Heliyon

ELSEVIER

Atmospheric levels of BPA associated with particulate matter in an urban environment

Natalia Soledad Graziani, Hebe Carreras, Eduardo Wannaz[*]

Instituto Multidisciplinario de Biología Vegetal, CONICET and FCEFyN, Universidad Nacional de Córdoba, Av. Vélez Sarsfield 1611, X5016 GCA Córdoba, Argentina

[*] Corresponding author.

E-mail addresses: eduardo.wannaz@unc.edu.ar, eduwannaz@gmail.com (E. Wannaz).

Abstract

Bisphenol A (BPA) is an Endocrine disruptor compound (EDC), capable of interfering with hormone related pathways and cause adverse effects, for example, in the reproductive system in wildlife and humans. As it has multiple applications and uses, mainly in the manufacturing of plastic and resin products, increasing amounts are being released into the environment with resulting enhanced exposure pathways. Since negative effects have been demonstrated even at low doses, it has been increasingly studied. However, these investigations have focused on BPA leaching from food cans, drinks and other consumer products as the oral exposure route is believed to be the most important. Consequently, the potential hazards of dermal or inhalation exposure resulting from the current levels of BPA in the atmosphere are poorly understand. Of the few studies reporting BPA occurrence in air, none of these were carried out in South American countries. Thus, the aim the present investigation was to assess BPA current levels in particulate matter in an urban environment (Córdoba, Argentina) and to analyze its spatial-temporal trends and the influence of meteorological parameters using a medium volume air sampler. Our results suggested a spatial trend that decreased according to the distance to emission sources in the order: Industrial area > City Centre > University campus, whose levels were comparable to other urban cities and with

the temporal trend showing a higher prevalence of the compound in the colder months. Wind speed (WS), temperature (T), atmospheric pressure (AP) and relative humidity (RL) were the most influenced variables. However daily values are unpredictable and depend mainly on the emission sources and punctual events that release BPA into the atmosphere.

No risk factor could be quantified since there is no reference value for this exposure route; Nevertheless, this study represents the first approach and provides data about this emergent pollutant in and Argentinean city, thus contributing to global BPA studies.

Keywords: Atmospheric science, Environmental science, Health sciences

1. Introduction

Natural or synthetic endocrine disruptor compounds (EDCs) are capable of altering endocrine functions through the imitation or blocking of endogenous hormones (Schug et al., 2011). Since humans are constantly exposed to this type of environmental chemicals, they have received increasing attention becoming a public concern. For example, some studies have shown a poorer ovarian response in humans, a reduced number of mature oocytes and normally fertilized oocytes, and a decreased probability of fertilization with increased serum BPA in Asian women and men, as well as a correlation between higher urinary BPA and lower sperm count (Rochester, 2013). Furthermore, over recent decades, the concentration of these synthetic compounds has increased significantly in the environment, leading to an enhanced exposure of animals and humans to EDCs. Among the main endocrine disruptors, bisphenol A (BPA) is one of the most used since the 1960s. It is an estrogenic monomer used in the production of polycarbonates, epoxy resins, unsaturated polyester-styrene resins and flame retardants, as well as in other types of plastic, such as polyvinyl chloride (PVC) and polyethylene terephthalate (PET). These materials are used as coating agents in cans, in powder paints, as additives in thermal paper, and in building materials, dental sealants, medical tubing, toys, water pipes, and soft drinks or water bottles (Staples et al., 1998).

Due to this wide variety of products and applications, BPA is now one of the most produced chemicals worldwide, with a total production capacity exceeding 3.8 million tons (Michalowicz, 2014) and its releases to the environment are more than 450 tons per year (USEPA, 2010). The main producer is USA (23%), followed by Germany, Japan and Taiwan (Huang et al., 2012). China is another important producer and also generates tons of industrial solid waste and household garbage that may contribute to its levels in the environment (Li and Wang, 2015).

Despite the ongoing environmental discussions about the hazards of plastic materials, their production is growing worldwide at a rate of 5% per year (Simoneit

et al., 2005; Wang et al., 2017). In addition, the development of recycling techniques for polymeric materials during recent years has contributed to this increment (Fukazawa et al., 2002; Hamad et al., 2013). Large amounts of plastic are disposed daily in landfills and subjected to intentional or incidental open-fire burning with the consequent volatilization of products, which are then incorporated into the environment (Simoneit et al., 2005). Thus, humans are exposed not only to environmental levels of BPA via inhalation, dermal exposure or ingestion (Kang et al., 2006), but also to BPA derivates released during the handling, unloading, heating and accidental spillage of waste (Staples et al., 1998).

Several authors have concluded that oral exposure related to food is the major source of BPA in all age groups of non-occupationally exposed human subjects (Dekant and Völkel, 2008). Therefore, the United States Environmental Protection Agency (EPA) established a reference dose for humans (Rfd) of 50 µg BPA/kg body weight (BW) day^{-1}, based on toxicological studies (USEPA, 2010). Related to this, according to Kang et al. (2006), the daily human intake of BPA is < 1 µg kg^{-1} body weight (BW) day^{-1}, which is 50-fold less than the reference dose. However, this values are only based on food consumption and, as mentioned above, most studies have focused on this pathway (Vandenberg et al., 2007), with little being known about the potential hazards of inhalation exposure due to the current levels of BPA in the environment. Regarding dermal exposure, Demierre et al. (2012) reported that a total daily uptake via the skin corresponds to 9.3 µg day^{-1}. More recent studies, have estimated a daily intake of 0.025 µg BPA kg^{-1} BW day^{-1} for the general population (Yang et al., 2019) while for occupationally exposed population there have been reported values of 1.42 ug day^{-1} and 71 ug day^{-1} (Rocha et al., 2015); 15−20 ug day^{-1} (Ndaw et al., 2016) and up to ∼250 ug day^{-1} (Fan et al., 2015). Nevertheless they are all far below the TDI (tolerable daily intake) of 50 µg BPA kg^{-1} BW day^{-1} so the European Union concluded that there is no concern arising from dermal exposure in comparison to other exposure sources (Demierre et al., 2012). It is important to highlight that TDI is based on the supposition that the primary route of exposure is oral intake, therefore dermal exposure as well as inhalation, should be treated as different intake pathways, thus TDI seems to be inadequate in these cases. On the other hand, despite the fact that BPA concentration has been widely studied in aquatic environments and in soils (Fromme et al., 2002; Staples et al., 1998), there are only a few studies related to its occurrence in air (Berkner et al., 2004; Matsumoto et al., 2005). Moreover, to date, there are no studies regarding BPA levels in South America, neither in air nor in other compartments, except for some studies carried out in Brazilian rivers (Montagner and Jardim, 2011; Moreira et al., 2011). Therefore, to our knowledge, this is the first investigation that assesses the temporal and spatial distribution of particle-bounded BPA in the atmosphere of Córdoba, Argentina, with the contribution of meteorological parameters also being analyzed. The results provide valuable

