Food Research International 42 (2009) 1410–1414

Contents lists available at ScienceDirect

Food Research International

journal homepage: www.elsevier.com/locate/foodres

Effect of detergents in the release of bisphenol A from polycarbonate baby bottles

J. Maia [a], J.M. Cruz [b], R. Sendón [a], J. Bustos [c], J.J. Sanchez [c], P. Paseiro [a,*]

[a] Department of Analytical Chemistry, Nutrition and Bromatology, Faculty of Pharmacy, University of Santiago de Compostela, 15782-Santiago de Compostela, Spain
[b] Chemical Engineering Department, University of Vigo, E.T.S.E.I., 36200 Vigo (Pontevedra), Spain
[c] National Center for Food, Spanish Agency for Food Safety and Nutrition, 28.220 Majadahonda (Madrid), Spain

ARTICLE INFO

Article history:
Received 4 May 2009
Accepted 8 July 2009

Keywords:
Bisphenol A
Detergent
Polycarbonate
Depolymerization
Migration

ABSTRACT

Bisphenol A (BPA) is a monomer crucial for the production of polycarbonate (PC). Recently, it has been verified that BPA is able to migrate from PC baby bottles into food simulants and numerous studies indicate that BPA may affect human health.

In this work, five different detergents and bleach were tested to verify if they were able increasing the BPA release from the PC. High performance liquid chromatography (HPLC) with fluorescence detection (FLD) and gas chromatography coupled to mass spectrometry (GC–MS) were used to quantify and identify BPA. Of all detergents tested, only with one was not detected a BPA concentration higher than the control. In the worst case, BPA levels detected were about 500 times higher than the control and the concentration kept high even after rinsing the PC samples three times. In the case of bleach, while it was in contact with the PC, the BPA released was not detected.

1. Introduction

Polycarbonate (Fig. 1) is a thermoplastic polymer with high transparency, low weight and high heat and impact resistance. Due to these characteristics and the fact that it can be easily worked, PC has been widely used. PC may be found at compact disks, drinking bottles, eyeglasses lenses, mobile phones, plastic food containers, houseware, as a replacement of glass in several products, such as baby feeding bottles, and many other products (Nérin, Fernández, Domeño, & Salafranca, 2003; Wong, Leo, & Seah, 2005).

The key building block of PC is bisphenol A (BPA; 2,2′-bis(4-hydroxyphenyl)propane; CAS No. 80-05-7) (Fig. 1). This monomer is one of the highest production-volume chemical in the world (Lang et al., 2008) and it is present in quite a lot of products that people are in contact with every day. In 2003, more than 2 million metric tons were produced globally and its demand increases up to 6–10% every year. The main application of BPA production is for the manufacture of epoxy resins (Paseiro Losada, López Mahía, Vázquez Odériz, Simal Lozano, & Simal Gándara, 1991; Simal Gándara et al., 1992) and, especially, of PC (21% and 72%, respectively) (Chapin et al., 2007). BPA can also be used as additive in PVC films (López-Cervantes & Paseiro-Losada, 2003).

Several studies have already confirmed that BPA release from PC baby bottles takes place (Biles, McNeal, Begley, & Hollifield, 1997; Brede, Fjeldal, Skjevrak, & Herikstad, 2003; Maragou, Makri, Lampi, Thomaidis, & Koupparis, 2007). This release is a consequence of BPA free monomers migration (Paseiro Losada, Paz Abuín, Vázquez Odériz, Simal Lozano, & Simal Gándara, 1991), but this is not the main reason, because the residual level of the free monomer and the BPA diffusion from the PC to the beverage is very low (Paseiro Losada, Simal Lozano, Paz Abuín, López Mahía, & Simal Gándara, 1993; Simal Gándara, Paz Abuín, Paseiro Losada, & Simal Lozano, 1993). Rather than that, most of the BPA released results from PC degradation (Bierdermann-Brem & Grob, 2009). Accordingly to some authors, BPA may have an effect on the human health.

