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Original article

## Quantifying the influence of meteorological variables on particle-bound PAHs in urban environments

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### ABSTRACT

Seasonal trend of polycyclic aromatic hydrocarbons (PAHs) associated with PM<sub>10</sub> particles was assessed in an urban area in Córdoba, Argentina, during 2011–2013. In addition, we quantified the contribution of temperature, relative humidity, rainfall, atmospheric pressure and wind speed on total and individual PAH concentrations. Mean PAH concentrations were higher during autumn and winter. Temperature was the meteorological parameter that affected the most total and individual PAHs. Its influence was stronger in lightweight than in heavy PAHs, since they are more easily partitioned in the vapor phase. We found also that temperature has a lagged effect on PAHs concentration with the strongest association observed with 2-day lag temperature. Wind speed and relative humidity were also significant predictors for both light and heavy PAHs. Despite the fact that we observed a decreasing trend in the concentration levels of PAHs, they are still high enough to pose a risk to human health due to chronic exposure.

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### 1. Introduction

Polycyclic aromatic hydrocarbons (PAHs) are ubiquitous contaminants in the environment produced by the combustion of fossil fuels and organic waste (Liu et al., 2007; Miet et al., 2009). In urban and industrial environments, they are almost entirely attributed an anthropogenic origin due to traffic domestic heating, thermal power stations and industrial emissions (Rehwagen et al., 2005; Dong and Lee, 2009).

PAHs are semi volatile compounds that can be found in the atmosphere in both the vapor and particle-associated phases (Motelay-Massei et al., 2003). Low molecular weight PAHs (LPAHs) tend to be more concentrated in the vapor phase, while high molecular weight PAHs (HPAHs) are often associated with particulate matter (Barrado et al., 2012). The fact that most of the PAHs are found in aerosol form, suggests they are highly dependent on climatic conditions and they can be transported over long distances, as well. PAHs are recognized as mutagenic compounds

and known to be carcinogenic in animals and humans; therefore, they have been classified as priority pollutants by the US Environmental Protection Agency (US EPA) and by the European community. A number of studies have already assessed the influence of climatic condition on PAH concentrations. Indeed, seasonal fluctuations with higher concentrations detected in winter have already been determined in many different urban and rural sampling areas (Fang et al., 2004; Lee et al., 2006; Li et al., 2006). Most of these studies have employed correlations, principal component or cluster analyses which provide qualitative information on the association between PAHs and meteorological factors. However, these analyses cannot be used to predict variations in their concentrations associated to changes in meteorological conditions.

On the other hand, during the past decade PAHs distribution in the environment have been widely studied, because of their recognized toxicity with a negative environmental impact. Thus, numerous studies have been undertaken to determine the level of PAH pollution in atmospheric deposition in Europe and Asia (Gryniewicz et al., 2002; He et al., 2014), however results from Latin American countries are still scarce. In Argentina, the concentration of particle-bound PAHs has been measured only in the urban area of La Plata and Córdoba city (Rehwagen et al., 2005; Carreras et al., 2013).

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The purpose of our study was to analyze seasonal variations of PAHs in the city of Cordoba as well as quantify the influence of temperature, relative humidity, rainfall, atmospheric pressure and wind speed on total and individual PAH concentrations.

## 2. Materials and methods

### 2.1. Study site

Cordoba is the second largest city in Argentina, located at the center of the country (31° 24' S, 64° 11' W) at an altitude of approximately 400 m above sea level. It has a population of 1.3 million and an irregular topography. Its general structure is funnel-shaped, with an increasing positive slope from the center towards the surrounding area. This somewhat concave formation reduces the air circulation and causes frequent thermal inversions both in autumn and winter (Olcese and Toselli, 2002). The climate is sub-humid, with an average annual rainfall of 790 mm, concentrated mainly in summer. The mean annual temperature is 17.4 °C and the prevailing winds come from the NE, S and SE.

