 12 surveys available for teenagers, 15 surveys available for adults, seven surveys available for elderly, and six surveys available for very elderly. Only surveys covering more than one day, and thus appropriate for calculating chronic exposure, were selected. For each survey, food consumption data are coded according to the FoodEx classification system.

4.5. Parameters used to assess non-dietary exposure

4.5.1. Inhalation absorption

For inhalation the same absorption factor as for ingestion, i.e.1, was assumed.

4.5.2. Dermal absorption

Bisphenol A penetration of skin has been investigated *in vitro* by using Franz cells with human skin (Zalko et al., 2011; Demierre et al., 2012), pig skin (Kaddar et al., 2008; Zalko et al., 2011,) and rat skin (Marquet et al., 2011). Penetration has been assessed also *in vivo* in rats (Marquet et al., 2011). Since rat skin has been shown to absorb BPA 10 times as fast as human skin (Marquet et al., 2011), the results with rat skin are too conservative and will not be used for deriving a human absorption fraction in this assessment. The pig and human skin *in vitro* studies have been conducted over different durations (24 h, 48 h and 72 h), all by using ^{14}C labelled BPA. All studies show increasing penetration with time and no study was conducted over a large enough time span to reach the maximum absorption. Thus, the determined absorption fractions in the human and pig skin *in vitro* studies that range between 10 and 47 % may underestimate the actual absorption.

In a study by Biedermann et al. (2010), an attempt was made to investigate dermal absorption by exposing living humans. Here, not the transfer to blood was assessed, but BPA was applied in different forms to the finger tips of a human volunteer and recovery from the finger tips was determined for different exposure times by measuring BPA in the extraction solution. The calculated amounts that remained in the skin after extraction can be seen as upper boundary values for dermal absorption, even if not all BPA remaining in the skin will finally reach the blood stream.

In one experiment BPA was dissolved in ethanol (10 mg/ml) and 1µl of this solution was applied directly to the skin of finger tips. For this experiment a recovery of 40 % after 1.5 h was reported (determined by extraction from skin with ethanol over 30seconds), from which a maximal dermal absorption fraction of 60 % can be deduced. Another experiment with the same amount of BPA in a larger volume of solvent (10 µl, 1 mg/ml) showed a recovery < 5 %, which implies that the maximal dermal absorption of BPA can reach 95-100 % if BPA is applied dissolved in ethanol. Ethanol may act as a transport mediator for BPA into the skin, thus enhancing the absorption fraction. Therefore, the dermal absorption fraction derived for BPA in ethanol may be used for BPA in formulations that have similar vehicle properties as ethanol (e.g. emulsions such as body lotions and creams).

In the same study, Biedermann et al. (2010) also investigated the dermal absorption from finger tips after touching of thermal paper. In this experiment 27 % absorption was derived after 2 h, if hands were not washed in the meantime. Some uncertainty in this experiment is associated with the fact that non labelled material was used and, hence, the varying amount transferred from thermal paper to skin in different experiments is introduced into the absorption fraction (the amount recovered after 2 h divided by the amount recovered immediately for deriving the absorption fraction). Another uncertainty is that apparently only one volunteer had been used for the experiments. The value of 27 % was therefore considered as too precise and rounded up to 30 %.

In the light of the *in vitro* studies failing to provide a reliable upper boundary for dermal absorption, the study of Biedermann et al. (2010) was used for the dermal exposure assessment. Specifically, the absorption fraction of 30 % was used for dermal exposure from thermal paper. For BPA in cosmetics the absorption fraction of 60 % was used because in cosmetics BPA is present in the dissolved form and absorption may be enhanced by substances acting as vehicle.

4.6. Exposure estimation

4.6.1. General assumptions for calculation

For each source of exposure (dietary; non-dietary oral, inhalation and dermal) and in each age group (infants (0-1 year), toddlers (1-3 years), other children (3-10 years), teenagers (10-18 years), women (18-45 years), men (18-45 years), other adults (45-65 years), elderly and very elderly (over 65 years) (EFSA, 2011), a scenario for average exposure and a scenario for high exposure has been developed. Only average exposure from the different sources have been added together to assess total exposure. In order to quantify the relative impact of each source, the assumptions made in the exposure assessments were aimed at obtaining a similar degree of conservativeness among the different sources.

In the case of infants, due to their very monotonous dietary pattern, loyalty was considered. Thus, high exposure was assessed considering that some infants might be systematically exposed to products containing a higher concentration of BPA, e.g. an infant formula containing a high concentration of BPA or a baby bottle releasing more BPA than other bottles. In other age classes, an average BPA concentration was considered and high chronic exposure was assessed considering higher levels of consumption or of contact with products containing BPA.

As far as possible, exposure to total BPA from dietary and other sources has been calculated. Where possible, exposure to conjugated and unconjugated BPA has been assessed separately, i.e. through food.

Biomonitoring studies have been used to assess how much total BPA is excreted in urine, allowing the estimation of exposure from all sources to total BPA. These estimates have been compared to the total calculated exposure value, as a check of plausibility. In addition, biomonitoring studies might be able to identify the existence of unrecognised source of exposure.

4.6.2. Exposure estimation from dietary sources

Dietary exposure to BPA in infants aged less than 6 months has been assessed by means of a model diet based on a standard level of consumption combined with BPA concentration in human milk or infant formula. Average and high BPA concentration values have been used to assess average and high chronic dietary exposure.

Dietary exposure from colostrum and human milk

Initial human milk (colostrum), which is produced during the first to approximately 5 days after delivery, differs from mature human milk. The assessment of exposure to BPA in the first few days of life has therefore been considered separately.

The quantity of initial human milk consumed by infants on their very first day of life is very small; it was estimated to be 44 ± 71 g (mean ± SD) by Neville et al. (1988) and as low as 15 ± 11 g by Santoro et al. (2010). The quantity of initial human milk consumed increases steadily each day and reaches around 500 g/day on the fifth day of life (Neville et al., 1988). Taking an average consumption of 250 g over the first 5 days, and assuming an average body weight for a newborn of 3.25 kg, an average consumption rate of 75 g/kg bw/day (rounded by 5-gram steps) is obtained. For infants aged 5 days to 3 months the average level of consumption of 150 g/kg bw/day considered by US EPA (US EPA, 2011) to derive exposure factors in the first month of life was used here. Since human milk consumption per kg bw decreases steadily from month 1 to month 3, the level of consumption observed at month 1 allows to perform a conservative assessment of exposure for this age class up to 3 months old. For infants aged up to 3 months and breastfed with mature human milk a level of consumption of 150 mg/kg bw/day and for breastfed infants aged 4 to 6 months, the level of consumption established in the EFSA opinion on default assumptions (132 g/kg bw/day) (EFSA Scientific Committee, 2012) was considered.

Based on data from the scientific literature described in chapter 4.8.4, average exposure for infants aged 1-5 days was assessed assuming that initial human milk would contain 3 µg of total BPA/kg whereas high exposure was assessed assuming that initial human milk would contain 6.6 µg of total BPA/kg. The CEF Panel noted that only very few data from Europe and/or obtained by a reliable analytical method were available and therefore decided to take into account data from Japan, reporting the above BPA concentrations. The Panel noted, however, that these data had significant limitations, including the use of ELISA methodology and the fact that the samples dated back to 2000. These limitations were addressed in the uncertainty analysis. Results are presented in Table 5.

Table 5: Exposure to total BPA from initial human milk

	Consumption of initial human milk (g/kg bw/day)	Average exposure (ng/kg bw/day)	High exposure (ng/kg bw/day)
BPA concentration (µg/l)		3.0	6.6
Infants, day 1-5	75	225	495

Average exposure of breastfed infants from 6 days of age to 6 months was assessed considering that mature milk would contain 0.4 µg of unconjugated BPA/kg and 0.9 µg of total BPA/kg whereas high exposure was assessed considering that mature milk would contain 1.2 µg of unconjugated BPA/kg and 2.6 µg of total BPA/kg. Results are presented in Table 6.

Table 6: Exposure to total and unconjugated BPA from mature human milk

	Consumption of mature human milk (g/kg bw/day)	Average exposure (ng/kg bw/day)		High exposure (ng/kg bw/day)	
		Unconjugated BPA	Total BPA	Unconjugated BPA	Total BPA
BPA concentration (µg/l)		0.4	0.9	1.2	2.6
Infants, 0-3 months	150	60	135	180	390
Infants, 4-6 months	132	53	119	158	343

Dietary exposure from infant formula

The highest level of consumption per kg bw is observed during the first months of life of formula-fed infants. The level of consumption considered (150 g/kg bw/day) is the one which has been considered for water consumption in infants in the recent CEF opinion on the criteria to be used for safety evaluation of recycling processes (EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2011b). The scenario is that of a 5 kg infant consuming 0.75 l of water per day for the reconstitution of infant formula, as suggested by WHO (2003).

Infant formula may be purchased as powder or ready to use (liquid). According to the European Dietetic Food Industry Association (email to EFSA dated 27 June 2013) canned liquid infant formula is not offered in cans in Europe and therefore exposure is not considered here. For powdered infant formula, the factor that is generally considered to calculate the quantity of reconstituted infant formula based on the quantity of powder (1/7) was used (EFSA, 2010).

A specific exposure assessment was performed for infants fed with such formulae, based on the average and high BPA concentration observed in European samples.

In Table 23, reporting exposure to BPA for the general population, only powdered infant formula (canned and not canned) and liquid infant formula not canned have been considered. A unique value, without distinction between these 3 types of formula, has been used based on the following considerations:

For powdered infant formula – canned, based on 10 European analytical data, an average concentration of 0.3 µg/kg and a high concentration of 2.2 µg/kg were considered (see chapter 4.3.5 "Occurrence in food" and Appendix III). Dietary exposure would amount to 6 ng/kg bw/day in an infant fed about 21 g/kg bw/day of infant formula powder (equivalent to 150 g/kg bw/day of ready to drink liquid infant formula) containing an average concentration of 0.3 µg/kg. Since infant formula powder is diluted in water, the baseline BPA contamination of drinking water reported in Table 3 was also considered (middle bound 0.2 µg/kg). Overall, exposure to BPA from the consumption of 150 ml/kg bw/day of reconstituted formula would be 36 ng/kg bw/day at the average (150 x 0.2 + 150 x 0.3 x 1/7) with more estimated BPA deriving from the water than from the powder. High exposure would be 77 ng/kg bw/day (150 x 0.2 + 150 x 2.2 x 1/7).

For powdered infant formula – not canned, only one analytical data was available for Europe (under the limit of detection, middle bound 0.9 µg/kg) whereas no data were available in Europe for liquid infant formula – not canned. Exposure from the consumption of 150 ml/kg bw/day of either reconstituted formula or of liquid infant formula – not canned would mainly derive from the background contamination of water and, based on a middle bound value of 0.2 µg/kg, would be in the range of 30 ng/kg bw/day.

The Panel noted that for these 3 types of formula, BPA concentration values in formulae and water were low and rather uncertain. Overall, no significant difference in exposure is expected between canned infant formula powder and non-canned infant formula (either liquid or powder).

Rough estimates of 30 ng/kg bw/day for average exposure and of 80 ng/kg bw/day for high exposure were therefore considered for these three types of products.

Dietary exposure from water coolers with PC reservoirs, PC water filters and old waterpipes repaired with epoxy resins

Water dispensers (also known as water coolers with PC reservoirs) and water filters can be used at household level (e.g. fridge water dispensers), at work places and in schools. The water coolers with PC reservoirs hold a large bottle (ca 10 l) on top which are often made from PC and are exchanged with a new bottle when empty. When referring to PC coolers in this opinion the actual bottle is meant. Regular consumers of water from these reservoirs are exposed to an additional source of exposure compared to the general population. The same is true for households living in buildings where old water pipes have been repaired with epoxy resins that release BPA into tap water.

Additional chronic exposure to BPA in these specific population groups was assessed considering total water consumption in each age class, as reported in Table 25. Data on the consumption of drinking water was derived from the EFSA Comprehensive European Food Consumption Database (Comprehensive Database) for all age classes, from toddlers to very elderly, at individual level. The median of average consumption and the highest observed 95th percentile are reported and were used to assess average and high exposure. For PC water dispensers, only average exposure was assessed since it is unlikely that high consumption of water would derive exclusively from PC dispensers. For water pipes, high exposure was assessed considering average consumption of water and high BPA concentration that may occur in some buildings.

For infants, the consumption of 150 ml/kg bw/day of water for the reconstitution of infant formula was considered. The use of water coolers with PC reservoirs was not considered for infants since it was considered unlikely that infant formula would be reconstituted with water from such a water dispenser.

For water coolers with PC reservoirs and PC filters, migration values of respectively 0.81 µg/l and 0.04 µg/l were considered (see Table 2 in Chapter 4.3.4. "Estimated migration values for specific PC food contact materials used in the exposure assessment"). For water pipes, the average and high exposure was assessed based on average and high BPA concentration in cold water in those buildings where water pipes had been repaired with a two components technique leading to high release of BPA

(see Chapter 4.3.5. Occurrence in food and Appendix II) of 0.1 and 1.1 µg/l, respectively, combined with the median of average water consumption. Results are presented in Table 7.

Table 7: Exposure to BPA from drinking water in specific population groups based on chronic(a) water consumption as reported in the EFSA Comprehensive Database

	Median of mean water consumption (g/kg bw/day)	Highest 95th percentile of water consumption (g/kg bw/day)	Average exposure (b) (ng/kg bw/day)				High exposure (c) (ng/kg bw/day)
			Water coolers with PC reservoirs	Water pipes			PC filters
BPA (ng/kg bw/day)			**0.81 µg/l**	**0. 1 µg/l**	**1.1 µg/l**	**0.04 µg/l**	**0.04 µg/l**
Toddlers	26.6	95.6	22	2.7	29	1.1	3.8
Other children	19.2	68.8	16	1.9	21	0.8	2.8
Teenagers	10.9	39.4	9	1.1	12	0.4	1.6
Women 18–45 years	9.8	39.2	8	1.0	11	0.4	1.6
Men 18–45 years	7.7	33.8	6	0.8	8	0.3	1.4
Other adults 45–65 years	8.5	32.3	7	0.9	9	0.3	1.3
Elderly and very elderly	10.5	28.6	9	1.1	12	0.4	1.1

(a) In order to assess chronic water consumption, only surveys with at least two survey days were considered.
(b) considering median water consumption
(c) considering high water consumption.

Dietary exposure from PC kettles, PC tableware, cookware and old PC baby bottles

BPA may migrate into food and beverages through contact with PC food contact materials such as tableware used to heat foods and beverages in microwave ovens, tableware used when the food or beverage is eaten (mugs, beakers, plates, bowls), water kettles used to boil water for preparing hot drinks such as coffee, tea or rehydrated soups. Since migration increases with temperature, time of contact and surface of contact, it is likely to be highest when hot beverages are prepared with water heated in a PC kettle or consumed in PC mugs or cups. The case of infant formula reconstituted with water heated in a PC water kettle and of infant fed with formula from an old PC baby bottle bought before the EU ban must also be considered. PC tableware and PC kettles are used only by a fraction of the population but in this fraction of the population who use them regularly it needs to be assessed as an additional source of exposure to BPA.

The migration value chosen to represent average potential migration from PC kettles into water was 0.11 µg/kg. This value is an estimate of BPA concentration in water that would be warmed twice in a kettle and left in it for a total of about 50 minutes (see Table 2 in Chapter 4.3.4. "Estimated migration values for specific PC food contact materials used in the exposure assessment"). It was considered that water heated in a kettle could be used to prepare hot beverages such as coffee (espresso excluded) or tea. Individual consumption data from the Comprehensive Database have been used to estimate the exposure to BPA from kettles. Average and high (95th percentile) exposure have been assessed for each survey and in each age class for the exposure to BPA from PC kettles. Summary data are presented in Table 8. As expected, the highest estimated exposure from PC kettles was observed in other adults and elderly due to their higher consumption of coffee and tea.

Table 8: Exposure to BPA in specific population groups using PC kettles, based on chronic(a) consumption of beverages that could be prepared with hot water, as reported in the EFSA Comprehensive Database

Age group	Median of average consumption of beverages (g/kg bw/day)	Highest 95th percentile of beverages (g/kg bw/day)	Exposure to BPA from PC kettles (ng/kg bw/day)	
			Average	High
Toddlers	0.4	19.3	0.04	2.1
Other children	0.4	16.0	0.05	1.8
Teenagers	1.0	15.4	0.11	1.7
Women 18-45 years	3.3	25.8	0.4	2.8
Men 18-45 years	1.9	23.6	0.2	2.6
Other adults 45-65 years	2.0	29.4	0.2	3.2
Elderly and very elderly	2.5	27.4	0.3	3.0

(a) In order to assess chronic water consumption, only surveys with at least two survey days were considered.

For infants fed with infant formula reconstituted from powder, dietary exposure related to the use of PC kettles to warm the water was assessed considering a water consumption of 150 ml/kg bw/day.

For breastfed infants, the additional exposure from consumption of herbal tea prepared with water heated in a PC kettle was estimated considering the consumption of one small baby bottle (100 ml) per day for a 5 kg infant.

Chronic dietary exposure to BPA from tableware and from cookware was also estimated for age classes from toddlers to elderly with the use of individual consumption data from the Comprehensive Database. In this case all eating occasions of food and beverages which may be consumed hot were

assumed to contain a BPA concentration level equal to 0.09 and 0.29 µg/kg, respectively. These values are the estimated migration during 15 minutes of contact between the food and the tableware (see Table 1 in Chapter 4.3.4. "BPA migration into food simulants"). All food and beverages, with the exception of "alcoholic beverages", "drinking water", "fruit and fruit products" and "fruit and vegetable juices", at the first level of the FoodEx system, were assumed to be consumed hot. Average and high (95th percentile) exposure have been assessed for each survey and in each age class for the exposure to BPA from tableware. Results are presented in Table 9. The highest estimated exposure from PC tableware was observed for toddlers due to their higher consumption of beverages per kg bw. This age class is also the one in which regular use of PC tableware is most likely to occur since `unbreakable` plastic mugs and beakers are often used for toddlers.

Table 9: Exposure to BPA in specific population groups using PC tableware or cookware containing BPA, based on chronic consumption of food that could be consumed warm, as reported in the EFSA Comprehensive Database

Age group	Median of average consumption of food (g/kg bw/day)	Highest 95th percentile of food (g/kg bw/day)	Exposure to BPA (ng/kg bw/day)		Exposure to BPA from (ng/kg bw/day)	
			PC tableware		Cookware	
			Average	High	Average	High
Toddlers	64.6	156.9	6	14	19	46
Other children	46.7	96.6	4	9	14	28
Teenagers	26.0	54.9	2	5	8	16
Women 18-45 years	22.4	52.2	2	5	6	15
Men 18-45 years	22.7	49.2	2	4	7	14
Other adults 45-65 years	21.9	51.0	2	5	6	15
Elderly and very elderly	20.8	49.0	2	4	6	14

The case of infants fed with formula in old PC baby bottles that would have been bought before the EU ban was also considered by combining the consumption level of 150 ml/kg bw/day with an average migration of 0.89 µg/l and a high migration of 4.56 µg//l (see Table 1 in Chapter 4.3.4. "BPA migration into food simulants").

In the 2006 opinion of EFSA a unique value of 5 µg/kg was considered for migration from tableware. The consumption of food in contact with tableware was extremely conservative, in particular for toddlers: 3 kg for a 60 kg adult (50 g/kg bw/day) and 2 kg for a 11 kg toddler (182 g/kg bw/day). Estimated exposure from this source was therefore one order of magnitude higher as compared to the present assessment: 250 ng/kg bw/day in adults and 900 ng/kg bw/day in toddlers.

Assessment of dietary exposure based on the EFSA Comprehensive database

Dietary exposure from 12 months old toddlers to elderly has been estimated using individual consumption data from the EFSA Comprehensive European Food Consumption Database (Comprehensive Database) combined with available concentration data derived from the scientific literature or from EFSA's call for data. In order to consider separately women of childbearing age, in the present assessment the adult age group has been broken down in three subgroups, comprising women from 18 to 45 years old, men from 18 to 45 years old and other adults from 45 to 65 years old. Elderly and the very elderly were merged. Dietary exposure in toddlers (12 to 36 months) was used as estimate for the dietary exposure in infants aged 6 to 12 months.

The average BPA concentration in each food category has been assessed by merging data from different sources or scientific publications (see Chapter 4.3.5). Chronic exposure was estimated by

multiplying the average BPA concentration for each FoodEx level 1 food group (see Appendix V for details) and type of packaging (canned or non-canned) with their respective consumption amount per kg body weight separately for each individual in the database, calculating the sum of exposure for each survey day for the individual and then deriving the daily average for the survey period. Average and 95th percentile exposure was calculated for the total survey population separately for each survey and age class. Details on surveys are given in Table 10.

Only a limited number of dietary surveys included in the Comprehensive Database included information on the type of packaging (canned or non-canned, in particular). The number and percentages of food codes specific for canned products per country and per survey are presented in Table 11.

1684 **Table 10:** Dietary exposure by country survey and age group and scenarios under the middle bound assumption

Country	Survey	Age group	Number of subjects	Middle Bound					
				Scenario 1 (ng/kg bw/day)		Scenario 2 (ng/kg bw/day)		Scenario 2 / Scenario 1	
				Mean	P95	Mean	P95	Mean	P95
United Kingdom	NDNS	Men 18-45 years	459	59	109	112	182	1.9	1.7
United Kingdom	NDNS	Women 18-45 years	587	49	91	107	191	2.2	2.1
Denmark	Danish_Dietary_Survey	Adolescents	479	64	117	137	248	2.1	2.1
United Kingdom	NDNS	Adults 45-65 years	678	51	94	120	201	2.3	2.1
Czech Republic	SISP04	Men 18-45 years	446	55	97	120	220	2.2	2.3
Denmark	Danish_Dietary_Survey	Men 18-45 years	781	51	80	109	182	2.1	2.3
Ireland	NSIFCS	Adults 45-65 years	358	48	85	124	203	2.6	2.4
Italy	INRAN_SCAI_2005_06	Other children	193	120	206	267	502	2.2	2.4
Ireland	NSIFCS	Men 18-45 years	282	55	90	126	218	2.3	2.4
Czech Republic	SISP04	Adults 45-65 years	801	41	75	102	186	2.5	2.5
Spain	AESAN	Women 18-45 years	160	56	126	161	313	2.8	2.5
Spain	AESAN	Men 18-45 years	141	57	100	142	249	2.5	2.5
Italy	INRAN_SCAI_2005_06	Adolescents	247	70	121	169	302	2.4	2.5
Italy	INRAN_SCAI_2005_06	Men 18-45 years	575	50	83	125	209	2.5	2.5
Hungary	National_Repr_Surv	Men 18-45 years	244	46	85	123	217	2.7	2.5
Czech Republic	SISP04	Adolescents	298	59	109	152	277	2.6	2.6
Czech Republic	SISP04	Other children	389	78	142	198	363	2.5	2.6
Denmark	Danish_Dietary_Survey	Elderly and very elderly	329	47	74	111	190	2.4	2.6
Denmark	Danish_Dietary_Survey	Adults 45-65 years	1 117	47	76	115	201	2.4	2.7
Denmark	Danish_Dietary_Survey	Women 18-45 years	924	49	79	119	211	2.4	2.7
Denmark	Danish_Dietary_Survey	Other children	490	102	165	253	446	2.5	2.7
Spain	AESAN_FIAB	Men 18-45 years	367	54	92	148	249	2.7	2.7
Ireland	NSIFCS	Women 18-45 years	318	47	82	123	223	2.6	2.7
Italy	INRAN_SCAI_2005_06	Adults 45-65 years	1 055	47	78	124	219	2.7	2.8
Italy	INRAN_SCAI_2005_06	Women 18-45 years	683	52	87	138	242	2.7	2.8
Spain	AESAN_FIAB	Adolescents	86	63	101	156	293	2.5	2.9

Country	Survey	Age group	Number of subjects	Middle Bound					
				Scenario 1 (ng/kg bw/day)		Scenario 2 (ng/kg bw/day)		Scenario 2 / Scenario 1	
				Mean	P95	Mean	P95	Mean	P95
Czech Republic	SISP04	Women 18-45 years	419	38	67	97	195	2.6	2.9
Germany	National_Nutrition_Survey_II	Adolescents	1 011	41	87	121	252	2.9	2.9
Germany	National_Nutrition_Survey_II	Men 18-45 years	2 517	46	91	127	264	2.8	2.9
Italy	INRAN_SCAI_2005_06	Elderly and very elderly	518	44	70	116	206	2.6	2.9
Hungary	National_Repr_Surv	Adults 45-65 years	503	38	67	113	199	3.0	3.0
Finland	DIPP	Toddlers	497	111	228	316	688	2.8	3.0
Hungary	National_Repr_Surv	Elderly and very elderly	286	35	60	107	183	3.1	3.1
Finland	FINDIET_2007	Men 18-45 years	333	37	59	101	184	2.7	3.1
Sweden	Riksmaten_1997_98	Women 18-45 years	354	42	73	137	228	3.3	3.1
Spain	AESAN_FIAB	Adults 45-65 years	207	52	90	163	283	3.1	3.1
Sweden	Riksmaten_1997_98	Men 18-45 years	352	41	67	127	209	3.1	3.1
Finland	DIPP	Other children	933	87	140	248	440	2.9	3.1
Spain	enKid	Adolescents	209	62	111	190	350	3.0	3.2
Sweden	NFA	Other children	1 473	79	147	263	476	3.3	3.2
Hungary	National_Repr_Surv	Women 18-45 years	327	41	69	120	224	2.9	3.3
Spain	AESAN_FIAB	Women 18-45 years	407	61	99	182	329	3.0	3.3
Bulgaria	NUTRICHILD	Toddlers	428	137	253	431	846	3.1	3.3
Sweden	Riksmaten_1997_98	Adults 45-65 years	504	43	71	141	238	3.3	3.4
Finland	FINDIET_2007	Adults 45-65 years	821	33	57	103	194	3.2	3.4
Spain	NUT_INK05	Adolescents	651	61	103	201	352	3.3	3.4
Germany	National_Nutrition_Survey_II	Women 18-45 years	3 285	38	73	124	251	3.2	3.4
Cyprus	Childhealth	Adolescents	303	41	77	142	269	3.5	3.5
Sweden	NFA	Adolescents	1 018	50	88	163	309	3.2	3.5
Finland	FINDIET_2007	Elderly and very elderly	463	29	51	97	179	3.3	3.5
Germany	National_Nutrition_Survey_II	Elderly and very elderly	2 496	38	70	125	247	3.3	3.5
France	INCA2	Men 18-45 years	517	37	60	121	211	3.3	3.5
Bulgaria	NUTRICHILD	Other children	433	127	223	409	790	3.2	3.6
Germany	National_Nutrition_Survey_II	Adults 45-65 years	4 617	40	75	127	268	3.2	3.6

Country	Survey	Age group	Number of subjects	Middle Bound					
				Scenario 1 (ng/kg bw/day)		Scenario 2 (ng/kg bw/day)		Scenario 2 / Scenario 1	
				Mean	P95	Mean	P95	Mean	P95
Netherlands	DNFCS_2003	Women 18-45 years	398	41	80	142	286	3.5	3.6
Finland	FINDIET_2007	Women 18-45 years	421	33	56	109	205	3.2	3.6
Spain	enKid	Other children	156	96	179	298	668	3.1	3.7
Spain	NUT_INK05	Other children	399	92	148	312	556	3.4	3.8
Netherlands	DNFCS_2003	Men 18-45 years	352	49	89	175	335	3.6	3.8
Spain	AESAN	Adults 45-65 years	109	50	86	158	331	3.2	3.9
Netherlands	VCP_kids	Other children	957	79	160	290	635	3.7	4.0
Greece	Regional_Crete	Other children	839	96	165	345	674	3.6	4.1
France	INCA2	Adolescents	973	43	73	156	307	3.7	4.2
France	INCA2	Adults 45-65 years	947	36	55	138	230	3.8	4.2
Belgium	Diet_National_2004	Men 18-45 years	365	40	69	158	290	4.0	4.2
France	INCA2	Women 18-45 years	812	35	55	132	235	3.8	4.3
Latvia	EFSA_TEST	Men 18-45 years	376	42	76	172	333	4.1	4.4
Germany	DONALD_2006_2008	Other children	660	57	86	215	381	3.8	4.4
Germany	DONALD_2006_2008	Toddlers	261	72	108	235	487	3.3	4.5
France	INCA2	Elderly and very elderly	348	34	51	137	231	4.0	4.6
France	INCA2	Other children	482	75	117	314	550	4.2	4.7
Netherlands	VCP_kids	Toddlers	322	97	178	375	857	3.9	4.8
Latvia	EFSA_TEST	Other children	189	60	112	264	544	4.4	4.9
Latvia	EFSA_TEST	Adolescents	470	44	78	187	381	4.3	4.9
Latvia	EFSA_TEST	Adults 45-65 years	547	34	63	161	309	4.7	4.9
Belgium	Diet_National_2004	Adolescents	584	37	65	161	345	4.3	5.3
Latvia	EFSA_TEST	Women 18-45 years	383	33	61	153	328	4.6	5.4
Belgium	Diet_National_2004	Adults 45-65 years	554	36	61	168	341	4.6	5.6
Finland	STRIP	Other children	250	70	108	362	620	5.2	5.8
Belgium	Regional_Flanders	Other children	625	81	131	415	813	5.1	6.2
Belgium	Diet_National_2004	Elderly and very elderly	1 230	35	59	183	375	5.2	6.3
Belgium	Diet_National_2004	Women 18-45 years	385	34	57	170	388	5.0	6.8

Country	Survey	Age group	Number of subjects	Middle Bound					
				Scenario 1 (ng/kg bw/day)		Scenario 2 (ng/kg bw/day)		Scenario 2 / Scenario 1	
				Mean	P95	Mean	P95	Mean	P95
Belgium	Regional_Flanders	Toddlers	36	104		551		5.3	
Italy	INRAN_SCAI_2005_06	Toddlers	36	145		312		2.1	
Spain	enKid	Toddlers	17	116		390		3.4	

1685 **Table 11:** Presence of canned food codes in Comprehensive Database per country and survey

Country	Survey	Number of national food codes			Number of FoodEx codes		
		Canned	All	Percentage	Canned	All	Percentage
Germany	National_Nutrition_Survey_II	1,694	22,387	8 %	168	817	21 %
United Kingdom	NDNS	210	3,228	7 %	87	678	13 %
Netherlands	VCP_kids	43	1,194	4 %	39	429	9 %
Sweden	Riksmaten_1997_98	57	1,055	5 %	44	487	9 %
Denmark	Danish_Dietary_Survey	22	315	7 %	21	233	9 %
Spain	AESAN	39	709	6 %	32	366	9 %
Sweden	NFA	67	1,529	4 %	46	528	9 %
Netherlands	DNFCS_2003	177	3,485	5 %	47	554	8 %
Spain	AESAN_FIAB	36	572	6 %	32	381	8 %
Spain	NUT_INK05	24	602	4 %	21	293	7 %
Ireland	NSIFCS	61	1,681	4 %	38	536	7 %
Czech Republic	SISP04	28	502	6 %	19	313	6 %
Cyprus	Childhealth	10	244	4 %	9	179	5 %
Italy	INRAN_SCAI_2005_06	15	1,085	1 %	13	462	3 %
Finland	STRIP	10	917	1 %	9	331	3 %
Bulgaria	NUTRICHILD	12	511	2 %	8	308	3 %
Spain	enKid	6	385	2 %	6	248	2 %
Hungary	National_Repr_Surv	10	536	2 %	8	357	2 %
Greece	Regional_Crete	6	376	2 %	5	257	2 %
Finland	FINDIET_2007	5	1,042	0 %	5	400	1 %
Finland	DIPP	5	925	1 %	5	413	1 %
Latvia	EFSA_TEST	5	1,300	0 %	5	488	1 %
France	INCA2	1	1,251	0 %	1	570	0 %
Belgium	Diet_National_2004	0	2,229	0 %	0	750	0 %
Belgium	Regional_Flanders	0	940	0 %	0	360	0 %
Germany	DONALD_2006_2008	0	3,769	0 %	0	680	0 %

1686

Two scenarios were therefore considered:

- Scenario 1. Only food specifically codified as canned in the dietary survey are assigned the corresponding occurrence level for BPA.

- Scenario 2: At FoodEx level 4, any food which has been codified as canned in at least one survey is always considered to be consumed as canned in all dietary surveys included in the Comprehensive Database. The corresponding average occurrence of BPA in canned products is consequently always assigned to these foods. In order to avoid an artificial overestimate of exposure to BPA, exceptions have been made for products which are consumed in large quantities in many EU countries and would generally not be consumed as canned. For these foods only those effectively codified as canned in the original survey have been assigned with the BPA occurrence in canned food. The exceptions were as follows: apple, beef meat, cow milk (all types), cream (all types), crème fraiche (all types), croissant, mandarins, oranges, potatoes fried, potatoes and potato products, poultry, rice and sour cream (all types).

Presentation of results:

Table 12 presents the minimum, median and maximum values for the average and 95th percentile in each age class, for lower bound, middle bound and upper bound, under scenario 1. Table 13 presents the same results under scenario 2. The highest levels of exposure were estimated for toddlers and other children, up to 857 and 813 ng/kg bw/day respectively for the 95th percentile under the middle bound scenario. Overall, among the population older than 6 months, infants and toddlers presented the highest estimated average (375 ng/kg bw/day) and high (857 ng/kg bw/day) dietary exposure. The CEF Panel considered that this was mainly due to their higher consumption of foods and beverages per kg bw.

Due to a very low percentage of left censored samples, mainly among canned foods, the techniques used to model data under the Limit of Detection (LOD) or quantification (LOQ) had a very small impact on the average concentration in the different food categories and, consequently, on the exposure. On average, exposure estimates calculated by the middle bound technique were 4–30 % (Scenario 1) and 4–12 % (Scenario 2), respectively, higher than those calculated by the lower bound method. Compared to the upper-bound estimates, the middle-bound estimates were 4–19 % (Scenario 1) and 2–8 % (Scenario 2) lower.

Table 10 reports for each survey age group the average and 95th percentile for each scenario. The ratio between scenario 2 and scenario 1 is lowest in countries where many food codes were available for canned products and/or where canned products are largely consumed. It is the case for UK men and women from 18 to 45 years old where the ratio is 1.9 and 2.2 at the average and 1.7 and 2.1 at the 95th percentile, respectively. The highest difference has been noted in Belgian toddlers with a ratio equal to 5.0 and 6.8 for the average and the 95th percentile, respectively.

Table 14 presents the number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and scenario. Under scenario 1, the percentage contribution to BPA from non-canned foods was predominant (but less than 50 %) in the large majority of dietary survey. Under this scenario, only for one survey (related to males from 18 to 45 years old) canned foods resulted to contribute between 50 and 75 % of average BPA exposure. Under scenario 2, canned products dominated in all surveys with the percentage contribution to BPA from non-canned foods mainly ranging between 10-25 %. Canned foods contributed up to more than 90 %, this is the case of one dietary survey among toddlers: “Fish and other seafood”

The number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. non-canned), FoodEx level 1 food category and scenario is reported in Table 10. Under scenario 1, non-canned “meat and meat products” turned out to be a major contributor to BPA average exposure in the large majority of countries and age classes. “Vegetables and vegetable

products" was the only canned food category that contributed up to 25-50 % in some of the population groups. "Meat and meat products" was the major contributor among the non-canned food categories also under scenario 2 but never exceeded 10-25 % of total exposure. On the other hand, the canned versions for "vegetables and vegetable products", "meat and meat products" and "composite food" were the major sources of average BPA exposure.

Under scenario 2, dietary exposure in women of childbearing age was slightly higher (132 and 388 ng/kg bw/day for average and high exposure, respectively) than that to men of the same age (126 and 355 ng/kg bw/day for average and high exposure, respectively). This may be due to different food items consumed by women as reported in the individual surveys.

1743 **Table 12:** Dietary exposure estimates for Scenario 1

Age class	Number of surveys	Lower Bound (ng/kg bw/day)					
		Average			95th percentile		
		Minimum	Median	Maximum	Minimum	Median	Maximum
Toddlers	7 (4)	55	92	131	94	178	241
Other children	15	51	73	118	78	135	207
Teenagers	12	34	51	67	60	89	112
Women 18-45 years	15	31	38	58	51	69	119
Men 18-45 years	15	34	45	55	56	81	103
Other adults 45-65 years	14	30	39	50	52	71	88
Elderly and very elderly	6	27	33	43	47	57	68

Age class	Number of surveys	Middle Bound (ng/kg bw/day)					
		Average			95th percentile		
		Minimum	Median	Maximum	Minimum	Median	Maximum
Toddlers	7 (4)	72	111	145	108	203	253
Other children	15	57	81	127	86	147	223
Teenagers	12	37	55	70	65	95	121
Women 18-45 years	15	33	41	61	55	73	126
Men 18-45 years	15	37	49	59	59	85	109
Other adults 45-65 years	14	33	42	52	55	75	94
Elderly and very elderly	6	29	35	47	51	60	74

Age class	Number of surveys	Upper Bound (ng/kg bw/day)					
		Average			95th percentile		
		Minimum	Median	Maximum	Minimum	Median	Maximum
Toddlers	7 (4)	88	126	159	135	223	267
Other children	15	63	90	135	94	157	235
Teenagers	12	41	59	74	70	100	127
Women 18-45 years	15	35	44	64	58	78	132
Men 18-45 years	15	39	53	64	63	90	115
Other adults 45-65 years	14	35	45	55	58	79	100
Elderly and very elderly	6	31	38	50	54	64	78

1744

1745

1746 **Table 13:** Dietary exposure estimates for Scenario 2

Lower Bound (ng/kg bw/day)							
Age class	**Number of surveys**	**Average**			**95th percentile**		
		Minimum	**Median**	**Maximum**	**Minimum**	**Median**	**Maximum**
Toddlers	7 (4)	212	356	516	445	721	817
Other children	15	184	275	393	337	525	766
Teenagers	12	114	150	190	237	288	357
Women 18-45 years	15	91	125	172	179	225	363
Men 18-45 years	15	94	118	164	170	204	314
Other adults 45-65 years	14	95	118	158	172	213	321
Elderly and very elderly	6	90	110	172	169	194	352

Middle Bound (ng/kg bw/day)							
Age class	**Number of surveys**	**Average**			**95th percentile**		
		Minimum	**Median**	**Maximum**	**Minimum**	**Median**	**Maximum**
Toddlers	7 (4)	235	375	551	487	767	857
Other children	15	198	290	415	363	550	813
Teenagers	12	121	159	201	248	304	381
Women 18-45 years	15	97	132	182	191	235	388
Men 18-45 years	15	101	126	175	182	218	335
Other adults 45-65 years	14	102	126	168	186	224	341
Elderly and very elderly	6	97	116	183	179	206	375

Upper Bound (ng/kg bw/day)							
Age class	**Number of surveys**	**Average**			**95th percentile**		
		Minimum	**Median**	**Maximum**	**Minimum**	**Median**	**Maximum**
Toddlers	7 (4)	257	395	587	504	812	886
Other children	15	212	306	440	392	584	868
Teenagers	12	128	168	212	259	320	403
Women 18-45 years	15	104	139	192	200	244	413
Men 18-45 years	15	108	134	186	193	230	360
Other adults 45-65 years	14	109	133	179	198	235	364
Elderly and very elderly	6	103	122	195	192	216	396

1747

1748

1749 **Table 14:** Percentage of average dietary exposure according to the type of packaging and scenario

Age group	Packaging type	Total number of surveys	Number of dietary surveys															
			Scenario 1 % average BPA contribution (Middle Bound)								Scenario 2 % average BPA contribution (Middle Bound)							
			< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	75 – 90 %	>90 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	75 – 90 %	>90 %
Toddlers	Canned	7	3	0	1	1	2	0	0	0	0	0	0	0	0	1	5	1
	Not canned		0	0	0	0	0	2	1	4	0	0	1	5	1	0	0	0
Other children	Canned	15	3	0	2	3	4	0	0	0	0	0	0	0	0	1	14	0
	Not canned		0	0	0	0	0	4	3	8	0	0	0	14	1	0	0	0
Teenagers	Canned	12	4	0	1	6	2	0	0	0	0	0	0	0	0	1	11	0
	Not canned		0	0	0	0	0	2	6	4	0	0	0	11	1	0	0	0
Women 18-45 years	Canned	15	4	0	2	5	4	0	0	0	0	0	0	0	0	1	14	0
	Not canned		0	0	0	0	0	4	5	6	0	0	0	14	1	0	0	0
Men 18-45 years	Canned	15	4	0	1	6	3	1	0	0	0	0	0	0	0	3	12	0
	Not canned		0	0	0	0	1	3	6	5	0	0	0	12	3	0	0	0
Other adults 45-65 years	Canned	14	4	0	2	5	3	0	0	0	0	0	0	0	0	1	13	0
	Not canned		0	0	0	0	0	3	5	6	0	0	0	13	1	0	0	0
Elderly and very elderly	Canned	7	3	0	0	2	2	0	0	0	0	0	0	0	0	0	7	0
	Not canned		0	0	0	0	0	2	2	3	0	0	0	7	0	0	0	0

1750

4.6.3. Exposure from non-dietary sources

While exposure to food mainly involves oral exposure, for non-food sources also the exposure routes inhalation and dermal absorption have to be considered. Inhalation is a relevant route for the sources outdoor and indoor air. For dust both ingestion and inhalation can occur. Dermal exposure has to be considered for BPA present on the surface of consumer products such as thermal paper or through cosmetics. All the equations used to calculate exposure form the non-food sources are given in Appendix IV.

In a first step, all possible non-food sources of exposure have been assessed with regard to their concentrations, migration and transfer potential for BPA (see chapter 4.3.6). For the quantitative assessment the most important source/route combinations have been selected that most probably will contribute to daily exposure. They are listed in Table 15 and the relevant population groups are given for each source/route combination.

Table 15: Overview of sources, population groups exposed and routes considered in the quantitative assessment

Exposure routes	Sources and population groups exposed				
	Air	**Dust**	**Thermal paper**	**Toys**	**Cosmetics**
Inhalation	all ages	all ages	n/a	n/a	n/a
Ingestion	n/a	all ages	all ages excluded infants	infants and toddlers	n/a
Dermal Absorption	n/a	n/a	all ages excluded infants and toddlers	n/a	all ages

n/a = not relevant for this route for all age groups

The following sources have not been assessed quantitatively: surface water ingestion, dermal exposure to water (both surface and tap water; e.g. during bathing and showering), cigarette filters (ingestion, inhalation) and medical devices other than dental materials, for the following reasons: Surface water ingestion while swimming can be regarded as minor both on an acute and chronic level compared to other sources such as drinking water. Also dermal exposure to surface water is negligible compared to dermal exposure to e.g. thermal paper. Cigarette filters have been suspected to be a source of exposure (Braun et al., 2011), but no evidence could be generated that BPA is actually used in cigarette filters. Medical devices are dealt with by SCENIHR in a separate opinion and do not represent a chronic exposure pathway for the whole population. One exception is dental materials that are commonly used in dental surgery both for children and adults, either as dental fillers (adults) or as fissure sealants (children).

Ingestion

The non-food sources evaluated for ingestion include dust, toys and other articles intended to be mouthed (infants, toddlers), dental materials (all age groups except infants and toddlers) and transfer from hands to food after touching of thermal paper by the parent. For ingestion, an absorption fraction of 1 was used.

Dust

For the average and the high scenario, the average BPA concentrations (C_{dust}) derived in Chapter 4.3.6 were multiplied with average and high dust ingestion rates (q_{dust}) according to Trudel et al., 2008 (see Table 16), respectively, and divided by age specific bodyweights bw as described above. For all calculations the same

absorption rate ($r_{absorption}$) of 1 for ingestion was used. Newborns (infants, 0-5 days) were assumed not to be exposed to dust via ingestion, but only to fine dust in air (included in calculation for air). Dust ingestion rates are commonly derived from soil ingestion rates as a proxy and thus are considered quite uncertain (Trudel et al., 2008). They are assumed to comprise both inhalation and ingestion as inhaled particles can be cleared from the thoracic tract and subsequently be ingested. Inhalation and ingestion thus cannot be separated.

The following equation was used to derive the exposure estimates:

$$E_{dust} = \frac{C_{dust} \cdot q_{dust}}{bw} \cdot r_{absorption}$$

Table 16: Values for dust ingestion (mg/day) according to Trudel et al. (2008) and estimates for exposure from dust (ng/kg bw/day)

	Average scenario		High scenario	
Age group	**q_{dust}** (mg/d)	**E_{dust}** (ng/kg bw/d)	**q_{dust}** mg/d)	**E_{dust}** (ng/kg bw/d)
infants	9.0	2.63	106	31.0
toddlers	9.0	1.10	106	12.9
children	26	1.27	95	4.63
teenagers	5.2	0.17	138	4.58
adults	5.2	0.11	138	2.88

The derived exposure values of 0.11 ng/kg bw/d in adults to 2.63 ng/kg bw/d in infants are low for the average scenario. In the high scenario the exposure ranged from 2.9 ng/kg bw/d (adults) to 31 ng/kg bw/d (infants). It should be noted, that the high scenario is not intended to reflect situations in houses with high BPA concentrations in dust, but addresses only variation due to behavioural aspects.

Toys (rattles) and pacifiers with PC shields

Data for migration of BPA from rattles and pacifiers with PC shields into saliva was used for this assessment (see chapter 4.3.6). The amount of substance migrating from pacifiers was adjusted to 24 h by linear extrapolation from the incubation time of 7.75 h. For rattles no extrapolation was needed, since the incubation time was 24 h. The resulting amount of substance that leached over 24 h from a product ($q_{product}$) was used in the equation below: 141.2 ng for rattles and 987.1 ng for pacifiers. Then, the migration over 24 h for the average scenario was corrected by average or high daily sucking times, yielding a fraction of the day that the rattle or pacifier is sucked (f_{time}). For the average exposure from plastic toys sucking times for users and non users as reported by Juberg et al. (2001) were used and for the high exposure P75 daily sucking times reported by Bremmer and van Veen (2002) (see Table 17). To calculate exposure from pacifiers with PC shields for toddlers, the P75 was directly taken Juberg et al. (2001).

In the migration experiments the toys were completely submersed. Therefore, in order to account for realistic exposure situations, it was further assumed that for toys (rattles) only 50 % of the toy surface is sucked ($f_{surface}$: 0.5). For pacifiers only the shield and ring are made of PC. Therefore, the available surface was assumed to be 25 % ($f_{surface}$: 0.25; only one side and only parts of the shield that are near to the mouth, approach according to Lassen et

al., 2011). The following equation was used to assess exposure to toys and pacifiers with PC shields:

$$E_{toy} = \frac{q_{product} * f_{time} * f_{surface}}{bw} * r_{absorption}$$

Table 17: Values for factors dealing with sucking times $_{ftime}$ and estimates for exposure from rattles/pacifiers with PC shields

	Average scenario			High scenario		
Age group	**f_{time} (d^{-1})**	**Reference**	**E_{toy} (ng/kg bw/d)**	**f_{time} (d^{-1})**	**Reference**	**E_{toy} (ng/kg bw/d)**
Toy, infants	0.012	Juberg et al. 2001	0.33	0.04	Bremmer and van Veen, 2002	1.24
Toy, toddlers	0.001	Juberg et al. 2001	0.02	0.04	Bremmer and van Veen, 2002	0.51
Pacifier, infants	0.15	Juberg et al. 2001	7.57	0.20	Bremmer and van Veen, 2002	9.77
Pacifier, toddlers	0.32	Juberg et al. 2001	6.60	1.49	Juberg et al. 2001	10.0

Using this approach, exposure values of 0.33 and 0.02 ng/kg bw/d for the average and 1.24 and 0.51 ng/kg bw/d for the high scenario for infants and toddlers' exposure to rattles (as a proxy for PC mouthing toys) were derived.

For pacifiers with PC shields due to longer sucking times higher exposure was calculated with 7.57 and 6.60 ng/kg bw/d for the average scenario infants and toddlers, and 9.77 and 10.0 ng/kg bw/d for the high exposure scenario. It has however to be mentioned that only 10-20 % of the shields of pacifiers may be made of PC, so that this exposure value is valid only for a specific consumer group.

Dental materials

For the dental materials exposure scenarios the procedure described in von Goetz et al. (2010) was used. Three different scenarios were assessed. One for children (target group: children in the age of 8-12 year) who are receiving dental sealants which are applied to protect their new (adult) molars. Another scenario is for teenagers (age 12-16) who receive lingual bonded retainers, and a third scenario describes a dental restoration (filling of a molar) in adults. All these scenarios refer to acute exposure events. Therefore, it is assumed that a combination of these scenarios is not needed.

Concentrations in saliva after transfer were used together with the amount of swallowed saliva per day (q_{saliva}: adults: 720 ml/day (Rudney et al., 1995); children 500 ml/day (Watanabe et al., 1995). It should be noted that for

1834 the calculation of the average values, a baseline value of 0.5 ng BPA/ml saliva (C_{saliva}) was used with the following
1835 equation:

1836 $$E_{dental} = \frac{C_{saliva} * q_{saliva}}{bw} * r_{absorption}$$

1837 Since the baseline level is very low (the level before treatment is the same as about 24h after treatment), it could be
1838 argued whether this value really represents exposure to dental material. Therefore, exposure to dental materials was
1839 not included in the total exposure calculation.

1840 This is in line with other risk assessments of BPA that have so far generally concluded that exposure from dental
1841 materials does not contribute significantly to total exposure (ECB, 2008; EFSA, 2006a; NTP-CERHR, 2008). This
1842 is also concluded in a recently published report by the Swedish National Board of Health and Welfare (ISBN: 978-
1843 91-87169-48-9, June 2012) addressing "Bisphenol A in dental materials". This report summarises research on *in*
1844 *vitro* and *in vivo* studies related to BPA from dental materials, and concludes that there is a possibility of low-dose
1845 exposure to BPA from dental materials, either as a contaminant (very low amounts) or from degradation of Bis-
1846 DMA.

1847 *Thermal paper: transfer to food*

1848 After touching thermal paper, e.g. during shopping, BPA on the fingers can be transferred to food and consequently
1849 be ingested, either by the person itself or a child. This may happen e.g. if a parent shops, gets a thermal paper
1850 receipt, and directly afterwards eats a shopped fruit or gives a piece of fruit to a toddler or child. In Biedermann et
1851 al. (2010) the transfer of BPA from contaminated hands back to dry paper was investigated and no BPA was
1852 detected (<LOD). However, since the same study revealed that transfer to wet and greasy fingers was much higher
1853 than to dry fingers, transfer to more lipophilic and/or wet surfaces, such as to food, cannot be compared to dry
1854 paper.

1855 No experimental data are available for transfer to food after touching thermal paper. In order to investigate this
1856 pathway, a transfer of 1 % from skin (f_{trans}) to food was hypothesised. It was assumed that only a fraction of 0.7
1857 (corresponding to 70 % absorption) is available for transfer (f_{avail}), because it was shown that BPA is taken up by
1858 the skin with a fraction of around 0.3 (corresponding to 30 % absorption, see Chapter 4.3.6). These fractions were
1859 combined with the assumption that 2, 2 and 4 transfer events ($q_{handling}$) for toddlers, children and adults (adults: e.g.
1860 1 shopping, 1 canteen meal or bus ticket), respectively occur per week (2/7, 2/7 and 4/7 per day) and that three
1861 fingers (n_{finger}) have touched the thermal paper. For the transferred amount of BPA from thermal paper to finger tips
1862 (a_{finger}) the mean value given by Lassen et al. (2011) was used, which is 1.4 µg/finger tip. The following equation
1863 was used to calculate exposure:

1864 $$E_{tp-food} = \frac{a_{finger} * n_{finger} * f_{avail} * f_{trans} * q_{handling}}{bw} * r_{absorption}$$

1865 This calculation yields exposures of 0.7 (toddlers), 0.3 (children), and 0.3 ng/kg bw/d (adults). Since there is no
1866 data available on the frequency of such unfavorable events, nor on transfer rates, this exposure estimate was not
1867 included in the calculation of exposure for the general public and specific consumer groups.

1868 Inhalation

1869 BPA concentrations in outdoor and indoor air (C_{air}) are low, with indoor air levels being slightly higher (see
1870 Chapter 4.3.6). For the calculation of an average value therefore the assumption was made that people spend 100 %
1871 of their time indoors. Average and high intake rates of air (q_{air}) are taken from Trudel et al. (2008) (see Table 18).

As absorption fraction ($r_{absorption}$) 1 was used (see chapter 4.5.1) and the following equation was used for the assessment:

$$E_{air} = \frac{C_{air} * q_{air}}{bw} * r_{absorption}$$

Table 18: Values for air intake rates q_{air} according to Trudel et al. (2008) and estimates for exposure from inhalation

	Average exposure		High exposure	
Age group	**q_{air}** (m^3/day)	**E_{air}** (ng/kg bw/day)	**q_{air}** (m^3/day)	**E_{air}** (ng/kg bw/day)
infants	12.0	2.40	28.8	5.76
toddlers	16.8	1.40	40.8	3.40
children	21.6	0.72	55.2	1.84
teenagers	50.4	1.15	91.2	2.07
adults	50.4	0.72	91.2	1.30

The average exposure values range from 0.72 (adults) to 2.4 ng/kg bw/day (infants). High exposure levels range from 1.3 (adults) to 5.76 ng/kg bw/d (infants).

Dermal

Thermal paper

In this exposure assessment it was assumed that children, teenagers and adults come into contact with thermal paper from shopping/canteen receipts, credit card receipts, bus tickets or parking tickets. The number of handling events qhandling for teenagers and adults for the high exposure was taken from a use study by Lassen et al. (2011) (4.6 handlings per day). Handling events for the average exposure were assumed as 1 per day for teenagers and adults, deduced from the credit card receipts handled by Danish consumers above 12 years (259 per year) from Lassen et al. (2011). Children were assumed to come into contact with thermal paper 0.5 times a day in the average exposure and maximally 2 times a day on a chronic basis.

The paper is handled mainly by the finger tips of three fingers (nfinger) of one (average exposure) or two hands (high exposure). Each finger has a BPA load available for absorption (afinger) of 1.4 µg/handling (Lassen et al., 2011). Thermal paper is covered with BPA only on one side, but since consumers handle the receipts usually by folding it away (with touching on both sides) and since the exposure studies present a average of all fingers holding the receipt, this fact was not considered separately, but assumed to be contained in the amount available for absorption. A dermal absorption fraction, $r_{absorption}$, of 0.3 (corresponding to 30 % absorption, see Chapter 4.3.6) was used.

The following equation was used for the assessment:

$$E_{tp-dermal} = \frac{a_{finger} * n_{finger} * q_{handling}}{bw} r_{absorption}$$

The estimates of exposure from dermal contact with thermal paper are summarised in Table 19.

Table 19: Values for qhandling and estimates for exposure from dermal contact with thermal paper

	Average exposure		High exposure	
Age group	**$q_{handling}$** (1/day)	**$E_{tp\text{-}dermal}$** (ng/kg bw/day)	**$q_{handling}$** (1/day)	**$E_{tp\text{-}dermal}$** (ng/kg bw/day)
children	0.5	20.6	2.0	165
teenagers	1.0	28.1	4.6	259
adults	1.0	17.7	4.6	163

From these average assumptions the exposure of 20.6, 28.1 and 17.7 ng/kg bw/day was derived for children, teenagers and adults, respectively. For the high exposure, exposure ranges from 259 (teenagers) to 163 ng/kg bw/d (adults).

Cosmetics

Exposure to cosmetics in the form of body lotion is possible for all age groups. Medians and P95 for amounts of body lotion used by adults ($q_{cosmetics}$) were taken from Hall (2007). For infants, toddlers, children and teenagers the amount used by adults was corrected by a factor for the different body surfaces (see Table 20). Mean body surfaces for adults of 1.85 m^2 were taken from Tikuisis et al. (2001) and for the other age groups from van Engelen and Prud'homme de Lodder (2007) (see Table 20). Dermal absorption was assumed to be a fraction of 0.6 (corresponding to 60 % absorption, see chapter 4.3.6). The retention factor f_{ret} for leave-on cosmetics is 1. A retention factor characterises a cosmetic regarding the fraction for substance staying on the skin (e.g. for rinse-off cosmetics it is 0.1).

The exposure was calculated with the following equation:

$$E_{cosmetics} = \frac{C_{cosmetics} * q_{cosmetics} * f_{ret}}{bw} * r_{absorption}$$

Table 20: Body surfaces, derived parameter values for qcosmetics and estimates for exposure from cosmetics

		Average exposure		High exposure	
Age group	**body surface (m^2)**	**$q_{cosmetics}$ (g/d)**	**$E_{cosmetics}$ (ng/kg bw/d)**	**$q_{cosmetics}$ (g/d)**	**$E_{cosmetics}$ (ng/kg bw/d)**
infants	0.31	0.77	2.87	1.51	5.61
toddlers	0.44	1.09	1.70	2.14	3.32
children	0.84	2.09	1.29	4.09	2.53
teenagers	1.4	3.48	1.47	6.81	2.88
adult	1.85	4.60	1.22	9.00	2.39

For the average exposure the exposure ranges from 1.2 (adults) to 2.9 ng/kg bw/d (infants). High exposure ranges from 2.4 (adults) to 5.6 ng/kg bw/d (infants).

Assessment of non-food average and high exposure

An average and a high scenario were calculated for all sources. For the average scenario, an attempt was made to choose average values for all parameters, including parameters describing frequency of use. For the high scenario, the same average parameters were used for absorption rates and occurrence data, but in line with the methodology used to assess exposure from food, the frequency of use parameters were modified to account approximately for a 95th percentile of the population. If not mentioned otherwise, the arithmetic mean was used for each parameter, but in some cases only medians and percentiles were available. In order to follow a similar approach to that of exposure from food, behavioural parameters were derived considering both users and non users in the general population.The estimates for average and high exposure are included in Table 21.

For calculations for specific population groups (e.g. users of pacifiers with PC shields), behavioural data were only taken from the group of users (see Table 22).

Exposure estimates were given per bodyweight. For the different age groups, different default bodyweights were used. For infants the default bodyweight of 5 kg for 1-3 months old infants was used (EFSA Scientific Committee, 2012). For toddlers the default bodyweight of 12 kg for 1-3 years old children was used (EFSA Scientific Committee, 2012). For children and teenagers default values of 30 kg for 9 year old children and of 44 kg for 15 year-old teenagers were used (van Engelen and Prud'homme de Lodder, 2007). For adults, the default bodyweight of 70 kg was used (EFSA Scientific Committee, 2012).

1941 **Table 21:** Average and high exposure for non-food sources

Average scenario	Exposure (ng/kg bw/day)					
	Infants	Toddlers	Children	Teenagers	Adults	Elderly/ Very elderly
bodyweights	5	12	30	44	70	70
Age (years)	<1	1-3	3-10	11-17	18-65	>65
Ingestion						
Dust	2.63	1.10	1.27	0.17	0.11	0.11
Toys, rattles	0.33	0.02	n/a	n/a	n/a	n/a
Inhalation						
Air	2.40	1.40	0.72	1.15	0.72	0.72
Dermal						
thermal paper	n/a	n/a	20.6	28.1	17.7	17.7
cosmetics, body lotion	2.87	1.70	1.29	1.47	1.22	1.22
High scenario	**Exposure (ng/kg bw/day)**					
	Infants	Toddlers	Children	Teenagers	Adults	Elderly Very elderly
bodyweights	5	12		60	70	70
Age (Years)	<1	1-3	3-10	11-17	18-65	>65
Ingestion						
Dust	31.0	12.9	4.63	4.58	2.88	2.88
Toys, rattles	1.24	0.51	n/a	n/a	n/a	n/a
thermal paper: transfer to food	n.a.	11.8	4.70	6.41	4.03	4.03
Inhalation						
Air	5.76	3.40	1.84	2.07	1.30	1.30
Dermal						
thermal paper	n/a	n/a	165	259	163	163
cosmetics, body lotion	5.61	3.32	2.53	2.88	2.39	2.39

1942 n/a = not relevant for this age group

1943 **Table 22:** Specific population groups, non-food sources

1944

	Exposure (ng/kg bw/day)						
	Infants	**Toddlers**	**Children**	**Teenagers**	**Adults**	**Elderly**	**Very elderly**
bodyweights	5	12	30	44	70	70	70
Average scenario, pacifiers with PC shields	7.57	6.60	n/a	n/a	n/a	n/a	n/a
High scenario, pacifiers with PC shields	9.77	6.60	n/a	n/a	n/a	n/a	n/a

1945 n/a = not relevant for this age group

4.7. Total exposure

In this chapter, total exposure to BPA was estimated by using modelling calculations. Exposure modelling involved the assessment of chronic exposure (absorbed dose) to BPA through different sources (diet, thermal paper, air, dust, toys, cosmetics, dental sealants) and routes of exposure (oral, inhalation and dermal) in the EU population. Analytical/experimental BPA concentrations were combined with food consumption (including human milk) to estimate dietary exposure and concentration data in and from non-food sources with behaviour patterns to estimate non-dietary exposure. Then, total average exposure was calculated by adding up average exposure from all dietary and non-dietary sources. Total high exposure was calculated by adding up high levels of exposure from the two highest sources and average exposure levels from all other sources.

These modelled calculations aimed to assess the total daily amount of BPA absorbed by the body by any route. The absorption factors considered in these calculations were 1 for oral, 1 for inhalation and 0.3 for dermal. Modelling allows estimation of exposure from all the sources of exposure which could be identified and quantified individually. In order to quantify the relative impact of each source, the assumptions made in the exposure estimates were aimed at obtaining a similar degree of conservativeness among the different sources.

In all population groups, diet was always one of the two highest sources of high exposures. The other highest sources of exposure were air in the first days of life, dust in infants and toddlers and dermal exposure from thermal paper in all other age classes. Results are presented in Table 23.

The percentage contribution of each source to total average exposure is presented in Table 24. Exposure through the diet was the main source of average exposure to BPA in all population groups (from 78 to 99 %), followed, with the exception of children aged less than 3 years old, by dermal exposure through thermal paper (from 7 to 15 %). The CEF Panel is aware of an ongoing study on BPA pharmacokinetic and dermal exposure in cashiers sponsored by the National Institute of Environmental Health Sciences (NIEHS) under the National Toxicology Program (NTP). The results of this study will be considered by the CEF Panel as they will be an additional source of information regarding the absorption of BPA from thermal paper.

Exposure to BPA from further sources was assessed in specific populations groups or in consumers with specific consumption patterns. The aim was to identify possible additional sources of exposure to BPA which could lead to levels of exposure significantly higher than those estimated for the general population. Average and high exposure from these further sources are presented in Table 25. In most cases, exposure from these further sources was less than 20 % of the estimated high exposure for the age class. In a few cases, exposure from these further sources was higher. It was the case for infants fed using old PC baby bottles and infants living in buildings with old water pipes repaired with epoxy resins and fed with formula reconstituted with tap water.

1978 **Table 23:** Exposure to BPA from all sources in the general population (ng/kg bw/day)

		Infants 0-6 months (breastfed)		Infants 0-6 months (formula fed)	Infants	Toddlers	Other children	Teenagers	Women	Men	Other adults	Elderly and very elderly
	1-5 days	6 days - 3 months	4 - 6 months	0- 6 months	6-12 months	1-3 years	3-10 years	10-18 years	18-45 years	18-45 years	45-65 years	65 years and over
Ingestion:												
Dust (average)		2.6	2.6	2.6	2.6	1.1	1.3	0.2	0.1	0.1	0.1	0.1
Dust (high)		31.0	31.0	31.0	31.0	12.9	4.6	4.6	2.9	2.9	2.9	2.9
Toys (average)		0.3	0.3	0.3	0.3	0.02						
Toys (high)		1.2	1.2	1.2	1.2	0.5						
Dietary exposure from food and beverages (average)	225	135	119	30	375	375	290	159	132	126	126	116
Dietary exposure from food and beverages (high)	495	390	343	80	857	857	813	381	388	335	341	375
Sum of all ingestion sources (average)	**225**	**138**	**122**	**33**	**378**	**376**	**292**	**159**	**132**	**127**	**126**	**116**
Inhalation:												
Air (average)	2.4	2.4	2.4	2.4	2.4	1.4	0.7	1.1	0.7	0.7	0.7	0.7
Air (high)	5.8	5.8	5.8	5.8	5.8	3.4	1.8	2.1	1.3	1.3	1.3	1.3
Sum of all inhalation sources (average)	**2.4**	**2.4**	**2.4**	**2.4**	**2.4**	**1.4**	**0.7**	**1.1**	**0.7**	**0.7**	**0.7**	**0.7**
Dermal:												
Thermal paper (average)							21	28	18	18	18	18
Thermal paper (high)							165	259	163	163	163	163
Cosmetics (average)		2.9	2.9	2.9	2.9	1.7	1.3	1.5	1.2	1.2	1.2	1.2
Cosmetics (high)		5.6	5.6	5.6	5.6	3.3	2.5	2.9	2.4	2.4	2.4	2.4
Sum of all dermal sources (average)		**3**	**3**	**3**	**3**	**2**	**22**	**30**	**19**	**19**	**19**	**19**
Total exposure from all sources (average)	**228**	**143**	**127**	**38**	**383**	**379**	**314**	**190**	**152**	**146**	**145**	**136**
Total exposure (high) calculated as two highest plus sum of the average of all other sources	**501**	**427**	**380**	**117**	**894**	**873**	**981**	**642**	**553**	**500**	**506**	**540**

1979

1980 **Table 24:** Main sources of exposure to BPA from all sources in the general population (% of average)

	Infants 0-6 months (breastfed)			Infants 0-6 months (formula fed)	Infants	Toddlers	Other children	Teenagers	Women	Men	Other adults	Elderly and very elderly
	1-5 days	6 days - 3 months	4 - 6 months	0- 6 months	6-12 months	1-3 years	3-10 years	10-18 years	18-45 years	18-45 years	45-65 years	65 years and over
Dust (ingestion)	0.0	1.8	2.1	6.9	0.7	0.3	0.4	0.1	0.1	0.1	0.1	0.1
Toys (ingestion)	0.1	0.2	0.3	0.9	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Dietary exposure from food and beverages (ingestion)	98.8	94.3	93.5	78.5	97.9	98.9	92.4	83.7	87.0	86.5	86.4	85.5
Air (inhalation)	1.1	1.7	1.9	6.3	0.6	0.4	0.2	0.6	0.5	0.5	0.5	0.5
Thermal paper (dermal)	0.0	0.0	0.0	0.0	0.0	0.0	6.6	14.8	11.7	12.1	12.2	13.0
Cosmetics (dermal)	0.0	2.0	2.3	7.5	0.7	0.4	0.4	0.8	0.8	0.8	0.8	0.9

1981

1982 **Table 25:** Exposure from further sources in specific population groups (ng/kg bw/day)

	Infants 0-6 months (breastfed)			Infants 0-6 months (formula fed)	Infants	Toddlers	Other children	Teenagers	Women	Men	Other adults	Elderly and very elderly
	1-5 days	6 days - 3 months	4 - 6 months	0-6 months	6-12 months	1-3 years	3-10 years	10-18 years	18-45 years	18-45 years	45-65 years	65 years and over
Residents of buildings with old water pipes repaired with epoxy resins (average)				15	2.7	2.7	1.9	1.1	1.0	0.8	0.9	1.1
Residents of buildings with old water pipes repaired with epoxy resins (high)				165	29	29	21	12	11	8	9	12
Users of PC tableware (average)					6	6	4	2	2	2	2	2
Users of PC tableware (high)					14	14	9	5	5	4	5	4
Users of PC kettles (average)				16.5	0.04	0.04	0.05	0.11	0.4	0.2	0.2	0.3
Users of PC kettles (high)					2.1	2.1	1.8	1.7	2.8	2.6	3.2	3.0
Consumers of water from PC filters (average)				6	1.1	1.1	0.8	0.4	0.4	0.3	0.3	0.4
Consumers of water from PC filters (high)					3.8	3.8	2.8	1.6	1.6	1.4	1.3	1.1
Consumers of water from water coolers with PC reservoirs (average)					22	22	16	9	8	6	7	9
Users of PC baby pacifiers (average)	8	8	8	8	8	7						
Users of PC baby pacifiers (high)	10	10	10	10	10	10						
Infants fed with formula in old PC baby bottles (average)				135								
Infants fed with formula in old PC baby bottles (high)				684								
Breastfed infants consuming herbal tea prepared with water warmed in a PC kettle (average)	2	2	2									
Breastfed infants consuming herbal tea prepared with water warmed in a PC kettle (high)	4	4	4									
Users of cookware (average)					19	19	14	8	6	7	6	6
Users of cookware (high)					46	46	28	16	15	14	15	14

4.8. Biomonitoring

4.8.1. General introduction

Biomonitoring is a direct approach to estimate the human exposure from all sources and *via* all uptake routes (Angerer et al., 2007; Hengstler et al., 2011). The approach is called direct because it can be directly related to the dose which has actually entered the systemic circulation. A number of sensitive analytical methods have been developed to measure low concentrations including trace amounts of BPA in biological samples such as urine and blood (Dekant and Völkel, 2008; WHO, 2011b; Asimakopoulos et al., 2012), the by far most approved biological matrices for human biomonitoring (Angerer et al., 2007). Yet the detection and quantification of BPA-related biomarkers in these matrices is per se not sufficient to arrive at reliable and valid estimates of exposure. What is additionally required to interpret BPA biomonitoring data and to translate these data into daily exposure estimates is a detailed understanding of the potential analytical/methodological pitfalls (see Appendix I) and of the toxicokinetics of BPA.

As a non persistent chemical with an elimination half-life of a few hours, BPA is rapidly removed from circulation *via* conjugation and subsequent renal excretion (Völkel et al., 2002; Doerge et al., 2010a). Toxicokinetic studies with oral administration of stable isotope-labelled (deuterated) BPA in humans have shown that BPA is almost completely excreted in urine in the conjugated form and that the elimination process is essentially complete within 24 h after exposure (Völkel et al., 2002; Völkel et al., 2008, Teeguarden et al, 2011). Urine is therefore the matrix of choice for biomonitoring, and the urinary concentration of total (unconjugated plus conjugated) BPA is the biomarker of choice to estimate BPA exposure (Calafat et al., 2008). Information on the presence and concentration of unconjugated and total BPA in serum is useful, and will additionally be compiled in this chapter, in order to inform toxicological risk assessment. However, given the exposure in the ng/kg bw range, the high first-pass metabolism in the liver, and the elimination characteristics of BPA, low serum concentrations of unconjugated and total BPA are to be expected. In addition, it has been shown that generally less than 1 % of total serum BPA is in the unconjugated form after oral administration (Doerge et al., 2010a; Taylor et al., 2011). Hence, the detection of unconjugated serum BPA becomes an analytical challenge that is additionally complicated by contamination and the instability of BPA conjugates (see Appendix I). Also compiled in this chapter is information on unconjugated and total BPA in human milk to enable the estimation of BPA exposure in breastfed infants.

4.8.2. Biomonitoring studies on urinary levels

Methodological aspects

Data on urinary levels of total BPA in humans were retrieved from scientific journals, from official websites of national health surveys (e.g. NHANES, CHMS, German Federal Environment Agency, Flemish human biomonitoring program), and from yet unpublished sources (e.g. DEMOCOPHES). Quality criteria for urinary BPA data were assessed, and a literature quality table was developed for the methodical aspects and study aspects. The quality of each study was assessed on the basis of the criteria given in Appendix I.

As a general rule, only data published from 2006 onwards were considered. Since then, substantial methodological improvements have been achieved both in terms of sensitivity and specificity by using MS-based analytical techniques. Moreover, efforts have been improved / implemented to preserve sample integrity and to reduce external contamination; more recent data should therefore be of higher quality than older data. Furthermore, the more recent data will provide an up-to-date indication of the current exposure to BPA.

A specific inclusion criterion for data on urinary BPA is that the biomonitoring studies have been performed in the European region. Only these data are included for estimating daily exposure to BPA for different age groups of the European populations. Data on urinary BPA levels in non-European populations are, however, presented for comparative purposes.

To compare the distribution characteristics of the urinary concentration of total BPA between the different studies, box-percentile plots (Esty and Banfield, 2003) comprising the 5th, 12.5th, 25th, 37.5th, 50th, 62.5th, 75th, 87.5th and 95th percentiles are used. In contrast to the practice in the food area (s. Chapter 4.3.5), the geometric mean (GM) rather than arithmetic mean (AM) was chosen as a measure of central tendency of the distribution for several reasons. Firstly, the urinary concentration of total BPA is approximately log-normally distributed (Figure 1), so that the GM rather than the AM is the most appropriate measure of central tendency. Secondly, since the GM of a log-normal distribution equals the median, the median can be used instead in cases when only the median is reported. Finally, biomonitoring studies on urinary BPA always report the GM and/or the median, whereas the AM is only rarely given. The GM and the 95th percentile of the volume-based total BPA concentrations are used to derive estimates of average and high daily BPA exposures. For comparative purposes, daily BPA exposures are also calculated from creatinine-based BPA concentrations.

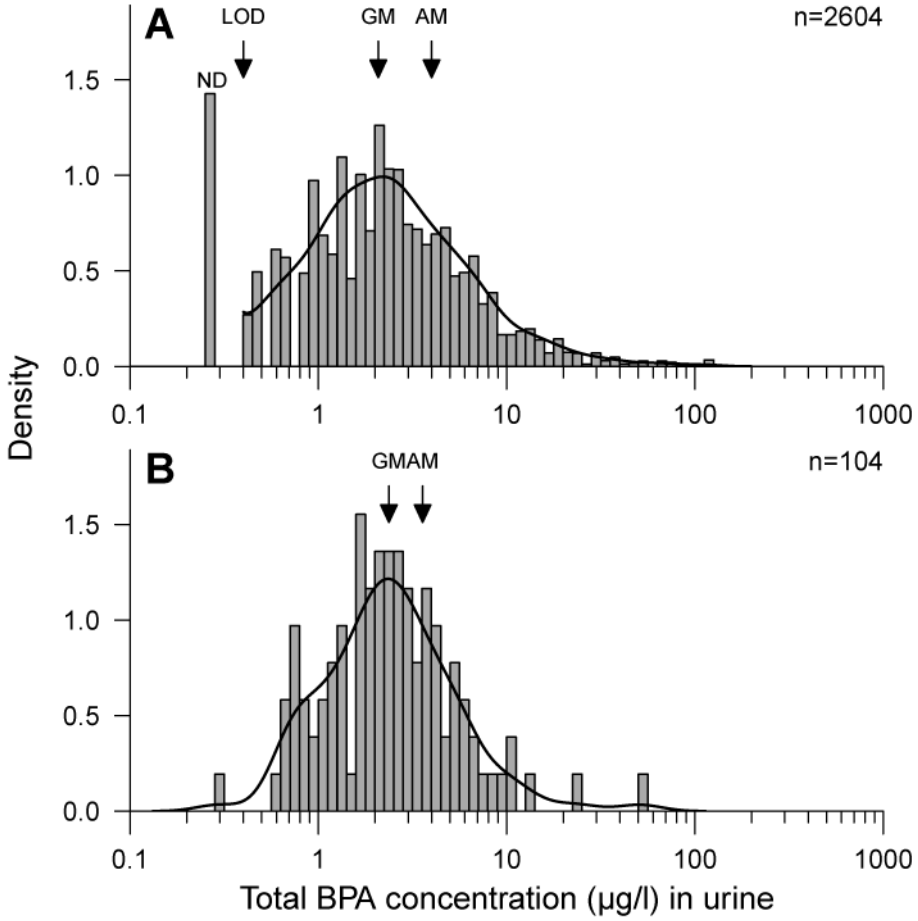


Figure 1: Lognormal distribution shape of urinary BPA concentration. Shown are the histogram and density plot of the total BPA concentration in urine for two example data sets. (A) NHANES 2005–2007 data for the total US population, (B) children of the Duisburg birth cohort study (Kasper-Sonnenberg et al., 2012). Arrows indicate the location of the geometric mean (GM), arithmetic mean (AM), and the limit of detection (LOD). The number of subjects (n) is additionally given. ND, fraction of nondetects.

Information about the specific distribution characteristics of urinary BPA concentration has consequences on how to handle left-censored data, i.e. observations below the limit of detection. Using a lower-bound approach (i.e. setting all undetected observations to zero) would make the GM calculation unfeasible, whereas the upper-bound approach (i.e. setting them to the LOD) would introduce a positive bias and, thereby, would overestimate the average concentration. Hornung and Reed (1990) have shown that the substitution of non detectable values by LOD/$\sqrt{2}$ is most appropriate for log-normally distributed data with moderate geometric standard deviations (GSD < 3) and low non

detection rates (<30 %). For larger GSD values, the middle-bound approach (i.e. setting the nondetectable values to LOD/2) is recommended (Hornung and Reed, 1990).

The geometric standard deviation (GSD), which is a unit-less multiplicative factor, is only very rarely reported in the biomonitoring studies on urinary BPA. However, for the freely available raw data of the US National Health and Nutrition Survey (NHANES, online), the GSD can be calculated. Using the volume-based urinary BPA concentrations of the last four survey periods and a grouping in four age classes (Figure 4), the average GSD can be calculated to be 2.9±0.2 (mean±standard deviation, range: 2.5–3.1, n=16 GSD values). Taking additionally the low non detection rates (2.4–12 %, Figure 4) into account, the replacement of nondetectable values by LOD/√2 is recommended according to Hornung and Reed (1990), and this setting has also been chosen by NHANES (Lakind et al., 2012). Using a value of LOD/2 instead of LOD/√2 for imputation would lower the GMs in Figure 4 by only 2.5±1.2 % (n=16, range: 0.7–4.7 %), which is a negligible effect. In conclusion, according to Hornung and Reed (1990) the impact of the imputation procedure is negligible as long as the non detection rates do not exceed 15 %.