Data obtained from *in vitro* assays shown that BPA has weak estrogenic activity and there is some concern about how it may affect the endocrine system by mimicking estradiol. This may happen even in the presence of very low BPA doses (Krishnan, Stathis, Permuth, Tokes, & Feldman, 1993; Laws, Carey, Ferrell, Bodman, & Cooper, 2000). There are also *in vivo* studies, made in mammals (mainly in mice and rats) that point to other adverse effects caused by BPA. Some examples can be cited: increase in hormonally mediated cancers, such as prostate and breast cancer; abnormalities in reproductive organs; a decline in semen quality; early sexual maturation in females; metabolic disorders including insulin resistant diabetes and obesity; and neurobehavioral problems such as autism and attention deficit hyperactivity disorder (Farabollini, Porrini, & Dessì-Fulgheri, 1999; Gupta, 2000; Howdeshell, Hotchkiss, Thayer, Vandenbergh, & Vom Saal, 1999; Timms et al., 2005; Vom Saal et al., 2007). These data, plus the fact that

* Corresponding author. Tel.: +34 981 598450; fax: +34 981 594912.
E-mail address: perfecto.paseiro@usc.es (P. Paseiro).

0963-9969/$ - see front matter © 2009 Elsevier Ltd. All rights reserved.
doi:10.1016/j.foodres.2009.07.003

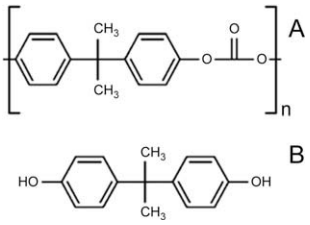

Fig. 1. Molecular structure of PC (A) and BPA (B).

detectable levels of BPA are present in the urine of more than 90% of the US population (Calafat et al., 2005), makes the BPA an alarming issue.

However, BPA safety aspects are increasingly controversial. Agencies such as the US Food and Drugs Administration (FDA), European Food Safety Authority (EFSA) and Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE), supported by the plastic industry, argue that, whenever BPA is used accordingly to the existing legislation, it does not represent any risk to human health when used in food contact materials (Aguilar et al., 2008; Philbert et al., 2008; Scientific Committee on Toxicity, Ecotoxicity and the Environment, 2002). On the other hand, the Health Canada chose to take a preventive approach and concluded that BPA should be considered as a substance that may constitute a danger to human life or health (Environment Canada, 2008). For this reason, Canada has announced the preparation of regulations to ban the importation, advertising and sale of PC baby bottles containing BPA. Measures to reduce the amount of BPA that is released into the environment will also be taken. Several US states, such as California, are already preparing legislation to take similar provisions.

To wash baby bottles, as well as other childcare products, it is usual to use ordinary dishwashing detergents and hot water. Depending on people habits, the baby bottles washing procedure can include hand wash, machine wash, brushing, soaking in hot soapy water, etc.

The aim of this work was to test the effect of different detergents on PC baby bottles, to see if commercially available detergents may increase the BPA release due to a depolymerization effect on the material. To do this, migration tests, at controlled times and temperatures, were performed by incubating PC samples with several detergent solutions. The PC samples were then rinsed with distilled water and migration tests were repeated, this time with distilled water. This last procedure was repeated for three times in order to check if the BPA release would continue after the removal of the residual detergent. After each incubation, the solutions were analyzed directly by HPLC–FLD.

2. Experimental

2.1. Chemicals and reagents

Bisphenol A, 99+% (Aldrich-Chemie, Steinheim, Germany); water obtained using Milli-Q apparatus from Millipore Ireland B.V. (Carringtwohill, Ireland); sodium hypochlorite, sol. reagent grade 10–13% (Sigma–Aldrich, Steinheim, Germany).

2.2. Samples

Baby bottles made of PC, bought in a local supermarket, were chosen to be our study samples. These baby bottles were produced in China.

The samples were prepared by cutting each baby bottle in several pieces with an area of 10 cm^2. The cuts were made vertically seeking that the pieces were representative of the different bottle zones. For each test, three pieces were used and each one was cut in half in order to fit the headspace vials where they were incubated.

The detergents were also bought in a local supermarket and two of them were hand dishwashing detergents (detergents A and B), while the other three were machine dishwashing detergents (detergents C, D and E). The detergent concentration used was 10 g l^{-1}. Bleach was also tested. Instead of using a commercial brand we used a sodium hypochlorite solution, at a concentration of 1 g l^{-1}. These concentrations were chosen because they seem to be close to the real use concentrations.

