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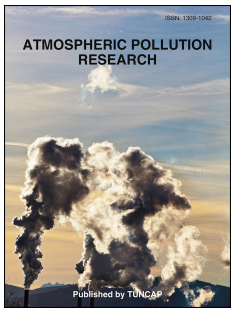

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Ambient $PM_{2.5}$, Polycyclic Aromatic Hydrocarbons and Biomass Burning Tracer in Mae Sot District, Western Thailand

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Abstract— Levoglucosan, as a biomass tracer, and sixteen polycyclic aromatic hydrocarbons (PAHs) in $PM_{2.5}$ ambient air samples collected from Tak Province during smoke and non-smoke episodes were analyzed. The average $PM_{2.5}$, levoglucosan and total PAHs concentrations in the smoke episode were 61.64 ± 22.85, 1.00 ± 0.41 µg/m^3 and 6.32 ± 2.26 ng/m^3, respectively. This figures were significantly higher than those recorded during the non-episode (13.76 ± 5.58, 0.12 ± 0.03 µg/m^3 and 2.59 ± 0.15 ng/m^3, respectively). The predominant PAHs proportions were comprised of Phenanthrene and Benzo[*ghi*]perylene and levoglucosan concentrations revealed a strong correlation with $PM_{2.5}$ concentrations, which indicated the source of $PM_{2.5}$ from biomass burning. Toxicity equivalent (TEQ_{BaP}) and the mutagenic equivalent (MEQ_{BaP}) levels during the smoke episode were significantly higher than those in the non-smoke episode. Furthermore, lifetime lung cancer risk recorded during smoke episode exceeded the acceptable cancer risk that has been recommended by US-EPA. These results suggest that this area was not only exposed to PAHs that originated from traffic combustion, but was also exposed from biomass burning emissions, particularly during biomass burning season when there is an increased risks of cancer and mutation. Although the exposure time in this area is relatively short, the high dose period of exposure occurs repetitively every year. In addition, backward trajectories showed that most of the air mass was generated from western region of Thailand and they were throughout the burning region not only emitted from local areas, but also from outside the country during the smoke episode.

Keywords: $PM_{2.5}$, PAHs, Levoglucosan, Biomass burning tracer, Lung cancer risk

1. INTRODUCTION

The northern region of Thailand is well known for experiencing severe smoke-caused air pollution every year during the dry season (Wiwatanadate and Liwsrisakun, 2011). Open burning is one of the crucial issues in northern Thailand, Laos and Myanmar and in some areas of Vietnam and Cambodia. (Streets et al., 2003; Hyer et al., 2010; Lee et al., 2011; Huang et al., 2013; Popovicheval et al., 2017). The pivotal instances of open burning activities that occur in the dry season include agriculture waste burning, rice straw burning and the lighting of wildfires (Garivait et al., 2008; Chantara et al., 2012). Traditional slash-and-burn agricultural processes are in part the causes of this problem. This traditional practice is hard to change because it is an inexpensive method that farmers have used to clear surface biomass for generations, leading to faster crop rotation, and enabling farmers to limit the presence of undesirable weeds, pests and plant-based diseases (Pongpiachan, 2016; Kim Oanh et al., 2011a). Biomass burning is considered a potential major source of toxic compounds releases into the atmosphere such as particulate matters (PM), polycyclic aromatic hydrocarbons (PAHs), water-soluble organic carbon (WSOC), elemental carbon (EC), organic carbon, (OC), anhydrosugars and volatile organic carbons (VOC) (Lemieuxa et al., 2004; Estrellan and Iino, 2010; Chuang et al., 2013; Zhu et al., 2015). These compounds produce a local, regional, and global impacts on air quality (Chen et al., 2017). At impact present, fine particles of smaller than 2.5 μm in diameter ($PM_{2.5}$) are a critical topic of study (Pongpiachan et al., 2015a, 2017a). The smaller size fraction of these particles enables them to be absorbed into the alveolar region of the respiratory system with far better efficiency than coarse particles (Harrison and Yin, 2000). $PM_{2.5}$ has been tied to increased risks of severe asthma attacks as well as to an increased use of medication among children who have asthma (Slaughter et al., 2003). Furthermore, PAHs are known to be important substances that are attached to PM. There are a large group of organic compounds that have two or more fused aromatic (benzene) rings. They are formed mainly as a result of pyrolytic processes, especially in the incomplete combustion of organic materials during industrial and anthropogenic processes (Hagedorn et. al., 2009 and WHO, 2000). They have a relatively low in water solubility, but highly lipophilic. The main sources of PAHs emission include not only biomass burning, but also motor vehicles, industrial processes, domestic heating, waste incineration and tobacco smoke (Ré-Poppi and Santiago-Silva, 2005). Consequently, these substances end up being widely distributed as environmental contaminants (Igwe and Ukaogo, 2015). Human exposure to these substances can occur in indoor and outdoor environments by inhalation, the ingestion of food and through contact with skin (Pongpiachan et al., 2015b, 2017b). PAHs have received an increased amount of attention in recent years in a range of air pollution studies because some of these compounds are highly carcinogenic or mutagenic (IARC, 1983). Although there are hundreds of PAHs, perhaps the most important is benzo[*a*]pyrene (BaP) (WHO, 2000). BaP is commonly present along with other PAHs in cigarette smoke, grilled and broiled foods, and as a by-product in several industrial processes. It can be easily dispersed into the

ambient air, indoor air, and in some water sources (ATSDR, 1995). In addition, Levoglucosan (1,6-anhydro-B-D-glucopyranose) is a major component of the particles emitted by biomass burning (Simoneit et. al., 1999). It is derived from pyrolysis of cellulose and hemicellulose at high temperatures (Fine et. al., 2001). Several previous studies have recommended using levoglucosan as an indicator for the tracer of biomass burning emissions (Simoneit, 2002; Hu et. al., 2013). For example, Fartas et al. (2009) conducted studies on the determination of levoglucosan in PM_{10} and the possible sources of emissions occurring from the combustion of selected plants found in Malaysia, and also found that the burning of softwoods produced more levoglucosan than hardwoods.

In focusing on air pollution problems in 8 provinces of upper North of Thailand, the numerous studies of the air quality, characteristic of PM_{10}, $PM_{2.5}$ and theirs sources especially in the city such as Chiang Mai, Lamphun and Lampang provinces have been reported (Pengchai et. al., 2009; Chantrara et. al., 2009; Phoothiwut and Junyapoom, 2013; Pongpiachan, 2013; Tsai et al., 2013; Wiriya et al., 2013; Khamkaew et al., 2016; Thepnuan et al., 2019). These problems has not only affected the visibility, the tourism, and the economy, but also the health of the people living in the areas resulting in a lot of extra-governmental budgets spent on air pollution related-treatments. Tak Province is located in the lower North of Thailand. It is the second largest province of the North of Thailand after Chiang Mai Province. The unique position and geography of Tak province with its western side join and form a long boundary with the Republic of the Union of Myanmar as well as other provinces in upper Northern Thailand such as Chiang Rai, Chiang Mai and Mae Hong Son Province, has made it especially vulnerable to the seasonal air pollution. However, there were only a few air pollution studies and none on the $PM_{2.5}$ studies in Tak Province regarding both the components and its toxicity. Moreover, investigation of the emission of biomass smoke particles to the $PM_{2.5}$ is important to find out the air pollution source in this area. The first goal of this study is to investigate the concentrations of PAHs and levoglucosan in $PM_{2.5}$ in suburban areas and the second goal is to assess the health risks of local people for providing empirical data with regard to the potential health effects.

2. MATERIALS AND METHODS

2.1 Study Site

The sampling site in this study was situated in Mae Sot District, Tak Province (Fig. 1), which is a district that is located in the west of Tak Province and shares a border with Myanmar. Mae Sot District is distinctive as a trade hub. It is referred to as a Special Economic Zone (SEZ) and it is home to considerable Burmese migrant and refugee populations. The study site was situated in Mae Sot Basin, which is surrounded by high mountains with tropical and pine forests. It is located at 16° 43' 07" N, 98° 33' 56" E (MSO) and is primarily situated along the roadside at an altitude of 212.71 m above sea level. $PM_{2.5}$ pollution has rarely been reported on in this area and there are no recorded data on the characteristics of PM for this study site. Whereas the eight provinces of northern Thailand are

currently receiving a lot of attention with regard to the air pollution issue and the concentrations and components of PM for these eight provinces are presently being studied.

2.2 Sample Collection

$PM_{2.5}$ samples were collected continuously for five days per month during the period of March to July 2016 by Mini Volume Portable Air Sampler at a flow rate of 5 L/min within twenty-four-hour in ambient air. The filters that were used to collect the samples were 47mm fiber film filters (Ø = 47 mm, Pallflex T60A20, Pall Corporation, Putnam, CT, USA). The filters were weighed at least three times before and after sampling with a microbalance (AB135-S/FACT Dual Range Analytical Balance, METTLER TOLEDO) with an accuracy of 0.01 microgram after being conditioned in an electronic desiccator at 25 ºC and with relative humidity of 10% for 24 hr. After collection, sample filters were stored in a refrigerator at approximately -20°C until analysis. PM_{10} concentrations and meteorological data were obtained from the Pollution Control Department (PCD) and Northern Meteorological Center, respectively.