The above decision of using the GM leads to an estimate for the average daily BPA exposure which is lower than the AM-based estimate. The reason for this so-called AM–GM inequality is the log-normal distribution shape of the urinary BPA data. To convert GM-based estimates into AM-based estimates, which are then comparable to those derived from the modelling approach, a multiplicative conversion factor of k = exp[0.5×LN(GSD)] is introduced. Using the GSD values of the NHANES data (see above), an average value for k of 1.7±0.1 (n=16, range: 1.5–1.9) is obtained, which is well in line with the directly calculated average AM/GM ratio of 1.9±0.4 (n=16). Additional information on the AM/GM ratio is obtained from the Canadian Health Measures Survey CHMS 2007–2009 with an average value of 1.9±0.1 (n=4) and from a few European studies with values of 1.5 from the Duisburg cohort study (Kasper-Sonnenberg, personal communication), and 1.8 from the German Environmental Survey for Children (GerES IV). A conversion factor of 1.8 is therefore used in this opinion to convert GM-based estimates into AM-based estimates.

For US National Health and Nutrition Survey (NHANES), descriptive statistics were calculated for specific age classes (see Chapter 4.4 Food consumption) by using the statistical computing environment R (R Core Team, 2012) in combination with the R survey package (Lumley, 2004, 2012), which has been recently used, for example, by Lakind et al. (2012). The outcome of the statistical procedures was checked by comparing the predictions for the default NHANES age groups with published data (CDC, 2012). All graphical figures were generated using the R lattice package (Sarkar, 2008).

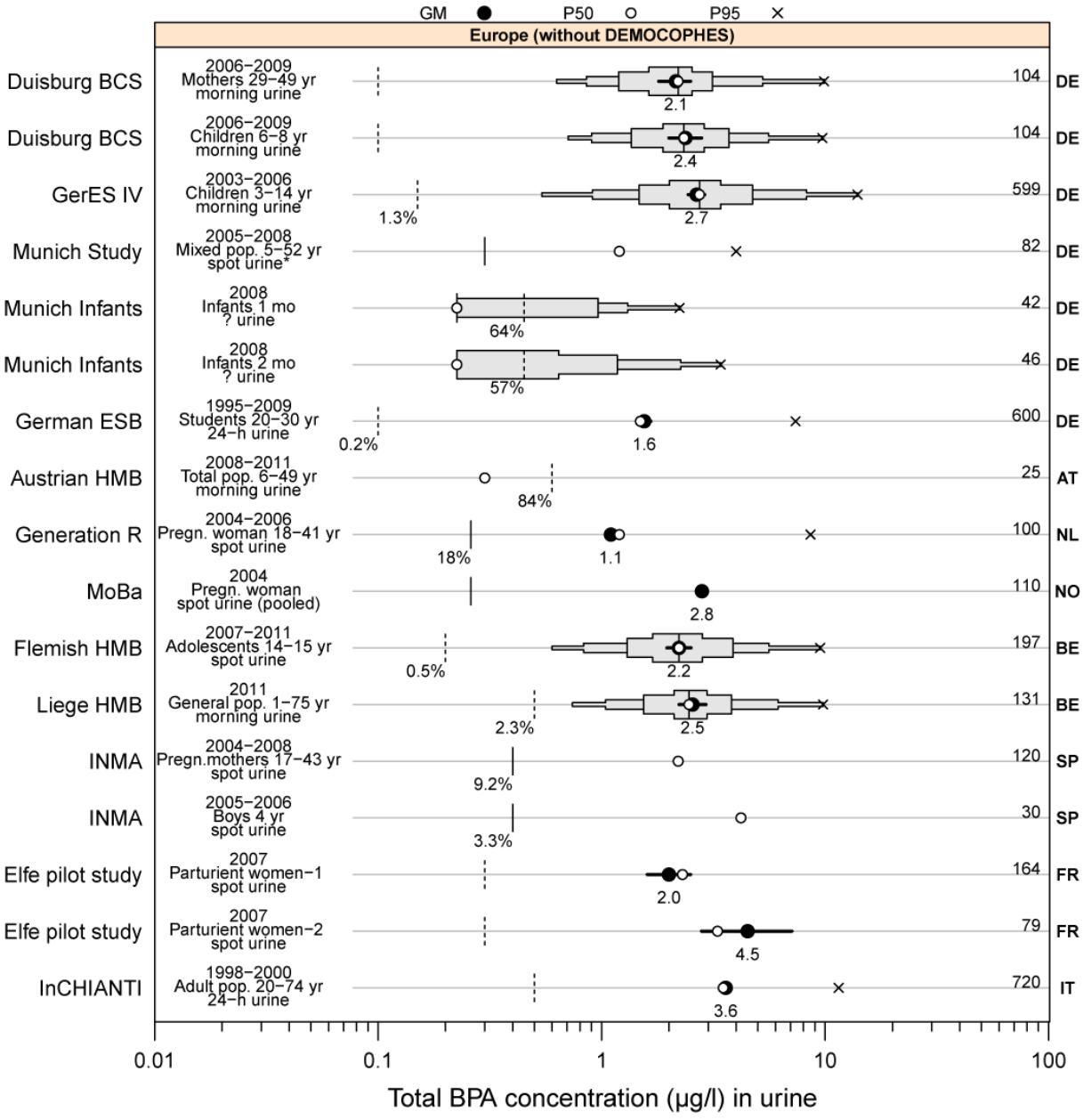
Urinary BPA concentrations (volume-based data)

Since 2006, a relatively large amount of data on total BPA concentration in urine have become available in selected populations from various regions, including North and South America, Europe, Africa, Asia and Australia. The studies comprise large-scaled, population-based cross-sectional studies, a spectrum of smaller-scaled studies on specific population groups, usually from a single location or region, as well as retrospective studies and prospective longitudinal studies.

As shown in Figure 2, European human biomonitoring (HBM) data on urinary total BPA are available from the German Environmental Survey for Children (GerES IV) (Becker et al., 2009; Kolossa-Gehring et al., 2012), the German Environmental Specimen Bank (ESB) study (Koch et al., 2012; Kolossa-Gehring et al., 2012), the Duisburg birth cohort study (BCS) (Kasper-Sonnenberg et al., 2012), two Munich studies (Völkel et al., 2008, Völkel et al., 2011), the Austrian HBM study (Hohenblum et al., 2012), the Flemish and Liege HBM studies (Milieu en Gezondheid, 2010; Pirard et al., 2012; Schoeters et al., 2012), the Generation R (Rotterdam) study (Ye et al., 2008a), the Norwegian mother and child birth cohort (MoBa) study (Ye et al., 2009a), the Spanish environment and childhood (INMA) project (Casas et al., 2011), the French Elfe pilot study (Vandentorren et al.,

2105 2011), the Italian InCHIANTI study (Galloway et al., 2010). Findings from the European-wide pilot
2106 study DEMOCOPHES (Joas et al., 2012) are shown in Figure 3.

2107



2108

2109 **Figure 2:** Urinary BPA concentrations of European studies (without DEMOCOPHES, see Figure 3).
2110 Shown are the concentrations of total urinary BPA from different European studies. Box-percentile
2111 plots (gray-shaded areas) show the distributional characteristics comprising the 5th, 12.5th, 25th, 37.5th,
2112 50th, 62.5th, 75th, 87.5th, and 95th percentiles. Filled circles with associated values and error bars
2113 indicate the geometric means and the 95 % confidence intervals. The 50th and 95th percentiles are
2114 shown by open circles and crosses. The number of subjects is given on the right. Vertical solid and
2115 dashed lines indicate the LOD and the LOQ, respectively. The proportion of measured values below
2116 the LOD (or LOQ) is given as a percentage. Additionally given are the sampling periods and sampling
2117 populations, and the kind of urine sampling ("?" means that no info on the urine sampling was
2118 available).

The fourth German Environmental Survey (GerES IV) is a representative study focussing on the chemical exposure of children (Becker et al., 2009; Kolossa-Gehring et al., 2012). Morning urine samples were collected from 3–14 year old children in 2003–2006. The concentration of total BPA was measured by GC-MS/MS with a LOQ of 0.15 µg/l. BPA was detected in 98.7 % of the $n = 599$ samples with a geometric mean of 2.7 µg/l and a 95th percentile of 14.0 µg/l (Becker et al. 2009) (Figure 2). The uncertainty in the geometric mean as expressed by the 95th percentile confidence interval corresponded to a relative margin of error of 8–9 %. An analysis by age groups revealed a significantly higher BPA concentration (GM: 3.55 µg/l) in the age category 3–5 years compared to the 6–8 yrs, 9–11 yrs, and 12–14 yrs age categories (GM: 2.22–2.72 µg/l).

By using historical samples from the German Environmental Specimen Bank (ESB), Koch et al. (2012) analysed retrospectively the extent of BPA body burden in the German population from 1995–2009 based on a total of 600 24-h urine samples. According to the ESB concept, samples were taken annually from approximately 60 male and 60 female students (20–30 years old) at each of four university cities (two from East Germany and two from West Germany). Total and unconjugated BPA was determined by HPLC-MS/MS with an LOQ of 0.1 µg/l. In the stored urine samples, total BPA was quantifiable in 99.8 % with a geometric mean of 1.6 µg/l (relative margin of error: 7 %) and a 95th percentile of 7.4 µg/l (Koch et al., 2012) (Figure 2). Unconjugated BPA was quantifiable in <15 % of the samples. Total BPA concentrations (geometric mean) decreased over time from 1.9 µg/l in 1995 to 1.3 µg/l in 2009, but 24-h urine volumes (mean) increased from 1.6 litres in 1995 to 2.1 litres in 2009. The derived daily exposures therefore remained rather constant at a geometric mean of 39 ng/kg bw/day (95 % confidence interval (CI): 37–42 ng/kg bw/day) and a 95th percentile of 171 ng/kg bw/day.

Within the framework of the Duisburg birth cohort study (Duisburg BCS), 208 morning urine samples of 104 mother-child pairs (29–49 and 6–8 years old) were collected in 2006–2009 (Kasper-Sonnenberg et al., 2012). Total BPA was measured by LC-MS/MS with an LOQ of 0.1 µg/l. Total BPA was quantifiable in all samples. The geometric mean concentration was 2.1 µg/l (95 % CI: 1.8–2.5 µg/l) in the mothers and 2.4 µg/l (95 % CI: 2.0–2.8 µg/l) in the children (Figure 2); the relative margin of error was 14–19 %. The 95th percentile of total urinary BPA was 8.4 µg/l for the mothers and 9.7 µg/l for the children. The BPA concentrations between children and mothers showed a low but significant correlation ($r_{Spearman} = 0.22$, p-value ≤ 0.05).

In the Munich infants study (Völkel et al., 2011), females who were participating in a birthing class in Munich were randomly selected, and 47 mother-infant pairs finally entered into the study. Urine was sampled from each infant at one month and two months of age in 2008. Total and unconjugated BPA was measured by HPLC-MS/MS with a LOQ of 0.45 µg/l. Unconjugated BPA was only detectable in 3.3 % of the samples. Total BPA was detected in 35.7 % of the first-month samples and in 43.5 % of the second-month samples (Figure 2). The 95th percentile of total urinary BPA for the first-month and second-month samples was 2.2 µg/l ($n = 42$) and 3.4 µg/l ($n = 45$), respectively. Note that these P95 values are different from those reported in the study (9.6 and 5.1 µg/l) in which the subset of detectable values was used to derive the 95th percentile. The distributional shape of the total BPA concentration was quite unusual with a 95th percentile (P95) more than 10–15-fold higher than the median (P50) (Figure 2). A typical range for the P95-to-P50 ratio from other studies is 5–6.

The second Munich study (Völkel et al., 2008) analysed spot urine samples from different sources, comprising 62 (multiple) samples from 21 co-workers (19–52 years old) as well as single samples from 31 women (18–41 years old) and 30 children (5–6 years old). The samples were collected in 2005–2008. Total BPA was measured by HPLC-MS/MS with a LOQ of 0.3 µg/l. The median concentration and 95th percentile of this heterogeneous data set was 1.2 and 4.0 µg/l, respectively (Figure 2).

The first population-based human biomonitoring study in Austria (Hohenblum et al., 2012) was performed in 2008–2011 and included 150 volunteers (6-49 years old) from 50 families from five different Austrian regions. Ten woman-child-men groups living in the same household were randomly

selected per region. 25 out of 100 collected first morning urine samples were analysed for total urinary BPA concentration. Questionnaire data were used to pre-select participants who might have a higher exposure (e.g. due to occupation, frequent use of canned food/beverages, use of plastic bottles). Total BPA was quantified by HPLC-MS/MS with an LOQ of 0.6 µg/l. Total BPA was detected in 16 % of the samples; the maximum BPA concentration was 11 µg/l (Figure 2). The detection rate was remarkably low compared to the typical rates reported in other European studies.

The Flemish Environment and Health Survey 2007–2011 cycle-2 (FLEHS II) focussed on obtaining reference values for a wide range of age-specific biomarkers of exposure in a representative sample of the Flemish population (Schoeters et al., 2012). BPA data from FLEHS II were provided by the Flemish Center of Expertise on Environment and Health, financed and steered by the Ministry of the Flemish Community. BPA was measured in spot urine samples of n = 197 teenagers (14–15 year old) by GC-MS with an LOQ of 0.2 µg/l (Milieu en Gezondheid, 2010). Total BPA was detected in 99.5 % of the samples. After adjusting for age, gender, and urinary creatinine, a geometric mean for the total BPA concentration of 2.2 µg/l (relative margin of error: 12–13 %) was obtained (Figure 2). The 95th percentile was 9.5 µg/l.

The Liege HMB study analysed urinary levels of environmental contaminants of a general Belgian population (1–75 years old) living in Liege and surrounding areas (Pirard et al., 2012). Morning urine samples were collected in 131 subjects in 2011, and total urinary BPA was quantified by GC-MS/MS with a LOQ of 0.50 µg/l. Total BPA was quantifiable in 97.7 % with a geometric mean of 2.6 µg/l and a 95th percentile of 9.8 µg/l (Figure 2). BPA levels in urine of people living in the same home and collected at the same time were fairly correlated ($r_{Pearson}$ = 0.88).

The Generation R study is a population-based birth cohort study in Rotterdam (Jaddoe et al., 2007). Multiple spot urine samples were collected from 9 778 pregnant females (18–41 years old) at 21–38 weeks of gestation. BPA was measured in a subset of urine samples collected from 100 women after 20 weeks of gestation in 2004–2006 (Ye et al., 2008a). BPA was quantified by GC-MS/MS with a LOD of 0.26 µg/l. Total BPA was detected in 82 % of the samples with a geometric mean of 1.1 µg/l and a 95th percentile of 8.6 µg/l (Figure 2).

Within the framework of the Norwegian mother and child birth cohort (MoBa) study, 110 urine spot samples were collected in 2004 from pregnant woman at 17–18 weeks of gestation (Ye et al., 2009a). Urine samples from groups of 11 subjects each were combined to make 10 pooled samples. As in the Generation R study, BPA was quantified by GC-MS/MS with a LOD of 0.26 µg/l. The geometric mean of the total BPA concentration in the 10 pooled samples was 2.8 µg/l (Figure 2).

The INMA (Infancia y Medio Ambiente) project is a population-based birth cohort study in Spain. 120 pregnant women (17–43 years old) were selected at random from four different regions and 30 children (4-year old boys) were selected from a fifth region. Spot urine samples were collected from the women during the 3rd trimester of pregnancy in 2004–2008, and from the children in 2005–2006. Urinary BPA was quantified by HPLC-MS/MS with a LOD of 0.4 µg/l. In the pregnant women, total urinary BPA was detected in 90.8 % of the samples with a median concentration of 2.2 µg/l (Figure 2). The 4-year old boys had a median concentration of 4.2 µg/l; the detection rate was 96.7 %.

The French longitudinal study of children (Elfe: Etude Longitudinale Française depuis l'Enfance) is a national cohort study examining the effects of environmental exposure on children's health (Vandentorren et al., 2011). Prior to this study, a pilot survey was conducted in two regions for validation purposes, which included the collection of spot urine samples from parturient women having a natural delivery (n = 164) or a Caesarean/forceps delivery (n = 79) in hospital maternity units. Total and unconjugated BPA was quantified by GC-MS with an LOQ of 0.3 µg/l. Total BPA was quantifiable in 96.9 % of all samples. The geometric mean concentration was 2.0 µg/l (95 % CI: 1.6–2.5 µg/l) in the natural-delivery group and 4.5 µg/l (95 % CI: 2.8–7.1 µg/l) in the Caesarean/forceps-delivery group (Figure 2). The higher values in women who had Caesarean sections

(or forceps delivery) suggest a contamination from medical devices either from catheterisation or urine probes when biomonitoring at delivery (Vandentorren et al., 2011).

To estimate daily BPA excretion levels in a large European cohort, Galloway et al. (2010) selected participants from the InCHIANTI study, a representative population-based study conducted in Chianti. 24-h urinary samples were collected from 720 participants (20–74 years old) in 1998–2000. During the three days before sample collection, the subjects consumed a diet free of meat and fish. Total BPA levels were measured by HPLC-MS/MS with a LOQ of 0.5 µg/l. The geometric mean and 95th percentile of the total BPA concentration in urine was 3.6 µg/l (relative margin of error: 5 %) and 11.5 µg/l (Figure 2), respectively.

DEMOCOPHES (Demonstration of a study to Coordinate and Perform Human Biomonitoring on a European Scale) is a pilot study funded by DG Research in the 7th Framework Programme (FP7/2007–2013) and aiming to demonstrate the harmonisation of HBM in Europe (Joas et al., 2012). DEMOCOPHES is a cross-sectional study of the European population's exposure to various substances using human biomarker data collected in 17 European countries from a non representative sampling of mother-child pairs in 2011–2012 (Joas et al., 2012). It is designed to cover an urban and a rural part of each country, involving mother-child pairs comprising an equal number of 6–11 year old boys and girls, and their mothers (Kolossa-Gehring et al., 2012). Urinary BPA was measured on a voluntary basis in only a few countries (Sweden, Luxembourg, Denmark, Slovenia, Belgium) using MS-based methods. Sweden recruited 100 mother-child pairs and reported geometric mean BPA concentrations of 1.2 µg/l for the mothers and 1.5 µg/l for children (M. Berglund, pers. communication) (Figure 3). In Luxembourg, 60 mother-child pairs were sampled, and the total BPA concentration was measured by LC-MS with LOQs of 1.0 and 2.0 µg/l (A. C. Gutleb, pers. communication). The geometric mean concentrations were 1.7 (mothers) and 1.8 µg/l (children). Denmark recruited 145 mother-child pairs from an urban area near Copenhagen and a rural area near Roskilde (Frederiksen et al., 2013). The study was additionally funded by the Danish Health and Medicines Authority, the Danish environmental protection agency and the Danish veterinary and food administration. The total BPA concentration was measured by LC-MS/MS, and the geometric mean concentrations were 2.0 µg/l (mothers) and 1.9 µg/l (children). In Slovenia, 155 mother-child pairs were recruited, and the median BPA concentrations were of 0.7 µg/l for the mothers and 2.0 µg/l for the children (M. Horvat, pers. communication). In Belgium, 129 mother-child pairs were sampled in the urban region of Brussels and in a rural area in the West of the country. Geometric mean concentrations of BPA were 2.6 µg/l for the mothers and 2.4 µg/l for the children (Covaci et al., 2012).

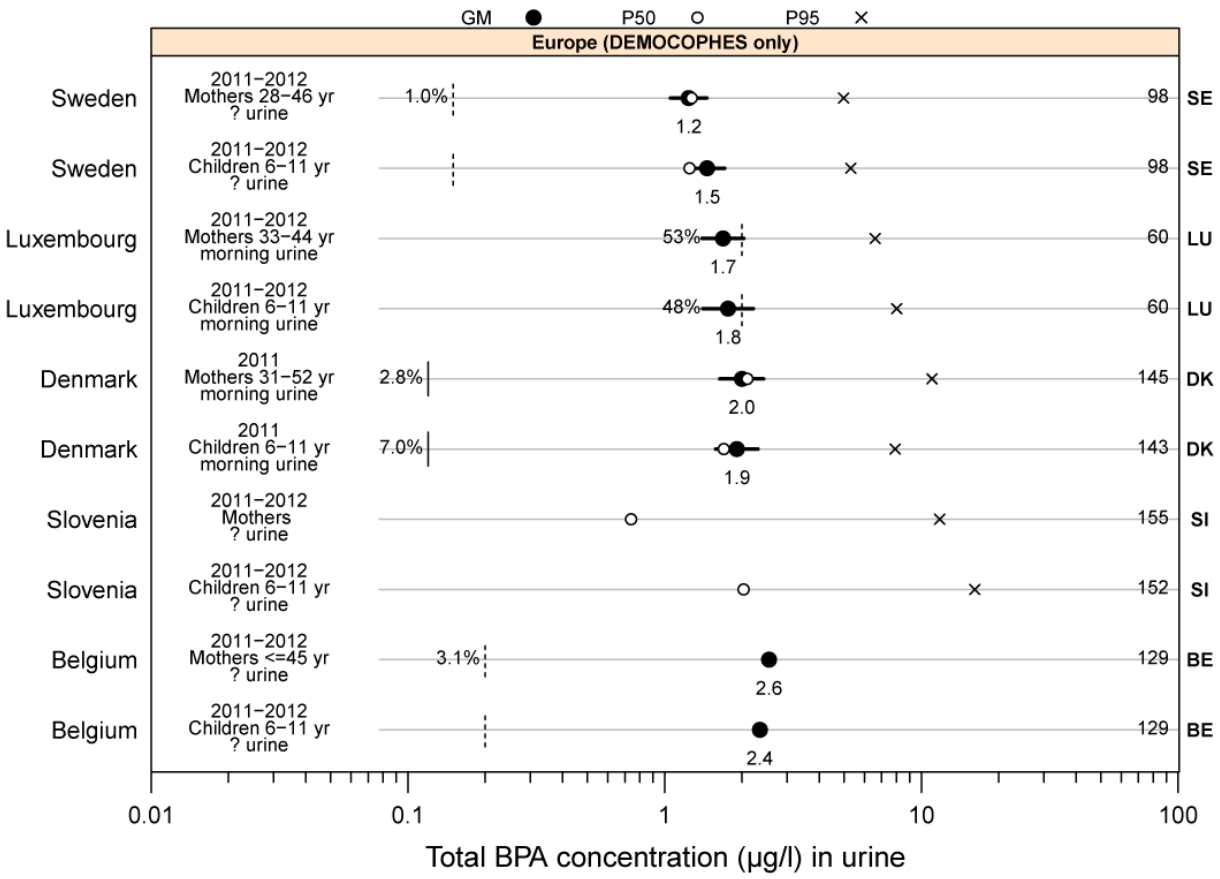


Figure 3: Urinary BPA concentrations in European mother-child studies from DEMOCOPHES. Shown are the concentrations of total urinary BPA in mothers and their 6–11 year old children for individual European countries. Open circles with associated numbers and error bars indicate the geometric means and the 95th confidence intervals. The 50th and 95th percentiles are shown by open circles and crosses. The number of subjects is given on the right. Vertical solid and dashed lines indicate the LOD and the LOQ, respectively. The proportion of measured values below the LOD (or LOQ) is given as a percentage. Additionally given are the sampling periods and the kind of urine sampling ("?" means that no info on the urine sampling was available). For references, see main text.

Among the non-European data, the largest data sets on urinary BPA levels have been generated within the framework of the US National Health and Nutrition Survey (NHANES) and the Canadian Health Measures Survey (CHMS). Because of their large sample size and their cross-sectional, nationally representative, population-based character, these surveys are used here for comparative purposes to provide reference values on average and high concentrations of total BPA in urine.

Both North American surveys used spot urine samples and measured the concentration of total BPA. The surveys differed slightly in their analytical procedures (Lakind et al., 2012). For example, the NHANES analysed the samples by HPLC-MS/MS with a LOD of 0.4 µg/l and a LOQ of 1.2 µg/l; measurements below the LOD were assigned a value of LOD/√2. The CHMS used GC-MS/MS with a LOD of 0.2 µg/l and a LOQ of 0.82 µg/l; missing values (<LOD) were assigned a value of LOD/2. Both surveys performed reagent-blank checks, but only the CHMS found results slightly above the LOD that were subtracted from the reported data.

In the last four NHANES surveys, covering the periods from 2003–2004 to 2009–2010, BPA was detected among the different age classes in 88–98 % of the 6 to >80 years old participants (n = 2 517–2 749 subjects in total) with a geometric mean of 1.5–3.7 µg/l (relative margin of error: 7–27 %) and a 95th percentile of 8.2–19.4 µg/l (CDC, 2012) (Figure 4).

In the CHMS 2007–2009 cycle-1 survey, BPA was detected among the different age classes (Figure 4) in 6–12 % of the 6–79 years old participants (n = 5 476 subjects in total) with somewhat lower geometric means of 0.9–1.5 µg/l (relative margin of error: 7–18 %) and somewhat lower 95th percentiles of 5.2–8.4 µg/l (Health Canada, 2010). Recent data from the CHMS 2009–2011 cycle-2 survey do not differ from those found in 2007–2009 cycle-1 survey period (Figure 4).

Given the survey differences in geometric means and 95th percentiles of the urinary BPA levels, it can be speculated whether analytic differences such as CHMS-specific background subtraction could have led to a bias in the results. Lakind et al. (2012) examined this issue as well as the differences in the survey methodologies (e.g. participant selection, urine sampling, fasting time) and concluded that the survey differences are unlikely to have substantial impacts on inter-survey comparisons of BPA exposures.

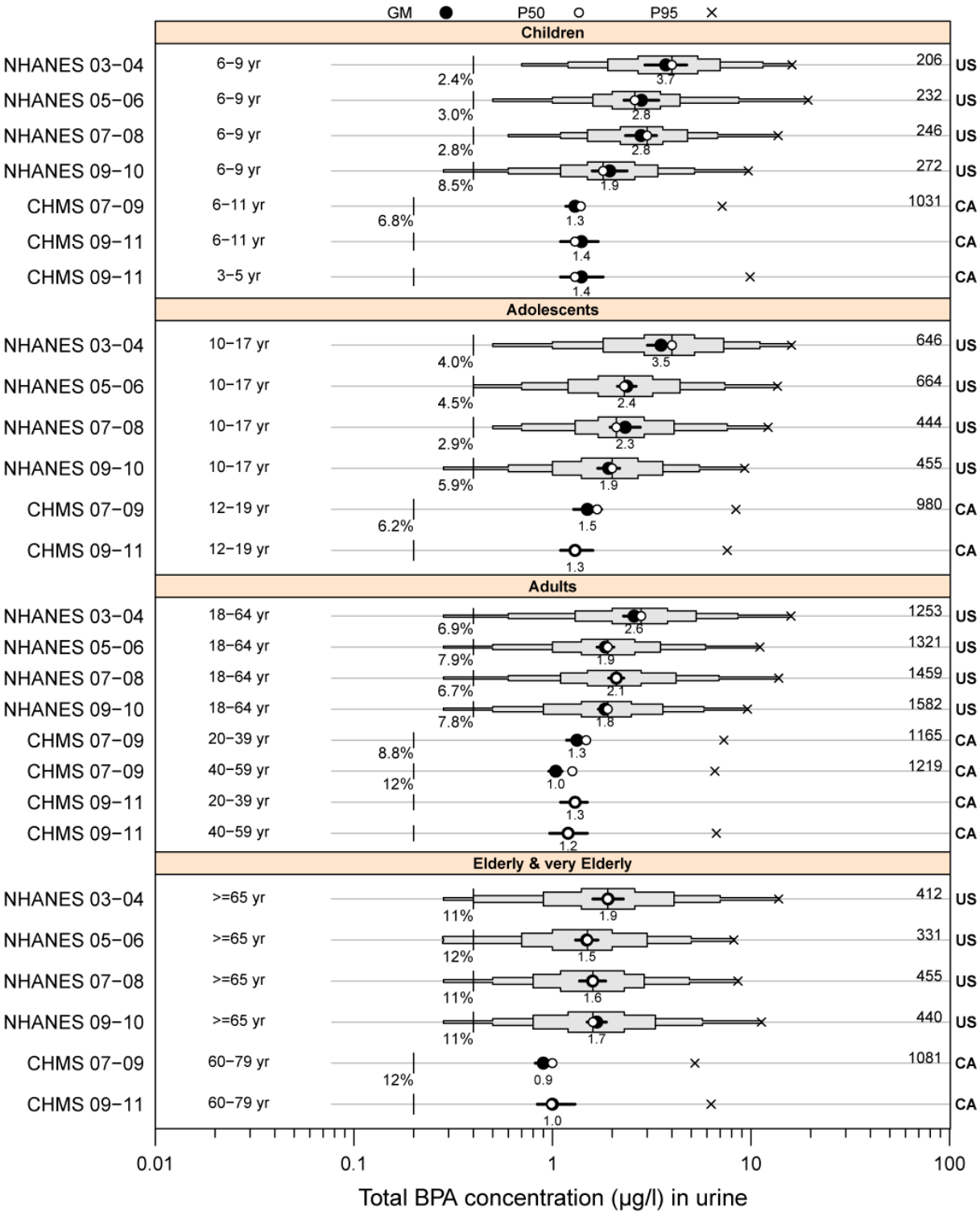


Figure 4: Urinary BPA concentrations of the large-sized North-American surveys grouped by the age classes and survey period. Shown are the concentrations of total urinary BPA from the US National Health and Nutrition Examination Survey (NHANES) and the Canadian Health Measures Survey (CHMS). Box-percentile plots (gray-shaded boxes) show the distributional characteristics comprising the 5th, 12.5th, 25th, 37.5th, 50th, 62.5th, 75th, 87.5th, and 95th percentiles. Filled circles with associated values and error bars indicate the geometric means and the 95 % confidence intervals. The 50th and 95th percentiles are shown by open circles and crosses. The number of subjects is given on the right. Vertical lines indicate the LOD. The proportion of measured values below the LOD is given as percentages. Country codes are shown on the right.

Further data from biomonitoring studies on urinary BPA levels are available from North and South America, Africa, Asia and Australia. The only study on newborns is included here; for all others see Appendix VII.

Nachman et al. (2013) analysed the urinary BPA concentrations in 12 healthy newborns (7–44 days old), whose mothers were recruited from the newborn nursery at the Johns Hopkins Hospital/USA. On the day of sample collection, the newborns had received infant formula or human milk, or a mixture of both. Unconjugated and glucuronidated BPA in urine was quantified by HPLC-MS/MS with a LOQ of 0.1 µg/l. Samples were analysed in two technical replicates each, and unconjugated BPA was detected only in one of the 24 replicates. The geometric mean of the glucuronidated BPA concentration was 0.74 µg/l which corresponded in terms of the mass of the unconjugated form to a value of 0.42 µg/l with an associated 95 % confidence interval of 0.29–0.61 µg/l. As almost no data are available for infants, this study is considered for the estimation of daily BPA exposure in infants.

To conclude, a relatively large amount of information on urinary BPA concentration is available for the European region. Only a few of the larger-sized European studies, however, can be assumed to be representative such as the German Environmental Survey (GerES IV) for a population of children in Germany, the Flemish Environment and Health Survey (FLEHS II) for the 14–15 years old teenagers of the Flemish population, the INMA project for pregnant women in Spain, and the InCHIANTI study for the 20–74 year olds from the Chianti region. All age classes are covered except the 1–3 years old toddlers. The analytical sensitivity to detect and quantify BPA varied between the different studies with LODs of 0.05–0.4 µg/l and LOQs of 0.1–2.0 µg/l. The distributional characteristics of the total BPA concentrations in terms of shape and spread are generally quite homogeneous across the different studies. On a $\log_{10}$-transformed scale, the distributions appear symmetrical, and the similarity of the geometric mean (GM) and the median (P50) indicate that the GM rather than the arithmetic mean is the appropriate measure for the central tendency. For the European studies, the GM of the total BPA concentrations is in general localised in the range between 1.1–3.6 µg/l (Figure 2 and 3), which is in agreement with the results of the large-sized North-Americ survey NHANES and CHMS (Figure 4). Exceptions from this general tendency are the Munich infants study and the Austrian HBM study with median values far below 0.6 µg/l, the Slovenian DEMOCOPHES study with a median value of 0.7 µg/l for the mothers, and the Elfe pilot study on parturient women having a Caesarean/forceps delivery (GM = 4.5 µg/l). An additional finding relevant for the estimation of high exposures is the 95th percentile (P95), which, for studies with spot-urine sampling, is 5–6-fold higher than the median value.

Creatinine-based BPA concentrations in urine

Expressing urinary BPA concentration as creatinine-based data (µg BPA/g creatinine) rather than volume-based data (µg BPA/l urine) is an alternative that aims to correct for urinary dilution. Depending on which basis is chosen, assumptions on daily urinary output (volume) or daily creatinine excretion (mass) are required to estimate BPA exposure. Many factors contribute to the daily variability in creatinine output as discussed in detail by Lakind and Naiman (2008). Creatinine-based BPA concentrations in urine are available only for a few European studies comprising the Duisburg birth cohort study (Duisburg BCS), the German Environmental Specimen Bank study (German ESB), the Flemish and Liege HBM studies, the birth cohort study in Rotterdam (Generation R), and the Norwegian mother and child birth cohort study (MoBa). The descriptive statistics (GM, P50, P95) with associated information on gender, age, and sampling are given in Table 26. The data for the North-American surveys NHANES and CHMS are included for comparative purposes. For the European studies except the MoBa study, the geometric means of the creatinine-based total BPA concentrations are in the range between 1.7–2.5 µg/g creatinine which conforms with the results of NHANES and CHMS (GM: 1.3–4.8 µg/g creatinine). The MoBa study on pregnant women is distinguished by a considerably higher value of 5.9 µg/g creatinine. The P95-to-P50 ratio for the studies with spot-urine sampling is 4.4–5.2 (European studies) and 3.3–6.7 (NHANES and CHMS), respectively, which is similar to that found for the volume-based data. Remarkably, the P95-to-P50

ratio for the German ESB study is only 3.6 which indicates a reduced variability very likely due to the 24-h urine sampling design.

Table 26: Descriptive statistics for creatinine-based BPA concentrations in urine. The table shows the geometric mean (GM), median (P50), and the 95th percentile (P95) of the creatinine-adjusted BPA concentration (µg/g creatinine) for the European studies and for the North-American surveys NHANES and CHMS. M: male, F: female, 24hU: 24-h urine, MU: morning urine, SU: spot urine.

Study	Gender	age	Sampling	GM	P50	P95
				(µg/g creatinine)		
German ESB	MF	20–30 yr	24hU	1.8	1.7	6.2
Duisburg BCS	F	29–49 yr	MU	2.3	2.1	10.0
Duisburg BCS	MF	6–8 yr	MU	1.8	1.7	6.2
Generation R	pregnant F	18–41 yr	SU	1.7	1.6	8.3
MoBa	pregnant F		SU	5.9	–	–
Flemish HMB	MF	14–16 yr	SU	1.7	1.5	7.5
Liege HMB	MF	7–75 yr	MU	2.5	2.3	13.7
NHANES03–05	MF	6–9 yr	SU	4.8	4.7	15.7
NHANES05–06	MF	6–9 yr	SU	3.4	3.0	22.5
NHANES07–09	MF	6–9 yr	SU	3.6	3.3	20.8
NHANES09–10	MF	6–9 yr	SU	2.7	2.6	9.9
CHMS07–09	MF	6–11 yr	SU	2.0	1.9	9.8
NHANES03–05	MF	10–17 yr	SU	2.9	2.9	12.2
NHANES05–06	MF	10–17 yr	SU	1.9	1.7	11.9
NHANES07–09	MF	10–17 yr	SU	2.0	1.8	7.0
NHANES09–10	MF	10–17 yr	SU	1.7	1.6	7.2
CHMS07–09	MF	12–19 yr	SU	1.3	1.3	6.4
NHANES03–05	MF	18–64 yr	SU	2.4	2.4	9.8
NHANES05–06	MF	18–64 yr	SU	1.8	1.6	8.7
NHANES07–09	MF	18–64 yr	SU	2.0	1.9	9.1
NHANES09–10	MF	18–64 yr	SU	1.9	1.8	7.7
CHMS07–09	MF	20–39 yr	SU	1.5	1.5	6.8
CHMS07–09	MF	40–59 yr	SU	1.3	1.3	7.5
NHANES03–05	MF	≥65 yr	SU	2.3	2.3	12.1
NHANES05–06	MF	≥65 yr	SU	1.8	1.6	8.8
NHANES07–09	MF	≥65 yr	SU	2.2	2.1	9.3
NHANES09–10	MF	≥65 yr	SU	1.9	1.8	8.4
CHMS07–09	MF	60–79 yr	SU	1.3	1.3	7.6

Estimation of daily BPA exposure from volume-based urinary BPA concentration

Estimation of BPA exposure based on volume-based urinary BPA concentration is used in the present opinion as a plausibility check for the calculated exposure estimates for BPA uptake via food and non-food sources. Volume-based urinary BPA data are given preference over creatinine-based data because these are supported by a larger number of European studies. Based on measured urinary concentration of total BPA C_{BPA} (µg/l), the daily BPA exposure $\dot{m}_{BPA}$ (ng/kg bw/day) was calculated by

$$\dot{m}_{BPA} = \frac{C_{BPA} \times \dot{V}_{urine}}{W}$$

where $\dot{V}_{urine}$ (ml/day) is the urinary output rate and W (kg) is the body weight (Lakind and Naiman 2008; UBA, 2012). Depending on whether body weight is available from the studies, either study-specific individual or mean values, or generic values derived by linear interpolation from body weight

vs. age relationships taken from literature, were used. Literature data were also used for the urinary output rate except for cases where study-specific individual urinary volumes from 24-h urine sampling were available. Lakind and Naiman (2008) provide detailed discussion on the range and variability of age/gender-specific body weight and urinary output rate.

Table 27 shows the body-weight and urinary output-rate parameters which were used to translate urinary BPA concentration into daily exposure. Parameters are given only for European studies and the North American surveys. Generic values for body weight were taken from the German National Health Interview and Examination Survey 1998 (Bergmann and Mensink, 1999), the German Health Interview and Examination Survey for Children and Adolescents (Stolzenberg et al., 2007), the Italian National Food Consumption Survey INRAN-SCAI 2005–06 (Leclercq et al., 2009), and from the reference values given by the International Commission on Radiological Protection (ICRP) (Valentin, 2002). For the urinary output rate, generic values were taken from Valentin (2002) and from Willock and Jewkes (2000). For comparative purposes, daily BPA exposures for the large-sized population-based surveys from North America (NHANES, CHMS) were also calculated, based on the survey-specific, individual body weights and on the generic urine volumes taken from ICRP reference tables (Valentin, 2002).

Estimates for the average and high levels of daily BPA exposure were calculated by using the geometric mean (GM), the median (P50) and the 95th percentile (P95) of the urinary BPA concentration of spot urine samples, first morning urine samples, and 24-h urine samples. Because of BPA's short elimination half-life, spot urinary concentrations primarily reflect the exposure that occurred within a relatively short period before urine collection (WHO, 2011a). Nevertheless, the single spot-sampling approach may adequately reflect the average BPA exposure of a population, provided the samples are collected from a large number of individuals and at random in relation to meal ingestion and bladder-emptying times.

The 95th percentile (P95) of urinary BPA concentration is used to obtain estimates for high BPA exposures. It is, however, noted that the P95 has different interpretations depending on whether spot urine samples, first morning urine samples, or 24-h samples are used. For spot urine samples, the P95 is related to the 95 % probability that a single, randomly collected sample from a randomly selected subject has an urinary BPA concentration not exceeding the 95th percentile. This is important as urinary BPA concentrations of repeated urine collections from individuals may vary up to two orders of magnitude (Ye et al., 2011; Teeguarden et al., 2011; Christensen et al., 2012a). The variability of urinary BPA levels has been analysed from repeated/serial urine collections by using so-called nested random-effects models (Braun et al., 2011; Ye et al., 2011), which can adequately reflect the hierarchical structure of the main sources of variability: (1) between persons, (2) within person/between days, and (3) within person/within day. The study by Ye et al. (2011) revealed that the total variance in spot urine collections could be subdivided into 70 % within-day variability, 21 % between-day variability, and 9 % between-person variability. The substantial within-day variability is lacking in 24-h urine samples, so that the 95th percentile can be expected to be closer to the average concentration (GM, median) than in spot urine samples and first morning urine samples (Aylward et al., 2012).

Table 27: Body-weight and urinary output-rate parameters for the considered European and North American Studies. The table provides the parameters for body weight (W), urinary output rate ($\dot{V}_{urine}$), and the specific urinary output rate (spec. $\dot{V}_{urine}$), which were used to translate urinary BPA concentration into daily BPA exposure. Gender and age were taken into account when deriving generic parameter values from published parameter-age relationships by linear interpolation. Study-specific parameters are set in italic font. References from which these parameters were taken are: [1] Koch et al. (2012), [2] Bergmann and Mensink (1999), [3] Valentin (2002), [4] Stolzenberg et al. (2007), [5] Willock and Jewkes (2000), [6] Ye et al. (2009a), [7] Leclercq et al. (2009), [8] Galloway et al. (2010), [9] CDC (2012), [10] Health Canada (2012), [11] M. Kasper-Sonnenberg (pers. communication), [12] E. Den Hond (pers. communication), [13] A. Gutleb (pers. communication), [14] Frederiksen et al. (2013).

Study	Gender	Age	Sampling	W (kg)	$\dot{V}_{urine}$ (ml/day)	spec. $\dot{V}_{urine}$ (ml/kg/day)	Reference
German ESB	MF	20–30 yr	24hU	72	1 790	25	[1]
Duisburg BCS	F	29–49 yr	MU	71	1 200	17	[11, 3]
Duisburg BCS	MF	6–8 yr	MU	24	600	25	[11, 3]
DEMOCOPHES SE	F	28-46 yr	?	70	1 200	17	[4, 3]
DEMOCOPHES SE	MF	6-11 yr	?	27	600	22	[4, 3]
DEMOCOPHES LU	F	33-44 yr	MU	65	1 200	17	[13, 3]
DEMOCOPHES LU	MF	6-11 yr	MU	29	600	22	[13, 3]
DEMOCOPHES DK	F	31-52 yr	MU	67	1 200	17	[14, 3]
DEMOCOPHES DK	MF	6-11 yr	MU	31	600	22	[14, 3]
DEMOCOPHES SI	F	??-?? yr	?	70	1 200	17	[4, 3]
DEMOCOPHES SI	MF	6-11 yr	?	27	600	22	[4, 3]
DEMOCOPHES BE	F	≤45 yr	?	70	1 200	17	[4, 3]
DEMOCOPHES BE	MF	6–11 yr	?	27	600	22	[4, 3]
GerES IV	MF	3–5 yr	MU	16	475	30	[4, 3]
GerES IV	MF	6–8 yr	MU	24	580	25	[4, 3]
GerES IV	MF	9–11 yr	MU	34	700	21	[4, 3]
GerES IV	MF	12–14 yr	MU	49	1 000	20	[4, 3]
Munich Infants	MF	1 mo	?	4	194	48	[3, 5]
Munich Infants	MF	2 mo	?	5	237	48	[3, 5]
Generation R	pregnant F	18–41 yr	SU	74	2 000	27	[6]
MoBa	pregnant F		SU	74	2 000	27	[6]
Flemish HMB	MF	14–16 yr	SU	57	1 200	19	[12, 3]
Liege HMB	MF	7–11 yr	MU	34	600	18	[2, 3]
Liege HMB	MF	12–19 yr	MU	65	1 200	19	[2, 3]
Liege HMB	MF	20–39 yr	MU	75	1 400	19	[2, 3]
Liege HMB	MF	40–59 yr	MU	79	1 400	18	[2, 3]
Liege HMB	MF	60–75 yr	MU	78	1 400	18	[2, 3]
INMA	pregnant F	17–43 yr	SU	74	2 000	27	[6]
INMA	MF	4 yr	SU	18	475	26	[2, 3]
Elfe pilot study	parturient F		SU	74	2 000	27	[6]
InCHIANTI	MF	20–40 yr	24hU	70	1 530	22	[7, 8]
InCHIANTI	MF	41–65 yr	24hU	70	1 690	24	[7, 8]
InCHIANTI	MF	66–74 yr	24hU	70	1 540	22	[7, 8]
NHANES	MF	6–>65 yr	SU	29–83	600–1 400	17–21	[9, 3]
CHMS	MF	6–79 yr	SU	33–80	650–1 400	18–19	[10, 3]

The results for daily BPA exposure for the European studies and for the North-American surveys (NHANES, CHMS) are shown in Figure 5. The data were grouped by the age classes as defined in Chapter 4.4 on food consumption. Age-specific estimates were available for all age classes except the 1–3 year old toddlers. As no data are available for this age group, an estimate was derived by extrapolation from 3–5 year old children to be able to make a comparison with the modelled estimate.

The GM and P50 values for average daily BPA exposure (as derived from volume-based BPA concentrations) are in good agreement among the European studies (Figure 5). Age classes with a relatively large coverage of European countries such as the children and adults, indicate a notable variability across the countries with the lowest exposures in Sweden (DEMOCOPHES SE) and Slovenia (DEMOCOPHES SI), and elevated exposures in Italy (InCHIANTI), Germany (GerES IV), and Spain (INMA). The Panel noted that the urine collection periods cover a wide range from 1998–2000 (InCHIANTI) to 2011–2012 (DEMOCOPHES).

For the infants, only two studies are available with BPA exposure data of 20 ng/kg bw/day for 7–44 day old newborns (US study at Johns Hopkins Hospital) and of <10 ng/kg BW/day (P95: 107–164 ng/kg BW/day) for 1–2 month old infants (Munich infants study). For the children, there is a tendency to higher values in younger (3–5 year old) children (107 ng/kg bw/day) compared to older (5–10 years old) ones (58 ng/kg bw/day). In teenagers and adults, the estimated daily BPA exposure is lower for both groups, at 49 ng/kg bw/day. For the elderly, only sparse data are available from the Liege HBM study (23 subjects in that age class) with a daily BPA uptake of 40 ng/kg bw/day, and from the InCHIANTI study with an uptake of 73 ng/kg bw/day. Essentially no data are available for the very elderly (≥75 years). In comparison to the North American surveys, the European data for the children, teenagers, and adults appear to be more similar to the NHANES data than to the CHMS data. Table 28 summarises the age-specific daily BPA exposures which are used as estimates of average BPA exposure.

To obtain estimates for high BPA exposure, the reported 95th percentiles from the different studies were used. The estimates for high BPA exposure were 136 ng/kg bw/day for infants, 676 ng/kg BW/day for 3–5 years old children, 311 ng/kg BW/day for 5–10 years old children, 225 ng/kg bw/day for the teenagers, 234 ng/kg bw/day for the adults, and 203 ng/kg bw/day for the (very) elderly (see Table 28). It should be noted that, apart from using the study-specific 95th percentiles, the mean P95-to-P50 ratio of 5.5 (as obtained by averaging over all spot-urine and morning-urine data shown in Figure 5) could be multiplied by the average BPA exposures to obtain estimates for the high BPA exposure.

Table 28: Daily BPA exposure as estimated from urinary BPA levels in different European studies. Estimates of the average and high daily BPA exposure were calculated from the geometric means and 95th percentiles of the volume-based urinary concentrations of total BPA. For each age class, the minimum, median, and maximum was obtained from the data available in each age class. Studies with multiple subgroups per age class were merged by calculating the mean of the geometric means and the 95th percentiles and by summing up the sample sizes of the subgroups. The number of studies and the sample-size range of participants is given for each age class.

Age class	Age (years)	No of studies	Sample size	Average daily exposure (ng/kg bw/day)		
				Minimum	Median	Maximum
Infants	0–1	2	12–88	<10	n/a	20
Toddlers	1–3	0	n/a	n/a	n/a	n/a
Children	3–5	2	30–137	105	107	109
Children	5–10	8	21–152	33	49	67
Teenagers	10–18	3	22–317	47	48	55
Adults	18–65	13	45–569	13	39	95
Elderly	65–75	2	23–452	40	57	73
Very Elderly	≥75	0	n/a	n/a	n/a	n/a

Age class	Age (years)	No of studies	Sample size	High daily exposure (ng/kg bw/day)		
				Minimum	Median	Maximum
Infants	0–1	1	88	n/a	136	n/a
Toddlers	1–3	0	n/a	n/a	n/a	n/a
Children	3–5	1	137	n/a	676	n/a
Children	5–10	6	60–152	118	204	380
Teenagers	10–18	2	197–317	200	228	256
Adults	18–65	8	60–569	85	184	291
Elderly	65–75	1	452	n/a	203	n/a
Very Elderly	≥75	0	n/a	n/a	n/a	n/a

n/a: not available

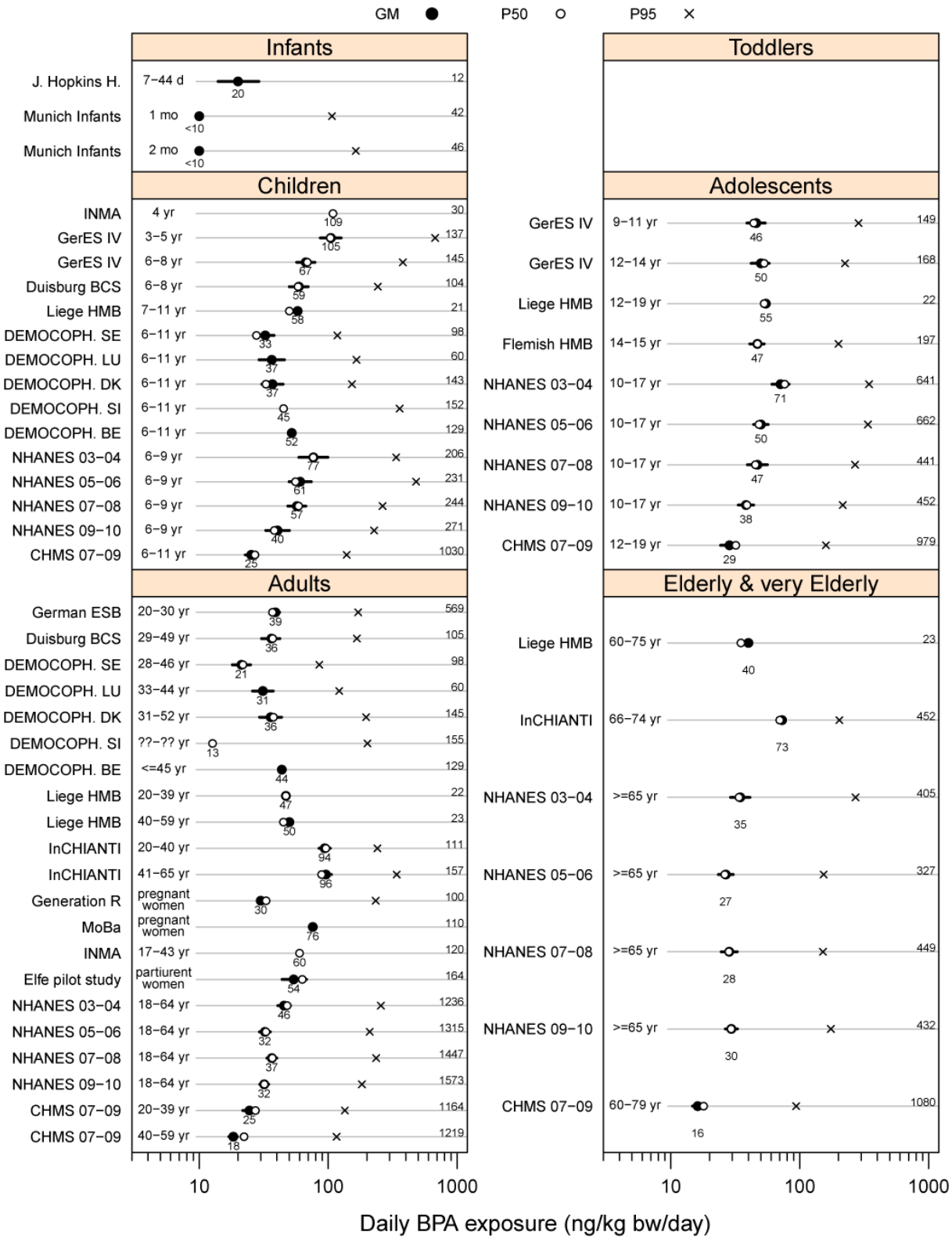


Figure 5: Daily BPA exposure as estimated from volume-based urinary BPA concentrations. The age-specific estimates for daily BPA exposure from the different studies are grouped by the age classes as defined in Chapter 4.4 on food consumption. Filled circles with associated numbers and error bars indicate the geometric means and the 95th percentile confidence intervals. The 50th and 95th percentiles are shown by open circles and crosses. The number (n) of subjects is given on the right. Age ranges and specific population groups (pregnant and parturient women) are indicated. The

studies comprise the European studies, large-sized population-based surveys from North America (NHANES, CHMS), and the US study from the Johns Hopkins Hospital on newborns.

Estimation of daily BPA exposure from creatinine-based urinary BPA concentration

The estimation of daily BPA exposure from creatinine-based urinary BPA concentrations lead to slightly different values than those obtained from volume-based urinary BPA concentrations (see Appendix VII). For the few European studies providing information on creatinine-based BPA levels, there is a tendency for lower BPA exposures in children, teenagers and adults, and a tendency for slightly higher exposures for the (very) elderly. These differences are (at least partly) explainable by daily urinary output rates that deviate from the generic values from literature. For the derivation of reference values for the comparison with BPA uptake via food and non-food resources, the volume-based BPA exposures will be used because these are better supported by a larger number of European studies.

4.8.3. Biomonitoring studies on serum levels

Methodological aspects

The detectability and concentration range of serum BPA is one of the most controversially discussed topics in the scientific literature on BPA (Dekant and Völkel, 2008; Vandenberg et al., 2010; Hengstler et al., 2011; Teeguarden et al., 2012; vom Saal et al., 2012; Vandenberg et al., 2013). In order to set the background for the assessment of human biomonitoring studies on serum BPA levels, the principal findings from the available toxicokinetic studies in humans and non human primates are briefly summarised in the following paragraphs.

In the few toxicokinetic studies in humans (Völkel et al., 2002) and rhesus monkeys (Doerge et al., 2010a; Taylor et al., 2011; Patterson et al., 2013), stable isotope-labelled BPA (deuterated) was administered to avoid any interference by possible contamination of samples with free BPA from environmental sources and medical devices. The administration of oral or intravenous doses of 64–400 µg/kg bw resulted in a transient increase in the serum concentrations of conjugated and total BPA up to 34–190 µg/l within the first hour (Figure 6), which was then followed by an approximately linear decrease (on a log-transformed scale) during the next hours. Unconjugated BPA was not detectable in the study by Völkel et al. (2002), because of the relatively high LOD, but was quantifiable in the three other studies in concentrations being 0.2–2.8 % (oral administration) and 8–29 % (intravenous injection) of the total BPA concentration during the first 4 h after dosing. In case of oral administration, the maximum levels of unconjugated BPA in serum did not exceed 1 and 4 µg/l at doses of 100 and 400 µg/kg bw, respectively (Doerge et al., 2010a; Taylor et al. 2011; Patterson et al., 2013). After intravenous injection of 100 µg/kg bw, however, much higher maximum levels of 34–39 µg/l were observed (Doerge et al., 2010a; Patterson et al., 2013).

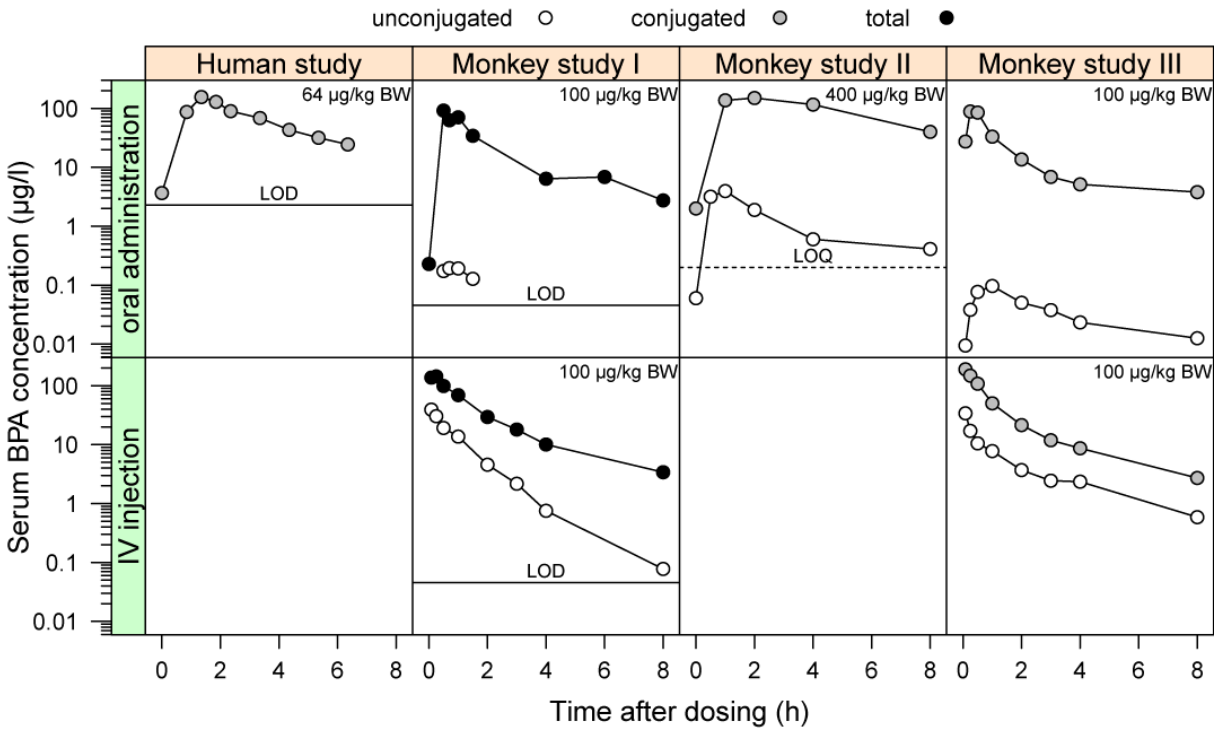


Figure 6: Time course of serum levels of unconjugated, conjugated and total BPA in toxicokinetic studies in adult humans and monkeys with oral administration and intravenous (IV) injection of isotope-labelled (deuterated) BPA. The serum concentrations of BPA are expressed as µg/l of unconjugated BPA. Solid and dashed horizontal lines indicate the LOD and LOQ, respectively. Data shown in the columns from left to right were taken from Völkel et al. (2002), Doerge et al. (2010a), Taylor et al. (2011) and (Patterson et al., 2013) with the applied dose given in each column.

Biomonitoring studies on urinary BPA levels have indicated average-to-high daily BPA uptakes in the general population of 39–676 ng/kg bw/day (see medians in Table 28), which are 2–3 orders of magnitude lower than the doses administered in the toxicokinetic studies mentioned above. Provided that this daily uptake is mainly food-related, and knowing that the kinetics are linear up to a dose of 100 000 µg/kg bw (Taylor et al., 2011), the Panel noted that even peak serum concentrations would be expected to be below 0.1 µg/l for the toxicologically relevant, unconjugated BPA. The Panel considered that detection of such low concentrations of unconjugated BPA without interferences from contamination is an analytical challenge. However, a significant uptake through the dermal route would increase the proportion of unconjugated BPA in the total BPA serum concentration, so that higher peak serum concentrations of unconjugated BPA are to be expected. In a general population having average-to-high daily BPA uptakes of 50–1 000 ng/kg bw/day, serum concentrations of conjugated or total BPA would only infrequently be expected to exceed a level of 1 µg/l.

These predictions are supported by the findings of a controlled exposure study, in which 24-hour urine and serum profiles of total BPA were measured in 20 human volunteers who ingested 100 % of one of three specified meals comprising standard grocery store food items for breakfast, lunch, and dinner (Teeguarden et al., 2011). The diet was rich in canned foods and juices to represent a potentially high BPA dietary exposure. Only 6 out of 20 subjects (i.e. 30 %) showed consistently detectable serum concentrations of total BPA within a few hours after food uptake (Figure 7). The individual peak serum concentrations in this subset of volunteers ranged from 0.6 to 1.3 µg/l and occurred within 2–3 hours after food consumption. These transient elevations of serum levels were associated with inter-meal urinary BPA excretion of 183–573 ng/kg bw. Overall, total BPA was detected in 27 % of the 320 serum samples collected from the 20 volunteers. The concentration of unconjugated BPA was always below the LOD of 0.3 µg/l. Comparing the derived doses and the detectable maximum concentrations

of total BPA of Teeguarden et al. (2011) with those of Völkel et al. (2002) suggests conformity with the assumption of linearity of BPA kinetics and its conjugated metabolites.

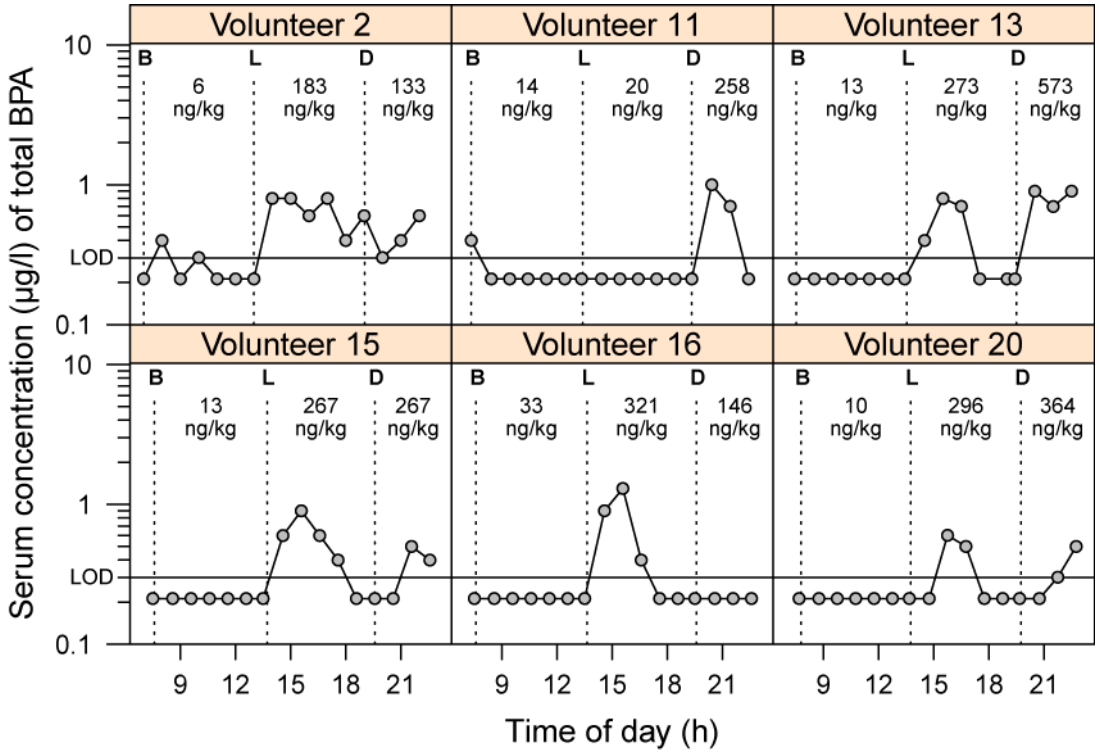


Figure 7: Time course of total BPA serum concentration in human volunteers ingesting a controlled diet enriched with canned food. Shown are the data of a subset of volunteers (6 out of 20) with consistently detectable concentrations of total serum BPA. Total BPA concentrations below the LOD of 0.3 µg/l are set to a value of LOD/√2. Vertical dotted lines indicate the meal times (B, breakfast; L, lunch; D, dinner). The per-body-weight amount of total BPA eliminated via urinary excretion during each intermeal period is given. Data were taken from Teeguarden et al. (2011).

Serum BPA concentrations

Data on serum levels of unconjugated, conjugated, and total BPA in humans were retrieved from peer-reviewed scientific papers (published since 2006) which were identified by a systematic literature search. The analytical methods for the determination of serum BPA comprised LC-UV, LC-FLD, LC-ECD, LC-MS and LC-MS/MS, GC-MS and GC-MS/MS, and RIA (see Appendix I for method description). Of the 26 human-biomonitoring studies reporting first-publication data, one study (Sajiki et al., 2008) was excluded as no information on the proportion of values below the LOD/LOQ was available. Additionally excluded were the patient-related subsets of four studies (Cobellis et al., 2009; Kaddar et al., 2009; Yang et al., 2009; Bloom et al., 2011) and one study reporting only patient-related data (Shao et al., 2012), because patients could have been in contact with BPA-containing medical devices.

The study groups comprised the general population (Liu et al., 2006a; He et al., 2009; Kaddar et al., 2009; Liao and Kannan, 2012a) as well as specific age classes such as children (Ye et al., 2012), teenagers (Geens et al., 2009b), adults (Fukata et al., 2006; Dirtu et al., 2008; Genuis et al., 2012; Santhi et al., 2012a), and seniors (Olsen et al., 2012). Additional data were available for more specific demographic groups such as students (Koch et al., 2012), male partners of female patients undergoing *in vitro* fertilisation (IVF) (Bloom et al., 2011), healthy women (Cobellis et al., 2009), female hospital controls (Yang et al., 2009), nursing women (Gyllenhammar et al., 2012), and pregnant women (Lee

et al., 2008; Padmanabhan et al., 2008; Wan et al., 2010; Chou et al., 2011; Kosarac et al., 2012; Unal et al., 2012). Also analysed were blood-bank samples (Ye et al., 2008b, 2009b).

Because of the large number of studies on pregnant women, and also taking account of the terms of reference to consider specifically this group (amongst others), this demographic group is considered separately from the remaining general population.

For the assessment of reported serum BPA levels, the following aspects were specifically assessed:

- the proportion of detectable/quantifiable values in relation to the LOD/LOQ
- the proportion of unconjugated BPA in the total BPA serum concentration
- the average serum concentrations of unconjugated (U), conjugated (C) and total (T) BPA for studies reporting ≥50 % detectable values.

To provide an overview of the study results, a Cleveland dot plot was used to visualise the average serum BPA concentrations and the proportions of detectable values (Figure 8). Pie charts displaying the proportion of detectable values were positioned at the respective LOD/LOQ of the study, and the average serum BPA concentrations (small geometric symbols) are shown for studies reporting ≥50 % detectable values. For symbols and pie charts, gray and black filling colours were used for unconjugated BPA and conjugated/total BPA, respectively. The serum concentrations of unconjugated, conjugated and total BPA combined are expressed in µg/l of unconjugated BPA.

To show the influence of decreasing analytical limits on the proportion of detectable BPA levels, the studies were ordered according to their LOD/LOQ, and the pie charts displaying the proportion of detectable values were positioned at the respective analytical limit (Figure 8). Some of the studies report an LOD, some of them an LOQ, and some report both LOD and LOQ. In the latter case, only that analytical limit was displayed which the study authors considered as censoring limit for reportable and non reportable concentrations. Across the different studies, the analytical limit for detecting the different BPA parameters (i.e. unconjugated, conjugated and total BPA concentrations) varied by almost two orders of magnitude (0.01–0.82 µg/l). In spite of this large variation in analytical sensitivity, the Panel noted that a consistent pattern such as an increasing proportion of detectable values with decreasing LOD/LOQ did not emerge. Overall, the detection rate for unconjugated and conjugated and/or total BPA varied largely from 0 % to 100 %. Given the findings of the controlled exposure study in human volunteers (Teeguarden et al., 2011), with unconjugated BPA being undetectable and total BPA being detectable in only 27 % of the 320 serum samples collected from the 20 volunteers, the Panel considered detection rates close to 100 % for conjugated and/or total BPA in serum, as an implausible result. High detection rates for unconjugated BPA in serum are even more implausible.

Only a few studies provide information on more than one serum BPA parameter (i.e. unconjugated, conjugated and total BPA). These studies were used to determine the proportion of unconjugated BPA in the total BPA concentration, where both unconjugated and total BPA were detectable and quantifiable in the same sample. Gyllenhammar et al. (2012) reported detection rates of 25 % and 21 % (at slightly different LODs of 0.5 and 0.8 µg/l) for unconjugated and total BPA, respectively. In 15 % of the samples, the authors reported that unconjugated BPA could be detected and accounted for one half to all of the total BPA. Ye et al. (2008b) reported unconjugated and total BPA in only one of 15 blood-bank samples at a similar concentration of 1.5 µg/l (i.e. all BPA present was in the unconjugated form). Koch et al. (2012) quantified both unconjugated and total BPA in only 7 of 60 plasma samples, reporting that unconjugated BPA accounted for the predominant share (90–100 %) of total BPA. Similarly, Ye et al. (2012) detected total BPA in only 3 of 24 pooled serum samples, and unconjugated BPA in 2 pooled samples only. The mean percentage of unconjugated BPA in samples with detectable total BPA was 67 %. Kosarac et al. (2012) reported detection rates of 67 % and 17 %

for unconjugated and conjugated BPA, respectively, again implying that serum BPA was essentially unconjugated.

The findings of these authors appear to indicate (i) that the detection of total BPA in a sample made the parallel detection of unconjugated BPA very likely, and (ii) that all serum BPA (if detected) was essentially unconjugated. The Panel considered that this is extremely unlikely given the findings of the toxicokinetic studies mentioned above, in which stable isotope-labelled BPA (deuterated) was administered to avoid any interference by possible contamination of samples with free BPA from environmental sources and medical devices.

Although also providing information on more than one serum BPA parameter (i.e. unconjugated, conjugated and total BPA), the study by Liao and Kannan (2012a) is notable for the fact that serum concentrations of unconjugated and conjugated (sulfated, glucuronidated) were directly measured *via* solid-phase extraction (SPE) and LC-MS/MS. The LODs of 0.01 µg/l for unconjugated BPA and 0.05 µg/l for conjugated BPA were the lowest reported for all studies reviewed in this opinion (Figure 8). Unconjugated, sulfated and glucuronidated BPA were detected in 75 %, 50 % and 50 % of the samples with geometric means of 0.035 µg/l, 0.065 µg/l and 0.115 µg/l (all concentrations values expressed in terms of unconjugated BPA). Based on these geometric mean concentrations, unconjugated BPA accounted for only 16 % of total BPA. It should be noted that the authors also analysed the serum samples by enzymatic deconjugation and liquid-liquid extraction (LLE) for the determination of total BPA. Using this method, unconjugated and total BPA were both detected in 100 % of the samples with geometric means of 0.049 µg/l and 0.075 µg/l. The geometric mean of 0.049 µg/l for unconjugated BPA (as obtained by LLE but without enzymatic deconjugation) agreed well with the 0.035 µg/l as obtained by SPE. The value of 0.075 µg/l for total BPA (as obtained by LLE with enzymatic deconjugation) was, however, considerably lower than would be expected from the sum of the SPE-derived concentrations for unconjugated and conjugated BPA forms.

Of the remaining studies not involving pregnant women, five studies (Dirtu et al., 2008; Kaddar et al., 2009; Yang et al., 2009; Bloom et al., 2011; Olsen et al., 2012) report detection rates of ≥50 % for unconjugated and total BPA and provide statistically feasible descriptive statistics with median concentrations up to 3.8 µg/l (Figure 8, upper panel). The results of two of these studies are presented below as examples.

Olsen et al. (2012) studied the serum concentration of total BPA in 1 016 seniors (all aged 70 years old) living in the community of Uppsala, Sweden. Blood samples were collected in the morning after overnight fast. Total BPA was detected in 98 % of the samples (LOD: 0.2 µg/l) with a median concentration of 3.8 µg/l. Assuming, as a rough calculation, a blood volume of 5 litre, a serum fraction of 0.55, and a body weight of 70 kg, this median concentration would translate into an *instantaneous* body burden of 150 ng/kg bw, the amount of BPA distributed among the other tissues not yet included. Given the large sample size, it could be concluded from these data that half of the Uppsala senior population has an *instantaneous* body burden of higher than 150 ng/kg bw in the morning after an overnight fast. However, taking into account the average-to-high *daily* BPA uptake among the elderly of 60–200 ng/kg bw/day as estimated from biomonitoring studies on urinary BPA, the Panel found it difficult to envisage a community-wide exposure scenario which could lead to such a high BPA body burden already in the morning after an overnight fast.

As a second example, Bloom et al. (2011) studied the serum concentration of unconjugated BPA in 27 couples undergoing *in vitro* fertilisation (IVF). On the day of oocyte retrieval, fasting and nonfasting blood specimens were collected from female patients and male partners, respectively. Unconjugated BPA was detected in 85 % (women) and 52 % (men) of the samples (LOD: 0.3 µg/l) with median concentrations of 3.3 µg/l (women) and 0.48 µg/l (men). The high serum concentration in the women will not be further discussed here as the female patients could have been in contact with BPA-containing medical devices. For the male partners, however, a simple back calculation can be used to put their serum concentrations into perspective. According to commonly accepted kinetic concepts, the following equation (Renwick, 2008; Mielke and Gundert-Remy, 2009) can be used to calculate the

dose rate D (ng/kg bw/h) from the steady-state serum concentration C_{ss} (µg/l), the serum clearance Cl (L/h), the fraction absorbed f_a, and the body weight bw (kg):

$$D = \frac{C_{ss} \times Cl}{f_a \times bw}.$$

An estimate for the serum clearance (Cl) of 80 L/h for a 70-kg human can be derived from the allometric scaling relationship provided by Doerge et al. (2012). Assuming a steady-state concentration (C_{ss}) of 0.48 µg/l, a body weight (bw) of 70 kg, and a fraction (f_a) of 0.3 of systemically available BPA (e.g. 30 % bioavailability *via* the dermal route), the calculation yields a dose rate (D) of 1 800 ng/kg bw/h. In other words, to sustain a steady-state serum concentration (C_{ss}) of 0.48 µg/l over a period of say 1 h would require a continuous external exposure of 1 800 ng/kg bw/h. According to Bloom et al. (2011), half of the male participants had serum concentrations of unconjugated BPA of 0.48 µg/l or higher under nonfasting conditions. Again, the Panel considered that it is very difficult to envisage a realistic exposure scenario that would lead to exposures equal to or exceeding 1 800 ng/kg bw per hour and even per day.

Given the unrealistic exposure implications for reported serum BPA concentrations in the µg/l range, the Panel considered that it is difficult to explain the high detection rates and the average concentrations of unconjugated and total BPA in the serum of pregnant women (Figure 8). As already discussed elsewhere (Koch et al., 2012), these results may be due to methodological differences in terms of detection technique (selectivity), LOD/LOQ (sensitivity), and within-laboratory and pre-analytical blank issues causing such results, but this can only be a matter of speculation.

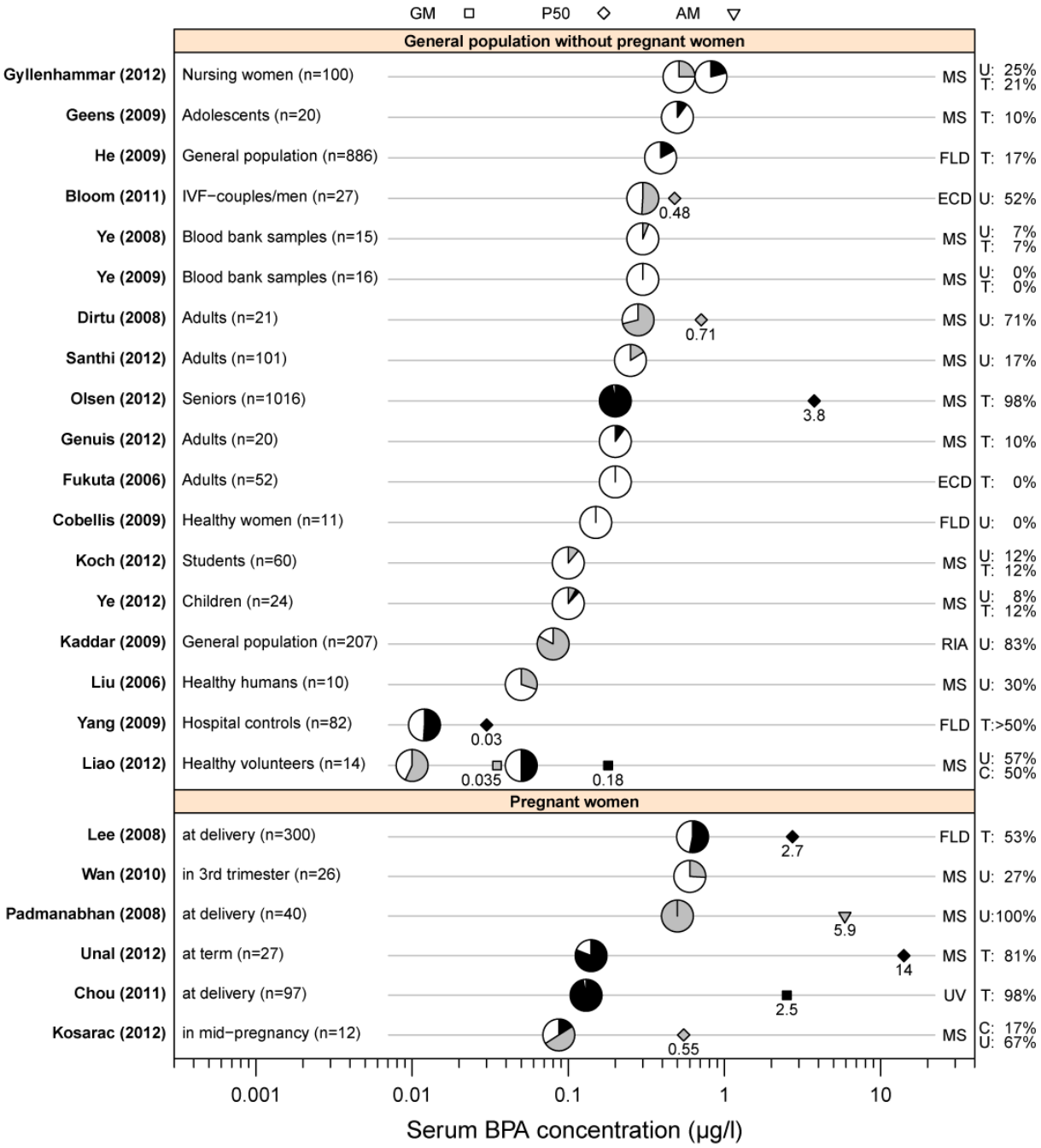


Figure 8: Cleveland dot plot showing the average serum BPA concentrations (small geometrical symbols) and the proportions of detectable/quantifiable values (pie charts). Pie charts displaying the proportion of detectable/quantifiable values were positioned at the respective LOD/LOQ. A gray filling colour is used for unconjugated (U) BPA, whereas black filling colour is used for conjugated (C) and total (T) BPA. Average serum concentrations are only shown for studies reporting ≥50 % detects. The different geometrical symbols indicate the geometric mean (squares), the median (diamonds), and the arithmetic mean (triangles). Information on the study groups, the number of subjects (n), the analytical method, and the percentage of detectable/quantifiable values are given. All serum concentrations are expressed in µg/l of unconjugated BPA For references, see main text.