The main sources of air pollution in Cordoba city are automobile sources, with a strong relationship with primary pollutants (CO, NO<sub>x</sub> and PM<sub>10</sub>). The city also has an important industrial development of mainly metallurgic and mechanical industries (Amarillo and Carreras, 2012). Previous studies have found that in Cordoba urban area, the average PM<sub>10</sub> and PM<sub>2.5</sub> values are almost 2.5 and 2.8 times higher respectively than the corresponding EU limit values for air quality (24-h limit value of 40 µg/m<sup>3</sup> for PM<sub>10</sub> and 25 µg/m<sup>3</sup> for PM<sub>2.5</sub>, López et al., 2011). Indeed, previous studies had shown that increases in the airborne particulate were related to a higher morbidity due to respiratory diseases (Amarillo and Carreras, 2012, Carreras et al., 2008).

### 2.2. Meteorological parameters

Daily meteorological data were obtained from the meteorological station of the National Meteorological Service located at the Cordoba Airport, 9.5 km north from the city center (−31.31° S, −64.21° W, altitude 484 masl). This meteorological station is considered as a source of the most reliable data. For the present study, we considered the influence of mean temperature (*T*), relative humidity (RH), atmospheric pressure (AP), wind speed (*W*), mixing layer height (LH) and rainfall (*R*).

### 2.3. Sample collection

Sampling was carried out for 24 h every day with no heavy rain from August 2011 to July 2013. A total of 424 samples were obtained. A medium volume sampler for PM<sub>10</sub> was located 7 m high on the roof of the Chemistry Department at the FCEFYN, Cordoba University. It was operated at a flow rate of 0.2 m<sup>3</sup> min<sup>−1</sup> to obtain a total sample volume of greater than 300 m<sup>3</sup> over the 24 h period.

Before sampling, glass fiber filters were baked at 200 °C for at least 24 h, to volatilize any organic contaminants. After sampling, the filters were wrapped in aluminum foil to avoid photo degradation. All freshly exposed filters were conditioned in a desiccator for 24 h to remove moisture. Exposed filters were then folded and rewrapped in aluminum foil, stored in sealed plastic bags and kept refrigerated (−18 °C) until the day of analysis (<1 month). The concentration of particles (µg m<sup>−3</sup>) was determined by differences in the filter weights before and after the 24 h exposures divided by the filtered air volume.

### 2.4. Sample preparation

The solvent-extracted organic matter (SEOM) was obtained immersing the filters in 30 mL methylene chloride (MC) (HPLC grade, Chromanorm) with ultrasound for two 30 min periods, to ensure that all compounds of interest had been extracted. In order to avoid MC evaporation (likely to take place during extraction, as it has a relatively low boiling point) and the subsequent possibility of losing the compounds of interest in the process, a cooling device was fitted over the flask's mouth containing both the sample and MC. The extracts were concentrated with a rotavapor, at 30 °C, and then evaporated under a soft nitrogen flow. The concentrates were filtered with syringe Teflon filters (0.22 µm) and brought up to 1 mL using acetonitrile.

### 2.5. Instrumental analysis

PAHs extracts were analyzed by High-Performance Liquid Chromatography (HPLC— Perkin Elmer series 200) using fluorescence detectors. Separation was performed by means of a reverse phase C-18 column (Luna 5u C18 100A). A gradient elution program was utilized using mobile phases of acetonitrile and distilled deionized water. The column temperature was 30 °C, and the flow rate of the mobile phase was 1.8 mL min<sup>−1</sup>. The changes in the composition of the mobile phase and detection wavelengths are shown in Table 1.

The system was calibrated for the 16 EPA target PAHs with a standard solutions (EPA 610 PAHs Mixture) purchased from Supelco (Argentina). Thus, we quantified the concentration of Naphthalene (NAP), Acenaphthene (AC), Fluorene (FL), Phenanthrene (PHE), Anthracene (ANT), Fluoranthene (FLU), Pyrene (PYR), Benzo[a]anthracene (BaA), Chrysene (CHR), Benzo[b]fluoranthene (BbF), Benzo[k]fluoranthene (BkF), Benzo[a]pyrene (BaP), Dibenzo[a,h]anthracene (DBahA), Benzo[g,h,i]perylene (BghiP). The system was calibrated with a six-point calibration curves for all PAHs, ranging from 1 to 100 µg L<sup>−1</sup> ( $R^2 > 0.95$ ,  $p < 0.001$ ). The PAH detection limits were found to be between 5 and 43 pg m<sup>−3</sup>. Several blank samples were analyzed and their mean PAH concentrations were subtracted from all other samples. The recoveries of PAHs from particulates ranged from 70% (Phe) to 99% (DBahA), while the coefficients of variation ranged from 0.5% (BkFl) to 1.3% (Phe).