In parallel with samples, a control was prepared by subjecting a PC sample to the same conditions but without using any detergent or bleach solution. In this case only distilled water was used all the time.

2.3. Apparatus and conditions

High performance liquid chromatography with fluorescence detection (HPLC–FLD) was performed using a Hewlett–Packard HP 1100 chromatograph equipped with a diode array detector and a fluorescence scanning detector arranged in series. The conditions used for determination of BPA by HPLC–FLD are listed in Table 1.

Gas chromatography with mass spectrometry (GC–MS) was carried out using a THERMO Finnigan TraceGC ultra chromatograph with a TraceDSQ mass detector and a TriPlus AS automatic injector. The conditions used for identification of BPA by GC–MS are listed in Table 2.

The oven used for the incubation tests was a Memmert, model ULE 400.

2.4. Identification of BPA by GC–MS

Successive dilutions of a standard 1000 mg l^{-1} BPA solution in ethanol were analyzed in full-scan and single ion monitoring (SIM) modes, according the conditions mentioned in table 2, in order to determinate the limit of detection of BPA by GC–MS. The limit of detection was achieved when the BPA peak was three times higher than the noise level.

Table 1
Conditions and instrument settings used for determination of BPA by HPLC–FLD.

Pump	Hewlett–Packard Quaternary HP 1100 pump		
Injection volume	50 µl		
Column temperature	25 °C		
Detector	Fluorescence (FL) HP 1100 Scan excitation range 200–280 nm Ultraviolet HP 1100 Scan range 190–400		
Degasser	HP 1100 vacuum degasser		
Wavelength	Fluorescence: excitation 225 nm, emission 305 nm Ultraviolet: Sig. 225 nm, Ref. 360 nm		
Column	Kromasil C18 25 × 0.36 cm I.D., 5 µm particle size		
Mobile phase	A: Milli-Q water B: Acetonitrile		
Flow	0.5 ml min^{-1}		
Gradient	Time (min)	A (%)	B
	0.00	70.00	30.00
	2.00	70.00	30.00
	30.00	20.00	80.00
Software	HP Chem Station		

Table 2
Conditions and instrument settings used for identification of BPA by GC–MS.

Constant flow	0.8 ml min^{-1}
Carrier gas	He
Column	
Dimensions	30 m × 0.250 mm
Film surface	1 μm
Liquid phase	DB-5MS
Injector temperature	225
Split mode	1:30
Injection volume	1.0 μl
Column temperature program	Isocratic (250 °C for 20 min)
Mass spectrometer, THERMO instrument Finnigan TraceDSQ	
Interphase temperature	200 °C
Electron energy	70 eV
Electron multiplier	1504 V
Full-scan	*m/z* 50–400
SIM mode	*m/z* 213 (base peak), 228
Electron impact	Positive mode
Spectrum library	Wiley
Software	XCalibur home page version 1.4 SRI, Windows XP

2.5. HPLC–FLD method validation

Calibration of the HPLC was made by injecting, in triplicate, seven BPA standard solutions in water. The solution concentrations were 10, 5, 1, 0.5, 0.1, 0.05, 0.01 mg l^{-1}. The calibration line of concentrations versus chromatographic peak areas was obtained by linear regression.

The limit of detection was calculated by successive dilutions of a BPA standard solution until the BPA peak-to-noise ratio was 3:1.

Precision measurement was estimated by injecting 10 times a 0.1 mg l^{-1} BPA solution in water and calculating the relative standard deviation.

2.6. Study of the effect of detergents on Polycarbonate

Baby bottles are always washed up and sterilized before being used. The following experiment was designed under extreme conditions to know the effect of detergents and bleach on PC.

Three baby bottle pieces (total area 30 cm^2) were immersed in a 15 ml solution of detergent or bleach in distilled water. The concentration of these solutions was 10 g l^{-1} for the detergents and 1 g l^{-1} for the bleach.

After being immersed in the solutions, the samples were incubated at 120 °C, for 1 h. Once the incubation was over, samples were allowed to cool down at room temperature for 30 min and 0.6 ml of the solution was taken for HPLC analysis in order to evaluate if the detergent was able to affect the PC during the washing and soaking procedure in very hot water. Then the samples were left further at 25 °C for 120 h (5 days) in the same solution. Once again, a small volume of solution was taken for HPLC analysis. The purpose was to see if the depolymerization process caused by detergent would occur, even at low temperatures.

