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Multi-Year PAH Behaviours in Atmospheric Particulates According to Land-Use Type

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ABSTRACT

The present study assessed the multi-year behaviours of high molecular-weight PAHs (four or more benzene ring) (benzo(*a*)anthracene, benzo(a)pyrene (BaP), benzo(*b*)fluoranthene, benzo(*k*)fluoranthene, chrysene, dibenzo(*a,h*)anthracene, and indeno(1,2,3-*cd*)pyrene) that are associated with airborne particulates in five areas of three cities (Seoul, Ulsan, and Gwangyang) and one rural area (Taeahn), utilizing four-year monitoring data. The order of the mean PAH concentrations was as follows (in descending order) residential-commercial complex (Seoul), traffic junction (Seoul), petrochemical plant (Ulsan), iron and steel plant (Gwangyang), and background area (Taeahn). With a few exceptions, individual congeners exhibited a concentration pattern similar to that of the total PAHs. The high PAH concentrations in the residential-commercial complex and traffic junction are likely attributable to the heavy traffic in Seoul, South Korea's capital. The inter-annual and seasonal modulations of PAHs were different for each area type, due to differences in land use. It is suggested that neither total PAHs nor BaP alone seem sufficient to index carcinogenicity associated with exposure to these substances. In most cases, the three meteorological parameters (relative humidity, temperature, and wind speed) exhibited a negative correlation with PAH concentrations, although only a few cases demonstrated a statistical significance at $p < 0.05$.

Keywords: Residential complex; Traffic junction; Petrochemical; Iron and steel industry; Background area.

INTRODUCTION

Polycyclic aromatic hydrocarbons (PAHs) are a class of complex compounds, which reflect a fused ring structure containing two or more benzene rings. These compounds are also defined as persistent organic compounds (van der Gon *et al.*, 2007). Exposure to PAHs has proven to be a matter of particular concern, as many of these compounds are ubiquitous; they have also been potentially linked to adverse health effects. In particular, there is increasing evidence for the ubiquitous presence of PAHs in both urban and suburban settings (Avino *et al.*, 2011; Cazier *et al.*, 2011; Nassar *et al.*, 2011; Zhang *et al.*, 2011; Cheng *et al.*, 2012). PAHs are emitted into the atmosphere through natural combustion as well as from anthropogenic sources (Mastral *et al.*, 2000; Ravindra *et al.*, 2008; Mari *et al.*, 2010; Li *et al.*, 2012). However, the contribution of natural sources to PAH emission into atmospheric environments may be insignificant (Wild and Jones, 1995). These pollutants are present at ambient temperature in the atmosphere, as the gas-phase or adsorbed on atmospheric particulates, depending upon their chemical and physical properties (Ravindra *et al.*, 2008; Tsai *et al.*, 2011; Wang *et al.*, 2011). Among the hundreds of PAH compounds found in the environment, 16 are classified as priority pollutants by the USEPA (2003). Additionally, some of these compounds have been shown to be carcinogens and teratogens (Villalobos-Pietrini *et al.*, 2006; Callén *et al.*, 2011).

The knowledge of typical atmospheric PAH features is suitable to establish control strategies for these pollutants, thereby improving atmospheric air quality. One possible approach gaining this type of information involves the examination of ambient monitoring data (Ravindra *et al.*, 2006). Furthermore, it has been suggested that ambient monitoring data obtained from areas of different source categories may illuminate different behaviours of atmospheric PAH pollution. While many studies have reported the atmospheric PAH levels in various microenvironments such

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as rural, urban and/or industrial areas (Tekasakul *et al.*, 2008; Wu *et al.*, 2010; Hanedar *et al.*, 2011), studies on multi-year atmospheric PAH behaviours according to land-use type are rare in the literature (Noth *et al.*, 2011). Accordingly, the present study was conducted to assess the multi-year behaviours of particulate PAHs in five categorical areas of three cities. This study focuses on the particulate-bound PAHs with higher molecular weights (4 or more benzene ring). Since the high mol ecular-weight PAHs have low solubility in water and a low vapor pressure, they are found almost exclusively in particle phase in the atmosphere (Ravindra *et al.*, 2008). Compared to gaseous PAHs, these particulate-bound PAHs evidence higher carcinogenicity, although they are less acutely toxic (Ravindra *et al.*, 2001). Major anthropogenic sources of atmospheric PAHs include incomplete combustion of fossil fuels, including domestic combustion, automobile fuel combustion, and industrial combustion, as well as a variety of industrial processes, such as petroleum refining and metal smelting (Choi *et al.*, 2007; Ravindra *et al.*, 2008; Ma *et al.*, 2010). To more accurately reflect these source behaviours, the area types selected for the current study included residential-commercial complex, traffic junction, the petrochemical industry, the iron and metal industry, and a background area.