### 2.6. Data analysis

The relationship between PAHs and meteorological parameters was first investigated calculating their Pearson correlation coefficients. After that, we run a multivariate regression analysis to quantify the contribution of different meteorological predictors on total and grouped PAHs. We used a stepwise variable selection method that starts regressing all meteorological variables while removing the weakest correlated variable and leaving those that explain best the distribution of the dependent variable. All meteorological variables were included as linear terms.

We also considered the possibility of a lagged effect of *T* on PAHs, using lag *T* instead of same day *T*, as predictor in the multivariate regression analysis. In addition, we run a regression stratified by seasons to assess the differential contribution of *T*. The analyses were performed using statistical software R 2.12.1.

## 3. Results and discussion

To evaluate different PAH behaviors, our data were examined individually and in groups. Thus, the following terms were used: total PAH (TPAH: sum of all individual PAH), low molecular weight

**Table 1**

Fluorescence excitation and emission wavelengths and composition of the mobile phase used in the present study.

Time (min)	Fluorescence excitation $\lambda_{exc}$ (nm)	Fluorescence emission $\lambda_{emi}$ (nm)	Mobile phase % (water:ACN)
0	290	430	40:60
3.2	260	352	20:80
6.2	250	390	10:90
7.1	270	440	10:90
7.8	250	390	0:100
11.5	290	430	0:100

PAH (LPAH: sum of 2–3 ring PAHs), and high-molecular weight PAH (HPAH: sum of 4–6 ring PAHs).

Since PAHs are absorbed predominantly on particulate matter, we first analyzed concentration of particles. The mean daily  $PM_{10}$  concentration during the study period was  $131.2 \mu\text{g}/\text{m}^3$  with a minimum of 10.80 and a maximum of  $822.4 \mu\text{g}/\text{m}^3$  (Fig. 1). As expected, cold seasons exhibited the highest  $PM_{10}$  concentrations ( $151.73 \pm 19.8 \mu\text{g}/\text{m}^3$ ) while summer and autumn the least  $PM_{10}$  levels ( $121.3 \pm 7.2$  and  $91.0 \pm 6.2 \mu\text{g}/\text{m}^3$ ). The extremely polluted air quality in winter season, is further reflected by the fact that out of 146 sampling days in winter, there were 54 days with  $PM_{10}$  exceeding the 24-h standard for coarse particles of  $150 \mu\text{g}/\text{m}^3$  (Fig. 1). The seasonal  $PM_{10}$  average concentrations were all higher than those measured in other Latin American cities. For example, the winter and summer average  $PM_{10}$  concentrations ( $151.73$  and  $91.0 \mu\text{g}/\text{m}^3$ ) were much higher than those in Mexico City ( $82.1$  and  $45.5 \mu\text{g}/\text{m}^3$ ; Valle-Hernández et al., 2010) and Santiago de Chile ( $129.4$  and  $64.2 \mu\text{g}/\text{m}^3$ ; Sienna and Rosazza, 2006).

The daily concentrations of  $PM_{10}$ -bound total PAHs ranged from  $0.33$  to  $49.33 \text{ ng}/\text{m}^3$  with an average of  $6.68 \pm 2.88 \text{ ng}/\text{m}^3$  (Table 2). These values are similar to those found in other cities worldwide, like Kuala Lumpur (Omar et al., 2002), Mumbai (Kulkarni and Venkataraman, 2000) and Santiago de Chile (Sienna et al., 2005). Overall, LPAHs are dominant due to the high concentrations of NAP. After that, the average concentrations of PAHs have the decreasing order of 4-ring > 3-ring > 6-ring > 5-ring. High concentrations of HPAHs are consistent with most reported in literature (He et al., 2014; Wang et al., 2015).