information about emergent organic pollutants in the atmosphere of a city in a developing country from South America, which can be employed to establish local guideline values.

2. Materials and methods

2.1. Study area

This study took place in Córdoba, the second largest city of Argentina, which is located at 31° 24′ S latitude and 64° 11′ W longitude, about 390 m above sea level, and surrounded by hills. It has a population of approximately 1.3 million inhabitants, with an average population density of 2274 inhabitants.km^{-2}.

A variety of industrial plants are located in the suburban areas surrounding the city, including major automobile factories, auto-part industries, agro-industries and food processing companies (López et al., 2011). The downtown area is the most seriously affected by air pollutants due to its high traffic density. Furthermore, Córdoba is located in a natural depression that reduces air circulation (Olcese and Toselli, 2002). The climate is sub-humid, with a mean annual precipitation of 800 mm (concentrated mainly in summer), a mean annual temperature of 18 °C and prevailing winds from the NE. During winter, there are frequent thermal inversions that increase air pollution problems in the city (López et al., 2011).

2.2. Sampling procedure and sites

As a preliminary study, a temporary sampling of the concentration of BAP in atmospheric particulate matter was carried out at the university campus (UC) of Córdoba, which is considered to be a semi-urban site with many trees and a significant percentage of bare soil (López et al., 2011). This sampling was carried out from June 2016 to May 2017, with a total of 55 samples being collected during working days.

Subsequently, in a second sampling, it was investigated whether there were any spatial differences in BPA concentrations at other sites in the city, with different characteristics. To carry this out, in addition to the UC site, two more sampling sites were chosen in the city. The second site was located in the city center (CC) and was characterized by high traffic, poor vegetation and high buildings that interfere with the dispersion of pollutants, while the third sampling site being located 7.7 km away from the city center, in an industrial zone (IA), with metallurgic and metal-mechanic industries. This sampling site was located inside an industrial plant, where several metallic parts are made (for tanks, radiators, covers, etc.). The sampling was performed during spring 2017, with a total of 38 samples collected during working days (Table 1).

Table 1. Concentrations of BPA (pg m^{-3}) in the atmosphere at the different sites studied in Córdoba, Argentina.

	Site	Sampling period	n	Mean	Standard Error	Minimum	Maximum
Temporal sampling	UC	June 2016 to May 2017	55	740	77	84	2454
Spatial sampling	UC	Spring 2017	23	403	73	89	1247
	CC		4	405	142	149	661
	IA		11	769	163	185	1775

2.3. Sample acquisition

Total suspended particles (TSP) samples were collected daily (24 ± 1 h) in glass fiber filters (110 mm of diameter, 0.3 μm pore) by employing a medium-volume sampler (Handi-Vol, Energética, Brazil) at 0.2 m^3 min^{-1} (approximately 300 m^3 every 24h) flow rate. This instrument was placed between 4 and 10 m above the ground in order to avoid obstruction of the air circulation.

Prior to use, the glass fiber filters (Pall, TX40HI20WW, USA) were baked at 450 °C for 5 h and stored in aluminum foil both before and after sample collection. The mass of the particles was determined gravimetrically as stated in the EPA Method IO-3.1 (USEPA, 1999), using the difference between the initial and final weights of each filter. Once the sample was collected, filters were stored at -20 °C in the dark until extraction was carried out to avoid BPA photolysis or volatilization.

2.4. BPA extraction and clean up

Considering that the analyte has a high tendency to sorb to glassware (Berkner et al., 2004), only glass material previously baked for 5 hs at 450 °C was employed to prevent external contamination. Filters containing particles were placed in closed reflux tubes that were filled with dichloromethane (JT Baker, USA) (Cao, 2010). Then, these tubes were placed in an ultrasound bath for 30 min, after which, the contents were mixed with vortex. These last two procedures were repeated 3 times to completely extract BPA from the matrix (Cao, 2010). Next, the extract was concentrated under a soft N$_2$ air current, filtered with a glass syringe and a Teflon filter (0.22 μm pore diameter), dried under a soft N$_2$ air current and then resuspended in 1 mL of acetonitrile:water (40:60). Acetonitrile was of analytical reagent grade HPLC (Merck, Germany) and water was obtained using a Milli-Q water purification system (Millipore Corp, USA).

To test the variability of our extraction method, three filters were divided into equal subsamples and each of these was extracted separately. The maximum variation observed in this method of extraction was 13 % (Table 2).

Table 2. Blanks and variability of the extraction method. Each simple corresponds to a filter with particulate matter that was divided in equal subsamples and extracted separated. The coefficient of variation (CV) is shown as a measure of relative variability.