METHODOLOGY

Five area types were selected for the current study (Fig. 1): STJ (Seoul metropolitan city, the capital of South Korea), traffic junction; UPI (Ulsan metropolitan city), petrochemical industry; GIM (Gwangyang city), iron and metal industry; SRC (Seoul metropolitan city), residential-commercial complex; and TBA (Taeahn), a background area. The three cities (Seoul, Ulsan, and Gwangyang) assessed herein have a population of 3.6 million with 3.3 million automobiles, a population of 1.1 million with 0.5 million automobiles, and a population of 0.2 million with 0.06 million automobiles. The background area (Taeahn) has a population of 0.06 million with 0.02 million automobiles. At each site, 24-h air samples were collected on a quarterly basis during 2005, but on a monthly basis since January 1, 2006. One sample was nominally collected on a monthly basis from each survey area and a total of 240 samples were analyzed for this study. Sampling and analysis were conducted in accordance with the Korean Standard Method for PAH measurements. Air sampling was conducted at a nominal flow rate of 0.22 m^3/min for total suspended particulate (TSP), using a medium-volume air sampler. For each sample, air was drawn through an ozone trap and then a quartz filter (8″ × 11″) in order to collect the particulates. A 2″ × 5″ strip of each filter was Soxhlet-extracted. This extract was analyzed using a gas chromatograph (GC)/mass spectrometer (MS, Simadzu QP5050) with a mass selective detector (MSD) and a capillary column (DB-5MS, 60 m × 0.25 mm i.d.). The MSD was run in selected ion monitoring mode (SIM) for optimum sensitivity. Quantification was carried out for the following 7 PAHs, all of which were included on the 16 USEPA priority list: benzo(*a*)anthracene (BaA, 4-ring); benzo(a)pyrene (BaP, 5-ring); benzo(*b*)fluoranthene (BbF, 5-ring); benzo(*k*)fluoranthene (BkF, 5-ring); chrysene (CHR, 4-ring); dibenzo(*a,h*)anthracene (DahA, 5-ring); and indeno(1,2,3-*cd*)pyrene (IcdP, 6-ring). The quality control and quality assurance program included the determination of method detection limits and recovery efficiencies, analysis of laboratory and field blank filters, internal standards in each sample extract, and laboratory surrogate standards. The analytical precision was < 10%. The estimated limits of detection limits derived from the standard deviation of spiked samples were 0.01–0.05 ng/m^3. Air monitoring data was not blank- or recovery-corrected.

Once the data were quality assured, the information was transferred to the Air Quality Management Bureau (AQMB) of the Korean Ministry of Environment. As such, the data presented in this study, comprising 24-h averaged concentrations, were extracted from the AQMB database for the selected monitoring stations over a 4-y period beginning in 2006, when quality assurance was initiated. Meanwhile, the meteorological data employed in the present study were obtained from the meteorological office (MO) of each city. Hourly observations of relative humidity, temperature, and wind speed were obtained from the MOs throughout the study period. The statistical analyses included an F-test and correlation procedures using SAS software (Version 9.1) on a personal computer.

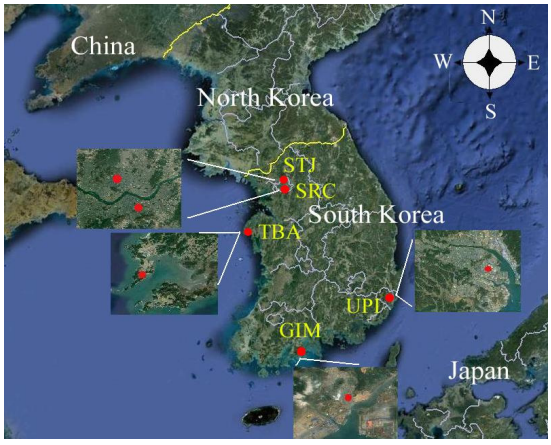

Fig. 1. Location of monitoring sites.