2.3 Chemicals Analysis

2.3.1 Extraction and PAHs Analysis

In this study, the 16 PAHs identified by the United States Environmental Protection Agency (US-EPA) were analyzed. The results are shown in Table 1. The half filter of each sample was cut into small pieces. After that, the filter samples were extracted ultrasonically twice with dichloromethane for 15min at 20° C and then centrifuged at 3,000 rpm for 15 min. Next, the solution was filtered through a 0.2 µm syringe PTFE filter and 0.1 mL hexane was added. Subsequently, the solution was evaporated by a gentle stream of nitrogen gas (N2). The residue solution was added with the internal standard (Semivolatile Internal Std Mix, Supelco, USA) and analyzed by gas chromatography system with mass spectrometry (GC-MS-QP2010 Series, Shimadzu Corporation, Japan). The column was a DB-5MS column (0.25 mm i.d. x 30 m and 0.25 µm film thickness, Agilent technologies Inc. USA) had an initial temperature of between 50 °C to 300 °C at 10 °C/min, and these conditions were then held for 10 min. High-purity helium was used as the carrier gas at a constant flow rate of 1.3 mL/min. The injection mode was splitless and the sample volume was 1 ml. MS detection was operated in selected ion monitoring mode (SIM). This protocol was carried out using the method reported by Wang et. al. (2016).

2.3.2 Extraction and Levoglucosan Analysis

The method for levoglucosan analysis was done following the procedure of Kumagai et al. (2010). Briefly, one quarter of each filter was extracted with 5 mL of dichloromethane/methanol (2:1). The internal standard (1 µL of Levoglucosan d_7) was added before sample extraction. The extract was further evaporated and dried under a stream of gaseous nitrogen. Levoglucosan was quantified by analyzing N, O-Bis (trimethylsilyl)-trifluoroacetamide with 10vol% Chlorotrimethylsilane (BSTFA

with 10vol% TMCS). Results of a gas chromatograph (Shimadzu Corporation, Japan) with a 60 m DB-5MS silica capillary column (Agilent technologies Inc. USA) connected to a mass spectrometry device (6890GC/5973MS, Agilent Technologies, CA, USA) was used for the analysis of levoglucosan.

2.3.3 Quality control for chemical analysis

The accuracy of PAHs analysis was performed using five replications of 0.1 µg/mL of mixed PAHs standard solution (Semivolatile Internal Standard Mix 2000 µg/ml in dichloromethane, Supelco, Merck, USA) to match the method used with samples. The results revealed high recovery levels (92 – 111%) for all PAHs compounds. The limit of detection (LOD) and limit of quantification (LOQ) values of GC-MS for 16-PAHs analysis were derived from six measurements of 0.005 µg/mL and were mixed with PAHs standard. Results were recorded at 0.28 – 2.40 ng/mL of LOD and 0.92 – 8.01 ng/mL of LOQ. The recovery levels of levoglucosan were achieved using the spiking method of standard solution (0.1, 0.5 and 2.0 mg/L). These concentrations were spiked in seven replications and the percent recoveries ranged from 96 – 120%. LOD and LOQ values of GC-MS for levoglucosan analysis were 0.04 mg/L and 0.13 mg/L, respectively. The results of repeatability of the method were evaluated in terms of percentage relative standard deviation (%RSD) ranged from 4.0 – 10.6%.

2.4 Health risk evaluation method

The carcinogenic and mutagenic health risks that occur as a result of PAHs exposure can be calculated by multiplying the concentrations of each PAH compound. The toxicity equivalents (TEQ) were calculated based on the toxicity equivalency factor (TEF) as Eq. 1 (Petry et al., 1996; Tsai et al., 2004; Wang et al., 2011; Benson et al., 2014; Mishra et al., 2016; Chen et al., 2019 and Ghanavati et al., 2019 are referred from Nisbet and LaGoy, 1992) and the mutagenic equivalents (MEQ) were computed from the mutagenic equivalency factor (MEF) as Eq. 2 (Durant et. al., 1999; Qu et al., 2015; Balgobin and Ramroop Singh, 2019). Additionally, BaP was classified as Group 1 by the International Agency for Research on Cancer (IARC) (Table 1) and was used as a representative marker for mixture exposure to PAHs.

$$TEQ_{BaP} = 0.01(CHR) + 0.01(BghiP) + 0.1(BaA) + 0.1(BbF) + 0.1(BkF) + 0.1(IND) + BaP + DahA \quad \text{(Equation 1)}$$

$$MEQ_{BaP} = 0.082(BaA) + 0.017(CHR) + 0.25(BbF) + 0.11(BkF) + 0.29(DahA) + 0.19(BghiP) + 0.31(IDP) + BaP \quad \text{(Equation 2)}$$

The inhalation cancer risk (ICR) was used for estimation of the cancer risk from exposure to PAHs which assessed the association between ICR and PAHs as a linear function of the TEQ

concentrations and the inhalation unit risk (IUR). IUR_{BaP} is the inhalation unit risk defined as the risk of cancer from a lifetime inhalation of unit mass of BaP ($m^3/\mu g$) which specifically, "the calculated, theoretical upper limit possibility of contracting cancer when exposed to BaP at a concentration of one microgram per cubic meter of air for a 70-year lifetime" (Bari et al., 2011; Jia et al., 2011; Manoli et al., 2016; Wang et al., 2012)

$$ICR = \Sigma TEQ_{BaP} \times IUR_{BaP} \quad \text{(Equation 3)}$$

Where: $IUR_{BaP} = 1.1 \times 10^{-6}$ m^3/ng (Cal-EPA, 2005; US-EPA, 2005)

$IUR_{BaP} = 8.7 \times 10^{-5}$ m^3/ng (WHO, 2000)

2.5 Backward trajectory calculation

The 24-hour backward trajectories were determined by using the Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model (Draxler and Hess, 1998). The backward trajectories are available online at https://ready.arl.noaa.gov/HYSPLIT_traj.php. The trajectories were computed every day at the same sampling site to investigate the air mass that was transported from the originated source to the receptor site.

3. RESULTS AND DISCUSSION

3.1 Concentration of PM

The data on PM_{10} concentrations were obtained from PCD, while $PM_{2.5}$ concentrations were collected from the ambient air samples using Mini Volume Portable Air Sampler during the period of March to July 2016 in Mae Sot District. The results varied from 65.8 – 128.2 and 3.3 – 122.0 $\mu g/m^3$, respectively and were highest in March. The seasonal and spatial variations in PM concentrations are presented in Fig 2. The mean averages of PM_{10} and $PM_{2.5}$ concentrations were recorded at 89.0 ± 17.2, 60.5 ± 27.9 $\mu g/m^3$ during the dry season (March - April), 84.6 ± 18.8, 63.9 ± 8.5 $\mu g/m^3$ and during the transitional period (May), respectively. In the wet season (June – July), the mean averages of $PM_{2.5}$ concentrations were recorded at 13.8 ± 5.6 $\mu g/m^3$ according to the data of PM_{10} concentrations of the PCD that were recorded during the period of January – May. The results of the $PM_{2.5}$ concentrations recorded during March – May (61.64 ± 22.85 $\mu g/m^3$) exceeded the acceptable 24-hr levels according to the National Ambient Air Quality Standards (NAAQS) in Thailand (50 $\mu g/m^3$). Accordingly, this period was classified as a smoke episode (episode). On the other hand, $PM_{2.5}$ concentrations recorded from late June to July (13.76 ± 5.58 $\mu g/m^3$) did not exceed the standards of the NAAQS which classified this period as a non-smoke episode (non-episode), while PM_{10} concentrations recorded during the sampling period exceeded 120 $\mu g/m^3$ only on the 26th of March. The variations of mean PM concentrations are illustrated in Table 1. The concentration values of $PM_{2.5}$ recorded during the episode were significantly higher than those recorded during the non-

episode. This was probably due to the open burning activities that typically occur during this period, which included the lighting of wildfires and the agricultural burning that takes place in rural areas and in neighboring countries. In addition, these results revealed a pattern that agreed with the pattern variations presented in previous reports produced in northern Thailand (Chantrara et. al., 2009; Pengchai et. al., 2009). In addition, the topography of the North of Thailand is mainly comprised of high mountains and pan basins, which give rise to temperature inversion or stagnant meteorological conditions in the dry season. Conditions involve low relative humidity levels, low dew point temperatures and light winds that are indicative of a stable atmosphere as has been previously reported (Amnuaylojaroen and Kreasuwun, 2012; Pungkhom and Jinsart, 2014).