4.8.4. Biomonitoring studies in human milk

Breastfed infants may be exposed to BPA via human milk as a consequence of exposure of the lactating mothers. BPA may occur in human milk in the unconjugated and conjugated forms by the lactational transfer from the maternal plasma compartment to the maternal milk compartment. The distribution of both BPA forms between the plasma and milk compartments may vary depending on the milk composition which changes in terms of protein and fat content within the first 3–5 days after delivery (Saint et al., 1984). Profound changes occur also in the milk concentration of sodium and chloride during the first 48 h post-partum, which are explained by the closure of tight junctions between the mammary epithelial cells that prevent plasma constituents from passing directly from the interstitial space into the milk (Neville and Walsh, 1996). It is therefore reasonable to consider initial human milk (colostrum), which is collected within the first few days after delivery, and mature human milk separately for exposure assessment. Additional arguments for a separate exposure assessment of newborns and infants receiving initial and mature human milk are (i) the three-fold higher activity of a human milk β-glucuronidase in initial milk compared to mature milk (Gourley and Arend 1986) and (ii) the possibility of a treatment-related elevated exposure of mothers staying in the hospital for a few days after delivery. The occurrence of BPA in human milk was analysed in eight small-scale studies carried out in Europe (Cariot et al., 2012), North America (Ye et al., 2006, 2008c; Duty et al., 2013) and South-East Asia (Otaka et al., 2003; Sun et al., 2004; Kuruto-Niwa et al., 2007; Yi et al., 2010).

In the study from France (Cariot et al., 2012), unconjugated BPA was quantified in initial human milk by isotope-dilution UPLC-MS/MS with a limit of detection (LOD) of 0.09 µg/l and a limit of quantification (LOQ) of 0.40 µg/l. Very much care was taken to avoid the cross-contamination by environmental BPA by using solvents and reagents of high analytical quality as well as pre-treated glassware. The milk was drawn manually and directly in pre-treated glass tubes, without any device, materials, wipes or gloves. Quality-control (QC) materials and standards were prepared from pooled human milk which derived from samples collected over several days from two donors (A. Cariot, pers. communication) who had been breastfeeding for over 1 month. Unconjugated BPA was absent in solvent blanks, and it was detected only in some of the pooled (mature) human milk used for standards and quality controls in concentrations (≤0.12 µg/l) markedly lower than the LOQ. To test the applicability of their analytical method, the authors analysed 3 samples which were collected from three donors within a few days after delivery. Unconjugated BPA was detected in all samples in concentrations of 0.80, 3.07, and 3.29 µg/l with a geometric mean of 2.0 µg/l (Figure 9). No information is available on whether the three donors stayed in the hospital and underwent medical procedures, which might have led to an additional, treatment-related non oral exposure resulting in higher-than-normal BPA levels in plasma and milk.

Initial human milk (colostrum) was also analysed by Kuruto-Niwa et al. (2007) for the presence of total BPA using an ELISA with an LOD of 0.3 µg/l. Milk samples were collected within three days after delivery from 101 healthy mothers from a local region in Japan in 2000–2001. Glass bottles were used for sample storage to avoid contamination. Total BPA was found in all 101 samples in a concentration range of 1.4–7.1 µg/l with a median of 3.0 µg/l (Figure 9). No information is available on the possible hospitalisation and medical treatment of the donors to exclude a treatment-related non oral exposure of the mothers. An additional uncertainty comes from the analytical method itself. The ELISA was originally developed for the determination of BPA in urine and proved to be sensitive to both unconjugated and glucuronidated BPA (Kodaira et al., 2000). A method comparison revealed a good correlation between ELISA and HPLC-FLD measurements of BPA in glucuronidase-treated urine samples (Kodaira et al., 2000). However, the cross-reactivity was only checked for a limited number of BPA-related compounds (Kodaira et al., 2000), so that an overestimation of BPA concentration by cross-reactivity with other structurally related compounds cannot be excluded (Dekant and Völkel, 2008; FAO/WHO, 2011; Asimakopoulos et al., 2012). Moreover, the ELISA was obviously not validated for other biological matrices such as human milk, so that the data should be interpreted with care.

The CEF Panel noted that only very few data from Europe and/or obtained by a reliable analytical method were available and therefore decided to take into account data from Japan, reporting an average BPA concentration of 3 µg/l in initial human milk. However, these data from Japan were obtained using ELISA methodology and samples dated back to 2000. These limitations were addressed in the uncertainty analysis.

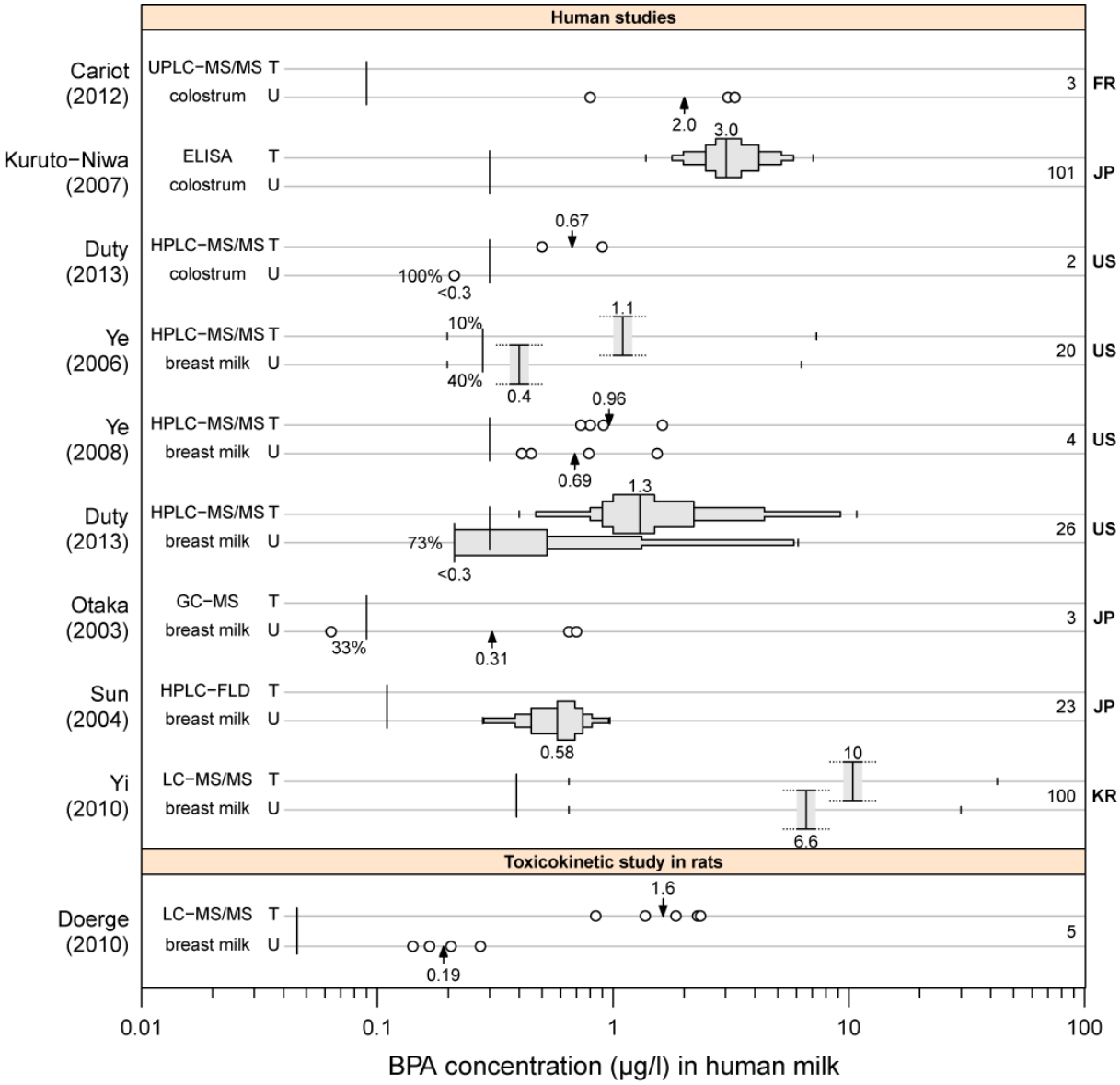


Figure 9: Summary figure of the study results on BPA in human milk. Shown are the concentrations of unconjugated (U) and total (T) BPA on a log10-transformed scale for the eight human studies and the single rat study by Doerge et al. (2010). Individual measurements (open circles) are shown for studies with small samples sizes (n < 20). For larger-scaled studies (n ≥ 20), box-percentile plots (gray-shaded boxes) are used to depict the distributional characteristics comprising the 5th, 12.5th, 25th, 37.5th, 50th, 62.5th, 75th, 87.5th and 95th percentiles, the median (vertical line within the boxes), and the minimum and maximum values (tick marks). Data from studies reporting only the median and the range are shown as incomplete boxplots. Vertical lines indicate the LOD. Concentrations below the LOD are set to a value of LOD/√2. Numbers associated with the data represent either the median value (larger-scaled studies) or the geometric mean (small-scale studies). The number of subjects and the country codes are shown on the right. All concentrations are expressed as µg/l of unconjugated BPA.

Three studies in the US quantified unconjugated and total BPA in human milk samples by isotope-dilution HPLC-MS/MS with an LOD of 0.3 µg/l (Ye et al., 2006, 2008c; Duty et al., 2013). QC materials for milk blanks were prepared by pooling human milk samples either taken from multiple donors (Ye et al., 2006) or purchased from Mother's Milk Bank between 2002–2003 (Ye et al., 2008c). In the first study, Ye et al. (2006) analysed 20 human milk samples from a group of lactating women without known occupational exposure. Unconjugated BPA was detected in 60 % of the samples with a median of 0.4 µg/l and a maximum of 6.3 µg/l (Figure 9). Total BPA was detected in 90 % of the samples with a median of 1.1 µg/l and a maximum of 7.3 µg/l. Comparison of the median concentrations of unconjugated and total BPA yielded a proportion of unconjugated BPA of 36 %. In the second study, Ye et al. (2008c) analysed milk samples of 4 donors only. The unconjugated and total BPA concentrations were in the range of 0.41–1.54 µg/l and 0.73–1.62 µg/l (Figure 9), respectively. The proportion of unconjugated BPA in the individual samples was quite high (50–99 %), and the authors (Ye et al., 2008c) acknowledged that they could not rule out the potential for contamination as information on the collection and storage of these four samples was not available.

In the third US study, Duty et al. (2013) analysed milk samples of 30 mothers with premature infants in a neonatal intensive care unit. Sample collection devices were pre-screened for BPA, and maternal milk was expressed by mechanical pumping and frozen in BPA-free storage containers. BPA-free breastpump disposable devices were made available to the mothers, however, the use of different systems by some mothers could not be excluded. The analytical measurements were performed by the same lab as in the other two US studies. Two human milk samples with concentrations of total BPA (222 and 296 µg/l) and unconjugated BPA (189 and 252 µg/l) were excluded as statistical outliers by the authors. Of the remaining 28 samples, two samples were collected from mothers within 3–5 days after delivery (S. Duty, pers. communication). The concentrations of unconjugated and total BPA in these 2 colostrum samples were <0.3 µg/l (i.e. below the LOD) and 0.67 µg/l (geometric mean), respectively. The remaining 26 mature-milk samples had median concentrations of <0.3 µg/l (unconjugated BPA) and 1.3 µg/l (total BPA) with unconjugated BPA accounting for less than 30 % (median value) of total BPA. Remarkably, the box-percentile plots for unconjugated and total BPA (Figure 9, percentiles kindly provided by S. Duty) revealed quite a large variability, which appears to be driven by unconjugated BPA. This variability may be related to the different exposures in the hospital and home environments.

The three remaining studies on BPA in human milk were carried out in Japan (Otaka et al., 2003; Sun et al., 2004) and South Korea (Yi et al., 2010). Otaka et al. (2003) analysed unconjugated BPA in human milk provided by 3 different volunteers and used GC-MS with an LOD of 0.09 µg/l and an LOQ of 0.21 µg/l. The authors reported the absence of contamination of reagents and materials as well as blank BPA concentrations below the LOD. Unconjugated BPA was detectable in two of the three samples in concentrations of 0.65 and 0.70 µg/l (Figure 9). Sun et al. (2004) used HPLC-FLD with an LOD of 0.11 µg/l to measure unconjugated BPA in samples from 23 healthy, primiparous and multiparous women. Glass tubes were used to avoid contamination. Unconjugated BPA was detected in all samples in a concentration range of 0.28–0.97 µg/l with a median of 0.61 µg/l (Figure 9). The last study by Yi et al. (2010) used LC-MS/MS and HPLC-FLD with an LOD of 0.39 µg/l and 0.6 µg/l, respectively, to measure unconjugated and total BPA. Milk samples were collected from 100 volunteers who had delivered within two weeks. The study revealed a substantial disagreement between the two analytical methods. Unconjugated BPA, for example, was detectable in all samples by LC-MS/MS but completely undetectable by HPLC-FLD, which led the authors to suspect an overestimation of BPA by LC-MS/MS in the lower concentrations range. In the high concentration range, the method disagreement was explained by poor resolution of HPLC-FLD. The median concentration of unconjugated and total BPA, as measured by LC-MS/MS, was 6.6 µg/l and 10 µg/l (Figure 9). Such high values were not found in other studies on total BPA in human milk and could possibly reflect a population-specific, elevated exposure to BPA.

To put the data on BPA in human milk in perspective, the results from animal studies should be taken into consideration. Valuable information on the lactational transfer of BPA and on the relative proportion of unconjugated BPA in animal milk is available from a controlled study in rats (Doerge et

al., 2010a), where dams were administered a daily oral dose of 100 µg/kg bw of stable isotope-labelled BPA. The isotope-labelled BPA was used to avoid contamination problems, and the dose was selected to be within the linear pharmacokinetic range at a level as close as possible to the range of proposed human exposure, yet high enough to measure both BPA forms (Doerge et al., 2010a). The analysis of milk samples, which were collected on day 7 postpartum at 1 h after dosing when BPA serum levels are maximal (Doerge et al., 2010b), revealed median concentrations of 0.19 µg/l and 1.6 µg/l for unconjugated and total BPA, respectively (Figure 9). The proportion of unconjugated BPA in the individual samples was low (8.7–12 %). So for an oral dose of 100 µg/kg bw, which is very high for humans, the median concentration of total BPA in rat milk is, unexpectedly, in the same order of magnitude as those in human milk. Physiological differences between rat and human cannot be excluded. For unconjugated BPA, the median concentration is an order of magnitude lower in rat milk compared to those reported for initial human milk (colostrum). Finally, the proportion of unconjugated BPA in rat milk is markedly lower than the reported proportions of <30 % (Duty et al., 2013), ~36 % (Ye et al., 2006), and 50–99 % (Ye et al., 2008c) for mature human milk.

To conclude, although anti-contamination measures have been taken during sample work-up and the analytical procedure, the issue of potential contamination during the collection and storage of human milk samples is not completely solved. Even if the collection procedure is under strict control, an uncertainty about a possible hospitalisation and medical treatment-related non oral exposure of the mothers remains. The measurement of only unconjugated BPA introduces an additional uncertainty about the concentration of conjugated BPA which should be taken into consideration in the exposure assessment. Given the presence of intestinal β-glucuronidases of bacterial origin in rats (Koldovsky et al., 1972; Rød and Midtvedt, 1977; Gadelle et al., 1985) and of a β-glucuronidase in human milk (Gaffney et al., 1986; Gourley and Arend 1986; Grazioso and Buescher 1996), one may expect a glucuronidase activity in the infant gut which may lead to a deconjugation of ingested glucuronidated BPA. There are several possible reasons why the proportions of unconjugated and conjugated BPA in human milk may vary. The first is the changing protein/fat composition of human milk within the first few days after delivery (Saint et al., 1984), which could affect the blood-to-milk transfer. The second is the presence of a β-glucuronidase in human milk. A third possibility is the maternal exposures via non oral routes which, for toxicokinetic reasons, may result in higher plasma fractions of unconjugated BPA.

Given the uncertainty and scarcity of the human milk data, a pragmatic approach to assess the exposure to BPA for breastfed newborns and infants could be a scenario based on unconjugated and total BPA in human milk that not only covers the lactational transfer of maternal BPA but also contributions of external BPA from collecting devices (e.g. breast milk pumps) and storage containers. To cover both average and high exposures, estimates of the central tendency and of an upper bound level should be derived. Estimates of the central tendency were obtained from all human studies except the study by Yi et al. (2010) (Table 29). For initial human milk, the average concentration of 3 µg/l for total BPA was taken from Kuruto-Niwa et al. (2007) as a conservative estimate, being aware that this relatively high estimate is not supported by the two observations from the study of Duty et al. (2013). For unconjugated BPA, the value of 2 µg/l from Cariot et al. (2012) was regarded as not reliable enough because of the very small sample size (n = 3) and of the lacking support from the study of Duty et al. (2013) in which unconjugated BPA was undetectable in the two initial-milk samples. Therefore, only an average concentration estimate for total BPA in initial human milk is provided (Table 29). For unconjugated BPA in mature human milk, a sample size-weighted mean of 0.4 µg/l was calculated from the moderately-sized studies of Ye et al. (2008c), Duty et al. (2013) and Sun et al. (2004). For total BPA in mature human milk, an estimate for the average concentration of 1.2 µg/l was taken from the moderately-sized studies of Ye et al. (2008c) and Duty et al. (2013). Table 29 summarises the derived estimates for the average concentration of unconjugated and total BPA.

Estimates for high exposures were derived from the interquartil range (IRQ) of the moderately-sized datasets for total BPA (Kuruto-Niwa et al., 2007; Duty et al., 2013) and unconjugated BPA (Sun et al., 2004). By noting that the $\log_{10}$-transformed BPA concentrations approximately follow a normal distribution (Figure 9), and that the standard deviation (σ) of a normal distribution is related to the IRQ

by IRQ = 1.35×σ, individual estimates for σ of 0.17, 0.30, and 0.16 could be derived for the three selected datasets. These individual estimates yielded an average σ of 0.21 on the $\log_{10}$-transformed scale. Naive 95 % one-sided confidence intervals were finally obtained by calculating a factor, $k = 10^{1.64\times\sigma} = 2.2$, which was then multiplied with the average BPA concentrations. The derived estimates for the high BPA exposure are given in Table 30.

Table 29: Database of average BPA concentrations (µg/l) in human milk used for exposure assessment. The average values represent either the median (larger-scale studies) or the geometric mean (small-scale studies).

Study/Author	Type of milk	No of samples	Average BPA concentration (µg/l) unconjugated	total
Cariot et al. (2012)	initial	3	2.0	n/a
Kuruto-Niwa et al. (2007)	initial	101	n/a	3.0
Duty et al. (2013)	initial	2	<0.3	0.7
Ye et al. (2006)	mature	20	0.4	1.1
Ye et al. (2008c)	mature	4	0.7	1.0
Duty et al. (2013)	mature	26	<0.3	1.3
Otaka et al. (2003)	mature	3	0.3	n/a
Sun et al. (2004)	mature	23	0.6	n/a

n/a: not available

Table 30: Average and high values used (µg/l) to estimate exposure to BPA from human milk.

Type of milk	BPA concentration (µg/l) unconjugated average	unconjugated high	total average	total high
initial	n/a	n/a	3.0	6.6
mature	0.4	0.9	1.2	2.6

n/a: not available

In the 2006 opinion, EFSA used a concentration of unconjugated BPA of <1.0 µg/l in human milk as a conservative estimate of potential dietary exposure to BPA.

In conclusion, the estimates for the average and high concentration of unconjugated and total BPA in mature human milk are supported by several small to medium-sized studies. In contrast, reliable estimates for initial human milk could not be derived because of the discrepancies between the studies and the low sample sizes in some of the studies. Nonetheless the average concentration of 3 µg/l for total BPA was taken from Kuruto-Niwa et al. (2007) as a conservative estimate, while being aware that this study has limitations and that this relatively high estimate is not supported by the two observations from the study of Duty et al. (2013).

The uncertainty arising from the unreliable estimates for initial human milk is further increased by the fact that milk production during the first five days is of a transitional character. The milk production rate increases more or less linearly during the first days after delivery, reaching a plateau of ~600 ml/day on day 5 (Neville and Walsh, 1996). This process is accompanied by compositional changes in protein and fat content (Saint et al., 1984) and in β-glucuronidase activity (Gourley and Arend 1986), which may affect the proportion of unconjugated BPA in the concentration of total BPA of maternal origin. Last but not least, there is the possibility of an exposure from medical devices for mothers staying in the hospital for a few days after delivery.

4.9. Discussion of total exposure estimates

The current draft opinion is focused on the modelled exposure (absorbed dose) of consumers to BPA (through different routes), taking into account different absorption factors for the different routes of exposure, and on the comparison of these exposure estimates with the total daily urinary excretion of BPA, assessed by urinary biomonitoring. The opinion also systematically evaluates the uncertainty in these estimates (chapter 4.9.3). The estimates do not reflect the proportion of the BPA dose bioavailable (unconjugated BPA) after absorption by the body and subsequent metabolism. The conversion of the exposure estimates from each source into internal (bioavailable) doses of BPA has not yet been considered. This conversion into internal doses needs to be considered in the subsequent step of risk characterisation of BPA. Uncertainties affecting the parameters that will be used for this conversion are not considered in the present document but will be taken into consideration in later steps of the risk assessment of BPA.

4.9.1. Comparison with biomonitoring studies

The estimates for the average and high total exposure to BPA in the general population as obtained by the modelling approach in Chapter 4.7 (Total exposure) are compared with the biomonitoring estimates.

Comparison of average total exposure

The estimates for the average total exposure as obtained by the modelling approach and by biomonitoring approach are shown in Table 31.

For the age class 'Infants', the average total exposure as estimated by the modelling approach ranged from 38 ng/kg bw/day (formula-fed 0–6 month olds) *via* 127 ng/kg bw/day (breastfed 4–6 months olds), 143 ng/kg bw/day (breastfed 6 days to 3 months olds), 228 ng/kg bw/day (breastfed 1–5 days olds) to 383 ng/kg bw/day (6–12 months olds). The biomonitoring approach estimated the average total exposure for 1–2 months old infants to be <10–20 ng/kg bw/day, which is at least 2–4-fold lower than the modelled estimate of 38 ng/kg bw/day for formula-fed infants.

The average total exposure of toddlers was only estimated by the modelling approach as no biomonitoring data were available. The modelling approach gave an estimate of 379 ng/kg bw/day.

For the 3–10 years old children, an average total exposure of 314 ng/kg bw/day was obtained by the modelling approach. The biomonitoring approach gave estimates of 107 and 49 ng/kg bw/day for 3–5 year old children and 5–10 year old children, respectively, which were 3–6-fold lower than the figure obtained by the modelling approach.

For the teenagers, adults, and the elderly and very elderly, a decreasing trend of BPA exposure from 190 *via* 145–152 to 136 ng/kg bw/day was observed in the modelled estimates. Similarly, the biomonitoring approach indicated a decreasing trend with values of 48 and 39 ng/kg bw/day for the teenagers and adults. The somewhat higher value of 57 ng/kg bw/day for the biomonitoring data in the elderly may be biased towards higher values because of the low number of only two biomonitoring studies. Again, the biomonitoring estimates are 2–4-fold lower than those obtained by the modelling approach.

To summarise, the estimates for the average total exposure as obtained by modelling and biomonitoring methods agree with each other within an order of magnitude. More specifically, the modelling approach gave estimates which were approximately 4-fold higher (38–383 ng/kg bw/day vs. <10–107 ng/kg bw/day) than those obtained by the biomonitoring approach. There are two important aspects which may contribute to these discrepancies. The first one is the statistical procedure by which averages are derived. The second one is the scenario for modelling the dietary and non-dietary exposure.

The exposure estimation *via* modelling (ingestion, dermal and inhalation exposure) is based on the calculation of arithmetic means (AM), whereas the estimation *via* urinary biomonitoring is based on geometric means (GM). In case of biomonitoring, the decision to use GMs was justified by the log-normal distribution shape of the urinary BPA data (see Chapter 4.8.2.1). To convert GM-based estimates into AM-based estimates, which are then comparable to those obtained by the modelling approach, a multiplicative conversion factor of 1.8 was derived (see Chapter 4.8.2.1). The different statistical procedures for calculating central tendencies may at least partly explain the discrepancies between the two approaches.

The second source for the discrepancy between the two approaches could be the scenario chosen for modelling the dietary exposure. Two scenarios (with lower-bound, middle-bound, and upper-bound handling of left-censored data) were considered in the dietary exposure estimation (see Chapter 4.6.2.1). In scenario 1, only food specifically codified as canned in the dietary survey are assigned the corresponding occurrence level for BPA. In scenario 2, any food at FoodEx level 4 which has been codified as canned in at least one survey is always considered to be consumed as canned in all dietary surveys considered in the Comprehensive Database. Scenario 2 and the middle-bound approach was chosen for the total exposure estimation. As scenario 2 might overestimate the dietary exposure, this may also partly explain the discrepancies between the estimates of modelling approach and the biomonitoring approach.

An additional source of discrepancy may be related to the conservativeness of the assumptions made to assess exposure to non-food sources.

Table 31: Average total exposure to BPA as estimated by the modelling approach and by biomonitoring. For some age classes such as infants and children, several values are given which refer to subgroups among the age classes.

Age class	Age (years)	Average total exposure (ng/kg bw/day)	
		Modelling approach[(a)]	Biomonitoring[(b)]
Infants	0 – 1	38/127/143/228/383	< 10-20
Toddlers	1 – 3	379	not available
Children	3 – 10	314	49-107
Teenagers	10 – 18	190	48
Adults	18 – 65	145-146-152	39
Elderly & very elderly	≥65	136	57

(a) Total exposure assessed by adding estimated exposure from inhalation, ingestion and dermal contact (see Chapter 4.7). For some age classes several values are given which refer to sub-groups among the age class (see Table 23 for details).
(b) When biomonitoring data were available for more than one age class, several values are given.

Comparison of high total exposure

The estimates for the high total exposure as obtained by the modelling approach and by biomonitoring approach are shown in Table 32.

For the age class 'Infants', the high total exposure as estimated by the modelling approach ranged from 117 ng/kg bw/day (formula-fed 0–6 month olds) via 380 ng/kg bw/day (breastfed 4–6 months olds), 427 ng/kg bw/day (breastfed 6 days to 3 months olds), 501 ng/kg bw/day (breastfed 1–5 days olds) to 894 ng/kg bw/day (6–12 months olds). The biomonitoring approach estimated the high total exposure for 1–2 months old infants to be 136 ng/kg bw/day, which corresponds well with the modelled estimate of 117 ng/kg bw/day for formula-fed infants.

No biomonitoring data were available for toddlers aged 1–3 years, therefore an estimate was derived by extrapolation from 3–5 year old children to be able to make a comparison with the modelled estimate. The modelling approach gave an estimate of 873 ng/kg bw/day.

For the 3–10 years old children, a high total exposure of 981 ng/kg bw/day was obtained by the modelling approach. The biomonitoring approach gave estimates of 676 and 204 ng/kg bw/day for 3–5 year old children and 5–10 year old children, respectively, which were 1.5–5-fold lower than the figure obtained by the modelling approach.

For the teenagers, adults, and the elderly and very elderly, high total exposures of 642, 500–553, and 540 ng/kg bw/day were obtained by the modelling approach. The biomonitoring approach gave values of 228, 184, and 203 ng/kg bw/day. Again, the biomonitoring estimates are 2.7–2.8-fold lower than those obtained by the modelling approach.

To summarise, the estimates for the high total exposure as obtained by modelling and biomonitoring methods agree with each other within an order of magnitude. More specifically, the modelling approach gave estimates which were approximately 3-fold higher than those obtained by the biomonitoring approach. Again, the statistical procedures to arrive at high exposure estimates and the scenario for modelling the dietary exposure have to be discussed to explain the discrepancies.

Both the modelling and the biomonitoring methods use the 95th percentile (P95) of the distribution of the dietary daily intakes and of the urinary total BPA concentration to derive high-exposure estimates. In the biomonitoring, however, the P95 of the urinary total BPA concentration has different interpretations depending on whether spot urine samples, first morning urine samples, or 24-h samples are used. For spot urine samples, the 95th percentile is related to the 95 % probability that a single, randomly collected sample from a randomly selected subject has an urinary BPA concentration not exceeding the P95. This is important as urinary BPA concentrations of repeated urine collections from individuals may vary by up to two orders of magnitude. Some studies exist which indicate that the total variance can be subdivided into 70 % within-day variability, 21 % between-day variability, and 9 % between person variability. Thus, taking the P95 of the urinary BPA concentration as a measure for deriving high exposure estimates is a conservative approach, as the real long-term average value for high exposure is lower. It can therefore be concluded that the 3-fold discrepancy between estimates derived by the modelling approach and by the biomonitoring approach could be somewhat higher.

An important source for the discrepancy between the two approaches is probably the scenario chosen for modelling the dietary exposure, which is discussed in detail in the Chapter 4.9.1.1 on average total exposure and the choice of the highest 95th percentile observed in all surveys available in the Comprehensive Database as high dietary exposure. The biomonitoring studies for the European region are generally not based on a representative sampling of the population and may, therefore, not have captured high levels of exposure that may occur in specific geographic areas or specific population groups.

An additional source of discrepancy may be related to the conservativeness of the assumptions made to assess high exposure to non-food sources.

Table 32: High total exposure to BPA as estimated by the modelling approach and by biomonitoring.

Age class	Age (years)	High total exposure (ng/kg bw/day)	
		Modelling approach	Biomonitoring
Infants	0 – 1	117/380/427/501/894	136
Toddlers	1 – 3	873	not available
Children	3 – 10	981	204 – 676
Teenagers	10–18	642	228
Other adults	18–65	500 – 506 – 553	184
Elderly & very elderly	≥65	540	203

[(a)] Total exposure assessed by adding estimated exposure from inhalation, ingestion and dermal contact (see Chapter 4.7). For some age classes several values are given which refer to sub-groups among the age class (see Table 23 for details).
[(b)] When biomonitoring data were available for more than one age class, several values are given.

4.9.2. Comparison with values from other exposure assessments

According to its terms of reference, the present opinion considers only European data on food consumption, BPA occurrence and migration, and urinary BPA concentration for estimating the exposure of the general population in the European region via modelling and biomonitoring approaches. The panel noted that there are other extensive exposure estimations outside Europe such as those based on urinary biomonitoring data from US National Health and Nutrition Survey (NHANES) and the Canadian Health Measures Survey (CHMS) (Lakind et al., 2012). For NHANES, which covers the periods from 2003–2004 to 2009–2010, there is a pronounced temporal variability in urinary BPA concentration with indications for a decline in urinary BPA concentration (Melzer et al., 2010; Lakind et al., 2012; Wells et al., 2013), especially in the 6–11 year olds (Wells et al., 2013), which suggests that the exposure may have decreased over the last decade. However, the EFSA evaluation focuses on European data where, given the data available, detection of trends in changes in exposure (whether decreases or increases) is not yet possible.

FAO/WHO Expert Meeting on Bisphenol A

The FAO/WHO Expert Meeting on Bisphenol A (FAO/WHO, 2011) estimated dietary exposure to BPA in adults by means of model diets based on the budget method and concentration data on canned food (average and maximum concentrations) retrieved from the literature or based on expert judgement. The Expert Meeting considered a variety of possible scenarios with respect to the frequency of consumption of packaged food, from the worst-case scenario (100 %) to the best-case scenario (25 %). Consequently, a number of estimates were derived for the mean and 95th percentile exposure. The potential dietary exposure for children from 6 to 36 months of age was also based on the budget method and considered a variety of food patterns related to the consumption of liquid food (human milk or infant formula) and the introduction of solid food (fruits, desserts, vegetables and meat), primarily packaged in glass with coated metal lids. Dietary exposure to BPA in infants (0–6 months of age) was assessed by means of consumption data on infant formula and human milk retrieved from the literature. The Expert Meeting assumed a mean consumption of 130 ml/kg bw per day and a 95th percentile consumption of 174 ml/kg bw per day for all food consumption patterns based exclusively on infant formula or human milk or mixtures of the two. Six scenarios were considered in order to cover different patterns with respect to the consumption of human milk (breast, glass or polycarbonate bottles), liquid infant formula (glass or polycarbonate bottles) and powdered infant formula (glass or polycarbonate bottles). Except for human milk, all concentration data used in the calculations were expressed as unconjugated BPA.

The mean exposure of exclusively breastfed babies (0–6 months) to BPA was estimated to be 0.3 µg/kg bw per day, and exposure at the 95th percentile was estimated to be 1.3 µg/kg bw per day. Exposure estimates were generally higher for infants fed with liquid compared with powdered formula and for infants fed using PC compared with non-PC bottles. The highest estimated exposure occurred in infants 0–6 months of age who are fed with liquid formula out of PC bottles: 2.4 µg/kg bw per day at the mean and 4.5 µg/kg bw per day at the 95th percentile. For children older than 3 years, the highest exposure estimates did not exceed 0.7 µg/kg bw per day at the mean and 1.9 µg/kg bw per day at the 95th percentile. For adults, highest exposure estimates did not exceed 1.4 µg/kg bw per day at the mean and 4.2 µg/kg bw per day at the 95th percentile.

Based on the limited published or review data available on exposure to BPA from non-food sources, the Expert Meeting considered that the upper range of mean exposure from inhalation of free BPA (concentrations in indoor and outdoor air) was approximately 0.003 µg/kg bw per day for the general population. Indirect ingestion (dust, soil and toys) was considered to be approximately 0.03 µg/kg bw per day in infants and approximately 0.0001 µg/kg bw per day in children and adults. The Expert Meeting was unable to provide an estimate of exposure from thermal papers because of insufficient data on dermal absorption and observational studies on use patterns. Exposure to BPA from dental treatment was not taken into account because it was considered as short term and unlikely to contribute substantially to chronic exposure.

ANSES

The assessment of exposure carried out by the French Agency for Food, Environmental and Occupational Health & Safety (ANSES, 2013) within its risk assessment to BPA is the only assessment quantifying sources of exposure other than the diet in Europe. A systematic approach was here used to identify and characterise the sources, routes and levels of exposure as well as the categories of population to be studied. Two groups referred to as the general population (including vulnerable populations) and professionals handling end products intended for the general public as part of their activities (outside of fabrication, processing, distribution and disposal) were investigated. In the former group, children over 3 years of age, adults and pregnant women were classified as three subgroups. In its exposure assessment ANSES took into account the oral route (food and beverage, drinking water, dust), inhalation route (indoor and outdoor air) and dermal route (thermal paper).

ANSES analysed 1 319 composite food and beverage samples which were collected in the context of a total dietary study conducted between 2007 and 2009 for unconjugated BPA concentrations. Concentration data of BPA in matrices other than foods were retrieved from the scientific literature and from reports of especially commissioned French studies on indoor air and dust from 30 selected homes, on tap water from the water distribution network and bottled water (spring water, natural mineral water, waters made drinkable through treatment) and on the frequency and concentration of BPA in 50 receipts collected in various French retail stores.

Total exposure to BPA was estimated by combining exposure levels from the various matrices by means of a probabilistic Monte Carlo approach which included also other variables, such as food consumption (in terms of type and quantity), body weight and respiratory volume. In order to accommodate for the reduced systemic bioavailability of unconjugated BPA from food, the exposure estimates were multiplied with factor 0.03 (equivalent to 3 % systemic bioavailability) to give the internal exposure from this particular source. The individual estimated exposure values derived from air, dust and food were then combined to calculate a total internal dose. In addition, the internal exposure caused by handling thermal tickets was calculated separately.

In order to compare the values from the ANSES report for total exposure with values from this exposure assessment, average internal dose values from the ANSES report from food and sedimented dust were divided by a factor 0.03 and summed to the average internal dose from air and thermal paper. The same calculation cannot be carried out for the 95th percentile. These results, together with those from the other studies presented in this chapter, are given in Table 33.

The dietary source was identified as the major contributor to the total average internal exposure with 84 % (pregnant women), 78 % (adults) and 70 % (for children > 3 years). When analysing this source further it became apparent that food products packed in cans (representing approximately 50 % of total dietary exposure), some food items of animal origin (with meat, offal and charcuterie representing 17 % of total dietary exposure) and a background level contamination (representing 25 – 30 % of total exposure) were responsible for these high levels. ANSES reported that about 85 % of the 1 207 analysed food samples were reported to be contaminated with a BPA background level of < 5 µg/kg.

The exposure resulting from thermal paper is calculated separately and not included in the total exposure because of the high uncertainty. The values are reported as internal exposure but can be taken also as external exposure because the conversion factor is 1. For the study population "consumers-pregnant women handling thermal receipts", the internal dose varies from 0.029 to 140 ng/kg bw/day for the exposure model using an absorption flow determination, to 0.009 to 260 ng/kg bw/day for the exposure model using an absorption rate determination. The 95th percentiles used for the comparison with the toxicological points of reference in the risk assessment, are 50 ng/kg bw/day and 80 ng/kg bw/day respectively. The average for both is 20 ng/kg bw/day.

For the study population "consumers-adult handling thermal receipts", the internal dose varies from: 0.017 to 150 ng/kg bw/day for the exposure model using an absorption flow determination, to 0.021 to 260 ng/kg bw/day for the exposure model using an absorption rate determination. The 95th percentiles are respectively 58 and 89 ng/kg bw/day, the averages are 20 and 30 ng/kg bw/day (ANSES, 2013)

Belgium

Dietary exposure to BPA was assessed in Belgium (Geens et al., 2010) by means of analytical data from 45 canned beverages and 21 canned food items from the Belgian market. Using detailed information from the national food consumption survey, the BPA intake of adults through canned foods and beverages was estimated to be 0.015 and 0.086 µg/kg bw/day for the mean and the 95th percentile, respectively.

FACET

BPA was also used, as an example, to validate a software, developed within the DG Research-funded project FACET, to assess the exposure to chemical migrants from food packaging. In order to estimate exposure to BPA, concentration distributions in foods packed in light metal packaging such as food and beverage cans, metal closures, aerosol cans and tubes were linked probabilistically via the software tool to the amounts of each food item consumed, as recorded in the UK National Diet and Nutrient Survey (NDNS) involving 19–64 year olds. The output from the FACET tool has also been verified using a semi-deterministic approach using packaging data from the UK.

The estimates of exposure to BPA from foods packed in light metal packaging using the probabilistic FACET tool were 0.13 (mean) and 0.59 (97.5th percentile) µg/kg bw/day in UK consumers of these foods. The major contributors were canned foods such as beer, soup, cider, carbonates, preserved pasta and ready meals, fruit and vegetables. Values obtained by probabilistic modelling were within the minimum and maximum ranges obtained by using a semi-deterministic approach.

Conclusions

- Exposure to BPA carried out by the FAO/WHO Expert Meeting on Bisphenol A are far higher than others due to the use of a conservative model diet
- Other exposure estimates are in the same order of magnitude
- Only EFSA and ANSES estimated exposure to BPA by summing up different sources

- Only EFSA considered all routes; whereas only diet and thermal paper were considered by ANSES.

- Exposure from canned food is one of the major contributors to dietary exposure to BPA for all age groups

- Exposure levels are higher in children aged over 3 years

Table 33: Exposure estimates to BPA

Population groups	Reference	Source of exposure	Exposure to BPA		
			Mean	95th percentile/ High	Conservative estimate based on standard assumptions
Adults	ANSES(d), 2013	Ingestion (sedimented dust and food), inhalation (air), and dermal(a) (thermal paper)	68		
Children aged over 3			69		
Pregnant women			78		
Infant (3 months)	EFSA, 2006a	Human milk only			200
		Infant formula fed with glass or non-PC bottle			2 300
		Infant formula fed with PC bottle			11 000 (b) (4 000 (c))
Infant (6 months)		Infant formula fed with PC bottle and commercial foods/beverages			13 000 (b) (8 300 (c))
Children (1.5 years)		2 kg commercial			5 300
Adults		3 kg commercial			1 500
Infants 0-6 months	EFSA, 2013	Ingestion (dust, migration from toys), inhalation (air) and dermal exposure (thermal paper and cosmetics)	228	501	
Infants (0-6 months,			38	117	
Infants (6 days - 3			143	427	
Infants (4 - 6			127	380	
Infants (6-12			383	894	
Toddlers (1-3 years)			379	873	
Other children (3-10			314	981	
Teenagers (10-18			190	642	
Men (18-45 years)			146	500	
Women (18-45			152	553	
Other adults (45-65			145	506	
Elderly and very			136	540	
Adults	FACET	Canned food and beverages	130	590	
Adults	Geens et al., 2010	Canned food and beverages	15	86	
Adults	FAO/WHO, 2011	Canned food and beverages	1 400	4 200	
Children (6 - 36			700	1 900	
Infants (0-6 months)		Infant formula and/or human	300	1 300	

(a) Only for adults and pregnant women
(b) Based on the upper value of 50 µg BPA/litre of infant formula
(c) Based on the typical value of 10 µg BPA/litre of infant formula
(d) For comparison purposes, mean external exposure to BPA was back calculated based on ANSES estimates of systematically bioavailable BPA and on the correction factor of 0.03 used for ingestion. This could not be performed for high percentile.

4.9.3. Evaluation of uncertainty in total exposure through expert judgement

Uncertainties affecting the exposure assessment were evaluated systematically, as recommended by EFSA (EFSA, 2006b; EFSA, 2009). The approach taken follows the principles suggested by EFSA (EFSA, 2006b), adapted to the needs of the present assessment. A detailed description of the approach is provided in Appendix VIII. Appendix VIII also contains a detailed analysis of uncertainties affecting the parameters used in exposure assessment and of their combined impact on the uncertainty affecting the calculated estimates of exposure for each source (dietary and non-dietary) as reported in chapter 4.6 (exposure estimation). Urinary biomonitoring data provide a direct estimate of the dose which has actually entered the systemic circulation and high total exposure could also be assessed based on these data (chapter 4.8.2. biomonitoring studies on urinary BPA levels). Therefore Appendix VIII also contains an analysis of uncertainties affecting the estimates of total exposure assessment obtained from urinary biomonitoring. The present chapter summarises the results of the detailed uncertainty evaluations from Appendix VIII and derives overall conclusions on the uncertainty of the estimates of total exposure.

The overall evaluation of uncertainty in total exposure was focused on high (rather than average) total exposure estimate, as this is of particular interest for risk characterisation. As stated in the opinion of the Scientific Committee on a request from EFSA related to Exposure Assessments, in order to be protective for the whole of Europe, international calculations should provide exposure estimates that are equal to or greater "than the best estimates carried out at national levels" (EFSA, 2005). It is therefore assumed that the purpose of the exposure assessment is to estimate high total BPA exposure in the EU country where this estimate is highest. The 95th percentile was chosen as an approximate target percentile for each population group assessed. The present uncertainty analysis is therefore aimed at evaluating how much higher or lower than the calculated estimate the real 95th percentile of total BPA exposure might be, for the selected population groups in the EU country with highest exposure.

According to its terms of reference, the present opinion should "consider specifically the exposure situation for the supposedly most vulnerable groups of the population (e.g. pregnant women, infants and children, etc.) ". In Chapter 4.7, total high exposure was estimated for the different sub-groups of the population (e.g. breastfed infants, children of 3 to 10 years of age, women of reproductive age, etc.) through modelling, by adding up high exposures for the two sources with the highest 95th percentiles plus average exposure from the other sources. Among children aged more than one year, the highest calculated total exposure was observed in toddlers (1-3 years). Uncertainty in the assessment of total exposure was therefore analysed in detail for the four following groups: women of child bearing age, toddlers, breastfed infants in the first few days of life and formula-fed infants (see Tables 34, 35, 36 and 37).

Modelling and biomonitoring provide independent estimates of the real high exposure. Therefore, Tables 34-37 summarise the evaluation of uncertainties for each estimate, and show how they have been used to derive overall conclusions about uncertainty in assessing what value the real high total exposure might take in each of the four population groups.

The first step of the analysis was to assess the uncertainties around the estimate of total exposure obtained by adding up exposure from the different sources. The total estimate is affected by the uncertainties associated with the assessment for each source and route of exposure (which are analysed in Appendix VIII) and by the uncertainties associated with the model used i.e. the way exposures from the different sources are added up. As a second step, uncertainties of the estimates of total high exposure obtained from urinary biomonitoring, which are described in Appendix VIII, are considered.

Finally, the interval within which the real total high exposure may lie is assessed, based on the outcome of the two first steps.

The scale used to evaluate the impact of the source of uncertainty on the estimates of exposure is shown in Figure 10 (for more discussion of the scale, see Appendix VIII). Plus symbols indicate the real value could be higher than the estimate, while minus symbols indicate the real value could be lower than the estimate.

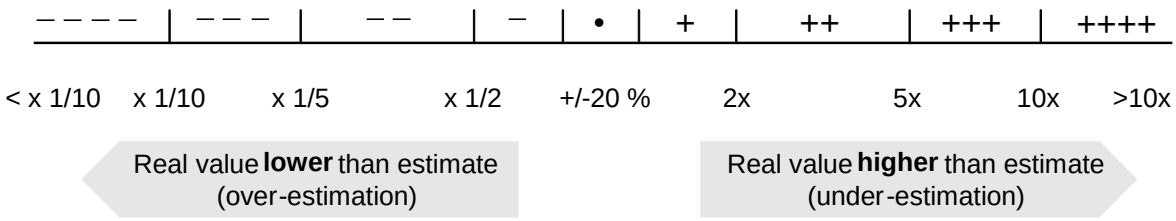


Figure 10: Scale used for evaluating the impact of uncertainties on estimates of total exposure to BPA

It is important to note that the scale is used to indicate the expected direction and width of the uncertainty but the relative likelihood of different values within the range was not assessed (except in the overall conclusions, see later). Thus, if the uncertainty is described with - -/+, it indicates that the real value may fall in an interval ranging from five times lower than the estimate to 2 times higher than the estimate. In this case, it does not necessarily imply that there is a higher probability for the real value to have been overestimated than underestimated.

The first step of the uncertainty analysis is described in the first part of Tables 34, 35, 36 and 37. The second column reports, for each source of exposure, the outcome of the detailed analysis of uncertainty that is presented in Appendix VIII. The third column reports the contribution of the single source of exposure to the modeled estimate of high total exposure. The contribution of this single source of exposure to the average total exposure is reported in the fourth column. Based on the assessments reported in columns 2 to 4 and on expert judgment, the expected overall impact of each source of uncertainty on the possible under- or over-estimation of the highest 95th percentile is reported in the fifth column. Finally, at the end of the first step, the impacts of uncertainty for each source are combined with the assessment of the uncertainty in the model and lead to the assessment of overall uncertainty around the estimated high total exposure.

For example, in the model used to assess high total exposure in women of child-bearing age (Table 34), average exposure from the air is responsible for 0.1 % of estimated high exposure. Additional information provided in the table is that average estimated exposure to BPA from air represents only 0.5 % of the estimated average exposure in this age class. Therefore, while the uncertainty in the estimate of average exposure from air alone is indicated with the symbols -/++, it is expected to have a very low impact on the possible under- or over-estimation of the highest 95th percentile when combining all BPA sources because of its low percentage contribution to the estimated exposure. This is indicated by the symbol ●.

Similarly, for thermal paper, in the model used to assess high total exposure in women of child-bearing age (Table 34), high exposure from thermal paper is responsible for 29 % of estimated high exposure. Additional information provided by the table is that the average estimated exposure to BPA from thermal paper represents 12 % of the estimated average exposure in this age class. Overall, while the uncertainty in the estimate of high exposure from thermal paper is indicated with the symbols – – /++, it is expected to have a reduced impact on the possible under- or over-estimation of the highest 95th percentile in this age class when combining all BPA sources, because it contributes only 29 % of the total estimate. This is indicated by the symbol –/+.

After evaluating the impact of uncertainty for each source of exposure on the total exposure, the overall uncertainty of the total exposure considering all the individual contributions is considered by expert judgement. The assessment for women of child-bearing age is shown in the final row of step 1

in Table 34. Considering the potential for the real value to be lower, three sources of uncertainty could make the real value up to 2-fold lower while the others are within 20 %. Overall it is judged that the real value could be up to 5-fold below the estimate (hence – –). Considering the potential for the real value to be higher, one source of uncertainty could make the real value up to 2-fold higher while the others are within 20 %. Overall it is judged that the real value could be up to 2-fold above the estimate (hence +). Combining these judgements leads to an assessment that the real value could lie between 5 times below and 2 times above the estimated value. This is represented in symbols as – –/+ and numerically as approximately 110–1 100 ng/kg bw/day (Table 34). It is emphasised that this should be regarded as an expert judgement and therefore approximate.

In the second step, the uncertainty of the estimate based on biomonitoring data is considered. This is assessed in detail in Appendix VIII and summarised in step 2 of Table 34, 35, 36 and 37.

In the third step, the estimates from both modelling and biomonitoring (where available) and their respective uncertainties (as evaluated in steps 1 and 2) are taken into account and are presented in Figures 11, 12, 13 and 14. The estimates of highest 95th percentile of total exposure in any EU country from modeling and urinary biomonitoring were in the same order of magnitude and the intervals describing uncertainty around these values largely overlap. Overall the Panel concludes that all values covered by the combined uncertainty intervals for the two estimates remain plausible.

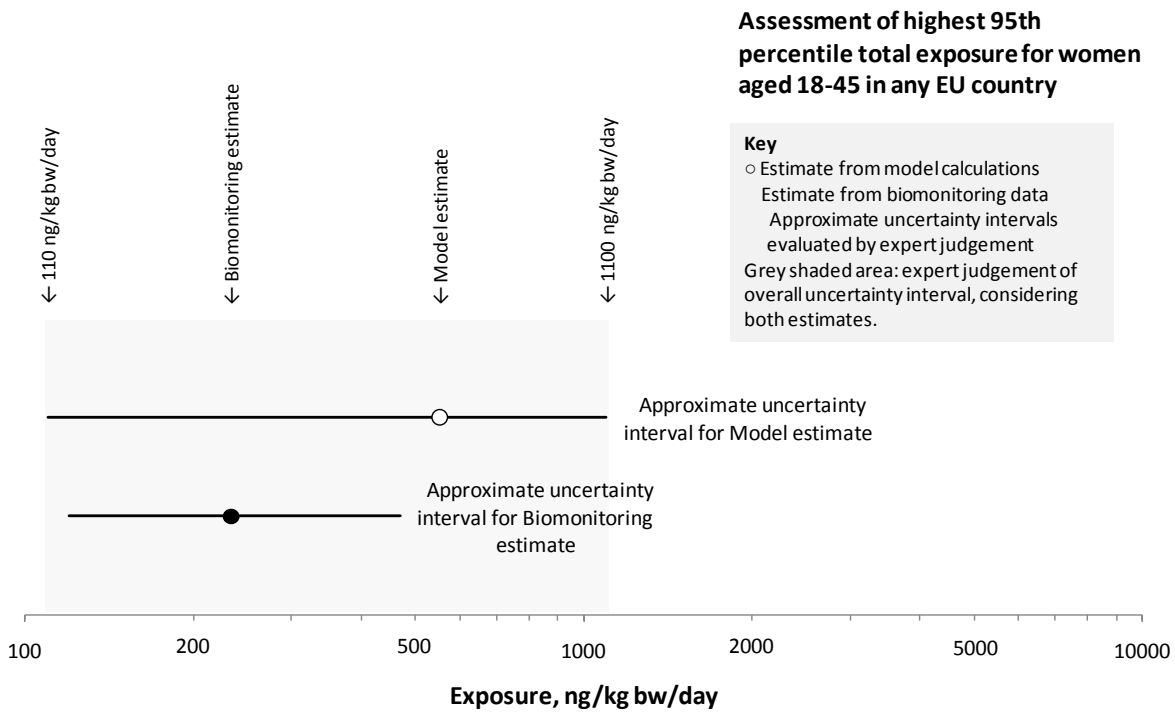
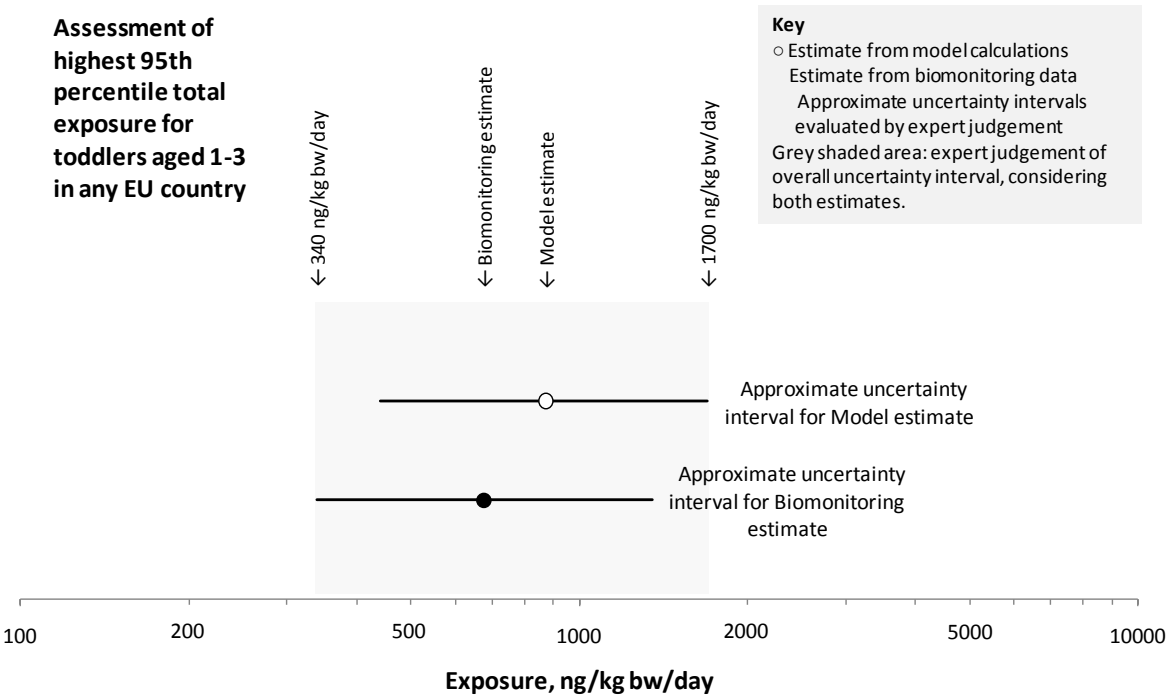


Figure 11: Overall evaluation of uncertainty for total high exposure of women of child-bearing age (18-45 years), plotted on a log scale. The real value may lie anywhere in the grey area.



(a) No biomonitoring data were available for this age group, therefore the values used were extrapolated from children 3-5 years

Figure 12: Overall evaluation of uncertainty for total high exposure of toddlers (a) (1-3 years), plotted on a log scale. It is considered that the real value may lie anywhere in the grey area.

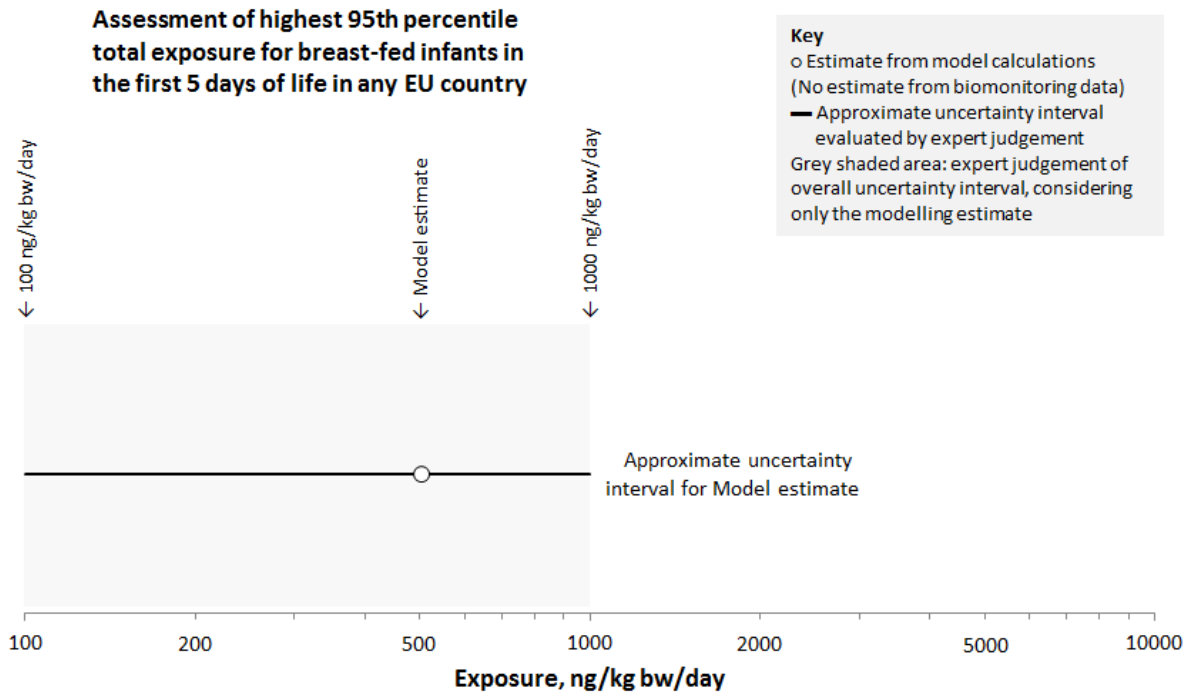


Figure 13: Overall evaluation of uncertainty for total high exposure of breastfed infants in the first 5 days of life (no biomonitoring data were available for this age group), plotted on a log scale. It is considered that the real value may lie anywhere in the grey area.

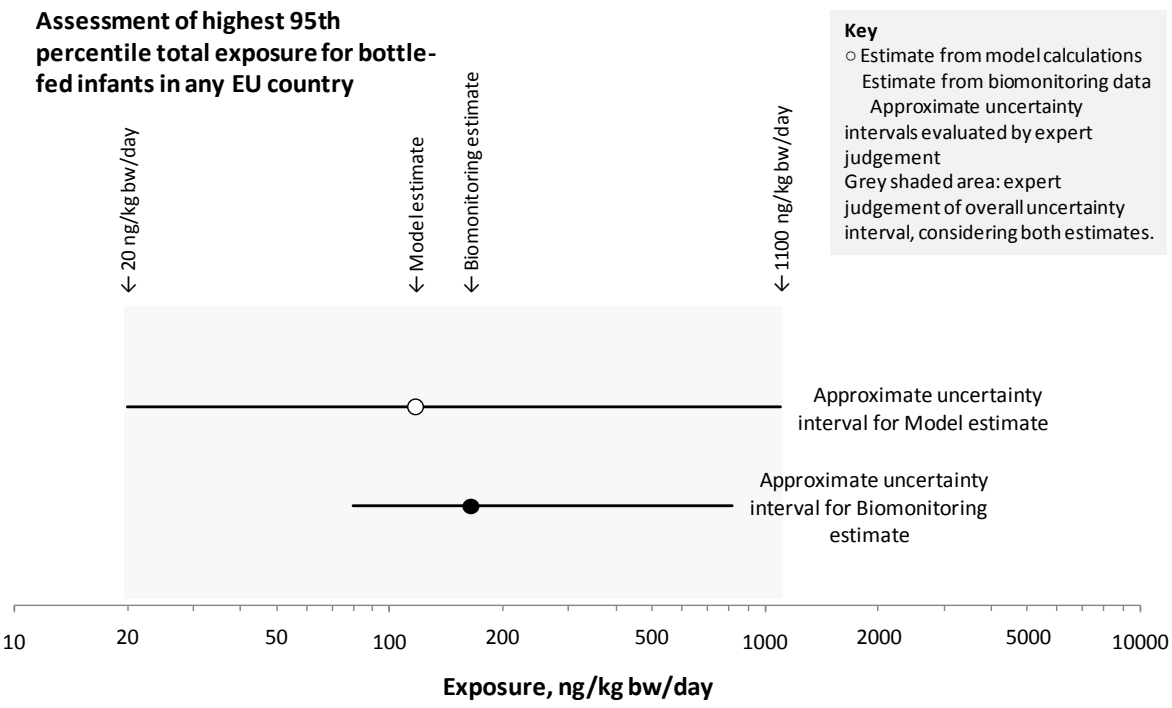


Figure 14: Overall evaluation of uncertainty for total high exposure of bottle-fed infants, plotted on a log scale. It is considered that the real value may lie anywhere in the grey area.

3287 **Table 34:** Evaluation of uncertainties affecting the assessment of high total exposure in women of child bearing age (18 to 45 years) (a)

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of estimated exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
FIRST STEP: UNCERTAINTIES AFFECTING THE MODEL BASED ESTIMATE OF HIGH TOTAL EXPOSURE FOR WOMEN AGED 18–45 OBTAINED BY ADDING UP EXPOSURE FROM THE DIFFERENT SOURCES (553 ng/kg bw/day)				
DIET Assessment of high dietary exposure to BPA	– / ●	70 %	87 %	– / ●
DERMAL EXPOSURE Assessment of high exposure from dermal contact with thermal paper	– –/++	29 %	12 %	–/+
Assessment of average exposure from cosmetics	– –/++	0.2 %	0.8 %	●
AIR INHALATION Assessment of average exposure from air	–/ ++	0.1 %	0.5 %	●
DUST INGESTION (AND INHALATION) Assessment of average exposure from dust	–/ +	0.02 %	0.1 %	●
MODELLING OF HIGH TOTAL EXPOSURE BY ADDING UP HIGH DIETARY EXPOSURE, HIGH EXPOSURE FROM THERMAL PAPER AND AVERAGE EXPOSURE FROM OTHER SOURCES. No information is available on the probability that women of child-bearing age who are highly exposed to BPA through the diet may also be highly exposed to thermal paper containing BPA. High total exposure (553 ng/kg bw/day) was assessed by adding up high exposure in the two sources of exposure leading to the highest 95th percentile plus average exposure from the other sources. If these events were independent this calculation would overestimate the real 95th percentile of total exposure. High total exposure assessed considering only high exposure in the source leading to the highest 95th percentile (diet) plus average exposure from all other sources would lead to 406 ng/kg bw/day (i.e. 74 % of the estimate). The probability for women of child bearing age to be highly exposed to all BPA sources is unknown. A more conservative model that would cover this case would be to add up high 95th percentiles from all sources, leading to an exposure of 557 ng/kg bw /bw/day i.e. 101 % of the estimate considered.				–/●
OVERALL UNCERTAINTY AROUND THE VALUE OF 553 NG/KG BW /BW/DAY AS A MODEL BASED ESTIMATE OF THE HIGHEST 95TH PERCENTILE OF TOTAL EXPOSURE FOR WOMEN AGED 18–45 IN ANY EU COUNTRY Considering the potential for the real value to be lower, three sources of uncertainty could make the real value up to 2 fold lower while the others are within 20 %. Overall it is judged that the real value could be up to 5 fold below the estimate (hence – –). Considering the potential for the real value to be higher, one source of uncertainty could make the real value up to 2 fold higher while the others are within 20 %. Overall it is judged that the real value could be up to 2 fold above the estimate (hence +).				– –/+ Based on exposure modelling, the real highest 95th percentile in any EU country may lie between approximately 110 and 1 100 ng/kg bw/day

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of estimated exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
SECOND STEP: UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR WOMEN AGED 18–45 FROM URINARY BIOMONITORING DATA (234 ng/kg bw /bw/day)				
Based on urinary biomonitoring, high total exposure is estimated to be 234 ng/kg bw/day in women of child-bearing age (see Appendix VIII). The main sources of uncertainty in this estimate are the sampling uncertainty due to limitations in the representativity of the available information on total BPA concentration in urine, the distribution uncertainty in the 95th percentile, and the uncertainty in the specific urinary output rate.				–/+ Based on biomonitoring data, the real highest 95th percentile in any EU country may lie between approximately 120 and 470 ng/kg bw/day
THIRD STEP: OVERALL CONCLUSION ON UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR WOMEN AGED 18–45				
The estimates of highest 95th percentile of total exposure in any EU country from modelling and urinary biomonitoring were in the same order of magnitude and the intervals describing uncertainty around these values largely overlap. Overall the Panel concludes that all values covered by the combined uncertainty intervals for the two estimates remain plausible. In this case, that implies an overall uncertainty interval of 110 to 1 100 ng/kg bw/day				Overall, the real highest 95th percentile in any EU country may lie between approximately 110 and 1 100 ng/kg bw/day,

(a) The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 10 for key to symbols.

3297

3298 **Table 35:** Evaluation of uncertainties affecting the assessment of high total exposure in toddlers (1-3 years) (a)

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
FIRST STEP: UNCERTAINTIES AFFECTING THE MODEL BASED ESTIMATE OF HIGH TOTAL EXPOSURE FOR TODDLERS (c) (1–3 YRS) OBTAINED BY ADDING UP EXPOSURE FROM THE DIFFERENT SOURCES (873 ng/kg bw/day)				
DIET				
Assessment of high dietary exposure to BPA	-/+	98 %	99 %	-/+
Assessment of average from toys	- -/+	0.0 %		●
DERMAL EXPOSURE				
Assessment of average exposure from dermal contact with thermal paper (assumed to be zero for toddlers)	●	0 %	0 %	●
Assessment of average exposure from cosmetics	- -/++	0.2 %	0.4 %	●
AIR INHALATION				
Assessment of average exposure from air	-/++	0.2 %	0.4 %	●
DUST INGESTION (AND INHALATION)				
Assessment of high exposure from dust	- - -/+	0.1 %	0.3 %	●
MODELLING OF HIGH TOTAL EXPOSURE BY ADDING UP HIGH DIETARY EXPOSURE, HIGH EXPOSURE FROM THERMAL PAPER AND AVERAGE EXPOSURE FROM OTHER SOURCES No information is available on the probability that toddlers who are highly exposed to BPA through the diet may also be highly exposed to BPA from dust. High total exposure (873 ng/kg bw/day) was assessed by adding up high exposure in the two sources of exposure leading to the highest 95th percentile plus average exposure from the other sources. If these events were independent this calculation would overestimate the real 95th percentile of total exposure. High total exposure assessed considering only high exposure in the source leading to the highest 95th percentile (diet) plus average exposure from all other sources would lead to 860 ng/kg bw/day (i.e. 98 % of the estimate). The probability for toddlers to be highly exposed to all BPA sources is unknown. A more conservative model that would cover this case would be to add up high 95th percentiles from all sources, leading to an exposure of 877 ng/kg bw/day (less than 1 % more than the estimate). Thus the estimate is dominated so strongly by the dietary source that the method of adding other sources has very little impact on the total.				●
OVERALL UNCERTAINTY AROUND THE VALUE OF 873 NG/KG BW/DAY AS A MODEL BASED ESTIMATE OF THE HIGHEST 95 %ILE OF TOTAL EXPOSURE FOR TODDLERS (1–3 YRS) IN ANY EU COUNTRY The estimate of total exposure for toddlers aged 1-3 years is dominated so strongly by the dietary source that uncertainties affecting the other sources have very little impact (each less than 20 %) on the uncertainty of the total exposure. Hence the uncertainty of the total exposure is very similar to				–/+ Based on exposure modelling, the real highest 95th percentile

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
that for the dietary source alone. Overall, it is judged that the real total exposure could be up to a factor of two above or below the estimate (hence –/+).				in any EU country may lie between approximately 440 and 1 700 ng/kg bw/day
SECOND STEP: UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR TODDLERS (1–3 YRS) FROM URINARY BIOMONITORING DATA (676 ng/kg bw/day)				
As no biomonitoring data are available for toddlers aged 1-3 years, an estimate was derived by extrapolation from biomonitoring data for children aged 3–5 years. Based on this, high total exposure is estimated to be 676 ng/kg bw/day in toddlers. The main sources of uncertainty in this estimate are the sampling uncertainty due to limitations in the representativity of the available information on total BPA concentration in urine, the distribution uncertainty in the 95th percentile, and the uncertainty in the specific urinary output rate. The extrapolation from children to toddlers is considered to contribute little uncertainty (see Table 12 in Appendix VIII).				–/+ Based on biomonitoring data, the real highest 95th percentile in any EU country may lie between approximately 340 and 1 350 ng/kg bw/day
THIRD STEP: OVERALL CONCLUSION ON UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR TODDLERS (1–3 YRS)				
The estimates of highest 95th percentile of total exposure in any EU country from modeling and urinary biomonitoring were in the same order of magnitude and the intervals describing uncertainty around these values largely overlap. Overall the Panel concludes that all values covered by the combined uncertainty intervals for the two estimates remain plausible. In this case, that implies an overall uncertainty interval of 340 to 1 700 ng/kg bw/day.				Overall, the real highest 95th percentile in any EU country may lie between approximately340 and 1 700 ng/kg bw/day

(b) The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 10 for key to symbols.
(c) No biomonitoring data were available for this age group, therefore the values used were extrapolated from children (3-5 years old)

3303

3304 **Table 36:** Evaluation of uncertainties affecting the assessment of high total exposure in breastfed infants in the first five days of life (a)

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the estimated average total exposure (%)	Ratio between high and average exposure from this source	Expected overall impact on the uncertainty of high total exposure
FIRST STEP: UNCERTAINTIES AFFECTING THE MODEL BASED ESTIMATE OF HIGH TOTAL EXPOSURE FOR BREASTFED INFANTS IN FIRST 5 DAYS OF LIFE, OBTAINED BY ADDING UP EXPOSURE FROM THE DIFFERENT SOURCES (501 ng/kg bw/day)					
DIET					
Assessment of high dietary exposure to BPA (via initial human milk/colostrum)	– –/+	99 %	99 %	2	– –/+
Assumption of zero exposure from mouthing of toys	●	0 %	0 %	-	●
DERMAL EXPOSURE					
Assumption of zero exposure from dermal contact with thermal paper	●	0 %	0 %	-	●
Assumption of zero exposure from cosmetics	●	0 %	0 %	-	●
AIR INHALATION					
Assessment of high exposure from air	- - -/+ +	0.2 %	1 %	2	●
DUST INGESTION (AND INHALATION)					
Assumption of zero exposure from dust	●	0 %	0 %	-	●
MODELLING OF HIGH TOTAL EXPOSURE BY ADDING UP HIGH DIETARY EXPOSURE, HIGH EXPOSURE FROM THERMAL PAPER AND AVERAGE EXPOSURE FROM OTHER SOURCES. In the first few days of life, breastfed infants are assumed to be exposure only to two sources of exposure: diet and air. The probability that those infants who are highly exposed to BPA through the diet may also be highly exposed to BPA from air is unknown. High total exposure (501 ng/kg bw/day) was assessed by adding up high exposure in these two sources of exposure. If these events were independent such calculation would overestimate the real 95th percentile of total exposure. High total exposure assessed considering only high exposure in the source leading to the highest 95th percentile (diet) plus average exposure from the other source would lead to 495 ng/kg bw/day (nearly 100 % of the estimate considered. Thus the estimate is dominated so strongly by the dietary source that the method of adding the second source has virtually no impact					●
OVERALL UNCERTAINTY AROUND THE VALUE OF 501 NG/KG BW/DAY AS A MODEL BASED ESTIMATE OF THE HIGHEST 95 %ILE OF TOTAL EXPOSURE FOR BREASTFED INFANTS IN THE FIRST 5 DAYS OF LIFE IN ANY EU COUNTRY The estimate of total exposure for breastfed infants in the first 5 days of life is so dominated by the dietary source that uncertainties affecting other sources have very little impact (each less than 20 %) on the uncertainty of the total exposure. Hence the uncertainty of the total exposure is					– –/+ Based on exposure modelling, the real highest 95th percentile in any EU country may lie

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the estimated average total exposure (%)	Ratio between high and average exposure from this source	Expected overall impact on the uncertainty of high total exposure
the same as that for the dietary source alone. Overall, it is judged that the real total exposure could be up to a factor of two above or five below the estimate (hence – –/+).					between approximately 100 and 1 000 ng/kg bw/day
SECOND STEP: UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR BREASTFED INFANTS IN THE FIRST 5 DAYS OF LIFE FROM URINARY BIOMONITORING DATA (164 ng/kg bw/day)					
There are few data for urinary biomonitoring in infants, only 2 small-sized studies from DE and US. Of these studies, only the US study provides individual data including information on whether infants were breastfed, formula-fed, or both. None of these individual data referred to 1-5 days old infants. In principle, it could be possible to extrapolate from older infants to 1-5 days old infants, based on information on physiological/developmental differences, but this approach would also need to consider the feeding conditions, i.e. the differences in (free & total) BPA concentration between initial milk (colostrum) and mature milk, thus introducing additional uncertainties. Therefore, the overall assessment for 1-5 day old infants is based only on the model estimates (Step 1, above).					Not applicable
THIRD STEP: OVERALL CONCLUSION ON UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR BREASTFED INFANTS IN THE FIRST 5 DAYS OF LIFE					
Due to the lack of urinary biomonitoring data for this group, the assessment of exposure is based on the modelling estimate alone. The uncertainty is therefore the same as was evaluated for the model estimate (Step 1, see above and Figure 13).					The real highest 95th percentile in any EU country may lie between approximately 100 and 1 000 ng/kg bw/day.

(a) The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 10 for key to symbols.