Sample	Subsample	BPA (ng mL^{-1})	Mean	SE	CV(%)
Blank 1		1.14			
Blank 2		2.13			
Blank 3		1.47			
Blank 4		3.26			
Blank 5		2.94			
1	a	76.23	75.41	1.82	6.37
	b	69.32			
	c	71.39			
	d	74.65			
	e	73.58			
	f	79.11			
	g	83.59			
2	a	23.93	25.05	1.35	13.24
	b	31.28			
	c	26.28			
	d	23.14			
	e	22.88			
	f	22.80			
3	a	421.74	471.39	17.28	9.7
	b	513.27			
	c	493.47			
	d	497.62			
	e	469.90			
	f	508.39			
	g	395.36			

2.5. Determination of BPA

High-performance liquid chromatography (Thermo scientific dionex ultimate 3000, Germering, Germany) was preferred over gas chromatography because in the latter case a complex derivatization step is necessary. Samples were analyzed employing a reversed phase C18 column (LiChrospher® 100 RP-18, Merck, Darmstadt, Germany) at 40 °C with an isocratic mobile phase of acetonitrile: water (40:60, v/v) at 1 mL min^{-1} flow rate, as used by other authors (Kang and Kondo 2002, 2005; Ng and Reuter, 2008). A fluorescence detector (FLD-3100 and FLD-3400 RS, Thermo scientific dionex, Germering, Germay) was utilized to perform the measurement at 275 nm for excitation and 313 nm for emission, as suggested by Ng and Reuter (2008).

BPA concentrations were calculated employing a 12 point BPA calibration curve ranging from 0.96 to 1000 ng mL^{-1} (R^2 = 0.9944), obtained from a BPA stock

solution (Sigma Aldrich, USA). Due to the inclusion of BPA in many sampling devices/materials the quantification of BPA in control samples is a problem (Berkner et al., 2004; Ye et al., 2013), however recent studies have demonstrated that it is possible to measure BPA in the absence of laboratory contamination, with appropriate and rigorous control tests (Vandenberg et al., 2014; Vom Saal and Welshons, 2014).

Since low concentrations of BPA are regularly measured in control samples (Berkner et al., 2004; Ye et al., 2013), prior to their use the solvents and blank filters were tested for the absence of BPA. In addition, every preparation of samples was accompanied by a blank sample that was extracted and analyzed together with the collected samples, with the results showing no considerable contamination in the blanks (less than 5% found in the samples). The values obtained were regarded as reliable if their content was more than two times higher than the blank sample (Berkner et al., 2004). The limits of detection (LOD) and quantification (LOQ) for BPA were 6 and 12 ng mL^{-1}, respectively, calculated as the analyte concentration corresponding to the sample blank value plus three standard deviation for the LOD and ten standard deviations for LOQ (Shrivastava and Gupta, 2011). The identification of BPA in the samples was only based on the retention times in order to conform with the standards.

2.6. Meteorological parameters

Daily meteorological data were obtained from the meteorological station of the National Meteorological Service located at Córdoba Airport, 9.5 km north of the city center (31o18′54″ S, 64o12′44″ W, altitude 484 m.a.s.l.). For the present study, we considered the influence of mean temperature (T), relative humidity (RH), atmospheric pressure (AP), wind speed (WS), wind direction (WD) and heliophany (H). The latter variable denotes the average sunlight hours.

2.7. Data analysis

InfoStat software coupled to R (Di Rienzo et al., 2018) was employed for the statistical analysis, with BPA concentrations being expressed as mean ± standard deviation. All data was ln transformed to conform to normality. ANOVA was employed for the monthly and spatial analysis, while general lineal models were used for seasonal analysis, with variance being modeled with the identity function (varIdent) to correct heteroscedasticity. The LSD Fisher post hoc test was used to detect differences between months and seasons, as well as variations among the sampling sites. Differences with a p value <0.05 were considered to be statistically significant. The Spearman correlation coefficients were employed to assess the relationship between BPA and the meteorological parameters.

3. Results and discussion

3.1. Atmospheric BPA levels

The average BPA concentration at the UC site was 740 pg m^{-3} during the sampling conducted between June 2016 and May 2017, which had a range of 84−2454 pg m^{-3} (Table 1). These concentration values of BPA obtained in the atmosphere of Córdoba, were similar to those reported for other urban environments (Table 3). Fu and Kawamura (2010) concluded that BPA has become ubiquitous in the environment, with concentrations ranging from 1 to 17400 pg m^{-3}, and much smaller concentrations at remote sites such as polar regions (1−11 pg m^{-3}) (Huang et al., 2012). In agreement, the values found in our study were comparable to the ones reported in other cities including Auckland (1.6 million inhabitants), Christchurch (0.4 million inhabitants) (Fu and Kawamura, 2010) and Osaka (2.6 million inhabitants) (Matsumoto et al., 2005). Higher concentrations were reported for urban areas in China where very high levels (20000 ± 2700 pg m^{-3}) were found in Guangzhou, one of the China's largest and most populated cities (Li and Wang, 2015). Other big and populated cities like Xi'an and New Delhi also exhibit high levels: 47000 ± 41000 pg m^{-3} and 18000 ± 14000 pg m^{-3} respectively (Li et al., 2014). On the other hand, the concentrations found in the city of Córdoba did not correlate with the amount of TSP in the atmosphere, which may indicate that the sources of BPA are different from the sources of TSP (Fig. 1).

3.2. Temporal variation

During the year of sampling at the UC site, the BPA concentrations in the atmosphere revealed seasonal differences with significantly higher atmospheric concentrations in the winter and lower concentrations in the spring (Fig. 2). These results

Table 3. Concentrations of BPA in the atmosphere (pg m^{-3}) at worldwide urban sites.