3.2 $PM_{2.5}$-bound PAHs and their health risks

The total concentrations of 16 US-EPA PAHs ranged from 2.29 - 11.36 ng/m^3 and the highest total 16PAHs were recorded on March 26th 11.36 ng/m^3, while 2.29 ng/m^3 was recorded as the lowest concentration on June 5th. The variations of mean and standard deviation (SD) of each PAHs compound during both the episode and non-episode periods are shown in Table 1. The proportion of each PAH compound to the total of the PAHs was calculated in percentage to show the profile of the PAHs and the temporal variations of $PM_{2.5}$ bound PAHs during the sampling period, and the results are presented in Fig. 3. The major PAHs compounds in this study included PHE 21%, BghiP 17%, BbF and IDP 12%. Additionally, the descending order of 5 PAHs compounds (ng/m^3) were BghiP > PHE > IDP > BbF > BaP recorded during the smoke period and PHE > BghiP > DahA > IDP > BaP recorded during the non-smoke period. The results show that the concentrations of PHE and BghiP were dominant during both episodes. Previous studies reported that Southeast Asia countries, i.e. Thailand, normally found a high proportion of BghiPe and IDP compounds (Kim Oanh et al., 2000; Chantara et al., 2009; Chuesaard et al., 2014). Several previous research studies have suggested the use of PAH compounds as tracers to identify the sources of pollution. These results concurred with those of other studies (Freeman and Cattel, 1990; Bari et. al., 2010), where it was found that BghiP, IDP, BaP, BbF, BkF were the predominant PAHs, while smaller proportions of BaA, PHE, ANT, PYR, FLA and CHR were found in the softwood burning emissions. Accordingly, Shen et al. (2011) found that the dominant particulate-bound PAHs were PHE (20 ± 12%), FLA (11 ± 7%), FLO (11 ± 7%), and PYR (10 ±7%) from indoor crop residue burning in a typical rural stove. Likewise, Chuesaard et al. (2014) referred from Kulkarni and Venkataraman (2000) considered using the BaP as a biomass burning tracer, while BhgiP and IDP as the vehicle exhaust tracers. On the other hand, Miguel et al. (1998) reported that the higher molecular weight PAHs (HMW-PAHs) such as BaP and DahA were the dominant emissions occurring from gasoline vehicles, while diesel trucks emitted HMW-PAHs such as BaP, BghiP, IDP and DahA at higher concentrations than duty vehicles. And also Boström et al. (2002) presented the BghiP as a source specific indicator for gasoline vehicles.

Fig. 5 illustrated the temporal variations of $PM_{2.5}$, levoglucosan and tPAHs levels, which reveal a similar variation pattern. Moreover, the total PAHs concentrations during the episode were higher than those of the non-episode with statistical significance ($p < 0.05$). Accordingly, the concentrations of each PAHs compound recorded during the smoke episode were higher than those recorded during the non-episode, whilst pyrene (PYR) levels in the non-episode period were higher than in the episode period. Consequently, the concentrations of BghiP, PHE, IDP, BbF and BaP recorded during the smoke episode indicated that the possible sources of air pollution at this study site occurred not only from vehicle emission, but also from the increased biomass burning activities that go on during the dry season. These results agree with those of other studies (Chantara et. al., 2010; Phoothiwut and Junyapoom, 2013; Chuesaard et. al., 2014).

The health risks were evaluated by calculating the TEQ_{BaP} and MEQ_{BaP} values, particularly BaP, which is classified as a Class 1 carcinogenic to humans by the International Agency for Research on Cancer (IARC). The individual PAHs concentrations used in the calculation included CHR, BghiP, BaA, BbF, BkF, IND, BaP and DahA. The ambient TEQ_{BaP} and MEQ_{BaP} levels in this study ranged from 0.41 – 3.14 ng/m^3 and 0.34 – 3.35 ng/m^3, respectively. The highest values of TEQ_{BaP} and MEQ_{BaP} were recorded on the 26th March 2016. This study found that the TEQ_{BaP} and MEQ_{BaP} values recorded during the episode were three times significantly higher than those recorded in the non-episode, as illustrated in Table 1. In the smoke episode, the mean TEQ_{BaP} values in MSO (1.57 ± 0.67 ng/m^3) was lower than those in April 2010 (3.70 ng/m^3) and higher than those in January – March 2011 (0.25 ng/m^3) while those in the non-smoke episode (0.53 ± 0.07 ng/m^3) was higher than those in Aug – Nov 2010 (0.18 ng/m^3) in Chiang Mai Province (Wiriya et al., 2013). Therefore, the study by Wiriya et al. (2016) showed that the TEQ value of PM_{10}-bound PAHs obtained from the burning of leaf litter was higher than in the burning of maize residue and rice straw. Notably, the burning of leaf litter emitted higher amounts of pollutants than the burning of agricultural residue. The relationship between TEQ_{BaP}, MEQ_{BaP} and tPAHs concentrations are shown in Table 3, both of correlation coefficient between TEQ_{BaP} and tPAHs value and MEQ_{BaP} and tPAHs value in MSO were strongly significant ($p < 0.01$). Furthermore, the correlation between BaP equivalent (TEQ_{BaP} and MEQ_{BaP}) and tPAHs in the smoke episode was greater than those in the non-smoke episode and the correlation coefficient of mutagenic equivalent was higher than carcinogenic equivalent in both episodes.

The potential inhalation lifetime lung cancer risk (LLCR) of human exposure to PAHs was evaluated by calculating the ICR based on BaP equivalent. This study employed the IUR recommended by California Environmental Protection Agency (Cal-EPA) and the United States Environmental Protection Agency (US-EPA) of 1.1×10^{-6} ng/m^3 and World Health Organization (WHO) of 8.7×10^{-5} ng/m^3 for 70 years of lifetime (Cal-EPA, 2005; US-EPA, 2005; WHO, 2000; Jia et al., 2011). The mean $ICR_{\sum 8PAHs\text{-}BaP}$ values in this study were 1.27×10^{-6} and 10.02×10^{-5} for carcinogenic risk and 1.28×10^{-6}, 10.09×10^{-5} for mutagenic risk based on EPA and WHO,

respectively. In addition, the ICR values of both TEQ_{BaP} and MEQ_{BaP} obtained during smoke episode were 3 times significantly higher than that of the non-smoke episode. Moreover, the LLCR values that calculated based on WHO were higher than the values that were calculated based on EPA (Fig. 4). The similar results were also observed in Zaragoza, Spain (Callén et al., 2014), Amazon region (de Oliveira Alves et al., 2015) and Kuala Lumpur, Malaysia (Sulong et al., 2017). Furthermore, it was found that the values of LLCR based on both of EPA and WHO techniques were higher than the estimation of the previous study for the 70 years of lifetime ICR from of $PM_{2.5}$-bounded PAHs (ΣPAH12-BaPeq) of northern Thailand in 2012 - 2013 (Pongpiachan et al., 2015a). However, these values were lower than those in Guangzhou, China (5.98×10^{-4}) (Liu et al., 2015). Thereupon based on US-EPA, the acceptable level of risk is one chance in a million (LLCR = 10^{-6}) or less of developing human cancer over a lifetime (70 years) and one case in ten thousand people (LLCR = 10^{-4}) is a very high potential risk (Greene and Morris, 2006; Sun et al., 2016). The LLCR values in the smoke episode were over the acceptable cancer risk for 2 cancer cases per million people in this study area. The period of PAHs exposure from biomass burning season is not a long time, but the dose exposure is extremely high and repeated in every year.

3.3 Levoglucosan concentrations

Levoglucosan, 1,6-anhydride of glucose, is the product of the degradation of products from cellulose and it is stable in the atmosphere. It represents a fraction marker for biomass burning in the form of wood burning and wildfires (Hoffmann et. al., 2010). The concentrations of levoglucosan in this study ranged from 0.08 – 1.95 µg/m^3 and the average concentration values recorded in the episode and non-episode were 1.00 ± 0.41 and 0.12 ± 0.03 µg/m^3, respectively (Table 1). The temporal variations of levoglucosan are shown in Fig. 5. This factor has cleared the biomass burning pattern with very high emissions occurring in March through May and low emissions during the remaining months. Accordingly, the results show that the levoglucosan concentrations were significantly higher during the episode than those recorded in the non-episode. The correlation between levoglucosan, $PM_{2.5}$ and individual PAHs compounds during the smoke and non-smoke episodes were calculated, and the Pearson correlation coefficients (r) are listed in Table 2. Levoglucosan concentrations revealed a significantly strong correlation with $PM_{2.5}$, FLU, BbF, BaP, IDP, DahA, BghiP and tPAHs during the smoke episode, while it was not significantly correlated during the non-smoke episode. These results corresponded with those of previous research studies involving biomass tracers, which found that levoglucosan was a concomitant in the fractions of PM being emitted from biomass burning. Consequently, these studies revealed that there was an increase in biomass burning emissions resulting in increased air pollutant concentrations in the ambient air (Bari et. al., 2010; Chuesaard et. al., 2014).