3317 **Table 37:** Evaluation of uncertainties affecting the assessment of high total exposure in formula-fed infants (0-6 months) (a)

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
FIRST STEP: UNCERTAINTIES AFFECTING THE MODEL BASED ESTIMATE OF HIGH TOTAL EXPOSURE FOR FORMULA-FED INFANTS OBTAINED BY ADDING UP EXPOSURE FROM THE DIFFERENT SOURCES (117 ng/kg bw/day)				
DIET				
Assessment of high dietary exposure to BPA	-/+++	69 %	79 %	–/+++
Assessment of average from toys	- - -/+	0.3 %	0.9 %	●
DERMAL EXPOSURE				
Assessment of average exposure from dermal contact with thermal paper (assumed to be zero for infants)	●	0 %	0 %	●
Assessment of average exposure from cosmetics	--/++	3 %	7 %	●
AIR INHALATION				
Assessment of average exposure from air	- -/++	2 %	6 %	●
DUST INGESTION (AND INHALATION)				
Assessment of high exposure from dust	- - -/+	27 %	7 %	– –/+
MODELLING OF HIGH TOTAL EXPOSURE BY ADDING UP HIGH DIETARY EXPOSURE, HIGH EXPOSURE FROM THERMAL PAPER AND AVERAGE EXPOSURE FROM OTHER SOURCES. No information is available on the probability that formula fed infants who are highly exposed to BPA through the diet may also be highly exposed to BPA in dust. High total exposure (117 g/kg bw/day) was assessed by adding up high exposure in the two sources of exposure leading to the highest 95th percentile plus average exposure from the other sources. If these events were independent such calculation would overestimate the real 95th percentile of total exposure. High total exposure assessed considering only high exposure in the source leading to the highest 95th percentile (diet) plus average exposure from all other sources would lead to 88 ng/kg bw/day (i.e. 75 % of such estimate). The probability for women of child bearing age to be highly exposed to all BPA sources is unknown. A more conservative model that would cover this case would be to add up high 95th percentiles from all sources, leading to an exposure of 124 ng/kg bw/day i.e. 106 % of the estimate considered.				–/●
OVERALL UNCERTAINTY AROUND THE VALUE OF 117 NG/KG BW/DAY AS A MODEL BASED ESTIMATE OF THE HIGHEST 95 %ILE OF TOTAL EXPOSURE FOR FORMULA-FED INFANTS IN ANY EU COUNTRY Considering the potential for the real value to be lower, one source of uncertainty could make the real value up to 5-fold lower while two others could make it up to 2-fold lower. Overall it is judged that the real value could be up to 5-fold below the estimate (hence – –). Considering the potential for the real value to be higher, one source of uncertainty could make the real value up to 10-fold higher while one other could make it up to 2-fold higher. Overall it is judged that the real value could be up to 10-fold above the estimate (hence +++).				– –/+++ Based on exposure modelling, the real highest 95th percentile in any EU country may lie between approximately 20 and 1 100 ng/kg bw/day

Source of uncertainty	Uncertainty of the estimated exposure from each single source	Contribution of the single source of exposure to the estimated high total exposure (%)	Contribution of the single source of exposure to the average total exposure (%)	Expected overall impact on the uncertainty of high total exposure
SECOND STEP: UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR FORMULA-FED INFANTS FROM URINARY BIOMONITORING DATA (164 ng/kg bw/day)				
Based on urinary biomonitoring, high total exposure is estimated to be 164 ng/kg bw/day in infants (see Appendix VIII). The main sources of uncertainty in this estimate are the sampling uncertainty due to limitations in the representativity of the available information on total BPA concentration in urine, the distribution uncertainty in the 95th percentile, and the uncertainty in the specific urinary output rate				–/++ Based on biomonitoring data, the real highest 95th percentile in any EU country may lie between approximately 80 and 820 ng/kg bw/day
THIRD STEP: OVERALL CONCLUSION ON UNCERTAINTIES AFFECTING THE ASSESSMENT OF HIGH TOTAL EXPOSURE FOR FORMULA–FED INFANTS				
The estimates of highest 95th percentile of total exposure in any EU country from modeling and urinary biomonitoring were in the same order of magnitude and the intervals describing uncertainty around these values largely overlap. Overall the Panel concludes that all values covered by the combined uncertainty intervals for the two estimates remain plausible. In this case, that implies an overall uncertainty interval of 20 to 1 100 ng//kg bw/day				Overall, the real highest 95th percentile in any EU country may lie between approximately 20 and 1 100 ng/kg bw/day,

(a) The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 10 for key to symbols.

CONCLUSIONS

The current exposure assessment of BPA from all sources shows that diet is the main source of exposure to BPA in all population groups (from 78 to 99%). Canned food and non-canned meat and meat products are the two main dietary contributors to BPA exposure in the large majority of countries and age classes.

Among the population older than 6 months, infants and toddlers had the highest estimated average (375 ng/kg bw/day) and high (857 ng/kg bw/day) dietary exposure. The CEF Panel considered that this was mainly due to their higher consumption of foods and beverages per kg bw. The modelled dietary exposure for teenagers, adults (including women of child bearing age) and elderly/very elderly ranged from 116 to 159 ng/kg bw/day for the average exposure and from 341 to 388 ng/kg bw/day for the high exposure, respectively. Dietary exposure to BPA estimated by EFSA in 2006 in the population older than 6 months was far higher (up to 5 300 ng/kg bw/day in toddlers) compared with the current assessment (up to 375 ng/kg bw/day for toddlers), due to the lack of data at that time which led to the use of very conservative assumptions in relation to both the level of consumption of canned food and the estimated BPA concentration in these foods.

Dietary exposure to BPA estimated by EFSA in 2006 in the population 0 to 6 months was also far higher (up to 11 000 ng/kg bw/day in infants aged 3 months in one of the scenarios considered) compared with the current assessment (up to 225 ng/kg bw/day for infants of 1–5 days), due to the lack of data at that time leading to very conservative assumptions in relation to BPA concentration in infant formula and to BPA migration from PC bottles.

Dietary exposure in women of childbearing age was slightly higher (132 and 388 ng/kg bw/day for average and high exposure, respectively) than that in men of the same age (126 and 355 ng/kg bw/day for average and high exposure, respectively). This may be due to different food items consumed by women as reported in the individual surveys.

The uncertainty around the estimates of dietary exposure based on the EFSA comprehensive database was judged as relatively low.

For the age class 'Infants' (0–6 months), the average total exposure as estimated by the modelling approach ranged from 38 ng/kg bw/day to 228 ng/kg bw/day. The modelled average total exposure for the population older than 6 months ranged from 314 to 383 ng/kg bw/day in infants, toddlers and children aged 3 to 10 years, and from 136 to 190 ng/kg bw/day in teenagers, adults and elderly and very elderly.

For the age class 'Infants' (0–6 months), the high total exposure as estimated by the modelling approach ranged from 117 ng/kg bw/day to 501 ng/kg bw/day. The modelled high total exposure for populations older than 6 months ranged from 873 to 981 ng/kg bw/day in infants, toddlers and children aged 3 to 10 years, and from 500 to 642 ng/kg bw/day in teenagers, adults and elderly and very elderly.

In addition to diet as the main contributor to total exposure thermal paper was the second source of exposure in all population groups above 3 years of age (from 7 to 15%). The uncertainty around the estimate of exposure to BPA from thermal paper was judged to be considerably higher than that around dietary exposure. The Panel considers that more data would be needed in relation to BPA absorption through the skin and to patterns of thermal paper handling by the general population in order to provide a refined estimate of exposure through this source which would reduce uncertainty in the estimate of total exposure to BPA. The CEF Panel is aware of an ongoing study on BPA pharmacokinetic and dermal exposure in cashiers sponsored by the National Institute of Environmental Health Sciences (NIEHS) under the National Toxicology Program (NTP). The results

of this study will be considered by the CEF Panel as they will be an additional source of information regarding the absorption of BPA from thermal paper.

Dust was the second source of exposure to BPA in children under the age of 3 years (except for infants in the first few days of life).

Average exposure to BPA from other sources such as toys and cosmetics was estimated to be less than 0.3 ng/kg bw/day and 2.9 ng/kg bw/day, respectively, in all population groups.

Biomonitoring estimates based on urinary BPA concentrations are in good agreement with modelled BPA exposures from all sources, suggesting that no major exposure sources have been missed for the modelled exposure assessment.

REFERENCES

Ackerman LK, Noonan GO, Heiserman WM, Roach JA, Limm W and Begley TH, 2010. Determination of bisphenol A in US infant formulas: updated methods and concentrations. Journal of Agricultural and Food Chemistry, 58, 2307-2313.

Ahmadkhaniha R, Salimi M and Rastkari N, 2013. Comparison of single-walled carbon nanotubes, multi-walled carbon nanotubes and C_{18} as adsorbents for the solid phase extraction of bisphenol A and bisphenol F in canned food. Fullerenes, Nanotubes and Carbon Nanostructures, 21, 604-616.

Alin J and Hakkarainen M, 2012. Migration from polycarbonate packaging to food simulants during microwave heating. Polymer Degradation and Stability, 97, 1387-1395.

Amiridou D and Voutsa D, 2011. Alkylphenols and phthalates in bottled waters. Journal of Hazardous Materials, 185, 281-286.

Angerer J, Ewers U and Wilhelm M, 2007. Human biomonitoring: state of the art. International Journal of Hygiene and Environmental Health, 210, 201-228.

ANSES (Agence Nationale de Sécurité Sanitaire de l'Alimentation, de l'Environnement et du Travail), 2013. Opinion of the French Agency for Food, Environmental and Occupational Health and Safety on the assessment of the risks associated with bisphenol A for human health, and on toxicological data and data on the use of bisphenols S, F, M, B, AP, AF and BADGE, 13 pp.

AOC (Resin Selector), 2013. Market Segment: Food Processing. Available online: http://www.corrosionresins.com/web/corrosion/selector/market/food_processing/

Asimakopoulos AG, Thomaidis NS and Koupparis MA, 2012. Recent trends in biomonitoring of bisphenol A, 4-t-octylphenol, and 4-nonylphenol. Toxicology Letters, 210, 141-154.

ASTM (American Society for Testing and Materials), 1997. Standard practice for collection of floor dust for chemical analysis D5438-94. Annual Book of ASTM Standards, 11.03, 7 pp.

Atkins P 2012. Sample preparation, extraction and analysis of imported children's toys for bisphenol A and phthalates. SPEX CertiPrep.

Atkinson JC, Diamond F, Eichmiller F, Selwitz R and Jones G, 2002. Stability of bisphenol A, triethylene-glycol dimethacrylate, and bisphenol A dimethacrylate in whole saliva. Dental Materials, 18, 128-135.

Aylward LL, Kirman CR, Adgate JL, McKenzie LM and Hays SM, 2012. Interpreting variability in population biomonitoring data: role of elimination kinetics. Journal of Exposure Science and Environmental Epidemiology, 22, 398-408.

Baggiani C, Anfossi L and Giovannoli C, 2007. Solid phase extraction of food contaminants using molecular imprinted polymers. Analytica Chimica Acta, 591, 29-39.

3406 Baggiani C, Baravalle P, Giovannoli C, Anfossi L and Giraudi G, 2010. Molecularly imprinted
3407 polymer/cryogel composites for solid-phase extraction of bisphenol A from river water and wine.
3408 Analytical and Bioanalytical Chemistry, 397, 815-822.

3409 Ballesteros O, Zafra A, Navalon A and Vilchez JL, 2006. Sensitive gas chromatographic-mass
3410 spectrometric method for the determination of phthalate esters, alkylphenols, bisphenol A and their
3411 chlorinated derivatives in wastewater samples. Journal of Chromatography A, 1121, 154-162.

3412 Ballesteros-Gomez A, Ruiz FJ, Rubio S and Perez-Bendito D, 2007. Determination of bisphenols A
3413 and F and their diglycidyl ethers in wastewater and river water by coacervative extraction and
3414 liquid chromatography-fluorimetry. Analytica Chimica Acta, 603, 51-59.

3415 Ballesteros-Gomez A, Rubio S and Pérez-Bendito D, 2009. Analytical methods for the determination
3416 of bisphenol A in food. Journal of Chromatography A, 1216, 449-469.

3417 Basheer C, Lee HK and Tan KS, 2004. Endocrine disrupting alkylphenols and bisphenol-A in coastal
3418 waters and supermarket seafood from Singapore. Marine Pollution Bulletin, 48, 1161-1167.

3419 Baugros JB, Giroud B, Dessalces G, Grenier-Loustalot MF and Cren-Olive C, 2008. Multiresidue
3420 analytical methods for the ultra-trace quantification of 33 priority substances present in the list of
3421 REACH in real water samples. Analytica Chimica Acta, 607, 191-203.

3422 Becerra V and Odermatt J, 2012. Detection and quantification of traces of bisphenol A and bisphenol
3423 S in paper samples using analytical pyrolysis-GC/MS. Analyst, 137, 2250-2259.

3424 Becker K, Goen T, Seiwert M, Conrad A, Pick-Fuss H, Muller J, Wittassek M, Schulz C and Kolossa-
3425 Gehring M, 2009. GerES IV: phthalate metabolites and bisphenol A in urine of German children.
3426 International Journal of Hygiene and Environmental Health, 212, 685-692.

3427 Bergmann KE and Mensink GBM, 1999. Körpermaße und Übergewicht. Gesundheitswesen, 61, S115-
3428 S120.

3429 Beronius A, Ruden C, Hakansson H and Hanberg A, 2010. Risk to all or none? A comparative
3430 analysis of controversies in the health risk assessment of Bisphenol A. Reproductive Toxicology,
3431 29, 132-146.

3432 Beronius A and Hanberg A (Karolinska Institutet, Institute of Environmental Medicine), 2011.
3433 Sources of exposure to bisphenol A. 2/2011, 46 pp.

3434 Biedermann S, Tschudin P and Grob K, 2010. Transfer of bisphenol A from thermal printer paper to
3435 the skin. Analytical and Bioanalytical Chemistry, 398, 571-576.

3436 Biedermann-Brem S, Grob K and Fjeldal P, 2008. Release of bisphenol A from polycarbonate baby
3437 bottles: mechanisms of formation and investigation of worst case scenarios. European Food
3438 Research and Technology, 227, 1053-1060.

3439 Biedermann-Brem S and Grob K, 2009. Release of bisphenol A from polycarbonate baby bottles:
3440 water hardness as the most relevant factor. European Food Research and Technology, 228, 679-
3441 684.

3442 Bloom MS, Kim D, Vom Saal FS, Taylor JA, Cheng G, Lamb JD and Fujimoto VY, 2011. Bisphenol
3443 A exposure reduces the estradiol response to gonadotropin stimulation during *in vitro* fertilization.
3444 Fertility and Sterility, 96, 672-677.

3445 Boeniger MF, Lowry LK and Rosenberg J, 1993. Interpretation of urine results used to assess
3446 chemical exposure with emphasis on creatinine adjustments: a review. American Industrial
3447 Hygiene Association Journal, 54, 615-627.

3448 Bono-Blay F, Guart A, de la Fuente B, Pedemonte M, Pastor MC, Borrell A and Lacorte S, 2012.
3449 Survey of phthalates, alkylphenols, bisphenol A and herbicides in Spanish source waters intended
3450 for bottling. Environmental Science and Pollution Research, 19, 3339-3349.

3451 BPF (British Plastics Federation), 2013. Polycarbonate PC. Available online:
3452 http://www.bpf.co.uk/Plastipedia/Polymers/Polycarbonate.aspx

3453 Bradley EL, Read WA and Castle L, 2007. Investigation into the migration potential of coating
3454 materials from cookware products. Food Additives and Contaminants, 24, 326-335.

3455 Bradley EL, Honkalampi-Hamalainen U, Weber A, Andersson MA, Bertaud F, Castle L, Dahlman O,
3456 Hakulinen P, Hoornstra D, Lhuguenot JC, Maki-Paakkanen J, Salkinoja-Salonen M, Speck DR,
3457 Severin I, Stammati A, Turco L, Zucco F and von Wright A, 2008a. The BIOSAFEPAPER project
3458 for *in vitro* toxicity assessments: preparation, detailed chemical characterisation and testing of
3459 extracts from paper and board samples. Food and Chemical Toxicology, 46, 2498-2509.

3460 Bradley EL, Driffield M, Harmer N, Oldring PKT and Castle L, 2008b. Identification of potential
3461 migrants in epoxy phenolic can coatings. International Journal of Polymer Analysis and
3462 Characterization, 13, 200-223.

3463 Braun JM, Kalkbrenner AE, Calafat AM, Bernert JT, Ye X, Silva MJ, Barr DB, Sathyanarayana S and
3464 Lanphear BP, 2011. Variability and predictors of urinary bisphenol A concentrations during
3465 pregnancy. Environmental Health Perspectives, 119, 131-138.

3466 Bremmer HJ and van Veen MP (RIVM), 2002. Children's toys fact sheet. 612810012/2002, 70 pp.

3467 Brenn-Struckhofova Z and Cichna-Markl M, 2006. Determination of bisphenol A in wine by sol-gel
3468 immunoaffinity chromatography, HPLC and fluorescence detection. Food Additives and
3469 Contaminants, 23, 1227-1235.

3470 Brugnera MF, Trindade MAG and Boldrin Zanoni MV, 2010. Detection of bisphenol A on a screen-
3471 printed carbon electrode in CTAB micellar medium. Analytical Letters, 43, 2823-2836.

3472 Cacho JI, Campillo N, Viñas P and Hernández-Córdoba M, 2012. Stir bar sorptive extraction coupled
3473 to gas chromatography-mass spectrometry for the determination of bisphenols in canned beverages
3474 and filling liquids of canned vegetables. Journal of Chromatography A, 1247, 146-153.

3475 Cacho JI, Campillo N, Viñas P and Hernández-Córdoba M, 2013. Stir bar sorptive extraction with EG-
3476 Silicone coating for bisphenols determination in personal care products by GC-MS. Journal of
3477 Pharmaceutical and Biomedical Analysis, 78-79, 255-260.

3478 Calafat AM, Ye X, Wong LY, Reidy JA and Needham LL, 2008. Exposure of the US population to
3479 bisphenol A and 4-tertiary-octylphenol: 2003-2004. Environmental Health Perspectives, 116, 39-
3480 44.

3481 Callan AC, Hinwood AL, Heffernan A, Eaglesham G, Mueller J and Odland JO, 2012. Urinary
3482 bisphenol A concentrations in pregnant women. International Journal of Hygiene and
3483 Environmental Health, in press.

3484 Campbell CG, Borglin SE, Green FB, Grayson A, Wozei E and Stringfellow WT, 2006. Biologically
3485 directed environmental monitoring, fate, and transport of estrogenic endocrine disrupting
3486 compounds in water: A review. Chemosphere, 65, 1265-1280.

3487 Cantonwine D, Meeker JD, Hu H, Sánchez BN, Lamadrid-Figueroa H, Mercado-García A,
3488 Fortenberry GZ, Calafat AM and Tellez-Rojo MM, 2010. Bisphenol A exposure in Mexico City
3489 and risk of prematurity: a pilot nested case control study. Environmental Health, 62, 2-7.

3490 Cao XL and Corriveau J, 2008a. Migration of bisphenol A from polycarbonate baby and water bottles
3491 into water under severe conditions. Journal of Agricultural and Food Chemistry, 56, 6378-6381.

3492 Cao XL and Corriveau J, 2008b. Determination of bisphenol A in water by isotope dilution headspace
3493 solid-phase microextraction and gas chromatography/mass spectrometry without derivatization.
3494 Journal of AOAC International, 91, 622-629.

3495 Cao XL and Corriveau J, 2008c. Survey of bisphenol A in bottled water products in Canada. Food
3496 Additives and Contaminants: Part B, 1, 161-164.

3497 Cao XL, Dufresne G, Belisle S, Clement G, Falicki M, Beraldin F and Rulibikiye A, 2008. Levels of
3498 bisphenol A in canned liquid infant formula products in Canada and dietary intake estimates.
3499 Journal of Agricultural and Food Chemistry, 56, 7919-7924.

Cao XL, Corriveau J, Popovic S, Clement G, Beraldin F and Dufresne G, 2009a. Bisphenol A in baby food products in glass jars with metal lids from Canadian markets. Journal of Agricultural and Food Chemistry, 57, 5345-5351.

Cao XL, Corriveau J and Popovic S, 2009b. Levels of Bisphenol A in canned soft drink products in Canadian markets. Journal of Agricultural and Food Chemistry, 57, 1307-1311.

Cao XL, Corriveau J and Popovic S, 2009c. Migration of bisphenol A from can coatings to liquid infant formula during storage at room temperature. Journal of Food Protection, 72, 2571-2574.

Cao XL, Corriveau J and Popovic S, 2010a. Sources of low concentrations of bisphenol A in canned beverage products. Journal of Food Protection, 73, 1548-1551.

Cao XL, Corriveau J and Popovic S, 2010b. Bisphenol A in canned food products from Canadian markets. Journal of Food Protection, 73, 1085-1089.

Cao XL, Perez-Locas C, Dufresne G, Clement G, Popovic S, Beraldin F, Dabeka RW and Feeley M, 2011. Concentrations of bisphenol A in the composite food samples from the 2008 Canadian total diet study in Quebec City and dietary intake estimates. Food Additives and Contaminants: Part A, 28, 791-798.

Cariot A, Dupuis A, Albouy-Llaty M, Legube B, Rabouan S and Migeot V, 2012. Reliable quantification of bisphenol A and its chlorinated derivatives in human breast milk using UPLC–MS/MS method. Talanta, 100, 175-182.

Casas L, Fernandez MF, Llop S, Guxens M, Ballester F, Olea N, Irurzun MB, Rodriguez LS, Riano I, Tardon A, Vrijheid M, Calafat AM, Sunyer J and Project I, 2011. Urinary concentrations of phthalates and phenols in a population of Spanish pregnant women and children. Environment International, 37, 858-866.

CDC (Centers for Disease Control and Prevention), 2012. Fourth National Report on Human exposure to Environmental Chemicals, Updated Tables September 2012, 307 pp.

CEH (Chemical Economics Handbook), 2008. Polycarbonate resins. Marketing Research Report, November 2008; 580.1100A-580.1104Z

CEH (Chemical Economics Handbook), 2010. Bisphenol A. Marketing Research Report, July 2010; 619.5000A-619.5002D

CEN, 2005. Determination of 2,2-bis(4-hydroxyphenyl)propane (bisphenol A) in food simulants.

Chen TC, Shue MF, Yeh YL and Kao TJ, 2010. Bisphenol A occurred in Kao-Pin River and its tributaries in Taiwan. Environmental Monitoring and Assessment, 161, 135-145.

Chen X, Wang C, Tan X and Wang J, 2011. Determination of bisphenol A in water via inhibition of silver nanoparticles-enhanced chemiluminescence. Analytica Chimica Acta, 689, 92-96.

Chou WC, Chen JL, Lin CF, Chen YC, Shih FC and Chuang CY, 2011. Biomonitoring of bisphenol A concentrations in maternal and umbilical cord blood in regard to birth outcomes and adipokine expression: a birth cohort study in Taiwan. Environmental Health, 94, 1-10.

Christensen KL, Lorber M, Koch HM, Kolossa-Gehring M and Morgan MK, 2012a. Population variability of phthalate metabolites and bisphenol A concentrations in spot urine samples versus 24- or 48-h collections. Journal of Exposure Science and Environmental Epidemiology, 22, 632-640.

Christensen KL, Lorber M, Koslitz S, Bruning T and Koch HM, 2012b. The contribution of diet to total bisphenol A body burden in humans: Results of a 48 hour fasting study. Environment International, 50, 7-14.

Chung SY, Kwon H, Choi YH, Karmaus W, Merchant AT, Song KB, Sakong J, Ha M, Hong YC and Kang D, 2012. Dental composite fillings and bisphenol A among children: a survey in South Korea. International Dental Journal, 62, 65-69.

3546 Cobellis L, Colacurci N, Trabucco E, Carpentiero C and Grumetto L, 2009. Measurement of bisphenol
3547 A and bisphenol B levels in human blood sera from healthy and endometriotic women. Biomedical
3548 Chromatography, 23, 1186-1190.

3549 Cooper JE, Kendig EL and Belcher SM, 2011. Assessment of bisphenol A released from reusable
3550 plastic, aluminium and stainless steel water bottles. Chemosphere, 85, 943-947.

3551 Cousins IT, Staples CA, Klecka GM and Mackay D, 2002. A multimedia assessment of the
3552 environmental fate of bisphenol A. Human and Ecological Risk Assessment, 8, 1107-1135.

3553 Covaci A, Geens T, Koppen G, Caeyers T, Govarts E, Dewolf M-C, Van de Mieroop E, Nelen V and
3554 D. A, 2012. Bisphenol-A and triclosan in urine of Belgian DEMOCOPHES mother-child pairs. Do
3555 we need to worry? Proceedings of the Cyprus Presidency Conference "Human Biomonitoring -
3556 Linking Environment to Health and Supporting Policy", Larnaca Cyprus, 22-25 October 2012.

3557 Cunha SC, Almeida C, Mendes E and Fernandes JO, 2011. Simultaneous determination of bisphenol
3558 A and bisphenol B in beverages and powdered infant formula by dispersive liquid-liquid micro-
3559 extraction and heart-cutting multidimensional gas chromatography-mass spectrometry. Food
3560 Additives and Contaminants, 28, 513-526.

3561 Cunha SC, Cunha C, Ferreira AR and Fernandes JO, 2012. Determination of bisphenol A and
3562 bisphenol B in canned seafood combining QuEChERS extraction with dispersive liquid-liquid
3563 microextraction followed by gas chromatography-mass spectrometry. Analytical and Bioanalytical
3564 Chemistry, 404, 2453-2463.

3565 De Coensel N, David F and Sandra P, 2009. Study on the migration of bisphenol-A from baby bottles
3566 by stir bar sorptive extraction-thermal desorption-capillary GC-MS. Journal of Separation Science,
3567 32, 3829-3836.

3568 Deblonde T, Cossu-Leguille C and Hartemann P, 2011. Emerging pollutants in wastewater: a review
3569 of the literature. International Journal of Hygiene and Environmental Health, 214, 442-448.

3570 Dekant W and Völkel W, 2008. Human exposure to bisphenol A by biomonitoring: methods, results
3571 and assessment of environmental exposures. Toxicology and Applied Pharmacology, 228, 114-134.

3572 Demierre AL, Peter R, Oberli A and Bourqui-Pittet M, 2012. Dermal penetration of bisphenol A in
3573 human skin contributes marginally to total exposure. Toxicology Letters, 213, 305-308.

3574 Dirtu AC, Roosens L, Geens T, Gheorghe A, Neels H and Covaci A, 2008. Simultaneous
3575 determination of bisphenol A, triclosan, and tetrabromobisphenol A in human serum using solid-
3576 phase extraction and gas chromatography-electron capture negative-ionization mass spectrometry.
3577 Analytical and Bioanalytical Chemistry, 391, 1175-1181.

3578 Dodson RE, Nishioka M, Standley LJ, Perovich LJ, Brody JG and Rudel RA, 2012. Endocrine
3579 disruptors and asthma-associated chemicals in consumer products. Environmental Health
3580 Perspectives, 120, 935-943.

3581 Doerge DR, Vanlandingham M, Twaddle NC and Delclos KB, 2010a. Lactational transfer of
3582 bisphenol A in Sprague-Dawley rats. Toxicology Letters, 199, 372-376.

3583 Doerge DR, Twaddle NC, Vanlandingham M and Fisher JW, 2010b. Pharmacokinetics of bisphenol A
3584 in neonatal and adult Sprague-Dawley rats. Toxicology and Applied Pharmacology, 247, 158-165.

3585 Doerge DR, Twaddle NC, Vanlandingham M and Fisher JW, 2012. Pharmacokinetics of bisphenol A
3586 in serum and adipose tissue following intravenous administration to adult female CD-1 mice.
3587 Toxicology Letters, 211, 114-119.

3588 Dorn PB, Chou CS and Gentempo JJ, 1987. Degradation of bisphenol A in natural waters.
3589 Chemosphere, 16, 1501-1507.

3590 Dupuis A, Migeot V, Cariot A, Albouy-Llaty M, Legube B and Rabouan S, 2012. Quantification of
3591 bisphenol A, 353-nonylphenol and their chlorinated derivatives in drinking water treatment plants.
3592 Environmental Science and Pollution Research, 19, 4193-4205.

3593 Duty SM, Mendonca K, Hauser R, Calafat AM, Ye X, Meeker JD, Ackerman R, Cullinane J, Faller J
3594 and Ringer S, 2013. Potential sources of bisphenol A in the neonatal intensive care unit. Pediatrics,
3595 131, 482-489.

3596 ECB (European Chemicals Bureau), 2008. European Union Risk Assessment Report-4,4'-
3597 isopropylidenediphenol (bisphenol A). Available online:
3598 http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/15069/1/lbna24589enn.pdf.
3599 JRC Scientific and Technical Reports, 164 pp.

3600 ECB (European Chemicals Bureau), 2010. European Union risk assessment report 4,4'-
3601 isopropylidenediphenol (bisphenol A). Available online:
3602 http://esis.jrc.ec.europa.eu/doc/risk_assessment/REPORT/bisphenolareport325.pdf. 3rd Priority
3603 List, 37, 695 pp.

3604 EFSA (European Food Safety Authority), 2005. Opinion of the Scientific Committee on a request
3605 from EFSA related to Exposure assessments. The EFSA Journal, 2005, 249, 1-26.

3606 EFSA (European Food Safety Authority), 2006a. Opinion of the Scientific Panel on Food Additives,
3607 Flavourings, Processing Aids and Materials in Contact with Food on a request from the
3608 Commission related to 2,2-bis(4-hydroxyphenyl)propane (bisphenol A). The EFSA Journal, 2006,
3609 428, 1-75.

3610 EFSA (European Food Safety Authority), 2006b. Guidance of the Scientific Committee on a request
3611 from EFSA related to Uncertainties in dietary exposure assessment. The EFSA Journal, 2006, 438,
3612 1-54.

3613 EFSA (European Food Safety Authority), 2008. Scientific Opinion of the Panel on Food additives,
3614 Flavourings, Processing Aids and Materials in Contact with Food (AFC) on a request from the
3615 Commission on the toxicokinetics of Bisphenol A. The EFSA Journal 759, 1-10.

3616 EFSA (European Food Safety Authority), 2009. Guidance of the Scientific Committee on
3617 Transparency in the scientific aspects of risk assessments carried out by EFSA. The EFSA Journal,
3618 2009, 1051, 1-22.

3619 EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2010.
3620 Scientific Opinion on bisphenol A: evaluation of a study investigating its neurodevelopmental
3621 toxicity, review of recent scientific literature on its toxicity and advice on the Danish risk
3622 assessment of Bisphenol A, EFSA Journal 2010; 8(9):1829, 116 pp.

3623 EFSA (European Food Safety Authority), 2010. Scientific Report of EFSA: Management of left-
3624 censored data in dietary exposure assessment of chemical substances. EFSA Journal,
3625 2010;8(3):1557, 96 pp.

3626 EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2011a.
3627 Statement on the ANSES reports on bisphenol A. EFSA Journal 2011;9(12):2475, 10 pp.

3628 EFSA (European Food Safety Authority), 2011. Guidance of EFSA: Use of the EFSA comprehensive
3629 European food consumption database in exposure assessment. EFSA Journal, 2011;9(3):2097, 34
3630 pp.

3631 EFSA Panel on Food Contact Materials, Enzymes, Flavourings and Processing Aids (CEF), 2011b.
3632 Scientific opinion on the criteria to be used for safety evaluation of a mechanical recycling process
3633 to produce recycled PET intended to be used for manufacture of materials and articles in contact
3634 with food. EFSA Journal, 2011;9(7):2184, 25 pp.

3635 EFSA Panel on Plant Protection Products and their Residues (PPR), 2012. Guidance on the use of
3636 probabilistic methodology for modelling dietary exposure to pesticide residues. EFSA Journal,
3637 2012;10(10):2839, 95 pp.

3638 EFSA Scientific Committee (SC), 2012. Guidance on selected default values to be used by the EFSA
3639 Scientific Committee, Scientific Panels and Units in the absence of actual measured data. EFSA
3640 Journal, 2012;10(3):2579, 32 pp.

Ehlert KA, Beumer CW and Groot MC, 2008. Migration of bisphenol A into water from polycarbonate baby bottles during microwave heating. Food Additives and Contaminants: Part A, 25, 904-910.

Ehrlich S, Williams PL, Missmer SA, Flaws JA, Berry KF, Calafat AM, Ye X, Petrozza JC, Wright D and Hauser R, 2012. Urinary bisphenol A concentrations and implantation failure among women undergoing *in vitro* fertilization. Environmental Health Perspectives, 120, 978-983.

Eliades T, 2007. Orthodontic materials research and applications: part 2. Current status and projected future developments in materials and biocompatibility. American Journal of Orthodontics and Dentofacial Orthopedics, 131, 253-262.

Eliades T, Hiskia A, Eliades G and Athanasiou AE, 2007. Assessment of bisphenol-A release from orthodontic adhesives. American Journal of Orthodontics and Dentofacial Orthopedics, 131, 72-75.

Eliades T, Voutsa D, Sifakakis I, Makou M and Katsaros C, 2011. Release of bisphenol-A from a light-cured adhesive bonded to lingual fixed retainers. American Journal of Orthodontics and Dentofacial Orthopedics, 139, 192-195.

Elobeid MA, Almarhoon ZM, Virk P, Hassan ZK, Omer SA, ElAmin M, Daghestani MH and AlOlayan EM, 2012. Bisphenol A detection in various brands of drinking bottled water in Riyadh, Saudi Arabia using gas chromatography/mass spectrometer. Tropical Journal of Pharmaceutical Research, 13, 455-459.

Er B and Sarimehmetoğlu B, 2011. The Investigation of bisphenol A presence in canned tuna fish using high-performance-liquid chromatography method. Journal of Animal and Veterinary Advances, 10, 2859-2862.

Esty WW and Banfield JD, 2003. The box-percentile plot. Journal of Statistical Software, 8, 1-14.

FAO/WHO (Food and Agriculture Organization/World Health Organization), 2011. Toxicological and health aspects of bisphenol A. Proceedings of the Joint FAO/WHO Expert Meeting on Bisphenol A (BPA), Ottawa, Canada, 59 pp.

Fasano E, Bono-Blay F, Cirillo T, Montuori P and Lacorte S, 2012. Migration of phthalates, alkylphenols, bisphenol A and di(2-ethylhexyl)adipate from food packaging. Food Control, 27, 132-138.

Fenlon KA, Johnson AC, Tyler CR and Hill EM, 2010. Gas-liquid chromatography-tandem mass spectrometry methodology for the quantitation of estrogenic contaminants in bile of fish exposed to wastewater treatment works effluents and from wild populations. Journal of Chromatography A, 1217, 112-118.

Ferrer E, Santoni E, Vittori S, Font G, Mañes J and 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 Chemistry, 126, 360-367.

Feshin DB, Fimushkin PV, Brodskii ES, Shelepchikov AA, Mir-Kadyrova EY and Kalinkevich GA, 2012. Determination of bisphenol A in foods as 2,2-bis-(4-(isopropoxycarbonyloxy)phenyl)propane by gas chromatography/mass spectrometry. Journal of Analytical Chemistry, 67, 460-466.

Fleisch AF, Sheffield PE, Chinn C, Edelstein BL and Landrigan PJ, 2010. Bisphenol A and related compounds in dental materials. Pediatrics, 126, 760-768.

Flint S, Markle T, Thompson S and Wallace E, 2012. Bisphenol A exposure, effects, and policy: a wildlife perspective. Journal of Environmental Management, 104, 19-34.

Focazio MJ, Kolpin DW, Barnes KK, Furlong ET, Meyer MT, Zaugg SD, Barber LB and Thurman ME, 2008. A national reconnaissance for pharmaceuticals and other organic wastewater contaminants in the United States- II) untreated drinking water sources. Science of the Total Environment, 402, 201-216.

3689 Frederiksen H, Nielsen JK, Morck TA, Hansen PW, Jensen JF, Nielsen O, Andersson AM and
3690 Knudsen LE, 2013. Urinary excretion of phthalate metabolites, phenols and parabens in rural and
3691 urban Danish mother-child pairs. International Journal of Hygiene and Environmental Health, in
3692 press.

3693 Fu P and Kawamura K, 2010. Ubiquity of bisphenol A in the atmosphere. Environmental Pollution,
3694 158, 3138-3143.

3695 Fukata H, Miyagawa H, Yamazaki N and Mori C, 2006. Comparison of Elisa- and LC-MS-based
3696 methodologies for the exposure assessment of bisphenol A. Toxicology Mechanisms and Methods,
3697 16, 427-430.

3698 Fukazawa H, Hoshino K, Shiozawa T, Matsushita H and Terao Y, 2001. Identification and
3699 quantification of chlorinated bisphenol A in wastewater from wastepaper recycling plants.
3700 Chemosphere, 44, 973-979.

3701 Fung EY, Ewoldsen NO, St Germain HAJ, Marx DB, Miaw CL, Siew C, Chou HN, Gruninger SE and
3702 Meyer DM, 2000. Pharmacokinetics of bisphenol A released from a dental sealant. Journal of the
3703 American Dental Association, 131, 51-58.

3704 Gadelle D, Raibaud P and Sacquet E, 1985. β-Glucuronidase activities of intestinal bacteria
3705 determined both *in vitro* and *in vivo* in gnotobiotic rats. Applied and Environmental Microbiology,
3706 49, 682-685.

3707 Gaffney PT, Buttenshaw RL, Ward M and Diplock RD, 1986. Breast milk β-glucuronidase and
3708 neonatal jaundice. The Lancet, 327, 1161-1162.

3709 Gallard H, Leclercq A and Croué JP, 2004. Chlorination of bisphenol A: kinetics and by-products
3710 formation. Chemosphere, 56, 465-473.

3711 Gallart-Ayala H, Moyano E and Galceran MT, 2010a. On-line solid phase extraction fast liquid
3712 chromatography-tandem mass spectrometry for the analysis of bisphenol A and its chlorinated
3713 derivatives in water samples. Journal of Chromatography A, 1217, 3511-3518.

3714 Gallart-Ayala H, Nunez O, Moyano E and Galceran MT, 2010b. Field-amplified sample injection-
3715 micellar electrokinetic capillary chromatography for the analysis of bisphenol A, bisphenol F, and
3716 their diglycidyl ethers and derivatives in canned soft drinks. Electrophoresis, 31, 1550-1559.

3717 Gallart-Ayala H, Moyano E and Galceran MT, 2011. Analysis of bisphenols in soft drinks by on-line
3718 solid phase extraction fast liquid chromatography-tandem mass spectrometry. Analytica Chimica
3719 Acta, 683, 227-233.

3720 Galloway T, Cipelli R, Guralnik J, Ferrucci L, Bandinelli S, Corsi AM, Money C, McCormack P and
3721 Melzer D, 2010. Daily bisphenol A excretion and associations with sex hormone concentrations:
3722 results from the InCHIANTI adult population study. Environmental Health Perspectives, 118,
3723 1603-1608.

3724 Garcia-Prieto A, Lunar L, Rubio S and Pérez-Bendito D, 2008. Decanoic acid reverse micelle-based
3725 coacervates for the microextraction of bisphenol A from canned vegetables and fruits. Analytica
3726 Chimica Acta, 617, 51-58.

3727 Geens T, Roosens L, Neels H and Covaci A, 2009a. Assessment of human exposure to bisphenol-A,
3728 triclosan and tetrabromobisphenol-A through indoor dust intake in Belgium. Chemosphere, 76,
3729 755-760.

3730 Geens T, Neels H and Covaci A, 2009b. Sensitive and selective method for the determination of
3731 bisphenol-A and triclosan in serum and urine as pentafluorobenzoate-derivatives using GC-
3732 ECNI/MS. Journal of Chromatography B, 877, 4042-4046.

3733 Geens T, Apelbaum TZ, Goeyens L, Neels H and Covaci A, 2010. Intake of bisphenol A from canned
3734 beverages and foods on the Belgian market. Food Additives and Contaminants, 27, 1627-1637.

3735 Geens T, Goeyens L and Covaci A, 2011. Are potential sources for human exposure to bisphenol-A
3736 overlooked? International Journal of Hygiene and Environmental Health, 214, 339-347.

3737 Geens T, Aerts D, Berthot C, Bourguignon JP, Goeyens L, Lecomte P, Maghuin-Rogister G, Pironnet
3738 AM, Pussemier L, Scippo ML, Van Loco J and Covaci A, 2012a. A review of dietary and non-
3739 dietary exposure to bisphenol-A. Food and Chemical Toxicology, 50, 3725-3740.

3740 Geens T, Goeyens L, Kannan K, Neels H and Covaci A, 2012b. Levels of bisphenol-A in thermal
3741 paper receipts from Belgium and estimation of human exposure. Science of the Total Environment,
3742 435-436, 30-33.

3743 Gehring M, Vogel D, Tennhardt L, Weltin D and Bilitewski B, 2004. Bisphenol A contamination of
3744 wastepaper, cellulose and recycled paper products. In: Waste Management and the Environment II.
3745 WIT Transactions on Ecology and the Environment. Eds Brebbia CA, Kungulos S, Popov V and
3746 Itoh H, Southampton, Boston, Massachusetts, 294-302.

3747 Genuis SJ, Beesoon S, Birkholz D and Lobo RA, 2012. Human excretion of bisphenol A: blood, urine,
3748 and sweat (BUS) study. Journal of Environmental and Public Health, 2012, 10 pp.

3749 Gibson R, Becerril-Bravo E, Silva-Castro V and Jimenez B, 2007. Determination of acidic
3750 pharmaceuticals and potential endocrine disrupting compounds in wastewaters and spring waters
3751 by selective elution and analysis by gas chromatography-mass spectrometry. Journal of
3752 Chromatography A, 1169, 31-39.

3753 Gómez MJ, Mezcua M, Martinez MJ, Fernández-Alba AR and Agüera A, 2006. A new method for
3754 monitoring oestrogens, N-octylphenol, and bisphenol A in wastewater treatment plants by solid-
3755 phase extraction–gas chromatography–tandem mass spectrometry. International Journal of
3756 Environmental Analytical Chemistry, 86, 3-13.

3757 Gonzalez-Castro MI, Olea-Serrano MF, Rivas-Velasco AM, Medina-Rivero E, Ordonez-Acevedo LG
3758 and De Leon-Rodriguez A, 2011. Phthalates and bisphenols migration in Mexican food cans and
3759 plastic food containers. Bulletin of Environmental Contamination and Toxicology, 86, 627-631.

3760 Gourley GR and Arend RA, 1986. β-Glucuronidase and hyperbilirubinaemia in breast-fed and
3761 formula-fed babies. The Lancet, 22, 644-646.

3762 Grazioso CF and Buescher ES, 1996. Inhibition of neutrophil function by human milk. Cellular
3763 Immunology, 168, 125-132.

3764 Grumetto L, Montesano D, Seccia S, Albrizio S and Barbato F, 2008. Determination of bisphenol A
3765 and bisphenol B residues in canned peeled tomatoes by reversed-phase liquid chromatography.
3766 Journal of Agricultural and Food Chemistry, 56, 10633-10637.

3767 Guart A, Bono-Blay F, Borrell A and Lacorte S, 2011. Migration of plasticizers phthalates, bisphenol
3768 A and alkylphenols from plastic containers and evaluation of risk. Food Additives and
3769 Contaminants, 28, 676-685.

3770 Guerra P, Eljarrat E and Barcelo D, 2010. Simultaneous determination of hexabromocyclododecane,
3771 tetrabromobisphenol A, and related compounds in sewage sludge and sediment samples from Ebro
3772 River basin (Spain). Analytical and Bioanalytical Chemistry, 397, 2817-2824.

3773 Gyllenhammar I, Glynn A, Darnerud PO, Lignell S, van Delft R and Aune M, 2012. 4-Nonylphenol
3774 and bisphenol A in Swedish food and exposure in Swedish nursing women. Environment
3775 International, 43, 21-28.

3776 Hadjmohammadi MR and Saeidi I, 2010. Determination of bisphenol A in Iranian packaged milk by
3777 solid-phase extraction and HPLC. Monatshefte für Chemie, 141, 501-506.

3778 Hall B, Tozer S, Safford B, Coroama M, Steiling W, Leneveu-Duchemin MC, McNamara C and
3779 Gibney M, 2007. European consumer exposure to cosmetic products, a framework for conducting
3780 population exposure assessments. Food and Chemical Toxicology, 45, 2097-2108.

Han DH, Kim MJ, Jun EJ and Kim JB, 2012. Salivary bisphenol-A levels due to dental sealant/resin: a case-control study in Korean children. Journal of Korean Medical Science, 27, 1098-1104.

Hansch C, Leo A and Hoekman D, 1995. Exploring QSAR - Hydrophobic, Electronic, and Steric Constants. Washington, DC: American Chemical Society, 131 pp.

He Y, Miao M, Herrinton LJ, Wu C, Yuan W, Zhou Z and Li DK, 2009. Bisphenol A levels in blood and urine in a Chinese population and the personal factors affecting the levels. Environmental Research, 109, 629-633.

Health Canada (Bureau of Chemical Safety, Food Directorate, Health Products and Food Branch), 2010. Report on Human biomonitoring of environmental chemicals in Canada: Results of the Canadian health measures survey cycle 1 (2007-2009). Available online: http://www.hc-sc.gc.ca/ewh-semt/contaminants/human-humaine/chms-ecms-eng.php

Health Canada (Bureau of Chemical Safety, Food Directorate, Health Products and Food Branch), 2012. Health Canada's updated assessment of bisphenol A (BPA) exposure from food sources. 6 pp.

Hegnerová K and Homola J, 2010. Surface plasmon resonance sensor for detection of bisphenol A in drinking water. Sensors and Actuators B, 151, 177-179.

Heinze JE, 2011. Comment on "High levels of bisphenol A in paper currencies from several countries, and implications for dermal exposure". Environmental Science & Technology, 45, 9464.

Hengstler JG, Foth H, Gebel T, Kramer PJ, Lilienblum W, Schweinfurth H, Völkel W, Wollin KM and Gundert-Remy U, 2011. Critical evaluation of key evidence on the human health hazards of exposure to bisphenol A. Critical Reviews in Toxicology, 41, 263-291.

Hoekstra EJ and Simoneau C, 2013. Release of bisphenol A from polycarbonate: a review. Critical Reviews in Food Science and Nutrition, 53, 386-402.

Hohenblum P, Steinbichl P, Raffesberg W, Weiss S, Moche W, Vallant B, Scharf S, Haluza D, Moshammer H, Kundi M, Piegler B, Wallner P and Hutter HP, 2012. Pollution gets personal! A first population-based human biomonitoring study in Austria. International Journal of Hygiene and Environmental Health, 215, 176-179.

Honkanen JO and Kukkonen JVK, 2006. Environmental temperature changes uptake rate and bioconcentration factors of bisphenol a in tadpoles of *Rana temporaria*. Environmental Toxicology and Chemistry, 25, 2804-2808.

Hornung RW and Reed LD, 1990. Estimation of average concentration in the presence of nondetectable values. Applied Occupational and Environmental Hygiene, 5, 46-51.

Horwitz W, 2003. The certainty of uncertainty. Journal of AOAC International, 86, 109-111.

Hu Y, Zheng Y, Zhu F and Li G, 2007. Sol-gel coated polydimethylsiloxane/beta-cyclodextrin as novel stationary phase for stir bar sorptive extraction and its application to analysis of estrogens and bisphenol A. Journal of Chromatography A, 1148, 16-22.

Huang YQ, Wong CK, Zheng JS, Bouwman H, Barra R, Wahlstrom B, Neretin L and Wong MH, 2012. Bisphenol A (BPA) in China: a review of sources, environmental levels, and potential human health impacts. Environment International, 42, 91-99.

Ignatius CM, Francis EE, Emeka EN, Elvis NS and Ebele JI, 2010. BPA and environmental estrogen in potable water sources in Enugu municipality, South-East, Nigeria. Bulletin of Environmental Contamination and Toxicology, 85, 534-537.

IHS (Information Handling Services), 2013. Polycarbonate Resins. Available online: http://www.ihs.com/products/chemical/planning/ceh/polycarbonate-resins.aspx

IUCLID (International Uniform ChemicaL Information Database), 2000. IUCLID Dataset, European Chemicals Bureau. Available online: http://esis.jrc.ec.europa.eu/doc/IUCLID/data_sheets/80057.pdf

3828 Jackson WJ and Darnell WR, 1985. Process for foaming cellulose acetate rod. US Patent 4,507,256,
3829 USA.

3830 Jaddoe VW, Bakker R, van Duijn CM, van der Heijden AJ, Lindemans J, Mackenbach JP, Moll HA,
3831 Steegers EA, Tiemeier H, Uitterlinden AG, Verhulst FC and Hofman A, 2007. The Generation R
3832 Study Biobank: a resource for epidemiological studies in children and their parents. European
3833 Journal of Epidemiology, 22, 917-923.

3834 Jiang M, Zhang JH, Mei SR, Shi Y, Zou LJ, Zhu YX, Dai K and Lu B, 2006. Direct enrichment and
3835 high performance liquid chromatography analysis of ultra-trace bisphenol A in water samples with
3836 narrowly dispersible bisphenol A imprinted polymeric microspheres column. Journal of
3837 Chromatography A, 1110, 27-34.

3838 Jiang X, Tian W, Zhao C, Zhang H and Liu M, 2007. A novel sol-gel-material prepared by a surface
3839 imprinting technique for the selective solid-phase extraction of bisphenol A. Talanta, 72, 119-125.

3840 Jiao Y, Ding L, Fu S, Zhu S, Li H and Wang L, 2012. Determination of bisphenol A, bisphenol F and
3841 their diglycidyl ethers in environmental water by solid phase extraction using magnetic multiwalled
3842 carbon nanotubes followed by GC-MS/MS. Analytical Methods, 4, 291-298.

3843 Joas R, Casteleyn L, Biot P, Kolossa-Gehring M, Castano A, Angerer J, Schoeters G, Sepai O,
3844 Knudsen LE, Joas A, Horvat M and Bloemen L, 2012. Harmonised human biomonitoring in
3845 Europe: activities towards an EU HBM framework. International Journal of Hygiene and
3846 Environmental Health, 215, 172-175.

3847 Johnson BO, Burke FM, Harrison R and Burdette S, 2012. Quantitative analysis of bisphenol A
3848 leached from household plastics by solid–phase microextraction and gas chromatography–mass
3849 spectrometry (SPME–GC–MS). Journal of Chemical Education, 1555-1560.

3850 Jones-Otazo HA, Diamond ML and Richardson GM, 2005. An interagency comparison of screening-
3851 level risk assessment approaches. Risk Analysis, 25, 841-853.

3852 Josephson J, 2006. No dental dilemma for BPA. Environmental Health Perspectives, 114, A404.

3853 Joskow R, Boyd Barr D, Barr JR, Calafat AM, Needham LL and Rubin C, 2006. Exposure to
3854 bisphenol A from bis-glycidyl dimethacrylate–based dental sealants. Journal of the American
3855 Dental Association, 137, 253-262.

3856 JRC (Joint Research Centre), 2009. Guidelines for performance criteria and validation procedures of
3857 analytical methods used in controls of food contact materials. Available online:
3858 http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/1326/1/method_perf_guideline
3859 s_final_ed2009.pdf

3860 Juberg D, Alfano K, Coughlin RJ and Thomson KM, 2001. An observational study of object mouthing
3861 behavior by young children. Pediatrics, 107, 135-142.

3862 Kaddar N, Harthe C, Dechaud H, Mappus E and Pugeat M, 2008. Cutaneous penetration of bisphenol
3863 A in pig skin. Journal of Toxicology and Environmental Health, Part A: Current Issues, 71, 471-
3864 473.

3865 Kaddar N, Bendridi N, Harthe C, de Ravel MR, Bienvenu AL, Cuilleron CY, Mappus E, Pugeat M
3866 and Dechaud H, 2009. Development of a radioimmunoassay for the measurement of bisphenol A in
3867 biological samples. Analytica Chimica Acta, 645, 1-4.

3868 Kang JH and Kondo F, 2006a. Bisphenol A in the surface water and freshwater snail collected from
3869 rivers around a secure landfill. Bulletin of Environmental Contamination and Toxicology, 76, 113-
3870 118.

3871 Kang JH and Kondo F, 2006b. Distribution and biodegradation of bisphenol A in water hyacinth.
3872 Bulletin of Environmental Contamination and Toxicology, 77, 500-507.

3873 Kang JH, Kondo F and Katayama Y, 2006c. Human exposure to bisphenol A. Toxicology, 226, 79-89.

3874 Kang JH, Asai D and Katayama Y, 2007. Bisphenol A in the aquatic environment and its endocrine-
3875 disruptive effects on aquatic organisms. Critical Reviews in Toxicology, 37, 607-625.

3876 Kang YG, Kim JY, Kim J, Won PJ and Nam JH, 2011. Release of bisphenol A from resin composite
3877 used to bond orthodontic lingual retainers. American Journal of Orthodontics and Dentofacial
3878 Orthopedics, 140, 779-789.

3879 Kasper-Sonnenberg M, Wittsiepe J, Koch HM, Fromme H and Wilhelm M, 2012. Determination of
3880 bisphenol A in urine from mother-child pairs-results from the Duisburg birth cohort study,
3881 Germany. Journal of Toxicology and Environmental Health, Part A: Current Issues, 75, 429-437.

3882 Kawaguchi M, Ito R, Endo N, Okanouchi N, Sakui N, Saito K and Nakazawa H, 2006. Liquid phase
3883 microextraction with in situ derivatization for measurement of bisphenol A in river water sample
3884 by gas chromatography-mass spectrometry. Journal of Chromatography A, 1110, 1-5.

3885 KEMI (Swedish Chemicals Agency), 2012. Bisfenol A i leksaker och barnartiklar. 6/12, 55 pp.

3886 Kim K, Park H, Yang W and Lee JH, 2011. Urinary concentrations of bisphenol A and triclosan and
3887 associations with demographic factors in the Korean population. Environmental Research, 111,
3888 1280-1285.

3889 Kitada Y, Kawahata H, Suzuki A and Oomori T, 2008. Distribution of pesticides and bisphenol A in
3890 sediments collected from rivers adjacent to coral reefs. Chemosphere, 71, 2082-2090.

3891 Klecka G, Zabik J, Woodburn K, Naylor C, Staples C and Huntsman B, 2007. Exposure analysis of
3892 C8- and C9- alkylphenols, alkylphenol ethoxylates, and their metabolites in surface water systems
3893 within the United States. Human and Ecological Risk Assessment, 13, 792-822.

3894 Klecka GM, Staples CA, Clark KE, Van Der Hoeven N, Thomas DE and Hentges SG, 2009. Exposure
3895 analysis of bisphenol A in surface water systems in North America and Europe. Environmental
3896 Science & Technology, 43, 6145-6615.

3897 Kleywegt S, Pileggi V, Yang P, Hao C, Zhao X, Rocks C, Thach S, Cheung P and Whitehead B, 2011.
3898 Pharmaceuticals, hormones and bisphenol A in untreated source and finished drinking water in
3899 Ontario, Canada-occurrence and treatment efficiency. Science of the Total Environment, 409,
3900 1481-1488.

3901 Koch HM, Kolossa-Gehring M, Schroter-Kermani C, Angerer J and Bruning T, 2012. Bisphenol A in
3902 24 h urine and plasma samples of the German Environmental Specimen Bank from 1995 to 2009: a
3903 retrospective exposure evaluation. Journal of Exposure Science and Environmental Epidemiology,
3904 22, 610-616.

3905 Kodaira T, Kato I, Li J, Mochizuki T, Hoshino M, Usuki Y, Oguri H and Yanaihara N, 2000. Novel
3906 ELISA for the measurement of immunoreactive bisphenol A. Biomedical Research Tokyo, 21,
3907 117-121.

3908 Koh CH, Khim JS, Villeneuve DL, Kannan K and Giesy JP, 2006. Characterization of trace organic
3909 contaminants in marine sediment from Yeongil Bay, Korea: 1. Instrumental analyses.
3910 Environmental Pollution, 142, 39-47.

3911 Koldovský O, Palmieri M and Jumawan J, 1972. Comparison of activities of acid β-galactosidase, β-
3912 glucuronidase, n-acetyl-β-glucosaminidase and α-galactosidase in jejunum and ileum of adult and
3913 suckling rats. Comparative Biochemistry and Physiology, Part B, 43, 1-8.

3914 Kolossa-Gehring M, Becker K, Conrad A, Schroter-Kermani C, Schulz C and Seiwert M, 2012.
3915 Environmental surveys, specimen bank and health related environmental monitoring in Germany.
3916 International Journal of Hygiene and Environmental Health, 215, 120-126.

3917 Kontominas MG, Goulas AE, Badeka AV and Nerantzaki A, 2006. Migration and sensory properties
3918 of plastics-based nets used as food-contacting materials under ambient and high temperature
3919 heating conditions. Food Additives and Contaminants, 23, 634-641.

3920 Kosarac I, Kubwabo C, Lalonde K and Foster W, 2012. A novel method for the quantitative
3921 determination of free and conjugated bisphenol A in human maternal and umbilical cord blood
3922 serum using a two-step solid phase extraction and gas chromatography/tandem mass spectrometry.
3923 Journal of Chromatography B, 898, 90-94.

3924 Krapivin AS, Samsonova JV, Uskova NA, Ivanova NL and Egorov AM, 2007. Enzyme-linked
3925 immunosorbent assay for bisphenol A: Assay optimization and its application for surface water
3926 analysis. Toxicological and Environmental Chemistry, 89, 161-172.

3927 Kubwabo C, Kosarac I, Stewart B, Gauthier BR, Lalonde K and Lalonde PJ, 2009. Migration of
3928 bisphenol A from plastic baby bottles, baby bottle liners and reusable polycarbonate drinking
3929 bottles. Food Additives and Contaminants: Part A, 26, 928-937.

3930 Kuklenyik Z, Ekong J, Cutchins CD, Needham LL and Calafat AM, 2003. Simultaneous measurement
3931 of urinary bisphenol A and alkylphenols by automated solid-phase extractive derivatization gas
3932 chromatography/mass spectrometry. Analytical Chemistry, 75, 6820-6825.

3933 Kuroda N, Kinoshita Y, Sun Y, Wada M, Kishikawa N, Nakashima K, Makino T and Nakazawa H,
3934 2003. Measurement of bisphenol A levels in human blood serum and ascitic fluid by HPLC using a
3935 fluorescent labeling reagent. Journal of Pharmaceutical and Biomedical Analysis, 30, 1743-1749.

3936 Kuruto-Niwa R, Tateoka Y, Usuki Y and Nozawa R, 2007. Measurement of bisphenol A
3937 concentrations in human colostrum. Chemosphere, 66, 1160-1164.

3938 Lacroix MZ, Puel S, Collet SH, Corbel T, Picard-Hagen N, Toutain PL, Viguie C and Gayrard V,
3939 2011. Simultaneous quantification of bisphenol A and its glucuronide metabolite (BPA-G) in
3940 plasma and urine: applicability to toxicokinetic investigations. Talanta, 85, 2053-2059.

3941 Lakind JS and Naiman DQ, 2008. Bisphenol A (BPA) daily intakes in the United States: estimates
3942 from the 2003-2004 NHANES urinary BPA data. Journal of Exposure Science and Environmental
3943 Epidemiology, 18, 608-615.

3944 Lakind JS, Levesque J, Dumas P, Bryan S, Clarke J and Naiman DQ, 2012. Comparing United States
3945 and Canadian population exposures from National Biomonitoring Surveys: bisphenol A intake as a
3946 case study. Journal of Exposure Science and Environmental Epidemiology, 22, 219-226.

3947 Lassen C, Mikkelsen HS and Brandt UK (Danish Ministry of the Environment, Environmental
3948 Protection Agency), 2011. Migration of bisphenol A from cash register receipts and baby dummies.
3949 Survey of Chemical Substances in Consumer Product, 110 2011, 67 pp.

3950 Le HH, Carlson EM, Chua JP and Belcher SM, 2008. Bisphenol A is released from polycarbonate
3951 drinking bottles and mimics the neurotoxic actions of estrogen in developing cerebellar neurons.
3952 Toxicology Letters, 176, 149-156.

3953 Leclercq C, Arcella D, Piccinelli R, Sette S, Le Donne C and Turrini A, 2009. The Italian National
3954 Food Consumption Survey INRAN-SCAI 2005-06: main results in terms of food consumption.
3955 Public Health Nutrition, 12, 2504-2532.

3956 Lee YJ, Ryu HY, Kim HK, Min CS, Lee JH, Kim E, Nam BH, Park JH, Jung JY, Jang DD, Park EY,
3957 Lee KH, Ma JY, Won HS, Im MW, Leem JH, Hong YC and Yoon HS, 2008. Maternal and fetal
3958 exposure to bisphenol A in Korea. Reproductive Toxicology, 25, 413-419.

3959 Lewis RJS, 2001. Hawley's Condensed Chemical Dictionary. 147, New York, NY.

3960 Li X, Lin L, Zou S, Lan C and Luan T, 2006. Determination of bisphenol A in landfill leachate by
3961 solid phase microextraction with headspace derivatization and gas chromatography-mass
3962 spectrophotometry. Chinese Journal of Analytical Chemistry, 34, 325-328.

3963 Li L, Wang J, Zhou S and Zhao M, 2008. Development and characterization of an immunoaffinity
3964 monolith for selective on-line extraction of bisphenol A from environmental water samples.
3965 Analytica Chimica Acta, 620, 1-7.

Li Y and Liu J, 2010. Dispersive liquid–liquid microextraction based on ionic liquid in combination with high-performance liquid chromatography for the determination of bisphenol A in water. International Journal of Environmental Analytical Chemistry, 90, 880-890.

Li X, Ying GG, Su HC, Yang XB and Wang L, 2010. Simultaneous determination and assessment of 4-nonylphenol, bisphenol A and triclosan in tap water, bottled water and baby bottles. Environment International, 36, 557-562.

Li J, Kuang D, Feng Y, Zhang F and Liu M, 2011. Voltammetric determination of bisphenol A in food package by a glassy carbon electrode modified with carboxylated multi-walled carbon nanotubes. Microchimica Acta, 172, 379-386.

Li X, Ying GG, Zhao JL, Chen ZF, Lai HJ and Su HC, 2013. 4-Nonylphenol, bisphenol-A and triclosan levels in human urine of children and students in China, and the effects of drinking these bottled materials on the levels. Environment International, 52, 81-86.

Liao C and Kannan K, 2011a. Widespread occurrence of bisphenol A in paper and paper products: implications for human exposure. Environmental Science & Technology, 45, 9372-9379.

Liao C and Kannan K, 2011b. High levels of bisphenol A in paper currencies from several countries, and implications for dermal exposure. Environmental Science & Technology, 45, 6761-6768.

Liao C and Kannan K, 2011c. Reply to comment on "High levels of bisphenol A in paper currencies from several countries, and implications for dermal exposure". Environmental Science & Technology, 45, 9465-9466.

Liao C and Kannan K, 2012a. Determination of free and conjugated forms of bisphenol A in human urine and serum by liquid chromatography-tandem mass spectrometry. Environmental Science & Technology, 46, 5003-5009.

Liao C, Liu F and Kannan K, 2012b. Bisphenol S, a new bisphenol analogue, in paper products and currency bills and its association with bisphenol A residues. Environmental Science & Technology, 46, 6515-6522.

Liao C, Liu F, Guo Y, Moon HB, Nakata H, Wu Q and Kannan K, 2012. Occurrence of eight bisphenol analogues in indoor dust from the United States and several Asian countries: implications for human exposure. Environmental Science & Technology, 46, 9138-9145.

Lide DR, 1994. Handbook of Data on Organic Compounds. G.W.A. Milne (eds.). CRC Press, Inc. Boca Raton, FL, 4081 pp.

Lide DR, 2004. CRC handbook of chemistry and physics 85th edition. CRC, New York, NY, 2712 pp.

Lim LW and Takeuchi T, 2006. On-line precolumn enrichment of bisphenol A using boronate column in microcolumn liquid chromatography. Journal of Chromatography A, 1106, 139-145.

Lim DS, Kwack SJ, Kim KB, Kim HS and Lee BM, 2009a. Risk assessment of bisphenol A migrated from canned foods in Korea. Journal of Toxicology and Environmental Health, Part A: Current Issues, 72, 1327-1335.

Lim DS, Kwack SJ, Kim KB, Kim HS and Lee BM, 2009b. Potential risk of bisphenol A migration from polycarbonate containers after heating, boiling, and microwaving. Journal of Toxicology and Environmental Health, Part A: Current Issues, 72, 1285-1291.

Lin WC, Wang SL, Cheng CY and Ding WH, 2009. Determination of alkylphenol residues in breast and commercial milk by solid-phase extraction and gas chromatography-mass spectrometry. Food Chemistry, 114, 753-757.

Liu M, Hashi Y, Pan F, Yao J, Song G and Lin JM, 2006a. Automated on-line liquid chromatography-photodiode array-mass spectrometry method with dilution line for the determination of bisphenol A and 4-octylphenol in serum. Journal of Chromatography A, 1133, 142-148.

4011 Liu XY, Nakamura C, Tanimoto I, Miyake S, Nakamura N, Hirano T and Miyake J, 2006b. High
4012 sensitivity detection of bisphenol A using liposome chromatography. Analytica Chimica Acta, 578,
4013 43-49.

4014 Liu X, Ji Y, Zhang H and Liu M, 2008. Elimination of matrix effects in the determination of bisphenol
4015 A in milk by solid-phase microextraction-high-performance liquid chromatography. Food
4016 Additives and Contaminants: Part A, 25, 772-778.

4017 Loganathan SN and Kannan K, 2011. Occurrence of bisphenol A in indoor dust from two locations in
4018 the eastern United States and implications for human exposures. Archives of Environmental
4019 Contamination and Toxicology, 61, 68-73.

4020 Lopez-Espinosa MJ, Granada A, Araque P, Molina-Molina JM, Puertollano MC, Rivas A, Fernandez
4021 M, Cerrillo I, Olea-Serrano MF, Lopez C and Olea N, 2007. Oestrogenicity of paper and cardboard
4022 extracts used as food containers. Food Additives and Contaminants, 24, 95-102.

4023 Lorenz C, von Goetz N, Scheringer M, Wormuth M and Hungerbuhler K, 2011. Potential exposure of
4024 German consumers to engineered nanoparticles in cosmetics and personal care products.
4025 Nanotoxicology, 5, 12-29.

4026 Lu J, Wu J, Stoffella PJ and Chris Wilson P, 2012. Isotope dilution-gas chromatography/mass
4027 spectrometry method for the analysis of alkylphenols, bisphenol A, and estrogens in food crops.
4028 Journal of Chromatography A, 1258, 128-135.

4029 Lu J, Wu J, Stoffella PJ and Wilson PC, 2013. Analysis of bisphenol A, nonylphenol, and natural
4030 estrogens in vegetables and fruits using gas chromatography-tandem mass spectrometry. Journal of
4031 Agricultural and Food Chemistry, 61, 84-89.

4032 Lumley T, 2004. Analysis of complex survey samples. Journal of Statistical Software, 9, 1-19.

4033 Lumley T, 2012. Survey: analysis of complex survey samples. R package version 3.28-2.

4034 Magi E, Di Carro M and Liscio C, 2010. Passive sampling and stir bar sorptive extraction for the
4035 determination of endocrine-disrupting compounds in water by GC-MS. Analytical and
4036 Bioanalytical Chemistry, 397, 1335-1345.

4037 Mahalingaiah S, Meeker JD, Pearson KR, Calafat AM, Ye X, Petrozza J and Hauser R, 2008.
4038 Temporal variability and predictors of urinary bisphenol A concentrations in men and women.
4039 Environmental Health Perspectives, 116, 173-178.

4040 Maia J, Cruz JM, Sendón R, Bustos J, Sanchez JJ and Paseiro P, 2009. Effect of detergents in the
4041 release of bisphenol A from polycarbonate baby bottles. Food Research International, 42, 1410-
4042 1414.

4043 Maia J, Cruz JM, Sendón R, Bustos J, Cirugeda ME, Sanchez JJ and Paseiro P, 2010. Effect of amines
4044 in the release of bisphenol A from polycarbonate baby bottles. Food Research International, 43,
4045 1283-1288.

4046 Mao L, Sun C, Zhang H, Li Y and Wu D, 2004. Determination of environmental estrogens in human
4047 urine by high performance liquid chromatography after fluorescent derivatization with *p*-
4048 nitrobenzoyl chloride. Analytica Chimica Acta, 522, 241-246.

4049 Maragou NC, Lampi EN, Thomaidis NS and Koupparis MA, 2006. Determination of bisphenol A in
4050 milk by solid phase extraction and liquid chromatography-mass spectrometry. Journal of
4051 Chromatography A, 1129, 165-173.

4052 Maragou NC, Makri A, Lampi EN, Thomaidis NS and Koupparis MA, 2008. Migration of bisphenol
4053 A from polycarbonate baby bottles under real use conditions. Food Additives and Contaminants:
4054 Part A, 25, 373-383.

4055 Mariscal-Arcas M, Rivas A, Granada A, Monteagudo C, Murcia MA and Olea-Serrano F, 2009.
4056 Dietary exposure assessment of pregnant women to bisphenol-A from cans and microwave
4057 containers in Southern Spain. Food and Chemical Toxicology, 47, 506-510.

4058 Marquet F, Payan JP, Beydon D, Wathier L, Grandclaude MC and Ferrari E, 2011. *In vivo* and *ex vivo*
4059 percutaneous absorption of [^{14}C]-bisphenol A in rats: a possible extrapolation to human absorption?
4060 Archives of Toxicology, 85, 1035-1043.

4061 Martin MD, 2007. Exposure to bisphenol A (BPA) from dental sealants is detectable in saliva and
4062 urine, and varies significantly between sealant formulations. Journal of Evidence-Based Dental
4063 Practice, 7, 79-80.

4064 Martin-Esteban A and Tadeo JL, 2006. Selective molecularly imprinted polymer obtained from a
4065 combinatorial library for the extraction of bisphenol A. Combinatorial Chemistry & High
4066 Throughput Screening, 9, 747-751.

4067 Matsumoto H, Adachi S and Suzuki Y, 2005. Bisphenol A in ambient air particulates responsible for
4068 the proliferation of MCF-7 human breast cancer cells and its concentration changes over 6 months.
4069 Archives Environmental Contamination and Toxicology, 48, 459-466.

4070 McClatchey KD, 2002. Clinical laboratory medicine. Lippincott Williams and Wilkins, Philadelphia,
4071 PA, 1693 pp.

4072 Melzer D, Rice NE, Lewis C, Henley WE and Galloway TS, 2010. Association of urinary bisphenol a
4073 concentration with heart disease: evidence from NHANES 2003/06. PLoS One, 5, e8673.

4074 Mendum T, Stoler E, VanBenschoten H and Warner JC, 2011. Concentration of bisphenol A in
4075 thermal paper. Green Chemistry Letters and Reviews, 4, 81-86.

4076 Mercea P, 2009. Physicochemical processes involved in migration of bisphenol A from polycarbonate.
4077 Journal of Applied Polymer Science, 112, 579-593.

4078 Merten C, Ferrari P, Bakker M, Boss A, Hearty A, Leclercq C, Lindtner O, Tlustos C, Verger P,
4079 Volatier J and Arcella D, 2011. Methodological characteristics of the national dietary surveys
4080 carried out in the European Union as included in the European Food Safety Authority (EFSA)
4081 Comprehensive European Food Consumption Database. Food Additives and Contaminants: Part A,
4082 28, 975-995.

4083 Mielke H and Gundert-Remy U, 2009. Bisphenol A levels in blood depend on age and exposure.
4084 Toxicology Letters, 190, 32-40.

4085 Mielke H, Partosch F and Gundert-Remy U, 2011. The contribution of dermal exposure to the internal
4086 exposure of bisphenol A in man. Toxicology Letters, 204, 190-198.

4087 Milieu en Gezondheid (Steunpunt Milieu en Gezondheid), 2010. Vlaams Humaan
4088 Biomonitoringsprogramma 2007-2011. 206 pp.

4089 Mita L, Bianco M, Viggiano E, Zollo F, Bencivenga U, Sica V, Monaco G, Portaccio M, Diano N,
4090 Colonna A, Lepore M, Canciglia P and Mita DG, 2011. Bisphenol A content in fish caught in two
4091 different sites of the Tyrrhenian Sea (Italy). Chemosphere, 82, 405-410.

4092 Mohapatra DP, Brar SK, Tyagi RD and Surampalli RY, 2010. Physico-chemical pre-treatment and
4093 biotransformation of wastewater and wastewater sludge-fate of bisphenol A. Chemosphere, 78,
4094 923-941.

4095 Molina-Garcia L, Fernandez-de Cordova ML and Ruiz-Medina A, 2012. Analysis of Bisphenol A in
4096 milk by using a multicommuted fluorimetric sensor. Talanta, 96, 195-201.

4097 Moors S, Blaszkewicz M, Bolt HM and Degen GH, 2007. Simultaneous determination of daidzein,
4098 equol, genistein and bisphenol A in human urine by a fast and simple method using SPE and GC-
4099 MS. Molecular Nutrition and Food Research, 51, 787-798.

4100 Moreno MJ, D'Arienzo P, Manclus JJ and Montoya A, 2011. Development of monoclonal antibody-
4101 based immunoassays for the analysis of bisphenol A in canned vegetables. Journal of
4102 Environmental Science and Health, Part B, 46, 509-517.

4103 Morgan MK, Jones PA, Calafat AM, Ye X, Croghan CW, Chuang JC, Wilson NK, Clifton MS,
4104 Figueroa Z and Sheldon LS, 2011. Assessing the quantitative relationships between preschool

children's exposures to bisphenol A by route and urinary biomonitoring. Environmental Science & Technology, 45, 5309-5316.

Mudiam MK, Jain R, Dua VK, Singh AK, Sharma VP and Murthy RC, 2011. Application of ethyl chloroformate derivatization for solid-phase microextraction-gas chromatography-mass spectrometric determination of bisphenol-A in water and milk samples. Analytical and Bioanalytical Chemistry, 401, 1695-1701.

Nachman RM, Fox SD, Golden WC, Sibinga E, Veenstra TD, Groopman JD and Lees PS, 2013. Urinary free bisphenol A and bisphenol A-glucuronide concentrations in newborns. Journal of Pediatrics, 162, 870-872.

Nahar MS, Soliman AS, Colacino JA, Calafat AM, Battige K, Hablas A, Seifeldin IA, Dolinoy DC and Rozek LS, 2012. Urinary bisphenol A concentrations in girls from rural and urban Egypt: a pilot study. Environmental Health, 11, 1-8.

Nakada N, Tanishima T, Shinohara H, Kiri K and Takada H, 2006. Pharmaceutical chemicals and endocrine disrupters in municipal wastewater in Tokyo and their removal during activated sludge treatment. Water Research, 40, 3297-3303.

Nam SH, Seo YM and Kim MG, 2010. Bisphenol A migration from polycarbonate baby bottle with repeated use. Chemosphere, 79, 949-952.

Neville MC, Keller R, Seacat J, Lutes V, Neifert M, Casey C, Allen J and Archer P, 1988. Studies in human lactation: milk volumes in lactating women during the onset of lactation and full lactation. American Journal of Clinical Nutrition, 48, 1375-1386.

Neville MC and Walsh CT, 1996. Effects of drugs on milk secretion and composition. In: Drugs and Human Lactation. Elsevier, Amsterdam, 15-46.

NHANES (National Health and Nutrition Examination Survey), Available online: http://www.cdc.gov/nchs/nhanes.htm

Niu Y, Zhang J, Wu Y and Shao B, 2011. Simultaneous determination of bisphenol A and alkylphenol in plant oil by gel permeation chromatography and isotopic dilution liquid chromatography-tandem mass spectrometry. Journal of Chromatography A, 1218, 5248-5253.

Niu Y, Zhang J, Wu Y and Shao B, 2012. Analysis of bisphenol A and alkylphenols in cereals by automated on-line solid-phase extraction and liquid chromatography tandem mass spectrometry. Journal of Agricultural and Food Chemistry, 60, 6116-6122.

Noonan GO, Ackerman LK and Begley TH, 2011. Concentration of bisphenol A in highly consumed canned foods on the US market. Journal of Agricultural and Food Chemistry, 59, 7178-7185.

NTP-CERHR (National Toxicology Program US Department of Health and Human Services-Center for the Evaluation of Risks to Human Reproduction), 2008 Monograph on the potential human reproductive and developmental effects of bisphenol A. 321 pp.

O'Neil MJ, 2006. The Merck Index. In: An Encyclopedia of Chemicals, Drugs, and Biologicals. Whitehouse Station, NJ: Merck and Co., Inc, 211 pp.

Oca ML, Ortiz MC, Herrero A and Sarabia LA, 2013. Optimization of a GC/MS procedure that uses parallel factor analysis for the determination of bisphenols and their diglycidyl ethers after migration from polycarbonate tableware. Talanta, 106, 266-280.

Oehlmann J, Oetken M and Schulte-Oehlmann U, 2008. A critical evaluation of the environmental risk assessment for plasticizers in the freshwater environment in Europe, with special emphasis on bisphenol A and endocrine disruption. Environmental Research, 108, 140-149.

Oldring P, Castle L, O'Mahony C and Dixon J, 2013. Estimates of exposure to bisphenol A from light metal packaging, using food consumption and packaging usage data: a refined deterministic approach and a fully probabilistic (FACET) approach. Food Additives and Contaminants. Submitted.

4152 Olsen L, Lampa E, Birkholz DA, Lind L and Lind PM, 2012. Circulating levels of bisphenol A (BPA)
4153 and phthalates in an elderly population in Sweden, based on the Prospective Investigation of the
4154 Vasculature in Uppsala Seniors (PIVUS). Ecotoxicology and Environmental Safety, 75, 242-248.

4155 Östberg T and Noaksson E, 2010. BPA i svenska kvitton. Proceedings of the tillämpad grön KemI,
4156 Jämtlands läns landsting.

4157 Otaka H, Yasuhara A and Morita M, 2003. Determination of bisphenol A and 4-nonylphenol in human
4158 milk using alkaline digestion and cleanup by solid-phase extraction. Analytical Sciences, 19, 1663-
4159 1666.

4160 Ou J, Hu L, Hu L, Li X and Zou H, 2006. Determination of phenolic compounds in river water with
4161 on-line coupling bisphenol A imprinted monolithic precolumn with high performance liquid
4162 chromatography. Talanta, 69, 1001-1006.

4163 Ouchi K and Watanabe S, 2002. Measurements of bisphenol A in human urine using liquid
4164 chromatography with multi-channel coulometric electrochemical detection. Journal of
4165 Chromatography B, 780, 365-370.

4166 Ozaki A, Yamaguchi Y, Fujita T, Kuroda K and Endo G, 2004. Chemical analysis and
4167 genotoxicological safety assessment of paper and paperboard used for food packaging. Food and
4168 Chemical Toxicology, 42, 1323-1337.

4169 Ozaki A, Kawasaki C, Kawamura Y and Tanamoto K, 2006. Migration of bisphenol A and
4170 benzophenones from paper and paperboard products used in contact with food Journal of the Food
4171 Hygienic Society of Japan, 47, 99-104.

4172 Padmanabhan V, Siefert K, Ransom S, Johnson T, Pinkerton J, Anderson L, Tao L and Kannan K,
4173 2008. Maternal bisphenol-A levels at delivery: a looming problem? Journal of Perinatology, 28,
4174 258-263.

