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Environmental Exposure to Polycyclic Aromatic Hydrocarbons and Total Suspended Particulates in a Taiwanese Temple

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Polycyclic aromatic hydrocarbons (PAHs), a large group of highly carcinogenic and mutagenic hydrocarbons generated mainly during carbonization or incomplete combustion (Bjϕrseth and Ramdahl, 1985), have been given more attention due to their presence in the indoor atmospheres of houses and workplaces (Viau et al, 2000; Angerer et al, 1997). Burning incense during worshipping ceremonies or even on a daily basis is an important religious rite for most Chinese and it also is done in numerous families and almost all temples in Taiwan. The incense smoke emitted has been proven to contain PAHs (Schoenotol and Gibbard, 1967). The mutagenic effect of Chinese incense smoke has been also identified by Ames test (Rasmussen, 1987). Furthermore, epidemiologic studies have shown an increase in cancer incidence among workers exposed to PAHs (Kawai et al. 1967; Gibbs 1985). Incense burning, however, results in non-stop emissions of PAHs due to its nature of slow and incomplete combustion, and thereby emission from it should be studied in more detail. Unfortunately, few studies have been conducted on this emission source.

This study reports on indoor and outdoor exposure to PAHs and total suspended particulate (TSP) in a Taiwanese temple. The realistic concentrations of the investigated contaminants obtained here were also compared with some values that were reported by other researchers or promulgated by certain countries and organizations (Sheldon et al. 1993; Meek et al. 1994; Slooff et al. 1989; DFG 1999; ATSDR, 1995). Our results could provide information useful in considering risks or regulations for indoor PAH and TSP pollutants generated by incense burning inside Taiwanese temples.

MATERIALS AND METHODS

The investigated Taiwanese temple with an interior volume of 216 m^3 was located in the suburban area of Tainan City in southern Taiwan. The procedure for the sampling and the analysis of each sample was based on previous study (Lee et al. 1995). Ambient air samples for both particulate- and gas-phase PAHs were collected with several standard semi-volatile

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sampling trains (Model PS-1, General Metal Works, USA) and sampled during daytime from 8:00 AM to 5:00 PM. The glass fiber filter (Cat. 1820-101, Whatman International, UK) mounted in PS-1 was used to collect TSP and the particulate PAHs. The glass cartridge which was also mounted in PS-1, and contained a 5-cm polyurethane foam (PUF) plug followed by 3-cm XAD-2 resin (20-60 mesh, Sigma Chemical, USA), and finally a 2-cm PUF plug was used to collect the gas-phase PAHs. Prior to use, glass fiber filters were cleaned with distilled-deionized water and heated at 450 °C, and PUF cartridges were cleaned by sequential Soxhlet extractions with distilled-deionized water, methanol, dichloromethane and a mixture of n-hexane/acetone (50/50 v/v) and finally dried to remove residual solvent at 45 °C. Glass fiber filters were weighed before and after sampling to determine the amounts of particles collected. During sample transportation and storage, PUF plug and resin were stored in clean screw-capped jars with Teflon cap liners while glass fiber filters were stored on a pre-baked glass plate and wrapped with aluminum foil.

Twenty-one PAHs, naphthalene (Nap), acenaphthylene (AcPy), acenaphthene (Acp), fluorene (Flu), phenanthrene (PA), anthracene (Ant), fluoranthene (FL), pyrene (Pyr), cyclopenta[c,d]pyrene (CYC), benz[a]anthracene (BaA), chrysene (Chr), benzo[b]fluoranthene (BbF), benzo[k]fluoranthene (BkF), benzo[e]pyrene (BeP), benzo[a]pyrene (BaP), perylene (PER), indeno[1,2,3-cd]pyrene (IND), dibenz[a,h]anthracene (DBA), benzo[b]chrysene (BbC), benzo[ghi]perylene (BghiP), and coronene (Cor) were identified and quantified for each collected sample according to the following experimental procedure: after final weighing, each sample was placed into a Soxhlet extractor and extracted for 24 hr with a mixture of n-hexane/dichloromethane (50/50 v/v). The extract was then concentrated under ultra pure nitrogen, cleaned up and re-concentrated to exactly 0.5 mL using a procedure similar to those presented in the previous study (Lee, 1991). Finally, the samples were placed in the auto-sampler (Model 3365, Hewlett Packard, USA) and analyzed by a gas chromatography/ mass spectrometer (GC/MS) (Model 5890/5972, Hewlett Packard, USA). The injection volume was 1 μL. The capillary column used was HP Ultra2 (50 m × 0.32 mm × 0.17 μm).