To find out if the release of BPA would stop after the detergent was removed, the PC pieces were washed in distilled water and immersed again, but this time in distilled water, for 1 h at 120 °C. Next, they were left cooling down at room temperature for 30 min, an aliquot of the water was taken for HPLC analysis and the PC pieces were washed again in distilled water. This process was repeated once a day, for two more days. The objective was to verify if the rinsing procedure was enough to stop or to diminish the eventual BPA release.

3. Results and discussion

3.1. Identification of BPA by GC–MS

With the GC–MS analytical method described in Table 2, BPA elution time was 10.52 min (Fig. 2). BPA was identified by comparing the mass spectra of the peak obtained in the samples with that of a BPA standard solution and also comparing against the XCalibur library.

The limit of detection in full-scan mode was 50 mg l^{-1}, while in SIM mode [m/z = 213, corresponding to the demethylated fragment of BPA, and m/z = 228, corresponding to BPA (López-Cervantes & Paseiro-Losada, 2003)] was 0.5 mg l^{-1}.

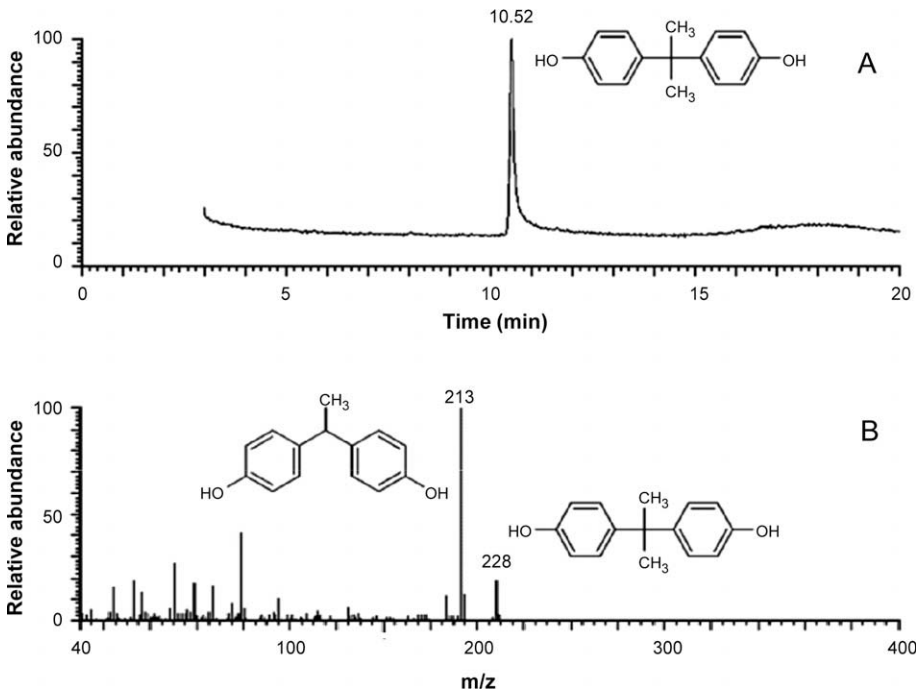

Fig. 2. GC–MS chromatogram of an acetonitrile extract of a PC baby bottle sample (A) and a mass spectrum of the 10.52 min peak of the chromatogram (B).

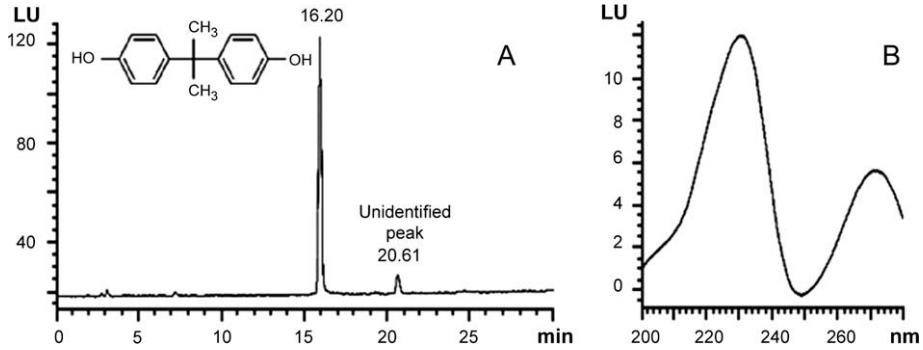

Fig. 3. HPLC chromatogram obtained from detergent B first measure, after being 1:100 dilution (A) and fluorescence excitation spectrum (λ_{em} = 305 nm) of the BPA peak (B).