RESULTS AND DISCUSSION

Comparison of Area Types for Ambient PAH Concentrations

The mean concentrations of individual PAHs monitored at five categorical areas are presented in Table 1. Rather than median values, mean values are reported in this table for the purpose of comparison with previous studies. The mean concentrations of total PAHs were higher in two areas of Seoul compared to other areas. With a few exceptions (BaA and CHR for GIM), the mean concentrations of individual PAHs exhibited a pattern similar to that of the total PAHs. As described earlier, Seoul had many more automobiles than other cities; automobiles are one of most important sources of PAHs (Ravindra *et al.*, 2008; Hanedar *et al.*, 2011). The ratios of BbF to BkF were 2.38 and 2.44

Table 1. Mean concentrations (ng/m^3) ± standard deviations of PAHs for the present (2006–2009) and other studies.

Location	BaA	BaP	BbF	BkF	CHR	DahA	IcdP	Σ PAHs
SRC[a]	0.39 ± 0.41	0.55 ± 0.74	1.48 ± 2.24	0.62 ± 0.71	0.95 ± 0.91	0.21 ± 0.23	0.38 ± 0.43	3.70 ± 4.59
STJ[b]	0.43 ± 0.43	0.45 ± 0.39	1.15 ± 1.62	0.47 ± 0.48	0.87 ± 0.72	0.19 ± 0.28	0.33 ± 0.28	3.49 ± 3.49
UPI[c]	0.31 ± 0.42	0.26 ± 0.35	0.50 ± 0.67	0.22 ± 0.24	0.41 ± 0.46	0.16 ± 0.30	0.15 ± 0.16	1.24 ± 1.65
GIM[d]	0.08 ± 0.12	0.08 ± 0.13	0.24 ± 0.42	0.13 ± 0.16	0.53 ± 0.75	0.05 ± 0.10	0.10 ± 0.14	1.05 ± 0.88
TBA[e]	0.06 ± 0.08	0.04 ± 0.06	0.11 ± 0.13	0.10 ± 0.13	0.09 ± 0.13	0.04 ± 0.08	0.06 ± 0.09	0.37 ± 0.47
Istanbul, Turkey[f]	1.4	1.1	1.2	0.4	1.7	NA	NA	NC
Hiroshima, Japan[g]	0.19 ± 0.18	0.52 ± 0.42	NA	NA	0.26 ± 0.22	NA	0.45 ± 0.39	NC
Calcutta, India[h]	22.9 ± 15.2	14.3 ± 9.6	23.7 ± 17.9	NA	NA	NA	NA	NC
Guangzhou, China[i]	1.4 ± 1.3	2.3 ± 2.1	2.6 ± 2.1	2.7 ± 2.3	2.7 ± 2.3	0.3 ± 0.4	2.7 ± 1.9	NC
La Plata, Argentina[j]	0.50	0.75	1.42	0.45	0.59	0.18	0.99	NC

[*] NA, not available; and NC, not calculated.
[a] Residential-commercial complex, Seoul, present study.
[b] Traffic junction, Seoul, present study.
[c] Petrochemical industrial area, Ulsan, present study.
[d] Iron and metal industrial area, Gwangyang, present study.
[e] background area, Taeahn, present study.
[f] Hanedar *et al.* (2011): urban area; and sampling period, September 2006–November 2007.
[g] Tham *et al.* (2008): at campus of Hiroshima University, urban area; and sampling period, January 2006–January 2007.
[h] Karar and Gupta (2006): PAHs in PM_{10} at industrial complex; and sampling period, November 2003–November 2004.
[I] Li *et al.* (2006): urban area; and sampling period, April 2001–March 2002.
[j] Rehwagen *et al.* (2005): petrochemical plants; and sampling period, 1999–2002.