Location	Type	Range	Sampling date	Reference
Chennai, India	PM10	200−17400	2007	Fu and Kawamura (2010)
Mumbai, India	PM10	100−9820	2008	Fu and Kawamura (2010)
Beijing, China	PM2.5	380−1260	2007	Fu and Kawamura (2010)
Yufa, China	PM2.5	230−860	2007	Fu and Kawamura (2010)
Hong Kong, China	PM2.5	30−690	2007	Fu and Kawamura (2010)
Osaka, Japan	TSP	10−1920	2000−2001	Matsumoto et al., (2005)
Sapporo, Japan	TSP	70−930	2008−2009	Fu and Kawamura (2010)
Auckland, New Zealand	PM10	4−1340	2004	Fu and Kawamura (2010)
Christchurch, New Zealand	PM10	95−1480	2004	Fu and Kawamura (2010)
Córdoba, Argentina	TSP	84−2454	2016−2017	Present Study
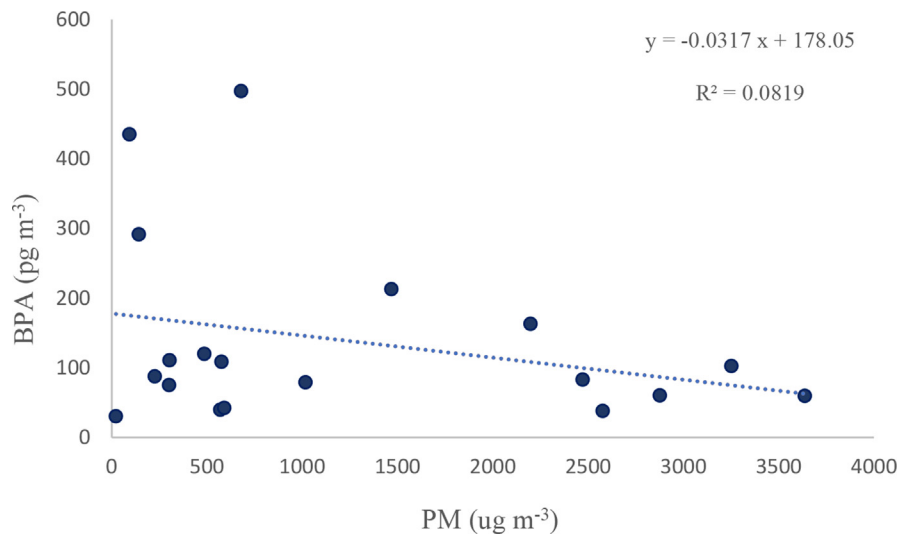
Fig. 1. - Concentration of Bisphenol A (pg m^{-3}) as a function of airborne particulate matter (PM) (ug m^{-3}) sampled in the university campus (UC). The equation of the regression and the value of R^2 are shown on the top of the graph.
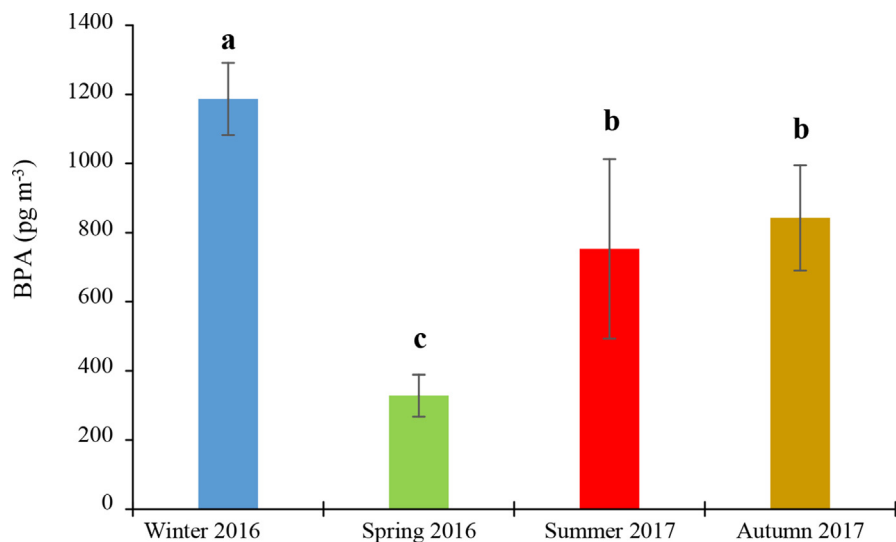
Fig. 2. Seasonal particle-bounded BPA concentration in winter (June−August), spring (September−November), summer (December−February) and autumn (March−May). Different letters indicate statistical difference with $p < 0.05$.

were in agreement with previous studies, where a higher BPA concentration was reported in the coldest months (Cecinato et al., 2017; Matsumoto et al., 2005). These observations may be explained by the climatic events occurring during the warmer months, such as higher temperatures, a greater intensity of sunlight, the increased height of the mixing layer and the frequent rain washing the particles in the atmosphere, as mentioned in studies carried out on other pollutants such as polycyclic aromatic hydrocarbons (Lee et al., 2001; Xia et al., 2013).

Nevertheless, during the summer, BPA levels recorded in our study were very variable, with high concentrations found in January, which were probably due to the occurrence of a bus catching fire in a car park near the university campus, only 500 m from the sampling site. On that particular day, the concentration of BPA in the atmosphere reached 2554 pg m^{-3}.

In terms of monthly variation, June showed the highest level of BPA, while October, November and December had the lowest concentrations (Fig. 3). In addition to the fact that a seasonal trend was seen in BPA levels, we also observed a strong daily variation, which could be attributed to the presence of local BPA sources. For example, during May, very different BPA daily levels were observed (Fig. 4). We discarded the possibility that this high variability may have been due to the extraction method, since subsamples from the same filter showed variation coefficients below 15%.