Table 3
Migration of BPA from PC baby bottle samples into the detergent solutions, determined by HPLC–FLD.

Solutions	Conc. ($g\,l^{-1}$)	Type of detergent	No. of exp. (n)	Concentration of BPA ($mg\,l^{-1}$)				
				120 °C, 1 h	+25 °C, 120 h	+120 °C, 1 h (1st rising with distilled water)	+120 °C, 1 h (2nd rising with distilled water)	+120 °C, 1 h (3rd rising with distilled water)
Dist. water		–	3	0.109	0.086	0.024	0.030	0.018
Det. A	10	Hand wash	3	0.022	0.020	0.012	0.027	0.018
Det. B	10	Hand wash	3	54.8	67.0	40.4	44.6	31.0
Det. C	10	Machine wash	3	3.03	2.65	0.111	0.065	0.030
Det. D	10	Machine wash	3	0.809	0.645	0.016	0.020	0.014
Det. E	10	Machine wash	3	30.9	33.3	0.037	0.050	0.041
Bleach	1	–	3	n.d.	n.d.	0.045	0.043	0.020

n.d. – not detected.

3.2. HPLC method performance

The calibration line obtained by injecting seven standard solutions with a concentration ranging 0.01–10 $mg\,l^{-1}$ showed a good linearity correlation ($R^2 > 0.9999$) and is represented by the following equation:

$$y = 2399.4x - 21.385$$

The limit of detection determined was 5.0 $\mu g\,l^{-1}$ and was obtained by running successive dilutions of a BPA stock solution until the BPA peak-to-noise ratio was 3:1.

The estimated relative standard deviation was 1.53%, after injecting 10 times a 0.1 $mg\,l^{-1}$ BPA solution.

With the HPLC analytical method described in Table 1, BPA elution time was 16.20 min (Fig. 3).

3.3. BPA release from baby bottles

The results values of the migration of BPA from PC baby bottles into the different solutions assayed in this work are shown in Table 3.

Migration of BPA was detected in the control sample, when distilled water was used (without any detergent or bleach). BPA concentration found was 0.109 $mg\,l^{-1}$, after the first incubation period.

Comparing all detergents, detergent A is the only one that seems to decrease the BPA release from PC, being BPA concentrations 4–5 times lower than those found in the distilled water, during the firsts two measurements. For this detergent, after rinsing the PC samples, the levels of BPA became close to that found in the control. In the case of bleach, although higher pH values should increase the BPA release (Krishnan et al., 1993), the levels were not detectable while the PC was in contact with the sodium hypochlorite solution (first two measurements). However, once the sodium hypochlorite is removed from the samples by rinsing, and the samples are incubated in distilled water only, BPA release is observed.

Detergents B, C, D and E showed a higher release of BPA than that observed in the control. BPA levels were approximately 500, 28, 7 and 283 times higher, respectively, than the concentration found in the control in the first measurement. After the PC samples were rinsed, BPA levels diminished to levels comparable to those found in the control either after the first rinsing (for detergents D and E) or the third rinsing (for detergent C). For detergent B, even though there was a visible decrease of the BPA levels during the rinsing procedures, significant concentrations (31.0–44.6 $mg\,l^{-1}$) could still be detected. These concentrations were 1680–1860 times higher than those found in the control samples.

4. Conclusions

BPA safety issue is presently a controversial issue and this monomer presence in a large percentage of the population indicates us that better knowledge about BPA origin is necessary.

These results also demonstrate that the BPA detected in the detergent solutions (except for the detergent A solution) does not result from diffusion, because BPA concentrations are much higher than that found in the control. Instead, BPA is primarily a product from a PC degradation process originated by the contact between detergent and PC at high temperatures.