for SRC and STJ, respectively. On the basis of one proposed diagnostic ratio (Pandey *et al.*, 1999; Park *et al.*, 2002), a BbF/BkF ratio larger than 0.5 is suggestive of the presence of automobile emissions. Additionally, automobiles in Seoul account for approximately 84% of PM_{10} emissions, which are associated closely with PAH emissions in this area (Choi, 2012). As such, it is suggested that the automobile exhaust effect might exceed the other sources of PAHs, such as petrochemical and iron and metal industries, in regard to ambient PAH levels. Although, as compared to STJ (traffic junction area), SRC revealed somewhat higher mean concentrations for both the individual and total PAHs, these differences were not statistically significant, with the exception of BaA. This may be explained in that, similar to STJ, SRC is one of the highest traffic areas in Seoul, with slow traffic all day long, even though the area has been officially designated as a residential-commercial complex area. UPI (petrochemical industrial area), which evidenced the third highest mean concentrations for both individual and total PAHs, exhibited a higher mean concentration of total PAHs relative to GIM (iron and metal industrial area). These differences are difficult to explain as a function of industrial type, since the number of automobiles in UPI was significantly higher than in GIM. The combined effects of industrial type and automobiles might result in the observed differences in PAH levels between the two areas. As anticipated, TBA (background area) exhibited the lowest ambient levels of both individual and total PAHs. The mean concentration of total PAHs in TBA was one-tenth that of SRC.

Table 1 also demonstrates the mean concentrations of individual PAHs measured in other studies. The PAH mean concentrations were within the range reported in other studies. With a few exceptions, the PAH concentrations obtained in the present study were slightly or substantially lower than those in Istanbul, Turkey (Hanedar *et al.*, 2011), in Calcutta, India (Karar and Gupta, 2006), in Guangzhou, China (Li *et al.*, 2006), and in La Plata, Argentina (Rehwagen *et al.*, 2005). However, the mean concentrations of three PAHs (BaA, BaP, and CHR) were higher in the present study than in Hiroshima, Japan (Tham *et al.*, 2008). The difference between the current and previous studies of PAH concentrations may be attributable to the combined effects of such emission parameters as emission source type and emission strength and regional environmental conditions, including temperature and dispersion or turbulence variability.

Variation in Annual Mean Concentrations by Area Types

The general trends were evaluated for the individual and total PAH concentrations obtained from five areas over a 4-y period (Fig. 2). The variations in the annual mean PAH concentrations exhibited unpredictable behavior, depending upon both the area type and the types of PAHs. For SRC, the mean concentrations of the three PAHs (BaP, BbF, and BkF) exhibited a decreasing pattern, whereas the mean concentrations of other PAHs fluctuated. With regard to STJ, the mean concentration of BkF only evidenced an apparent increasing trend. For UPI, the mean concentrations of three PAHs (BaA, BbF, and DahA) evidenced a decreasing trend, whereas the other three PAHs (BaP, BkF, and IcdP) exhibited an increasing trend. The variations in mean PAH concentrations evidenced no trend for GIM, but exhibited an increasing trend for most of the PAHs observed in TBA, likely due to the increase in the number of automobiles (Choi, 2012). Nevertheless, those variations in the annual PAH concentrations monitored in the five areas are not reasonably explicable, as the necessary information could not be obtained.