4175 Pandelova M, Piccinelli R, Lopez WL, Henkelmann B, Molina-Molina JM, Arrebola JP, Olea N,
4176 Leclercq C and Schramm KW, 2011. Assessment of PCDD/F, PCB, OCP and BPA dietary
4177 exposure of non-breast-fed European infants. Food Additives and Contaminants: Part A, 28, 1110-
4178 1122.

4179 Patterson TA, Twaddle NC, Roegge CS, Callicott RJ, Fisher JW and Doerge DR, 2013. Concurrent
4180 determination of bisphenol A pharmacokinetics in maternal and fetal rhesus monkeys. Toxicology
4181 and Applied Pharmacology, 267, 41-48.

4182 Pedersen SN and Lindholst C, 1999. Quantification of the xenoestrogens 4-tert-octylphenol and
4183 bisphenol A in water and in fish tissue based on microwave assisted extraction, solid phase
4184 extraction and liquid chromatography-mass spectrometry. Journal of Chromatography A, 864, 17-
4185 24.

4186 Peng X, Wang Z, Yang C, Chen F and Mai B, 2006. Simultaneous determination of endocrine-
4187 disrupting phenols and steroid estrogens in sediment by gas chromatography-mass spectrometry.
4188 Journal of Chromatography A, 1116, 51-56.

4189 Pérez Bendito MD, Bravo SR, Reyes ML and Prieto AG, 2009. Determination of bisphenol A in
4190 canned fatty foods by coacervative microextraction, liquid chromatography and fluorimetry. Food
4191 Additives and Contaminants: Part A, 26, 265-274.

4192 Pérez-Palacios D, Fernández-Recio MA, Moreta C and Tena MT, 2012. Determination of bisphenol-
4193 type endocrine disrupting compounds in food-contact recycled-paper materials by focused
4194 ultrasonic solid-liquid extraction and ultra performance liquid chromatography-high resolution
4195 mass spectrometry. Talanta, 99, 167-174.

4196 Pirard C, Sagot C, Deville M, Dubois N and Charlier C, 2012. Urinary levels of bisphenol A, triclosan
4197 and 4-nonylphenol in a general Belgian population. Environment International, 48, 78-83.

4198 Podlipna D and Cichna-Markl M, 2007. Determination of bisphenol A in canned fish by sol–gel
4199 immunoaffinity chromatography, HPLC and fluorescence detection. European Food Research and
4200 Technology, 224, 629-634.

4201 Polydorou O, Trittler R, Hellwig E and Kümmerer K, 2007. Elution of monomers from two
4202 conventional dental composite materials. Dental Materials, 23, 1535-1541.

4203 Polydorou O, Beiter J, Konig A, Hellwig E and Kümmerer K, 2009a. Effect of bleaching on the
4204 elution of monomers from modern dental composite materials. Dental Materials, 25, 254-260.

4205 Polydorou O, Hammad M, Konig A, Hellwig E and Kümmerer K, 2009b. Release of monomers from
4206 different core build-up materials. Dental Materials, 25, 1090-1095.

4207 Polydorou O, König A, Hellwig E and Kümmerer K, 2009c. Long-term release of monomers from
4208 modern dental-composite materials European Journal of Oral Sciences, 117, 68-75.

4209 Poustka J, Dunovská L, Hajšlová J, Holadová K and Poustková I, 2007. Determination and
4210 Occurrence of Bisphenol A, Bisphenol A Diglycidyl Ether, and Bisphenol F Diglycidyl Ether,
4211 Including Their Derivatives, in Canned Foodstuffs' from the Czech Retail Market. Czech Journal
4212 of Food Sciences, 25, 221-229.

4213 Pulgar R, Olea-Serrano MF, Novillo-Fertrell A, Rivas A, Pazos P, Pedraza V, Navajas JM and Olea N,
4214 2000. Determination of bisphenol A and related aromatic compounds released from bis-GMA-
4215 based composites and sealants by high performance liquid chromatography. Environmental Health
4216 Perspectives, 108, 21-27.

4217 Quintana JB, Rodil R, Muniategui-Lorenzo S, Lopez-Mahia P and Prada-Rodriguez D, 2007.
4218 Multiresidue analysis of acidic and polar organic contaminants in water samples by stir-bar
4219 sorptive extraction-liquid desorption-gas chromatography-mass spectrometry. Journal of
4220 Chromatography A, 1174, 27-39.

4221 R Core Team (The R Development Core Team), 2012. R: A language and environment for statistical
4222 computing. Version 2.11.1 (2010-05-31), 1706 pp.

4223 Rastkari N, Yunesian M and Ahmadkhaniha R, 2011. Levels of bısphenol A and bısphenol F In
4224 canned foods in iranıan markets. Iranian Journal of Environmental Health Science & Engineering,
4225 8, 95-100.

4226 Rathee M, Malik P and Singh J, 2012. Bisphenol A in dental sealants and its estrogen like effect.
4227 Indian Journal of Endocrinology and Metabolism, 16, 339-342.

4228 Renwick AG, 2008. Toxicokinetics: pharmacokinetics in toxicology. In: Principles and Methods of
4229 Toxicology. Ed Hayes AW, Informa Health Care, 178-203.

4230 Rød TO and Midtvedt T, 1977. Origin of intestinal β-glucuronidase in germfree, mono-contaminated
4231 and conventional rats. Acta Pathologica Et Microbiologica Scandinavica Section B-Microbiology,
4232 85, 271-276.

4233 Rudel RA, Camann DE, Spengler JD, Korn LR and Brody JG, 2003. Phthalates, alkylphenols,
4234 pesticides, polybrominated diphenyl ethers, and other endocrine-disrupting compounds in indoor
4235 air and dust. Environmental Science and Technology, 37, 4543-4553.

4236 Rudel RA, Dodson RE, Perovich LJ, Morello-Frosch R, Camann DE, Zuniga MM, Yau AY, Just AC
4237 and Brody JG, 2010. Semivolatile endocrine-disrupting compounds in paired indoor and outdoor
4238 air in two Northern California communities. Environmental Science & Technology, 44, 6583-6590.

4239 Rudney JD, Ji Z and Larson CJ, 1995. The prediction of saliva swallowing frequency in humans from
4240 estimates of salivary flow rate and the volume of saliva swallowed. Archives of Oral Biology, 40,
4241 507-512.

4242 Ruiz FJ, Rubio S and Pérez-Bendito D, 2007. Vesicular coacervative extraction of bisphenols and
4243 their diglycidyl ethers from sewage and river water. Journal of Chromatography A, 1163, 269-276.

Rykowska I and Wasiak W, 2006. Properties, threats, and methods of analysis of bisphenol A and its derivatives. Acta Chromatographica, 16, 7-27.

Saint L, Smith M and Hartmann PE, 1984. The yield and nutrient content of colostrum and milk of women from giving birth to 1 month post-partum. British Journal of Nutrition, 52, 87-95.

Sajiki J, 2003. Simple and accurate determination of bisphenol A in red blood cells prepared with basic glycine buffer using liquid chromatography electrochemical detection. Journal of Chromatography B, 783, 367-375.

Sajiki J, Miyamoto F, Fukata H, Mori C, Yonekubo J and Hayakawa K, 2007. Bisphenol A (BPA) and its source in foods in Japanese markets. Food Additives and Contaminants, 24, 103-112.

Sajiki J, Hasegawa Y, Hashimoto H, Makabe Y, Miyamoto F, Yanagibori R, Shin J, Shimidzu Y and Morigami T, 2008. Determination of bisphenol A (BPA) in plasma of hemodialysis patients using three methods: LC/ECD, LC/MS, and ELISA. Toxicology Mechanisms and Methods, 18, 733-738.

Salapasidou M, Samara C and Voutsa D, 2011. Endocrine disrupting compounds in the atmosphere of the urban area of Thessaloniki, Greece. Atmospheric Environment, 45, 3720-3729.

Salgueiro-Gonzalez N, Turnes-Carou I, Muniategui-Lorenzo S, Lopez-Mahia P and Prada-Rodriguez D, 2012a. Fast and selective pressurized liquid extraction with simultaneous in cell clean up for the analysis of alkylphenols and bisphenol A in bivalve molluscs. Journal of Chromatography A, 1270, 80-87.

Salgueiro-Gonzalez N, Concha-Grana E, Turnes-Carou I, Muniategui-Lorenzo S, Lopez-Mahia P and Prada-Rodriguez D, 2012b. Determination of alkylphenols and bisphenol A in seawater samples by dispersive liquid-liquid microextraction and liquid chromatography tandem mass spectrometry for compliance with environmental quality standards (Directive 2008/105/EC). Journal of Chromatography A, 1223, 1-8.

Sambe H, Hoshina K, Hosoya K and Haginaka J, 2006. Simultaneous determination of bisphenol A and its halogenated derivatives in river water by combination of isotope imprinting and liquid chromatography-mass spectrometry. Journal of Chromatography A, 1134, 16-23.

Sangiorgi G, Ferrero L, Ferrini BS, Lo Porto C, Perrone MG, Zangrando R, Gambaro A, Lazzati Z and Bolzacchini E, 2013. Indoor airborne particle sources and semi-volatile partitioning effect of outdoor fine PM in offices. Atmospheric Environment, 65, 205-214.

Santhi VA, Sakai N, Ahmad ED and Mustafa AM, 2012a. Occurrence of bisphenol A in surface water, drinking water and plasma from Malaysia with exposure assessment from consumption of drinking water. Science of the Total Environment, 427-428, 332-338.

Santhi VA, Hairin T and Mustafa AM, 2012b. Simultaneous determination of organochlorine pesticides and bisphenol A in edible marine biota by GC-MS. Chemosphere, 86, 1066-1071.

Santillana MI, Ruiz E, Nieto MT, Bustos J, Maia J, Sendon R and Sanchez JJ, 2011. Migration of bisphenol A from polycarbonate baby bottles purchased in the Spanish market by liquid chromatography and fluorescence detection. Food Additives and Contaminants: Part A, 28, 1610-1618.

Santoro WJ, Martinez FE, Ricco RG and Moyses Jorge S, 2010. Colostrum ingested during the first day of life by exclusively breastfed healthy newborn infants. Journal of Pediatrics, 156, 29-32.

Sarkar D, 2008. Lattice: Multivariate Data Visualization with R. Springer, New York, 265 pp.

Sasaki N, Okuda K, Kato T, Kakishima H, Okuma H, Abe K, Tachino H, Tuchida K and Kubono K, 2005. Salivary bisphenol-A levels detected by ELISA after restoration with composite resin. Journal of Materials Science: Materials in Medicine, 16, 297-300.

Schoeters G, Den Hond E, Colles A, Loots I, Morrens B, Keune H, Bruckers L, Nawrot T, Sioen I, De Coster S, Van Larebeke N, Nelen V, Van de Mieroop E, Vrijens J, Croes K, Goeyens K and

4290 Baeyens W, 2012. Concept of the Flemish human biomonitoring programme. International Journal
4291 of Hygiene and Environmental Health, 215, 102-108.

4292 Schoringhumer K and Cichna-Markl M, 2007. Sample clean-up with sol-gel enzyme and
4293 immunoaffinity columns for the determination of bisphenol A in human urine. Journal of
4294 Chromatography B, 850, 361-369.

4295 Segovia-Martinez L, Moliner-Martinez Y and Campins-Falco P, 2010. A direct Capillary Liquid
4296 Chromatography with electrochemical detection method for determination of phenols in water
4297 samples. Journal of Chromatography A, 1217, 7926-7930.

4298 Sendón García R, Sanches Silva A, Cooper I, Franz R and Paseiro Losada P, 2006. Revision of
4299 analytical strategies to evaluate different migrants from food packaging materials. Trends in Food
4300 Science and Technology, 17, 354-366.

4301 Shao B, Han H, Tu X and Huang L, 2007a. Analysis of alkylphenol and bisphenol A in eggs and milk
4302 by matrix solid phase dispersion extraction and liquid chromatography with tandem mass
4303 spectrometry. Journal of Chromatography B, 850, 412-416.

4304 Shao B, Han H, Li D, Ma Y, Tu X and Wu Y, 2007b. Analysis of alkylphenol and bisphenol A in
4305 meat by accelerated solvent extraction and liquid chromatography with tandem mass spectrometry.
4306 Food Chemistry, 105, 1236-1241.

4307 Shao M, Chen YH and Li XY, 2012. Consecutive online separation and determination of
4308 polybrominated diphenyl ethers, phthalate esters and bisphenol A in human serum by Gas
4309 Chromatography-Mass Spectrometry. Chinese Journal of Analytical Chemistry, 40, 1139-1146.

4310 Sheng N, Wei F, Zhan W, Cai Z, Du S, Zhou X, Li F and Hu Q, 2012. Dummy molecularly imprinted
4311 polymers as the coating of stir bar for sorptive extraction of bisphenol A in tap water. Journal of
4312 Separation Science, 35, 707-712.

4313 Simoneau C, Valzacchi S, Morkunas V and Van den Eede L, 2011. Comparison of migration from
4314 polyethersulphone and polycarbonate baby bottles. Food Additives and Contaminants: Part A, 28,
4315 1763-1768.

4316 Simoneau C, Van den Eede L and Valzacchi S, 2012. Identification and quantification of the migration
4317 of chemicals from plastic baby bottles used as substitutes for polycarbonate. Food Additives and
4318 Contaminants: Part A, 29, 469-480.

4319 Singh SP, Azua A, Chaudhary A, Khan S, Willett KL and Gardinali PR, 2010. Occurrence and
4320 distribution of steroids, hormones and selected pharmaceuticals in South Florida coastal
4321 environments. Ecotoxicology, 19, 338-350.

4322 Swedish National Board of Health and Welfare 2012. Bisphenol A in dental materials. Available
4323 online: http://www.socialstyrelsen.se/Lists/Artikelkatalog/Attachments/18772/2012-6-48.pdf. 59
4324 pp.

4325 Stackelberg PE, Gibs J, Furlong ET, Meyer MT, Zaugg SD and Lippincott RL, 2007. Efficiency of
4326 conventional drinking-water-treatment processes in removal of pharmaceuticals and other organic
4327 compounds. Science of the Total Environment, 377, 255-272.

4328 Stavrakakis C, Colin R, Hequet V, Faur C and Le Cloirec P, 2008. Analysis of endocrine disrupting
4329 compounds in wastewater and drinking water treatment plants at the nanogram per litre level.
4330 Environmental Technology, 29, 279-286.

4331 Stolzenberg H, Kahl H and Bergmann KE, 2007. Body measurements of children and adolescents in
4332 Germany. Results of the German Health Interview and Examination Survey for Children and
4333 Adolescents (KiGGS). Bundesgesundheitsblatt Gesundheitsforschung Gesundheitsschutz, 50, 659-
4334 669.

4335 Sun Y, Nakashima K, Takahashi M, Kuroda K and Nakashima K, 2002. Determination of bisphenol A
4336 in rat brain by microdialysis and column switching highperformance liquid chromatography with
4337 fluorescence detection. Biomedical Chromatography, 16, 319-326.

4338 Sun Y, Irie M, Kishikawa N, Wada M, Kuroda N and Nakashima K, 2004. Determination of bisphenol
4339 A in human breast milk by HPLC with column-switching and fluorescence detection. Biomedical
4340 Chromatography, 18, 501-507.

4341 Sun C, Leong LP, Barlow PJ, Chan SH and Bloodworth BC, 2006. Single laboratory validation of a
4342 method for the determination of bisphenol A, bisphenol A diglycidyl ether and its derivatives in
4343 canned foods by reversed-phase liquid chromatography. Journal of Chromatography A, 1129, 145-
4344 148.

4345 Szymański A, Rykowska I and Wasiak W, 2006. Determination of bisphenol A in water and milk by
4346 micellar liquid chromatography. Acta Chromatographica, 17, 161-172.

4347 Taylor JA, Vom Saal FS, Welshons WV, Drury B, Rottinghaus G, Hunt PA, Toutain PL, Laffont CM
4348 and VandeVoort CA, 2011. Similarity of bisphenol A pharmacokinetics in rhesus monkeys and
4349 mice: relevance for human exposure. Environmental Health Perspectives, 119, 422-430.

4350 Teeguarden JG, Calafat AM, Ye X, Doerge DR, Churchwell MI, Gunawan R and Graham MK, 2011.
4351 Twenty-four hour human urine and serum profiles of bisphenol a during high-dietary exposure.
4352 Toxicological Sciences, 123, 48-57.

4353 Teeguarden JG, Calafat AM and Doerge DR, 2012. Adhering to fundamental principles of
4354 biomonitoring, BPA pharmacokinetics, and mass balance is no ‘‘flaw’’. Toxicological Sciences,
4355 125, 321-325.

4356 Terasaki M, Shiraishi F, Fukazawa H and Makino M, 2007. Occurrence and estrogenicity of phenolics
4357 in paper-recycling process water: pollutants originating from thermal paper in waste paper.
4358 Environmental Toxicology and Chemistry. 26, 2356-2366.

4359 Thompson M, 2000. Recent trends in inter-laboratory precision at ppb and sub-ppb concentrations in
4360 relation to fitness for purpose criteria in proficiency testing. Analyst, 125, 385-386.

4361 Thompson M (Royal Society of Chemistry), 2004. The amazing Horwitz function. Available online:
4362 http://www.rsc.org/pdf/amc/brief17.pdf. amc technical brief, 2 pp.

4363 Tikuisis P, Meunier P and Jubenville CE, 2001. Human body surface area: measurement and
4364 prediction using three dimensional body scans. European Journal of Applied Physiology, 85, 264-
4365 271.

4366 Troiano R and Goodman W 2010. UHPLC separation and detection of bisphenol A (BPA) in plastics.
4367 Available online: http://www.perkinelmer.com/CMSResources/Images/44-
4368 74055app_BPAinToys.pdf

4369 Trudel D, Horowitz L, Wormuth M, Scheringer M, Cousins IT and Hungerbuhler K, 2008. Estimating
4370 consumer exposure to PFOS and PFOA. Risk Analysis, 28, 251-269.

4371 Tsai WT, 2006. Human health risk on environmental exposure to bisphenol-A: a review. Journal of
4372 Environmental Science and Health, Part C, 24, 225-255.

4373 UBA (Human Biomonitoring Commission of the German Federal Environment Agency), 2012.
4374 Substance monograph on bisphenol A (BPA)-reference and human biomonitoring (HBM) values
4375 for BPA in urine. Bundesgesundheitsblatt Gesundheitsforschung Gesundheitsschutz, 55, 1215-
4376 1231.

4377 Unal ER, Lynn T, Neidich J, Salazar D, Goetzl L, Baatz JE, Hulsey TC, Van Dolah R, Guillette LJJ
4378 and Newman R, 2012. Racial disparity in maternal and fetal-cord bisphenol A concentrations.
4379 Journal of Perinatology, 32, 844-850.

4380 US EPA (US Environmental Protection Agency), 2011. Exposure Factors Handbook. Available
4381 online: http://www.epa.gov/ncea/efh/pdfs/efh-complete.pdf. EPA/600/R-09/052F

4382 Valentin J, 2002. Basic anatomical and physiological data for use in radiological protection: reference
4383 values. Annals of the ICRP, 32, 1-277.

van Engelen JGM and Prud'homme de Lodder LCH (RIVM), 2007. Non-food products: How to assess children's exposure?, 75 pp.

Van Landuyt KL, Snauwaert J, De Munck J, Peumans M, Yoshida Y, Poitevin A, Coutinho E, Suzuki K, Lambrechts P and Van Meerbeek B, 2007. Systematic review of the chemical composition of contemporary dental adhesives. Biomaterials, 28, 3757-3785.

Van Landuyt KL, Nawrot T, Geebelen B, De Munck J, Snauwaert J, Yoshihara K, Scheers H, Godderis L, Hoet P and Van Meerbeek B, 2011. How much do resin-based dental materials release? A meta-analytical approach. Dental Materials, 27, 723-747.

Vandenberg LN, Hauser R, Marcus M, Olea N and Welshons WV, 2007. Human exposure to bisphenol A (BPA). Reproductive Toxicology, 24, 139-177.

Vandenberg LN, Chahoud I, Heindel JJ, Padmanabhan V, Paumgartten FJ and Schoenfelder G, 2010. Urinary, circulating, and tissue biomonitoring studies indicate widespread exposure to bisphenol A. Environmental Health Perspectives, 118, 1055-1070.

Vandenberg LN, Hunt PA, Myers JP and vom Saal FS, 2013. Human exposures to bisphenol A: mismatches between data and assumptions. Reviews on Environmental Health, 28, 37-58.

Vandentorren S, Zeman F, Morin L, Sarter H, Bidondo ML, Oleko A and Leridon H, 2011. Bisphenol-A and phthalates contamination of urine samples by catheters in the Elfe pilot study: implications for large-scale biomonitoring studies. Environmental Research, 111, 761-764.

Varelis P and Balafas D, 2000. Preparation of 4,4'-(1-[$^{2}H_{6}$]methylethylidene)bis-[2,3,5,6-$^{2}H_{4}$]phenol and its application to the measurement of bisphenol A in beverages by stable isotope dilution mass spectrometry. Journal of Chromatography A, 883, 163-170.

Viganò L, Mandich A, Benfenati E, Bertolotti R, Bottero S, Porazzi E and Agradi E, 2006. Investigating the estrogenic risk along the river Po and its intermediate section. Archives of Environmental Contamination and Toxicology, 51, 641-651.

Viñas P, Campillo N, Martinez-Castillo N and 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. Analytical and Bioanalytical Chemistry, 397, 115-125.

Viñas P, Lopez-Garcia I, Campillo N, Rivas RE and Hernandez-Cordoba M, 2012. Ultrasound-assisted emulsification microextraction coupled with gas chromatography-mass spectrometry using the Taguchi design method for bisphenol migration studies from thermal printer paper, toys and baby utensils. Analytical and Bioanalytical Chemistry, 404, 671-678.

VKM (Norwegian Scientific Committee for Food Safety), 2008. Assessment of four studies on developmental neurotoxicity of bisphenol A. Opinion of the Scientific Panel on Food Additives, Flavourings, Processing Aids, Materials in Contact with Food and Cosmetics, 08/404-6, 61 pp.

Völkel W, Colnot T, Csanady GA, Filser JG and Dekant W, 2002. Metabolism and kinetics of bisphenol A in humans at low doses following oral administration. Chemical Research in Toxicology, 15, 1281-1287.

Völkel W, Kiranoglu M and Fromme H, 2008. Determination of free and total bisphenol A in human urine to assess daily uptake as a basis for a valid risk assessment. Toxicology Letters, 179, 155-162.

Völkel W, Kiranoglu M and Fromme H, 2011. Determination of free and total bisphenol A in urine of infants. Environmental Research, 111, 143-148.

vom Saal FS, Nagel SC, Coe BL, Angle BM and Taylor JA, 2012. The estrogenic endocrine disrupting chemical bisphenol A (BPA) and obesity. Molecular and Cellular Endocrinology, 354, 74-84.

4430 von Goetz N, Wormuth M, Scheringer M and Hungerbuhler K, 2010. Bisphenol A: how the most
4431 relevant exposure sources contribute to total consumer exposure. Risk Analysis, 30, 473-487.

4432 Wan Y, Choi K, Kim S, Ji K, Chang H, Wiseman S, Johes PD, Khim JS, Park S, Park J, Lam MW and
4433 Giesy J, 2010. Hydroxylated polybrominated diphenyl ethers and bisphenol A in pregnant women
4434 and their matching fetuses: placental transfer and potential risks. Environmental Science &
4435 Technology, 44, 5233-5239.

4436 Wang X, Zeng H, Zhao L and Lin J-M, 2006. Selective determination of bisphenol A (BPA) in water
4437 by a reversible fluorescence sensor using pyrene/dimethyl β-cyclodextrin complex. Analytica
4438 Chimica Acta, 556, 313-318.

4439 Wang X, Diao CP and Zhao RS, 2009. Rapid determination of bisphenol A in drinking water using
4440 dispersive liquid-phase microextraction with in situ derivatization prior to GC-MS. Journal of
4441 Separation Science, 32, 154-159.

4442 Watanabe S, Ohnishi M, Imai K, Kawano E and Igarashi S, 1995. Estimation of the total saliva
4443 volume produced per day in five-year-old children. Archives of Oral Biology, 40, 781-782.

4444 Watanabe T, Yamamoto H, Inoue K, Yamaguchi A, Yoshimura Y, Kata K, Nakazawa H, Kuroda K
4445 and Nakashima K, 2001. Development of sensitive high-performance liquid chromatography with
4446 fluorescence detection using 4-(4,5-diphenyl-1H-imidazol-2-yl)-benzoyl chloride as a labeling
4447 reagent for determination of bisphenol A in plasma samples. Journal of Chromatography B, 762, 1-
4448 7.

4449 Wei X, Huang Y, Wong MH, Giesy JP and Wong CK, 2011. Assessment of risk to humans of
4450 bisphenol A in marine and freshwater fish from Pearl River Delta, China. Chemosphere, 85, 122-
4451 128.

4452 Wells EM, Jackson LW and Koontz MB, 2013. Decline in urinary bisphenol A concentrations in the
4453 United States. Epidemiology, 24, 167-168.

4454 Wen Y, Zhou BS, Xu Y, Jin SW and Feng YQ, 2006. Analysis of estrogens in environmental waters
4455 using polymer monolith in-polyether ether ketone tube solid-phase microextraction combined with
4456 high-performance liquid chromatography. Journal of Chromatography A, 1133, 21-28.

4457 Weschler CJ and Nazaroff WW, 2012. SVOC exposure indoors: fresh look at dermal pathways.
4458 Indoor Air, 22, 356-377.

4459 WHO (World Health Organization), 2003. Domestic water quantity, service level and health.
4460 Available online: http://www.who.int/water_sanitation_health/diseases/WSH03.02.pdf

4461 WHO (World Health Organization), 2009. Principles and methods for the risk assessment of
4462 chemicals in food. Available online:
4463 http://www.who.int/foodsafety/chem/principles/en/index1.html. Environmental health criteria; 240.

4464 WHO (World Health Organization), 2011a. Background Paper on Sources and occurrence of
4465 bisphenol A relevant for exposure of consumers. Proceedings of the FAO/WHO Expert Meeting on
4466 Bisphenol A (BPA), Ottawa, Canada, 2-5 November 2010, 69 pp.

4467 WHO (World Health Organization), 2011b. Background Paper on Chemistry and analytical methods
4468 for determination of bisphenol A in food and biological samples. Proceedings of the FAO/WHO
4469 Expert Meeting on Bisphenol A (BPA), Ottawa, Canada, 2-5 November 2010, 29 pp.

4470 Willock J and Jewkes F, 2000. Making sense of fluid balance in children. Paediatric Nursing, 12, 37-
4471 42.

4472 Wilson NK, Chuang JC, Morgan MK, Lordo RA and Sheldon LS, 2007. An observational study of the
4473 potential exposures of preschool children to pentachlorophenol, bisphenol-A, and nonylphenol at
4474 home and daycare. Environmental Research, 103, 9-20.

4475 Wu SY, Xu Q, Chen TS, Wang M, Yin XY, Zhang NP, Shen YY, Wen ZY and Gu ZZ, 2010.
4476 Determination of bisphenol A in plastic bottled drinking water by high performance liquid

4477 chromatography with solid-membrane extraction based on electrospun nylon 6 nanofibrous
4478 membrane. Chinese Journal of Analytical Chemistry, 38, 503-507.

4479 WUR (Rijkswaterstaat Institute for Coastal and Marine Management (RIKZ)), 2001. Chemical study
4480 on bisphenol A. Available online: http://edepot.wur.nl/174301. RIKZ/2001.027, 94 pp.

4481 Xiao J, Shao B, Wu XY, Sun XJ and Wu YN, 2011. A study on bisphenol A, nonylphenol, and
4482 octylphenol in human urine samples detected by SPE-UPLC-MS. Biomedical and Environmental
4483 Sciences, 24, 40-46.

4484 Yamamoto T and Yasuhara A, 2002. Chlorination of bisphenol A in aqueous media: formation of
4485 chlorinated bisphenol A congeners and degradation to chlorinated phenolic compounds.
4486 Chemosphere, 46, 1215-1223.

4487 Yang M, Park MS and Lee HS, 2006. Endocrine disrupting chemicals: human exposure and health
4488 risks. Journal of Environmental Science and Health, Part C 24, 183-224.

4489 Yang YJ, Hong YC, Oh SY, Park MS, Kim H, Leem JH and Ha EH, 2009. Bisphenol A exposure is
4490 associated with oxidative stress and inflammation in postmenopausal women. Environmental
4491 Research, 109, 797-801.

4492 Ye X, Kuklenyik Z, Needham LL and Calafat AM, 2006. Measuring environmental phenols and
4493 chlorinated organic chemicals in breast milk using automated on-line column-switching-high
4494 performance liquid chromatography-isotope dilution tandem mass spectrometry. Journal of
4495 Chromatography B, 831, 110-115.

4496 Ye X, Pierik FH, Hauser R, Duty S, Angerer J, Park MM, Burdorf A, Hofman A, Jaddoe VW,
4497 Mackenbach JP, Steegers EA, Tiemeier H and Longnecker MP, 2008a. Urinary metabolite
4498 concentrations of organophosphorous pesticides, bisphenol A, and phthalates among pregnant
4499 women in Rotterdam, the Netherlands: the Generation R study. Environmental Research, 108, 260-
4500 267.

4501 Ye X, Tao LJ, Needham LL and Calafat AM, 2008b. Automated on-line column-switching HPLC-
4502 MS/MS method for measuring environmental phenols and parabens in serum. Talanta, 76, 865-
4503 871.

4504 Ye X, Bishop AM, Needham LL and Calafat AM, 2008c. Automated on-line column-switching
4505 HPLC-MS/MS method with peak focusing for measuring parabens, triclosan, and other
4506 environmental phenols in human milk. Analytica Chimica Acta, 622, 150-156.

4507 Ye X, Pierik FH, Angerer J, Meltzer HM, Jaddoe VW, Tiemeier H, Hoppin JA and Longnecker MP,
4508 2009a. Levels of metabolites of organophosphate pesticides, phthalates, and bisphenol A in pooled
4509 urine specimens from pregnant women participating in the Norwegian Mother and Child Cohort
4510 Study (MoBa). International Journal of Hygiene and Environmental Health, 212, 481-491.

4511 Ye X, Wong LY, Jia LT, Needham LL and Calafat AM, 2009b. Stability of the conjugated species of
4512 environmental phenols and parabens in human serum. Environment International, 35, 1160-1163.

4513 Ye X, Wong LY, Bishop AM and Calafat AM, 2011. Variability of urinary concentrations of
4514 bisphenol A in spot samples, first morning voids, and 24-hour collections. Environmental Health
4515 Perspectives, 119, 983-988.

4516 Ye X, Zhou X, Wong LY and Calafat AM, 2012. Concentrations of bisphenol A and seven other
4517 phenols in pooled sera from 3-11 year old children: 2001-2002 National Health and Nutrition
4518 Examination Survey. Environmental Science & Technology, 46, 12664-12671.

4519 Yi B, Kim C and Yang M, 2010. Biological monitoring of bisphenol A with HLPC/FLD and
4520 LC/MS/MS assays. Journal of Chromatography B, 878, 2606-2610.

4521 Yonekubo J, Hayakawa K and Sajiki J, 2008. Concentrations of bisphenol A, bisphenol A diglycidyl
4522 ether, and their derivatives in canned foods in Japanese markets. Journal of Agricultural and Food
4523 Chemistry, 56, 2041-2047.

4524 Zafra-Gómez A, Ballesteros O, Navalón A and Vílchez JL, 2008. Determination of some endocrine
4525 disrupter chemicals in urban wastewater samples using liquid chromatography–mass spectrometry.
4526 Microchemical Journal, 88, 87-94.

4527 Zafra-Gómez A, Morales JC, Ballesteros O and Navalón A, 2009. Sensitive gas chromatographic-
4528 mass spectrometric (GC-MS) method for the determination of bisphenol A in rice-prepared dishes.
4529 Food Additives and Contaminants: Part A, 26, 1209-1216.

4530 Zalko D, Jacques C, Duplan H, Bruel S and Perdu E, 2011. Viable skin efficiently absorbs and
4531 metabolizes bisphenol A. Chemosphere, 82, 424-430.

4532 Zhang ZL, Hibberd A and Zhou JL, 2006. Optimisation of derivatisation for the analysis of estrogenic
4533 compounds in water by solid-phase extraction gas chromatography-mass spectrometry. Analytica
4534 Chimica Acta, 577, 52-61.

4535 Zhang Z, Alomirah H, Cho HS, Li YF, Liao C, Minh TB, Mohd MA, Nakata H, Ren N and Kannan K,
4536 2011a. Urinary bisphenol A concentrations and their implications for human exposure in several
4537 Asian countries. Environmental Science & Technology, 45, 7044-7050.

4538 Zhang HC, Yu XJ, Yang WC, Peng JF, Xu T, Yin DQ and Hu XL, 2011b. MCX based solid phase
4539 extraction combined with liquid chromatography tandem mass spectrometry for the simultaneous
4540 determination of 31 endocrine-disrupting compounds in surface water of Shanghai. Journal of
4541 Chromatography B, 879, 2998-3004.

4542 Zheng Z, Du Y, Wang Z, Feng Q and Wang C, 2013. Pt/graphene-CNTs nanocomposite based
4543 electrochemical sensors for the determination of endocrine disruptor bisphenol A in thermal
4544 printing papers. Analyst, 138, 693-701.

4545 Zhu R, Zhao W, Zhai M, Wei F, Cai Z, Sheng N and Hu Q, 2010. Molecularly imprinted layer-coated
4546 silica nanoparticles for selective solid-phase extraction of bisphenol A from chemical cleansing and
4547 cosmetics samples. Analytica Chimica Acta, 658, 209-216.

4548 Zimmerman-Downs JM, Shuman D, Stull SC and Ratzlaff RE, 2010. Bisphenol A blood and saliva
4549 levels prior to and after dental sealant placement in adults. Journal of Dental Hygiene, 84, 145-150.

4550

4551

4552 **ABBREVIATIONS**

ABS	Acrylonitrile-butadiene-styrene
AM	Arithmetic mean
BADGE	Bisphenol A-diglycidyl ether
bw	body weight
CI	Confidence interval
DMA	Dimethacrylate
ECD	Electrochemical detection
EEA	European Economic Area
EMA	Ethoxylate dimethacrylate
EU	European Union
FDA	Food and Drug Administration
FLD	Fluorescence detection
GC	Gas chromatography
GM	Geometric mean
GMA	Glycidyl methacrylate
GSD	Geometric standard deviation
HDPE	High density polyethylene
HPLC	High performance liquid chromatography
LB	Lower bound
LC	Liquid chromatography
LDPE	Low density polyethylene
LOD	Limit of detection
LOQ	Limit of quantification
MB	Middle bound
MS	mass spectrometry
PA	Polyamide
PBPK	Physiologically based pharmacokinetic modelling
PC	Polycarbonate
PEI	Polyetherimides
PES	Polyethersulfone
PM10	Particulate Matter with diameter less then 10 µm
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinylchloride
RIA	Radio-Immunoassay
SCENIHR	Scientific Committee on emerging and newly identified health risks
SML	Specific migration limit
TBBPA	Tetrabromobisphenol A
UB	Upper bound
US	United States
UV	Ultraviolett

APPENDICES

APPENDIX I: SAMPLING AND METHODS OF ANALYSIS

The Appendix I describes the criteria considered for the inclusion of data in the assessment of the exposure to BPA, as well as for assessment of the quality of the biomonitoring studies.

When considering the inclusion of data in the assessment of the exposure to BPA it is essential that the methodology used to derive the data is of an appropriate quality. This Appendix describes the quality criteria applied to ensure, as far as possible, the quality of the data considered in this opinion.

The criteria for inclusion/exclusion of data (and methodology) for consideration for the opinion for BPA are given below and are based on the performance characteristics of the method. Performance characteristic means functional quality that can be attributed to an analytical method. This may be for instance specificity, accuracy, trueness, precision, repeatability, reproducibility, recovery, detection capability and ruggedness. The JRC guidelines on performance criteria and validation procedures of analytical methods used in controls of food contact materials" were used as the basis to define the criteria for all methods considered in this opinion (JRC, 2009).

In terms of method performance the main criteria to consider are:

- The recovery of the method
- The repeatability of the method
- The limit of detection and/or limit of quantification

Recovery

Recovery means the percentage of the true concentration of a substance recovered during the analytical procedure. For inclusion the recovery should be in a range, as described in Table 38:

Table 38: Ranges of recovery

Concentration	Mean recovery (%)
≤ 10 parts per billion (ppb, µg/kg)	40 – 120
100-10 ppb	60 – 110
≥ 100 ppb	80 – 110

For the purpose of the exposure assessment in this report, data were not corrected for the recovery. Correction for recovery is aimed at reducing the uncertainty in concentration data, but since the technique used to estimate it varies among laboratories, such a correction may at the end introduce even more uncertainty in the concentration data. Data derived from analytical determinations with recoveries outside the above mentioned criteria were excluded.

Repeatability

Repeatability is defined (IUPAC) as precision under repeatability conditions (i.e. same operator, instrument, laboratory, and within a short time interval). Repeatability (r) is often expressed as a relative standard deviation RSD_r (%) derived from replicate analyses of either a certified reference

material or a fortified material. For inclusion of data the criteria applied was that the repeatability (RSD_r) should not exceed the level calculated by the Horwitz Equation. The Horwitz equation actually describes the reproducibility (R) between different labs as a function of concentration and expressed as relative standard deviation RSD_R (%). Setting the reproducibility measure (RSD_R) as the limit for the repeatability (RSD_r) is explained by the fact that the RSD_r is generally one-half to two-thirds of the RSD_R. For very low concentrations, the reproducibility is somewhat better than expected from the Horwitz equation and approaches a constant value of 33 % (Horwitz, 2003). Similarly, Thompson (2000, 2004) concluded an invariant value of 20–25 % for concentrations below 10 ppb. In the Table 39 a limit value of 25 % was chosen for concentrations of 1 and 10 ppb.

Table 39: The RSD calculated using the Horwitz equation for concentration range from 1ppb to 1 ppm

Concentration	Relative standard deviation (RSD_r, %)
1 ppb	25*
10 ppb	25*
100 ppb	22.6
1 ppm	16.0

* The RSD calculated using the Horwitz equation is > 25 %. However it has been shown (Horwitz, 2003, Thompson 2000, 2004) that at concentrations of less than 10 ppb there is a tendency for an invariant RSD of 20-25 % and so 25 % was selected as the criteria for acceptable repeatability.

Limit of detection/limit of quantification

Analytical limits of detection are usually expressed as multiples of the signal-to-noise ratio (S/N) of the (chromatographic) background signal with the limit of detection (LOD) being 3×S/N and the limit of quantification being 10×S/N. In some biomonitoring studies reporting the unavoidable presence of background BPA contamination (e.g. Völkel et al., 2011), somewhat higher multiples of the S/N are used to report only values above the background contamination.

Food samples below the limit of quantification or reporting (left-censoring limit) were handled through the substitution method: the lower bound (LB) value was obtained by assigning a value of zero to all the samples reported as less than the left-censoring limit, the middle bound (MB) value by assigning half of the left-censoring limit and the upper bound (UB) by assigning the left-censored limit as the sample result (see paragraph 4.3.3. Occurrence data in food). The average BPA concentration in each food category was therefore assessed as LB, MB and UB. Therefore, in a study where all samples give a quantifiable BPA concentration, the limits of detection and quantification are of no relevance in the assessment of average LB, MB or UB BPA concentrations. In a study in which BPA concentrations are reported in some samples as < LOD or < LOQ, the MB and UB average BPA concentration of the specific food category will be influenced by the left-censoring limits, and this will influence the assessment of exposure to BPA. Criteria were therefore set to avoid the possibility that samples with a very high left-censoring limit would artificially increase the assessment of average MB and UB BPA concentration in some food categories. For occurrence data in food, methods reporting LOD values greater than 15 µg/kg or LOQ values greater than 50 µg/kg were excluded from the assessment of average BPA concentration, and therefore from the exposure assessment. For biomonitoring data methods reporting LOD values greater than 0.4 µg/kg or LOQ values greater than 1.3 µg/kg were excluded from the exposure assessment.

Supplementary criteria to be considered when assessing method performance were:

- The selectivity of the method, i.e. whether or not interferences had been considered (e.g. Ackermann et al., 2010)
- Whether or not measures had been taken to reduce or avoid background contamination
- Whether or not the method-performance data described have been derived for an appropriate matrix and at a concentration relevant to the levels measured in the samples

Specifically for biomonitoring studies, it is necessary to detect and quantify BPA in different biological matrices (urine, serum, human milk) in the unconjugated and the conjugated form. Complicating problems for all of these matrices are the (i) the artefactual contamination with trace levels of unconjugated BPA from environmental sources and (ii) the instability of BPA conjugates due to spontaneous or enzymatic hydrolysis during collection, storage and analysis (Vandenberg et al., 2010; Hengstler et al., 2011; WHO, 2011a). Therefore, the documentation of measures to preserve sample integrity and to reduce external contamination was taken into account when deciding whether a study is considered valid and relevant to be included for this opinion.

Many different approaches have been reported for the determination of BPA and conjugated BPA. These are reviewed in Chapter 4.2.3. One of these approaches involves the use of enzyme-linked immunosorbent assay (ELISA). ELISA kits for the determination of BPA in biological samples are commercially available and have been used to determine BPA levels in such matrices. In such cases the selectivity of the ELISA technique should be considered. ELISA cannot differentiate between conjugated and unconjugated BPA and it has also been reported that cross-reactivity occurs with other structurally similar substances. In this evaluation data generated for biological samples derived using ELISA methodology were only included where there was a data gap and in all cases the data derived using this technique were considered with caution. Specific examples are included in the narrative in Chapter 4.7.

Samples

No quality criteria were established for sampling methods. The country of origin of the samples was considered and, in most cases, non-EU data were excluded (see Chapter 4.2). Where information was provided samples taken for determination of BPA concentration or of migration of BPA were considered to be representative of those available on the market. However in many cases this information was not given.

Methods of analysis

The approach used to extract BPA from any sample (including all of the potential sources of exposure given in Chapter 3) is dependent on the matrix being tested. Methodology typically involves extraction of the analyte from the matrix and may be followed by clean-up of the extracts to eliminate any interferences, concentration to achieve the desired method sensitivity and/or derivatisation to provide BPA in a form suitable for analysis. Analytical approaches described in the literature include: liquid chromatography (LC) with ultra-violet (UV), fluorescence (FLD), electrochemical (ECD) or mass spectrometric (MS or MS/MS) detection, gas chromatography (GC) with MS detection and immunoaffinity methods (e.g. enzyme-linked immunosorbent assays - ELISA). An overview of the methodology for the determination of BPA in and migrating from food contact materials, in foods, in biological samples, in non-food potential sources of exposure (including outdoor air, surface water, dust, indoor air, paper products, children toys and pacifiers with PC shield and medical devices) is presented below. Ballesteros-Gómez et al. (2009) reviewed methods describing the determination of BPA in foods and a WHO/FAO background paper on "Chemistry and Analytical Methods for Determination of BPA in Food and Biological Samples" was prepared by Cao (WHO, 2011b).

Extraction and migration of BPA from food contact materials

Material types that may contain BPA and that are used in food contact applications include PC plastics, epoxy coatings applied to metal substrates and recycled paper and board. To extract all of the residual BPA from a material or article requires some degree of interaction between the material and the extraction solvent. This interaction, referred to for plastics as swelling of the polymer, allows for extraction from the entire material rather than just from the surface. For polar materials such as paper and board and polycarbonate plastics the greatest interaction occurs with polar solvents. For less polar materials such as epoxy resins the greatest interaction occurs with less polar solvents. The solubility of the BPA in the extraction solvent must also be considered. BPA is soluble in acetic acid and is very soluble in ethanol, benzene and diethyl ether (Lide, 2004). Only a limited number of methods have been reported for the determination of BPA in food contact materials and articles as in most cases a migration test into a food simulant or solvent rather than an exhaustive extraction has been carried out.

Extraction tests – Mercea (2009) and Ehlert et al. (2008) described the determination of residual BPA in PC by dissolution of the polymer in dichloromethane followed by subsequent precipitation with methanol. Dissolution of PC in methylene chloride and precipitation with acetone has also been described to determine residual BPA concentration in the polymer (Nam et al, 2010). In such studies all of the BPA will remain in solution and so is amenable to direct analysis by techniques such as LC-FLD. When determining the concentration of residual BPA in a PC plastic, care should be taken to avoid hydrolysis of the polymer, since this could lead to an overestimation of the BPA levels present that could migrate into a foodstuff under normal conditions of use. Alkaline conditions have been reported to hydrolyse the PC polymer, and the hardness of the water has also been postulated to play a role in the degradation (Biedermen-Brem et al., 2008; Biedermen-Brem and Grob, 2009). For epoxy coated metal substrates for which the coating is usually < 10 µm it is generally accepted that acetonitrile affords exhaustive extraction. Given the solubility of BPA in ethanol and the polarity of paper and board substrates, then extraction in this solvent is conventionally used for the exhaustive extraction of this matrix. It is rare that sensitivity is an issue when analysing extracts of PC or epoxy coated food contact materials and articles, and therefore the extracts generated are usually analysed directly.

Migration – Regulation (EU) No 10/2011 (EU, 2011) defines food simulants and migration test conditions for food contact plastics and is applicable to PC plastics. These food simulants intended to mimic the migration of a given substance that could, under the worst foreseeable conditions of use, migrate into a foodstuff. For consumer protection purposes it is the intention that migration into food simulants should exceed that which will occur into a food. A CEN Technical Specification was published in 2005 describing methodology for the determination of BPA in conventional EU food simulants (CEN, 2005). In this procedure aqueous food simulants are analysed directly by LC-UV and oil samples dissolved in hexane and extracted into methanol/water. The methanol/water extracts are then analysed directly by LC-UV. The aforementioned Regulation also permits the substitution of food simulants with more severe extraction solvents, provided that the substitution is based on scientific evidence that the substitute food simulants (extraction solvent) used overestimate the migration compared to the regulated food simulants. The majority of the methods available for food contact materials and articles describe the determination of BPA in these regulated or substituted food simulants (solvents). The exposed simulants/solvents may be analysed directly by LC-FLD or LC-MS/MS (e.g. Santillana et al., 2011), analysed using solid-phase micro-extraction and GC-MS (e.g. Cao and Corriveau, 2008b), concentrated using solid phase extraction (SPE) and analysed by GC-MS (e.g. Guart, 2011; Fasano et al., 2012), concentrated using SPE, derivatised and analysed by GC-MS (e.g. Ehlert et al., 2008; Kubwabo et al., 2009). Direct analysis of water as a food simulant using an ELISA method has also been reported (Cooper et al., 2011) however concerns regarding sensitivity, selectivity and cross-reactivity have been raised for this method of analysis (Chapter 4.2.1).

Extraction of BPA from food

For foodstuffs solvent extraction is the most common technique used for the isolation of BPA from the food matrix. The solvent used and the extraction conditions are dependent on the specific food type.

Acetonitrile is the most commonly used extraction solvent for solid foodstuffs. In addition to the extraction of BPA acetonitrile will also precipitate any proteins that are present, thereby effectively performing a clean-up step alongside the extraction. In addition to the removal of proteins from the matrix the separation of the BPA from the fat also facilitates improved analytical performance, and this has been reported to be achieved using alkanes (hexane, heptanes and isooctane) along with the acetonitrile. For liquid foodstuffs and beverages BPA extraction using ethyl acetate, chloroform or dichloromethane has been reported (Ballesteros-Gómez et al., 2009), however SPE techniques are more extensively used to isolate the BPA from these matrices (e.g. Maragou et al., 2006; Ackermann et al., 2010; Gallart-Ayala et al., 2011; Bono-Blay et al., 2012). Other extraction techniques for reported in the literature have been summarised by Ballesteros-Gómez et al. (2009) and include pressurised liquid extraction (Ferrer et al., 2011), coacervative microextraction (García-Prieto et al., 2008; Pérez Bendito et al., 2009), microwave assisted extraction (Pedersen and Lindholst, 1999; Basheer et al., 2004), solid-phase micro-extraction (Cao and Corriveau, 2008b), stir bar sorptive extraction (Magi et al., 2010), molecularly imprinted polymers (Baggiani et al., 2007, 2010) and matrix solid phase dispersion extraction (Shao et al., 2007a).

Although some methods report the direct analysis of the solvent extracts using LC and GC separation techniques, in most cases additional sample clean-up and concentration steps are required to achieve the desired selectivity and sensitivity. SPE clean-up is the most commonly reported technique (Cao et al., 2009a; Yonekubo et al., 2008; Grumetto et al., 2008), however some methods describing the use immunoaffinity columns for sample clean-up have also been reported (Brenn-Struckhofova and Cichna-Markel, 2006; Podlipna and Cichna-Markel, 2007), along with others describing gel permeation chromatographic methods (Poustka et al., 2007; Gyllenhammar et al., 2012).

As mentioned in Chapter 2 of this opinion, animals that have been exposed to BPA have the potential to contain conjugated BPA, and so food products of animal origin may further contribute to BPA exposure. The methods used to derive the BPA data for animal products and used in the exposure assessment in this opinion were scrutinised to assess whether or not the reported concentration was that of unconjugated BPA or total BPA (conjugated + unconjugated). None of these methods, published in the scientific literature or obtained through the EFSA call for data, described deconjugation steps in the approach. For several methods BPA concentrations were determined after derivatisation (Cao et al., 2008; Geens et al., 2010; Cunha et al., 2011; Feshin et al., 2012). In these examples it is possible that deconjugation would occur during the derivatisation step, especially if a strong acid or base were used. However no scientific data is available to support this, and therefore it was assumed that the reported BPA concentrations for all data are for unconjugated BPA only. Given the rapid elimination and the short half-live of BPA, it seems unlikely that significant concentrations of the conjugates will accumulate in animals intended for food following exposure during their lifetime. ANSES (ANSES, 2013) reported that the levels of unconjugated BPA and total BPA (conjugated + unconjugated) were similar in the meat products that they tested.

Extraction of BPA from biological samples

A number of sensitive methods have been developed to quantitate low concentrations of BPA in blood and urine samples from non-intentionally exposed human subjects (Dekant and Volkel, 2008; WHO 2011b; Asimakopoulos et al., 2012). In biological samples BPA can exist in both the conjugated and unconjugated form. BPA-glucuronide is the most commonly found BPA conjugate along with lower levels of BPA-sulphate. Consequently, methods to determine total BPA in biological samples include an enzymatic deconjugation step using β-glucuronidase and sulphatase. Even if a study is focused only on unconjugated BPA, the information on total or conjugated BPA is needed for quality-control purposes. Additional quality criteria include the information on extraction recovery and the use of surrogate standards to monitor the extent of the deconjugation reaction. In addition to the deconjugation step sample work-up procedures comprise the clean-up, which is generally based on SPE and/or liquid-liquid extraction (LLE). The most common solvent used for the extraction of BPA from biological samples is acetonitrile. As discussed above for foodstuffs one advantage of using acetonitrile as the extraction solvent is the simultaneous precipitation of endogenous proteins in the matrix (Völkel et al., 2011). Recent trends for biomonitoring of BPA have been described by

Asimakopoulos et al. (2012) and include an overview of the methodology applied to these matrices. The authors summarise that "ethyl acetate (Schöringhumer and Cichna-Markl, 2007), chloroform (Kuroda et al., 2003), diethyl ether (Ouchi and Watanabe, 2002), isopropanol (Atkinson et al., 2002) and ammonium hydroxide (Kaddar et al., 2009) were also reported for analyte(s) extraction or/and protein precipitation purposes. n-Hexane, ethanol and petroleum ether were particularly used for lipid removal from matrix (Sajiki, 2003; Lin et al., 2009)". As for liquid foodstuffs SPE extraction can be applied to liquid matrices (usually following dilution with water and deconjugation with enzymes) or it can be applied as a clean-up and concentration step to achieve the sensitivity required for these matrices. Examples of the use of SPE in sample extraction, clean-up and concentration include BPA determination in urine (Moors et al., 2007; Calafat et al., 2008; Teeguarden et al., 2011), human colostrom (Kuruto-Niwa et al., 2007) and human milk (Cariot et al., 2012). Additional information is given in section 4.8 of the opinion.

Extraction of BPA from non-food sources

Environmental samples - outdoor air – To determine the concentration of BPA in air samples, the sample is first collected onto a filter and the filter is extracted using solvent. Sample clean up methods, concentration and derivatisation steps are then all similar to other matrices. Fu and Kawamura (2010) used an aerosol sampling technique to obtain the samples. The resulting filters were ultrasonicated in dichloromethane/methanol (2:1, v/v), evaporated to dryness and derivatised with BSTFA with 1 % trimethylsilyl chloride in pyridine. Following dilution with hexane the derivatives were analysed by GC-MS. Sangiorgi et al. (2013) compared indoor and outdoor BPA in particulate matter. The filter samples were extracted with methanol and analysed directly by LC-MS/MS. Wilson et al. (2007) described methodology for the sampling of outdoor air using a 10 mm inlet, to collect targeted chemicals in a glass cartridge containing a quartz fibre filter followed by XAD-2 resin. Soxhlet extraction of the filter using dichloromethane, sample concentration by SPE using fluorisil and analysis by GC-MS.

Environmental samples – surface water – many of the extraction techniques described for the determination of BPA in surface water are consistent with those reported and described above for food and beverages and for food simulants. Other examples include the extraction of BPA from with coacervates made up of decanoic acid reverse micelles with analysis using LC-FLD (Ballesteros-Gómez et al., 2007), SPE methodology using magnetic multiwalled carbon nanotubes followed by GC-MS/MS to determine BPA in river water as well as snow and drinking water (Jiao et al., 2012) and detection via inhibition of luminol chemiluminescence (CL) by BPA on the silver nanoparticles (AgNPs)-enhanced luminol-$KMnO_4$ CL system (Chen et al., 2011). Krapivin et al. (2007) reviewed a range of ELISA methods for the determination of BPA in surface water samples.

Indoor air – Methods described for the determination of BPA in indoor air are consistent with those for outdoor air.

Dust – Wilson et al. (2007) described the collection of house dust using an HVS3 vacuum sampler (ASTM, 1997). Dust samples were sonicated with 10 % diethyl ether/hexane to extract the BPA from the matrix. Sample concentration and analysis was consistent with the air samples. Geens et al. (2009a) reported similar methodology for dust samples with the BPA being extracted into hexane:acetone (3:1), clean up by SPE using fluorisil but with analysis by LC-MS/MS. Völkel et al. (2008) measured BPA in dust collected by residents in homes using regular vacuum cleaners. Sonication of the dust in methanol released the BPA and, following the addition of water, the extracts were analysed using SPE-LC-MS/MS. Loganathan and Kannan (2011) determined BPA in house dust. The BPA was extracted into ethyl acetate, solvent swapped into methanol and analysed by LC-MS/MS.

Paper products (including thermal papers) – As mentioned above, paper is a polar matrix and so to ensure exhaustive extraction polar solvents are generally used to extract the BPA. Biedermann et al. (2010) extracted BPA from thermal paper samples by immersion in methanol overnight at 60C, extracts were then diluted prior to analysis by LC-FLD. Liao and Kannan (2011a, b) and Geens et al. (2012b) also used methanol to extract BPA from paper samples. Mendum et al. (2011) used ethanol as

the extraction solvent for thermal receipts. Another study reported the use of pyrolysis GC-MS to determine BPA in paper samples (Becerra and Odermatt, 2012) although the authors state that "The reliability of quantification with an internal standard should be further investigated".

Children's toys and teats – Methods of analysis reported for the determination of BPA in plastic toys are consistent with those for the extraction of BPA from plastic food contact materials, e.g. dissolution in a solvent with subsequent polymer precipitation, solvent extraction using microwave digestion and solvent extraction. Atkins (2012) described the dissolution of PVC toys in tetrahydrofuran with polymer precipitation using hexane and compared the extraction efficiency with that of a simpler microwave digestion method. Another method for determination of BPA released from toys described the use of water and 0.07 M hydrochloric acid. The contact conditions were 24 hours at 40°C for water according to EN 14372 and 24 hours at 37°C for the acidic medium. In this study the extraction methods used were intended to mimic the exposure of children to BPA from this source (Troiano and Goodman, 2010). In this the transfer of BPA to water or to a saliva simulant to determine exposure from these articles was considered, as well as the concentration of BPA in the plastic portion of the toys itself. Methods of analysis for the determination of BPA in saliva simulant include ultrasound-assisted emulsification liquid–liquid microextraction (Viñas et al., 2012). Methodology for the determination of BPA in plastic toys and in physiological saline solution was described by KemI (2012). Ground plastics were soxhlet extracted with either methanol or dichloromethane and analysed directly by GC-MS. The toys were also exposed to physiological saline solution (37 °C, 10 min, 30 min and 2 h with stirring) and the extract analysed by GC-MS. Other samples were exposed to artificial saliva (24 °C, 24 h).

Medical devices (dental sealants) – The extraction media used for the determination of BPA in resin based dental materials are included in the review of the exposure from these sources (Van Landuyt et al., 2011). The extraction solvents included water, acetonitrile, ethanol, ethanol/water, artificial saliva or saliva simulant, phosphate buffer or citrate/phosphate buffer.

Instrumental Analysis

The analytical methods reported to be used for the determination of BPA in all matrices described above include: LC-UV, LC-FLD, LC-ECD, LC-MS and LC-MS/MS, GC-MS and GC-MS/MS and ELISA.

GC methods – Although some methods describe the direct analysis of solvent extracts containing BPA by GC-MS or GC-MS/MS many involve derivatisation to achieve repeatable data. Cao (WHO, 2011b) concluded that "derivatisation is always recommended for quantitative analysis by GC-MS". Improved accuracy and sensitivity can be achieved by the derivatisation of the free hydroxyl functional groups on BPA (WHO, 2011b). Silylation using N-O-bis(trimethylsilyl) trifluoroacetamide (BSTFA) (Fu and Kawamura, 2010; Viñas et al., 2010) or N',N'-methyl-(tert-butyldimethylsilyl) trifluoroacetamide (MTBSTFA) (Becker et al., 2009) and acetylation using acetic anhydride (Cao et al., 2008b, 2009a, b; Viñas et al., 2010; Cunha et al., 2011) are the most common derivatisation techniques used for BPA. The use of isopropyl chloroformate to form diether derivatives (Feshin et al., 2012), pentafluorobenzylbromide (Kuklenyik et al., 2003), pentafluorobenzoylchloride (Geens et al., 2009b, 2012b), pentafluoropropionic anhydride (Dirtu et al., 2008), trifluoroactic anhydride (Varelis and Balafas, 2000) has also been described.

LC methods – The majority of the LC methods reported use reverse phase chromatography for the determination of BPA. More recently the use of UPLC methods has also been described (Lacroix et al., 2011; Xiao et al., 2011; Cariot et al., 2012; Perez-Palacios et al., 2012) for the determination of BPA in biological samples. Although BPA is a weak chromophore and so can be detected by UV the sensitivity of the analysis is low when compared with other detectors. The CEN Technical Specification for the determination of BPA (CEN, 2005) uses UV detection at 280 nm to determine BPA concentrations in food simulants however none of the more recently developed methods use this detector. LC-FLD methodology with excitation wavelengths in the range 224 to 235 nm or 275 nm and emission wavelengths in the range 300-320 nm have been described and reviewed by Cao (WHO,

2011b). Although BPA is a relatively strong fluorophore (due to the conjugation) the addition of a stronger fluorophore to BPA using 4-(4,5-diphenyl-1H-imidazol-2-yl)benzoyl chloride (Watanabe et al., 2001; Sun et al., 2002; Kuroda et al., 2003) or p-nitrobenzoyl chloride (Mao et al., 2004) prior to analysis by LC-FLD has also been proposed to improve the method sensitivity. The lack of selectivity of these methods compared to MS methods means that the data derived may overestimate the concentration of BPA present in the samples. Although ECD detection affords better selectivity than UV and FLD methods (it is electrically specific for phenolic compounds) there are only limited applications described in the literature. Sajiki et al. (2007) used LC-ECD and LC-MS for the detection of BPA in canned foods and concluded that although LC-ECD is specific for phenols and MS for the mass of BPA the best selectivity is afforded by the tandem MS/MS techniques and so this is preferred for quantifying BPA.

For both GC-MS and LC-MS methods of analysis isotope-dilution mass spectrometry based on stable isotope-labelled (^{2}H or ^{13}C) BPA as an internal standard is considered as the most specific, selective, accurate and precise detection method for measuring trace levels of BPA (WHO, 2011b).

ELISA methods (Fukata, 2006) – Commercial ELISA kits for the determination of BPA are available and have been used to determine BPA in biological samples. They are not selective for the unconjugated form and so concentrations measured using this technique are for total BPA. The main issue with ELISA methods is the cross-reactivity with other compounds that are structurally similar to BPA.

APPENDIX II: EFSA CALL FOR DATA

The Appendix II contains details on the quality of data received through the EFSA call for data, for the following categories: Food and beverages intended for human consumption, Migration data from food contact materials and Occurrence data in food contact materials.

Food and beverages intended for human consumption

The European Economic Area (EEA) countries (European Union, plus Iceland, Liechtenstein and Norway and Switzerland submitted BPA occurrence data from different kind of food, 2 076 results were reported from 2004 to 2012.

Regarding the 1 592 results submitted on unconjugated BPA determination the method was accredited by ISO/IEC 17 025 procedure for 71 % of the results and internally validated for 29 %. Regarding the 484 results submitted on determination of Bisphenol total the method was accredited by ISO/IEC 17 025 procedure for 12 % of the results, the procedure was internally validated for 42 % and not validated for 12 %, no information was provided for 33 % of the results (12 % of results submitted from accredited laboratories and 21 % of results submitted from non accredited laboratories).

Information about the method of analysis was provided for 100 % of the results. The following methods were reported for the determination of Bisphenol unconjugated in 1 592 samples analysed: GC-MS-MS (71 % of the samples) and LC-MS/MS (29 % of the samples). The following methods were reported for the determination of Bisphenol total in 484 samples analysed: LC-MS/MS (48.1 % of the samples). GC-MS (18.4 % of the samples), HPLC-FD (12.8 % of the samples), HPLC-UV (8.5 % of the samples), GC-MS-MS (6.4 % of the samples) and HPLC with standard detection methods (5.8 % of the samples).

For the determination of Bisphenol unconjugated LODs were reported as below the limit of 15 µg/kg for 693 results (ranging from 0.008 to 13.9 µg/kg) and greater than 15 µg/kg for 1 (one) result (29.8 µg/kg). LOQs were reported as below the limit of 50 µg/kg for 717 results (ranging from 0.024 to 41.7 µg/kg) and greater than 50 µg/kg for 1 (one) result (89.4 µg/kg).

For the determination of Bisphenol total LODs were reported as below the limit of 15 µg/kg for 344 results (ranging from 0.0003 to 3.667 µg/kg) and greater than 15 µg/kg for 34 results (ranging from 16.67 to 105 µg/kg). LOQs were reported as below the limit of 50 µg/kg for 396 results (ranging from 0.001 to 50 µg/kg) and greater than 50 µg/kg for 33 results (210 µg/kg).

The food samples across food groups classified according to the FoodEx classification system level 1 were: drinking water (23 %), vegetables and vegetable products (15 %), meat and meat products (10 %), composite food (8 %), milk and dairy products (7 %), grains and grain-based products (7 %), fish and other seafood (7 %), fruit and fruit products (5 %), alcoholic beverages (4 %), non-alcoholic beverages (4 %), legumes, nuts and oilseeds (3 %), starchy roots and tubers (2 %), snacks, desserts, and other foods (2 %), animal and vegetable fats and oils (1 %), herbs, spices and condiments (1 %), sugar and confectionary (1 %), eggs and egg products (1 %), fruit and vegetable juices (1 %).

The vast majority of the samples at the 2nd level of the FoodEx classification were: “tap water” (13 %), “bottled water” (9 %), “fruiting vegetables (4 %), “fish meat (4 %), “livestock meat” (4 %), Some of the analysed foods were canned in tinplate varnished or partly varnished (5 %), in metal (4 %) and tinplate (2 %).

Migration data from food contact materials

The method for the determination of BPA was accreditated by ISO/IEC 17 025 procedure for 34 % of the 988 submitted results, the procedure was validated internally for 30 % (including results from non

accreditated laboratories) and accreditated by an other third party quality assessment procedure for 36 %.

Information about the method of analysis was provided for 100 % of the results. The following methods were reported: HPLC-FL (52 % of the samples), HPLC with standard detection methods (23 % of the samples), HPLC-UV (11 % of the samples), GC-MS (6 % of the samples), LC-MS-MS (6 % of the samples), LC-MS (2 % of the samples).

LODs were reported as below the limit of 15 µg/kg for 748 results (ranging from 0.006 to 15 µg/kg) and greater than 15 µg/kg for 92 results (ranging from 20 to 40 µg/kg)µg/kg. LOQs were reported as below the limit of 50 µg/kg for 872 results (ranging from 0.018 to 50 µg/kg) and greater than 50 µg/kg for 103 results (ranging from 60 to 500 µg/kg).

All the data and results value are converted to µg/kg. If the result of the overall migration in the original results was expressed as mg/dm2 the conversion rate was 1 mg/dm^2 equal to 6 mg/kg of packaged food as reported in Consideration n.26 of the Regulation EU No 10/2011.

Occurrence data in food contact materials

The method for the determination of BPA was validated internally for 1 % of the samples analyzed. No information was provided on the accreditation of the method for the remaining 99 % of the sample analysed.

Information about the method of analysis was provided for 43 % of the 545 submitted results. The following methods were reported: HPLC with standard detection methods (25 % of the samples), GC-MS (16 % of the samples), HPLC-FL (1 % of the samples). Classification of the method of analysis was not possible for 57 % of the samples (submitted as “EG-Referenzmethode” and “Nicht in einer offiziellen Sammlung enthaltene Methode”).

LODs were reported as below the limit of 15 µg/kg for 321 results (ranging from 0.0033 to 10 000 µg/kg) and greater than 15 µg/kg for 212 results (ranging from 90 to 42 800 µg/kg). LOQs were reported as below the limit of 50 µg/kg for 330 results (ranging from 0.01 to 40 µg/kg) and greater than 50 µg/kg for 203 results (ranging from 90 to 42 800 µg/kg).

APPENDIX III: FOOD CATEGORIES

Appendix III provides a comprehensive description of all data made available in relation to BPA concentration in food and beverages. Data are described separately for "Canned food categories" and "Non-canned food categories", making use of the EFSA FoodEx categories. European data from the literature and from the EFSA's call for data are first described separately and then pooled. Non-European data are then described for comparison purpose only. Note that in this appendix the term "BPA means unconjugated BPA

Canned food categories

For canned food, the overall number of samples was 638 of which 342 samples were from the literature an296 d samples were from the call for data.

"Grains and grain-based products", canned

One sample for "Grains and grain-based products" was available from the literature data in Belgium (Geens et al., 2010). The corn grain sample had a BPA concentration of 67.4 µg/kg.

Concentration data on "Grains and grain-based products" were provided through the call for data by France and Ireland with a total of 18 samples. The samples were mainly corn grains. The BPA concentrations ranged from 23.1 µg/kg (corn grain, France) to 47.5 µg/kg (corn grain, France). Mean BPA concentration (middle bound) was 34.9 µg/kg.

When all European data for canned grains and grain-based products were pooled, average BPA concentration (middle bound) was 36.6 µg/kg.

Concentration values in samples from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007), Korea (Lim et al., 2009a, Kawamura (personal communication), 2013), Canada (Cao et al., 2011), China (Niu et al., 2012) and Iran (Ahmadkhaniha et al., 2013) were within the same range as the samples from Europe.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

"Vegetables and vegetable products", canned

Concentration data in 50 samples of canned "Vegetables and vegetable products" were available from the literature in Russia (Feshin et al., 2012), Belgium (Geens et al., 2010), Spain (Garcia-Prieto et al., 2008) and Italy (Grumetto et al., 2008). Most of the analysed samples referred to canned tomato products (Grumetto et al., 2008). The BPA concentrations ranged from below LOD/LOQ (40 %) to 116.3 µg/kg (mushrooms, Geens et al., 2010). Mean BPA concentration (middle bound) was 26.0 µg/kg.

Concentration data on canned "Vegetables and vegetable products" were provided through the call for data by Germany, Switzerland, Ireland, Finland and Norway for a total of 73 samples. Around half of the samples were sweet corn, while coconut milk, sauerkraut, tomatoes and other vegetable products constituted the other half. The BPA concentrations ranged from below LOD/LOQ (18 %) to 100.1 µg/kg (mushrooms, Germany). Mean BPA concentration (middle bound) was 21.7 µg/kg.

When all European data for canned vegetable and vegetable products were pooled, average BPA concentration (middle bound) was 23.5 µg/kg.

Concentration values in samples from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007; Yonekubo et al., 2008; Kawamura (personal communication), 2013), Korea (Lim et al., 2009a), Iran

(Ahmadkhaniha et al., 2013) and Canada (Cao et al., 2010a) were within the same range as the samples from Europe.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

"Legumes, nuts and oilseeds", canned

Concentration data in two samples of canned "Legumes, nuts and oilseeds" were available from the literature in Spain (Garcia-Prieto et al., 2008). The peas had a BPA concentration of 69.0 µg/kg and the green beans had a BPA concentration of 103.0 µg/kg. The average BPA concentration (middle bound) was 120.5 µg/kg.

Concentration data on legumes, nuts and oilseeds were provided through the call for data by Ireland, Germany, France, and Finland for a total of 18 samples. The samples were of beans and peas. The BPA concentration ranged from below LOD/LOQ (33 %) to 137.0 µg/kg (green peas, Ireland). The average BPA concentration (middle bound) was 28.8 µg/kg.

When all European data for legumes, nuts and oilseeds samples were pooled, average BPA concentration (middle bound) was 34.6 µg/kg.

Concentration values in samples from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007), Korea (Lim et al., 2009a), and Canada (Cao et al., 2010a, 2011), were in the same range as the samples from Europe. However, one study from the USA (Noonan et al., 2011) showed BPA concentrations of some canned beans and peas with BPA values up to 730 µg/kg, while the average BPA concentration in canned vegetables in the Noonan et al. (2011) study was 87.8 µg/kg.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Fruit and fruit products, canned

Concentration data in 7 samples of canned "Fruit and fruit products" were available from the literature in Belgium (Geens et al., 2010) and Spain (Garcia-Prieto et al., 2008). The analysed samples were from canned fruit. BPA concentrations varied from 7.8 µg/kg (canned mixed fruit, Garcia-Prieto et al., 2008) to 24.4 µg/kg (canned mango, Garcia-Prieto et al., 2008). Mean BPA concentration (middle bound) was 15.9 µg/kg.

Concentration data on fruit and fruit products were provided through the call for data by Ireland, Germany, France and Norway for a total of 14 samples. The samples were mostly of canned fruit, in addition to two samples of dried prunes and one sample of jam. The BPA concentration varied from below LOD/LOQ (21 %) to 107.0 µg/kg (dried prunes, Ireland). Mean BPA concentration (middle bound) was 12.2 µg/kg.

When all European canned fruit and fruit products were pooled, average BPA concentration (middle bound) was 13.4 µg/kg.

Concentration values in fruit and fruit products from Japan (Sajiki et al., 2007; Kawamura (personal communication), 2013), Korea (Lim et al., 2009a), Canada (Cao et al., 2011), USA (Noonan et al., 2011), and most of the concentrations from Singapore (Sun et al., 2006) were within the same range as the samples from Europe. However, Sun et al., (2006) reported canned mango with a BPA concentration of 160 µg/kg.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in theEFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Meat and meat products, canned

Concentration data in 31 samples of canned "Meat and meat products" were available from the literature in Czech Republic (Poustka et al., 2007), Russia (Feshin et al., 2012), Spain (Bendito et al., 2009), and Belgium (Geens et al., 2010). The analysed samples were mostly of pate from pork liver (16 samples) and luncheon meat (11 samples). BPA concentrations ranged from below the level of quantification (39 %) to 51.1 µg/kg (luncheon meat, Czech Republic). Mean BPA concentration (middle bound) was 14.7 µg/kg.

Concentration data on meat and meat products were provided through the call for data by Ireland, Finland, and France for a total of 16 samples. The samples were of different meat and meat products. The BPA concentration ranged from below the limit of quantification (38 %) to 203.0 µg/kg (bacon, Ireland). Mean BPA concentration (middle bound) was 64.2 µg/kg.

When all European data for canned meat and meat products were pooled, average BPA concentration (middle bound) was 31.5 µg/kg.

Concentration values in meat samples from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007; Kawamura (personal communication), 2013), Korea (Lim et al., 2009a), and Canada (Cao et al., 2011) were in the same range as the samples from Europe.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

"Fish and other seafood", canned

Concentration data in 107 samples of canned "Fish and seafood" were available from the literature in Czech Republic (Poustka et al., 2007), Portugal (Cuhna et al., 2012), Belgium (Geens et al., 2010), and Spain (Perez-Bendito et al., 2009). The analysed samples were of tuna, mackerel, sardines and other fish and seafood. The BPA concentrations ranged from below LOD/LOQ (20 %) to 169.3 µg/kg (tuna in oil, Geens et al., 2010). Mean BPA concentration (middle bound) was 39.5 µg/kg.

Concentration data on fish and other seafood were provided through the call for data by Germany, Finland, Switzerland, Ireland, Norway, and France for a total of 67 samples. The samples were of tuna, sardines, mackerel and other fish and seafood. The BPA concentration ranged from below level of detection (33 %) to 198 µg/kg (cod and whiting, Ireland). Mean BPA concentration (middle bound) was 33.0 µg/kg.

When all European data for canned fish and seafood samples were pooled, average BPA concentration (middle bound) was 37.0 µg/kg.

Concentration values in samples from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007; Yonekubo et al., 2008; Kawamura (personal communication), 2013), Korea (Lim et al., 2009a) and Canada (Cao et al., 2011) were within the same range as the samples from Europe.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Milk and dairy products, canned

Concentration data in 19 samples of canned "Milk and dairy products" were available from the literature in Spain (Molina-Garcia et al., 2012) and Greece (Maragou et al., 2006). The analyzed samples were of liquid milk (9 samples), evaporated milk (7 samples), and milk powder (3 samples). BPA concentrations varied from below the level of detection (63 %) to 15.2 µg/kg (evaporated milk, Maragou et al., 2006). Mean BPA concentration (middle bound) was 2.6 µg/kg.

Concentration data on milk and dairy products were provided through the call for data by Germany for 3 samples. The samples were of liquid milk. BPA concentration varied from 0.7 µg/kg to 35.9 µg/kg. Mean BPA concentration (middle bound) was 19.8 µg/kg.

When all European data for canned milk and dairy products were pooled, average BPA concentration (middle bound) was 4.9 µg/kg.

The concentration value in evaporated milk from Canada (Cao et al., 2011) was in the same range as the European samples.

The FAO/WHO opinion (2011) and the EFSA opinion (2006) did not assigned a specific BPA value for canned milk and diary products.

Sugar and confectionary, canned

The only sample in this food category was available from the literature in Belgium (Geens et al., 2010). The BPA concentration in the fruit sauce was 0.2 µg/kg.

This concentration value was used in the exposure assessment. However, the only foods consumed in this category were fruit sauce and molasses, and these foods were not consumed in large quantities and do not make an impact on the total exposure.

Fruit and vegetable juices, canned

Concentration data in 5 samples of canned "Fruit and vegetable juice" were available from the literature in Belgium (Geens et al., 2010). The analyzed samples of fruit juice varied in BPA concentrations from 0.8 µg/kg to 4.7 µg/kg. The average BPA concentration (middle bound) was 2.7 µg/kg.

The FAO/WHO opinion (2011) assigned an BPA value of 23.2 µg/kg to the canned non-carbonated liquids, while in the EFSA opinion (2006) 10 µg/kg was used for canned liquid beverages.

Non-alcoholic beverages, canned

The food category "Non-alcoholic beverages" includes canned beverages such as soft drinks, both carbonated and non-carbonated, coffee and tea. Concentration data in 54 samples of canned "Non-alcoholic beverages" were available from the literature in Belgium (Geens et al., 2010), Portugal (Cunha et al., 2011), and Spain (Gallart-Ayala et al., 2011; Cacho et al., 2012). The samples were of canned soft drinks (49 samples) and canned tea (5 samples). The BPA concentrations ranged from below the limit of detection (26 %) to 8.1 µg/kg (citrus soda, Geens et al., 2010). Mean BPA concentration (middle bound) was 0.5 µg/kg.

Concentration data on "Non-alcoholic beverages" were provided through the call for data by Germany and Norway for a total of 11 samples. Two of the samples were coffee and the rest soft drinks. BPA concentration ranged from below the level of detection (27 %) to 1.5 µg/kg (in coffee, Germany). Mean BPA concentration (middle bound) was 0.5 µg/kg.

When all European data for canned non-alcoholic beverages were pooled, average BPA concentration (middle bound) was 0.5 µg/kg.

From the literature outside Europe, Lim et al. (2009a) found high BPA concentrations in 7 out of 8 samples of canned coffee and tea from Korea. The highest BPA concentration was 136.14 µg/kg, and 6 of the samples were in the range 10.64-38.28 µg/kg (Lim et al., 2009a). Concentration values in samples from Canada (Cao et al., 2009b, 2010a, 2011) and Japan (Kawamura (personal communication), 2013) were in the same range as the samples from Europe.