Analytical-reagent grade methanol, dichloromethane, n-hexane and acetone (Merck, Germany) were used as the solvents. A PAH standard containing 16 PAH compounds (Concentration: 100-2000 μg/mL) (Mixture 610-M, Supelco, USA) and five individual PAH compounds (Concentration: 10 μg/mL) (Merck, Germany) were used to create calibration curves. The recovery efficiency of PAHs ranged from 0.78 to 0.95 and averaged 0.86. The method detection limits (MDL) were between 0.025 ng (BaA) and 0.742 ng (Nap).

RESULTS AND DISCUSSION

Table 1 gives the total-PAH concentrations and the particulate-phase distributions in the ambient air of inside, outside the investigated temple, and intersection with heavy traffic (Lee et al. 1995). PAH concentrations found in the indoor air of the temple ranged from 4739 to 11130 ng/m^3 with a mean concentration (± standard deviation) of 8888 ± 3598 ng/m^3. The mean value of total-PAH concentrations was approximately 19 times higher than that found outside the temple (468 ± 156 ng/m^3), and close to that found at the traffic-source (8110 ng/m^3). Table 1 also shows the particulate-phase PAHs accounted only for 3.2% of total-PAHs, which was quite different from those measured at the intersection (46.1%). These results implied that there were high concentrations of total-PAHs within the temple, and their sources probably differed from traffic-source.

Table 1. Total-PAH concentrations and particulate-phase distributions in the ambient air of various sampling sites.

Sampling site	Total-PAH concentration		Particulate-phase distribution	
	Range (ng/m^3)	Mean (ng/m^3)	Range (%)	Mean (%)
Inside the temple[a]	4739-11130	8888	2.42-3.71	3.20
Outside the temple[a]	360-647	468	5.56-18.7	11.7
Traffic-source[b]	4500-11300	8110	28.3-67.3	46.1

[a] Measured in this study (n = 6); R. H. = 56.5-81.0%; Temp. = 28.0-30.0 °C

[b] Results reported by Lee et al. (1995) (n = 4)

Table 2. TSP concentrations in the ambient air of various sampling sites and National Air Quality Standard (NAQS) in Taiwan.

Sampling site	TSP concentration		
	Range (μg/m^3)	Mean (μg/m^3)	NAQS[c]
Inside the temple[a]	429-991	790	1. yearly geometric mean: 130 μg/m^3. 2. 24-hr mean value: 250 μg/m^3.
Outside the temple[a]	59.7-87.1	72.5	
Traffic-source[b]	177-338	255	

[a] Measured in this study (n = 6); R. H. = 56.5-81.0%; Temp. = 28.0-30.0 °C

[b] Results reported by Lee et al. (1995) (n = 4)

[c] Promulgated by EPA, Taiwan (1999)

Table 2 shows the TSP concentrations in the indoor air of the investigated temple were between 429 and 991 μg/m^3 and averaged 790 μg/m^3. The mean TSP concentration was approximately 3 and 11 times higher than that of the traffic-source (255 μg/m^3) (Lee et al. 1995) and outside the temple (72.5 μg/m^3), and significantly exceeded both the yearly geometric mean (130 μg/m^3) and the 24-hr mean value (250 μg/m^3) of National Air Quality Standard (NAQS) (Taiwan EPA 1999) in Taiwan. However, NAQS is set for ambient not indoor air so it is just a reference standard in this study.