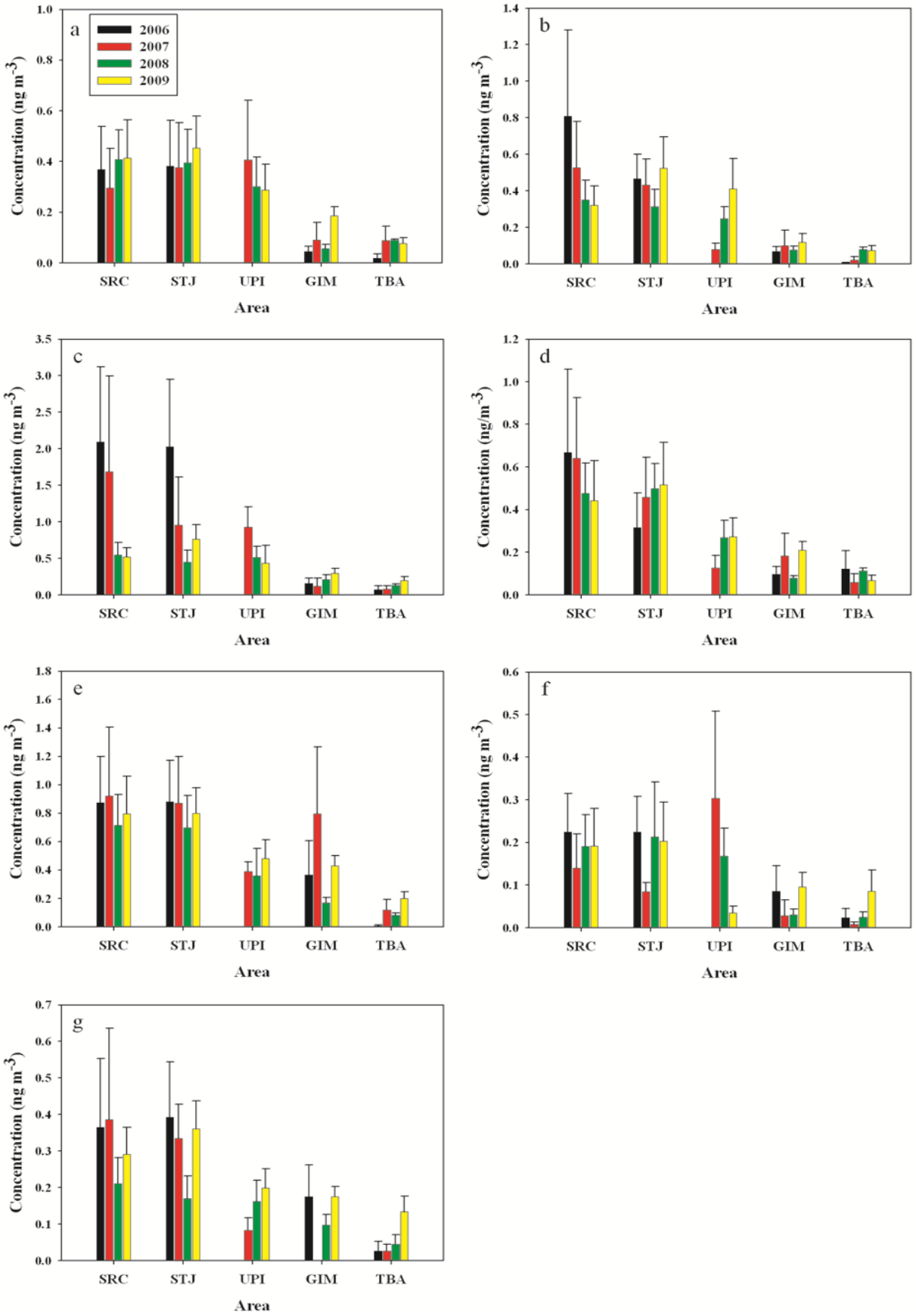

Fig. 2. Annual mean concentrations (ng/m^3) and standard errors of PAHs for five categorical areas: a, BaA; b, BaP; c, BbF; d, BkF; e, CHR; f, DahA; g, IcdP; and h, sum of 7 PAHs.

Seasonal Variations in PAH Concentrations

The seasonal mean PAH concentrations for five categorical areas are demonstrated in Table 2. It is worth noting that different categorical areas exhibited different seasonal variations. Unlike the annual variations, in the SRC and STJ the concentrations of both individual and total PAHs exhibited a distinct seasonal variation, with a maximum in the winter and a minimum in the summer. The mean concentration of total PAHs in the winter was 4.2 times higher than that in the summer for the SRC, and it was 6.7 times higher for the STJ. This seasonal pattern is consistent with the results reported by several researchers (Tham *et al.*, 2008; Ma *et al.*, 2010; Hanedar *et al.*, 2011; Nassar *et al.*, 2011). The elevated PAH concentrations in the winter are attributed to the higher PAH emissions and the prevailing meteorological conditions. More heating fuel is typically consumed in Korean urban areas during wintertime, thereby elevating the ambient air levels of PAHs in urban areas (Lee *et al.*, 2006). Another possible reason for the higher PAH emissions in the winter is the lower combustion efficiency of gasoline or other fuels during the winter, thereby causing higher concentrations in the ambient air. This is supported by Deng *et al.* (2011), who found that high emissions from motor vehicles are associated with cold ambient temperatures. Higher atmospheric stability is generally observed in South Korea during the winter due to the more frequent occurrence of low inversion heights, thereby elevating ground-level air pollution (Lee *et al.*, 2008). Moreover, a seasonal rain front usually occurs in South Korea during the summer months, thereby increasing the wet deposition of particulate PAHs (Vu *et al.*, 2011).