In agreement with the measured values of  $PM_{10}$ , we observed a strong seasonal variation in the concentrations of PAHs with minimum concentrations during warmer seasons and maximum concentrations during colder seasons. This trend could be due to several reasons. First, atmospheric PAHs can be photodegraded by hydroxyl radicals, which are more abundant during summertime (Totten et al., 2002; Mandalakis et al., 2003). Second, atmospheric

dispersion of particle bound PAHs is strongly affected by meteorological parameters such as wind speed that has been reported as one of the most important influencing factors on PAHs levels in air (Li et al., 2006; van Drooge and Ballesta, 2009). Third, low temperatures suppress the volatilization of PAH, while also promoting their condensation and deposition (Tremolada et al., 2009). Thus, the high concentrations of PAHs measured during autumn and winter may be due to the frequent temperature inversions that reduce atmospheric dispersion. A higher PAH concentration occurring in winter has been reported in many other urban atmospheres. Indeed several other studies from Europe and USA have reported that PAH levels in winter are generally higher by a factor of 1.5–10 than those in summer (Amodio et al., 2009). Similarly, Ray et al. (2012) found that the winter concentration of PAHs in Delhi, India was 25% higher than in summer. In contrast, during the summer season the lowest levels of PAH were observed suggesting the influence of both temperature and wind speed in the degradation and dispersion of PAHs, respectively. Kiss et al. (2001) proposed that volatilization, degradation, and uptake of PAH (into biota) can occur more rapidly during the summer, which can reduce the levels at which they are present.

The concentration of the most known carcinogenic substance BaP (range  $0.12$ – $0.46 \text{ ng}/\text{m}^3$ ) exceeds values found in small urban areas in China ( $0.05 \text{ ng}/\text{m}^3$ , Liu et al., 2013) but are lower than those found in industrial regions in Minas Gerais, Brazil ( $5.6 \text{ ng}/\text{m}^3$ , Menezes and Cardeal, 2012) and Mexico DF ( $4.0 \text{ ng}/\text{m}^3$ , Guzman-Torres et al., 2009). In Argentina, there is no regulation for PAHs but comparing our measurements with the international standard for carcinogenic compounds of  $1 \text{ ng}/\text{m}^3$ , the BaP levels do not exceed this limit even during the worse air quality period.

The prevailing meteorological conditions during the study period in Cordoba city are described in Table 3. This city has a mediterranean climate with markedly different seasons: while winters are dry and cold, summers are very hot, windy and humid, with frequent maximum  $T$  higher than  $30 \text{ }^\circ\text{C}$ .

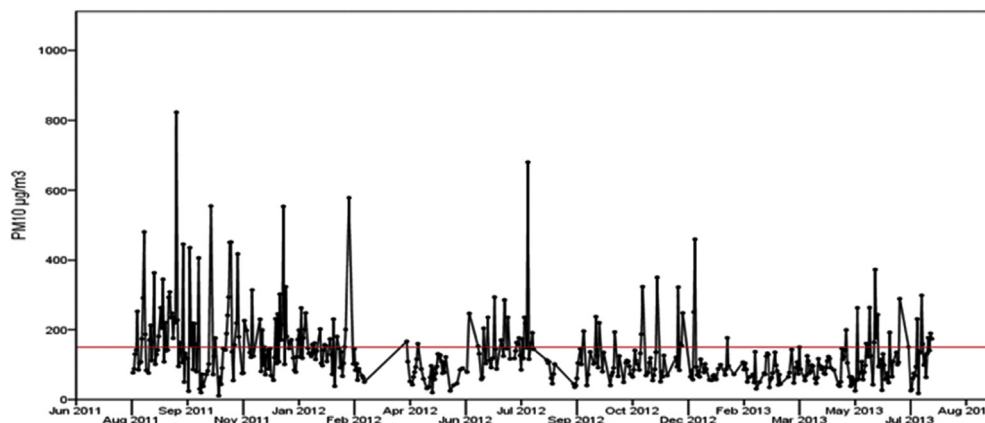


Fig. 1.  $PM_{10}$  concentrations during August 2011 to April 2013, in Cordoba city. Horizontal line represents the air quality standard for  $PM_{10}$  particles.