Furthermore, the relationship between TEQ_{BaP} and MEQ_{BaP} and levoglucosan concentrations are shown in Fig. 6. Both the correlation coefficients between the TEQ_{BaP} and levoglucosan values

and MEQ_{BaP} and levoglucosan values at this study site that were recorded during the smoke episode (r = 0.753 and 0.761, respectively) were higher than those in the non-smoke episode (r = -0.245 and -0.287, respectively). These results confirm that the increasing biomass influences in the smoke episode could potentially present a carcinogenic risk to health and a greater mutagenicity risk than those same influences in the non-smoke episode. Therefore, the health effects of PM exposure depend upon certain physical characteristics, such as the breathing mode of the inhabitants (volume and rate of a person), the size of particulates and the effectiveness of PM exposure by local area conditions (e.g. topography, weather and seasons), sources of PM, concentrations being emitted and microenvironments (Brown et al., 2013; Casati et al., 2007; Kim et al., 2015).

3.4 Relationship of $PM_{2.5}$ concentrations to other air pollutants and Meteorological concentrations

The correlation between $PM_{2.5}$ levels to air pollutants and meteorological concentrations was analyzed by Spearman rank correlation analysis. The results found that the $PM_{2.5}$ concentrations had a strong positive correlation with tPAHs, TEQ_{BaP} and MEQ_{BaP}, levoglucosan concentrations and temperature (Temp) ($p < 0.01$), and a significant degree of correlation with PM_{10} at $p < 0.05$ as is shown in Table 3. On the other hand, the increased precipitation (rain) and relative humidity (%RH) values established negative relationships with $PM_{2.5}$, tPAHs and levoglucosan with statistical significance at a confidence level of 0.01. Notably, wind speed (WS) yielded a negative correlation with $PM_{2.5}$, but without statistical significance. These results were similar to those of previous studies such as Owoade et al. (2012), Jayamurugan et al. (2013) and Chen et al. (2016). The effects of the meteorological parameters typically fluctuate in different regions of the world. The important factors such as temperature, %RH and wind can influence the dispersion, transport and removal of particulate matter in the atmosphere (Munir et al., 2017). The strong positive correlation between $PM_{2.5}$ and temperature was due to the temperature and could have affected the formation of secondary particulate matters in the atmosphere. This may have been the case as high temperatures pay a role in photochemical reaction precursors (Wang and Ogawa, 2015; Munir et al., 2013). Wet deposition is one of the important mechanisms for removing air pollution by washing out PM and organic pollutants from the atmospheric to the ground (Guo et al., 2014). The negative relation coefficient obtained might have been a result of the washout mechanism of rainfall that occurs by reducing the atmospheric particulate pollution and covariate factors of atmospheric pressure and ambient temperature (Guo et al., 2016; Chen et al. 2016). Additionally, high humidity can reduce $PM_{2.5}$ concentrations as particles expand to become too heavy to dwell in the ambient air. Conversely, when hygroscopic influences grow along with low humidity, the $PM_{2.5}$ concentrations are increased (Wang and Ogawa, 2015; Lou et al., 2017). The precipitation data recorded during the sampling of $PM_{2.5}$ are shown in Fig. 2. All the sampling during the non-episode period occurred on rainy days. Therefore, the rain that fell on the sampling days resulted in decreased concentrations of particulate matter,

tPAHs and levoglucosan. Thus, rainfall was an important factor that effectively resulted in a reduction the pollutants in the ambient air during the non-smoke episode. On the 28th of March during the smoke episode, the levels of PM and tPAHs in the pollutants were slightly reduced as a result of the rainfall. This might have been the case because it had rained for a short period of time and the amount of rainfall was minimal (2.2 mm) on this occasion. In addition, the air mass movement was not only influenced by the local area, but also from the direction of the southeast, as is illustrated in Fig. 7. Several studies have correlated the occurrence of rain fall with a reduction in the particulate matter present in the atmosphere with regard to the amount, frequency, intensity, and the number of consecutive rainy days. Qian et al. (2009) found that the frequency and amount of precipitation are both significantly decreased during periods of high pollution. On the other hand, Choi et al. (2008) reported that the aerosol concentrations over a number of days were positively associated with the days of moderate-rainfall frequency (10-20 mm/day), but negatively correlated with the days of light-rainfall frequency (5 mm/day).

3.5 Backward trajectories of air movement

The Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model (Internet-based) developed by the Air Resource Laboratory (ARL) of the National Oceanic and Atmospheric Administration (NOAA) was used to compute the 24-hr and 72-hr backward trajectory values at 16.00 UTC in each day of the sampling period in this study. The air mass trajectories were used to interpret the transport pathways of infinitesimally small particles as they move through time and space (Stohl, 1998; Wang et al., 2010). Backward trajectory recorded during each sampling month showed the arrival height of 500 m above ground level (AGL) (Fig. 7) identify the source regions and long-range transports of air mass (Kim Oanh and Leelaskultum, 2011b; Crosbie et al., 2014). As shown in the study done by Zhu et al. (2011), for the receptor heights of 100 - 1000 m, the transport pathways were not significantly different, whereas the trajectories could increase the length of transport with the receptor height. The 24 or 72 hr backward trajectories are frequently used in previous studies (Wiriya et al., 2013; Xin et al., 2016; Sun et al., 2017). The 24 hr backward trajectories, with its smaller trajectory position errors, have been used to investigate the possible sources of regional emission and elucidate the regional transport pathways (Wang et al., 2010; Godłowska et al., 2015). Increasing the time of trajectories could trace the increase in long-range transport pathway. In addition, Fig 7 illustrated the accumulated 7 days of hotspots during sampling in each month. Hotspot analysis is the primary tool to locate the patterns of the forest fire occurrence (Feltman et al., 2012; Said et al., 2017). In this study, hotspot data was derived from Moderate Resolution Imaging Spectroradiometer (MODIS) on board the Terra (EOS AM-1) and Aqua (EOS PM-1) satellites from the NASA's Earth Observatory website (https://earthdata.nasa.gov).

Most of the data model showed that the air mass movement was from a westerly direction. During the smoke episode, the major direction of the backward trajectory originated from the

continental area of Myanmar before arriving in Mae Sot District except for the backward trajectory in March which showed the originated of air mass not only from Myanmar but also from the local area and Cambodia (Fig 7(a-c). Besides, the map of hotspots shows that the accumulated 7 days hotspots were very high at both local area and neighboring countries. Likewise, the pathway of air mass moved over the areas showing a high accumulated number of hotspots before arriving at the study sites. Numerous studies have reported that the air mass transport pathways significantly correlated with spatial and temporal variations of pollutants. It was considered that under propitious atmospheric conditions the air mass parcels can transport particulate matters and other pollutants generated from the open-burning areas (Chuang et al., 2016; Liu et al., 2013; McGowan and Clark, 2008). As a result in this study, the close correlation between $PM_{2.5}$ and levoglucosan concentrations and the peak of hotspot activity during the smoke episode implies that the high concentration of $PM_{2.5}$ during this period was from biomass burning emission. Although the air mass trajectory during the non-smoke episode generated in the Andaman Sea, it was transported by the air mass that blew through Myanmar to the receptor site (Fig. 7(d-e). The air mass transport was not affected by the hotspot activity in this episode. This is concordant with the reported wind direction in Chiang Mai Province in 2010 and 2011 (Wiriya et al., 2013). In the details, the backward trajectory values recorded during the non-smoke episode moved directly from a southwest direction into the receptor sites and had a longer trajectory than those recorded during the smoke episode. This would indicate the presence of a significantly high-wind speed regimes (Pöhlker et al., 2018) and long-range transport pathway over the Andaman Sea and Myanmar to the sampling sites. In addition, the increase of the wind speed would decrease the concentration of $PM_{2.5}$ (Wang et al., 2017).

4. CONCLUSIONS

Mae Sot District is located in Tak Province, which is a province in western Thailand. Notably, the air pollution issue in Tak Province during the dry season is not different from that of northern Thailand. The results in this study reveal that the temporal variations of $PM_{2.5}$ concentrations, and other air pollutants such as PAHs and levoglucosan, displayed a similar trend with those of previous studies conducted in Chiang Mai, Lamphun and Lampang Provinces. The peak concentrations in those studies were recorded in March. The average of $PM_{2.5}$, tPAHs and levoglucosan were significantly higher when recorded in the smoke episode than in the non-smoke episode. The predominant compounds of PAHs are PHE and BghiP in both episodes that can emit from not only vehicles but also biomass combustion. However, all individual PAHs concentrations recorded during the smoke episode were significantly higher than the non-smoke episode except for PYR (indicating a vehicle source). The concentrations of levoglucosan were strongly correlated with $PM_{2.5}$ and tPAHs during the smoke episode that can identify the source emission from biomass burning. Thus, these results can confirm that the sources of air pollution at the study site not only involved traffic emissions but also occurred from increasing biomass burning during the dry season. The human

health risks of $PM_{2.5}$-bound PAHs exposure evaluated from TEQ_{BaP} and MEQ_{BaP} showed that the smoke episode could increase a potential human health risk higher than those in the non-episode period. As a result, the LLCR during smoke episode were higher than the acceptable level of augmentative human cancer risk over a 70-year lifetime (US-EPA). During the smoke episode, backward trajectories have revealed that the problematic air masses that moved over the burning areas as showed the high hotspot activity were not only emitted from local areas, they were also emitted from outside the country like Myanmar and Cambodia. Therefore, the most air masses were generated from the western region of Thailand.