In the current study was demonstrated that dishwashing detergents may increase the BPA release while the detergent is in contact with the PC. The BPA concentration decrease immediately near to control levels after rinsing the PC samples (0.018–0.041 $mg\,l^{-1}$, after the third rinsing), except for one of the cases tested (detergent B), where, although there is a visible decrease on BPA concentration, the levels detected are still very high (approximately 31.0 $mg\,l^{-1}$, after the third rising).

Further studies are needed in order to better understanding of the BPA release from PC mechanism that we are facing, and also would be need to perform more tests in the conditions required by the European Council Directive 82/711/EEC (European Commission, 1982), simulating real-use conditions.

Acknowledgements

The study was financially supported by the Ministerio de Ciencia e Innovación (Project Ref. AGL2008-04146).

Authors are grateful to the "Angeles Alvariño" Program financed by "Consellería de Innovación e Industria, Xunta de Galicia" for the postdoctoctoral contract of R. Sendón. The authors are also grateful to Ms. Patricia Blanco Carro, Ms. Cristina Casal Romero and Mr. Gonzalo Hermelo Vidal for their excellent technical assistance.

References

Aguilar, F. et al. (2008). Scientific opinion of the panel on food additives, flavourings, processing aids and materials in contact with food (AFC) on a request from the commission on the toxicokinetics of bisphenol A. *The EFSA Journal, 759*, 1–10.

Bierdermann-Brem, S., & Grob, K. (2009). Release of bisphenol A from polycarbonate baby bottles: Water hardness as the most relevant factor. *European Food Research and Technology, 228*(5), 679–684.

Biles, J. E., McNeal, T. P., Begley, T. H., & Hollifield, H. C. (1997). Determination of bisphenol-A in reusable polycarbonate food-contact plastics and migration to food-simulating liquids. *Journal of Agricultural and Food Chemistry, 45*(9), 3541–3544.

Brede, C., Fjeldal, P., Skjevrak, I., & Herikstad, H. (2003). Increased migration levels of bisphenol A from polycarbonate baby bottles after dishwashing, boiling and brushing. *Food Additives and Contaminants, 20*(7), 684–689.

Calafat, A. M., Kuklenyik, Z., Reidy, J. A., Caudill, S. P., Ekong, J., & Needham, L. L. (2005). Urinary concentrations of bisphenol A and 4-nonylphenol in a human reference population. *Environmental Health Perspectives, 113*(4), 391–395.

Chapin, R. E. et al. (2007). NTP-CERHR expert panel report on the reproductive and developmental toxicity of bisphenol A, 26 November 2007. <http://www.cerhr.niehs.nih.gov/chemicals/bisphenol/BPAFinalEPVF112607.pdf> Accessed 08.12.2008.

Environment Canada, Health Canada (2008). Screening assessment for the challenge phenol 4,4′-(1-methylethylidene)bis(bisphenol A), October, 2008. <http://www.ec.gc.ca/substances/ese/eng/challenge/batch2/batch2_80-05-7_en.pdf> Accessed 21.11.2008.

European Commission, Commission Directive 82/711/EEC of 18 October 1982. Laying down the basic rules necessary for testing migration of the constituents of plastic materials and articles intended to come into contact with foodstuffs. *Official Journal of the European Communities, L 297*.

Farabollini, F., Porrini, S., & Dessì-Fulgheri, F. (1999). Perinatal exposure to the estrogenic pollutant bisphenol A affects behavior in male and female rats. *Pharmacology Biochemistry and Behavior, 64*(4), 687–694.

Gupta, C. (2000). Reproductive malformation of the male offspring following maternal exposure to estrogenic chemicals. *Proceedings of the Society for Experimental Biology and Medicine, 224*(2), 61–68.

Howdeshell, K. L., Hotchkiss, A. K., Thayer, K. A., Vandenbergh, J. G., & Vom Saal, F. S. (1999). Environmental toxins: Exposure to bisphenol A advances puberty. *Nature, 401*(6755), 763–764.

Krishnan, A. V., Stathis, P., Permuth, S. F., Tokes, L., & Feldman, D. (1993). Bisphenol-A: An estrogenic substance is released from polycarbonate flasks during autoclaving. *Endocrinology, 132*(6), 2279–2286.

Lang, I. A., Galloway, T. S., Scarlett, A., Henley, W. E., Depledge, M., Walace, R. B., et al. (2008). Association of urinary bisphenol A concentration with medical disorders and laboratory abnormalities in adults. *JAMA – Journal of the American Medical Association, 300*(11), 1303–1310.