**Table 2**  
Seasonal concentrations (ng/m<sup>3</sup>) of individual and total PAHs collected in Cordoba, Argentina.

Season	Autumn		Winter		Spring		Summer		Overall mean
	Mean	SE <sup>a</sup>	Mean	SE	Mean	SE	Mean	SE	
NAP	3.220	0.320	4.000	0.396	2.608	0.229	2.094	0.351	2.88
ACE	0.206	0.072	0.269	0.030	0.147	0.010	0.049	0.007	0.16
FL	0.835	0.132	0.767	0.080	0.315	0.065	0.256	0.030	0.49
PHE	0.456	0.040	0.769	0.067	0.495	0.044	0.144	0.027	0.45
ANT	0.014	0.001	0.023	0.002	0.008	0.001	0.004	0.000	0.01
FLU	0.348	0.082	0.658	0.073	0.158	0.012	0.129	0.017	0.30
PYR	0.199	0.044	0.367	0.046	0.077	0.010	0.061	0.005	0.16
BaA	0.315	0.061	0.530	0.051	0.167	0.018	0.148	0.008	0.27
CHR	0.293	0.086	0.515	0.059	0.124	0.020	0.083	0.007	0.23
BbF	0.521	0.116	0.913	0.148	0.504	0.052	0.349	0.030	0.56
BkF	0.171	0.032	0.226	0.022	0.687	0.089	0.341	0.065	0.40
BaP	0.293	0.072	0.455	0.051	0.115	0.018	0.130	0.011	0.23
DBahA	0.122	0.019	0.113	0.011	0.030	0.005	0.035	0.004	0.07
BghiP	0.720	0.097	0.938	0.073	0.385	0.033	0.360	0.022	0.56
TPAHs	7.629	0.788	10.423	0.895	5.793	0.319	4.133	0.378	6.68

<sup>a</sup> SE: standard error.**Table 3**  
Seasonal means of meteorological parameters measured during the study period in Cordoba city.

	Autumn		Winter		Spring		Summer	
	Mean	SE <sup>a</sup>	Mean	SE	Mean	SE	Mean	SE
T (°C)	16.1	0.6	13.5	0.4	20.2	0.4	24.1	0.4
RH (%)	64.3	1.7	51.6	1.6	56.7	1.4	64.4	1.7
AP (hPa)	960.8	0.5	961.7	0.6	959.1	0.5	957.2	0.5
WS (km/h)	12.5	0.5	14.4	0.5	15.6	0.6	13.9	0.6
LH (km)	1243	66.7	1506	72.2	2077	79.8	1828	102.4
R (mm)	0.6	0.5	0.9	0.2	2.7	0.4	3.8	1.2

<sup>a</sup> SE: standard error.

It is already known that meteorological conditions can affect PAHs in terms of their generation, phase partition, accumulation, diffusion, removal, etc. However, each meteorological parameter has different contribution to PAHs concentration. Thus, we first assessed the influence of meteorological parameters on total and individual PAHs, categorized according to their benzene ring number, and calculated their Pearson's correlation coefficient (Table 4). Temperature was the parameter with higher correlation coefficients with total and 2-, 3-, 4-, 5- and 6-ring PAHs. These negative coefficients explain the higher PAH levels found during the cold season. Significant correlations were observed also with atmospheric pressure and wind speed, although with smaller coefficients. The inverse relationship between PAHs and WS shows the predominance of local sources because strong winds brought pollutants out of the study area, whereas low winds allow pollution levels to increase. This is consistent with results observed by Mantis et al. (2005) in Athens, Greece and Hong et al. (2007) in Xiamen, China. The significant and negative coefficients found between lightweight PAHs and surface thermal inversion, i.e. the height of

**Table 4**  
Pearson correlation coefficients between TPAHs or PAHs categorized according to their benzene ring number and selected meteorological parameters. Bold numbers indicate significant correlations (\*\* correlation is significant at the 0.01 level; \* correlation is significant at the 0.05 level).