As compared with the SRC and STJ, the other categorical areas revealed less distinct seasonal variations in the concentrations of individual or total PAHs. The mean concentration of total PAHs in the UPI were somewhat higher in the fall than in the winter, although the summer mean concentration was lower than the winter mean concentration. For GIM, the mean concentrations of total PAHs did not differ significantly between seasons. For the TBA, the spring mean concentrations of total PAHs were even higher than the mean concentrations in winter. The individual PAHs for these areas also showed unpredictable seasonal variations. Less distinct seasonal behavior of PAH concentrations for UPI and GIM may be due to the industrial emissions, as industrial activities are not significantly influenced by season. In addition, the seasonal variation in meteorological factors, such as wind speed, humidity, and mixing height, and long-range transport of PAHs might also result in the seasonal variations in PAH concentrations.

Among the particulate PAHs, BaP has been the subject of special concern because of its high carcinogenicity (Ravindra *et al.*, 2001). This pollutant has been frequently identified as an indicator of air quality associated with overall PAH carcinogenicity (EUD, 2004). For example, the EUD (2004) has proposed a target value of 1 ng/m^3 BaP for the total content in the PM_{10} fraction averaged over a calendar year. The mean BaP concentrations obtained from the five categorical areas in the present study did not exceed the European ambient BaP guideline value. However, the high standard deviation values for two categorical areas (SRC and STJ) during the winter indicated that, for a certain portion of monitoring days, the BaP concentrations have exceeded the European ambient BaP guideline value. Accordingly, the need for a strategy to reduce atmospheric BaP concentrations at the two categorical areas is suggested to minimize the health risks of residents associated with atmospheric BaP exposure.

If BaP only is used to evaluate PAHs' carcinogenic property, this evaluation is likely to be underestimated due to the carcinogenicity of other PAHs, even though it is considered to have the highest carcinogenicity among ambient PAHs (Ravindra *et al.*, 2001). Additionally, BaP has a relatively short life time under simulated sunlight conditions (ca. 5.3 h) due to photochemical reactions (Perraudin *et al.*, 2007). In order to tackle this disadvantage of using BaP as a representative PAH for carcinogenicity, the BaP-equivalent carcinogenic power (BaPE) is used (Liu *et al.*, 2009; Kong *et al.*, 2011). Table 3 reveals the BaPE levels for five categorical areas according to season. For both the SRC and STJ, the seasonal trends for the BaPE were similar to the seasonal BaP trends, with the maximum average BaPE values of 1.65 and 1.43 ng/m^3 for the winter, respectively, and the minimum average values of 0.54 and 0.19 ng/m^3 for the summer. The wintertime average BaPE values substantially exceeded the European ambient BaP guideline value, thereby justifying the use of BaPE values for the evaluation of atmospheric PAH carcinogenicity. Moreover, it is noteworthy that, for the GIM, the wintertime average BaPE value was at a minimum, while the wintertime mean concentration of total PAHs was reversed. This suggests that the concentration of total PAHs is not appropriate as a sole measure for the evaluation of PAHs' carcinogenic properties.

Relationship of PAH Concentrations with Meteorological Parameters

Table 4 indicates the correlation of the concentrations of individual and total PAHs with three meteorological parameters (relative humidity, temperature, and wind speed). The meteorological data were obtained from the MOs of each city, as site-specific meteorological data were unavailable. In most cases, the three meteorological parameters were negatively correlated with PAH concentrations, although only a few cases demonstrated a statistical significance at $p < 0.05$. This result is consistent with those of previous studies (Karar and Gupta, 2006; Tham *et al.* 2008). Strong wind speed would exert a dilution effect on the PAH concentrations (Karar and Gupta, 2006). High temperature would increase the evaporation of particulate PAHs from the particle to gas phase, whereas low temperature would increase the condensation of gaseous PAHs onto particles (Tham *et al*., 2008). In contrast to the present study and a previous study (Karar and Gupta, 2006), Mastral *et al.* (2000) reported a positive correlation between relative humidity and the concentrations of total PAHs. The reason for this difference remains unclear, although Mastral *et al.* (2000) has suggested that the positive correlation between relative humidity and PAH concentrations may be attributable to an elevated deposition of water vapor on the particles.

Table 2. Mean PAH concentrations (ng/m^3) ± standard deviation as a function of the season for five categorical areas.