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REFERENCES

Amnuaylojaroen, T., Kreasuwun, J., 2012. Investigation of fine and coarse particulate matter from burning areas in Chiang Mai, Thailand using the WRF/CALPUFF. Chiang Mai J. Sci. 39(2), 311-326.

ATSDR (Agency for Toxic Substances and Disease Registry), 1995. Toxicological profile for polycyclic aromatic hydrocarbons (PAHs). U.S. Department of Health & Human Services. Agency for Toxic Substances and Disease Registry.

Balgobin, A., Ramroop Singh, N., 2019. Source apportionment and seasonal cancer risk of polycyclic aromatic hydrocarbons of sediments in a multi-use coastal environment containing a Ramsar wetland, for a Caribbean island. Sci. Total Environ. 664, 474-486.

Bari, M.A., Baumbach, G., Kuch, B., Scheffknecht, G., 2011. Air pollution in residential areas from wood-fired heating. Aerosol and Air Quality Research 11, 749-757.

Bari, Md.A, Baumbach, G., Kuch, B., Scheffknecht, G., 2010. Particle-phase concentrations of polycyclic aromatic hydrocarbons in ambient air of rural residential areas in southern German. Air Qual Atmos Health, 3, 103-116.

Benson, N.U., Essien, J.P., Asuquo, F.E., Eritobor, A.L., 2014. Occurrence and distribution of polycyclic aromatic hydrocarbons in surface microlayer and subsurface seawater of Lagos Lagoon, Nigeria. Environ. Monit. Assess. 186, 5519-5529.

Boström, C.-E., Gerde, P., Hanberg, A., Jernström, B., Johansson, C., Kyrklund, T., Rannug, A., Törnqvist, M., Victorin, K., Westerholm, R., 2002. Cancer Risk Assessment, Indicators, and Guidelines for Polycyclic Aromatic Hydrocarbons in the Ambient Air. Environmental Health Perspectives, 110(3), 451-488.

Brown, J.S., Gordon, T., Price, O., Asgharian, B., 2013. Thoracic and respirable particle definitions for human health risk assessment. Particle and fibre toxicology 10, 12.

Cal-EPA. 2005. Air toxics hot spots program risk assessment guidelines. Oakland, CA: California Environmental Protection Agency, Office of Environmental Health Hazard Assessment, Air Toxicology and Epidemiology Section.

Callén, M.S., Iturmendi, A., López, J.M., 2014. Source apportionment of atmospheric PM2.5-bound polycyclic aromatic hydrocarbons by a PMF receptor model. Assessment of potential risk for human health. Environmental Pollution, 195, 167-177.

Casati, R., Scheer, V., Vogt, R., Benter, T., 2007. Measurement of nucleation and soot mode particle emission from a diesel passenger car in real world and laboratory in situ dilution. Atmos. Environ. 41, 2125-2135.

Chantara, S., Sillapapiromsuk, S., Wiriya, W., 2012. Atmospheric pollutants in Chiang Mai (Thailand) over a five-year period (2005–2009), their possible sources and relation to air mass movement. Atmos. Environ., 60, 88–98.

Chantara, S., Wangkarn, S., Sangchan, W., Rayanakorn, M., 2010. Spatial and temporal variations of ambient PM10-bound polycyclic aromatic hydrocarbons in Chiang Mai and Lamphun Provinces, Thailand. Desalin. Water Treat., 19, 17–25.

Chantrara, S., Wangkarn, S., Tengcharoenkul, U., Sangchan, W., Rayanakorn, M., 2009. Chemical Analysis of Airborne Particulates for Air Pollutants in Chiang Mai and Lamphun Provinces, Thailand. Chiang Mai J, Sci., 36(2), 123-135.

Chen, J., Li, C., Ristovski, Z., Milic, A., Gu, Y., Islam, M.S., Wang, S., Hao, J., Zhang, H., He, C., Guo, H., Fu, H., Miljevic, B., Morawska, L., Thai, P., LAM, Y.F., Pereira, G., Ding, A., Huang, X., Dumka, U.C., 2017. A review of biomass burning: Emissions and impacts on air quality, health and climate in China. Science of the Total Environment, 579, 1000–1034.

Chen, Q., Chen, Y., Luo, X.S., Hong, Y., Hong, Z., Zhao, Z., Chen, J., 2019. Seasonal characteristics and health risks of PM2.5-bound organic pollutants in industrial and urban areas of a China megacity. J. Environ. Manage. 245, 273-281.

Chen, T., He, J., Lu, X., She, J., Guan, Z., 2016. Spatial and Temporal Variations of PM2.5 and Its Relation to Meteorological Factors in the Urban Area of Nanjing, China. Int. J. Environ. Res. Public Health, 13, 921.

Choi, Y.-S., Ho, C.-H., Kim, J., Gong, D.-Y., Park, R.J., 2008. The Impact of Aerosols on the Summer Rainfall Frequency in China. Journal of Applied Meteorology and Climatology, 47(6), 1802-1813. doi:10.1175/2007jamc1745.1.

Chuang, M.T., Chou, C.C.-K., Sopajaree, K., Lin, N.-H., Wang, J.-L., Sheu, G.-R., Chang, Y.-J., Lee, C,-T., 2013. Characterization of aerosol chemical properties from near-source biomass burning in the northern Indochina during 7-SEAS/Dongsha experiment. Atmospheric Environment, 78, 72-81.

Chuang, M.-T., Lee, C.-T., Chou, C.C.K., Engling, G., Chang, S.-Y., Chang, S.-C., Sheu, G.-R., Lin, N.-H., Sopajaree, K., Chang, Y.-J., Hong, G.-J., 2016. Aerosol transport from Chiang Mai, Thailand to Mt. Lulin, Taiwan – Implication of aerosol aging during long-range transport. Atmos. Environ. 137, 101-112.

Chuesaard, T., Chetiyanukornkul, T., Kameda, T., Hayakawa, K., Toriba, A., 2014. Influence of Biomass Burning on the Levels of Atmospheric Polycyclic Aromatic Hydrocarbons and Their Nitro Derivatives in Chiang Mai, Thailand. Aerosol and Air Quality Research, 14, 1247–1257.

Crosbie, E., Sorooshian, A., Monfared, N.A., Shingler, T., Esmaili, O., 2014. A Multi-Year Aerosol Characterization for the Greater Tehran Area Using Satellite, Surface, and Modeling Data. Atmosphere (Basel), 5(2), 178–197. doi:10.3390/atmos5020178.

de Oliveira Alves, N., Brito, J., Caumo, S., Arana, A., Sandra de Souza Hacon, S., Artaxo, P., Hillamo, R., Teinilä, K., de Medeiros, S.R.B., de Castro Vasconcellos, P., 2015. Biomass burning in the Amazon region: Aerosol source apportionment and associated health risk assessment. Atmospheric Environment, 120, 277-285.

Draxler, R.R., Hess, G.D., 1998. An overview of the HYSPLIT-4 modeling system for trajectories, dispersion and deposition. Aust. Meteorol. Mag., 47, 295-308.

Durant, J., Lafleur, A., Busby, W., Donhoffner, L., Penman, B., Crespi, C., 1999. Mutagenicity of C24H14 PAH in human cells expressing CYP1A1. Mutation Research, 446, 1-14.

Estrellan, C.R., Iino, F., 2010. Toxic emissions from open burning. Chemosphere, 80, 193–207.

Fartas, F.M., Othman, M.R., Ali rajeb, F., Latif, M. T., 2009. Determination of levoglucosan in PM10 and biomass close burning residue sanples using anthrone-surfuric acid colorimetric method. Journal of Applied Sciences Research, 5(10), 1663-1667.

Feltman, J.A., Straka, T.J., Post, C.J., Sperry, S.L., 2012. Geospatial analysis application to forecast wildfire occurrences in South Carolina. Forests, 3(2), 265-282. doi: doi:10.3390/f3020265.

Fine, P.M., Cass, G.R., Simoneit, B.R.T., 2001. Chemical characterization of fine particle emissions from fireplace combustion of woods grown in the Northeastern United States. Environ. Sci. Technol., 35, 2665-75.

Freeman, D. J., Cattel, C.R., 1990. Wood burning as a source of atmospheric polycyclic hydrocarbons. Environmental Science and Technology, 24, 1581–1585.