	TPAHs	2-ring	3-ring	4-ring	5-ring	6-ring
T	<b>-0.43**</b>	<b>-0.30**</b>	<b>-0.53**</b>	<b>-0.29**</b>	<b>-0.45</b>	<b>-0.46**</b>
RH	-0.10	-0.05	<b>-0.13*</b>	-0.10	-0.09	<b>-0.20**</b>
AP	<b>0.23**</b>	<b>0.21**</b>	<b>0.27**</b>	<b>0.15*</b>	<b>0.18**</b>	<b>0.20**</b>
WS	<b>-0.25**</b>	<b>-0.21**</b>	<b>-0.20**</b>	<b>-0.18**</b>	<b>-0.24**</b>	<b>-0.23**</b>
LH	-0.11	<b>-0.15*</b>	<b>-0.19**</b>	0.03	-0.02	-0.14
R	-0.09	-0.09	-0.08	-0.06	-0.05	-0.06

the mixing layer, suggest that these compounds are mainly emitted by local sources.

To quantify the contribution of those meteorological parameters that we already observed have a significant association with PAHs, we run a multivariate regression analysis (Table 5).

Overall, the best predictor model was obtained for 3-ring PAHs, which explained a 41% of the observed concentrations, while the worst model was obtained for 4-ring PAHs that only explained 6% of their concentrations. Temperature was the parameter that affected the most total and individual PAHs. The highest coefficient was observed for TPAHs, meaning a 0.44 ug/m<sup>3</sup> increase in their concentrations with a 1 °C decrease in T. The influence of T was stronger in lightweight than in heavy PAHs, since the former compounds are more easily partitioned in the vapor phase than heavier PAHs. Several other studies already found an inverse relationship between PAHs and T, generally attributed to increases in emissions from heating during colder weather, and reductions in atmospheric degradation of PAHs (Li et al., 2006; Ravindra et al., 2008). Wind speed was also a significant predictor for almost all PAHs, except for 4-ring PAHs. The highest influence of wind speed was observed in 2-ring PAHs, showing a 0.13 ng/m<sup>3</sup> decrease in their concentration for every 1 km/h increase in wind speed. The dilution effect of wind suggests that PAHs originated mainly from local sources, because strong winds would sweep pollutants out of the study area, whereas light winds would allow pollution levels to increase (Hong et al., 2007). Relative humidity was a significant predictor for almost all PAHs, as well. Slight decreases in PAH concentrations were associated to ambient humidity increases. Chetwittayachan et al. (2002) and Hien et al. (2007) also found PAHs concentration negatively correlated with humidity. Finally, the concentration of 4-ring PAHs was the least dependent on meteorological parameters, since their concentration was only marginally affected by atmospheric pressure.

**Table 5**  
Results of the different regression models run to predict total and 2-, 3-, 4-, 5- and 6-ring PAHs concentration.

	Constant	$\beta$ for T	$\beta$ for WS	$\beta$ for RH	$\beta$ for AP	R <sup>2</sup>
TPAHs	22.95	-0.44	-0.27	-0.09		0.38
2-ring	10.86	-0.24	-0.13	-0.04		0.25
3-ring	32.18	-0.09	-0.04	-0.02	-0.03	0.41
4-ring	9.82				0.10	0.06
5-ring	1.41	-0.03	-0.02	-0.01		0.31
6-ring	2.14	-0.04	-0.02	-0.01		0.32

Considering that ambient  $T$  was the most important meteorological parameter affecting PAHs concentrations, we investigated if PAHs on a given day are affected by that day's temperature or the effect of a lagged temperature, that is the previous day's temperature, two previous days, etc. Thus, we run the same multivariate regression with lagged  $T$  as main predictor instead of same day  $T$ . Fig. 2 shows the increase in total PAHs concentration for a  $1\text{ }^{\circ}\text{C}$  decrease in ambient  $T$ , for 10 single day lags. The strongest association was observed two days before the collection of PAHs, thereafter the influence of  $T$  decreases and remains constant up to 10 days before the day PAHs were measured.