PAHs	Area	Concentration			
		Spring	Summer	Fall	Winter
BaA	SRC	0.53 ± 0.41	0.06 ± 0.03	0.27 ± 0.35	0.72 ± 0.45
	STJ	0.27 ± 0.13	0.11 ± 0.04	0.34 ± 0.34	0.99 ± 0.50
	UPI	0.23 ± 0.29	0.21 ± 0.15	0.40 ± 0.64	0.41 ± 0.40
	GIM	0.07 ± 0.06	0.15 ± 0.18	0.11 ± 0.11	0.01 ± 0.05
	TBA	0.02 ± 0.03	0.05 ± 0.06	0.11 ± 0.12	0.06 ± 0.05
BaP	SRC	0.45 ± 0.32	0.44 ± 1.11	0.37 ± 0.42	0.94 ± 0.58
	STJ	0.37 ± 0.17	0.11 ± 0.05	0.51 ± 0.47	0.84 ± 0.42
	UPI	0.14 ± 0.14	0.14 ± 0.10	0.41 ± 0.55	0.34 ± 0.31
	GIM	0.17 ± 0.19	0.05 ± 0.04	0.11 ± 0.14	0.02 ± 0.07
	TBA	0.01 ± 0.03	0.03 ± 0.03	0.08 ± 0.08	0.04 ± 0.06
BbF	SRC	0.77 ± 0.76	0.36 ± 0.74	1.10 ± 1.81	3.68 ± 4.03
	STJ	0.55 ± 0.50	0.21 ± 0.11	1.31 ± 1.80	2.55 ± 2.35
	UPI	0.39 ± 0.61	0.08 ± 0.13	0.83 ± 0.86	0.70 ± 0.55
	GIM	0.21 ± 0.17	0.21 ± 0.27	0.10 ± 0.16	0.44 ± 0.86
	TBA	0.02 ± 0.05	0.08 ± 0.09	0.24 ± 0.17	0.09 ± 0.10
BkF	SRC	0.50 ± 0.33	0.41 ± 0.89	0.37 ± 0.47	1.21 ± 0.78
	STJ	0.32 ± 0.15	0.12 ± 0.09	0.45 ± 0.42	1.00 ± 0.65
	UPI	0.20 ± 0.14	0.12 ± 0.10	0.29 ± 0.34	0.29 ± 0.25
	GIM	0.14 ± 0.12	0.11 ± 0.10	0.21 ± 0.27	0.07 ± 0.07
	TBA	0.06 ± 0.08	0.03 ± 0.04	0.14 ± 0.17	0.17 ± 0.18
CHR	SRC	1.06 ± 0.73	0.34 ± 0.08	0.60 ± 0.64	1.98 ± 1.21
	STJ	0.68 ± 0.28	0.22 ± 0.08	0.76 ± 0.61	1.82 ± 0.77
	UPI	0.27 ± 0.41	0.36 ± 0.28	0.34 ± 0.44	0.69 ± 0.64
	GIM	0.27 ± 0.23	0.36 ± 0.48	0.23 ± 0.22	1.26 ± 1.49
	TBA	0.10 ± 0.16	0.08 ± 0.07	0.14 ± 0.18	0.02 ± 0.03
DahA	SRC	0.18 ± 0.19	0.07 ± 0.07	0.14 ± 0.18	0.43 ± 0.35
	STJ	0.15 ± 0.19	0.07 ± 0.05	0.15 ± 0.18	0.40 ± 0.48
	UPI	0.07 ± 0.11	0.09 ± 0.20	0.12 ± 0.19	0.37 ± 0.53
	GIM	0.02 ± 0.03	0.07 ± 0.10	0.04 ± 0.09	0.08 ± 0.17
	TBA	0.06 ± 0.16	0.03 ± 0.05	0.01 ± 0.01	0.04 ± 0.07
IcdP	SRC	0.32 ± 0.18	0.06 ± 0.05	0.23 ± 0.30	0.88 ± 0.72
	STJ	0.31 ± 0.11	0.14 ± 0.08	0.31 ± 0.29	0.57 ± 0.44
	UPI	0.07 ± 0.07	0.15 ± 0.09	0.22 ± 0.24	0.18 ± 0.14
	GIM	0.15 ± 0.15	0.13 ± 0.18	0.10 ± 0.10	0.01 ± 0.02
	TBA	0.04 ± 0.07	0.06 ± 0.09	0.07 ± 0.11	0.06 ± 0.11
Σ PAHs	SRC	3.80 ± 2.42	1.53 ± 2.65	3.08 ± 3.49	6.36 ± 7.19
	STJ	2.64 ± 1.05	0.98 ± 0.31	3.82 ± 3.21	6.52 ± 4.79
	UPI	0.86 ± 1.09	0.48 ± 0.63	1.91 ± 2.32	1.71 ± 1.79
	GIM	1.02 ± 0.62	1.07 ± 0.61	0.84 ± 0.58	1.26 ± 1.47
	TBA	0.23 ± 0.38	0.33 ± 0.37	0.69 ± 0.61	0.23 ± 0.36

Table 3. BaP-equivalent carcinogenic power (BaPE, ng/m^3) for five categorical areas according to season*.