Garivait, S., Bonnet, S., Kamnoed, O., 2008. Air Pollutant Emissions from Paddy Residues Open Burning and their Potential for Bioenergy in the Greater Mekong Sub-Region (Cambodia, Lao PDR, Thailand and Vietnam). GMSARN International Journal, 2, 169 – 174.

Ghanavati, N., Nazarpour, A., Watts, M.J., 2019. Status, source, ecological and health risk assessment of toxic metals and polycyclic aromatic hydrocarbons (PAHs) in street dust of Abadan, Iran. Catena 177, 246-259.

Godłowska, J., Hajto, M.J., Tomaszewska, A.M., 2015. Spatial analysis of air masses backward trajectories in order to identify distant sources of fi ne particulate matter emission. Archives of Environmental Protection, 41(2), 28–35. doi:10.1515/aep-2015-0015.

Greene, N.A., Morris, V.R., 2006. Assessment of public health risks associated with atmospheric exposure to PM2.5 in Washington, DC, USA. International Journal of Environmental Research and Public Health, 3, 86-97.

Guo, L.-C., Bao, L.-J., She, J.-W., Zeng, E. Y., 2014. Significance of wet deposition to removal of atmospheric particulate matter and polycyclic aromatic hydrocarbons: A case study in Guangzhou, China. Atmospheric Environment, 83, 136-144. doi:https://doi.org/10.1016/j.atmosenv.2013.11.012

Guo, L.-C., Zhang, Y., Lin, H., Zeng, W., Liu, T., Xiao, J., Rutherford, S., You, J., Ma, W., 2016. The washout effects of rainfall on atmospheric particulate pollution in two Chinese cities. Environmental Pollution, 215, 195-202.

Hagedorn, H.-W., Scherer, G., Engl, J., Riedel, K., Cheung, F, Errington, G., Shepperd, J., McEwan, M., 2009. Urinary excretion of phenolic polycyclic aromatic hydrocarbons (OH-PAH) in nonsmokers and in smokers of cigarettes with different ISO tar yields. Analytical Toxicology, 33, 301-309.

Harrison, R.M., Yin, J., 2000. Particulate matter in the atmosphere: which particle properties are important for its effects on health. The Science of the Total Environment, 249, 85-101.

Hoffmann, D., Tilgner, A., Iinuma, Y., Herrmann, H., 2010. Atmospheric Stability of Levoglucosan: A Detailed Laboratory and Modeling Study. Environ. Sci. Technol., 44, 694–699.

Hu, Q.-H., Xie, Z.-Q., Wang, X.-M., Kang, H., Zhang, P., 2013. Levoglucosan indicates high levels of biomass burning aerosols over oceans from the Arctic to Antarctic. Scientific Reports, 3: 3119, 1-7. DOI: 10.1038/srep03119.

Huang, K., Fu, J.S., Hsu, N.C., Gao, Y., Dong, X., Tsay, S.-C., Lam, Y.F., 2013. Impact assessment of biomass burning on air quality in Southeast and East Asia during BASE-ASIA. Atmospheric Environment, 78, 291-302.

Hyer, E.J., Chew, B. N., 2010. Aerosol transport model evaluation of an extreme smoke episode in Southeast Asia. Atmospheric Environment, 44, 1422-1427.

IARC (International Agency for Research on Cancer), 1983. IARC monographs on the evaluation of the carcinogenic risk of chemicals to human. Polynuclear aromatic compounds, Part I, Chemical, environmental, and experimental data. World Health Organization, Geneva, Switzerland.

Igwe, J.C., Ukaogo, P.O., 2015. Environmental Effects of Polycyclic Aromatic Hydrocarbons. Journal of Natural Sciences Research, 5 (7), 117-131.

Jayamurugan, R., Kumaravel, B., Palanivelraja, S., Chockalingam, M. P., 2013. Influence of Temperature, Relative Humidity and Seasonal Variability on Ambient Air Quality in a Coastal Urban Area. International Journal of Atmospheric Sciences, 2013, 1-7.

564 Jia, Y., Stone, D., Wang, W., Schrlau, J., Tao, S., Simonich, S.L.M., 2011. Estimated reduction in cancer risk due to PAH
565 exposures if source control measures during the 2008 Beijing Olympics were sustained. Environmental Health
566 Perspectives, 119, 815-820.
567 Khamkaew, C., Chantara, S., Janta, R., Pani, S. K., Prapamontol, T., Kawichai, S., Wiriya, W., Lin, N.-H., 2016.
568 Investigation of Biomass Burning Chemical Components over Northern Southeast Asia during 7-SEAS/BASELInE
569 2014 Campaign. Aerosol and Air Quality Research, 16, 2655–2670.
570 Kim Oanh, N.T., Leelasakultum, K., 2011a. Characterization of particulate matter emission from open burning of rice straw.
571 Atmospheric Environment, 45, 493-502.
572 Kim Oanh, N.T., Leelasakultum, K., 2011b. Analysis of meteorology and emission in haze episode prevalence over
573 mountain-bounded region for early warning. Sci. Total Environ., 409, 2261-2271.
574 Kim, K.-H., Kabir, E., Kabir, S., 2015. A review on the human health impact of airborne particulate matter. Environ. Int. 74,
575 136-143.
576 Kulkarni, P., Venkataraman, C., 2000. Atmospheric Polycyclic Aromatic Hydrocarbons in Mumbai, India. Atmos. Environ.,
577 34, 2785–2790.
578 Kumagai, K., 2010. Determination of Dicarboxylic Acids and Levoglucosan in Fine Particles in the Kanto Plain, Japan, for
579 Source Apportionment of Organic Aerosols. Aerosol and Air Quality Research. doi:10.4209/aaqr.2009.11.0075.
580 Lee, C.T., Chuang, M.T., Lin, N.H., Wang, J.L., Sheu, G.R., Wang, S.H., Huang, H., Chen, H.W., Weng, G.H., Hsu, S.P.
581 (2011). The enhancement of biosmoke from Southeast Asia on PM2.5 water-soluble ions during the transport over
582 the Mountain Lulin site in Taiwan. Atmos. Environ., 45, 5784–5794.
583 Lemieuxa, P.M., Lutesb, C.C., Santoianni, D.A., 2004. Emissions of organic air toxics from open burning: a comprehensive
584 review. Progress in Energy and Combustion Science, 30, 1–32.
585 Liu, J., Man, R., Ma, S., Li, J., Wu, Q., Peng, J., 2015. Atmospheric levels and health risk of polycyclic aromatic
586 hydrocarbons (PAHs) bound to PM2. 5 in Guangzhou, China. Marine Pollution Bulletin, 100, 134-143.
587 Liu, N., Yu, Y., He, J., Zhao, S., 2013. Integrated modeling of urban–scale pollutant transport: application in a semi–arid
588 urban valley, Northwestern China. Atmospheric Pollution Research, 4, 306- 314.
589 Lou, C., Liu, H., Li, Y., Peng, Y., Wang, J., Dai, L., 2017. Relationships of relative humidity with PM2.5 and PM10 in the
590 Yangtze River Delta, China. Environ Monit Assess, 189: 582.
591 Manoli, E., Kouras, A., Karagkiozidou, O., Argyropoulos, G., Voutsa, D., Samara, C., 2016. Polycyclic aromatic
592 hydrocarbons (PAHs) at traffic and urban background sites of northern Greece: source apportionment of ambient
593 PAH levels and PAH-induced lung cancer risk. Environmental Science and Pollution Research, 23, 3556-3568.
594 McGowan, H., Clark, A., 2008. Identification of dust transport pathways from Lake Eyre, Australia using Hysplit. Atmos.
595 Environ. 42, 6915-6925.
596 Miguel, A.H., Kirchstetter, T.W., Harley, R.A., Hering, S.V., 1998. On-road Emissions of Particulate Polycyclic Aromatic
597 Hydrocarbons and Black Carbon from Gasoline and Diesel Vehicles. Environ. Sci. Technol., 32(4), 450–455.
598 Mishra, N., Ayoko, G.A., Morawska, L., 2016. Atmospheric polycyclic aromatic hydrocarbons in the urban environment:
599 Occurrence, toxicity and source apportionment. Environ. Pollut. 208, 110-117.
600 Munir, S., 2013. Modeling Particulate Matter Concentrations in Makkah, Applying a Statistical Modeling Approach.
601 Aerosol and Air Quality Research. doi:10.4209/aaqr.2012.11.0314.
602 Munir, S., Habeebullah, T. M., Mohammed, A. M. F., Morsy, E. A., Rehan, M., Ali, K., 2017. Analysing PM2.5 and its
603 Association with PM10 and Meteorology in the Arid Climate of Makkah, Saudi Arabia. Aerosol and Air Quality
604 Research, 17(2), 453-464. doi:10.4209/aaqr.2016.03.0117.
605 Nisbet, I.C.T., LaGoy, P.K., 1992. Toxic equivalency factors (TEFs) for polycyclic aromatic hydrocarbons (PAHs).
606 Regulatory Toxicology and Pharmacology, 16, 290-300.
607 Owoade, O.K., Olise, F.S., Ogundele, L.T., Fawole, O.G., Olaniyi, H. B., 2012. Orrelation between particulate matter
608 concentrations and meteorological parameters at a site in ILE-IFE, NIGERIA. Ife Journal of Science, 14(1), 88-93.