Based on these data, the FAO/WHO opinion (2011) assigned a different BPA concentration to carbonated beverages (cola, beer, soda, tonic) and non-carbonated beverages (tea, coffee, other), due to high values in canned tea and coffee in the Korean study (Lim et al. 2009a). The carbonated beverages were given a BPA concentration of 1.0 µg/kg in the exposure assessment, while the non-carbonated beverages were given a higher BPA concentration of 23.2 µg/kg. The EFSA opinion (2006) used 10 µg/kg as the BPA concentration for canned beverages.

The Panel observed that the high values in canned tea and coffee in the Korean study were not confirmed by any other study. Contrary to FAO/WHO (2011), the Panel decided not to distinguish between carbonated and non-carbonated soft drinks.

Alcoholic beverages, canned

Concentration data in 18 samples of canned alcoholic beverages were available from the literature in Portugal (Cuhna et al., 2011), Belgium (Geens et al., 2010), and Spain (Gallart-Ayala et al., 2011; Cacho et al., 2012). The analysed samples were all of beer. BPA concentrations ranged from below the level of detection (17 %) to 4.7 µg/kg (beer, Cuhna et al., 2011). Mean BPA concentration (middle bound) was 0.9 µg/kg.

Concentration data in 49 samples of canned alcoholic beverages were provided through the call for data by United Kingdom and Germany. The samples were mostly of beer. The BPA concentration ranged from below the level of qunantification (35 %) to 4.5 µg/kg (beer, Germany). Mean BPA concentration (middle bound) was 0.8 µg/kg.

When all European data for canned alcoholic beverages were pooled, average BPA concentration (middle bound) was 0.8 µg/kg.

The concentration values in alcoholic beverages from Canada (Cao et al., 2010a, 2011) and Japan (Kawamura (personal communication), 2013) were within the same range as the European samples.

The FAO/WHO opinion (2011) assigned a BPA value of 23.2 µg/kg to the canned non-carbonated liquids, while the EFSA opinion (2006) used 10 µg/kg for canned beverages.

Drinking water, canned

There were 11 European samples of canned drinking water available from the literature (1 sample) and from the call for data (10 samples). The average BPA concentration (middle bound) was 0.004 µg/kg. However, there was no reported consumption of canned water, and the concentration value has therefore not been used in the exposure assessment.

Herbs, spices and condiments, canned

Concentration data in 2 samples of canned “Herbs, spices and condiments” were provided through the call for data by Germany. The samples were of dressing and curry sauce, and the BPA concentrations were 0.6 µg/kg and 82.1 µg/kg respectively. The average BPA concentration (middle bound) was 41.4 µg/kg.

The two widely differing values imply a high uncertainty about the average concentration for this food category. However, this will have little impact on the assessment because the foods in this category were not consumed in large quantities.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Food for infants and small children, canned

Concentration data in 10 samples of canned "Food for infants and small children" were available from the literature in Portugal (Cuhna et al., 2011), Spain (Molina-Garcia et al., 2012), and Russia (Feshin et al., 2012). The analysed samples were of infant formula powder. The BPA concentrations ranged from below the level of quantification (70 %) to 2.2 µg/kg (Feshin et al. 2012). Mean BPA concentration (middle bound) was 0.3µg/kg, and the highest BPA concentration was 2.2 µg/kg.

The European Dietetic Food Industry Association has confirmed that canned liquid infant formula is presently not used in Europe (email to EFSA dated 27 June 2013), but are used in other parts of the world. Values from European manufactured canned infant formula was therefore not included in the opinion.

Cao et al., 2008b (Canada) analysed 16 samples of infant formula from USA and Canada. BPA concentration ranged from 2.27 µg/kg to 10.23 µg/kg. Mean BPA concentration was 5.98 µg/kg. Ackerman et al., (2010, USA) have provided BPA concentrations in 71 samples of canned infant formula. The infant formulas were both ready-to-feed and concentrated liquid. The BPA concentrations in liquid formula ranged from 0.56 to 11 µg/kg, with a average BPA concentration of 5.74 µg/kg. In addition Ackerman et al., (2010, USA) detected BPA in 1 sample of 14 powder formula products (0.40 µg/kg).

Earlier opinions have chosen different BPA concentrations for exposure from infant formula. The FAO/WHO report (2011) uses two average BPA concentration values for liquid infant formula of 4 µg/kg for the ready to feed formula, and 3.5 µg/kg for the concentrated liquid formula.

The EFSA opinions (2006) assumed a very conservative value BPA concentration of 100 µg/kg for both beverages and solid canned food consumed by infants.

Products for special nutritional use, canned

Concentration data in 14 samples of canned "Products for special nutritional use" were available from the literature in Portugal (Cunha et al., 2011), Belgium (Geens et al., 2010), Spain (Gallart-Ayala et al., 2011), and Russia (Feshin et al., 2012). All the 14 samples for special nutritional use from the European literature were canned soft drinks. The BPA concentration ranged from below LOD/LOQ (36 %) to 4.8 µg/kg (energy drink, Geens et al., 2010). The average BPA concentration (middle bound) was 1.2 µg/kg.

The FAO/WHO opinion (2011) assigned a BPA value of 23.2 µg/kg to the canned non-carbonated liquids, while the EFSA opinion (2006) used 10 µg/kg for canned beverages.

Composite food, canned

Concentration data in only 6 samples of canned composite foods were available from the literature in Belgium (Geens et al., 2010) and Spain (Bendito et al., 2009). The analysed samples were of soups and other dishes. The BPA concentrations ranged from below the level of quantification (one sample) to 73.1 µg/kg (in ravioli, Geens et al., 2010). Mean BPA concentration (middle bound) was 25.9 µg/kg.

Concentration data in 25 samples of canned composite food were provided through the call for data by Germany, Ireland, Finland, Norway and France. The samples were of soups, bean-based meals, pasta and other composite foods. The BPA concentrations ranged from below the limit of quantification (20 %) to 110 µg/kg (meat balls, Ireland). Mean BPA concentration (middle bound) was 39.6 µg/kg.

When all European data for canned composite foods were pooled, average BPA concentration (middle bound) was 37.0 µg/kg.

The concentration values in composite foods from Singapore (Sun et al., 2006), Japan (Sajiki et al., 2007; Yonekubo et al., 2008; Kawamura (personal communication), 2013), Canada (Cao et al., 2010a), and USA (Noonan et al., 2011) were within the same range as European samples. However, Sajiki et al., (2007) reported a canned crème soup with a value of 156 µg/kg, and canned brown sauces with very high BPA concentrations (428, 547 and 842 µg/kg). Yonekubo et al., (2008) reported a canned gratin sauce with a BPA concentration of 235 µg/kg.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Snacks, desserts, and other foods, canned

Concentration data in 1 sample of canned "Snacks, desserts, and other foods" were provided through the call for data by Ireland. The sample was of starchy pudding, and the BPA concentration was 52.0 µg/kg. This BPA concentration was used in the exposure assessment.

There are few concentration data in this food category. However, the consumption of foods in this category were only of custard and undefined snacks, and neither was not consumed in large quantities.

The FAO/WHO opinion (2011) assigned an overall BPA value of 36.7 µg/kg to the solid canned food, while in the EFSA opinion (2006) 50 µg/kg was used for canned solid foods.

Non-canned food categories

For non-canned food, concentration data from the literature were scarce, with only 246 samples overall, of which 159 were water samples. However, the call for data provided 1 637 samples of non-canned food, of which France is the main contributor with 1 433 samples (88 % of the total non-canned food samples).

Grains and grain-based products, non-canned

Concentration data in 1 sample of non-canned "Grains and grain-based products" was available from the literature in Belgium (Geens et al., 2010). The corn grain sample had a BPA concentration of 0.9 µg/kg.

Concentration data on grains and grain-based products were provided though the call for data by France, Ireland and Norway for a total of 95 samples. The samples were of grains, bread, cakes, breakfast cereals and other grain products. The BPA concentration ranged from below LOD/LOQ (43 %) to 11.9 µg/kg (flan, France). Mean BPA concentration (middle bound) was 1.0 µg/kg.

When all European data for non-canned grains and grain-based products were pooled, average BPA concentration (middle bound) was 1.0 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from cereal products within the same range as the European data.

Concentration values in samples from Japan (Sajiki et al., 2007), and Canada (Cao et al., 2011) were within the same range as the samples from Europe. However, one sample of cookies from Japan (Sajiki et al., 2007) had a BPA concentration of 14 µg/kg.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Vegetables and vegetable products, non-canned

Concentration data in 4 samples of non-canned "Vegetables and vegetable products" were available from the literature in Belgium (Geens et al., 2010). The BPA concentration in the varied vegetables ranged from 0.1 µg/kg to 1.0 µg/kg. Mean BPA concentration (middle bound) was 0.4 µg/kg.

Concentration data on non-canned "Vegetables and vegetable products" were provided through the call for data by France (199 samples), Norway (1 sample) and Ireland (1 sample) for a total of 201 samples. The BPA concentrations for the varied vegetables ranged from below LOD/LOQ (34 %) to 5.3 µg/kg (leaf vegetables, France). Mean BPA concentration (middle bound) was 1.2 µg/kg.

When all European data for canned vegetables and vegetable products were pooled, average BPA concentration (middle bound) was 1.2 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from vegetables within the same range as the European data.

Concentration values in samples from Canada (Cao et al., 2011), and USA (Noonan et al., 2011; Lu et al., 2012, 2013) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Starchy roots and tubers, non-canned

There was not found any BPA concentration data in non-canned "Starchy roots and tubers" in the European literature.

Concentration data on non-canned "Starchy roots and tubers" were provided through the call for data by France (44 samples), and Ireland (1 sample) for a total of 45 samples. All the samples were of potatoes. The BPA concentrations ranged from below LOD/LOQ (16 %) to 2.6 µg/kg (fried potatoes, France).

The average BPA concentration (middle bound) for starchy roots and tubers was 0.7 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from potatoes within the same range as the European data.

Concentration values in samples from Canada (Cao et al., 2011), were within the same range as the samples from Europe. Potatos from USA (Lu et al., 2012, 2013) had a BPA concentration of 4.3 µg/kg.

Legumes, nuts and oilseeds, non-canned

There was not found any BPA concentration data in the European literature.

Concentration data on non-canned "Legumes, nuts and oilseeds" were provided through the call for data by France (3 samples), and Ireland (2 samples) for a total of 5 samples. The samples were of oilseeds, beans, tree nuts and other seeds. The BPA concentration ranged from below LOD/LOQ (60 %) to 0.5 µg/kg (beans, France).

The average BPA concentration (middle bound) for legumes, nuts and oilseeds was 0.2 µg/kg.

Concentration values in samples from Singapore (Sun et al., 2006), and Japan (Sajiki et al., 2007) were within the same range as the samples from Europe. However, one sample of shelled seeds from Canada (Cao et al., 2011) had a BPA concentration of 0.7 µg/kg.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Fruit and fruit products, non-canned

Concentration data in 3 samples of non-canned "Fruit and fruit products" were available from the literature in Belgium (Geens et al., 2010). The BPA concentration in pineapple and olives ranged from 0.1 µg/kg to 1.3 µg/kg. Mean BPA concentration (middle bound) was 0.5 µg/kg.

Concentration data on non-canned "Fruit and fruit products" were provided through the call for data by France (79 samples), and Ireland (6 samples) for a total of 85 samples. The samples were of different fruits, dried fruits and jam. The BPA concentration ranged form below the level of quantification (73 %) to 2.1 µg/kg (grapefruit, France). Mean BPA concentration (middle bound) was 0.3 µg/kg.

When all European data for non-canned fruit and fruit products were pooled, average BPA concentration (middle bound) was 0.3 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from fruits within the same range as the European data.

Concentration values in samples from Japan (Sajiki et al., 2007) were within the same range as the samples from Europe.Fruit samples from USA (Lu et al., 2012, 2013) had a BPA concentration above the European level, with the highest BPA concentration for citrus of 9.0 µg/kg.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Glucuronated BPA in food of animal origin

Any BPA to which food production animals are exposed may conjugate and so may be present in their tissues as glucuronated BPA (ANSES, 2013). When BPA is measured in food of animal origin (e.g. meat, milk, eggs), it is possible that deconjugation occurs. Another potential source of unconjugated BPA in meat products is its migration from any food contact materials or from articles used in the processing of the product. With the exception of the data submitted by France through EFSA`s call, none of the methods, published in the scientific literature or obtained through the EFSA`s call, described deconjugation steps and so it was assumed that the BPA concentrations reported were for unconjugated BPA only. The levels of total and unconjugated BPA in foods of animal origin were reported by ANSES to be virtually the same (ANSES, 2013). Therefore the data on total BPA reported by France were merged with the other data from EFSA's call for data.

Meat and meat products, non-canned

Concentration data in 1 sample of non-canned "Meat and meat products" was available from the literature in Belgium (Geens et al., 2010). The BPA concentration of sausages was 0.9 µg/kg.

Concentration data of non-canned "Meat and meat products" were provided through the call for data by France (172 samples), Ireland (12 samples), and Norway (7 samples) for at total of 191 samples. The samples were of meat types, sausages and pates. The BPA concentration ranged from below the level of quantification (5 %) to 394.8 µg/kg (edible offal, France). The BPA concentration (middle bound) was 9.5 µg/kg.

When all European data for non-canned meat and meat products were pooled, average BPA concentration (middle bound) was 9.4 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from meat within the same range as the European data.

Concentration values in samples from China (Shao et al., 2007b), Canada (Cao et al., 2011), and Japan (Sajiki et al., 2007) were within the same range as the samples from Europe.Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Fish and other seafood, non-canned

Concentration data in 8 samples of non-canned "Fish and other seafood" were available from the literature in Spain (Salgueiro-Gonzalez et al. 2012a), and Belgium (Geens et al., 2010). Most of the analysed samples were of mussel. The BPA concentrations ranged from below LOD/LOQ (75 %) to 11.2 µg/kg (mussel, Salgueiro-Gonzalez et al., 2012a). The BPA concentration (middle bound) was 1.9 µg/kg.

Concentraton data on non-canned "Fish and other seafood" were provided through the call for data by France (66 samples), and Norway (2 samples) for a total of 68 samples. The samples were mostly of mussels, shrimps, salmon and trout. The BPA concentration ranged from below LOD/LOQ (3 %) to 97.9 µg/kg (salmon and trout, France). The BPA concentration (middle bound) was 8.1 µg/kg.

When all European data for non-canned fish and other seafood were pooled, average BPA concentration (middle bound) was 7.4 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from fish within the same range as the European data.

Concentration values in samples from China (Shao et al., 2007a, Wei et al., 2011), and Canada (Cao et al., 2011) were within the same range as the samples from Europe. Some fish and seafood samples from Malaysia (Santhi et al., 2012) had a BPA concentration above the European level, with the highest BPA concentration for squid of 729.0 µg/kg dry weight.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Milk and dairy products, non-canned

Concentration data in 1 sample of non-canned "Milk and dairy products" was available from the literature in Greece (Maragou, et al., 2006). The BPA concentration was below LOD/LOQ, and the middle bound value was 2.6 µg/kg.

Concentration data on non-canned "Milk and dairy products" were provided through the call for data by France (139 samples), Ireland (8 samples), and Norway (4 samples) for a total of 151 samples. The samples were mostly of yoghurt, cow's milk and other cheeses and types of milk. The BPA concentration ranged from below LOD/LOQ (52 %) to 6.1 µg/kg (Chantal cheese, France). The BPA concentration (middle bound) was 0.3 µg/kg).

When all European data for non-canned milk and dairy products were pooled, average BPA concentration (middle bound) was 0.3 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from dairy within the same range as the European data.

Concentration values in samples from China (Shao et al., 2007a; Liu et al., 2008), Canada (Cao et al., 2011), and Japan (Sajiki et al., 2007) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Eggs and egg products, non-canned

There was not found any BPA concentration data in non-canned "Eggs and egg products" in the European literature.

Concentration data on non-canned "Eggs and egg products" were provided through the call for data by France (13 samples), Ireland (1 sample), and Norway (1 sample) for a total of 15 samples. The samples were mostly whole eggs. The BPA concentration ranged from below LOD/LOQ (20 %) to 4.5 µg/kg (whole eggs, France).

The BPA concentration (middle bound) of non-canned eggs and egg products was 0.9 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from eggs within the same range as the European data.Concentration values in samples from China (Shao et al., 2007a) were within the same range as the samples from Europe. However, one of ten eggsamples from China (Shao et al., 2007a) had a BPA concentration of 10.45 µg/kg.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Sugar and confectionary, non-canned

Concentration data in 1 sample of non-canned "Sugar and confectionary" was available from the literature in Belgium (Geens et al., 2010). The BPA concentration was 0.3 µg/kg.

Concentration data on non-canned sugar and confectionary were provided through the call for data by France (14 samples), Ireland (4 samples), and Norway (1 sample) for a total of 19 samples. The samples were mostly chocolate and sugars. The BPA concentration ranged from below LOD/LOQ (42 %) to 2.6 (molasses and other syrups, France). The average BPA concentration (middle bound) was 0.5 µg/kg.

When all European data for non-canned sugar and confectionary were pooled, average BPA concentration (middle bound) was 0.5 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007), and Canada (Cao et al., 2011) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Animal and vegetable fats and oils, non-canned

There was not found any BPA concentration data in non-canned "Animal and vegetable fats and oils" in the European literature.

Concentration data on non-canned animal and vegetable fats and oils were provided through the call for data by France (20 samples), Ireland (4 samples), and Norway (2 samples) for a total of 26 samples. The samples were mostly butter and vegetable oils. The BPA concentrations ranged from below LOD/LOQ (46 %) to 1.4 µg/kg (margarine, and olive oil, France).

The BPA concentration (middle bound) of non-canned animal and vegetable fats and oils was 0.5 µg/kg.

A market basket study from Sweden (Gyllehammar et al., 2012) observed a BPA concentration from fats within the same range as the European data.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Fruit and vegetable juices, non-canned

Concentration data in 2 samples of non-canned "Fruit and vegetable juices" were available from the literature in Belgium (Geens et al., 2010). The BPA concentrations were below LOD/LOQ of 0.01 µg/kg.

Concentration data on non-canned fruit and vegetable juices were provided through the call for data by France (12 samples), Ireland (1 sample), and Norway (1 sample) for a total of 14 samples. The samples were all fruit juices. The BPA concentrations ranged from below LOD/LOQ (71 %) to 6.0 µg/kg (orange juice, France). The average BPA concentration (middle bound) was 0.8 µg/kg.

When all European data on non-canned fruit and vegetable juices were pooled, average BPA concentration (middle bound) was 0.7 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Non-alcoholic beverages, non-canned

Concentration data in 1 sample of non-canned "Non-alcoholic beverages" was provided through the literature in Belgium (Geens et al., 2010). The BPA concentrations were below LOD/LOQ of 0.01 µg/kg.

Concentration data on non-canned non-alcoholic beverages were provided through the call for data by France (68 samples), Ireland (3 samples), and Norway (1 sample) for a total of 72 samples. The samples were mostly from coffee, tea and hot chocolate. The BPA concentration ranged from below LOD/LOQ (64 %) to 1.7 µg/kg (black tea infusion, Ireland). The BPA concentration (middle bound) was 0.2 µg/kg.

When all European data on non-canned non-alcoholic beverages were pooled, average BPA concentration (middle bound) was 0.2 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007), and Canada (Cao et al., 2010a) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Alcoholic beverages, non-canned

Concentration data in 59 samples of non-canned "Alcoholic beverages" were available from the literature in Austria (Brenn-Struckhofova et al., 2006). All the samples were of wine. The BPA

concentrations ranged from below LOD/LOQ (22 %) to 2.1 µg/kg (wine, Brenn-Struckhofova et al., 2006). The BPA concentration (middle bound) was 0.5 µg/kg.

Concentration data on non-canned "Alcoholic beverages" were provided through the call for data by United Kingdom (14 samples), Germany (8 samples), France (8 samples), and Ireland (5 samples) for a total of 35 samples. The samples were of beer and wine. The BPA concentrations ranged from below LOD/LOQ (71 %) to 1.6 µg/kg (wine, France). The average BPA concentration (middle bound) was 0.5 µg/kg.

When all European data on non-canned alcoholic beverages were pooled, average BPA concentration (middle bound) was 0.5 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007), and Canada (Cao et al., 2010a, 2011) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Water, non-canned

BPA may be present in drinking water due to environmental contamination and/or epoxy resin linings in the drinking-water distribution network and/or migration from polycarbonate water dispensers or water filters. Any BPA present in drinking water may be transformed into chlorinated BPA due to the use of chlorination of water for disinfection purposes (Gallard et al., 2004). In common chlorinated drinking water ($pH \geq 6.5$; $[Cl_2] \geq 0.2$ mg/l) the half-life of BPA would be less than 3 h.

Concentration data in 159 non-canned "Water" were available from the literature in Spain (Guart et al., 2011; Bono-Blay et al., 2012) and Belgium (Geens et al., 2010). The samples were from well water, bottled water and water stored in PC carboys.

BPA was detected in only 6 samples out of 131 samples of well water to be used for bottling water in Spain (LOD 0.009 µg/kg, maximum 0.2 µg/kg) (Bono-Blay et al., 2012). BPA was not detected in one sample of bottled water from Belgium (Geens et al., 2010). BPA was not detected in any sample of bottled water in Spain made of HDPE (n= 7) or PET (n= 10) (LOD = 0.009 µg/kg) (Guart et al., 2011). However, BPA was detected in all 10 samples of water stored in PC coolers in Spain (Guart et al., 2011). The BPA concentrations ranged from below LOD/LOQ (90 %) to 4.4 µg/kg. The average BPA concentration (middle bound) was 0.2 µg/kg.

Concentration data on non-canned "Water" were provided through the call for data by France (396 samples), Germany (42 samples), Spain (17 samples), Ireland (2 samples), Plastics Europe (2 samples), and Norway (1 sample) for a total of 460 samples. All types of non-canned waters where pooled, since most consumers consume a variety of water from different sources. The samples were mostly from tap water, but also from bottled water in PET, glass and PC coolers. The BPA concentrations ranged from below LOD/LOQ (84 %) to 4.5 µg/kg (water stored in PC carboy, France). The average BPA concentration was 0.2 µg/kg.

When all European data on non-canned water were pooled, average BPA concentration (middle bound) was 0.2 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007) were within the same range as the samples from Europe.

The EFSA opinion (2006) did not assign a BPA value to non-canned water. In its exposure assessment, FAO/WHO (2011) observed that most BPA concentration in tap water are below 0.01

µg/l whereas BPA concentration in water packaged in PC bottles were just below 1 µg/l. This last value was used by FAO/WHO in the exposure assessment as a conservative scenario.

Herbs, spices and condiments, non-canned

Concentration data in 2 non-canned "Herbs, spices and condiments" were available from the literature in Belgium (Geens et al., 2010). The samples were from pickles and vegetable sauce with the same BPA concentration of 0.3 µg/kg.

Concentration data on non-canned "Herbs, spices and condiments" were provided through the call for data by France (8 samples), Ireland (8 samples), and Norway (1 samples) for a total of 17 samples. The samples were mainly soy sauce, dressing and some stock cubes. The BPA concentrations ranged from below LOD/LOQ (71 %) to 2.5 µg/kg (dressing, France). The average BPA concentration was 1.3 µg/kg.

When all European data on non-canned herbs, spices and condiments were pooled, average BPA concentration (middle bound) was 1.2 µg/kg.

Concentration values in samples from Canada (Cao et al., 2011) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Food for infants and small children, non-canned

Concentration data in 1 non-canned infant formula was available from the literature from Greece (Maragou, et al., 2006). The BPA concentration was below LOD/LOQ, and the middle bound BPA concentration was 0.9 µg/kg.

Concentration values in samples of baby foods contained in glass jars with metal lid from Canada (Cao et al., 2009a, 2011) were in the ranged below LOD to BPA concentration of 1.7 µg/kg.

Earlier opinions have chosen different BPA concentrations for exposure from infant formula. The FAO/WHO report (2011) used two average BPA concentration values for liquid infant formula of 4 µg/kg for the ready to feed formula, and 3.5 µg/kg for the concentrated liquid formula. The EFSA opinion (2006) did not assign a BPA concentration to non-canned food.

Composite food, non-canned

Concentration data in 3 non-canned "Composite food" was available from the literature from Belgium (Geens et al., 2010). The BPA concentration in the vegetable soups ranged between 0.1 µg/kg to 0.4 µg/kg. The average BPA concentration was 0.3 µg/kg.

Concentration data on non-canned "Composite food" were provided through the call for data by France (96 samples), Switzerland (7 samples), Ireland (2 samples), and Norway (2 samples) for a total of 107 samples. The samples were of different composite foods and dishes. The BPA concentration ranged from below the level of quantification (10 %) to 25.8 µg/kg (sandwich, France). The average BPA concentration (middle bound) was 2.4 µg/kg.

When all European data on non-canned composite foods were pooled, average BPA concentration (middle bound) was 2.4 µg/kg.

Concentration values in samples from Japan (Sajiki et al., 2007), and Canada (Cao et al., 2011) were within the same range as the samples from Europe.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Snacks, desserts, and other foods, non-canned

There was not found any BPA concentration data in non-canned "Snacks, desserts, and other foods" in the European literature.

Concentration data on non-canned "Snacks, desserts, and other foods" were provided through the call for data by France (25 samples), and Ireland (6 samples) for a total of 31 samples. The samples were of potato crisps and desserts. The BPA concentration ranged from below the level of quantification (68 %) to 0.4 µg/kg (potato crisps, France).

The average BPA concentration (middle bound) in non-canned snacks, desserts and other foods was 0.4 µg/kg.

Neither the EFSA opinion (2006) nor the FAO/WHO opinion (2011) did assign a BPA value to non-canned food.

Foods in glass jars with metal lids

BPA can be used in internal coating of metal lids for foods in glass jars, and residues of BPA in these coatings can migrate into foods, especially at elevated temperatures (Cao et al., 2009a). Migration of BPA from the coating on metal lids into foods is assumed to be low compared to canned foods (Cao et al., 2009a). There are not many available data on the BPA concentration in food from glass jars with metal lids.

However, baby foods in glass jars with metal lids are an important part of the diets for children aged 6 months and older. One Canadian study has determined the BPA concentration in 99 baby food products in glass jars (Cao et al., 2009a). The BPA levels in 15 % of the samples were less than the average LOD, and 70 % had BPA levels of less than 1 µg/kg. The average BPA level was 1.1 µg/kg.

Concentration data on 10 samples of fruit, vegetables and anchovy in glass jars were available from the literature in the Netherlands (Geens et al., 2010). The average BPA level was 0.60 µg/kg, with a range from 0.10 µg/kg in red cabbage to 1.28 µg/kg in pineapple.

As expected, the concentrations observed in foods in glass jars with metal lids was in line with that of non-canned food and lower than that in canned food. Concentration data from foods in glass jars with metal lids from the European market were therefore categorized with that of non-canned food in the exposure assessment.

Water from water pipes relined with epoxy resins

Data on BPA in drinking water were available from the literature. A survey performed in Sweden (KEMI, online) investigated if any BPA could be released in drinking water from aged water pipes relined with epoxy resins. Two different techniques for relining have been used in Sweden from 2006-2011, one so called one-component method where the composition of the material has been prepared industrially and the second one so called two-component method where the components are mixed on the spot. Both hot and cold water were collected and analysed. The concentrations in 31 samples of hot water ranged from below the LOQ of 0.01 µg/l (19 %) to 60 µg/l. Mean BPA concentration (middle bound) was 6.2 µg/l, and the 95th percentile (middle bound) was 60 µg/l.

In general the levels were low in cold water. A total of 19 samples of cold water from water pipes relined with the two-component method were analysed for BPA concentration, and the range was from

below the LOQ of 0.01 µg/l (66 %) to 1.1 µg/l. The average BPA concentration (middle bound) was 0.10 µg/l.

The ANSES opinion (2013) had a special attention on water networks renovated with epoxy resins. However, all the 46 samples analyzed had BPA concentrations below the LOQ of 0.025 µg/l.

5581 **APPENDIX IV: SUMMARY OF THE NON-DIETARY SOURCES**

5582 **Table 40:** Overview of the literature concerning non-food sources considered in the exposure assessment.

5583

Author	Country	Location	Unit	Min	Max	Mean	Median	95th percentile
Outdoor air								
Salapasidou et al., 2011	GR	urban traffic site	ng/m^3	0.06	18.6	6.78		
		industrial site	ng/m^3	LOD	47.3	13.2		
Wilson et al., 2007	USA	North Carolina	ng/m^3	1.0	1.5			
		Ohio	ng/m^3	0.7	0.9			
Rudel, 2010	USA	California	ng/m^3		2.0		0.5	
Matsumoto et al., 2005	J	urban ambient outdoor air	ng/m^3	0.02	1.92	0.51		
Fu and Kawamura, 2010	Worldwide		pg/m^3	1	17 400			
Surface water								
Klecka et al., 2007	North America		µg/l				0.08	
	Europe		µg/l				0.01	
Air								
ANSES, 2013	FR	30 homes	ng/m^3		5.3	1.0	0.6	
Wilson et al., 2007	USA	257 US homes	ng/m^3	0.9	193		1.82	11.1
Rudel et al., 2010	USA	50 Californian houses	ng/m^3	0.5	20		0.5	
Dust								
Völkel et al., 2008	DE	12 German homes	µg/kg	117	1 486		553	
Geens et al., 2009	BE	18 Belgian homes	ng/g	535	9 729		1 461	
Geens et al., 2009	BE	2 Belgian offices	ng/g	4 685	8 380			
ANSES, 2013	FR	25 French homes	mg/kg		20	5.8	4.7	
Paper products								
Biedermann et al., 2010	CH	thermal papers	g/kg	8	17	13.3		

Author	Country	Location	Unit	Min	Max	Mean	Median	95th percentile
Östbert and Noaksson, 2010	SE	receipts	g/kg	5	32			
Liao and Kannan, 2011a	USA	thermal paper receipts	g/kg	0.000001	13.9			
Liao and Kannan, 2011b	USA	paper currencies	mg/kg	0.001	82.7			
Gehring et al., 2004		recycled toilet paper	mg/kg	3.2	46.1			
Toys								
Vinas et al., 2012	ES	Toys and teats	µg/l	0.2	5.9			
KemI, 2012	SE	Toys and teats	µg/l	<0.1	2.1			
Lassen et al., 2011	DK	Pacifiers	ng/product		1 360	319		
Cosmetics								
Cacho et al., 2013	ES	Various cosmetic products	µg/kg	<LOQ	88			
Dodson et al., 2012	USA	Various cosmetic products	mg/kg	1	100			
Dental sealants								
Sasaki, 2005		Saliva	µg/l		100			
Kang, 2011		Saliva	µg/l		21	5		

5584
5585

APPENDIX V: SOURCES FOODEX LEVEL 1

The chronic exposure was estimated by multiplying the average BPA concentration for each FoodEx level 1 food group (s) and type of packaging (canned or non-canned) with their respective consumption amount per kg body weight, separately for each individual in the database, calculating the sum of exposure for each survey day for the individual and then deriving the daily average for the survey period. The dietary surveys used, by age class, are given in the Tables below:

Table 41: Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and scenario - Toddlers (Total number of surveys = 7)

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Animal and vegetable fats and oils	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Composite food	5	1	1	0	0	0	0	3	1	2	1	0
Canned	Fish and other seafood	6	1	0	0	0	0	0	5	1	1	0	0
Canned	Fruit and fruit products	6	1	0	0	0	0	3	3	1	0	0	0
Canned	Fruit and vegetable juices	7	0	0	0	0	0	2	2	3	0	0	0
Canned	Grains and grain-based products	6	1	0	0	0	0	6	1	0	0	0	0
Canned	Herbs, spices and condiments	7	0	0	0	0	0	3	3	1	0	0	0
Canned	Legumes, nuts and oilseeds	5	1	1	0	0	0	0	4	2	1	0	0
Canned	Meat and meat products	6	1	0	0	0	0	0	0	2	5	0	0
Canned	Milk and dairy products	7	0	0	0	0	0	5	2	0	0	0	0
Canned	Non-alcoholic beverages	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Products for special nutritional use	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Snacks, desserts, and other foods	7	0	0	0	0	0	3	1	0	2	1	0
Canned	Starchy roots and tubers	7	0	0	0	0	0	7	0	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Sugar and confectionary	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Vegetables and vegetable products	3	1	1	1	1	0	0	0	0	3	4	0
Not canned	Alcoholic beverages	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Animal and vegetable fats and oils	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Composite food	2	4	0	1	0	0	6	1	0	0	0	0
Not canned	Drinking water	0	5	1	1	0	0	1	6	0	0	0	0
Not canned	Eggs and egg products	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Fish and other seafood	0	5	0	2	0	0	5	2	0	0	0	0
Not canned	Food for infants and small children	0	5	0	2	0	0	4	2	1	0	0	0
Not canned	Fruit and fruit products	0	7	0	0	0	0	7	0	0	0	0	0
Not canned	Fruit and vegetable juices	0	5	2	0	0	0	5	2	0	0	0	0
Not canned	Grains and grain-based products	0	1	6	0	0	0	0	7	0	0	0	0
Not canned	Herbs, spices and condiments	6	1	0	0	0	0	7	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	2	5	0	0	2	4	1	0	0
Not canned	Milk and dairy products	0	1	5	1	0	0	0	7	0	0	0	0
Not canned	Non-alcoholic beverages	5	2	0	0	0	0	7	0	0	0	0	0
Not canned	Products for special nutritional use	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	6	1	0	0	0	0	7	0	0	0	0	0
Not canned	Starchy roots and tubers	1	6	0	0	0	0	6	1	0	0	0	0
Not canned	Sugar and confectionary	6	1	0	0	0	0	7	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	4	3	0	0	0	7	0	0	0	0	0

5593

5594 **Table 42:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5595 scenario - Other children (Total number of surveys = 15)

5596

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Animal and vegetable fats and oils	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Composite food	12	1	2	0	0	0	5	1	2	2	4	1
Canned	Fish and other seafood	9	5	1	0	0	0	0	6	8	1	0	0
Canned	Fruit and fruit products	11	4	0	0	0	0	3	8	4	0	0	0
Canned	Fruit and vegetable juices	12	2	1	0	0	0	3	7	5	0	0	0
Canned	Grains and grain-based products	14	1	0	0	0	0	8	6	1	0	0	0
Canned	Herbs, spices and condiments	15	0	0	0	0	0	6	5	3	1	0	0
Canned	Legumes, nuts and oilseeds	12	3	0	0	0	0	3	7	3	2	0	0
Canned	Meat and meat products	12	2	1	0	0	0	0	0	5	1	0	0
Canned	Milk and dairy products	14	0	0	0	1	0	9	5	1	0	0	0
Canned	Non-alcoholic beverages	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Products for special nutritional use	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Snacks, desserts, and other foods	15	0	0	0	0	0	5	5	2	2	1	0
Canned	Starchy roots and tubers	15	0	0	0	0	0	1	2	0	0	0	0
Canned	Sugar and confectionary	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Vegetables and vegetable products	8	3	2	1	1	0	0	0	2	7	6	0
Not canned	Alcoholic beverages	15	0	0	0	0	0	1	0	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Animal and vegetable fats and oils	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Composite food	6	3	1	4	1	0	9	6	0	0	0	0
Not canned	Drinking water	1	1	4	0	0	0	5	10	0	0	0	0
Not canned	Eggs and egg products	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fish and other seafood	0	1	4	1	0	0	1	3	0	0	0	0
Not canned	Food for infants and small children	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and fruit products	1	1	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and vegetable juices	0	1	5	0	0	0	1	2	0	0	0	0
Not canned	Grains and grain-based products	0	0	1	2	0	0	0	15	0	0	0	0
Not canned	Herbs, spices and condiments	12	3	0	0	0	0	1	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	2	1	2	0	6	5	4	0	0
Not canned	Milk and dairy products	0	5	9	1	0	0	2	13	0	0	0	0
Not canned	Non-alcoholic beverages	7	8	0	0	0	0	1	0	0	0	0	0
Not canned	Products for special nutritional use	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	13	2	0	0	0	0	1	0	0	0	0	0
Not canned	Starchy roots and tubers	0	1	0	0	0	0	1	2	0	0	0	0
Not canned	Sugar and confectionary	10	5	0	0	0	0	1	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	8	7	0	0	0	1	0	0	0	0	0

5600 **Table 43:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5601 scenario - Adolescents (Total number of surveys = 12)

5602

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	1	0	0	0	0	0	1	0	0	0	0	0
Canned	Animal and vegetable fats and oils	1	0	0	0	0	0	1	0	0	0	0	0
Canned	Composite food	9	1	2	0	0	0	4	2	1	2	3	0
Canned	Fish and other seafood	4	4	4	0	0	0	0	3	7	2	0	0
Canned	Fruit and fruit products	1	1	0	0	0	0	1	1	0	0	0	0
Canned	Fruit and vegetable juices	8	2	1	1	0	0	4	6	1	1	0	0
Canned	Grains and grain-based products	1	0	0	0	0	0	7	5	0	0	0	0
Canned	Herbs, spices and condiments	1	1	0	0	0	0	5	2	4	1	0	0
Canned	Legumes, nuts and oilseeds	1	2	0	0	0	0	1	6	3	2	0	0
Canned	Meat and meat products	1	1	1	0	0	0	0	0	0	1	2	0
Canned	Milk and dairy products	1	0	0	0	0	0	1	1	0	0	0	0
Canned	Non-alcoholic beverages	1	0	0	0	0	0	1	0	0	0	0	0
Canned	Products for special nutritional use	1	0	0	0	0	0	1	0	0	0	0	0
Canned	Snacks, desserts, and other foods	1	0	0	0	0	0	6	3	2	1	0	0
Canned	Starchy roots and tubers	1	0	0	0	0	0	1	2	0	0	0	0
Canned	Sugar and confectionary	1	0	0	0	0	0	1	0	0	0	0	0
Canned	Vegetables and vegetable products	5	2	2	2	1	0	0	0	1	7	4	0
Not canned	Alcoholic beverages	1	2	0	0	0	0	1	0	0	0	0	0
Not canned	Animal and vegetable fats and oils	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Composite food	3	3	3	2	1	0	7	5	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Drinking water	1	6	5	0	0	0	3	9	0	0	0	0
Not canned	Eggs and egg products	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fish and other seafood	0	4	8	0	0	0	8	4	0	0	0	0
Not canned	Food for infants and small children	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and fruit products	2	1	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and vegetable juices	3	8	1	0	0	0	1	0	0	0	0	0
Not canned	Grains and grain-based products	0	0	1	2	0	0	0	1	0	0	0	0
Not canned	Herbs, spices and condiments	8	4	0	0	0	0	1	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	0	9	3	0	2	6	4	0	0
Not canned	Milk and dairy products	0	1	1	0	0	0	5	7	0	0	0	0
Not canned	Non-alcoholic beverages	4	8	0	0	0	0	1	0	0	0	0	0
Not canned	Products for special nutritional use	1	1	0	0	0	0	1	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Starchy roots and tubers	0	1	0	0	0	0	1	0	0	0	0	0
Not canned	Sugar and confectionary	1	0	0	0	0	0	1	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	8	4	0	0	0	1	0	0	0	0	0

5606 **Table 44:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5607 scenario - Women 18-45 years (Total number of surveys = 15)

5608

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	14	1	0	0	0	0	1	0	0	0	0	0
Canned	Animal and vegetable fats and oils	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Composite food	11	3	0	1	0	0	4	5	1	2	3	0
Canned	Fish and other seafood	5	4	5	1	0	0	0	4	8	3	0	0
Canned	Fruit and fruit products	10	5	0	0	0	0	0	1	0	0	0	0
Canned	Fruit and vegetable juices	12	2	1	0	0	0	3	1	1	0	0	0
Canned	Grains and grain-based products	12	3	0	0	0	0	8	7	0	0	0	0
Canned	Herbs, spices and condiments	13	2	0	0	0	0	4	8	3	0	0	0
Canned	Legumes, nuts and oilseeds	10	4	0	1	0	0	2	8	3	2	0	0
Canned	Meat and meat products	11	2	2	0	0	0	0	0	1	1	0	0
Canned	Milk and dairy products	15	0	0	0	0	0	1	3	0	0	0	0
Canned	Non-alcoholic beverages	14	1	0	0	0	0	1	0	0	0	0	0
Canned	Products for special nutritional use	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Snacks, desserts, and other foods	15	0	0	0	0	0	7	6	1	1	0	0
Canned	Starchy roots and tubers	15	0	0	0	0	0	1	1	0	0	0	0
Canned	Sugar and confectionary	15	0	0	0	0	0	1	0	0	0	0	0
Canned	Vegetables and vegetable products	5	2	4	4	0	0	0	0	0	5	1	0
Not canned	Alcoholic beverages	6	9	0	0	0	0	1	2	0	0	0	0
Not canned	Animal and vegetable fats and oils	15	0	0	0	0	0	1	0	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA						Scenario 2 % average BPA					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Composite food	5	5	2	2	1	0	1	2	0	0	0	0
Not canned	Drinking water	1	7	5	2	0	0	3	1	0	0	0	0
Not canned	Eggs and egg products	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fish and other seafood	0	9	5	1	0	0	1	3	0	0	0	0
Not canned	Food for infants and small children	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and fruit products	3	1	0	0	0	0	1	0	0	0	0	0
Not canned	Fruit and vegetable juices	5	9	1	0	0	0	1	0	0	0	0	0
Not canned	Grains and grain-based products	0	2	1	1	0	0	0	1	0	0	0	0
Not canned	Herbs, spices and condiments	9	6	0	0	0	0	1	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	2	1	1	0	3	9	3	0	0
Not canned	Milk and dairy products	0	1	1	0	0	0	8	7	0	0	0	0
Not canned	Non-alcoholic beverages	4	1	1	0	0	0	8	7	0	0	0	0
Not canned	Products for special nutritional use	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Starchy roots and tubers	1	1	0	0	0	0	1	1	0	0	0	0
Not canned	Sugar and confectionary	15	0	0	0	0	0	1	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	5	1	0	0	0	1	0	0	0	0	0

5612 **Table 45:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5613 scenario - Men 18-45 years (Total number of surveys = 15)

5614

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	14	0	1	0	0	0	15	0	0	0	0	0
Canned	Animal and vegetable fats and oils	15	0	0	0	0	0	15	0	0	0	0	0
Canned	Composite food	10	3	2	0	0	0	5	4	1	2	3	0
Canned	Fish and other seafood	5	6	3	1	0	0	0	3	9	3	0	0
Canned	Fruit and fruit products	13	2	0	0	0	0	4	11	0	0	0	0
Canned	Fruit and vegetable juices	12	2	1	0	0	0	6	8	1	0	0	0
Canned	Grains and grain-based products	13	2	0	0	0	0	9	6	0	0	0	0
Canned	Herbs, spices and condiments	13	2	0	0	0	0	4	8	3	0	0	0
Canned	Legumes, nuts and oilseeds	9	4	1	1	0	0	1	8	4	2	0	0
Canned	Meat and meat products	10	3	2	0	0	0	0	0	0	13	2	0
Canned	Milk and dairy products	15	0	0	0	0	0	14	1	0	0	0	0
Canned	Non-alcoholic beverages	14	1	0	0	0	0	15	0	0	0	0	0
Canned	Products for special nutritional use	15	0	0	0	0	0	15	0	0	0	0	0
Canned	Snacks, desserts, and other foods	15	0	0	0	0	0	8	5	1	1	0	0
Canned	Starchy roots and tubers	15	0	0	0	0	0	14	1	0	0	0	0
Canned	Sugar and confectionary	15	0	0	0	0	0	15	0	0	0	0	0
Canned	Vegetables and vegetable products	6	0	6	2	1	0	0	0	0	5	10	0
Not canned	Alcoholic beverages	1	11	3	0	0	0	6	9	0	0	0	0
Not canned	Animal and vegetable fats and oils	15	0	0	0	0	0	15	0	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Composite food	7	4	2	2	0	0	13	2	0	0	0	0
Not canned	Drinking water	2	9	4	0	0	0	5	10	0	0	0	0
Not canned	Eggs and egg products	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Fish and other seafood	0	9	5	1	0	0	13	2	0	0	0	0
Not canned	Food for infants and small children	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Fruit and fruit products	11	4	0	0	0	0	15	0	0	0	0	0
Not canned	Fruit and vegetable juices	8	6	1	0	0	0	14	1	0	0	0	0
Not canned	Grains and grain-based products	0	2	13	0	0	0	0	15	0	0	0	0
Not canned	Herbs, spices and condiments	10	5	0	0	0	0	15	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	1	10	4	0	1	7	7	0	0
Not canned	Milk and dairy products	0	14	1	0	0	0	10	5	0	0	0	0
Not canned	Non-alcoholic beverages	4	11	0	0	0	0	8	7	0	0	0	0
Not canned	Products for special nutritional use	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Starchy roots and tubers	2	13	0	0	0	0	14	1	0	0	0	0
Not canned	Sugar and confectionary	15	0	0	0	0	0	15	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	10	5	0	0	0	15	0	0	0	0	0

5618 **Table 46:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5619 scenario – Other adults 45-65 years (Total number of surveys = 14)

5620

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	13	1	0	0	0	0	14	0	0	0	0	0
Canned	Animal and vegetable fats and oils	14	0	0	0	0	0	14	0	0	0	0	0
Canned	Composite food	10	3	0	1	0	0	3	5	2	1	3	0
Canned	Fish and other seafood	5	5	3	1	0	0	0	1	6	7	0	0
Canned	Fruit and fruit products	10	4	0	0	0	0	1	1	0	0	0	0
Canned	Fruit and vegetable juices	12	2	0	0	0	0	8	6	0	0	0	0
Canned	Grains and grain-based products	12	2	0	0	0	0	8	6	0	0	0	0
Canned	Herbs, spices and condiments	14	0	0	0	0	0	7	7	0	0	0	0
Canned	Legumes, nuts and oilseeds	10	2	1	1	0	0	1	7	4	2	0	0
Canned	Meat and meat products	10	3	1	0	0	0	0	0	0	1	1	0
Canned	Milk and dairy products	14	0	0	0	0	0	12	2	0	0	0	0
Canned	Non-alcoholic beverages	14	0	0	0	0	0	14	0	0	0	0	0
Canned	Products for special nutritional use	14	0	0	0	0	0	14	0	0	0	0	0
Canned	Snacks, desserts, and other foods	14	0	0	0	0	0	9	5	0	0	0	0
Canned	Starchy roots and tubers	14	0	0	0	0	0	13	1	0	0	0	0
Canned	Sugar and confectionary	14	0	0	0	0	0	14	0	0	0	0	0
Canned	Vegetables and vegetable products	6	0	7	1	0	0	0	0	0	4	10	0
Not canned	Alcoholic beverages	1	12	1	0	0	0	8	6	0	0	0	0
Not canned	Animal and vegetable fats and oils	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Composite food	7	3	1	3	0	0	12	2	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Drinking water	1	7	6	0	0	0	4	1	0	0	0	0
Not canned	Eggs and egg products	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Fish and other seafood	0	5	6	3	0	0	11	3	0	0	0	0
Not canned	Food for infants and small children	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Fruit and fruit products	2	12	0	0	0	0	14	0	0	0	0	0
Not canned	Fruit and vegetable juices	8	6	0	0	0	0	14	0	0	0	0	0
Not canned	Grains and grain-based products	0	2	1	0	0	0	0	1	0	0	0	0
Not canned	Herbs, spices and condiments	12	2	0	0	0	0	14	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	1	11	2	0	2	9	3	0	0
Not canned	Milk and dairy products	0	13	1	0	0	0	11	3	0	0	0	0
Not canned	Non-alcoholic beverages	4	9	1	0	0	0	8	6	0	0	0	0
Not canned	Products for special nutritional use	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Starchy roots and tubers	2	12	0	0	0	0	13	1	0	0	0	0
Not canned	Sugar and confectionary	14	0	0	0	0	0	14	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	5	9	0	0	0	14	0	0	0	0	0

5624 **Table 47:** Number of dietary surveys according to the percentage of average dietary exposure to BPA per type of packaging (canned vs. not canned) and
5625 scenario - Elderly and very elderly (Total number of surveys = 7)

5626

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Canned	Alcoholic beverages	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Animal and vegetable fats and oils	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Composite food	7	0	0	0	0	0	2	3	0	1	1	0
Canned	Fish and other seafood	4	2	0	1	0	0	0	1	3	3	0	0
Canned	Fruit and fruit products	6	1	0	0	0	0	1	3	3	0	0	0
Canned	Fruit and vegetable juices	6	1	0	0	0	0	5	2	0	0	0	0
Canned	Grains and grain-based products	7	0	0	0	0	0	5	2	0	0	0	0
Canned	Herbs, spices and condiments	7	0	0	0	0	0	5	2	0	0	0	0
Canned	Legumes, nuts and oilseeds	7	0	0	0	0	0	0	5	1	1	0	0
Canned	Meat and meat products	5	2	0	0	0	0	0	0	0	7	0	0
Canned	Milk and dairy products	7	0	0	0	0	0	4	3	0	0	0	0
Canned	Non-alcoholic beverages	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Products for special nutritional use	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Snacks, desserts, and other foods	7	0	0	0	0	0	5	1	1	0	0	0
Canned	Starchy roots and tubers	7	0	0	0	0	0	6	1	0	0	0	0
Canned	Sugar and confectionary	7	0	0	0	0	0	7	0	0	0	0	0
Canned	Vegetables and vegetable products	3	0	3	0	1	0	0	0	0	2	5	0
Not canned	Alcoholic beverages	0	7	0	0	0	0	5	2	0	0	0	0
Not canned	Animal and vegetable fats and oils	7	0	0	0	0	0	7	0	0	0	0	0

Packaging type	FoodEx Level 1 category	Number of dietary surveys (Middle Bound)											
		Scenario 1 % average BPA contribution						Scenario 2 % average BPA contribution					
		< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %	< 1 %	1 – 5 %	5 – 10 %	10 – 25 %	25 – 50 %	50 – 75 %
Not canned	Composite food	5	0	1	1	0	0	7	0	0	0	0	0
Not canned	Drinking water	1	3	3	0	0	0	2	5	0	0	0	0
Not canned	Eggs and egg products	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Fish and other seafood	0	3	2	2	0	0	7	0	0	0	0	0
Not canned	Food for infants and small children	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Fruit and fruit products	0	7	0	0	0	0	7	0	0	0	0	0
Not canned	Fruit and vegetable juices	5	2	0	0	0	0	7	0	0	0	0	0
Not canned	Grains and grain-based products	0	0	7	0	0	0	0	7	0	0	0	0
Not canned	Herbs, spices and condiments	6	1	0	0	0	0	7	0	0	0	0	0
Not canned	Legumes, nuts and oilseeds	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Meat and meat products	0	0	0	0	6	1	0	1	5	1	0	0
Not canned	Milk and dairy products	0	6	1	0	0	0	4	3	0	0	0	0
Not canned	Non-alcoholic beverages	1	6	0	0	0	0	4	3	0	0	0	0
Not canned	Products for special nutritional use	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Snacks, desserts, and other foods	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Starchy roots and tubers	0	7	0	0	0	0	6	1	0	0	0	0
Not canned	Sugar and confectionary	7	0	0	0	0	0	7	0	0	0	0	0
Not canned	Vegetables and vegetable products	0	2	5	0	0	0	7	0	0	0	0	0

5627

5628

5629 # APPENDIX VI: EQUATIONS AND PARAMETERS USED IN THE CALCULATION OF EXPOSURE FROM NON-DIETARY SOURCES

5630 **Table 48:** Overview of the equations and parameters used for calculating exposure from non-food sources.

5631

Pathway/Source	Formula	Parameters
		General: $\mathbf{r_{uptake}}$: uptake fraction (-) bw: bodyweight (kg bw) $\mathbf{E_{source}}$: Exposure contribution of respective source (ng/kg bw /d)
Ingestion/dust	$E_{dust} = \frac{C_{dust} \cdot q_{dust}}{bw} \cdot r_{absorption}$	$\mathbf{C_{dust}}$: concentration in dust (median) (ng/mg) $\mathbf{q_{dust}}$: dust ingestion (mg/d)
Ingestion/mouthing of toys Ingestion/mouthing of pacifiers	$E_{toy} = \frac{q_{product} * f_{time} * f_{surface}}{bw} * r_{absorption}$	$\mathbf{q_{product}}$: total amount of BPA that migrated into artificial saliva (ng) $\mathbf{f_{time}}$: correction factor sucking time per day/duration of migration experiment (1/d) $\mathbf{f_{surface}}$: correction factor for contact surface (-)
Ingestion/dental materials	$E_{dental} = \frac{C_{saliva} * q_{saliva}}{bw} * r_{absorption}$	$\mathbf{C_{saliva}}$: concentration of BPA in saliva after dental treatment (µg/l) $\mathbf{q_{saliva}}$: ingested saliva (mL/d)
Ingestion/thermal paper transfer to food	$E_{tp-food} = \frac{a_{finger} * n_{finger} * f_{avail} * f_{trans} * q_{handling}}{bw} * r_{absorption}$	a_{finger}: amount on finger after touching thermal paper (ng) n_{finger}: number of fingers touching thermal paper (-) f_{avail}: available fraction for transfer to food (-) f_{trans}: transfer fraction to food (-) $q_{handling}$: handling events with transfer (1/d)
Inhalation/air	$E_{air} = \frac{C_{air} * q_{air}}{bw} * r_{absorption}$	C_{air}: concentration in air (ng/m^3) q_{air}: quantity of inhaled air per day (m^3/d)
Dermal uptake/thermal paper	$E_{tp-dermal} = \frac{a_{finger} * n_{finger} * q_{handling}}{bw} r_{absorption}$	a_{finger}: amount on finger after touching thermal paper (ng) n_{finger}: number of fingers touching thermal paper (-) $q_{handling}$: handling events (1/d)
Dermal uptake/cosmetics	$E_{cosmetics} = \frac{C_{cosmetics} * q_{cosmetics} * f_{ret}}{bw} * r_{absorption}$	$C_{cosmetics}$: concentration in cosmetics (ng/mg) $q_{cosmetics}$: applied amount per day (mg/d) f_{ret}: retention factor (1 for leave-on) (-)

5632
5633

APPENDIX VII: BIOMONITORING

The Appendix VII contains estimation of daily BPA intake from creatinine-adjusted urinary concentration and a summary of the biomonitoring studies on urinary BPA levels are available from North and South America, Africa, Asia and Australia

Estimation of daily BPA intake from creatinine-BASED urinary concentration

For the estimation of daily BPA intake, volume-based BPA concentrations (µg BPA/l urine) are generally preferred over creatinine-based urinary concentrations (µg BPA/g creatinine) (Lakind and Naiman, 2008; Mahalingaiah et al., 2008; Geens et al., 2012b). The arguments against creatinine-based data are the (i) larger variation range of >1 000 % in urinary creatinine concentration compared to up to 300 % variation in daily urinary volume (Boeniger et al., 1993), and (ii) the differences in the physiological mode of urinary excretion (active secretion, filtration) between glucuronidated BPA and creatinine (Boeniger et al., 1993; Mahalingaiah et al., 2008). Although the large North American surveys (NHANES, CHMS) indicate an approximately 10-fold difference between the 5th and 95th percentiles in (spot urine) creatinine concentration (Health Canada, 2012), the comparison between (spot urine) creatinine concentration and daily urinary volume falls short, because in the latter the within-day variation is removed. Morever, although one may expect an increase in variability by dividing one fluctuating variable (BPA concentration) by another (creatinine concentration), there is de facto no increase in the P95-to-P50 ratio between volume-based BPA concentrations and creatinine-based urinary BPA concentrations. An additional argument for the use of creatinine-based concentrations instead of volume-based concentrations is the fact that the former is not dependent on the drinking behaviour. An example of changing drinking behavior is the retrospective study by Koch et al. (2012), who reported the increase in 24-h urine volume from 1.6 to 2.1 L in German students between 1995 and 2009, which was associated with a decrease in mean urinary creatinine concentration from 1.2 to 0.8 g/L. The daily urinary excretion of creatinine, in contrast, depends primarily on the muscle mass of the individual. A man excretes 14–16 mg/kg bw/day, and a woman 11–20 mg/kg bw/day, but the amount is fairly consistent for a given individual (McClatchey, 2002).

Based on creatinine-based urinary concentration of total BPA X_{BPA} (µg/g creatinine), the daily BPA exposure $\dot{m}_{BPA}$ (ng/kg bw/day) was calculated by

$$\dot{m}_{BPA} = \frac{X_{BPA} \times \dot{m}_{creatinine}}{W}$$

where $\dot{m}_{creatinine}$ (g/day) is the creatinine excretion rate and W (kg) is the body weight (Lakind and Naiman, 2008; UBA, 2012). Depending on whether body-weight is available from the studies, either study-specific individual or mean values, or generic values derived by linear interpolation from body weight vs. age relationships taken from literature, were used. Age-specific generic values on daily creatinine excretion were taken from Valentin (2002) except for cases where study-specific values from 24-h urine sampling were available. Table 49 shows the body-weight and creatinine excretion-rate parameters which were used to translate creatinine-based BPA concentration into daily BPA exposure. Generic values for the creatinine excretion rate were taken from ICRP reference tables (Valentin 2002).

Age-specific estimates were only available from a few European studies, and only for children, adolescents, adults and the (very) elderly. For the children, the creatinine-based BPA intakes tend to be lower than the volume-based BPA intakes (e.g. 39 *vs.* 53 ng/kg bw/day for the Duisburg birth cohort study). The same tendency applies for the adolescents and the adults except the German ESB study and the MoBa study (Figure 15). In the German ESB study, a sensible difference is not to be

expected because both (creatinine-based and volume-based) exposure estimates were derived from 24-h urine and creatinine excretions of the study participants rather than from generic values from literature. For the (very) elderly, the Liege HBM study indicates that the creatinine-based intake is somewhat higher than the volume-based intake (49 *vs*. 40 ng/kg bw/day).

The daily BPA intake as estimated from creatinine-adjusted urinary BPA concentrations are shown in Figure 1 (red symbols). For comparative purposes, estimates derived from volume-based urinary BPA concentrations (black symbols) are additionally shown.

Table 49: Body-weight and creatinine excretion-rate parameters for the considered European and North American Studies. Given are the parameters for body weight (W), creatinine excretion rate ($\dot{m}_{creatinine}$), and the specific creatinine excretion rate (spec. $\dot{m}_{creatinine}$). Gender and age were taken into account when deriving generic parameter values from published parameter-age relationships by linear interpolation. Study-specific parameters are set in italic font. References from which these parameters were taken are: [1] Koch et al. (2012), [2] Bergmann and Mensink (1999), [3] Valentin (2002), [4] Stolzenberg et al. (2007), [5] Ye et al. (2009), [6] CDC (2012), [7] Health Canada (2012), [8] M. Kasper-Sonnenberg (personal communication), [9] E. Den Hond (personal communication).

Study	**gender**	**age**	**Sampling**	W	$\dot{m}_{creatinine}$	**spec.** $\dot{m}_{creatinine}$	**Reference**
				(kg)	(ml/day)	(ml/kg/day)	
German ESB	MF	20–30 yr	24hU	*72*	1 000	14	[1]
Duisburg BCS	F	29–49 yr	MU	*71*	1 000	14	[8, 3]
Duisburg BCS	MF	6–8 yr	MU	*24*	458	17	[8, 3]
Generation R	pregnant F	18–41 yr	SU	74	1 000	14	[5, 3]
MoBa	pregnant F		SU	*74*	1 000	14	[5, 3]
Flemish HMB	MF	14–16 yr	SU	*57*	1200	21	[9, 3]
Liege HMB	MF	7–11 yr	MU	34	586	17	[2, 3]
Liege HMB	MF	12–19 yr	MU	65	1 200	19	[2, 3]
Liege HMB	MF	20–39 yr	MU	75	1 350	18	[2, 3]
Liege HMB	MF	40–59 yr	MU	79	1 350	17	[2, 3]
Liege HMB	MF	60–75 yr	MU	78	1 350	17	[2, 3]
NHANES	MF	6–>65 yr	SU	*29–83*	490–1350	16–18	[6, 3]
CHMS	MF	6–79 yr	SU	*33–80*	650–1 350	17–19	[7, 3]

The differences between creatinine-based and volume-based BPA exposure estimates among the European studies suggest that generic values for the daily urine volume overestimate the true daily urine volume in the children, adolescents, and the adults. In the (very) elderly, the situation seems to be reversed. This hypothesis is corroborated by North-American surveys (NHANES, CHMS), for which explanatory information on urinary creatinine concentration is additionally available (Table 50). For the adolescents and adults of the NHANES survey, the actual creatinine concentrations are higher than the generic predictions (e.g. 1.33 *vs*. 0.92 g/L for the adolescents), which explains the lower creatinine-based BPA exposure estimates compared to the volume-based BPA exposure estimates (Figure 1). In other words, US adolescents and adults produce less urine than expected from literature data and produce, therefore, a more concentrated urine. Using volume-based urinary BPA concencentrations in combination with generic values from literature on daily urinary output will consequently overestimate the daily BPA exposures for US adolescents and adults. Explanations for differences among the US (very) elderly and among the Canadian population groups can be derived in a similar manner.

In conclusion, the estimation of daily BPA exposure from creatinine-based urinary BPA concentrations lead to slighly different values than those obtained from volume-based urinary BPA concentrations. For the few European studies (with the exception of the German ESB study), there is a tendency for lower BPA exposures in children, adolescents and adults, and a tendency for slightly higher BPA exposures for the (very) elderly. These differences are (at least partly) explainable by daily urinary outputs that deviate from the generic values taken from literature. For the derivation of reference values for the comparison with BPA uptake via food and non-food resources, the volume-based BPA intakes will be used because these are more conservative and better supported by a larger number of European studies.

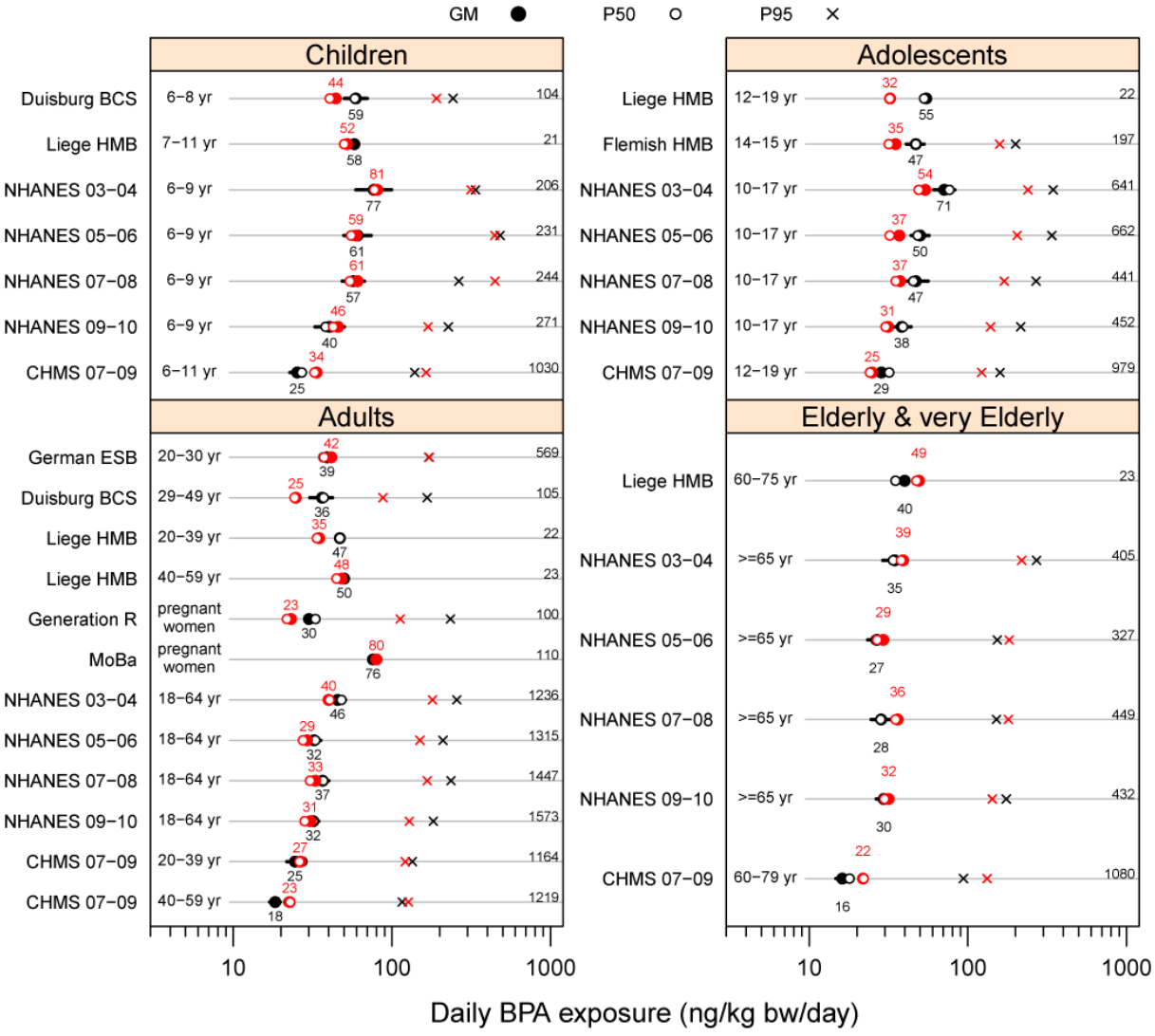


Figure 15: Daily BPA exposures as estimated from creatinine-based urinary BPA concentration. Shown are the estimates derived from creatinine-based (red) and volume-based (black) urinary BPA concentrations.

Table 50: Comparison of study-specific and generic urinary creatinine concentrations. Given are the (average) median creatinine concentration for different age groups of the NHANES survey (2003–2004, 2005–2006, 2007–2008, 2009–2010) survey and the CHMS 2007–2009 survey. Additionally given are the generic values which were obtained from Valentin (2002) by dividing the (age-specific) creatinine excretion rate by urinary output rate.

Age group	Urinary [Cr] (g/L)		Age group	Urinary [Cr] (g/L)	
	NHANES	Valentin		CHMS	Valentin
Children	0.86	0.82	6–11 yr	0.75	0.86
Adolescents	1.33	0.92	12–19 yr	1.33	1.00
Adults	1.20	0.96	20–39 yr	1.01	0.96
(Very) Elderly	0.91	0.96	40–59 yr	0.87	0.96
			60–79 yr	0.81	0.96

Biomonitoring studies on urinary BPA levels from non-European studies excluding NHANES and CHMS

Further data from biomonitoring studies on urinary BPA levels are available from North and South America, Africa, Asia and Australia.

Mahalingaiah et al. (2008) analysed 217 spot urine samples collected from *n*=82 male and female partners (seeking infertility evaluation and treatment in a hospital) in Massachusetts in 2004–2006 by HPLC/MS-MS (LOD = 0.36 µg/l) and detected total in 87 % of the samples with a geometric mean (GM) of 1.31 µg/l. Ehrlich et al. (2012) analysed 325 spot urine samples collected from *n*=137 18–45 years old women (undergoing *in-vitro* fertilisation in a hospital) in Boston in 2004–2010 by HPLC-MS/MS (LOD = 0.4 µg/l) and detected total BPA in 88 % of the samples with a GM of 1.53 µg/l and a 95th percentile of 6.04 µg/l. Morgan et al. (2011) analysed pooled serial spot urine samples collected from *n*=81 preschool children (2–5 years old) in Ohio in 2000–2001 by HPLC-MS/MS (LOD = 0.4 µg/l) and detected total BPA in 100 % of the samples with a GM of 4.8 µg/l and a 95th percentile of 20.8 µg/l.

Cantonwine et al. (2010) analysed spot urine samples collected from *n*=60 pregnant women from Mexico city in 2001–2003 by HPLC-MS/MS (LOD = 0.4 µg/l) and detected total BPA in 80 % with a GM of 1.5 µg/ml and a 95th percentile of 5.7 µg/l.

Nahar et al. (2012) used HPLC-MS/MS (LOD = 0.4 µg/l) and measured spot urine samples collected from *n*=57 healthy 10–13 year old premenstrual girls from rural and urban areas near Cairo in 2009; total BPA was detected in 79 % of the samples with a GM of 0.84 µg/l.

He et al. (2009) analysed spot urine samples from *n*=922 family members of industrial workers from east and middle mainland China by HPLC-FLD (LOD = 0.31 µg/l) and detected total BPA in 50 % of the samples with a GM of 0.87 µg/l. Within the framework of the Korean National HBM survey, Kim et al. (2011) analysed spot urine samples collected from *n*=1 870 subjects (18–69 years old) in 2009 by GC-MS (LOD = 0.05 µg/l) and detected total BPA in 99.8 % of the samples with a GM of 1.90 µg/l and a 95th percentile of 7.74 µg/l. Li et al. (2013) analysed morning urine samples from *n*=287 children and students (3–24 years old) from South China by GC-MS (LOD = 0.0005 µg/l) and detected total BPA in 100 % of the samples with a GM of 3.0 µg/l. Zhang et al. (2011a) analysed spot urine samples collected from *n*=296 subjects of the general population in seven Asian countries (China, India, Japan, Korea, Kuwait, Malaysia, Vietnam) in 2006–2010 by HPLC-MS/MS (LOQ = 0.1

µg/l) and detected total in 94.3 % of the samples with a GM of 1.2 µg/l. The GM for the individual countries ranged from 0.84 µg/l (Japan, *n*=36 samples) to 2.0 µg/l (Korea, *n*=32 samples).

Callan et al. (2012) used HPLC-MS/MS (limit of reporting: 0.48 and 1.30 µg/l for batch 1 and 2) and measured 1st morning urine samples collected from *n*=26 pregnant woman (25–39 years) from Western Australia in 2011; total BPA was detected in 85 % of the samples with a GM of 1.63 µg/l and a median of 2.41 µg/l.

APPENDIX VIII: EVALUATION OF UNCERTAINTIES IN THE EXPOSURE ASSESSMENT THROUGH EXPERT JUDGEMENT

This Appendix documents the approach taken to evaluating uncertainties affecting the Panel's exposure assessment for BPA and presents the detailed results for different parts of the exposure assessment.
The general approach is adapted from the method for qualitative evaluation of uncertainty that was suggested in EFSA guidance on dealing with uncertainties in exposure assessment (EFSA, 2006). The suggested approach comprised the following key steps:

- Systematically examine every part of the assessment for potential sources of uncertainty
- List the identified uncertainties in a table
- Evaluate the impact of each uncertainty on the outcome of the exposure assessment, using a suitable scale
- Evaluate the combined impact of all the uncertainties, considered together, on the outcome of the exposure assessment

The evaluation of uncertainties is approximate, using expert judgment. EFSA (2006) suggested expressing the evaluation on a qualitative scale, provided the scale was defined, and showed an example where this was done with combinations of '+' and '-' symbols. Subsequently it was realised that while helpful in indicating the relative magnitudes of uncertainties, a qualitative scale does not give any indication how large they are in absolute terms, which is in principle needed for risk management. For example, if an exposure estimate is 10, its uncertainty is evaluated as '++' and the corresponding TDI is 20, then the risk manager needs to know whether ++ means the true exposure could be larger by a factor of 2 or more, because that would imply potential exceedance of the TDI. Therefore, some later EFSA opinions provided quantitative scales for the symbols, notably the EFSA PPR Panel's guidance document on probabilistic modelling of dietary exposure (EFSA Plant Protection Products and their Residues Panel, 2012).

The general principles above have been applied to the exposure assessment, but the detailed methodology and format of the evaluation have been adapted to suit the differing needs of different parts of the assessment, as described below.

The uncertainty analysis is focussed on the parts of the assessment that contribute to the assessment of high total exposure (rather than average), since this is of particular interest for risk characterisation. The following chapters assess uncertainty for each of the individual sources of exposure, which contribute to the assessment of high total exposure, e.g. the assessment for the women of child bearing age combines high exposure for the dietary route and for dermal exposure via thermal paper with average exposure for all other sources. How the uncertainties for different sources combine is considered in chapter 4.9.3 of the main Opinion, in order to reach a conclusion on the overall uncertainty associated with the assessment of high total exposure. Uncertainties associated with the biomonitoring data on BPA in urine are also assessed below, since these data are used in chapter 4.9.3 as an additional line of evidence to support the overall conclusions about high total exposure.

Uncertainties affecting the estimation of exposures were evaluated using a tabular format similar to the original suggestions of EFSA (EFSA, 2006). The Panel's assessment of the impact of each uncertainty was expressed using symbols whose meaning is defined on a quantitative scale (Figure 10). Plus symbols mean that the true value of the exposure could be higher than the estimate; minus symbols mean that the true value could be lower; a dot (●) means the impact of the uncertainty is less than +/- 20 %. Since the evaluation is approximate, each symbol represents a range of possible values; for example, '++' means the true exposure is judged to be between 2 and 5 times the estimate. Pairs of symbols are used where the uncertainty spans a larger range; for example -/++ would mean the true

value exposure is judged to be between half and five times the estimate. However, the relative likelihood of different values within the range was not assessed.

It is emphasised that all the evaluations are approximate expert judgements and should not be interpreted as precise estimates.

1. Uncertainties in the assessment of dietary exposure (excluding breastfed infants)

Uncertainties affecting the estimation of high dietary exposures were evaluated by adding two extra columns to the tabular format suggested by EFSA (EFSA, 2006) (Table 51). The left hand column in Table 51 lists the sources of uncertainty identified, and the right hand column gives the Panel's evaluation of the impact of those uncertainties on its estimates of high exposure, using symbols from the scale in Figure 10. The two additional columns, in the centre of the table, identify the variable that is affected by each uncertainty, and the value(s) used for that variable in the Panel's calculation of high exposure.

The scale in Table 51 was also used to evaluate the combined impact of all the uncertainties on the assessment of high dietary exposures, which is shown in the bottom row of Table 1 together with a short explanation of how it was derived.

Table 51: Evaluation of uncertainties affecting the assessment of high dietary exposure The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 1 for key to symbols.

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
The Comprehensive Database includes nine surveys for toddlers, 17 surveys for other children, 12 surveys for adolescents, 15 surveys for adults, seven surveys for elderly and six surveys for very elderly. Consumption patterns in other Member States can be different.	Food consumption	Individual food consumption data	- / +
Food consumption data for women aged from 18 to 45 years old from 15 different surveys have been used as a proxy for women of child-bearing age. Younger and older women can still be considered in child-bearing age. Women can change their consumption patterns when becoming pregnant.	Food consumption	Individual food consumption data	●
Dietary data in the Comprehensive Database have been collected by means of different study designs, methodologies and protocols which could bias their results in a different way for each survey. In particular, the following parameters may affect the level of detail and the accuracy of the collected data: the dietary assessment method used, the description and codification of the food consumed, the number of days per subject, the sampling design and size, the management of under-reporters, the quantification of portion sizes, the software applications used and the non-dietary information collected. Furthermore, in some of the countries, data provided to EFSA	Food consumption	Individual food consumption data	-/+

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
came from relatively old national dietary surveys.			
Increasing the number of survey days (for both recalls and records) has the advantage of reducing the effect of study subjects' day-to-day variation, thus leading to an improved estimation of consumption variability. As survey duration increases, high percentiles consumption decreases. This might be particularly important for episodically consumed foods, as some kind of canned foods could be. Only food consumption data collected on more than one day per subject have been used to assess chronic exposure. The number of days per subject ranged from two to three in toddlers and from two to seven in women aged from 18 to 45 years old.	Food consumption	Individual food consumption data	-/●
Only a limited number of dietary surveys included in the Comprehensive Database presented information on the type of packaging (canned or non-canned, in particular). Two scenarios were therefore considered, 1) only food specifically codified as canned were considered as such 2) at FoodEx level 4, any food which has been codified as canned in at least one survey is always considered to be consumed as canned in all dietary surveys included in the Comprehensive Database. The ratio between the 95th percentiles calculated under scenario 2 and scenario 1 ranged from 4 to 4.8 in toddlers and from 2.1 to 6.8 among women aged from 18 to 45 years old.	Food consumption	Individual food consumption data	- / ● (scenario 2)
Different methods of analysis have been used to quantify BPA in food and beverages, all presenting an uncertainty. Occurrence data from different origins, Total Diet Studies (TDS), monitoring and literature. Data on occurrence of BPA in food retrieved from scientific journals can be biased towards positive results since negative results are not always published. Data from TDS can be biased due to the pooling of the food samples. Data from the literature represent 22 % of the samples. It is therefore expected that this bias produce limited effects.	BPA occurrence levels	Average BPA concentration assessed by merging data from different sources or publications.	●
Food samples below the limit of quantification or reporting were handled through the substitution method: the lower bound (LB) value was obtained by assigning a value of zero to all the samples reported as less than the left-censoring limit, the middle bound (MB) value by assigning half of the left-censoring limit and the upper bound (UB) by assigning the left-censored limit as the sample result. At the 95th percentile, MB exposure estimates	BPA occurrence levels	Average BPA occurrence for LB, MB and UB have been calculated.	●

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
were 4 to 20 % (scenario 1) and 2 to 9 % (scenario 2) higher than those calculated using the LB method and 2 to 20 % (scenario 1) and 2 to 8 % (scenario 2) lower than those calculated using the UB method.			
Bias could have been introduced by the limited number of samples for some of the categories and due to the large food categories, specific foods could present lower or higher levels. In particular, relatively high levels of BPA in non-canned meat and fish have been identified in many samples from France and one from Ireland. These are difficult to explain, more samples from different countries would have been useful.	BPA occurrence levels	Average BPA occurrence for each FoodEX level 1 food group and type of packaging (canned or non-canned).	- / +
Bias could have been introduced due to the limited number of samples and Member States represented. France data are, for example, predominant for non-canned food and beverages. BPA levels could be lower or higher in some of the Member States. On average, specific population groups could be exposed to systematically lower or higher levels than those calculated at EU level, e.g. through the consumption of specific brands.	BPA occurrence levels	Average BPA occurrence has been calculated at EU level.	- / +
In general, analytical determination performed in food were aimed at quantifying unconjugated BPA and would not allow to detect or quantify conjugated BPA (sulfated, glucuronidated) or chlorinated BPA. Based on ANSES specific analysis, conjugated BPA represent a very minor fraction of total BPA. A unique study was retrieved in which chlorinated BPA was quantified but it did not reach the quality criteria established by the Panel. Chlorinated BPA was not detectable in the serum samples collected from 14 healthy volunteers notwistanding the very low LOD (0.05 µg/l). This uncertainty is therefore likely to have a minor impact on the estimate of high exposure.	BPA occurrence levels	Total BPA	●
Data on body weight at subject level was used. Direct measurements were taken in some of the surveys, while in the remaining, self reported measures were used.	Body weight	Individual body weights	●
Toddlers: High levels of exposure have been estimated by means of the 95th percentile for the total population. A limited number of subjects were available for some of the age classes. In particular, in the case of toddlers the 95th percentile was assessed only for four surveys presenting at least 60 subjects per study.	BPA exposure	Highest 95th percentile among toddlers from 4 different dietary surveys	- / +
Women aged 18 to 45: High levels of exposure have been estimated by means of the 95th percentile for the total population. A limited number of subjects were available for	BPA exposure	Highest 95th percentile among women aged from 18 to 45 years old	- / ●

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
some of the age classes. In particular, in the case of women aged from 18 to 45 years old the 95th percentile was assessed for 15 surveys.		from 15 different dietary surveys	
Overall assessment: The main source of uncertainty in the assessment of dietary exposure to BPA is due to limitations in the representativity of the available information on food consumption and BPA occurrence in food. In the case of toddlers, the age group presenting the highest exposure estimates, only for four surveys it was possible to calculate the 95th percentile of exposure whereas this was possible for 15 dietary surveys in the case of women aged from 18 to 45 years old. Noteworthy is also the fact that food consumption data from different surveys presents different levels of bias due to the different study designs, methodologies and protocols used. Exposure could also have been under or over estimated due to the limited number of analytical BPA samples, mainly available for specific food categories and from a scarce number of Member States. A clear overestimation has been introduced in the assessment of dietary exposure to BPA by not correcting for usual intake and by assuming (scenario 2) that any food which has been codified as canned in at least one survey is always consumed as canned in all dietary surveys.			- / ● In women aged from 18 to 45 years old - / + In toddlers

2. Uncertainties in the assessment of exposure for breast-fed infants

Exposure of breast-fed infants is assessed separately from the rest of the population and involves only two variables: the concentration of BPA in human breast milk and the consumption of breast milk by infants (expressed per kg body weight). Uncertainties affecting this assessment are evaluated in the table 52.