3.3. Spatial analysis

Following the temporal study, a spatial study was also carried out. Three sites with well-marked differences were sampled in the city of Córdoba, with it being observed that the concentration of BPA in the atmosphere at the IA site was significantly higher than at the other study sites (Fig. 5). These results suggest the presence of local sources in the industrial area that may be releasing BPA or its derivatives, and are consistent with those reported by Salapasidou et al. (2011), who found higher levels of BPA at an urban-industrial site compared with an urban area in a city located in northern Greece. However, it is important to consider that the spatial sampling was carried out only during the lowest concentration
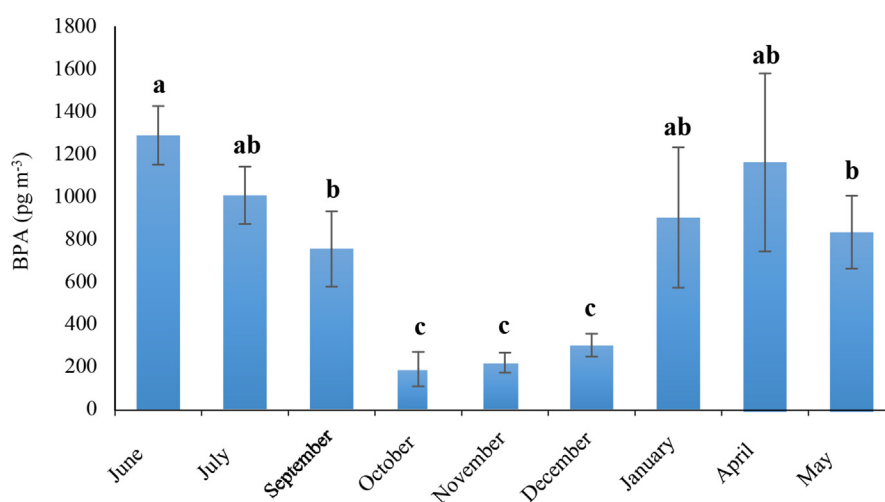
Fig. 3. Monthly particle-bounded BPA concentration. ANOVA between months, different letters indicate statistical differences with p < 0.05.
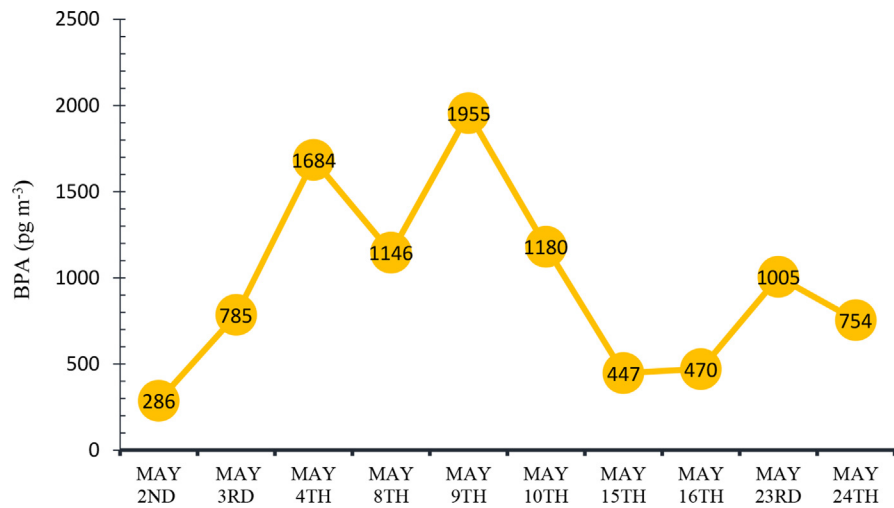
Fig. 4. BPA daily concentration values (pg m^{-3}) in the university campus (UC) during May 2017.

period (spring), so this is a limitation of this study since the differences could be considerably exacerbated in periods of higher concentrations.

3.4. Influence of meteorological parameters

It is well known that meteorological conditions can affect organic pollutants, with respect to their generation, phase partitioning, accumulation, diffusion, removal, etc
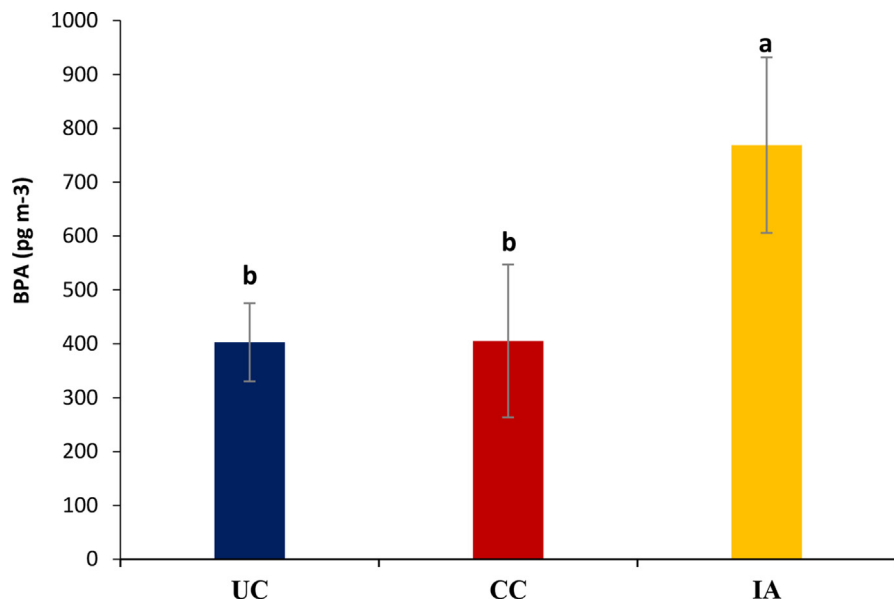

Fig. 5. - Concentrations of BPA (pg m^{-3}) at the sampling sites in the city of Córdoba during Spring 2017. ANOVA between sites, different letters indicate statistical difference with p < 0.05.