Pengchai, P., Chantara, S., Sopajaree, K., Wangkarn, S., Tengcharoenkul, U., Rayanakorn, M., 2009. Seasonal variation, risk assessment and source estimation of PM 10 and PM10-bound PAHs in the ambient air of Chiang Mai and Lamphun, Thailand. Environ Monit Assess., 154, 197–218.

Petry, T., Schmid, P., Schlatter, C., 1996. The use of toxic equivalency factors in assessing occupational and environmental health risk associated with exposure to airborne mixtures of polycyclic aromatic hydrocarbons (PAHs). Chemosphere 32 (4), 639-648.

Phoothiwut, S., Junyapoom, S., 2013. Size distribution of atmospheric particulates and particulate-bound polycyclic aromatic hydrocarbons and characteristics of PAHs during haze period in Lampang Province, Northern Thailand. Air Qual Atmos Health, 6, 397–405.

Pöhlker, C., Walter, D., Paulsen, H., Könemann, T., Rodríguez-Caballero, E., Moran-Zuloaga, D., Brito, J., Carbone, S., Degrendele, C., Després, V.R., Ditas, F., Holanda, B.A., Kaiser, J.W., Lammel, G., Lavrič, J., Ming, J., Pickersgill, D., Pöhlker, M.L., Praß, M., Ruckteschler, N., Saturno, J., Sörgel, M., Wang, Q., Weber, B., Wolff, S., Artaxo, P., Pöschl, U., Andreae, M.O., 2018. Land cover and its transformation in the backward trajectory footprint region of the Amazon Tall Tower Observatory. Atmos. Chem. Phys., 19(13), 8425-8470. doi:10.5194/acp-19-8425-2019.

Pongpiachan, S., 2013. Diurnal Variation, Vertical Distribution and Source Apportionment of Carcinogenic Polycyclic Aromatic Hydrocarbons (PAHs) in Chiang-Mai, Thailand. Asian Pacific Journal of Cancer Prevention, 14, 1851-1863.

Pongpiachan, S., 2016. Incremental lifetime cancer risk of $PM_{2.5}$ bound polycyclic aromatic hydrocarbons (PAHs) before and after the Wildland fire episode. Aerosol Air Qual. Res., 16, 2907-2919.

Pongpiachan, S., Hattayanone, M., Cao, J., 2017b. Effect of agricultural waste burning season on PM2. 5-bound polycyclic aromatic hydrocarbon (PAH) levels in Northern Thailand. Atmospheric pollution research, 8(6), 1069-1080.

Pongpiachan, S., Kositanont, C., Palakun, J., Liu, S., Ho, K.F., Cao, J., 2015b. Effects of day-of-week trends and vehicle types on $PM_{2.5}$-bounded carbonaceous compositions. Science of the Total Environment, 532, 484-494.

Pongpiachan, S., Liu, S., Huang, R., Zhao, Z., Palakun, J., Kositanont, C., Cao, J., 2017a. Variation in day-of-week and seasonal concentrations of atmospheric PM 2.5-bound metals and associated health risks in Bangkok, Thailand. Archives of environmental contamination and toxicology, 72(3), 364-379.

Pongpiachan, S., Tipmanee, D., Khumsup, C., Kittikoon, I., Hirunyatrakul, P., 2015a. Assessing risks to adults and preschool children posed by $PM_{2.5}$-bound polycyclic aromatic hydrocarbons (PAHs) during a biomass burning episode in Northern Thailand. Science of the Total Environment, 508, 435-444.

Popovicheval, O.B., Shonija, N.K., Persiantseva, N., Timofeev, M., Diapouli, E., Eleftheriadis, K., Borgese, L., Nguyen, X. A., 2017. Aerosol Pollutants during Agricultural Biomass Burning: A Case Study in Ba Vi Region in Hanoi, Vietnam. Aerosol and Air Quality Research, 17, 2762–2779.

Pungkhom, P., Jinsart, W., 2014. Health risk assessment from bush fire air pollutants using statistical analysis and geographic information system: A case study in Northern Thailand. Geoinformatics, 10(1), 17-24.

Punsompong, P., Chantara, S., 2018. Identification of potential sources of PM10 pollution from biomass burning in northern Thailand using statistical analysis of trajectories. Atmospheric Pollution Research, https://doi.org/10.1016/j.apr.2018.04.003.

Qian, Y., Gong, D., Fan, J., Leung, L. R., Bennartz, R., Chen, D., Wang, W., 2009. Heavy pollution suppresses light rain in China: Observations and modeling. Journal of Geophysical Research, 114. doi:10.1029/2008jd011575.

Qu, C., Li, B., Wu, H., Wang, S., Giesy, J.P., 2015. Multi-pathway assessment of human health risk posed by polycyclic aromatic hydrocarbons. Environ. Geochem. Health 37, 587-601.

Ré-Poppi, N., Santiago-Silva, M., 2005. Polycyclic aromatic hydrocarbons and other selected organic compounds in ambient air of Campo Grande City, Brazil. Atmospheric Environment, 39, 2839-2850.

Said, S. N. m. B. M., Zahran, E.-S. M. M., & Shams, S. (2017). Forest fire risk assessment using hotspot analysis in GIS. The Open Civil Engineering Journal, 11(1), 786-801.

Shen, G.F., Wang, W., Yang, Y.F., Ding, J.N., Xue, M., Min, Y.J., Zhu, C., Shen, H.Z., Li, W., Wang, B., Wang, R., Wang, X.L., Tao, S., Russell, A.G., 2011. Emissions of PAHs from indoor crop residue burning in a typical rural stove: Emission factors, size distributions, and gas-particle partitioning. Environ. Sci. Technol., 45, 1206–1212.

Simoneit, B.R., 2002. Biomass burning—a review of organic tracers for smoke from incomplete combustion. Applied Geochemistry 17, 129-162.

Simoneit, B.R.T., Schauer, J.J., Nolte, C.G., Oros, D.R., Elias, V.O., Fraser, M.P., Rogge, W.F., Cass, G.R., 1999. Levoglusan, a tracer for cellulose in biomass burning and atmospheric particles. Atmospheric Environment, 33, 173–182.

Slaughter, J.C., Lumley, T., Sheppard, L., Koenig, J.Q., Shapiro, G.G., 2003. Effects of ambient air pollution on symptom severity and medication use in children with asthma. Annals of Allergy, Asthma, & Immunology, 91(4): 346–353.

Stohl, A., 1998. COMPUTATION, ACCURACY AND APPLICATIONS OF TRAJECTORIESÐA REVIEW AND BIBLIOGRAPHY. Atmospheric Environment, 32(6), 947-966.

Streets, D., Yarber, K. ., 2003. Biomass burning in Asia: Annual and seasonal estimates and atmospheric emissions. GLOBAL BIOGEOCHEMICAL CYCLES, 17(4), 1099, doi:10.1029/2003GB002040.

Sulong, N.A., Latif, M.T., Khan, M.F., Amil, N., Ashfold, M.J., Wahab, M.I.A., Chan, K.M., Sahani, M., 2017. Source apportionment and health risk assessment among specific age groups during haze and non-haze episodes in Kuala Lumpur, Malaysia. Science of the Total Environment, 601-602, 556-570.

Sun, J., Huang, L., Liao, H., Li1, J., Hu, J., 2017. Impacts of Regional Transport on Particulate Matter Pollution in China: a Review of Methods and Results. Curr Pollution Rep, 3, 182–191. doi 10.1007/s40726-017-0065-5.

Sun, S., Xia, Z., Wang, T., Wu, M., Zhang, Q., Yin, J., Zhou, Y., Yang, H., Wang, W., Yu, Y., 2016. Pollution level, sources, and lung cancer risk of PM10-bound polycyclic aromatic hydrocarbons (PAHs) in summer in Nanjing, China. Journal of Chemistry, ID 4546290, 1-10.

Thepnuan, D., Chantara, S., Lee, C.-T., Lin, N.-H., Tsai, Y. I., 2019. Molecular markers for biomass burning associated with the characterization of PM2.5 and component sources during dry season haze episodes in Upper South East Asia. Science of the Total Environment, 658, 708–722.

Tsai, P.-J., Shih, T.-S., Chen, H.-L., Lee, W.-J., Lai, C.-H., Liou, S.-H., 2004. Assessing and predicting the exposures of polycyclic aromatic hydrocarbons (PAHs) and their carcinogenic potencies from vehicle engine exhausts to highway toll station workers. Atmos. Environ. 38, 333-343.