In addition the same analysis was done discriminating low and high weight PAHs (Fig. 3). We found that LPAHs (2 and 3-ring PAHs) were more affected by ambient  $T$  corresponding to two days previous the measurement of PAHs, while in heavier PAHs, with 5 and 6 benzene rings, the influence of lag  $T$  was barely noticed. This result is reflecting the higher dependence of LPAHs with ambient  $T$ , due to their low evaporation point. The reason we found stronger associations with ambient  $T$  from one and two previous days than with same day  $T$  is probably related to the fact that when there is a high (or low)  $T$  event, it is expected that  $T$  will remain high (or low) after a few days. In addition, to assess if the relationship between  $T$  and THAPs had a seasonal trend, we run a stratified regression analysis for every season. As expected, for every  $1\text{ }^{\circ}\text{C}$  decrease in ambient  $T$  we found almost twice increase in TPAHs during winter and autumn ( $0.850$  and  $0.676\text{ }\mu\text{g}/\text{m}^3$ ) than during spring and summer ( $0.079$  and  $0.476\text{ }\mu\text{g}/\text{m}^3$ ).

#### 4. Conclusions

Our results indicate that temperature is the most important factor affecting negatively PAHs concentrations in the city of Cordoba. The study demonstrates that not only same day  $T$  is associated to PAH concentrations, but also previous days  $T$  have a strong influence on PAHs. Wind speed is another meteorological factor affecting PAHs concentration.

Climate and land topography are both contributing to increase air pollution in the city of Cordoba. Most of the downtown area is located in a depression, which results in a high concentration of particle-bound PAHs in the lower atmosphere layers. This condition favors the formation of thermic inversion layers, mostly during wintertime, that prevents the dispersion of particles. Thus, the concentration of particles during cold months frequently exceeds the international standards. In fact, we found that the level of TPAHs were significantly higher during cold than warm months and the influence of  $T$  during the cold period was twice the influence of  $T$  during warmer months.

A key limitation of this study is the fact that we did not measure PAH levels on the gas phase, which would contribute to explain the seasonal variations of particle-bound PAHs. However, the fact that

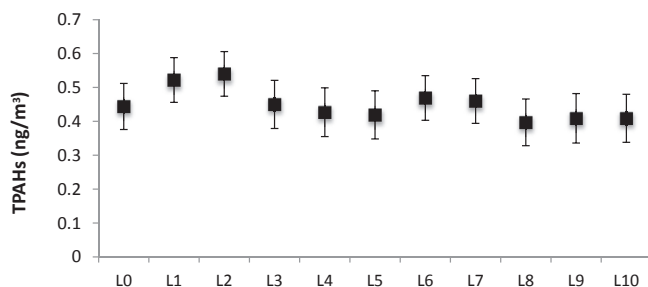


Fig. 2.  $\beta$  coefficients + SD for different lag  $T$  (L) as main predictors of TPAHs on multivariate regression models.

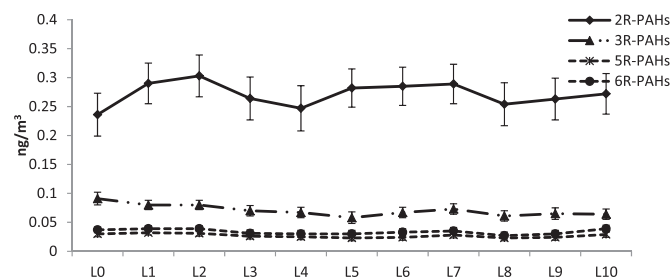


Fig. 3.  $\beta$  coefficients + SD for different lag  $T$  (L) as main predictors of light and heavy PAHs on multivariate regression models.

heavier PAHs are mostly associated to particulate matter, suggest that our estimations are valuable enough from a human health point of view since these compounds are the most carcinogenic ones. This is a fully justified reason to establish a continuous particle-monitoring network in the city of Cordoba.

Despite this weakness, to our knowledge this is the first comprehensive study that quantifies numerically the association between meteorological variables and particle-bound PAHs in and urban environment. Even though the concentration levels of PAHs were lower than the levels observed in a previous study in the same city (Carreras et al., 2013), they are still high enough to pose a risk to human health due to chronic exposure, which is particularly true for carcinogenic PAHs. Our results indicate the need for a continuous research in this field to control for adverse consequences on citizens' health.

#### Conflict of interest

There is no conflict of interest.

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