Table 52: Evaluation of uncertainties affecting the estimation of high exposure of breast-fed infants to BPA in human breast milk. The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 1 for key to symbols.

Source of uncertainty	Parameter affected	Impact of uncertainty on high exposure estimate
Analytical uncertainty for concentrations above LOD. Recovery: Not a problem in studies (6 of 8) using isotope-dilution mass spectrometry due to the implicit recovery correction.	BPA concentration in breast milk	●
Repeatability: Intra- and inter-day CV <15% for MS-based methods		●
Accuracy: < ±10% (intra- and interday)		●
Contamination of breast-milk samples. Only 3 out of 8 studies (all from the same lab) measured both unconjugated and total BPA. The median proportion of unconjugated BPA ranged from <30% to 76%. It is unclear whether the variable proportion in unconjugated BPA arises from contamination and/or from enzymatic deconjugation by a breast-milk β-glucuronidase during sample collection and storage.	BPA concentration in breast milk	– / ●
Sampling uncertainty Number of subjects ranges from *n*=3–4 in method-development studies to *n*=20–100 in other studies. The relatively low number of subjects per study and the non-representative sampling may result in a	BPA concentration in breast milk	–/+

Source of uncertainty	Parameter affected	Impact of uncertainty on high exposure estimate
sampling bias. This affects the study estimates for the central tendency and the variability, which both enter into the calculation of the high BPA concentration.		
Uncertainty about the variability of the population means The number of studies (N=8) is low, and only four studies (the moderately sized ones) were finally considered for the estimation of average and high concentrations of unconjugated and total BPA. The estimate for the average concentration of total BPA in initial breast milk (colostrum) is based on the sample mean of one study only. For mature breast milk, the estimate is based on taking the average of the sample means of two studies only. Based on this low number of studies, there is practically no information on variability of the sample means across different populations or countries. Information on this inter-country variability is especially relevant for the calculation of the high BPA concentration in order to capture high levels of exposure that may occur in specific geographic areas. The absence of this information leads to an uncertainty which is judged to by greater than 20% but lower than 200%.	BPA concentration in breast milk	●/+
Distribution uncertainty. There are generally not enough data per study to directly get a reliable empirical (non-parametric) estimate of the 95th percentile. However, the available raw data for the moderately sized ($n \geq 20$) studies suggest a log-normal distribution so that a parametric estimation of the 95th percentile appears feasible. Based on the interquartile ranges (IRQs) of three studies, and by assuming a log-normal distribution, an average standard deviation was derived which was then used (together with a mean value) to estimate the 95% percentile as a measure for the high BPA concentration. In principle, this estimate is conservative as the calculated standard deviation reflects not only the between-individual variability but also the within-individual variability, which would average out in the long term. (Repeated/serial milk collections are unfortunately not available to estimate the relative contributions of these two variabilities). The thus obtained generic estimates for high BPA concentration are plausible except for one study (Duty et al., 2013) which showed a more long-tailed distribution. This justifies the selection of a two-sided rather than one-sided uncertainty.	BPA concentration in breast milk	–/+
Uncertainty about regional differences. In the breast-milk database, the European countries are, in essence, not covered. However, based on the urinary BPA concentrations, there is no reason to assume a considerably different (or higher) BPA exposure of European mothers in comparison to the USA, for which three studies on breast milk are available, and which have the main impact on the calculation of estimates for average and high BPA concentrations in breast milk.	BPA concentration in breast milk	–/+
Measurement of breast milk consumption Different methods of measurement have been used to quantify human milk consumption, all presenting an uncertainty. The uncertainties are expected to be relatively small and tend to average out when the number of observations increase.	Breast milk consumption	●
Variation between individuals The average breast milk volume is given per kg body weight and thereby takes into account the size of the baby. However, after correction for body weight there will be some residual variation between children in their average consumption per day of colostrum and breast milk. EFSA (EFSA CONTAM Panel, 2012)) has previously	Breast milk consumption	-/+

Source of uncertainty	Parameter affected	Impact of uncertainty on high exposure estimate
used 800ml as an estimate of average intake of breast milk for 3 month olds with body weight 6.1 kg, and 1200ml for high intake, suggesting that variation of up to 50% is considered possible.		
Variation of consumption in the first days of life The volumes consumed increases approximately linearly from a few grams on day 1 to around 500 grams on day 5. Considering an average consumption of 250 g over the first 5 days, and assuming an average body weight for a newborn of 3.25 kg, an average consumption rate of 75 g/kg bw/day (rounded by 5-gram steps) is obtained. Because of the transitional character in the milk production and consumption rate, this estimate is associated with an uncertainty which is jugded to be larger than ±20% but smaller than ±200%.		-/+
Variation of consumption of breast milk in months 0-6 An estimated value of 150 g/kg bw/day already used in previous EFSA opinion has been used. The energy requirement and thereby the human milk consumption per kg body weight decreases steadily from month 1 to month 6 in exclusively breastfed children. The standard breast milk volume can be an underestimate the first months and an overestimate when the child reaches 6 months.		-/+
Overall assessment – mature breast milk There is no reason to assume all the individual uncertainties to be correlated. It is expected that the unidirectional but oppositely directed uncertainties on sample contamination and population-means variability would cancel out. The other bidirectional uncertainties add up and increase the overall uncertainty in both directions. However, the upward uncertainties are countered by the uncertainty relating to the question of whether the proportion of conjugated BPA in breast milk becomes systemically available. As a result, it is expected that overall, the true exposures will lie between 20-120% of the estimate. **Overall assessment – initial breast milk (colostrum)** The above assessment is valid for mature breast milk, for which the estimate is supported by several small to medium-sized studies. For initial breast milk (colostrum), a reliable estimate could not be derived because of the discrepancies between the three available studies and the low sample sizes in some of the studies. The uncertainty for initial breast milk is further increased by the fact that milk production during the first five days is of a transitional character with changes in milk production rate and milk composition (protein and fat content). Last but not least, there is the possibility of an exposure from medical devices of mothers staying in the hospital for a few days after delivery.		– –/● (mature breast milk) – –/+ (initial breast milk/ colostrum)

3. Uncertainties in the assessment of exposure in formula-fed infants

Table 53: Evaluation of uncertainties affecting the estimation of high dietary exposure (95th percentile) of 80 ng/kg bw/day in formula-fed infants. The evaluations are approximate expert judgements and should not be interpreted as precise estimates. See Figure 1 for key to symbols

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
The assumed consumption value of ready to eat infant	consumption of	150 g/kg	●

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
formula (independently of being prepared from a powder or liquid) is based on water consumption value (WHO 2003). Variability in the consumption between individuals is expected to be low. In its assessment of BPA, WHO used as 95th percentile of consumption of infant formula in infants 174 ml/kg bw (WHO, 2010)	formula/ kg body weight	bw/day	
Method of analysis – analytical determination CV ≤ 15 %	BPA occurrence levels	0	●
Sampling: Estimates were based on data from literature and only small number of samples were available (10 for canned powder, 5 for canned liquid formula. The values ranged from <LOD/LOQ to 2.7 µg/kg for canned formula; 47 % of the samples were below LOQ.	BPA occurrence levels	95th of BPA concentration (middle bound) was 2.2 µg/kg	- /+
Sampling: Estimates were based on data from literature and only one sample of non-canned formula below the LOD/LOQ:with the middle bound of 0.9 µg/kg.	BPA occurrence levels	LB, MB and UP for average and 95th	-/+
Uncertainty due to deconjugation of conjugated BPA: In general, analytical determinations performed in food aimed at quantifying unconjugated BPA and would not allow to detect or quantify glucuronated BPA. However, according to ANSES data, the proportion of conjugated BPA in formula was not significant.	BPA occurrence levels	Total BPA is assumed	●
BPA level in water: the water used to reconstitute infant formula from powder was assumed to contain 0.2 µg/kg of BPA (middle bound of all data on non-canned water), leading to an estimated exposure of 30 ng/kg bw day from water. However, the formula could be reconstituted systematically with water containing significantly more BPA in infants living in flats where old waterpipes have been lined with epoxyresins (high exposure from water would then be 165 ng/kg bw day, If the percentage of infants in this situation was more than 5 % in one of the EU countries, this would lead to a real highest 95th percentile in the EU more than twice the estimate of 80 ng/kg bw /day. Other cases such as water warmed in a PC kettle or water filtered with a PC filter would lead to very little additional exposure (see table 24 in paragraph 4.7).	BPA occurrence levels	0.2 µg/kg (back ground level water)	● / ++
Dilution factor in powder formula preparation of 7 is assumed. This can vary depending on the instruction of preparation.	BPA occurrence levels	1/7	●
The value used in the exposure assessment covers the most common types of packaging (powder or non- canned liquid infant formula) and baby bottles not releasing any BPA, whereas other cases can occur leading to a higher exposure, PC old bottles may still in use and can yield a high exposure of 684 ng/kg bw day. If the percentage of infants in this situation was more than 5 % in one of the EU	BPA occurrence levels		● / +++

Source of uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
countries, this would lead to a real highest 95th percentile 8 times higher than the estimate of 80 ng/kg bw /day.			
Overall assessment: The main sources of uncertainty in the high level of exposure from infants formula are related to the lack of knowledge on the percentage of infants for whom more BPA is present in the water used to reconstitute infant formula, for whom old PC bottles bought before the 2011 ban would be used. This percentage could be higher than 5 % in some countries, leading to significantly higher 95Th percentile.			- / +++

4. Uncertainties in the assessment of (average and high) non-dietary exposure

Some sources of exposure were considered to be negligible or zero for toddlers and infants and were therefore not included in the exposure assessment. For both infants and toddlers, exposure from thermal paper was excluded. For infants in the first 5 days of life, exposure via toys, cosmetics and dust were also excluded. There is high confidence in these assumptions, so their uncertainty is represented by dots in the relevant tables in chapter 4.9.3 of the Opinion.

Assessment of average non-dietary exposure

The estimates of average exposure from non-dietary sources is intended to have the same level of conservativeness as the estimate of dietary exposure performed under scenario 2. Thus, in scenario 2 for dietary exposure all foods that may be canned are considered to be canned. To correspond with this, all thermal paper is assumed to contain BPA. The effect of this on the exposure estimates is considered below.

Table 54: Evaluation of variability and uncertainties affecting the assessment of average BPA exposure of dust ingestion by different age groups. See Figure 1 for key to symbols.

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
In order not to multiply too many worst case parameters (so as to achieve a realistic worst case) for this parameter an average (mean) value was used. Concentrations in dust are assessed in three European studies. The median value from the study with the middle median values was used.	C_{dust}	1.461 mg/kg	+
Method of analysis: trace analytics +/- 15 %	C_{dust}	1.461 mg/kg	●
Dust ingestion rates in general are very uncertain. They are derived from soil ingestion studies. No specific dust ingestion studies are available to date. In this assessment average values from Trudel et al., 2008 were used. For infants no data at all are available. Therefore, the value for toddlers was used, which introduces more conservatism.	q_{dust}	9 mg/d (infants, toddlers) 5 mg/d (adults)	- - (infants) ● (toddlers) ● (adults)
This absorption factor is a placeholder for an	$r_{absorbtion}$	1 (fraction)	--

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
absorption factor that takes into account that (1) different particle sizes have different absorption fractions (2) a fraction of the dust will be inhaled and another fraction will be ingested. Suitable absorption fractions will be derived for the whole opinion. For the time being the absorption fraction of 1 represents the worst case (equals external exposure).			
For infant body weight a value for 1-3 months old infants was used (EFSA Scientific Committee, 2012). For toddlers also a value on the conservative side was used. Adult female body weights vary: about 70 % are below the EFSA default value of 70 kg. (EFSA Scientific Committee, 2012)	body weight	5 kg (infants) (12 kg) (toddlers) 70 kg (adults)	- (infants - (toddlers) -/+ (adults)
Overall assessment. Since toddler data on dust ingestion were also used for infants, who normally will have less exposure to dust, the true value for exposure for infants may be below the calculated values. The general uncertainty with dust ingestion rates may affect the exposure in both directions. For the time being a large uncertainty is also associated to the absorption fraction of 1, which represents a worst case and will be refined for the full opinion.			- - / + (infants) - / + (toddlers, adults)

Table 55: Evaluation of variability and uncertainties affecting the assessment of average BPA exposure from toys in infants and toddlers. See Figure 1 for key to symbols

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
This average amount of leaching from toys was derived from one migration study with toys bought in Sweden. Toys will vary largely, so this value may not be representative. However, toys made of polycarbonate are not frequent on the market, so the true average value is likely to be closer to 0.	q_{toy}	141 ng	- -
Method of analysis: trace analytics +/- 15 %	q_{toy}	141 ng	●
The time fraction that the toy is sucked per day will be highest for continuous sucking (1) and lowest for not sucking. Average sucking times have been used that were observed in children of different age classes.	f_{time}	0.012 day^{-1} (infants) 0.001 day^{-1} (toddlers)	●
The fraction of surface in contact with the mouth zone of the baby will be variable depending on the toy. Many different sizes of toys are available. Here, as an example we assessed a rattle: For a rattle approximately 0.5 of the rattle will be in contact with saliva. It is assumed that all of that saliva is subsequently ingested. This may not be true, not ingested saliva may reduce the effective surface up to 5 times.	$f_{surface}$	0.5	- - / +
For infant body weight a value for 1-3 months old infants was used (EFSA Scientific Committee, 2012). For toddlers also a value on the conservative side.	body weight	5 kg (infants) 12 kg (toddlers)	- (infants) - (toddlers)

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
Overall assessment. Because of the small fraction of PC toys on the market this value may be an overestimation for average exposure.			- --/+

5869

5870 **Table 56:** Evaluation of variability and uncertainties affecting the assessment of average BPA
5871 exposure from air inhalation all age groups See Figure 1 for key to symbols.

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
Concentrations of BPA in indoor air are only available for France in a limited study. It is not clear whether levels of BPA in indoor air will vary between countries in Europe. For this assessment it was assumed that people spend 100 % of their time indoors. Since outdoor levels of BPA seem to be slightly smaller, this may result in a slight overestimation (not much, because on average people in industrialized countries spend 90 % of their time indoors). However, in one study for Greece levels in outdoor air were as high as 6 ng/m^3. People in Greece, however, may spend more time outdoors than people in Northern Europe. Taking into account the high levels in outdoor air in Greece (which were not used in the assessment), there may be an underestimation for Greece and other Southern countries.	C_{air}	1.0 ng/m^3	- / ++
Method of analysis and sampling together can affect the measurement so that the variation may be +/- 100 %	C_{air}	1.0 ng/m^3	-/+
Inhalation rates vary with the activity profile. Therefore, the highest uncertainty is associated with the mix of activities during the day. Here, average values associated with an activity pattern proposed by Trudel et al, 2008 were used for the different consumer groups.	q_{air}	12 m^3/d (infants) 16.3 m^3/d (toddlers) 50.4 m^3/d (adults)	-/+ (infants) -/+ (toddlers) -/+ (adults)
The inhalation absorption fraction was assumed to be 1. Since BPA will mainly be inhaled with small particles that can also be exhaled, it is unclear how much BPA will be release during the residence time in the lung.	$r_{absorption}$	1 (fraction)	- -
For infant body weight a value for 1-3 months old infants was used (EFSA Scientific Committee, 2012). For toddlers also a value on the conservative side. Adult female body weights vary: about 70 % are below the EFSA default value of 70 kg. (EFSA Scientific Committee, 2012)	body weight	5 kg (infants) 12 kg (toddlers) 70 kg (adults)	- (infants) - (toddlers) -/+
Overall assessment. The activity profile will be very different for different			- - / ++

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
subpopulations and different cultures. Also levels in indoor air are only available for one country in Europe and may be different in other countries.			(infants) - / ++ (toddlers, adults)

5872

5873 The estimates of average exposure from non-dietary sources is intended to have the same level of
5874 conservativeness as the estimate of dietary exposure performed under scenario 2. Therefore, all
5875 thermal paper is estimated to contain BPA.

5876 **Table 57:** Evaluation of variability and uncertainties affecting the assessment of average level
5877 dermal exposure to BPA from thermal paper for adults. See Figure A for key to symbols.

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
The amount left on the fingers after handling thermal papers depends on the wetness and greasiness of the touching skin. If the paper is handled very shortly, not pressed and the fingers are dry it can be assumed that no BPA is transferred at all. The highest amount transferred was observed for wet fingers (Lassen, 2011). The average value presumably is on the conservative side, since it was derived by pressing hardly a thermal paper during 10 s (with dry fingers).	q_{finger}	1.4 µg	-
Method of analysis – analytical determination CV ≤ 15%	q_{finger}	1.4 µg	●
Maximum is 10. Normally people grasp paper with thumb and 1 or 2 finger tips. More contact can occur for those who fold their tickets, but the two little fingers are not involved. Based on the limited data available, 3 fingers per handling event is thought to be a average case.	n_{finger}	3	- / +
This value is based on the number of credit card receipts/person/year in Denmark.	$q_{handling}$	1 per day	- - / ++
The dermal absorption fraction can range from 0 to 1. In order not to multiply too many worst-case parameters for this parameter an average value was used, which was determined for uptake into the skin.	$r_{absorbtion}$	0.3 (fraction)	-/+
Not all thermal papers contain BPA. Presumably today around 80 % thermal papers contain BPA and the percentage may be declining due to public debate.	Occurrence of BPA in thermal paper	100 % (Upper bound)	-
Adult female body weights vary: about 70 % are below the EFSA default value of 70 kg (EFSA Scientific Committee, 2012).	bw	70 kg	+
Overall assessment For two parameters data are lacking completely, which is why the assessment is highly uncertain. It is not clear, in which direction the true value may lie.			- - -/++

5878

5879 **Table 58:** Evaluation of variability and uncertainties affecting the assessment of average level of
5880 BPA exposure from cosmetics from all age groups. See Figure 1 for key to symbols

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
Only one study on 6 products is available to date. This data is not representative. One product was chosen for an exemplary assessment: a face cream as a proxy for body lotion. The range of possible concentrations of BPA therefore is not really known. The highest boundary may be 10 ppm, since this is an acceptable level for impurities in a product.	$C_{cosmetics}$	0.031 µg/g	--/++
Method of analysis: trace analytics +/- 15 %	$C_{cosmetics}$	0.031 µg/g	●
Application rates of body lotion have been assessed in a large study on the European level for adults. Data is considered as robust. For infants and children, however, use data had to be extrapolated from adult data.	$q_{cosmetics}$	0.77 g/d (infants) 1.1 g/d toddlers) 4.6 g/d (adults)	- /+ (infants, toddlers) ● (adults)
It was assumed that only one cosmetic was used (a worst case body lotion). In reality, some individuals using body lotion will also use other cosmetics leading to some additional BPA exposure.	$q_{cosmetics}$		+
An absorption rate for BPA in cosmetics is not available. Therefore, an absorption rate from BPA in ethanol was used as a proxy. The amount absorbed rises with falling concentration. For a concentration of 1 mg/mL absorption rates up to 100 % have been reported by Biedermann, 2010. The absorption rate of 0.6 refers to a concentration of 10 mg/mL.	$r_{absorbtion}$	0.6	-/+
Absorption rates determined in the in vivo study by Biedermann et al., 2010 have been determined for absorption into the skin, and not into the blood. Uptake into the blood stream may be considerably lower	$r_{absorption}$	0.6	- -
For infant body weight a value for 1-3 months old infants was used. For toddlers also a value on the conservative side. Adult female body weights vary: about 70 % are below the EFSA default value of 70 kg. (EFSA Scientific Committee, 2012)	Body weight	5 kg (infants) 12 kg (toddlers) 70 kg (adults)	-
Overall assessment. Overall the uncertainties in absorbtion rate and behavioural data may level out, but large uncertainties remain.			--/++

Assessment of high non-dietary exposure

Table 59: Evaluation of variability and uncertainties affecting the assessment of exposure from thermal paper for women of child-bearing age. Note that evaluations in columns 4 and 5 of the table are approximate expert judgements and should not be interpreted as precise estimates. See Figure 1 for key to symbols.

Source of variability or uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
The approximation of a 95th percentile was performed by combining three average parameter values (q_{finger}, bodyweight and $r_{absorption}$) with approximate 75th percentile values for two other parameters and an upper bound for another (BPA occurrence). It is uncertain whether this approach leads to the true 95th percentile. The more parameters introduced as the 75th percentile, the higher will be the percentile. For two 75th percentile and two average parameters the 95th percentile is more likely to be slightly underestimated than overestimated.	(all)		●/+
The amount left on the fingers after handling thermal papers depends on the wetness and greasiness of the touching skin. If the paper is handled very shortly, not pressed and the fingers are dry it can be assumed that no BPA is transferred from the paper to the fingers at all. The highest amount of 30 µg transferred was observed for wet fingers (Lassen, 2011). In order not to multiply too many worst-case parameters (so as to achieve a realistic worst case) for this parameter an average value was used. However, this average presumably is on the conservative side, since it was derived by pressing hard on a thermal paper for 10 s (with dry fingers), which is not always done when handling receipts.	q_{finger} quantity on the finger	1.4 µg (Average value)	- -/++
Method of analysis – analytical determination CV ≤ 15%	q_{finger}	1.4 µg	●
Maximum is 10. Normally people grasp paper with thumb and 1 or 2 finger tips. More contact can occur for those who fold their tickets, but the two little fingers are not involved. Based on the limited data available, 6 fingers per handling event is thought to be an approximate 75th percentile and suitable for making an estimate of high exposure when combined with the other input variables.	n_{finger} number of fingers	6 (Approx. 75th percentile)	●
The used value was determined as a worst case by Lassen et al, 2011 from a use study with shopping receipts (3.6) and added safety value for unknown papers, e.g. bus tickets. The frequency of handling may occasionally and for special people be much higher, but presumably not more than 10 events (7 shopping, 2 bus, 1 canteen ticket) on a regular basis.	$f_{handling}$ frequency of handling	4.6 / day (Approx. 75th percentile)	-/+
The dermal absorption fraction can range from 0 to 1. In order not to multiply too many worst-case parameters for this parameter an average value was used, which was determined for uptake into the skin.	$r_{absorbtion}$	0.3 (fraction) (Average value)	-/+
Adult female body weights vary: about 70 % are below the EFSA default value of 70 kg.	bw body weight	70 kg (Average value)	-/+
Not all thermal papers contain BPA. Presumably today around 80 % thermal papers contain BPA and the percentage may be declining due to public debate.	Occurrence of BPA in thermal paper	100 % (Upper bound)	-
Overall assessment. The largest uncertainty arises from the variability of people's skin wetness and greasiness, and behavioural factors. From the combination of a conservative average for the amount on the fingers and the absorption fraction and approximate 75th			--/++

Source of variability or uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
percentiles for the both use parameters, a 95th percentile was targeted. In order to roughly check our assumptions to achieve a P95, we performed a Monte Carlo simulation by applying the full parameter range given above. In this Monte Carlo simulation the 95th percentile was estimated to be about 400 ng/kg bw /d in comparison to 163 ng/kg bw /d for the deterministic evaluation, meaning that there is the possibility of underestimating the 95th percentile. However, the assumption that the controlled experiment mimics worst-case touching of thermal paper, may have led to overestimation. Overall, we estimate that the true 95th percentile may be between 2-5 times lower and 2-5 times higher than the estimated 95th percentile.			

5888

5889 **Table 60:** Evaluation of variability and uncertainties affecting the assessment of exposure from dust
5890 ingestion in infants and toddlers. See Figure A for key to symbols

Source of variability or uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
The approximation of the 95th percentile was performed by combining two average parameter values ($C_{dust,}$ bodyweight) with higher percentile values for two other parameters. Supposed that these parameters would be 75th percentiles the approach would likely lead to a 95th percentile.			●/+
Concentrations in dust are assessed in three European studies. Here the median value from the study with the middle median values was used.	C_{dust}	1.461 mg/kg (Average value)	-/+
Method of analysis: trace analytics +/- 15 %	C_{dust}	1.461 mg/kg	●
Dust ingestion rates are very uncertain. They are derived from soil ingestion studies. No specific dust ingestion studies are available to date. It is assumed that the true value for dust ingestion is lower, because pika behavior contributes large amounts of data for toddlers. In this assessment the highest value used in a deterministic exposure assessment by Trudel et al., 2008 was used. For infants no data at all are available. Therefore, the value for toddlers was used, which introduces more conservatism.	q_{dust}	106 mg/d (Maximum derived value)	- - - / - - (infants) - - / + (toddlers)
It is assumed that 100 % dust is absorbed. This is clearly an upper bound and not a 75th percentile. Different particle sizes will be taken up with different effectiveness and by different organs (inhalation vs. ingestion). Since it is not clear, which systemic uptake rates will be used to derive the internal dose, we use the upper bound.	$r_{absorption}$	1 (fraction) (Upper bound)	- -
For infant body weight a value for 1–3 months old infants was used (EFSA Scientific Committee, 2012). For toddlers also a value on the conservative side.	body weight	5 kg (infant) 12 kg (toddler)	-
Overall assessment. Because of the very uncertain dust ingestion rates, for which high exposure values were used, and because of the upper bound used for dust absorption the true value for the 95th percentile may be below the			- - - / +

Source of variability or uncertainty (high scenario)	Parameter affected	Value used in assessment	Impact on high exposure estimate
calculated values. However, since dust concentrations were shown to be higher e.g. for France both uncertainties may level out.			

5891

5892 **Table 61:** Evaluation of variability and uncertainties affecting the assessment of high exposure from
5893 air inhalation by all consumer groups. See Figure 1 for key to symbols

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
The approximation of the 95th percentile was performed by combining two average parameters value ($C_{air,}$ bodyweight) with higher percentile values for two other parameters. Supposed that these parameters would be 75th percentiles the approach would likely lead to a 95th percentile.			-/+
Concentrations of BPA in indoor air are only available for France in a limited study. It is not clear whether levels of BPA in indoor air will vary further in Europe. For this assessment it was assumed that people spend 100 % of their time indoors. Since outdoor levels of BPA seem to be slightly smaller, this may result in a slight overestimation (not much, because on average people in industrialized countries spend 90 % of their time indoors). However, in one study for Greece levels in outdoor air were as high as 6 ng/m^3. People in Greece, however, may spend more time outdoors than people in Northern Europe. Taking into account the high levels in outdoor air in Greece (which were not used in the assessment), there may be an underestimation for Greece and other Southern countries.	C_{air}	1.0 ng/m^3 (Average value)	- / +++
Method of analysis: trace analytics +/- 15 %	C_{air}	1.0 ng/m^3	-/+
Inhalation rates vary with the activity profile. Therefore, the highest uncertainty is associated with the mix of activities during the day. Here, high scenario values associated with an activity pattern proposed by Trudel et al, 2008 were used for the different consumer groups, which rather represent higher percentles than the 75th percentile.	q_{air}	28.8 m^3/d (infants) 40.8 m^3/d (toddlers) 91.2 m^3/d (adults) (high estimates)	- - (infants) - - (toddlers) - - (adults)
The inhalation absorption fraction was assumed to be 1. Since BPA will mainly be inhaled with small particles that can also be exhaled, it is unclear how much BPA will be released during the residence time in the lung.	$r_{absorption}$	1 (fraction) (Upper bound)	- -
For infant body weight a value for 1-3 months old infants (EFSA Scientific Committee, 2012) was used. For toddlers also a value on the conservative side.	body weight	5 kg (infants) 12 kg (toddlers)	-

Source of variability or uncertainty (average scenario)	Parameter affected	Value used in assessment	Impact on average exposure estimate
		70 kg (adults)	
Overall assessment. It was not possible to use 75th percentiles in the assessment, rather high exposure values were used that will result in a higher exposure estimate than the 95th percentile. This overestimation will level out some of the possible underestimation due to the uncertain air concentrations.			- - - /+ + (infants) - - - /+ + (toddlers, adults)

5. Evaluation of uncertainties affecting the assessment of high total exposure based on biomonitoring data on total BPA concentration in urine

In this assessment, data for 3–5 year old children were taken as a surrogate, as no biomonitoring data are available for 1–3 year old toddlers. For women of child-bearing age, data for mothers, pregnant and parturient women were used. The evaluations are approximate expert judgements and should not be interpreted as precise estimates.

Table 62: Evaluation of uncertainties affecting the assessment of high total exposure in Women (W) of child-bearing age, Toddlers (T), and Infants (I) based on biomonitoring data on total BPA concentration in urine. See Figure 1 for key to symbols.

Source of uncertainty	Parameter affected	Value used in assessment	Impact on high exposure estimate
Analytical uncertainty for urinary BPA concentrations above LOD. Recovery: Not a problem since all studies use isotope-dilution mass spectrometry with recovery correction. Repeatability: Intra- and inter-day CV <21 %. Accuracy: < ±20 % (intra- and interday). Taken together, the overall analytical uncertainty is regarded to be within ±20 %.	BPA concentration in urine C_{BPA} (µg/l)	Range of the 95th percentiles W: 5–12 µg/l T: 23 µg/l I: 2.2–3.4 µg/l	W: ● T: ● I: ●
Contamination of urine samples. Most studies report only total BPA concentration in urine but only a few studies additionally report the concentration of unconjugated BPA. It can however be expected that contamination of urine samples during collection and storage is generally under control. A small proportion of total BPA might be from contamination which would then result in a slight overestimation, so tends to be conservative.	BPA concentration in urine C_{BPA} (µg/l)	Range of 95th percentiles W: 5–12 µg/l T: 23 µg/l I: 2.2–3.4 µg/l	W: ● T: ● I: ●
Sampling uncertainty Number of subjects per study is 60–164 (Women), 30–137 (Toddlers), and 12–46 (Infants). The relatively low number of subjects in some studies may result in a sampling bias. Moreover, only a few European studies (GerES IV, INMA) can be assumed to be representative for a specific age class and geographical region. The database contains 10 studies for Women from 10 different European countries (but only 6 have reported a 95th percentile [P95]), two European studies for	BPA concentration in urine C_{BPA} (µg/l)	Range of the 95th percentiles W: 5–12 µg/l T: 23 µg/l I: 2.2–3.4 µg/l	W: ● / + T: ● / + I: ● / ++

Source of uncertainty	Parameter affected	Value used in assessment	Impact on high exposure estimate
"Toddlers" (but only one has reported a 95th percentile), and one European study for Infants. Biomonitoring studies may, therefore, not have captured high levels of exposure that may occur in specific geographic areas or specific population groups.			
Distribution uncertainty Most studies provide the 95th percentile (P95) of the distribution of the BPA concentration in individual urinary samples. The P95 is used to obtain estimates for high BPA exposures. Many studies report data for spot urine samples, for which the P95 relates to the 95 % probability that a single, randomly collected sample from a randomly selected subject has an urinary BPA concentration not exceeding the P95. This is important as urinary BPA concentrations of repeated urine collections from individuals may vary by up to two orders of magnitude. Some studies exist which indicate that the total variance can be broken down into 70 % within-day variability, 21 % between-day variability, and 9 % between person variability. Thus, taking the P95 of the reported values will over-estimate the 95th percentile of long-term average values (true value will tend to be lower).	BPA concentration in urine C_{BPA} (µg/l)	Range of the 95th percentiles W: 5–12 µg/l T: 23 µg/l I: 2.2–3.4 µg/l	W: – / ● T: – / ● I: – / ●
Uncertainty in specific urinary output rate The specific urinary output rate (ml/kg bw/day) is the urinary output rate (ml/day) divided by body weight (kg) For the urinary output rate, generic values were generally used to estimate the average urinary output rate per population subgroup. These generic values were derived by linear interpolation from urinary output *vs.* age relationships taken from literature. Some studies, however, collected 24-h urine samples and provided individual data for daily urinary output. The average of these experimental data can be compared with generic values to obtain a measure of possible bias. For example, the German ESB study (Koch et al., 2012) analyzed historical 24-h urine samples of 20-30 years old male and female students and reported an increase in urinary output rate from 1 500 ml/day in 1995 to 2 000 ml/day in 2009. The generic value for adults (averaged over males and females) is 1 400 ml/day. In this special case, the deviation of the average experimental values from the generic value is+ 7 % and +42 %. For body weight, also generic values were general used to estimate the average body weight per population subgroup. These generic values were derived by linear interpolation from body *vs.* age relationships taken from literature. Some studies, however, measured the individual body weights. The average of these experimental data can be compared with generic values to obtain a measure of possible bias. The available data suggest the uncertainty to be within ±20 %. Taken both parameters together, the range of values for the specific urinary output rate for Women is 17–27 ml/kg bw/day. For studies, for which the upper value was taken, the true value could be lower by a factor of	Specific urinary output rate spec. $\dot{V}_{urine}$ (ml/kg bw/ day)	Range of values W: 17–27 ml/kg bw/day T: 30 ml/kg bw/day I: 48 ml/kg bw/day	W: – / + T: – / + I: – / +

Source of uncertainty	Parameter affected	Value used in assessment	Impact on high exposure estimate
1.6. For studies, for which the lower value was taken, the true value could be higher by a factor of 1.6.			
Uncertainty about time trends in exposure Urinary samples were collected in different time periods, i.e. in 2004–2012 (Women), 2003–2006 ("Toddlers"), and 2008 (Infants). There could be changes in exposure over the years in exposure. A retrospective study using historical samples of students from the German Environmental Specimen Bank (ESB) indicated a gradual decline in the 95th percentiles from 1995 to 2001/2003, which, however, did not continue from 2003 on and seemed to be reversed to some extent from 2003 on (Koch et al., 2012). The results of US NHANES suggests a slight decline in the 95th percentiles of 257 ng/kg bw/day to 183 ng/kg bw/day for adults over the periods from 2003–2004 to 2009–2010.	Daily BPA exposure $\dot{m}_{BPA}$ (ng/kw bw/day)	Range of 95th percentiles W: 85–234 ng/kg bw/day T: 676 ng/kg bw/day I: 164 ng/kg bw/day	W: ● T: ● I: ●
Uncertainty in extrapolating from children to toddlers There is no indication that the exposure of 3–5 year old children, which was taken as a surrogate for the exposure of 1–3 year old toddlers, is substantially different from that of toddlers. The first line of evidence from the biomonitoring study GerES IV is that 3–5 year old children have a higher exposure than 6–8 year old children (Becker et al., 2009). Other biomonitoring studies on 4 year old children (INMA) and older children (Duisburg BCS, Liege HBM, DEMOCOPHES) provide additional support for this age dependency. The second line of evidence is that the modelling approach did not indicate substantial differences in the high total exposure between toddlers and the age class of 3–10 year old children. This provides an indirect indication for similar exposures between the toddlers and the surrogate group of 3–5 year old children, because this group can be expected to be in the upper tail of the modelled exposure distribution of the 3–10 year old children.	Daily BPA exposure $\dot{m}_{BPA}$ (ng/kw bw/day)	Range of 95th percentiles T: 676 ng/kg bw/day	T: ●
Overall assessment: The main sources of uncertainty in the estimation of high total exposure based on biomonitoring data is the sampling uncertainty due to limitations in the representativity of the available information on total BPA concentration in urine, the distribution uncertainty in the 95th percentile, and the uncertainty in the specific urinary output rate. The latter uncertainty is two-sided. The distribution uncertainty in the 95th percentile is one-sided so that the true value for high total exposure is likely to be lower than the estimate. The sampling uncertainty is also one-sided but orientated in the opposite direction so that the true value for high total exposure is likely to be higher than the estimate. Overall, the two uncertainties in opposite directions may cancel out to some extent, but the outcome could be positive or negative depending on their true magnitudes. Hence the overall assessment is that the true value could be either lower or higher than the estimate. The estimates for high total exposure are 234 ng/kg bw/day for women (W) of child-bearing age, 676 ng/kg bw/day for "Toddlers" (T), and 164 ng/kg bw/day for Infants (I). As a control check for the high total exposure estimate for Women (which was derived from the highest 95th percentile of 6 studies), a parametric statistic was calculated from the $\log_{10}$-transformed individual P95 values and, based on that, the value 10^(average+1.64×sigma) was then used as a proxy for a hypothetical European country with the highest P95. This control check yielded a value of 296 µg/l, which is 26 % higher than the chosen value of 234 µg/l for			W: – / + T: – / + I: – / ++

Source of uncertainty	Parameter affected	Value used in assessment	Impact on high exposure estimate
Women. No such control checking is possible for "Toddlers" and Infants. However, compared to the infant study (n=46), the study for "Toddlers" is a large-sized (n=137) representative study (GerES IV), which results in different uncertainty ratings.			

5904

5905 **APPENDIX IX: LITERATURE QUALITY TABLES**

5906 **Table 63:** Literature quality table – occurrence in food

5907

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Determination of bisphenol A in US infant formulas: updated methods and concentrations	Ackerman, L. K., Noonan, G. O., Heiserman, W. M., Roach, J. A, Limm, W. and Begley, T. H.	Journal of Agricultural and Food Chemistry	2010	58:4, 2307-2313	10.1021/jf90395 9u	Not considered	Not considered	Not considered	Excluded - samples from USA
Analytical methods for the determination of bisphenol A in food	Ballesteros-Gomez, A. Rubio, S. and Perez-Bendito, D.	Journal of Chromatography A	2009	1216:3, 449-460	10.1016/j.chroma.2008.06.037	Not considered	Not considered	Not considered	Excluded - analytical method review paper - no relevant data for calculation of exposure from food
Determination of bisphenol A in wine by sol-gel immunoaffinity chromatography, HPLC and fluorescence detection	Brenn-Struckhofova, Z. and Cichna-Markl, M.	Food Additives and Contaminants	2006	23:11, 1227–1235	10.1080/02652030600654382	46 white and 13 red wine samples of which 10 were taken directly from the wine vats, 21 had been filled	Austria	Filtered samples were cleaned-up by sol-gel immunoaffinity chromatography, using polyclonal BPA rabbit antibodies. Analysis was carried out by HPLC-FLD LOD (S:N=3) = 0.1 µg/L	Included - although the samples were obtained pre-2006 the paper described concentration data for wine which is not available elsewhere

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
						into glass bottles and 28 were purchased from supermarkets (packaged in glass bottles (n=17) or tetra-brik (n=11))		LOQ (S:N=6) = 0.2 µg/L Recovery = 74 - 81 % (average of three spiking levels: 0.4, 0.8 and 1.2 µg/L) Repeatability = not given Calibration = external standards 0.3 to 100 µg/L in mobile phase No information on prevention of contamination or blanks	
Stir bar sorptive extraction coupled to gas chromatography-mass spectrometry for the determination of bisphenols in canned beverages and filling liquids of canned vegetables	Cacho, J. I., Campillo, N., Viñas, P. and Hernández-Córdoba, M.	Journal of Chromatography A	2012	1247, 146-153	10.1016/j.chroma.2012.05.064	Beverages and filling liquids of vegetables (canned) 10 canned beverages, and 10 filling liquids of vegetables	Samples purchased in Spain	Following degassing and dilution with water the BPA was derivatised in situ with acetic anhydride, extracted using stir bar sorptive extraction, and analysed by thermal desorption GC-MS LOD = 2.5 ng/L in solution (3x st dev of the procedural blank) equates to 12.5 ng/L in sample (sample was diluted x5 with water prior to analysis) LOQ = 8.4 ng/L (10x st dev of the procedural blank) equates to 42 ng/L in sample (sample was diluted x5 with water prior to analysis) Recovery = 86-122 % at 0.1	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								µg/L and 97-105 % at 1 µg/L Repeatability = 1.9 % intraday and 3.1 % interday for water spiked with BPA at 0.5 µg/L. <10 % in matrix (recovery study) Calibration = external standards 0.02 to 2.5 µg/L in water Reported repeatable trace background levels of BPA of 10 ng/L - background concentration was subtracted from reported values	
Levels of bisphenol A in canned liquid infant formula products in Canada and dietary intake estimates	Cao, X. L., Dufresne, G., Belisle, S., Clement, G., Falicki, M., Beraldin, F. and Rulibikiye, A.	Journal of Agricultural and Food Chemistry	2008	56, 7919-7924	10.1021/jf80087 12	21 liquid formula products 17 milk based and 4 soy based	Samples purchased in Canada (5 originated in Switzerland - 1 soya and 4 milk based)	Following precipitation of the protein the BPA was extracted from the sample using SPE. Analysis was carried out by GC-MS after derivatisation with acetic anhydride LOD (S:N=3) = better than 0.1 µg/kg (instrument detection limit) LOQ = 0.5 µg/kg (equivalent to lowest calibration standard) Recovery = 85-94 % for five to eight replicates of infant formula spiked at 2.5, 8.0	Included - although the samples were obtained in Canada some were manufactured in Switzerland

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								and 20 µg/kg (rsd of recovery samples = 2.7-3.9 %) Repeatability = 5.0 % at 0.5 µg/kg and 2.8 % at 10.4 µg/kg External calibration 0.01 to 0.48 µg/L equivalent to 0.5-24 µg/kg Method blank prepared but its use is not described further	
Migration of bisphenol A from can coatings to liquid infant formula during storage at room temperature	Cao, X. L., Corriveau, J., Popovic, S	Journal of Food Protection	2009	72:12, 2571-2574	Not given	Not considered	Not considered	Not considered	Excluded - samples from Canada
Levels of bisphenol A in canned soft drink products in Canadian markets	Cao, X. L., Corriveau, J., Popovic, S	Journal of Agricultural and Food Chemistry	2009	57, 1307-1311	10.1021/jf80321 3g	Not considered	Not considered	Not considered	Excluded - samples from Canada
Bisphenol A in baby food products in glass jars with metal lids from Canadian markets	Cao, X. L., Corriveau, J., Popovic, S., Clement, G., Beraldin,	Journal of Agricultural and Food Chemistry	2009	57:12, 5345-5351	10.1021/jf900688	Not considered	Not considered	Not considered	Excluded - samples from Canada

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	F. and Dufresne, G.								
Bisphenol A in canned food products from Canadian markets	Cao, X. L., Corriveau, J., Popovic, S	Journal of Food Protection	2010	73, 1085-1089	Not given	Not considered	Not considered	Not considered	Excluded - samples from Canada
Sources of low concentrations of bisphenol A in canned beverage products	Cao, X. L., Corriveau, J., Popovic, S	Journal of Food Protection	2010	73, 1548-1551	Not given	Not considered	Not considered	Not considered	Excluded - samples from Canada
Concentrations of bisphenol A in the composite food samples from the 2008 Canadian total diet study in Quebec City and dietary intake estimates	Cao, X. L., Perez-Locas, C., Dufresne, G., Clement, G., Popovic, S., Beraldin, F., Dabeka, R. W. and Feeley, M.	Food Additives and Contaminants Part A	2011	28:6, 791-798	10.1080/19440049.2010.513015	Not considered	Not considered	Not considered	Excluded - samples from Canada
The contribution of diet to total bisphenol A body burden in humans: Results of a 48hour	Christensen, K. L., Lorber, M., Koslitz, S., Bruning,	Environment International	2012	50, 7-14	10.1016/j.envint.2012.09.002	Not considered	Not considered	Not considered	Excluded - biomonitoring data only - no relevant data for calculation of exposure from food

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
fasting study	T. and Koch, H. M.								
Simultaneous determination of bisphenol A and bisphenol B in beverages and powdered infant formula by dispersive liquid–liquid micro-extraction and heartcutting multidimensional gas chromatography-mass spectrometry	Cunha, S. C., Almeida, C., Mendes, E. and Fernandes, J. O.	Food Additives and Contaminants	2011	28:4, 513-526	10.1080/19440049.2010.551325	22 canned soft drinks, 8 canned beers, 7 canned infant formula (infant formula was reconstituted with water following on-pack instructions prior to analysis)	Samples purchased in Portugal (randomly purchased in local supermarkets)	BPA was extracted from the samples using disperse liquid-liquid micro-extraction with simultaneous derivatisation with acetic anhydride. Analysis was carried out by two-dimensional GC-MS LOD = 0.005 µg/L in canned beverages and 0.06 µg/L in reconstituted powdered infant formula (3x S:N) LOQ = 0.01 µg/L in canned beverages and 0.20 µg/L in reconstituted powdered infant formula (10x S:N) Recovery = 83 % for beverage spiked at 0.05 µg/L, 93 % for beverage spiked at 0.2 µg/L; 114 % for powdered infant formula spiked at 0.05 µg/L, 93 % for powdered infant formula spiked at 0.2 µg/L (six replicates of each) Repeatability = 8 % for beverage spiked at 0.05 µg/L, 8 % for beverage	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								spiked at 0.2 µg/L; 15 % for powdered infant formula spiked at 0.05 µg/L, 7 % for powdered infant formula spiked at 0.2 µg/L (six spiked replicates) Calibration = Matrix matched - 0.02-10 µg/L for beverages and 0.5-10 µg/L for infant formula BPA free bottled beverages and milk samples used as method blanks to check for background contamination	
Determination of bisphenol A and bisphenol B in canned seafood combining QuEChERS extraction with dispersive liquid-liquid microextraction followed by gas chromatography-mass spectrometry	Cunha, S. C., Cunha, C., Ferreira, A. R. and Fernandes, J. O.	Analytical and Bioanalytical Chemistry	2012	404, 2453-2463	10.1007/s00216-012-6389-5	47 canned seafood samples (23 canned tunas, 10 canned sardines, 3 canned mackerels, 3 canned squid, 3 canned octopuses, 2 canned mussels, 1 canned eel, 1 canned anchovy, 1	Samples purchased in Portugal (randomly purchased in local supermarkets)	BPA was extracted from the fish samples using acetonitrile with QuEChERS and DLLME clean-up. The extracted BPA was derivatised using acetic anhydride and the derivative analysed by GC-MS LOD = 0.2 µg/kg in the foodstuff (3x S:N) LOQ = 1 µg/kg in the foodstuff (corresponding to the lowest calibration standard) Recovery = 68-104 % for tuna, 71-104 % for sardines in sauce (spike levels = 1, 5	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
						canned codfish)		and 20 µg/kg) Repeatability = 8-21 % for tuna, 11-19 % for sardines in sauce (spike levels = 1, 5 and 20 µg/kg) Calibration = matrix matched standards in the range 1 to 150 µg/kg Muffled glassware was used - no plasticware - to minimise contamination. Method blanks were prepared periodically to check for background contamination	
The investigation of bisphenol A presence in canned tuna fish using high-performance-liquid chromatography method	Er, B. and Sarimehmetoğlu, B.	Journal of Animal and Veterinary Advances	2011	10, 2859-2862	Not given	160 canned tuna fish samples	Samples purchased in Turkey	Solvent extracted samples were cleaned-up by SPE. Analysis was carried out by HPLC-FLD LOD = 1.96 µg/L in solution LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = Not specified No information on prevention of contamination or blanks	Excluded - method performance criteria not defined and so criteria could not be confirmed to have been met
Simultaneous determination of bisphenol A, octylphenol, and	Ferrer, E., Santoni, E., Vittori, S., Font,	Food Chemistry	2011	126, 360-367	10.1016/j.foodchem.2	2 samples of powdered skimmed milk and 8	Spain and Italy (5 samples purchased	BPA was extracted using pressurised liquid extraction with a C18 dispersant. Analysis was carried out by	Excluded - reported concentrations are described as comparable to others

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
nonylphenol by pressurised liquid extraction and liquid chromatography–tandem mass spectrometry in powdered milk and infant formulas	G., Manes, J. and Sagratini, G.				010.1 0.098	powdered infant formula	from each)	LC-MS/MS LOD (S/N=3) = 5 µg/kg LOQ (S/N=10) = 16 µg/kg Recovery = 89-92 % for five replicates of infant formula and powdered skimmed milk spiked at 50 µg/kg and 500 µg/kg Repeatability = 12 to 14 % or five replicates of infant formula and powdered skimmed milk spiked at 50 µg/kg and 500 µg/kg Calibration = Matrix matched - concentration range 3 orders of LOQ No measures against contamination reported	in the literature however the values given in this paper are several orders of magnitude greater than these supposedly comparable values
Determination of bisphenol A in foods as 2,2-bis-(4-(isopropoxycarbonyloxy)phenyl)propane by gas chromatography/ mass spectrometry	Feshin, D. B., Fimushkin, P. V., Brodskii, E. S., Shelepchikov, A. A., Mir-Kadyrova, E. Y. and Kalinkevich, G. A.	Journal of Analytical Chemistry	2012	67:5, 460-466	Not given	One sample of each of: an energetic beverage, infant meat puree, infant formula feed, canned meat and canned vegetables	Samples purchased in Russia	Aqueous samples derivatised directly in the matrix with isopropyl chloroformate, other foods solvent extracts were derivatised following sample clean-up by SPE for fat containing samples. Analysis was carried out by GC-MS LOD < 0.05 µg/kg for energetic beverage, < 0.1 µg/kg for infant meat puree,	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								infant formula feed, canned meat and canned vegetables LOQ = not given Recovery = 103 % when 300 ng added - average of triplicate results, 104 % when 600 ng added (BPA spiked into apple juice mass of apple juice not given) Repeatability = 0.005 % given in paper - actually 3.8 % using data given (triplicate extracts of a meat puree sample at 1.33 µg/kg) Calibration = 5 to 1200 ng (in 20 mL water) A method blank was prepared in each batch to check for contamination	
Analysis of bisphenols in soft drinks by on-line solid phase extraction fast liquid chromatography-tandem mass spectrometry	Gallart-Ayala, H., Moyano, E. and Galceran, M. T.	Analytica Chimica Acta	2011	683, 227-233	10.1016/j.aca.2010.10.034	Eleven beverages (cola, soda, beer, tea and energy drinks)	Samples purchased in Spain	Beverage samples were analysed directly. BPA was concentrated using on-line SPE. Analysis was carried out by LC-MS LOD = 0.025 µg/L in the cola, 0.015 µg/L in the lemon soda and 0.025 µg/L in the tonic water (3x s:n ratio) LOQ = 0.085 µg/L in the cola, 0.050 µg/L in the	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								lemon soda and 0.085 µg/L in the tonic water (10x s:n ratio) Recovery = 98 % in the cola, 97 % in the lemon soda and 97 % in the tonic spiked at 0.5 µg/L, 98 % in the cola, 96 % in the lemon soda and 94 % in the tonic spiked at 0.2 µg/L (five replicates of each) Repeatability = 2.5 % in the cola, 4 % in the lemon soda and 3.5 % in the tonic spiked at 0.5 µg/L, 3 % in the cola, 5 % in the lemon soda and 5 % in the tonic spiked at 0.2 µg/L (five replicates of each) Calibration = 0.05 to 10 µg/L No measures against contamination reported	
Field-amplified sample injection-micellar electrokinetic capillary chromatography for the analysis of bisphenol A, bisphenol F, and their diglycidyl	Gallart-Ayala, H., Nunez, O., Moyano, E. and Galceran, M. T.	Electrophoresis	2010	31:9, 1550-1559	10.1002/elps.200900606	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
ethers and derivatives in canned soft drinks									
Decanoic acid reverse micelle-based coacervates for the microextraction of bisphenol A from canned vegetables and fruits	García-Prieto, L., Lunar, L., Rubio, S. and Pérez-Bendito, D.	Analytica Chimica Acta	2008	617, 51-58	10.1016/j.aca.2008.01.061	1 can of each of red peppers, sweetcorn, green beans, peas, fruit salad, peaches in syrup - all from Spain and 1 can of mango slices from Thailand	Samples purchased in Spain	BPA was extracted from the foods using coacervative microextraction. Analysis was carried out by LC-FLD LOD = 1.3 µg/kg peas (3x s:n ratio) LOQ = 9.3 µg/kg (not stated how determined) Recovery = 86 % for six replicates of peas spiked at 200 µg/kg Repeatability = 2.8 % for six replicates of peas spiked at 200 µg/kg Calibration = 0.14 to 20 ng BPA in acetonitrile (not expressed as a concentration) No measures against contamination reported	Included
Intake of bisphenol A from canned beverages and foods on the Belgian market	Geens, T., Zipora Apelbaum, T., Goeyens, L., Neels, H. and	Food Additives and Contaminants	2010	27:11, 1627-1637	10.1080/19440049.2010.508183	50 beverages (45 canned, 4 in PET and 1 in Tetra Pak) and 44 foods including	Samples purchased in Belgium	After degassing BPA was extracted from the beverage sample using SPE. BPA was extracted from solid content of canned foods using solvent. The liquid content of canned food was filtered.	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	Covaci, A.					fruits, vegetables, soups, fish and meat (27 canned, 1 in paper, 2 in Tetra Pak, 10 in glass and 4 in plastic containers)		Analysis was carried out by GC-MS after derivatisation with pentafluorobenzoylchloride LOD = not given LOQ = 0.02 µg/kg for beverages, 0.10 µg/kg for food (calculated from 3x st dev of the procedural blanks) Recovery = 95 % for beverages spiked at 4.4 µg/L, 93 % for foods spiked at 10.5 µg/kg Repeatability = within day = 0.8 - 5.5 % for beverages spiked at 4.4 µg/L and 2.8 % for foods spiked at 10.5 µg/kg; between day = 3.0 % for beverages spiked at 4.4 µg/L and 2.8 % for foods spiked at 10.5 µg/kg Calibration = not given Method blank prepared to determine any contamination through the procedure	
A review of dietary and non-dietary exposure to bisphenol-A	Geens, T., Aerts, D., Berthot, C., Bourguign	Food and Chemical Toxicology	2012	50, 3725-3740	10.1016/j.fct.2012.07.059	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from food

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	on, J. P., Goeyens, L., Lecomte, P., Maghuin-Rogister, G., Pironnet, A. M., Pussemier, L., Scippo, M. L., Van Loco, J. and Covaci, A.								
Determination of bisphenol A and bisphenol B residues in canned peeled tomatoes by reversed-phase liquid chromatography	Grumetto, L., Montesano, D., Seccia, S., Albrizio, S. and Barbato, F.	Journal of Agricultural and Food Chemistry	2008	56, 10633-10637	10.1021/jf800297z	42 canned tomato samples (38 from Italy, 4 from China). 26 samples had packaging coated with epoxyphenolic lacquer and 16 with low BADGE enamel	Samples purchased in Italy	BPA was extracted from the samples with solvent, concentrated and the solvent extracts passed down the SPE cartridges. Analysis was carried out by LC-UV and LC-FLD (fractions were collected and infused into an MS source for confirmation) LOD = 1.1 µg/kg (calculated as 3x st dev of the noise) LOQ = 3.7 µg/kg (calculated as 10x st dev of the noise) Recovery = 94.3 % BPA spiked at 100, 200, 300 and	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								500 µg/kg into blank tomatoes Repeatability = 2.63 % BPA spiked at 100, 200, 300 and 500 µg/kg into blank tomatoes Calibration = External calibration 50 to 1000 µg/L Control (previously verified as BPA free) tomato samples used as method blank matrices to determine any contamination through the procedure. No plastic ware was used in the laboratory	
4-Nonylphenol and bisphenol A in Swedish food and exposure in Swedish nursing women	Gyllenhammar, I., Glynn, A., Darnerud, P. O., Lignell, S., van Delft, R. and Aune, M.	Environment International	2012	43, 21-29	10.1016/j.envint.2012.02.010	Samples tested were composites of food groups	Samples purchased in Sweden	Solvent extracted samples were cleaned-up by gel permeation chromatography. Analysis was carried out by GC-MS following acetylation LOD = not given LOQ = 2-4 µg/kg fresh weight Recovery = not given Repeatability = not given Calibration = not given No measures against contamination reported	Excluded - samples were collected in 2005 - market basket with wide pooled samples. Some samples also have canned and non-canned food together.
Determination of bisphenol A in	Hadjmohammadi,	Monatshefte für	2010	141:5, 501-506	10.1007/s0	Not considered	Not considered	Not considered	Excluded - samples from Iran

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Iranian packaged milk by solid-phase extraction and HPLC	Mohammad Reza, Saeidi, Iman	Chemie			0706-010-0297-1				
Human exposure to bisphenol A	Kang, J. H., Kondo, F. and Katayama, Y.	Toxicology	2006	226:2-3, 79-89	10.1016/j.tox.2006.06.009	Not considered	Not considered	Not considered	Excluded - review paper from Japan - no relevant data for calculation of exposure from food
Risk assessment of bisphenol A migrated from canned foods in Korea	Lim, D. S., Kwack, S. J., Kim, K. B., Kim, H. S. and Lee, B. M.	Journal of Toxicology and Environmental Health. Part A	2009	72:21-22, 1327-1335	10.1080/15287390903212444	Not considered	Not considered	Not considered	Excluded - samples from Korea
On-line precolumn enrichment of bisphenol A using boronate column in microcolumn liquid chromatography	Lim, L. W. and Takeuchi, T.	Journal of Chromatography A	2006	1106:1-2, 139-145	10.1016/j.chroma.2005.09.003	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food
Elimination of matrix effects in the determination of bisphenol A in milk by solid-phase microextraction-	Liu, X., Ji, Y., Zhang, H. and Liu, M.	Food Additives and Contaminants	2008	25:6, 772-778	10.1080/02652030701713921	Not considered	Not considered	Not considered	Excluded - samples from China

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
high-performance liquid chromatography									
Isotope dilution-gas chromatography/ mass spectrometry method for the analysis of alkylphenols, bisphenol A, and estrogens in food crops	Lu, J., Wu, J., Stoffella, P. J. and Wilson, P. C.	Journal of Chromatography A	2012	1258, 128-135	10.1016/j.chroma.2012.08.033	Not considered	Not considered	Not considered	Excluded - samples from USA
Determination of bisphenol A in milk by solid phase extraction and liquid chromatography–mass spectrometry	Maragou, N.C., Lampi, E. N., Thomaidis, N. S. and Koupparis, M. A.	Journal of Chromatography A	2006	1129, 165-173	10.1016/j.chroma.2006.06.103	8 canned condensed milk and 1 canned powdered infant formula sample	Samples purchased in Greece	BPA was extracted from the milk samples using solid phase extraction. Analysis was carried out by LC-ESI-MS LOD = 1.7 µg/kg milk (3.3×SDn=10)/b) where SD is the st dev of the response of 10 replicate milk samples spiked at 5 µg/kg, b is the slope of the calibration line from 5 to 200 µg/L LOQ = 5.1 µg/kg milk ((10×SDn=10)/b) Recovery = 83 % for milk spiked at 5 µg/kg, 101 % for milk spiked at 50 µg/kg and	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								106 % for milk spiked at 500 µg/kg (intra-day, n=6); 97 % for milk spiked at 5 µg/kg, 97 % for milk spiked at 50 µg/kg and 104 % for milk spiked at 500 µg/kg (inter-day, n=6) Repeatability = 12.5 % for milk spiked at 5 µg/kg, 5.0 % for milk spiked at 50 µg/kg and 2.1 % for milk spiked at 500 µg/kg (intra-day, n=6); 17.6 % for milk spiked at 5 µg/kg, 5.8 % for milk spiked at 50 µg/kg and 5.2 % for milk spiked at 500 µg/kg (inter-day, n=6) Calibration = External calibration 5 to 700 µg/L Water and milk blanks were analysed in each batch to check for contamination	
Dietary exposure assessment of pregnant women to bisphenol-A from cans and microwave containers in Southern Spain	Mariscal-Arcas, M., Rivas, A., Granada, A., Monteagudo, C., Murcia, M. A. and Olea-	Food and Chemical Toxicology	2009	47, 506-510	10.1016/j.fct.2008.12.011	Not considered	Not considered	Not considered	Excluded - migration from food contact materials paper - no relevant data for calculation of exposure from food

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	Serrano, F.								
Selective Molecularly Imprinted Polymer Obtained from a Combinatorial Library for the Extraction of Bisphenol A	Martin-Esteban, A. and Tadeo, J. L.	Combinatorial Chemistry and High Throughput Screening	2006	9, 747-751	Not given	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food
Bisphenol A content in fish caught in two different sites of the Tyrrhenian Sea (Italy)	Mita, L., Bianco, M., Viggiano, E., Zollo, F., Benciveng a, U., Sica, V., Monaco, G., Portaccio, M., Diano, N., Colonna, A., Lepore, M., Canciglia, P. and Mita, D. G.	Chemosph ere	2011	82, 405-410	10.10 16/j.c hemo spher e.201 0.09. 071	Dorsal muscular tissue and liver samples of mullet, salpa, white bream, bass and ombrine	Samples obtained from two coastal regions of Italy	Solvent extracted samples were cleaned-up by SPE. Analysis was carried out by HPLC-UV or FLD and in some cases were validated by GC-MS LOD = not given LOQ = not given Recovery = not given Repeatability = not given Calibration = not given Samples stored in glass containers but no other measures described to reduce background contamination	Excluded - method performance criteria not defined and so criteria could not be confirmed to have been met
Analysis of bisphenol A in	Molina-García, L.,	Talanta	2012	96, 195-201	10.10 16/j.t	3 x Powdered	Samples purchased in	Following precipitation of the protein the BPA was	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
milk by using a multicommuted fluorimetric sensor	Fernández-de Córdova, M. L. and Ruiz-Medina, A.				alanta.2012.02.021	milk, 2 x powdered infant formula, 3 x liquid infant formula and 6 x liquid milk	Spain	extracted from the sample using SPE. Analysis was carried out using a multicommuted fluorimetric sensor LOD = 0.06 µg/L (paper doesn't describe how it was determined) LOQ = 0.2 µg/L (0.19 µg/kg) (not described how determined) Recovery = 93-106 % for four samples spiked at 0.5, 2.0 and 5.0 µg/L Repeatability = Intra-day = 3.4 % at 4 µg/L. Inter-day = 5.7 % at 4 µg/L Calibration = 0.2 to 5.0 µg/L No measures against contamination reported	
Development of monoclonal antibody-based immunoassays for the analysis of bisphenol A in canned vegetables	Moreno, M. J., D'Arienzo, P., Manclus, J. J. and Montoya, A.	Journal of Environmental Science and Health B	2011	46:6, 509-517	10.1080/03601234.2011.583871	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food
Assessing the quantitative relationships	Morgan, M. K., Jones, P.	Environmental Science	2011	45:12, 5309-5316	10.1021/es2005	Not considered	Not considered	Not considered	Excluded - biomonitoring paper - no relevant data for

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
between preschool children's exposures to bisphenol A by route and urinary biomonitoring	A., Calafat, A. M., Ye, X., Croghan, C. W., Chuang, J. C., Wilson, N. K., Clifton, M. S., Figueroa, Z. and Sheldon, L. S.	and Technology			37u				calculation of exposure from food
Simultaneous determination of bisphenol A and alkylphenol in plant oil by gel permeation chromatography and isotopic dilution liquid chromatography-tandem mass spectrometry	Niu, Y., Zhang, J., Wu, Y. and Shao, B.	Journal of Chromatography A	2011	1218:31, 5248-5253	10.1016/j.chroma.2011.06.005	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food
Analysis of bisphenol A and alkylphenols in cereals by automated on-line solid-phase	Niu, Y., Zhang, J., Wu, Y. and Shao, B.	Journal of Agricultural and Food Chemistry	2012	60:24, 6116-6122	10.1021/jf30140 1k	Not considered	Not considered	Not considered	Excluded - samples from China

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
extraction and liquid chromatography tandem mass spectrometry									
Concentration of bisphenol A in highly consumed canned foods on the US market	Noonan, G. O., Ackerman, L. K. and Begley, T. H.	Journal of Agricultural and Food Chemistry	2011	59:13, 7178-7185	10.1021/jf20107 6f	Not considered	Not considered	Not considered	Excluded - samples from USA
Assessment of PCDD/F, PCB, OCP and BPA dietary exposure of non-breast-fed European infants	Pandelova, M., Piccinelli, R., Levy Lopez, W., Henkelmann, B., Molina-Molina, J. M., Arrebola, J. P., Olea, N., Leclercq, C. and Schramm, K.-W.	Food Additives and Contaminants: Part A	2011	28:8, 1110-1122	10.1080/19440049.2011.583281	6 pooled samples of infant formula and 5 pooled samples of baby food representing the diet of babies aged 5 to 9 months of age (including jarred foods)	Samples purchased in seven EU countries (Germany, UK, France, Sweden, Italy, Portugal, Slovak Republic)	BPA was extracted from the infant formula samples using acetonitrile. BPA was extracted from the freeze-dried solid food samples using hexane and acetonitrile. Following solid phase extraction the extracts were evaporated to dryness and derivatised using BSTFA. Analysis was carried out by GC-MS. Chlorinated BPA determined as well as BPA LOD = 0.8 to 1.7 µg/kg for BPA and it's chlorinated derivatives in the infant formula and 1.5 to 3.3 µg/kg for BPA and it's chlorinated derivatives in the solid foods	Excluded - method performance criteria not well defined and so criteria could not be confirmed to have been met

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								and beverages (the paper doesn't describe how these were determined) LOQ = 2.6 to 5.8 µg/kg for BPA and it's chlorinated derivatives in the infant formula and 4.9 to 10.9 µg/kg for BPA and it's chlorinated derivatives in the solid foods and beverages (the paper doesn't describe how these were determined) Recovery = average recoveries were: 99.0 % (BPA), 101.2 % (ClBPA), 92.9 %, (Cl2BPA), 93.3 % (Cl3BPA) and 93.5 % (Cl4BPA) Repeatability = not given Calibration = not given No measures against contamination reported	
Determination of bisphenol A in canned fatty foods by coacervative microextraction, liquid chromatography and fluorimetry	Pérez Bendito, M. D., Rubio Bravo, S., Lunar Reyes, M. L. and García Prieto, A.	Food Additives and Contaminants: Part A	2009	26:2, 265-274	10.1080/02652030802368740	1 can of each of tuna in oil, mackerel in vegetable oil, sardines in olive oil, mussels in pickled sauce, meatballs	Samples purchased in Spain	BPA was extracted from the solid portion of the foods (the liquid portion was discarded) using coacervative microextraction. Analysis was carried out by LC-FLD LOD = 9 µg/kg (3x s:n ratio) LOQ = depends on sample	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
						and luncheon meat		mass taken for 200 mg sample method quantification limit is 29 µg/kg for tuna in oil; for 400 mg sample method quantification limit is 14 µg/kg for tuna in oil Recovery = 90-99 % for overspiked food samples spiked with 50 ng BPA with a mass of food of either 200 mg or 400 mg Repeatability = 6 % for tuna spiked with BPA at concentrations between 0.05 and 1.5 µg/kg Calibration = 0.2 to 60 ng BPA in acetonitrile (not expressed as a concentration) No measures against contamination reported	
Determination of bisphenol A in canned fish by sol–gel immunoaffinity chromatography, HPLC and fluorescence detection	Podlipna, D. and Cichna-Markl, M.	European Food Research and Technology	2007	224, 629-634	10.1007/s00217-006-0350-9	7 tuna in brine, 5 tuna in oil, 5 sardines in oil, 1 mackerel in brine and 1 mackerel in oil	Samples purchased in Austria	Solvent extracted samples were cleaned-up by sol-gel immunoaffinity chromatography, using polyclonal BPA rabbit antibodies. Analysis was carried out by HPLC-FLD LOD = 0.4 µg/L in solution, 0.4 µg/kg in tuna, 0.2 µg/kg	Included The highest value was obtained by analysing a sample after its sell by date

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								in sardines, 0.2 µg/kg in mackerel, 0.9 µg/L in brine, 1.8 µg/L in oil (all 3x s:n ratio) LOQ = 0.74 µg/L in solution, 0.8 µg/kg in tuna, 0.4 µg/kg in sardines, 0.4 µg/kg in mackerel, 1.9 µ/L in brine, 3.8 µg/L in oil (all 6x s:n ratio) Recovery = 45 % in tuna, 97 % in sardines, 83 % in mackerel, 61 % in brine, 31 % in oil Repeatability = Standard deviation of the recovery was 5 % in tuna, 12 % in sardines, 26 % in mackerel, 12 % in brine, 9 % in oil Calibration = External calibration 0.5 to 100 µg/L No measures against contamination reported	
Determination and occurrence of bisphenol A, bisphenol A diglycidyl ether, and bisphenol F diglycidyl ether, including their derivatives, in	Poustka, J., Dunovská, L., Hajšlová, J., Holadová, K. and Poustková, I.	Czech Journal of Food Sciences	2007	25:4, 221-229	Not given	1 can of each of sardines in oil, mackerel in oil, tuna fish, cod liver, luncheon meat and pate (pork)	Samples purchased in Czech Republic	Solvent extracted samples were cleaned-up by gel permeation chromatography. Analysis was carried out by HPLC-FLD LOD = 3 µg/kg luncheon meat LOQ = 10 µg/kg luncheon	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
canned foodstuffs' from the Czech retail market								meat Recovery = 83 % in pork luncheon meat spiked at 100 µg/kg Repeatability = Coefficient of variation = 3.0 % for pork luncheon meat spiked at 100 µg/kg Calibration = External calibration 2 to 100 µg/L No measures against contamination reported	
Levels Of bisphenol A and bisphenol F In canned foods in Iranian markets	Rastkari, N., Yunesian, M. and Ahmadkhaniha, R.	Iranian Journal of Environmental Health Science and Engineering	2011	8, 95-100	Not given	Not considered	Not considered	Not considered	Excluded - samples from Iran
Properties, threats, and methods of analysis of bisphenol a and its derivatives	Rykowska I. and Wasiak W.	Acta Chromatographica	2006	16, 7-27	Not given	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from food
Bisphenol A (BPA) and its source in foods in Japanese markets	Sajiki, J., Miyamoto, F., Fukata, H., Mori, C., Yonekubo,	Food Additives and Contaminants	2007	24:1, 103-112	10.1080/02652030600936383	Not considered	Not considered	Not considered	Excluded - samples from Japan

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	J. and Hayakawa, K.								
Fast and selective pressurized liquid extraction with simultaneous in cell clean up for the analysis of alkylphenols and bisphenol A in bivalve molluscs	Salgueiro-Gonzalez, N., Turnes-Carou, I., Muniategui-Lorenzo, S., Lopez-Mahia, P. and Prada-Rodriguez, D.	Journal of Chromatography A	2012	1270, 80-87	10.1016/j.chroma.2012.11.014	6 samples of molluscs	Samples obtained from Spain	BPA was extracted using selective pressurised liquid extraction with a simultaneous in cell clean up with analysis by LC-MS/MS LOD = 0.9 µg/kg in the foodstuff (average of procedural blanks + 3 x st dev of 10 procedural blanks) LOQ = 3.3 µg/kg in the foodstuff (average of procedural blanks + 10 x st dev of 10 procedural blanks) Recovery = 93-99 % (BPA spike levels into the mussels = 5, 50 and 500 µg/kg, seven replicates at each level) Repeatability = 3-8 % (BPA spike levels into the mussels = 5, 50 and 500 µg/kg, seven replicates at each level) Calibration = Quantification was achieved by standard addition. Linearity was demonstrated between 0.001 and 10,000 µg/kg Filters and sorbents were rinsed with solvent prior to	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								use. Procedural blanks were included to ensure background levels were low.	
Simultaneous determination of organochlorine pesticides and bisphenol A in edible marine biota by GC-MS	Santhi, V. A., Hairin, T. and Mustafa, A. M.	Chemosphere	2012	86:10, 1066-1071	10.1016/j.chemosphere.2011.11.063	Not considered	Not considered	Not considered	Excluded - samples from Malaysia
Analysis of alkylphenol and bisphenol A in meat by accelerated solvent extraction and liquid chromatography with tandem mass spectrometry	Shao, B., Han, H., Li, D., Ma, Y., Tu, X. and Wu, Y.	Food Chemistry	2007	105:3, 1236-1241	10.1016/j.foodchem.2007.02.040	Not considered	Not considered	Not considered	Excluded - samples from China
Analysis of alkylphenol and bisphenol A in eggs and milk by matrix solid phase dispersion extraction and liquid chromatography with tandem mass spectrometry	Shao, B., Han, H., Tu, X. and Huang, L.	Journal of Chromatography B	2007	850:1-2, 412-416	10.1016/j.jchromb.2006.12.033	Not considered	Not considered	Not considered	Excluded - samples from China
Single laboratory	Sun, C.,	Journal of	2006	1129:1,	10.10	Not	Not	Not considered	Excluded - review

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
validation of a method for the determination of bisphenol A, bisphenol A diglycidyl ether and its derivatives in canned foods by reversed-phase liquid chromatography	Leong, L. P., Barlow, P. J., Chan, S. H. and Bloodworth, B. C.	Chromatography A		145-148	16/j.chroma.2006.08.018	considered	considered		paper from Japan - no relevant data for calculation of exposure from food
Determination of bisphenol A in water and milk by micellar liquid chromatography	Szymański, A. and Wasiak, W.	Acta Chromatographica	2006	17, 161-172	Not given	Powdered milk and mineral water in PC bottles	Samples obtained from Poland	BPA was extracted from the water and reconstituted powdered milk samples using solid phase extraction. Analysis was carried out by micellar LC-UV LOD = 0.3 µg/L (3x s:n ratio) LOQ = 1.0 µg/L (10x s:n ratio) Recovery = 92.3 % for BPA spiked into water at 1 µg/L (after the SPE step?) six replicates Repeatability = 3.97 % for BPA spiked into water at 1 µg/L (after the SPE step?) six replicates Calibration = External	Excluded - method performance criteria not well defined and so criteria could not be confirmed to have been met

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								calibration 0.5 to 100 µg/L No measures against contamination reported	
Human exposure to bisphenol A (BPA)	Vandenberg, L. N., Hauser, R., Marcus, M., Olea, N. and Welshons, W. V.	Reproductive Toxicology	2007	24:2, 139-177	10.1016/j.reprotox.2007.07.010	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from food
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	Vinas, P., Campillo, N., Martinez-Castillo, N. and Hernandez-Cordoba, M.	Analytical and Bioanalytical Chemistry	2010	397:1, 115-125	10.1007/s00216-010-3464-7	9 canned food samples (peas, peas with carrots, sweet corn, artichoke, mushroom, bean shoot and mixed vegetables). Both the supernatant liquid contained in the can and the solid food were analysed (separately)	Samples obtained from Spain	BPA was extracted from the supernatant and food samples following dilution/slurrying with water using solid phase microextraction. Derivatisation with acetic anhydride and BSTFA were compared. Analysis was carried out by GC-MS LOD = 0.016 µg/L (derivatisation using acetic anhydride), 0.025 µg/L (derivatisation using BSTFA) - 3 x s:n of solvent standards LOQ = 0.055 µg/L (derivatisation using acetic anhydride), 0.083 µg/L (derivatisation using	Excluded - method performance criteria not well defined and so criteria could not be confirmed to have been met