(Amarillo and Carreras, 2016). Therefore, we evaluated the effect of some meteorological parameters on the concentration of BPA in the atmosphere by calculating the Spearman correlation coefficient (Table 4). These results revealed that the wind speed (WS) was inversely correlated with the concentration of BPA in the atmosphere (R^2 = -0.67; p < 0.001), indicating that during strong wind events, the amounts of BPA would be reduced in the atmosphere, especially if there are local sources. Temperature (T) also revealed a significant and negative correlation with the concentration of BPA in the atmosphere (R^2 = -0.55; p < 0.001). Although there are no previous studies carried out between concentrations of BPA in the atmosphere and temperature, we could extrapolate this result to that found in studies carried out with PAHs, because BPA (228.115 g mol^{-1}) presents a vapour pressure value (4×10^{-8} mm Hg at 25 °C), which is between benzo[a]anthracene (228.094 g mol^{-1}; 2.7×10^{-7} mm Hg at 25 °C) and chrysene (228.094 g mol^{-1}; 6.2×10^{-9} mm Hg at 25 °C) (NCBI, 2019). Higher temperatures not only increase PAHs participation in the vapor phase, but would also cause more degradation (He et al., 2014). Amarillo and Carreras (2016) also found this relationship for polycyclic aromatic hydrocarbons (PAHs) in Córdoba city. Furthermore, the intensity of sunlight during the warmer months could also contribute to a greater degradation of the compound. In contrast, the atmospheric pressure (AP) was positively correlated with the concentration of BPA in the atmosphere (R^2 = 0.37; p < 0.01) in the same way that the relative humidity (R^2 = 0.38; p < 0.01). On average, the AP in Córdoba is higher in the cold months, when thermal inversion events can occur due to the topography of Córdoba (López et al., 2011). Stable atmospheric conditions prevent the vertical diffusion of air pollutants, while calm conditions prevent horizontal transport. In these cases, air pollutants remain close to the surface of the city, which would explain this association. The relative humid (RH) is also higher in the colder months being negatively correlated with mean temperature. All these significant meteorological parameters are illustrated in Fig. 6. With respect to the other meteorological variables analyzed (WD and H) no correlation was revealed with BPA levels in the atmosphere.

Table 4. Spearman correlation coefficients between BPA air concentration and meteorological parameters. Bold numbers indicate significant correlations (**correlation is significant at the p ≤ 0.001 level; *correlation is significant at the p ≤ 0.01).

	T	RH	AP	WS	WD	H
BPA	−0.55**	0.38*	0.37*	−0.67**	−0.15	−0.23

T = mean temperature, RH = relative humidity, AP = Atmospheric pressure, WS = wind speed, WD = Wind direction, H = heliophany.
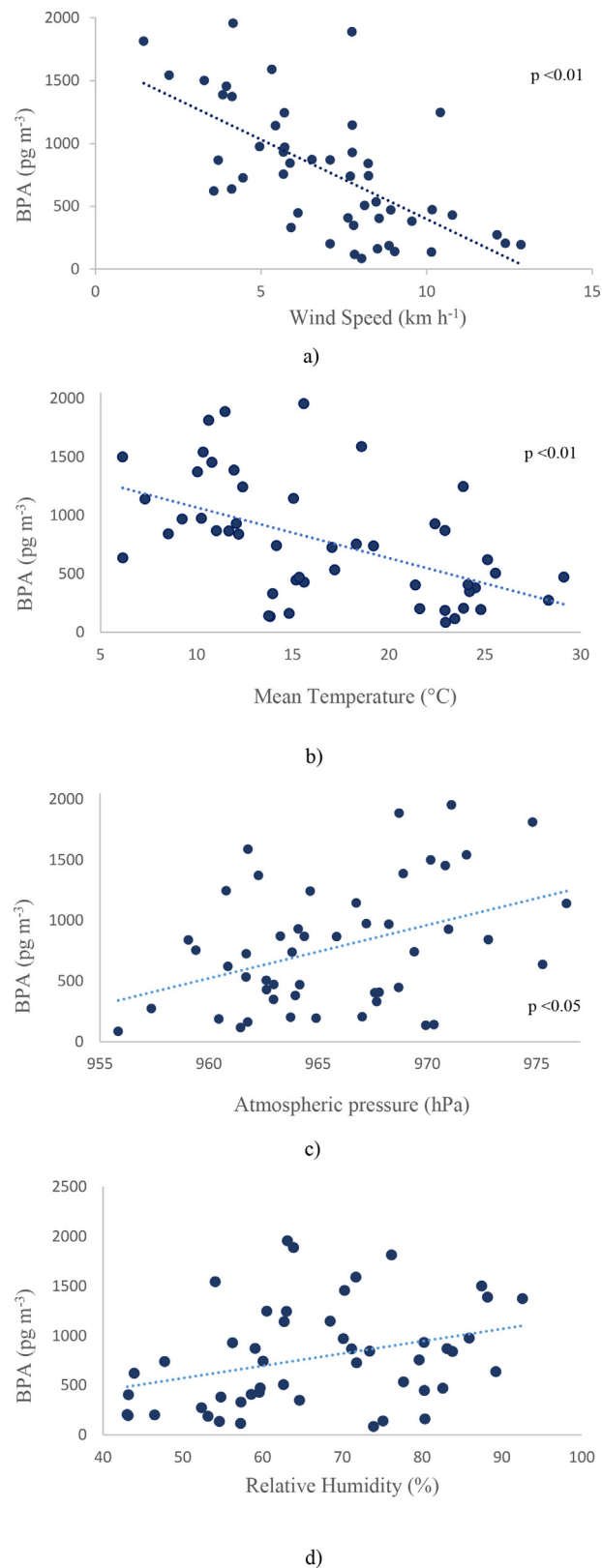
Fig. 6. Correlations for BPA concentrations in air and meteorological parameters: a) Wind speed, b) Mean temperature, c) Atmospheric pressure and d) Relative Humidity.