Season	BaPE				
	SRC	STJ	UPI	GIM	TBA
Spring	0.70	0.56	0.24	0.22	0.06
Summer	0.54	0.19	0.23	0.13	0.06
Fall	0.59	0.77	0.60	0.17	0.12
Winter	1.65	1.43	0.67	0.08	0.09

* BaPE = BaA × 0.06 + BbF × 0.07 + BkF × 0.07 + BaP + DahA × 0.6 + IcdP × 0.08 (Cecinato, 1997).

CONCLUSIONS

The results of the current study demonstrated that high molecular-weight PAH pollution varied with the land-use type. In particular, the industrial areas did not consistently evidence higher PAH pollution than the residential-commercial or other type of land uses. For example, the mean concentrations of total PAHs were higher in two areas

Table 4. Pearson correlation coefficients of PAH concentrations with meteorological parameters for five categorical areas*.

PAHs	Area	*R*, correlation coefficient		
		Relative humidity	Temperature	Wind speed
BaA	SRC	**–0.48**	**–0.69**	0.04
	STJ	**–0.38**	**–0.67**	0.08
	UPI	–0.04	–0.12	**–0.36**
	GIM	0.02	**–0.29**	–0.03
	TBA	–0.14	**–0.34**	–0.21
BaP	SRC	**–0.44**	**–0.73**	–0.19
	STJ	–0.15	**–0.82**	–0.14
	UPI	0.01	–0.10	–0.19
	GIM	–0.21	0.19	0.07
	TBA	–0.20	**–0.38**	–0.12
BbF	SRC	**–0.40**	**–0.72**	–0.11
	STJ	–0.18	**–0.75**	–0.13
	UPI	–0.18	**–0.36**	–0.21
	GIM	–0.17	–0.06	0.11
	TBA	–0.11	**–0.37**	–0.11
BkF	SRC	**–0.40**	**–0.68**	–0.17
	STJ	–0.11	**–0.76**	–0.05
	UPI	–0.12	**–0.30**	–0.14
	GIM	–0.07	0.18	0.02
	TBA	**–0.35**	**–0.49**	–0.19
CHR	SRC	**–0.44**	**–0.85**	–0.05
	STJ	**–0.28**	**–0.86**	–0.15
	UPI	0.01	–0.19	**–0.42**
	GIM	–0.03	0.04	–0.05
	TBA	–0.10	–0.20	–0.13
DahA	SRC	**–0.21**	**–0.49**	0.15
	STJ	**–0.43**	**–0.41**	–0.11
	UPI	–0.13	–0.14	–0.12
	GIM	–0.12	–0.03	0.13
	TBA	–0.06	**–0.29**	–0.08
IcdP	SRC	**–0.36**	**–0.78**	0.02
	STJ	–0.09	**–0.48**	**–0.32**
	UPI	0.04	–0.12	–0.13
	GIM	–0.08	**0.31**	0.01
	TBA	–0.13	**–0.30**	–0.04
ΣPAHs	SRC	–0.18	**–0.21**	–0.06
	STJ	**–0.29**	–0.18	–0.04
	UPI	**–0.24**	**–0.29**	–0.08
	GIM	–0.17	**–0.20**	–0.09
	TBA	–0.17	–0.18	–0.06

* Bold number represents statistical significance at $p < 0.05$.

of Seoul relative to other areas. With a few exceptions, the mean concentrations of individual PAHs exhibited a pattern similar to the total PAHs. It was suggested that the effects of automobile exhaust would exceed whose of the other PAH sources, such as petrochemical and iron and metal industries, on ambient PAH levels. As anticipated, the background area evidenced the lowest ambient levels of both individual and total PAHs. The variations in the annual mean high molecular-weight PAH concentrations evidenced unpredictable behavior, depending upon both the types of areas and PAHs. Different categorical areas exhibited different seasonal variations. Our results indicated that the use of the concentration of total PAHs or BaP is not appropriate as a sole measure for the evaluation of PAHs' carcinogenic properties.

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