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
								BSTFA) - 10 x s:n ratio of solvent standards Recovery = 84-112 % for BPA spiked into supernatant at 0.5 and 5 µg/L six replicates Repeatability = 5.12 % (derivatisation using BSTFA) and 5.43 % (derivatisation using acetic anhydride) - for solvent standards Calibration = External calibration - working range described as 0.05 to 10 µg/L (derivatisation with acetic anhydride) and 0.1 to 10 µg/L (derivatisation with BSTFA) - reported concentrations were outside this range No measures against contamination reported	
Bisphenol a: how the most relevant exposure sources contribute to total consumer exposure	von Goetz, N., Wormuth, M., Scheringer, M. and Hungerbuhler, K.	Risk Analysis	2010	30:3, 473-487	10.1111/j.1539-6924.2009.01345.x	Not considered	Not considered	Not considered	Excluded - exposure paper - no relevant data for calculation of exposure from food
Assessment of	Wei, X.,	Chemosph	2011	85:1, 122-	10.10	Not	Not	Not considered	Excluded - samples

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
risk to humans of bisphenol A in marine and freshwater fish from Pearl River Delta, China	Huang, Y., Wong, M. H., Giesy, J. P. and Wong, C. K.	ere		128	16/j.chemosphere.2011.05.038	considered	considered		from China
An observational study of the potential exposures of preschool children to pentachlorophenol, bisphenol-A, and nonylphenol at home and daycare	Wilson, N. K., Chuang, J. C., Morgan, M. K., Lordo, R. A. and Sheldon, L. S.	Environmental research	2007	44079.29236	10.1016/j.envres.2006.04.006	Not considered	Not considered	Not considered	Excluded - biomonitoring paper - no relevant data for calculation of exposure from food
Endocrine disrupting chemicals: human exposure and health risks	Yang, M., Park, M. S. and Lee, H. S.	Journal of Environmental Science and Health. Part C	2006	24:2, 183-224	10.1080/10590600936474	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from food
Concentrations of bisphenol A, bisphenol A diglycidyl ether, and their derivatives in canned foods in Japanese markets	Yonekubo, J., Hayakawa, K. and Sajiki, J.	Journal of Agricultural and Food Chemistry	2008	56, 2041-2047	10.1021/jf073106n	Not considered	Not considered	Not considered	Excluded - samples from Japan
Sensitive gas	Zafra-	Food	2009	26:8,	10.10	Not	Not	Not considered	Excluded - analytical

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
chromatographic-mass spectrometric (GC-MS) method for the determination of bisphenol A in rice-prepared dishes	Gómez, A., Morales, J. C., Ballesteros, O. and Navalón, A.	Additives and Contaminants		1209-1216	80/026520309029390663	considered	considered		method paper - no relevant data for calculation of exposure from food
Pt/graphene-CNTs nanocomposite based electrochemical sensors for the determination of endocrine disruptor bisphenol A in thermal printing papers	Zheng, Z., Du, Y., Wang, Z., Feng, Q. and Wang, C.	Analyst	2012	138:2, 693-701	10.1039/c2an36569c	Not considered	Not considered	Not considered	Excluded - non-food paper - no relevant data for calculation of exposure from food

5908

5909 **Table 64:** Literature quality table – occurrence in drinking water

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Alkylphenols and phthalates in	Amiridou, D. and	Journal of Hazardous	2011	185:1, 281-286	10.1016/j.jh	Bottled waters	Greece	BPA was extracted from the water samples using	To be included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
bottled waters	Voutsa, D.	Materials			azmat .2010. 09.03 1			dichloromethane, dried and evaporated to dryness. The extracts were derivatised using BSTFA. Analysis was carried out by GC-MS LOD = range of 2-30 ng/L reported for all analytes tested LOQ = Not given Recovery = 77-92 % (for alkylphenols spiked at 20, 50 and 100 ng/L) Repeatability = Not given Calibration = 10 to 200 ng/L (seven levels) Glassware, solvents and samples were handled carefully to avoid contamination. Method blank prepared to determine any contamination through the procedure. Results were corrected for blank values.	
Survey of phthalates, alkylphenols, bisphenol A and herbicides in Spanish source waters intended for bottling	Bono-Blay, F., Guart, A., de la Fuente, B., Pedemonte, M., Cinta Pastor, M., Borrell, A. and	Environmental Science and Pollution Research	2012	19, 3339–3349	10.1007/s11356-012-0851-y	131 water sources intended for drinking	Distributed all over Spain	BPA was extracted from the water samples using solid phase extraction. Analysis was carried out by GC-MS LOD = 0.009 µg/L in the water (3x s:n ratio) LOQ = 0.029 µg/L in the water (10x s:n ratio) Recovery = 89 % at 1 µg/L,	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	Lacorte, S.							93 % at 0.1 µg/L (HPLC water spiked with BPA) Repeatability = 5.4 % (HPLC water spiked with BPA at 0.01 µg/L, 93 % at 0.1 µg/L) Calibration = External calibration 5 to 1000 µg/L (samples enriched during the procedure to be in this range) Method blank prepared to determine any contamination through the procedure	
Survey of bisphenol A in bottled water products in Canada	Cao, X-L and Corriveau, J.	Food Additives and Contaminants Part B	2008	1:2, 161-164	10.1080/02652030802563290	Not considered	Not considered	Not considered	Excluded - samples from Canada
Determination of bisphenol A in water via inhibition of silver nanoparticles-enhanced chemiluminescence	Chen, X., Wang, C., Tan, X. and Wang, J.	Analytica Chimica Acta	2011	689:1, 92-96	10.1016/j.aca.2011.01.031	Not considered	Not considered	Not considered	Excluded - samples from China
Quantification of bisphenol A, 353-nonylphenol and their chlorinated derivatives in drinking water treatment plants	Dupuis, A., Migeot, V., Cariot, A., Albouy-Llaty, M.,	Environmental science and pollution research internation	2012	19:9, 4193-4205	10.1007/s11356-012-0972-3	8 Drinking water samples collected at the outlet of the 8 different	France	BPA was extracted from the water samples using solid phase extraction. Analysis was carried out by LC-MS/MS. LOD = 0.5 ng/L (3x s:n ratio	To be included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	Legube, B. and Rabouan, S.	al				drinking water treatment plants		– corrected for recovery) LOQ = 1.5 ng/L (10x s:n ratio – corrected for recovery) Recovery = 108 % for blank samples spiked at 20 and 40 ng/L Repeatability = 7 % intra-day RSD, 18 % inter-day RSD Calibration = 2 to 40 ng/L (five levels) Glassware was baked, high quality solvents and teflon seals were used to minimise contamination. Method blanks were prepared to determine any contamination through the procedure.	
Bisphenol A Detection in Various Brands of Drinking Bottled Water in Riyadh, Saudi Arabia Using Gas Chromatography/ Mass Spectrometer	Elobeid, M. A., Almarhoon, Z. M., Virk, P., Hassan, Z. K., Omer, S. A., El Amin, M., Daghestani, M. H. and Al Olayan, E. M.	Tropical Journal of Pharmaceutical Research	2012	13, 455-459	10.4314/tjpr.v11i3.15	Not considered	Not considered	Not considered	Excluded - samples from Saudi Arabia
Migration of plasticizers phthalates,	Guart, A., Bono-Blay, F.,	Food Additives and	2011	28, 676-685	10.1080/1944004	Bottled water packed in 10	No details are provided on the place	BPA was extracted from the water samples using solid phase extraction. Analysis	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
bisphenol A and alkylphenols from plastic containers and evaluation of risk	Borrell, A. and Lacorte, S.	Contaminants Part A			9.2011.555845	in PET bottles, 10 in PC coolers and 7 in HDPE bottles	of purchase or sampling but the authors are from Spain	was carried out by GC-MS LOD = 0.009 µg/L (3x standard deviation of the blank samples, n=5) LOQ = not given Recovery = 97 % for HPLC water spiked at 1 µg/L Repeatability = not given Calibration = 10 to 10000 µg/L Method blank prepared to determine any contamination through the procedure	
Surface plasmon resonance sensor for detection of bisphenol A in drinking water	Hegnerová, K. and Homola, J	Sensors and Actuators B	2010	151:1, 177-179	10.1016/j.snb.2010.09.025	Not considered	Not considered	Not considered	Excluded - analytical method review paper - no relevant data for calculation of exposure from drinking water
Sol-gel coated polydimethylsiloxane/beta-cyclodextrin as novel stationary phase for stir bar sorptive extraction and its application to analysis of estrogens and bisphenol A	Hu, Y., Zheng, Y., Zhu, F. and Li, G.	Journal of Chromatography A	2007	1148:1, 16-22	10.1016/j.chroma.2007.02.101	Not considered	Not considered	Not considered	Excluded - samples from China
BPA and environmental	Ignatius, C. M.,	Bulletin of Environm	2010	85:5, 534-537	10.1007/s0	Not considered	Not considered	Not considered	Excluded - samples from Nigeria

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
estrogen in potable water sources in Enugu municipality, South-East, Nigeria	Francis, E. E., Emeka, E. N., Elvis, N. S. and Ebele, J. I.	ental Contamination and Toxicology			0128-010-0111-0				
Direct enrichment and high performance liquid chromatography analysis of ultra-trace Bisphenol A in water samples with narrowly dispersible Bisphenol A imprinted polymeric microspheres column	Jiang, M., Zhang, J. H., Mei, S. R., Shi, Y., Zou, L. J., Zhu, Y. X., Dai, K. and Lu, B.	Journal of Chromatography A	2006	1110:1-2, 27-34	10.1016/j.chroma.2006.01.051	Not considered	Not considered	Not considered	Excluded - samples from China
A novel sol-gel-material prepared by a surface imprinting technique for the selective solid-phase extraction of bisphenol A	Jiang, X., Tian, W., Zhao, C., Zhang, H. and Liu, M.	Talanta	2007	72:1, 119-125	10.1016/j.talanta.2006.10.006	Not considered	Not considered	Not considered	Excluded - samples from China
Determination of bisphenol A, bisphenol F and their diglycidyl	Jiao, Y, Ding, L, Fu, S., Zhu, S., Li,	Analytical Methods	2012	4:1, 291-298	10.1039/c1ay05433c	Not considered	Not considered	Not considered	Excluded - samples from China

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
ethers in environmental water by solid phase extraction using magnetic multiwalled carbon nanotubes followed by GC-MS/MS	H. and Wang, L.								
Exposure to bisphenol A from bis-glycidyl dimethacrylate–based dental sealants	Joskow, R., Boyd Barr, D., Barr, J. R., Calafat, A. M., Needham, L. L. and Rubin, C.	Journal of the American Dental Association	2006	137, 253-262	Not given	Not considered	Not considered	Not considered	Excluded – dental sealants data only - no relevant data for calculation of exposure from drinking water
Liquid phase microextraction with in situ derivatization for measurement of bisphenol A in river water sample by gas chromatography-mass spectrometry	Kawaguchi, M., Ito, R., Endo, N., Okanouchi, N., Sakui, N., Saito, K. and Nakazawa, H.	Journal of Chromatography A	2006	1110:1-2, 1-5	10.1016/j.chroma.2006.01.061	Not considered	Not considered	Not considered	Excluded - samples from Japan
Simultaneous determination and assessment of 4-nonylphenol,	Li, X., Ying, G. G., Su, H. C., Yang,	Environment international	2010	36:6, 557-562	10.1016/j.envint.2010.	Not considered	Not considered	Not considered	Excluded - samples from China

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
bisphenol A and triclosan in tap water, bottled water and baby bottles	X. B. and Wang, L.				04.009				
Properties, threats, and methods of analysis of bisphenol a and its derivatives	Rykowska I. and Wasiak W.	Acta Chromatographica	2006	16, 7-27	Not given	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from drinking water
Occurrence of bisphenol A in surface water, drinking water and plasma from Malaysia with exposure assessment from consumption of drinking water	Santhi, V. A., Sakai, N., Ahmad, E. D. and Mustafa, A. M.	The Science of the Total Environment	2012	427-428, 332-338	10.1016/j.scitotenv.2012.04.041	Not considered	Not considered	Not considered	Excluded - samples from Malaysia
Dummy molecularly imprinted polymers as the coating of stir bar for sorptive extraction of bisphenol A in tap water	Sheng, N., Wei, F., Zhan, W., Cai, Z., Du, S., Zhou, X., Li, F. and Hu, Q.	Journal of Separation Science	2012	35:5-6, 707-712	10.1002/jssc.201100883	Not considered	Not considered	Not considered	Excluded - samples from China
Occurrence and distribution of steroids,	Singh, S. P., Azua, A.,	Ecotoxicology	2010	19:2, 338-350	10.1007/s10646-	Not considered	Not considered	Not considered	Excluded - samples from USA

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
hormones and selected pharmaceuticals in South Florida coastal environments	Chaudhary, A., Khan, S., Willett, K. L. and Gardinali, P. R.				009-0416-0				
Efficiency of conventional drinking-water-treatment processes in removal of pharmaceuticals and other organic compounds	Stackelberg, P. E., Gibs, J., Furlong, E. T., Meyer, M. T., Zaugg, S. D. and Lippincott, R. L.	The Science of the Total Environment	2007	377:2-3, 255-272	10.1016/j.scitotenv.2007.01.095	Not considered	Not considered	Not considered	Excluded - samples from USA
Human exposure to bisphenol A (BPA)	Vandenberg, L. N., Hauser, R., Marcus, M., Olea, N. and Welshons, W. V.	Reproductive Toxicology	2007	24:2, 139-177	10.1016/j.reprotox.2007.07.010	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from drinking water
Rapid determination of bisphenol A in drinking water using dispersive liquid-phase microextraction with in situ derivatization	Wang, X., Diao, C. P. and Zhao, R. S.	Journal of Separation Science	2009	32:1, 154-159	10.1002/jssc.200800436	Not considered	Not considered	Not considered	Excluded - samples from China

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
prior to GC-MS									
Determination of Bisphenol A in Plastic Bottled Drinking Water by High Performance Liquid Chromatography with Solid-membrane Extraction Based on Electrospun Nylon 6 Nanofibrous Membrane	Wu S. Y., Xu, Q, Chen, T. S., Wang, M., Yin, X. Y., Zhang, N. P., Shen, Y. Y., Wen, Z. Y. and Gu Z. Z.	Chinese Journal of Analytical Chemistry	2010	38:4, 503-507	10.1016/s1872-2040(09)60035-9	Not considered	Not considered	Not considered	Excluded - samples from China
Endocrine disrupting chemicals: human exposure and health risks	Yang, M., Park, M. S. and Lee, H. S.	Journal of Environmental Science and Health. Part C	2006	24:2, 183-224	10.1080/10590506009036474	Not considered	Not considered	Not considered	Excluded - review paper - no relevant data for calculation of exposure from drinking water

Table 65: Literature quality table – occurrence in food contact materials

5915

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Migration from polycarbonate packaging to food simulants during microwave heating	Alin, L. and Hakkarainen, M.	Polymer Degradation and Stability	2012	97:8, 1387-1395	10.1016/j.polymdegradstab.2012.05.017	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials
The BIOSAFEPAPER project for in vitro toxicity assessments: preparation, detailed chemical characterisation and testing of extracts from paper and board samples	Bradley, E. L., Honkalampi-Hamalainen, U., Weber, A., Andersson, M. A., Bertaud, F., Castle, L., Dahlman, O., Hakulinen, P., Hoornstra, D., Lhuguenot, J. C., Maki-Paakkanen, J., Salkinoja-Salonen, M., Speck, D. R., Severin, I., Stammati, A., Turco, L., Zucco, F. and von Wright, A.	Food and Chemical Toxicology	2008	46:7, 2498-2509	10.1016/j.fct.2008.04.017	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials
Investigation into the migration potential of coating materials from cookware products	Bradley, E. L., Read, W. A. and Castle, L.	Food Additives and Contaminants	2007	24:3, 326-335	10.1080/02652030601013711	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials
Migration and sensory properties	Kontominas, M. G., Goulas, A. E.,	Food Additives	2006	23:6, 634-641	10.1080/0265	Not considered	Not considered	Not considered	Excluded - migration data rather than

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
of plastics-based nets used as food-contacting materials under ambient and high temperature heating conditions	Badeka, A. V. and Nerantzaki, A.	and Contaminants			2030600643369				occurrence data was used for the determination of the exposure from food contact materials
Oestrogenicity of paper and cardboard extracts used as food containers	Lopez-Espinosa, M. J., Granada, A., Araque, P., Molina-Molina, J. M., Puertollano, M. C., Rivas, A., Fernandez, M., Cerrillo, I., Olea-Serrano, M. F., Lopez, C. and Olea, N.	Food Additives and Contaminants	2007	24:1, 95-102	10.1080/02652030600936375	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials
Physicochemical processes involved in migration of bisphenol A from polycarbonate	Mercea, P.	Journal of Applied Polymer Science	2009	112:2, 579-593	10.1002/app.29421	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials
Bisphenol A (BPA) and its source in foods in Japanese markets	Sajiki, J., Miyamoto, F., Fukata, H., Mori, C., Yonekubo, J. and Hayakawa, K.	Food Additives and Contaminants	2007	24:1, 103-112	10.1080/02652030600936383	Not considered	Not considered	Not considered	Excluded - migration data rather than occurrence data was used for the determination of the exposure from food contact materials

5916
5917

5918 **Table 66:** Literature quality table – migration from food contact materials

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Migration from polycarbonate packaging to food simulants during microwave heating	Alin, L. and Hakkarainen, M.	Polymer Degradation and Stability	2012	97:8, 1387-1395	10.1016/j.polymdegradstab.2012.05.017	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of exposure
Alkylphenols and phthalates in bottled waters	Amiridou, D. and Voutsa, D.	Journal of Hazardous Materials	2011	185(1):281-6	10.1016/j.jhazmat.2010.09.031	Not considered	Not considered	Not considered	Not considered	Excluded – not relevant - occurrence in drinking water rather than migration data reported
Release of bisphenol A from polycarbonate baby bottles: mechanisms of formation and investigation of worst case scenarios	Biedermann-Brem, S., Grob, K. and Fjeldal, P.	European Food Research and Technology	2008	227:4, 1053-1060	10.1007/s00217-008-0819-9	PC baby bottles from four producers	Samples purchased in Norway	100 ºC for 5 min	BPA was determined in the exposed water samples by direct analysis using LC-FLD LOD = 0.01 µg/L (5 x s:n ratio) LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = Not given Measurement uncertainty quoted as 20 % but no indication is given as to how this was calculated No information on prevention of contamination or blanks	Included
Investigation	Bradley, E.	Food	2007	24:3, 326-	10.108	26 non-	Samples	Olive oil:	BPA was determined in	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
into the migration potential of coating materials from cookware products	L., Read, W. A. and Castle, L.	Additives and Contaminants		335	0/0265203060101371 1	stick coated cookware products , 5 tested for the migration of BPA	were purchased in the UK	175ºC for 1 h; 95 % ethanol: 60ºC for 6 h; Acetic acid: 100ºC for 1 h	the exposed 10 % ethanol and 3 % acetic acid simulants by HPLC-FLD. The exposed olive oil was diluted with heptane and extracted with acetonitrile which was analysed by HPLC-FLD LOD =Not given for all simulants/products - 0.026 mg/dm^2 in acetic acid for one product tested LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = Not given No information on prevention of contamination or blanks	
Identification of Potential Migrants in Epoxy Phenolic Can Coatings	Bradley, E. L., Driffield, M., Harmer, N., Oldring, P. K. T. and Castle, L.	International Journal of Polymer Analysis and Characterisation	2008	13:3, 200-223	10.1080/1023666080207051 2	Not considered	Not considered	Not considered	Not considered	Excluded - migration data for can coatings not used in the exposure assessment, occurrence in food data used for all cases except specific populations *
Determination of bisphenol A	Brenn-Struckhofo	Food Additive	2006	23:11, 1227–1235	10.1080/0265	Not considered	Not considered	Not considered	Not considered	Excluded – not relevant - occurrence

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
in wine by sol-gel immunoaffinity chromatography, HPLC and fluorescence detection	va, Z. and Cichna-Markl, M.	s and Contaminants			2030600654382					in food rather than migration data reported
Migration of Bisphenol A from Polycarbonate Baby and Water Bottles into Water under Severe Conditions	Cao, X.-L. and Corriveau, J.	Journal of Agricultural and Food Chemistry	2008	56, 6378–6381	10.1021/jf800870b	5 polycarbonate baby bottles	Samples were purchased in Canada	70ºC for 2 h	Following the addition of sodium chloride the BPA was extracted from the sample using SPME. Analysis was carried out by GC-MS LOD = 0.5 µg/L LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = 5 to 600 µg/L Method blanks were prepared to determine any contamination through the procedure. Blank levels detected were subtracted from the reported concentrations.	Included NOTE: although the samples were from outside Europe the comprehensive number and range of sample types provided data not available in Europe
Determination of Bisphenol A in Water by Isotope Dilution	Cao, X.-L. and Corriveau, J.	Journal of AOAC Internati	2008	91, 622-629	Not given	3 polycarbonate baby bottles and	Samples were purchased in Canada	25ºC for 24h	Following the addition of sodium chloride the BPA was extracted from the sample using SPME.	Included NOTE: although the samples were from

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Headspace Solid-Phase Microextraction and Gas Chromatography/Mass Spectrometry Without Derivatization		onal				2 water bottles			Analysis was carried out by GC-MS LOD = 0.5 µg/L LOQ = Not given Recovery = Not given Repeatability = 9.7 % (n=6 replicates at 5 µg/L) and 8.9 % (n=6 replicates at 20 µg/L) Calibration = 2.5 to 40 µg/L Deactivated glassware was used. Method blanks were prepared to determine any contamination through the procedure	outside Europe the comprehensive number and range of sample types provided data not available in Europe
Assessment of bisphenol A released from reusable plastic, aluminium and stainless steel water bottles	Cooper, J. E., Kendig, E. L. and Belcher, S. M.	Chemosphere	2011	85:6, 943-947	10.1016/j.chemosphere.2011.06.060	Reusable bottles: Nalgene, 32 ounce loop-top polycarbonate bottles, Tritan™ copolyester bottles, one litre stainless steel	Sample were purchased in the USA	25ºC for 5 days	BPA was determined in the exposed water samples by direct analysis using ELISA LOD = 0.05 µg/L LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = 0.05 to 10 µg/L Method blank prepared to determine any	Included (data for PC bottle only) NOTE: although the samples were from outside Europe the comprehensive number and range of sample types provided data not available in Europe

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
						bottles, aluminium epoxy resin lined bottles and Eco-Care™ lined bottles			contamination through the procedure	
Study on the migration of bisphenol-A from baby bottles by stir bar sorptive extraction-thermal desorption-capillary GC-MS	De Coensel, N., David, F. and Sandra, P.	Journal of Separation Science	2009	32:21, 3829-3836	10.1002/jssc.200900349	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of exposure
Migration of bisphenol A into water from polycarbonate baby bottles during microwave heating	Ehlert, K. A., Beumer, C. W. and Groot, M. C.	Food Additives and Contaminants Part A	2008	25:7, 904-910	10.1080/02652030701867867	Eighteen types of PC bottles from throughout Europe	Samples were purchased in Europe	100ºC for 1 min	BPA was extracted from the exposed simulant samples using SPE. Analysis was carried out by GC-MS after derivatisation with N-methyl-N-(trimethylsilyl) trifluoroacetamide LOD = 0.1 µg/L LOQ = Not given	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									Recovery = 95 % (water spiked with BPA at 1 µg/L, n = 5) Repeatability = 2 % (water spiked with BPA at 1 µg/L, n = 5) Calibration = 0.05 to 5 µg/L No measures against contamination reported	
Migration of phthalates, alkylphenols, bisphenol A and di(2-ethylhexyl)adipate from food packaging	Fasano, E., Bono-Blay, F., Cirillo, T., Montuori, P. and Lacorte, S.	Food Control	2012	27:1, 132-138	10.1016/j.foodcont.2012.03.005	Eleven food packaging materials	Not given	40ºC for 10 days	BPA was extracted from the exposed simulant samples using SPE. Analysis was carried out by GC-MS LOD = 21 to 33 ng/L LOQ = Not given Recovery = 80% (from water spiked with 100 ng of BPA in 30, 50 or 100 mL simulant, n = 2) Repeatability = Not given Calibration = 0.01 to 1 µg/mL No measures against contamination reported	Included (data for PC baby bottles only)
Phthalates and bisphenols migration in Mexican food	Gonzalez-Castro, M. I., Olea-Serrano,	Bulletin of Environmental	2011	86:6, 627-631	10.1007/s00128-011-0266-3	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Mexico

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
cans and plastic food containers	M. F., Rivas-Velasco, A. M., Medina-Rivero, E., Ordonez-Acevedo, L. G. and De Leon-Rodriguez, A.	Contamination and Toxicology								
Migration of plasticizers phthalates, bisphenol A and alkylphenols from plastic containers and evaluation of risk	Guart, A., Bono-Blay, F., Borrell, A. and Lacorte, S.	Food Additives and Contaminants Part A	2011	28, 676-685	10.1080/19440049.2011.555845	10 Water samples packed in PC coolers. Migration solutions derived from PC exposed to water for 10 days at 40oC	No details are provided on the place of purchase or sampling but the authors are from Spain	40ºC for 10 days	BPA was extracted from the water samples using solid phase extraction. Analysis was carried out by GC-MS LOD = 0.009 µg/L (3x standard deviation of the blank samples, n=5) LOQ = Not given Recovery = 97 % for HPLC water spiked at 1 µg/L Repeatability = not given Calibration = 10 to 10000 µg/L Method blank prepared to determine any contamination through the	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									procedure	
Sol-gel coated polydimethylsiloxane/beta-cyclodextrin as novel stationary phase for stir bar sorptive extraction and its application to analysis of estrogens and bisphenol A	Hu, Y., Zheng, Y., Zhu, F. and Li, G.	Journal of Chromatography A	2007	1148:1, 16-22	10.1016/j.chroma.2007.02.101	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Human exposure to bisphenol A	Kang, J. H., Kondo, F. and Katayama, Y.	Toxicology	2006	226:2-3, 79-89	10.1016/j.tox.2006.06.009	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Migration of bisphenol A from plastic baby bottles, baby bottle liners and reusable polycarbonate drinking bottles	Kubwabo, C., Kosarac, I., Stewart, B., Gauthier, B. R., Lalonde, K. and Lalonde, P. J.	Food Additives and Contaminants Part A	2009	26:6, 928-937	10.1080/02652030802706725	New and used baby bottles, baby bottle liners and re-usable drinks bottles	Samples were purchased in Canada	40ºC for 8 hr, 1 and 10 days	BPA was extracted from the water samples using solid phase extraction and from the ethanol solutions using solid phase extraction following acidification. Analysis was carried out by GC-MS/MS after derivatisation with N-methyl-N-(trimethylsilyl) trifluoroacetamide	Included NOTE: although the samples were from outside Europe the comprehensive number and range of sample types provided data not available in Europe

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									LOD = 0.04 ng/L LOQ = 0.11 ng/L Recovery = 93 % (simulant spiked at 0.25 ng/L, n = 7) Repeatability = 9.7 % (simulant spiked at 0.25 ng/L, n = 7) Calibration = Not given Method blank prepared to determine any contamination through the procedure	
Bisphenol A is released from polycarbonate drinking bottles and mimics the neurotoxic actions of estrogen in developing cerebellar neurons	Le, H. H., Carlson, E. M., Chua, J. P. and Belcher, S. M.	Toxicology Letters	2007	176:2, 149-156	10.1016/j.toxlet.2007.11.001	New and used polycarbonate baby bottles	Samples were purchased or obtained (used bottles) in the USA	22ºC for 24, 72, 120 and 168 hr; 100ºC for 24 h	BPA was determined in the exposed water samples by direct analysis using ELISA LOD = 0.05 µg/L LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = 0.05 to 10 µg/L Method blank prepared to determine any contamination through the procedure	Included NOTE: although the samples were from outside Europe the comprehensive number and range of sample types provided data not available in Europe
Voltammetric determination of bisphenol A	Li, J., Kuang, D., Feng, Y.,	Microchimica Acta	2011	172:3-4, 379-386	10.1007/s00604-010-	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
in food package by a glassy carbon electrode modified with carboxylated multi-walled carbon nanotubes	Zhang, F. and Liu, M.				0512-0					
Simultaneous determination and assessment of 4-nonylphenol, bisphenol A and triclosan in tap water, bottled water and baby bottles	Li, X., Ying, G. G., Su, H. C., Yang, X. B. and Wang, L.	Environment International	2010	36:6, 557-562	10.1016/j.envint.2010.04.009	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Potential risk of bisphenol A migration from polycarbonate containers after heating, boiling, and microwaving	Lim, D. S., Kwack, S. J., Kim, K. B., Kim, H. S. and Lee, B. M.	Journal of Toxicology and Environmental Health. Part A	2009	72:21-22, 1285-1291	10.1080/15287390903212329	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Oestrogenicity of paper and cardboard	Lopez-Espinosa, M. J.,	Food Additives and	2007	24:1, 95-102	10.1080/02652030600	Not considered	Not considered	Not considered	Not considered	Excluded - migration data for paper and cardboard not used in

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
extracts used as food containers	Granada, A., Araque, P., Molina-Molina, J. M., Puertollano, M. C., Rivas, A., Fernandez, M., Cerrillo, I., Olea-Serrano, M. F., Lopez, C. and Olea, N.	Contaminants			0936375					the exposure assessment, occurrence in food data used for all cases except specific populations *
Effect of amines in the release of bisphenol A from polycarbonate baby bottles	Maia, J. Cruz, J. M.l, Sendón, R., Bustos, J., Cirugeda, M. E., Sanchez, J. J. and Paseiro, P.	Food Research International	2010	43:5, 1283-1288	10.1016/j.foodres.2010.03.014	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of exposure
Effect of detergents in	Maia, J., Cruz, J.	Food Research	2009	42:10, 1410-1444	10.1016/j.food	Not considered	Not considered	Not considered	Not considered	Excluded – model studies determining

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
the release of bisphenol A from polycarbonate baby bottles	M., Sendón, R., Bustos, J., Sanchez, J. J. and Paseiro, P.	International			res.2009.07.003					the worst case data rather than migration data reported
Migration of bisphenol A from polycarbonate baby bottles under real use conditions	Maragou, N. C., Makri, A., Lampi, E. N., Thomaidis, N. S. and Koupparis, M. A.	Food Additives and Contaminants Part A	2008	25:3, 373-383	10.1080/02652030701509998	Not considered	Not considered	Not considered	Not considered	Excluded - model studies determining the worst case data rather than migration data reported
Physicochemical processes involved in migration of bisphenol A from polycarbonate	Mercea, P.	Journal of Applied Polymer Science	2009	112:2, 579-593	10.1002/app.29421	Polycarbonate films, discs, plaques, containers and water coolers	Not given	Not considered	Aqueous food simulant samples were analysed directly by LC-FLD. LOD = 0.5 to 1 µg /L for water and 3 % acetic acid LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = Not given No measures against contamination reported	Excluded - studies with tailor made samples or at non standardised conditions
Application of ethyl chloroformate	Mudiam, M. K., Jain, R.,	Analytical and Bioanaly	2011	401:5, 1695-1701	10.1007/s00216-011-	Not considered	Not considered	Not considered	Not considered	Excluded – samples from India

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
derivatization for solid-phase microextraction -gas chromatography-mass spectrometric determination of bisphenol-A in water and milk samples	Dua, V. K., Singh, A. K., Sharma, V. P. and Murthy, R. C.	tical Chemistry			5226-6					
Bisphenol A migration from polycarbonate baby bottle with repeated use	Nam, S. H., Seo, Y. M. and Kim, M. G.	Chemosphere	2010	79:9, 949-952	10.1016/j.chemosphere.2010.02.049	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Migration of Bisphenol A and Benzophenones from Paper and Paperboard Products Used in Contact with Food	Ozaki, O., Kawasaki, C., Kawamura, Y. and Tanamoto, K.	Journal of the Food Hygienic Society of Japan	2006	47:3, 99-104	Not given	Not considered	Not considered	Not considered	Not considered	Excluded - migration data for paper and paperboard products not used in the exposure assessment, occurrence in food data used for all cases except specific populations *
Determination of bisphenol-type endocrine disrupting compounds in food-contact	Perez-Palacios, D., Fernandez-Recio, M. A.,	Talanta	2012	99, 167-174	10.1016/j.talanta.2012.05.035	Not considered	Not considered	Not considered	Not considered	Excluded - migration data for paper materials not used in the exposure assessment, occurrence in food

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
recycled-paper materials by focused ultrasonic solid-liquid extraction and ultra performance liquid chromatography-high resolution mass spectrometry	Moreta, C. and Tena, M. T.									data used for all cases except specific populations *
Bisphenol A (BPA) and its source in foods in Japanese markets	Sajiki, J., Miyamoto, F., Fukata, H., Mori, C., Yonekubo, J. and Hayakawa, K.	Food Additives and Contaminants	2007	24:1, 103-112	10.1080/02652030600936383	Not considered	Not considered	Not considered	Not considered	Excluded – samples from Asia
Migration of bisphenol A from polycarbonate baby bottles purchased in the Spanish market by liquid	Santillana, M. I., Ruiz, E., Nieto, M. T., Bustos, J., Maia, J., Sendon, R. and Sanchez, J.	Food Additives and Contaminants. Part A	2011	28:11, 1610-1618	10.1080/19440049.2011.589036	72 baby bottle samples from 12 brands	Samples were purchased in Spain	70ºC for 2h	Aqueous food simulant samples were analysed directly by LC-FLD LOD = 4 to 7 µg/kg LOQ = 30 µg/kg Recovery = 107-118 % (blank spiked with BPA at 0.12, 0.6 and 1.2 mg/kg, n	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
chromatography and fluorescence detection	J.								= 9) Repeatability = 3.4 to 5.8 % (blank spiked with BPA at 0.12, 0.6 and 1.2 mg/kg, n = 9) Calibration = 0.03 to 1.2 mg/kg No measures against contamination reported	
Revision of analytical strategies to evaluate different migrants from food packaging materials	Sendón García, R., Sanches Silva, A., Cooper, I., Franz, R. and Paseiro Losada, P.	Trends in Food Science and Technology	2006	17:7, 354–366	10.1016/j.tifs.2006.01.005	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of exposure
Identification and quantification of the migration of chemicals from plastic baby bottles used as substitutes for polycarbonate	Simoneau, C., Van den Eede, L. and Valzacchi, S.	Food Additives and Contaminants. Part A	2012	29:3, 469-480	10.1080/19440049.2011.644588	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of exposure
Ultrasound-assisted emulsification	Vinas, P., Lopez-Garcia, I.,	Analytical and Bioanaly	2012	404:3, 671-678	10.1007/s00216-012-	Not considered	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation of

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
microextraction coupled with gas chromatography-mass spectrometry using the Taguchi design method for bisphenol migration studies from thermal printer paper, toys and baby utensils	Campillo, N., Rivas, R. E. and Hernandez-Cordoba, M.	tical Chemistry			5957-z					exposure
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	Vinas, P., Campillo, N., Martinez-Castillo, N. and Hernandez-Cordoba, M.	Analytical and Bioanalytical Chemistry	2010	397:1, 115-125	10.1007/s00216-010-3464-7	Not considered	Not considered	Not considered	Not considered	Excluded – reported studies determining the transfer to saliva rather than migration data for food simulants

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Sample description	Country of origin of samples	Migration test conditions	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Sensitive gas chromatographic-mass spectrometric (GC-MS) method for the determination of bisphenol A in rice-prepared dishes	Zafra-Gómez, A., Morales, J. C., Ballesteros, O. and Navalón, A.	Food Additives and Contaminants	2009	26:8, 1209-1216	10.1080/02652030902939663	Not considered	Not considered	Not considered	Not considered	Excluded – not relevant - occurrence in food data rather than migration data reported
Optimization of a GC/MS procedure that uses parallel factor analysis for the determination of bisphenols and their diglycidylethers after migration from polycarbonate tableware	Oca, M.L., Ortiz, M.C., Herrero, A., Sarabia, L.A.	Talanta	2013	106, 266-280		PC cups	Not given	70ºC for 24h	BPA was determined in the simulant 50% ethanol by GC-MS after SPE extraction. Procedural blanks were analysed. LOD = 2.65 µg/l LOQ = Not given Recovery = 114% Repeatability = 5% Calibration = 0 to 90 µg/l with BPA-d16 as internal standard	Included

* Specific populations for which migration data was used to calculate the exposure were: populations consuming foods served in PC tableware; populations consuming water from PC coolers; populations consuming beverages prepared with water boiled in PC kettles; populations consuming water that has passed through PC filters; populations consuming foodstuffs cooked in cookware to which a BPA containing non-stick coating has been applied

Table 67: Literature quality table - occurrence in non-food matrices

5924

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Detection and quantification of traces of bisphenol A and bisphenol S in paper samples using analytical pyrolysis-GC/MS	Becerra, V. and Odermatt, J.	Analyst	2012	137:9, 2250-2259	10.1039/c2an15961a	Paper	Not considered	Not considered	Not considered	Excluded - analytical method paper - no relevant data for calculation of exposure from non-food sources
Release of bisphenol A from polycarbonate baby bottles: water hardness as the most relevant factor	Biedermann-Brem, S. and Grob, K.	European Food Research and Technology	2009	228:5, 679-684	10.1007/s00217-008-0978-8	Food contact material	Not considered	Not considered	Not considered	Excluded - food contact material and migration data only - no relevant data for calculation of exposure from non-food sources
Transfer of bisphenol A from thermal printer paper to the skin	Biedermann, S., Tschudin, P. and Grob, K.	Analytical and Bioanalytical Chemistry	2010	398:1, 571-576	10.1007/s00216-010-3936-9	Paper	13 thermal printing papers (receipts and recorders for chromatographic instruments)	Switzerland	BPA was extracted from the paper by immersion in methanol overnight at 60°C. Analysis was carried out by LC-FLD LOD = Not given LOQ = 0.05 µg in 10 mL ethanol Recovery = Not given Repeatability = 4 % for repeat (n =	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									6) analysis of an extract at 1.2 µg/mL Calibration = 0.1 to 50 µg/mL No measures against contamination reported (levels detected are high and so typical background levels would not influence the concentrations measured in the samples)	
Detection of Bisphenol A on a Screen-Printed Carbon Electrode in CTAB Micellar Medium	Brugnera, M. F., Trindade, M. A. G. and Zanoni, M. V. B.	Analytical Letters	2010	43:18, 2823-2836	10.1080/00032711003731332	River water and sewage	Not considered	Not considered	Not considered	Excluded - environmental data only - no relevant data for calculation of exposure from non-food sources
Dental composite fillings and bisphenol A among children: a survey in South Korea	Chung, S. Y., Kwon, H., Choi, Y. H., Karmaus, W., Merchant,	International Dental Journal	2012	62:2, 65-69	10.1111/j.1875-595X.2011.00089.x	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in the calculation of exposure from non-food sources

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	A. T., Song, K. B., Sakong, J., Ha, M., Hong, Y. C. and Kang, D.									
Dermal penetration of bisphenol A in human skin contributes marginally to total exposure	Demierre, A. L., Peter, R., Oberli, A. and Bourqui-Pittet, M.	Toxicology Letters	2012	213:3, 305-308	10.1016/j.toxlet.2012.07.001	Not applicable	Not considered	Not considered	Not considered	Excluded – absorption paper - no relevant data for calculation of exposure from non-food sources NOTE: This manuscript was considered for determination of the absorption of BPA, but excluded for methodological reasons
Orthodontic materials research and applications: part 2. Current status and projected future developments in	Eliades, T.	American Journal of Orthodontics and Dentofacial Orthopedi	2007	131:2, 253-262	10.1016/j.ajodo.2005.12.029	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination.

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
materials and biocompatibility		cs								
Assessment of bisphenol-A release from orthodontic adhesives	Eliades, T., Hiskia, A., Eliades, G. and Athanasiou, A. E.	American Journal of Orthodontics and Dentofacial Orthopedics	2007	131:1, 72-75	10.1016/j.ajodo.2006.08.013	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Release of bisphenol-A from a light-cured adhesive bonded to lingual fixed retainers	Eliades, T., Voutsa, D., Sifakakis, I., Makou, M. and Katsaros, C.	American Journal of Orthodontics and Dentofacial Orthopedics	2011	139:2, 192-195	10.1016/j.ajodo.2009.12.02	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Bisphenol A and related compounds in dental materials	Fleisch, A. F., Sheffield, P. E., Chinn, C., Edelstein, B. L. and Landrigan, P. J.	Pediatrics	2010	126:4, 760-768	10.1542/peds.2009-2693	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Assessment of human exposure to Bisphenol-A, Triclosan and	Geens, T., Roosens, L., Neels, H. and	Chemosphere	2009	76:6, 755-760	10.1016/j.chemosphere.	Dust	Dust from 18 houses and 2 offices collected using a vacuum	Belgium	Dust samples were filtered and BPA was extracted from the dust with a	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Tetrabromobisphenol-A through indoor dust intake in Belgium	Covaci, A.				2009.05.024		cleaner		mixture of hexane and acetone (3:1). Following solid phase extraction the samples were evaporated to dryness and reconstituted in methanol. A labelled BPA internal standard was used. Analysis was carried out by LC-MS/MS. LOD = Not given LOQ = 3 µg/kg of dust Recovery = Not given (results automatically corrected through use of labelled internal standard) Repeatability = 6 % for repeat (n = 6) analysis of a homogenised dust sample Calibration = Range not given	

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									(seven levels used) The procedural blank sample was taken into account when determining the method LOQ	
Levels of bisphenol-A in thermal paper receipts from Belgium and estimation of human exposure	Geens, T., Goeyens, L., Kannan, K., Neels, H. and Covaci, A.	The Science of the Total Environment	2012	435-436, 30-33	10.1016/j.scitotenv.2012.07.001	Paper	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation strategy of exposure from non-food sources
Salivary bisphenol-A levels due to dental sealant/resin: a case-control study in Korean children	Han, D. H., Kim, M. J., Jun, E. J. and Kim, J. B.	Journal of Korean Medical Science	2012	27:9, 1098-1104	10.3346/jkms.2012.27.9.1098	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Comment on "High levels of bisphenol A in paper currencies from several countries, and implications for dermal exposure	Heinze, J. E.	Environmental Science and Technology	2011	45:21, 9464	10.1021/es203169y	Paper	Not considered	Not considered	Not considered	Excluded - no primary data for calculation of exposure from non-food sources
Quantitative Analysis of	Johnson, B. O.,	Journal of Chemical	2012	89, 1555−1560	10.1021/ed	Consumer products	Not considered	Not considered	Not considered	Excluded – source not considered in

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Bisphenol A Leached from Household Plastics by Solid–Phase Microextraction and Gas Chromatography–Mass Spectrometry (SPME–GC–MS)	Burke, F. M., Harrison, R and Burdette, S.	Education			2003884					exposure calculation
No Dental Dilemma for BPA	Josephson, J.	Environmental Health Perspectives	2006	114:7, A404	None given	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Release of bisphenol A from resin composite used to bond orthodontic lingual retainers	Kang, Y. G., Kim, J. Y., Kim, J., Won, P. J. and Nam, J. H.	American Journal of Orthodontics and Dentofacial Orthopedics	2011	140:6, 779-789	10.1016/j.ajodo.2011.04.022	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
High levels of bisphenol A in paper currencies from several countries, and implications for dermal exposure	Liao, C. and Kannan, K.	Environmental Science and Technology	2011	45:16, 6761-6768	10.1021/es200977t	Paper	Not considered	Not considered	Not considered	Excluded – samples from various countries worldwide, obtained in USA

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Widespread occurrence of bisphenol A in paper and paper products: implications for human exposure	Liao, C. and Kannan, K.	Environmental Science and Technology	2011	45:21, 9372-9379	10.1021/es202507f	Paper	Not considered	Not considered	Not considered	Excluded – samples from USA, Japan, Korea and Vietnam
Bisphenol S, a new bisphenol analogue, in paper products and currency bills and its association with bisphenol A residues	Liao, C., Liu, F. and Kannan, K.	Environmental Science and Technology	2012	46:12, 6515-6522	10.1021/es300876n	Paper	Not considered	Not considered	Not considered	Excluded - not related to BPA - no relevant data for calculation of exposure from non-food sources
Occurrence of eight bisphenol analogues in indoor dust from the United States and several Asian countries: implications for human exposure	Liao, C., Liu, F., Guo, Y., Moon, H. B., Nakata, H., Wu, Q. and Kannan, K.	Environmental Science and Technology	2012	46:16, 9138-9145	10.1021/es302004w	Dust	Not considered	Not considered	Not considered	Excluded - samples from USA, China, Japan and Korea
Reply to Comment on "High Levels of Bisphenol A in Paper Currencies from Several Countries, and	Liao, C. and Kannan, K.	Environmental Science and Technology	2011	45, 9465-9466	10.1021/es203380e	Paper	Not considered	Not considered	Not considered	Excluded - no primary data for calculation of exposure from non-food sources

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Implications for Dermal Exposure"										
Occurrence of bisphenol A in indoor dust from two locations in the eastern United States and implications for human exposures	Loganathan, S. N. and Kannan, K.	Archives of Environmental Contamination and Toxicology	2011	61:1, 68-73	10.1007/s00244-010-9634-y	Dust	Not considered	Not considered	Not considered	Excluded - samples from USA
Exposure to Bisphenol A (BPA) from dental sealants is detectable in saliva and urine, and varies significantly between sealant formulations	Martin, M. D.	The Journal of Evidence-Based Dental Practice	2007	7:2, 79-80	10.1016/j.jebdp.2007.03.008	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Concentration of bisphenol A in thermal paper	Mendum, T., Stoler, E., VanBenschoten, H. and Warner, J. C.	Green Chemistry Letters and Reviews	2011	4:1, 81-86	10.1080/17518253.2010.502908	Paper	Not considered	Not considered	Not considered	Excluded - samples from USA
The contribution of dermal	Mielke, H., Partosch,	Toxicology Letters	2011	204:2-3, 190-198	10.1016/j.to	Not applicable	Not considered	Not considered	Not considered	Excluded – absorption paper - no

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
exposure to the internal exposure of bisphenol A in man	F. and Gundert-Remy, U.				xlet.2011.04.032					primary data for calculation of exposure from non-food sources
Assessing the quantitative relationships between preschool children's exposures to bisphenol A by route and urinary biomonitoring	Morgan, M. K., Jones, P. A., Calafat, A. M., Ye, X., Croghan, C. W., Chuang, J. C., Wilson, N. K., Clifton, M. S., Figueroa, Z. and Sheldon, L. S.	Environmental Science and Technology	2011	45:12, 5309-5316	10.1021/es200537u	Indoor air, outdoor air, house dust, indoor surface	Not considered	Not considered	Not considered	Excluded - samples from USA
Long-term release of monomers from modern dental-composite materials	Polydorou, O., König, A., Hellwig, E. and Kümmerer, K.	European Journal of Oral Science	2009	117, 68-75	None given	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Effect of	Polydorou,	Dental	2009	25:2,	10.10	Dental	Not considered	Not	Not considered	Excluded - dental

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
bleaching on the elution of monomers from modern dental composite materials	O., Beiter, J., König, A., Hellwig, E. and Kümmerer, K.	Materials		254-260	16/j.dental.2008.07.004			considered		sealants not included in exposure determination
Release of monomers from different core build-up materials	Polydorou, O., Hammad, M., König, A., Hellwig, E. and Kümmerer, K.	Dental Materials	2009	25:9, 1090-1095	10.1016/j.dental.2009.02.014	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Elution of monomers from two conventional dental composite materials	Polydorou, O., Trittler, R., Hellwig, E. and Kümmerer, K.	Dental Materials	2007	23:12, 1535-1541	10.1016/j.dental.2006.12.011	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Bisphenol A in dental sealants and its estrogen like effect	Rathee, M., Malik, P. and Singh, J.	Indian Journal of Endocrinology and Metabolism	2012	16:3, 339-342	10.4103/2230-8210.95660	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Occurrence of bisphenol A in	Santhi, V. A., Sakai,	The Science of	2012	427-428, 332-338	10.1016/j.s	Surface water	Not considered	Not considered	Not considered	Excluded – samples from Malaysia

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
surface water, drinking water and plasma from Malaysia with exposure assessment from consumption of drinking water	N., Ahmad, E. D. and Mustafa, A. M.	the Total Environment			citotenv.2012.04.041					
How much do resin-based dental materials release? A meta-analytical approach	Van Landuyt, K. L., Nawrot, T., Geebelen, B., De Munck, J., Snauwaert, J., Yoshihara, K., Scheers, H., Godderis, L., Hoet, P. and Van Meerbeek, B.	Dental Materials	2011	27:8, 723-747	10.1016/j.dental.2011.05.001	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination
Systematic review of the chemical composition of contemporary	Van Landuyt, K. L., Snauwaert,	Biomaterials	2007	28:26, 3757-3785	10.1016/j.biomaterials.2	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in total exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
dental adhesives	J., De Munck, J., Peumans, M., Yoshida, Y., Poitevin, A., Coutinho, E., Suzuki, K., Lambrechts, P. and Van Meerbeek, B.				007.04.044					
Ultrasound-assisted emulsification microextraction coupled with gas chromatography-mass spectrometry using the Taguchi design method for bisphenol migration studies from thermal printer paper, toys and baby utensils	Viñas, P., López-Garcia, I., Campillo, N., Rivas, R. E. and Hernandez-Córdoba, M.	Analytical and Bioanalytical Chemistry	2012	404:3, 671-678	10.1007/s00216-012-5957-z	Paper and Toys	Not considered	Not considered	BPA was extracted from the paper by immersion in water. Toys were immersed in saliva simulant. Derivatisation with acetic anhydride and BSTFA were compared. Analysis was carried out by GC. LOD = 0.1 µg/L LOQ = 0.3 µg/	Excluded – experimental setup not appropriate

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									Recovery = Not given Repeatability = 7.6 % (replicate, n=10, analyses of samples at 1 µg/L) Calibration = 0.1 to 3 µg/L No measures against contamination reported	
Bisphenol a: how the most relevant exposure sources contribute to total consumer exposure	von Goetz, N., Wormuth, M., Scheringer, M. and Hungerbuhler, K.	Risk Analysis	2010	30:3, 473-487	10.1111/j.1539-6924.2009.01345.x	Various - review paper	Not considered	Not considered	Not considered	Excluded – modelling paper - no primary data for calculation of exposure from non-food sources
SVOC exposure indoors: fresh look at dermal pathways	Weschler, C. J. and Nazaroff, W. W.	Indoor Air	2012	22:5, 356-377	10.1111/j.1600-0668.2012.00772.x	Indoor surfaces	Not considered	Not considered	Not considered	Excluded – modelling paper - no primary data for calculation of exposure from non-food sources
An observational study of the potential exposures of	Wilson, N. K., Chuang, J. C.,	Environmental Research	2007	103:1, 9-20	10.1016/j.envres.2006.	Indoor air, outdoor air, house dust,	Not considered	Not considered	Not considered	Excluded – samples from USA

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
preschool children to pentachlorophenol, bisphenol-A, and nonylphenol at home and daycare	Morgan, M. K., Lordo, R. A. and Sheldon, L. S.				04.006	indoor surfaces				
Molecularly imprinted layer-coated silica nanoparticles for selective solid-phase extraction of bisphenol A from chemical cleansing and cosmetics samples	Zhu, R., Zhao, W., Zhai, M., Wei, F., Cai, Z., Sheng, N. and Hu, Q.	Analytica Chimica Acta	2010	658:2, 209-216	10.1016/j.aca.2009.11.008	Cosmetics	Not considered	Not considered	Not considered	Excluded - samples from China
Stir bar sorptive extraction with EG-Silicone coating for 4 bisphenols determination in personal care products by GC-MS	Cacho, J. I., Campillo, N., Viñas, P. and Hernández-Córdoba, M.	Journal of Pharmaceutical and Biomedical Analysis	2013	78-79, 255-260	10.1016/j.jpba.2013.02.023	Personal care products	30 cosmetic and personal care products	Spain	Following dilution with water the BPA was extracted using stir bar sorptive extraction, and analysed by thermal desorption GC-MS LOD = 8.7 µg/kg LOQ = 29.2 µg/kg Recovery = 89-114 % (replicate, n=10, analysis of	Included NOTE: although the paper was published in 2013 the comprehensive number and range of sample types provided data not otherwise available in Europe

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
									three samples spiked with BPA and 40 and 160 µg/kg) Repeatability = 2.1-11 % (replicate, n=10, analysis of three samples spiked with BPA and 40 and 160 µg/kg) Calibration = 0. 5 - 20 µg/L No measures against contamination reported	
Endocrine Disruptors and Asthma-Associated Chemicals in Consumer Products	Dodson, R., Nishioka, M., Standley, L. J., Green Brody, J. and Rudel, R.A.	Environmental Health Perspectives	2012	120:7, 935-943	None given	Consumer products	Not considered	Not considered	Not considered	Excluded - samples from USA
Bisphenol A contamination of wastepaper, cellulose and	Gehring, M., Vogel, D., Tennhardt,	Waste Management and the Environment	2004	294-299	None given	Paper	Not considered	Not considered	Not considered	Excluded – source not considered in exposure calculation

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
recycled paper products	L., Weltin, D. and Bilitewski, B.	II								
Bisphenol A (BPA) in China: a review of sources, environmental levels, and potential human health impacts	Huang, Y. Q., Wong, C. K., Zheng, J. S., Bouwman, H., Barra, R., Wahlstrom, B., Neretin, L. and Wong, M. H.	Environment International	2012	42, 91-99	10.1016/j.envint.2011.04.010	Review paper	Not considered	Not considered	Not considered	Excluded - samples from China
Exposure to bisphenol A from bis-glycidyl dimethacrylate–based dental sealants	Joskow, R., Boyd Barr, D., Barr, J. R., Calafat, A. M., Needham, L. L. and Rubin, C.	Journal of the American Dental Association	2006	137, 253-262	None given	Dental	Not considered	Not considered	Not considered	Excluded - dental sealants not included in exposure determination
Bisphenol A i leksaker och barnartiklar – behov av exponeringsminsk	KEMI	Kemi Rapport Nr 6/12	2012	None given	None given	Toys and childcare articles	24 toys and childcare articles tested for BPA content and	Sweden	BPA was soxhlet extracted from the toys using methanol or dichloromethane	Included

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
ning? Rapport från ett regeringsuppdrag							migration		and the extract was analysed by GC/MS. LOD = 50 µg/L of leachate LOQ = Not given Recovery = Not given Repeatability = Not given Calibration = Not given No measures against contamination reportedg	
Bisphenol A I svenska kvitton	Oestberg, T and Noaksson, E	Jegrelius	2010	None given	None given	Thermal paper	Not considered	Not considered	Not considered	Excluded - no relevant data for calculation strategy of exposure from non-food sources
Semivolatile Endocrine-Disrupting Compounds in Paired Indoor and Outdoor Air in Two Northern California Communities	Rudel, R. A., Dodsom, R. E., Perovich, L. J., Morello-Frosch, R., Camann,	Environmental Science and Technology	2010	44, 6583-6590	10.1021/es100159c	Indoor and outdoor air	Not considered	Not considered	Not considered	Excluded – samples from USA

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
	D. E., Zuniga, M. M., Yau, A. Y., Just, A. C. and Green Brody, J.									
Occurrence and estrogenicity of phenolics in paper-recycling process water: pollutants originating from thermal paper in waste paper	Terasaki, M., Shiraishi, F., Fukazawa, H. and Makino, M.	Environmental Toxicology and Chemistry	2007	26:11, 2356-2366	None given	Paper recycling process water	Not considered	Not considered	Not considered	Excluded – environmental data only - no relevant data for calculation of exposure from non-food source
Risk to all or none? A comparative analysis of controversies in the health risk assessment of Bisphenol A	Beronius, A., Ruden, C., Hakansson, H. and Hanberg, A.	Reproductive Toxicology	2010	29:2, 132-146	10.1016/j.reprotox.2009.11.007	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Excluded – review paper - no primary data for calculation of exposure from non-food sources
Are potential sources for human exposure to bisphenol-A overlooked?	Geens, T., Goeyens, L. and Covaci, A.	International Journal of Hygiene and Environmental Health	2011	214, 339-347	10.1016/j.ijheh.2011.04.005	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Excluded – review paper - no primary data for calculation of exposure from non-food sources

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
Cutaneous penetration of bisphenol A in pig skin	Kaddar, N., Harthé, C., Déchaud, H., Mappus, E. and Pugeat, M.	Journal of Toxicology and Environmental Health, Part A	2008	71, 471-473	10.1080/15287390801906824	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Excluded – absorption paper - no relevant data for calculation of exposure from non-food sources NOTE: This manuscript was considered for determination of the absorption of BPA, but excluded for methodological reasons
In vivo and ex vivo percutaneous absorption of [14C]-bisphenol A in rats: a possible extrapolation to human absorption?	Marquet, F., Payan, J.-P., Beydon, D., Wathier, L., Grandclaude, M.-C. and Ferrari, E.	Archives of Toxicolology	2011	85, 1035-1043	10.1007/s00204-011-0651-z	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Excluded – absorption paper - no relevant data for calculation of exposure from non-food sources NOTE: This manuscript was considered for determination of the absorption of BPA, but excluded for methodological

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Reported data included or excluded from the calculation of the exposure to bisphenol A and reasoning
										reasons
Determination of free and total bisphenol A in human urine to assess daily	Völkel, W., Kiranoglu, M and Fromme, H.	Toxicology Letters	2008	179, 155-162	10.1016/j.toxlet.2008.05.002	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Excluded – biomonitoring data only - no relevant data for calculation of exposure from non-food sources
Viable skin efficiently absorbs and metabolizes bisphenol A	Zalko, D., Jacques, C., Duplan, H., Bruel, S. and Perdu, E.	Chemosphere	2011	82, 424-430	10.1016/j.chemosphere.2010.09.058	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	Not applicable for calculation of absorption	

Table 68: Literature quality table - occurrence in the environment

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
Sensitive gas chromatographic-mass spectrometric method for the determination of phthalate esters, alkylphenols, bisphenol A and their chlorinated derivatives in wastewater samples	Ballesteros, O. Zafra, A. Navalon, A. and Vilchez, J. L.	Journal of Chromatography A	2006	1121:2, 154-162	10.1016/j.chroma.2006.04.014	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination
Determination of bisphenols A and F and their diglycidyl ethers in wastewater and river water by coacervative extraction and liquid chromatography-fluorimetry	Ballesteros-Gomez, A., Ruiz, F. J., Rubio, S. and Perez-Bendito, D.	Analytica Chimica Acta	2007	603:1, 51-59	10.1016/j.aca.2007.09.048	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Multiresidue analytical methods for the ultra-trace quantification of 33 priority substances present in the list of REACH in real water samples	Baugros, J. B., Giroud, B., Dessalces, G., Grenier-Loustalot, M. F. and Cren-Olive, C.	Analytica Chimica Acta	2008	607:2, 191-203	10.1016/j.aca.2007.11.036	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Biologically directed environmental monitoring, fate, and	Campbell, C. G., Borglin,	Chemosphere	2006	65:8, 1265-1280	10.1016/j.chemosphere.200	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
transport of estrogenic endocrine disrupting compounds in water: A review	S. E., Green, F. B., Grayson, A., Wozei, E. and Stringfellow, W. T.				6.08.003					exposure determination
Bisphenol A occurred in Kao-Pin River and its tributaries in Taiwan	Chen, T. C., Shue, M. F., Yeh, Y. L. and Kao, T. J.	Environmental Monitoring and Assessment	2010	161:1-4, 135-145	10.1007/s10661-008-0733-4	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Determination of bisphenol A in water via inhibition of silver nanoparticles-enhanced chemiluminescence	Chen, X., Wang, C., Tan, X. and Wang, J.	Analytica Chimica Acta	2011	689:1, 92-96	10.1016/j.aca.2011.01.031	Industrial wastewater and river water	Not considered	Not considered	Not considered	Excluded - waste water and river water not included in exposure determination
Emerging pollutants in wastewater: a review of the literature	Deblonde, T., Cossu-Leguille, C. and Hartemann, P.	International Journal of Hygiene and Environmental Health	2011	214:6, 442-448	10.1016/j.ijheh.2011.08.002	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
Selective Molecularly Imprinted Polymer Obtained from a Combinatorial Library for the Extraction of Bisphenol A	Martin-Esteban, A. and Tadeo, J. L.	Combinatorial Chemistry and High Throughput Screening	2006	9, 747-751	None given	Not applicable	Not considered	Not considered	Not considered	Excluded - analytical method paper - not relevant for occurrence in the environment
Gas-liquid chromatography-tandem mass spectrometry methodology for the quantitation of estrogenic contaminants in bile of fish exposed to wastewater treatment works effluents and from wild populations	Fenlon, K. A., Johnson, A. C., Tyler, C. R. and Hill, E. M.	Journal of Chromatography A	2010	1217:1, 112-118	10.1016/j.chroma.2009.10.063	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
Bisphenol A exposure, effects, and policy: a wildlife perspective	Flint, S., Markle, T., Thompson, S. and Wallace, E.	Journal of Environmental Management	2012	104, 19-34	10.1016/j.jenvman.2012.03.021	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
A national reconnaissance for pharmaceuticals and other organic wastewater contaminants in the United States--II) untreated drinking water sources	Focazio, M. J., Kolpin, D. W., Barnes, K. K., Furlong, E. T.,	The Science of the Total Environment	2008	402:2-3, 201-216	10.1016/j.scitotenv.2008.02.021	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	Meyer, M. T., Zaugg, S. D., Barber, L. B. and Thurman, M. E.									
Ubiquity of bisphenol A in the atmosphere	Fu, P. and Kawamura, K.	Environmental Pollution	2010	158:10, 3138-3143	10.1016/j.envpol.2010.06.040	Outdoor air	Not considered	Not considered	Not considered	Excluded - outdoor atmosphere not included in exposure determination
On-line solid phase extraction fast liquid chromatography-tandem mass spectrometry for the analysis of bisphenol A and its chlorinated derivatives in water samples	Gallart-Ayala, H., Moyano, E. and Galceran, M. T.	Journal of Chromatography A	2010	1217:21, 3511-3518	10.1016/j.chroma.2010.03.028	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Exposure Analysis of Bisphenol A in Surface Water Systems in North America and Europe	Gary M Klecka, Charles A Staples, Kathryn E Clark, Nelly Van Der Hoeven, David E Thomas,	Environmental Science and Technology	2009	43, 6145-6150	10.1021/es900598e	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	and Steven G Hentges									
Determination of acidic pharmaceuticals and potential endocrine disrupting compounds in wastewaters and spring waters by selective elution and analysis by gas chromatography-mass spectrometry	Gibson, R., Becerril-Bravo, E., Silva-Castro, V. and Jimenez, B.	Journal of Chromatography A	2007	2007 Oct 26;1169(1-2):31-9	10.1016/j.chroma.2007.08.056	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination
A new method for monitoring oestrogens,N-octylphenol, and bisphenol A in wastewater treatment plants by solid-phase extraction–gas chromatography–tandem mass spectrometry	Gómez, María José, Mezcua, Milagros, Martinez, María José, Fernández-Alba, Amadeo R. and Agüera, Ana	International Journal of Environmental Analytical Chemistry	2006	86:1-2, 3-13	10.1080/03067310500247983	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination
Simultaneous determination of hexabromocyclododecane, tetrabromobisphenol A, and related compounds in sewage sludge and sediment samples from Ebro River	Guerra, P., Eljarrat, E. and Barcelo, D.	Analytical and Bioanalytical Chemistry	2010	2010 Aug;397(7):2817-24	10.1007/s00216-010-3670-3	Sewage water	Not considered	Not considered	Not considered	Excluded - sewage water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
basin (Spain)										
Environmental temperature changes uptake rate and Bioconcentration factors of bisphenol a in tadpoles of Rana temporaria <, Vol. 25, No 10, pp. 2804–2808, 2006.pdf>	Honkanen, J. O. and Kukkonen, J. V. K.	Environmental Toxicology and Chemistry	2006	25:10, 2804-2808	None given	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
Bisphenol A (BPA) in China: a review of sources, environmental levels, and potential human health impacts	Huang, Y. Q., Wong, C. K., Zheng, J. S., Bouwman, H., Barra, R., Wahlstrom, B., Neretin, L. and Wong, M. H.	Environment International	2012	2012 Jul;42:91-9	10.1016/j.envint.2011.04.010	Not applicable	Not considered	Not considered	Not considered	Excluded - review paper - not relevant for occurrence in the environment
BPA and environmental estrogen in potable water sources in Enugu municipality, South-East, Nigeria	Ignatius, C. M., Francis, E. E., Emeka, E. N., Elvis, N. S. and	Bulletin of Environmental Contamination and	2010	2010 Nov;85(5): 534-7	10.1007/s00128-010-0111-0	River water and rain water	Not considered	Not considered	Not considered	Excluded - river water and rain water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	Ebele, J. I.	Toxicology								
Direct enrichment and high performance liquid chromatography analysis of ultra-trace Bisphenol A in water samples with narrowly dispersible Bisphenol A imprinted polymeric microspheres column	Jiang, M., Zhang, J. H., Mei, S. R., Shi, Y., Zou, L. J., Zhu, Y. X., Dai, K. and Lu, B.	Journal of Chromatography A	2006	2006 Mar 31;1110(1-2):27-34	10.1016/j.chroma.2006.01.051	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Determination of bisphenol A, bisphenol F and their diglycidyl ethers in environmental water by solid phase extraction using magnetic multiwalled carbon nanotubes followed by GC-MS/MS	Jiao, Yanna, Ding, Li, Fu, Shanliang, Zhu, Shaohua, Li, Hui and Wang, Libing	Analytical Methods	2012	4:1, 291-298	10.1039/c1ay05433c	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Bisphenol A in the aquatic environment and its endocrine-disruptive effects on aquatic organisms	Kang, J. H., Asai, D. and Katayama, Y.	Critical Reviews in Toxicology	2007	2007;37(7):607-25	10.1080/10408440701493103	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
Bisphenol A in the surface water and freshwater snail	Kang, J. H. and Kondo,	Bulletin of Enviro	2006	2006 Jan;76(1):113-8.	10.1007/s00128-005-	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
collected from rivers around a secure landfill	F.	nmental Contamination and Toxicology			0896-4					relevant for occurrence in the environment
Distribution and biodegradation of bisphenol A in water hyacinth	Kang, J. H. and Kondo, F.	Bulletin of Environmental Contamination and Toxicology	2006	2006 Oct;77(4): 500-7.	10.1007/s00128-006-1092-x	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
Liquid phase microextraction with in situ derivatization for measurement of bisphenol A in river water sample by gas chromatography-mass spectrometry	Kawaguchi, M., Ito, R., Endo, N., Okanouchi, N., Sakui, N., Saito, K. and Nakazawa, H.	Journal of Chromatography A	2006	2006 Mar 31;1110(1-2):1-5	10.1016/j.chroma.2006.01.061	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Distribution of pesticides and bisphenol A in sediments collected from rivers adjacent to coral reefs	Kitada, Y., Kawahata, H., Suzuki, A. and	Chemosphere	2008	2008 May;71(11):2082-90	10.1016/j.chemosphere.2008.01.025	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	Oomori, T.									
Pharmaceuticals, hormones and bisphenol A in untreated source and finished drinking water in Ontario, Canada--occurrence and treatment efficiency	Kleyweg t, S., Pileggi, V., Yang, P., Hao, C., Zhao, X., Rocks, C., Thach, S., Cheung, P. and Whitehea d, B.	The Science of the Total Enviro nment	2011	2011 Mar 15;409(8): 1481-8	10.1016/j .scitotenv .2011.01. 010	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Characterization of trace organic contaminants in marine sediment from Yeongil Bay, Korea: 1. Instrumental analyses	Koh, C. H., Khim, J. S., Villeneu ve, D. L., Kannan, K. and Giesy, J. P.	Enviro nmenta l Pollutio n	2006	2006 Jul;142(1): 39-47	10.1016/j .envpol.2 005.09.0 05	Sediment	Not considered	Not considered	Not considered	Excluded - sediment not included in exposure determination
Enzyme-linked immunosorbent assay for bisphenol A: Assay optimization and its application for surface water analysis	Krapivin, A. S., Samsono va, J. V., Uskova, N. A., Ivanova,	Toxicol ogical and Enviro nmenta l Chemis	2007	89:1, 161-172	10.1080/ 0277224 0600954 246	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	N. L. and Egorov, Al. M.	try								
Development and characterization of an immunoaffinity monolith for selective on-line extraction of bisphenol A from environmental water samples	Li, L., Wang, J., Zhou, S. and Zhao, M.	Analytica Chimica Acta	2008	2008 Jul 14;620(1-2):1-7	10.1016/j.aca.2008.05.036	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Determination of Bisphenol A in Landfill Leachate by Solid Phase Microextraction with Headspace Derivatization and Gas Chromatography-Mass Spectrophotometry	Li, Xiangli, Lin, Li, Zou, Shichun, Lan, Chongyu and Luan, Tiangang	Chinese Journal of Analtical Chemistry	2006	34:3, 325-328	10.1016/s1872-2040(06)60018-2	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Dispersive liquid–liquid microextraction based on ionic liquid in combination with high-performance liquid chromatography for the determination of bisphenol A in water	Li, Yu and Liu, Jianlin	International Journal of Environmental Analytical Chemistry	2010	90:11, 880-890	10.1080/03067310903045455	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
High sensitivity detection of bisphenol A	Liu, X. Y.,	Analytica	2006	2006 Sep 18;578(1):	10.1016/j.aca.2006	Surface water	Not considered	Not considered	Not considered	Excluded - surface water

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
using liposome chromatography	Nakamura, C., Tanimoto, I., Miyake, S., Nakamura, N., Hirano, T. and Miyake, J.	Chimica Acta		43-9	.07.016					not included in exposure determination
Passive sampling and stir bar sorptive extraction for the determination of endocrine-disrupting compounds in water by GC-MS	Magi, E., Di Carro, M. and Liscio, C.	Analytical and Bioanalytical Chemistry	2010	2010 Jun;397(3):1335-45	10.1007/s00216-010-3656-1	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Physico-chemical pre-treatment and biotransformation of wastewater and wastewater sludge--fate of bisphenol A	Mohapatra, D. P., Brar, S. K., Tyagi, R. D. and Surampalli, R. Y.	Chemosphere	2010	2010 Feb;78(8):923-41	10.1016/j.chemosphere.2009.12.053	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination
Pharmaceutical chemicals and endocrine disrupters in municipal wastewater in Tokyo and their removal during activated sludge treatment	Nakada, N., Tanishima, T., Shinohara, H., Kiri, K.	Water Research	2006	2006 Oct;40(17):3297-303	10.1016/j.watres.2006.06.039	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	and Takada, H.									
A critical evaluation of the environmental risk assessment for plasticizers in the freshwater environment in Europe, with special emphasis on bisphenol A and endocrine disruption	Oehlmann, J., Oetken, M. and Schulte-Oehlmann, U.	Environmental Research	2008	108:2, 140-149	10.1016/j.envres.2008.07.016	Not applicable	Not considered	Not considered	Not considered	Excluded - environmental risk paper - not relevant for occurrence in the environment
Determination of phenolic compounds in river water with on-line coupling bisphenol A imprinted monolithic precolumn with high performance liquid chromatography	Ou, J., Hu, L., Hu, L., Li, X. and Zou, H.	Talanta	2006	2006 Jun 15;69(4):1001-6.	10.1016/j.talanta.2005.12.003	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Simultaneous determination of endocrine-disrupting phenols and steroid estrogens in sediment by gas chromatography-mass spectrometry	Peng, X., Wang, Z., Yang, C., Chen, F. and Mai, B.	Journal of Chromatography A	2006	2006 May 26;1116(1-2):51-6	10.1016/j.chroma.2006.03.017	Sediment	Not considered	Not considered	Not considered	Excluded - sediment not included in exposure determination
Multiresidue analysis of acidic and polar organic contaminants in water samples by stir-bar sorptive extraction-liquid desorption-gas chromatography-mass spectrometry	Quintana, J. B., Rodil, R., Muniategui-Lorenzo, S.,	Journal of Chromatography A	2007	2007 Dec 7;1174(1-2):27-39	10.1016/j.chroma.2007.07.088	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
	Lopez-Mahia, P. and Prada-Rodriguez, D.									
Vesicular coacervative extraction of bisphenols and their diglycidyl ethers from sewage and river water	Ruiz, F. J., Rubio, S. and Perez-Bendito, D.	Journal of Chromatography A	2007	2007 Sep 7;1163(1-2):269-76	10.1016/j.chroma.2007.06.024	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Determination of alkylphenols and bisphenol A in seawater samples by dispersive liquid-liquid microextraction and liquid chromatography tandem mass spectrometry for compliance with environmental quality standards (Directive 2008/105/EC)	Salgueiro-Gonzalez, N., Concha-Grana, E., Turnes-Carou, I., Muniategui-Lorenzo, S., Lopez-Mahia, P. and Prada-Rodriguez, D.	Journal of Chromatography A	2012	2012 Feb 3;1223:1-8	10.1016/j.chroma.2011.12.011	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Simultaneous determination of bisphenol A and its	Sambe, H., Hoshina,	Journal of Chrom	2006	2006 Nov 17;1134(1-2):16-23	10.1016/j.chroma.2006.08.	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
halogenated derivatives in river water by combination of isotope imprinting and liquid chromatography-mass spectrometry	K., Hosoya, K. and Haginaka, J.	atography A			072					exposure determination
A direct Capillary Liquid Chromatography with electrochemical detection method for determination of phenols in water samples	Segovia-Martinez, L., Moliner-Martinez, Y. and Campins-Falco, P.	Journal of Chromatography A	2010	2010 Dec 10;1217(50):7926-30	10.1016/j.chroma.2010.10.078	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Analysis of endocrine disrupting compounds in wastewater and drinking water treatment plants at the nanogram per litre level	Stavrakakis, C., Colin, R., Hequet, V., Faur, C. and Le Cloirec, P.	Environmental Technology	2008	2008 Mar;29(3):279-86.	10.1080/09593330802099452	Waste water	Not considered	Not considered	Not considered	Excluded -waste water not included in exposure determination
Human health risk on environmental exposure to Bisphenol-A: a review	Tsai, W. T.	Journal of Environmental Science and Health. Part C	2006	2006;24(2):225-55.	10.1080/10590500600936482	Not applicable	Not considered	Not considered	Not considered	Excluded - review paper - not relevant for occurrence in the environment
Investigating the	Vigano,	Archiv	2006	2006	10.1007/	Surface	Not	Not considered	Not	Excluded -

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
estrogenic risk along the river Po and its intermediate section	L., Mandich, A., Benfenati, E., Bertolotti, R., Bottero, S., Porazzi, E. and Agradi, E.	es of Environmental Contamination and Toxicology		Nov;51(4): 641-51.	s00244-005-0129-1	water	considered		considered	surface water not included in exposure determination
Selective determination of bisphenol A (BPA) in water by a reversible fluorescence sensor using pyrene/dimethyl ß-cyclodextrin complex	Wang, X., Zeng, H., Zhao, L. and Lin, J.-M.	Analytica Chimica Acta	2006	556:2, 313-318	10.1016/j.aca.2005.09.060	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Analysis of estrogens in environmental waters using polymer monolith in-polyether ether ketone tube solid-phase microextraction combined with high-performance liquid chromatography	Wen, Y., Zhou, B. S., Xu, Y., Jin, S. W. and Feng, Y. Q.	Journal of Chromatography A	2006	2006 Nov 10;1133(1-2):21-8	10.1016/j.chroma.2006.08.049	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Determination of some endocrine disrupter chemicals in urban wastewater samples using liquid chromatography–mass	Zafra-Gómez, Alberto, Ballesteros, Oscar,	Microchemical Journal	2008	88:1, 87-94	10.1016/j.microc.2007.10.003	Waste water	Not considered	Not considered	Not considered	Excluded - waste water not included in exposure determination

Title	Authors	Journal	Year	Reference (volume: issue, page number)	DOI	Category	Sample description	Country of origin of samples	Method description and quality parameters	Included/excluded and reasoning
spectrometry	Navalón, Alberto and Vílchez, José Luís									
MCX based solid phase extraction combined with liquid chromatography tandem mass spectrometry for the simultaneous determination of 31 endocrine-disrupting compounds in surface water of Shanghai	Zhang, H. C., Yu, X. J., Yang, W. C., Peng, J. F., Xu, T., Yin, D. Q. and Hu, X. L.	Journal of Chromatography B	2011	2011 Oct 15;879(28):2998-3004	10.1016/j.jchromb.2011.08.036	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination
Endocrine disrupting compounds in the atmosphere of the urban area of Thessaloniki, Greece	M. Salapasidou, C. Samara, D. Voutsa*	Atmospheric Environment	2011	45, 3720-3729	10.1016/j.atmosenv.2011.04.025	Atmosphere	Not considered	Not considered	Not considered	Excluded - outdoor atmosphere not included in exposure determination
Optimisation of derivatisation for the analysis of estrogenic compounds in water by solid-phase extraction gas chromatography-mass spectrometry	Zhang, Z. L., Hibberd, A. and Zhou, J. L.	Analytica Chimica Acta	2006	2006 Sep 1;577(1):52-61	10.1016/j.aca.2006.06.029	Surface water	Not considered	Not considered	Not considered	Excluded - surface water not included in exposure determination

5931