The PAH mass adsorbed on the air particulate and normalized by the particle mass is defined as particle-bound PAH composition (μg/g). Table 3 shows the particle-bound composition in the indoor air of the investigated temple ranged from 271 to 423 μg/g and averaged (± standard deviation) 359 ± 79 μg/g, which means approximately 1/2 and 1/43 of that of outside the temple (720 ± 295 μg/g) and the traffic-source (15400 μg/g) (Lee et al. 1995). To our knowledge, few countries have considered TSP standards along with toxic substances, such as the carcinogenic PAHs in the ambient particulate. This is probably limited by insufficient data on particle-bound PAHs to characterize the sources, evaluate the health effects and assess the risks.

Table 3. Particle-bound total-PAH compositions in the ambient air of various sampling sites.

Sampling site	Particle-bound total-PAH composition (μg/g)	
	Range	Mean (± standard deviation)
Inside the temple[a]	271-423	359 ± 79
Outside the temple[a]	500-1058	720 ± 295
Traffic-source[b]	5210-29200	15400

[a] Measured in this study (n = 6); R. H. = 56.5-81.0%; Temp. = 28.0-30.0 °C
[b] Results reported by Lee et al. (1995) (n = 4)

Figure 1 illustrates the profile of mean gas-phase individual-PAH concentrations in the indoor air of the investigated temple. The volatile PAHs, Nap (MW=128), AcPy (MW=152), Acp (MW=154) and FL (MW=202), had higher concentrations. AcPy was the principal PAH contaminant with a mean concentration (± standard deviation) of 5511 ± 2366 ng/m^3, representing on average 62% of total-PAHs, followed by Nap, Acp, FL and PA at 1367 ± 538, 818 ± 409, 498 ± 227, and 226 ± 87 ng/m^3. All other PAHs had individual mean concentration below 70 ng/m^3. This is a unique and characteristic pattern found in the indoor air of the investigated temple.

Figure 2 demonstrates the profile of mean particle-bound individual-PAH compositions in the indoor air of the investigated temple. In the descending order of the compositions (± standard deviation), the first five were DBA (MW=278), BghiP (MW=276), BaA (MW=228), CYC (MW=228) and BbC (MW=278) at 50.4 ± 57.4, 48.8 ± 10.2, 42.2 ± 32.9, 41.4 ± 37.2, and 33.7 ± 6.7 μg/g. The mean particle-bound individual-PAH compositions of CYC and BaA were higher than that expected because CYC and BaA were not heavier PAHs among the 21 investigated. This phenomenon might be an

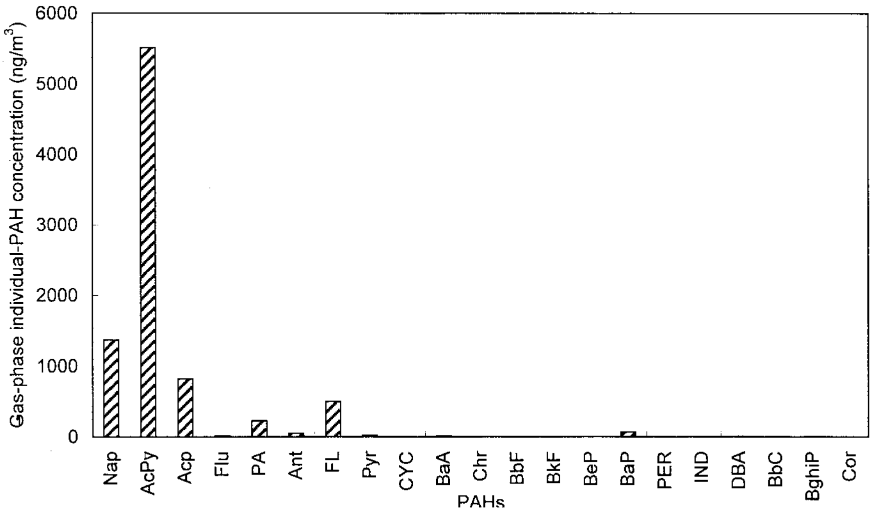

Figure 1. Mean gas-phase individual-PAH concentration (ng/m^3) in the indoor air of the investigated temple.