4. Conclusions

This study presents a first analysis of the occurrence (temporal and spatial variation) of Bisphenol A in an urban atmosphere in the city of Córdoba, Argentina. The results found are comparable to those reported in other cities of the world. Here, in addition, we observed a seasonal trend, with BAP levels increasing during the cold months but decreasing during the warm period. Regarding the different sites studied, a spatial gradient was observed within the city, with the highest concentrations being found at the industrial site. Nevertheless, it should be noted that this compound presented a high daily variation making the atmospheric concentration of BPA in the city very unpredictable. This might be due to the concentration depending on the presence of local sources that release BPA into the environment rather than depending on the concentration of atmospheric particulate matter. It was also observed that some meteorological parameters (WS, T, AP and RH) affected the concentration of this compound in the atmosphere.

The data presented will certainly contribute to improve the national dataset of emerging pollutants, which will help in implementing policies to protect the environment and human health. However, more studies are needed to obtain a better understanding of the potential risks associated with BPA particle bounded inhalation and to establish a reference toxicity value. In this study, no risk assessment was performed due to the lack of reference toxicity value for inhalation pathways. In addition, more studies are needed to assess concentrations of BPA in respirable particles (fine and ultrafine), which can deeply reach the respiratory system.

Declarations

Author contribution statement

Natalia Soledad Graziani: Performed the experiments; Analyzed and interpreted the data.

Hebe Carreras: Contributed reagents, materials, analysis tools or data.

Eduardo Wannaz: Conceived and designed the experiments; Analyzed and interpreted the data; Wrote the paper.

Funding statement

Competing interest statement

The authors declare no conflict of interest.

Additional information

No additional information is available for this paper.

Acknowledgements

References

Amarillo, A.C., Carreras, H., 2016. Quantifying the influence of meteorological variables on particle-bound PAHs in urban environments. Atmos. Pollut. Res. 7, 597−602.

Berkner, S., Streck, G., Herrmann, R., 2004. Development and validation of a method for determination of trace levels of alkylphenols and bisphenol A in atmospheric samples. Chemosphere 54, 575−584.

Cao, X., 2010. Background paper on chemistry and analytical methods for determination of bisphenol a in food and biological samples FAO/WHO expert meeting on bisphenol a (BPA) prepared by. World Heal. Organ. 1−29.

Cecinato, A., Romagnoli, P., Perilli, M., Balducci, C., 2017. Nonylphenol and bisphenol a in ambient particulates: a preliminary approach in Italy. Environ. Toxicol. Stud. J. 1, 1−8.

Dekant, W., Völkel, W., 2008. Human exposure to bisphenol A by biomonitoring: methods, results and assessment of environmental exposures. Toxicol. Appl. Pharmacol. 228, 114−134.

Demierre, A.-L., Peter, R., Oberli, A., Bourqui-Pittet, M., 2012. Dermal penetration of bisphenol A in human skin contributes marginally to total exposure. Toxicol. Lett. 213, 305−308.

Di Rienzo, J.A., Casanoves, F., Balzarini, M.G., Gonzalez, L., Tablada, M., Robledo, C.W., 2018. InfoStat, Versión 2018, Grupo InfoStat, FCA. Universidad Nacional de Córdoba, Argentina. http://www.infostat.com.ar.

Fan, R., Zeng, B., Liu, X., Chen, C., Zhuang, Q., Wang, Y., Hu, M., Lv, Y., Li, J., Zhou, Y., Lin, Z.Y.W., 2015. Levels of bisphenol-A in different paper products in Guangzhou, China, and assessment of human exposure via dermal contact. Environ. Sci. Process. Impacts 17, 667−673.

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

Fu, P., Kawamura, K., 2010. Ubiquity of bisphenol A in the atmosphere. Environ. Pollut. 158, 3138−3143.

Fukazawa, H., Watanabe, M., Shiraishi, F., Shiraishi, H., Shiozawa, T., Matsushita, H., Terao, Y., 2002. Formation of chlorinated derivatives of bisphenol a in waste paper recycling plants and their estrogenic activities. J. Health Sci. 48, 242−249.

Hamad, K., Kaseem, M., Deri, F., 2013. Recycling of waste from polymer materials: an overview of the recent works. Polym. Degrad. Stabil. 98, 2801−2812.

He, J., Fan, S., Meng, Q., Sun, Y., Zhang, J., Zu, F., 2014. Polycyclic aromatic hydrocarbons (PAHs) associated with fine particulate matters in Nanjing, China: distributions, sources and meteorological influences. Atmósfera. Environ. 89, 207−215.

Huang, Y.Q., Wong, C.K.C., Zheng, J.S., Bouwman, H., Barra, R., Wahlström, B., Neretin, L., Wong, M.H., 2012. Bisphenol A (BPA) in China: a review of sources, environmental levels, and potential human health impacts. Environ. Int. 42, 91−99.

Kang, J.H., Kondo, F., 2002. Bisphenol a degradation by bacteria isolated from river water. Arch. Environ. Contam. Toxicol. 43, 265−269.

Kang, J.H., Kondo, F., 2005. Bisphenol A degradation in seawater is different from that in river water. Chemosphere 60, 1288−1292.

Kang, J.H., Kondo, F., Katayama, Y., 2006. Human exposure to bisphenol A. Toxicology 226, 79−89.

Lee, S., Ho, K., Chan, L., Zielinska, B., Chow, J.C., 2001. Polycyclic aromatic hydrocarbons (PAHs) and carbonyl compounds in urban atmosphere of Hong Kong. Atmos. Environ. 35, 5949−5960.

Li, J., Wang, G., 2015. Airborne particulate endocrine disrupting compounds in China: compositions, size distributions and seasonal variations of phthalate esters and bisphenol A. Atmos. Res. 154, 138−145.

Li, J., Wang, G., Aggarwal, S.G., Huang, Y., Ren, Y., Zhou, B., Singh, K., Gupta, P.K., Cao, J., Zhang, R., 2014. Comparison of abundances, compositions and sources of elements, inorganic ions and organic compounds in atmospheric aerosols from Xi'an and New Delhi, two megacities in China and India. Sci. Total Environ. 476−477, 485−495.