Laws, S. C., Carey, S. A., Ferrell, J. M., Bodman, G. J., & Cooper, R. L. (2000). Estrogenic activity of octylphenol, nonylphenol, bisphenol A and methoxychlor in rats. *Toxicological Sciences, 54*(1), 154–167.

López-Cervantes, J., & Paseiro-Losada, P. (2003). Determination of bisphenol A in, and its migration from, PVC stretch film used for food packaging. *Food Additives and Contaminants, 20*(6), 596–606.

Maragou, N. C., Makri, A., Lampi, E. N., Thomaidis, N. S., & Koupparis, M. A. (2007). Migration of bisphenol A from polycarbonate baby bottles under real use conditions. *Food Additives and Contaminants, 25*(3), 373–383.

Nérin, C., Fernández, C., Domeño, C., & Salafranca, J. (2003). Determination of potential migrants in polycarbonate containers used for microwave ovens by high-performance liquid chromatography with ultraviolet and fluorescence detection. *Journal of Agricultural and Food Chemistry, 51*(19), 5647–5653.

Paseiro Losada, P., López Mahía, P., Vázquez Odériz, L., Simal Lozano, J., & Simal Gándara, J. (1991). Sensitive and rapid reversed phase high performance liquid chromatography–fluorescence method for determining bisphenol A diglycidyl ether in aqueous based food simulants. *Journal of the Association of Official Analytical Chemists, 74*(6), 925–928.

Paseiro Losada, P., Paz Abuín, S., Vázquez Odériz, L., Simal Lozano, J., & Simal Gándara, J. (1991). Quality control of cured epoxy resins: Determination of residual free monomers (m-xylylenediamine and bisphenol A diglycidyl ether) in the finished product. *Journal of Chromatography, 585*(1), 75–81.

Paseiro Losada, P., Simal Lozano, J., Paz Abuín, S., López Mahía, P., & Simal Gándara, J. (1993). Kinetics of the hydrolysis of bisphenol A diglycidyl ether (BADGE) in water-based food simulants. Implications for legislation on the migration of BADGE-type epoxy resins into foodstuffs. *Fresenius' Journal of Analytical Chemistry, 345*(7), 527–532.

Philbert, M. A. et al. (2008). Scientific peer-review of the draft assessment of bisphenol A for use in food contact applications (October 31, 2008). <http://www.fda.gov/OHRMS/DOCKETS/ac/08/briefing/2008-386b1-5.pdf> Accessed 21.11.08.

Scientific Committee on Toxicity, Ecotoxicity and the Environment (2002). Opinion on the result of the risk assessment of: Bisphenol A, Human Health Part (May 22, 2002). <http://ec.europa.eu/health/ph_risk/committees/sct/documents/out156_en.pdf> Accessed 26.11.08.

Simal Gándara, J., Paseiro Losada, P., López Mahía, P., Simal Lozano, J., & Paz Abuín, S. (1992). RP-HPLC–TSP-MS of epoxy resins bisphenol A diglycidyl ether type. *Journal of Chromatographic Science, 30*(1), 11–16.

Simal Gándara, J., Paz Abuín, S., Paseiro Losada, P., & Simal Lozano, J. (1993). Determination of bisphenols A and F in non-cured epoxy resins by RP-HPLC–Fluorescence techniques. *Journal of Chromatographic Science, 31*, 450–454.

Timms, B. G., Howdeshell, K. L., Barton, L., Bradley, S., Richter, C. A., & Vom Saal, F. S. (2005). Estrogenic chemicals in plastic and oral contraceptives disrupt development of the fetal mouse prostate and urethra. *Proceedings of the National Academy of Sciences of the United States of America, 102*(19), 7014–7019.

Vom Saal, F. S. et al. (2007). Chapel Hill bisphenol A expert panel consensus statement: Integration of mechanisms, effects in animals and potential to impact human health at current levels of exposure. *Reproductive Toxicology, 24*(2), 131–138.

Wong, K. O., Leo, L. W., & Seah, H. L. (2005). Dietary exposure assessment of infants to bisphenol A from the use of polycarbonate baby milk bottles. *Food Additives and Contaminants, 22*(3), 280–288.