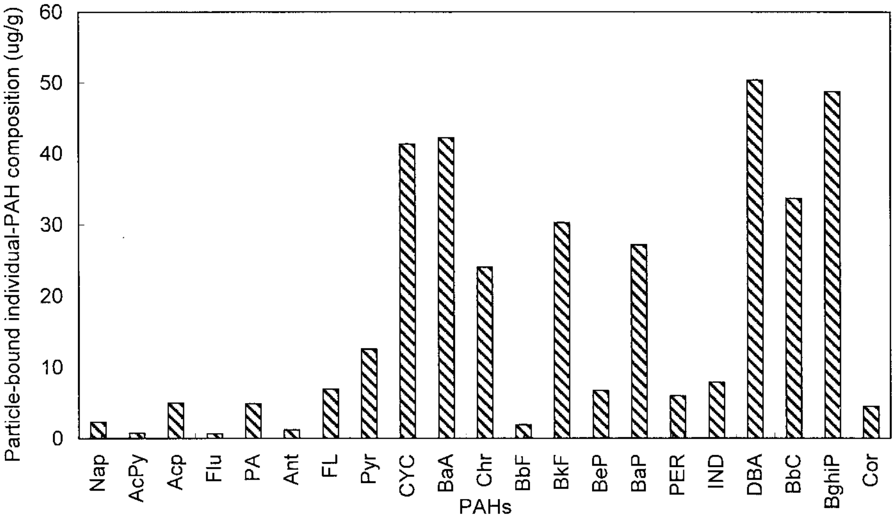

Figure 2. Mean particle-bound individual-PAH composition (μg/g) in the indoor air of the investigated temple.

important marker for the PAH emission of Taiwanese incenses.

Table 4 lists the comparison of PAH concentrations of various emission sources and PAH limits. The PAH concentrations measured in this study should be considered as high. In fact, the level of measured indoor BaP concentrations (57.6-98.2 ng/m^3) was much higher than that of residences with tobacco smoking (2.2 ng/m^3) and without specified indoor combustion source (0.83 ng/m^3) in California (Sheldon et al. 1993), and even exceeded that reported for cities near aluminum smelters in Canada (3-49 ng/m^3) (Meek et al. 1994). The indoor BaP concentrations measured also largely exceeded the maximum permissible risk level (1 ng/m^3 BaP) in the ambient air, which was suggested by Slooff et al. (1989) based on the carcinogenic potential of inhaled particulate PAHs. However, the level of measured indoor BaP concentrations did not exceed the German technical exposure limit (TRK value) for workplaces excluding pencil pitch production (2000 ng/m^3) (DFG 1999). Furthermore, the mean total-PAH concentration was only approximately 1/22 of 0.2 mg/m^3 that was a legally enforceable limit established by the Occupational Safety and Health Administration (OSHA) (ATSDR 1995).

Table 4. Comparison of PAH concentrations of various emission sources and PAH limits (ng/m^3)

Emission sources or PAH limits	BaP concentration	Mean total-PAH concentration
Incense Burning[a]	57.6-98.2	8888
Tobacco Smoking[b]	2.2	
No specified indoor combustion source[b]	0.83	
Aluminum smelters[c]	3-49	
Ambient air[d]	1	
Workplaces excluding pencil pitch production (DFG)[e]	2000	
Coal tar production (OSHA)[f]		200000

[a] Concentrations were 8-hr averages and measured in this study; R. H. = 56.5-81.0%; Temp. = 28.0-30.0 °C

[b] Concentrations were 24-hr averages and reported by Sheldon et al. (1993)

[c] Reported for cities near aluminum smelters in Canada (Meek et al. 1994)

[d] Maximum permissible risk level of BaP in the ambient air, suggested by Slooff et al. (1989)

[e] German technical exposure limit (TRK value) (DFG 1999)

[f] Reported by ATSDR (1995)

Results in this study indicate that people acting inside the investigated Taiwanese temple are exposed to high levels of both PAHs and TSP from indoor air pollution caused by incense burning. We suggest that it would be necessary to undertake a comprehensive epidemiologic study, including biological monitoring, to assess the potential harmfulness related to indoor exposure in Taiwanese temples. Of course, improving the ventilation of a Taiwanese temple is the most elementary requirement.

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