López, M.L., Ceppi, S., Palancar, G.G., Olcese, L.E., Tirao, G., Toselli, B.M., 2011. Elemental concentration and source identification of PM10 and PM2.5 by SR-XRF in Córdoba City, Argentina. Atmos. Environ. 45, 5450−5457.

Matsumoto, H., Adachi, S., Suzuki, Y., 2005. Bisphenol a in ambient air particulates responsible for the proliferation of MCF-7 human breast cancer cells and its concentration changes over 6 months. Arch. Environ. Contam. Toxicol. 48, 459−466.

Michałowicz, J., 2014. Bisphenol A − sources, toxicity and biotransformation. Environ. Toxicol. Pharmacol. 37, 738−758.

Montagner, C.C., Jardim, W.F., 2011. Spatial and seasonal variations of pharmaceuticals and endocrine disruptors in the Atibaia River, São Paulo State (Brazil). J. Braz. Chem. Soc. 22, 1452−1462.

Moreira, M., Aquino, S., Coutrim, M., Silva, J., Afonso, R., 2011. Determination of endocrine-disrupting compounds in waters from Rio das Velhas, Brazil, by liquid chromatography/high resolution mass spectrometry (ESI-LC-IT-TOF/MS). Environ. Technol. 32, 1409−1417.

NCBI, 2019. National Center for Biotechnology Information. PubChem Compound Database; CID=9171. https://pubchem.ncbi.nlm.nih.gov/compound/9171. (Accessed 5 February 2019).

Ndaw, S., Remy, A., Jargot, D., Robert, A., 2016. Occupational exposure of cashiers to Bisphenol A via thermal paper: urinary biomonitoring study. Int. Arch. Occup. Environ. Health 89, 935−946.

Ng, C.M., Reuter, W.M., 2008. Analysis of Bisphenol A in Toys by HPLC with Fluorescence Detection Liquid Chromatography. Application Note. PerkinElmer, Inc., USA. https://www.perkinelmer.com/lab-solutions/resources/docs/APP_Analysis-of-Bisphenol-A-in-Toys-012153_01.pdf.

Olcese, L.E., Toselli, B.M., 2002. Some aspects of air pollution in Córdoba, Argentina. Atmos. Environ. 36, 299−306.

Rocha, B.A., Azevedo, L.F., Gallimberti, M., Campiglia, A.D., Barbosa Jr., F., 2015. High levels of bisphenol A and bisphenol S in Brazilian thermal paper receipts and estimation of daily exposure. J. Toxicol. Environ. Health, Part A 78 (18), 1181−1188.

Rochester, J.R., 2013. Bisphenol A and human health: a review of the literature. Reprod. Toxicol. 42, 132−155.

Salapasidou, M., Samara, C., Voutsa, D., 2011. Endocrine disrupting compounds in the atmosphere of the urban area of Thessaloniki, Greece. Atmos. Environ. 45, 3720−3729.

Schug, T.T., Janesick, A., Blumberg, B., Heindel, J.J., 2011. Endocrine disrupting chemicals and disease susceptibility. J. Steroid Biochem. Mol. Biol. 127, 204−215.

Shrivastava, A., Gupta, V., 2011. Methods for the determination of limit of detection and limit of quantitation of the analytical methods. Chronicles Young Sci. 2, 21.

Simoneit, B.R.T., Medeiros, P.M., Didyk, B.M., 2005. Combustion products of plastics as indicators for refuse burning in the atmosphere. Environ. Sci. Technol. 39, 6961−6970.

Staples, C.A., Dome, P.B., Klecka, G.M., Oblock, S.T., Harris, L.R., 1998. A review of the environmental fate, effects, and exposures of bisphenol A. Chemosphere 36, 2149−2173.

USEPA, 1999. Method IO-3.1. Selection, Preparation and Extraction of Filter Material.

USEPA, 2010. Bisphenol a Action Plan (CASRN 80-05-7), 2010. Available from: https://www.epa.gov/sites/production/files/2015-09/documents/bpa_action_plan.pdf. (Accessed 24 September 2018).

Vandenberg, L.N., Hauser, R., Marcus, M., Olea, N., Welshons, W.V., 2007. Human exposure to bisphenol A (BPA). Reprod. Toxicol. 24, 139−177.

Vandenberg, L.N., Gerona, R.R., Kannan, K., Taylor, J.A., Van Breemen, R.B., Yuan, Y., Newbold, R.R., Padmanabhan, V., Vom Saal, F.S., Woodruff, T.J., 2014. A round robin approach to the analysis of bisphenol a (BPA) in human blood samples. Environ. Health: A Global Access Sci. Source 13 (1). Article number 25.

Vom Saal, F.S., Welshons, W.V., 2014. Evidence that bisphenol A (BPA) can be accurately measured without contamination in human serum and urine, and that BPA causes numerous hazards from multiple routes of exposure. Mol. Cell. Endocrinol. 398, 101−113.

Wang, Z., Liu, H., Liu, S., 2017. Low-dose bisphenol a exposure: a seemingly instigating carcinogenic effect on breast cancer. Adv. Sci. 4, 1600248.

Xia, Z., Duan, X., Tao, S., Qiu, W., Liu, D., Wang, Y., Wei, S., Wang, B., Jiang, Q., Lu, B., Song, Y., Hu, X., 2013. Pollution level, inhalation exposure and lung cancer risk of ambient atmospheric polycyclic aromatic hydrocarbons (PAHs) in Taiyuan, China. Environ. Pollut. 173, 150−156.

Yang, Y., Yang, Y., Zhang, J., Shao, B., Yin, J., 2019. Assessment of bisphenol A alternatives in paper products from the Chinese market and their dermal exposure in the general population. Environ. Pollut. 244, 238−246.

Ye, X., Zhou, X., Hennings, R., Kramer, J., Calafat, A.M., 2013. Potential external contamination with bisphenol A and other ubiquitous organic environmental chemicals during biomonitoring analysis: an elusive laboratory challenge. Environ. Health Perspect. 121, 283−286.