Tsai, Y.I., Sopajaree, K., Auranee Chotruksa, A., Hsin-Ching Wu, H.-C., Kuo, S.-C., 2013. Source indicators of biomass burning associated with inorganic salts and carboxylates in dry season ambient aerosol in Chiang Mai Basin, Thailand. Atmospheric Environment, 78, 93-104. https://doi.org/10.1016/j.atmosenv.2012.09.040.

US-EPA (United States of Environmental Protection Agency). 2005. Guidelines for Carcinogen Risk Assessment. Risk Assessment Forum. Washington, DC., EPA/630/P-03/001B, March 2005.

Wang, F., Chen, D.S., Cheng, S.Y., Li, J.B., Li, M.J., Ren, Z.H., 2010. Identification of regional atmospheric PM10 transport pathways using HYSPLIT, MM5–CMAQ and synoptic pressure pattern analysis. Environmental Modelling & Software, 25, 927–934.

Wang, J, Zhang, M., Bai, X., Tan, H., Li, S., Liu, J., Zhang, R., Wolters, M.A., Qin, X., Zhang, M., Lin, H., Li, Y., Li, J., Chen, L., 2017. Large-scale transport of PM2.5 in the lower troposphere during winter cold surges in China. Scientific Reports, 7(13238), 1-10. Doi: 10.1038/s41598-017-13217-2.

Wang, J., Chen, S., Tian, M., Zheng, X., Gonzales, L., Ohura, T., Mai, B., Simonich, S.L., 2012. Inhalation cancer risk associated with exposure to complex polycyclic aromatic hydrocarbon mixtures in an electronic waste and urban area in South China. Environmental Science and Technology, 46, 9745-9752.

Wang, J., Ogawa, S., 2015. Effects of Meteorological Conditions on PM2.5 Concentrations in Nagasaki, Japan. Int. J. Environ. Res. Public Health, 12, 9089-9101.

Wang, Q., Kobayashi, K., Lu, S., Nakajima, D., Wang, W., Zhang, W., Sekiguchi, K., Terasaki, M., 2016. Studies on size distribution and health risk of 37 species of polycyclic aromatic hydrocarbons associated with fine particulate matter collected in the atmosphere of a suburban area of Shanghai city, China. Environmental Pollution, 214, 149-160.

Wang, W., Huang, M,J., Kang, Y., Wang, H.-S., Leung, A.O.W., Cheung, K. C. and Wong, M.H., 2011. Polycyclic aromatic hydrocarbons (PAHs) in urban surface dust of Guangzhou, China: Status, sources and human health risk assessment. Science of the Total Environment, 409, 4519-4527.

WHO. 2000. World Health Organization. Air Quality Guidelines for Europe-second edition: chapter 5.9 polycyclic aromatic hydrocarbons (PAHs). Copenhagen, Denmark.

Wiriya, W., Chantara, S., Sillapapiromsuk, S., Lin, N.-H., 2016. Emission Profiles of PM10-Bound Polycyclic Aromatic Hydrocarbons from Biomass Burning Determined in Chamber for Assessment of Air Pollutants from Open Burning. Aerosol and Air Quality Research, 16, 2716–2727.

Wiriya, W., Prapamontol, T., Chantara, S., 2013. PM10-bound polycyclic aromatic hydrocarbons in Chiang Mai (Thailand): Seasonal variations, source identification, health risk assessment and their relationship to air-mass movement. Atmos. Res., 124, 109–122.

Wiwatanadate, P., Liwsrisakun, C., 2011. Acute effects of air pollution on peak expiratory flow rates and symptoms among asthmatic patients in Chiang Mai, Thailand. International Journal of Hygiene and Environmental Health, 214, 251-257.

Yagishita, M., Kageyama, S., Ohshima, S., Matsumoto, M., Aoki, Y., Goto, S., Nakajima, D., 2015. Atmospheric concentration and carcinogenic risk of polycyclic aromatic hydrocarbons including benzo[*c*]fluorene, cyclopenta[*c,d*]pyrene, and benzo[*j*]fluoranthene in Japan. Atmospheric Environment, 115, 263-268. doi:10.1016/j.atmosenv.2015.05.050.

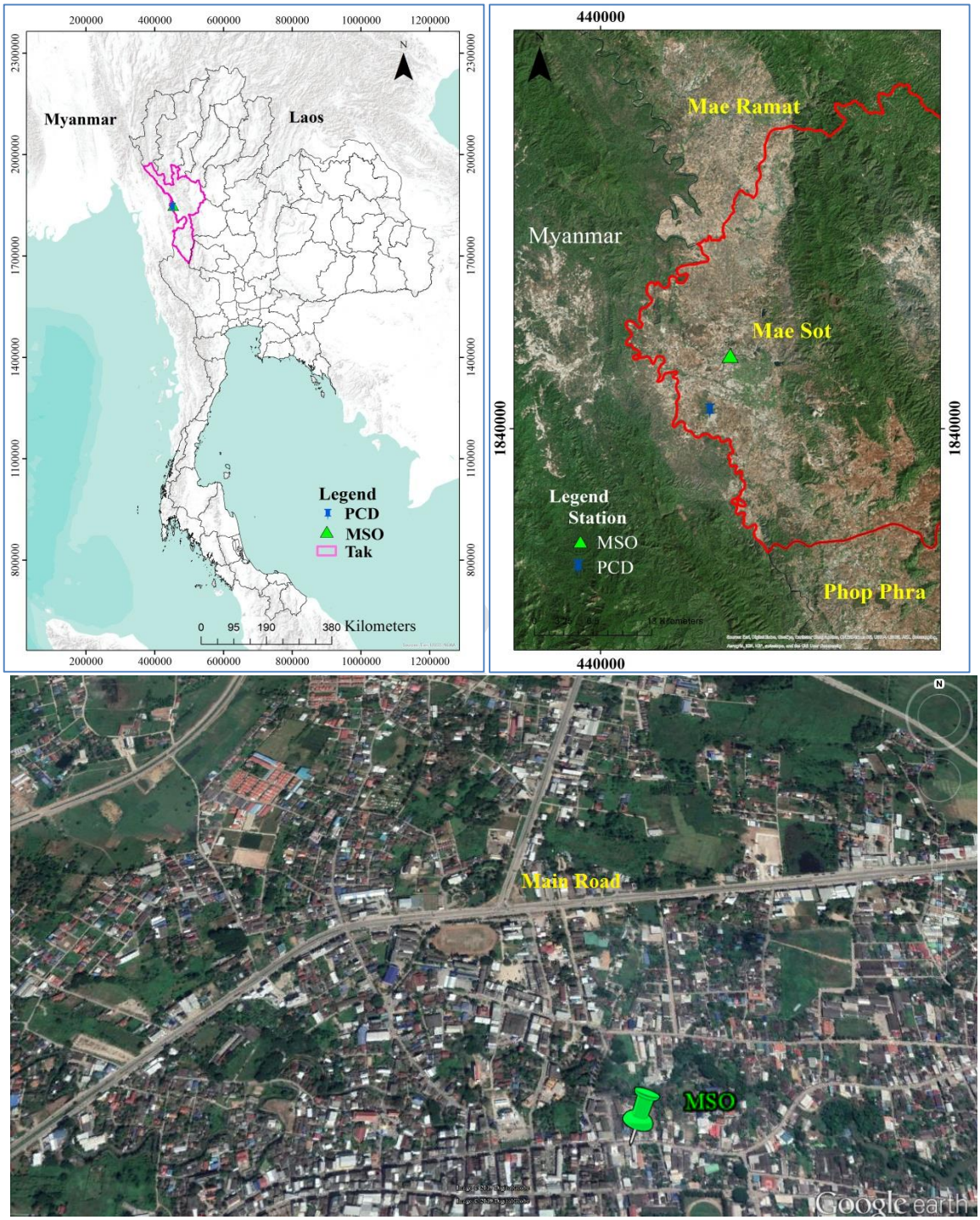
Yujie Xin, Y., Wang, G., Chen, L., 2016. Identification of Long-Range Transport Pathways and Potential Sources of PM10 in Tibetan Plateau Uplift Area: Case Study of Xining, China in 2014. Aerosol and Air Quality Research, 16, 1044–1054. doi: 10.4209/aaqr.2015.05.0296.

Yujie Xin, Y., Wang, G., Chen, L., 2016. Identification of Long-Range Transport Pathways and Potential Sources of PM10 in Tibetan Plateau Uplift Area: Case Study of Xining, China in 2014. Aerosol and Air Quality Research, 16, 1044–1054. doi: 10.4209/aaqr.2015.05.0296.

Zhu, C., Kawamura, K., Kunwar, B., 2015. Effect of biomass burning over the western North Pacific Rim: wintertime maxima of anhydrosugars in ambient aerosols from Okinawa. Atmos. Chem. Phys., 15, 1959–1973.

Zhu, L., Huang, X., Shi, H., Cai, X., Song, Y., 2011. Transport pathways and potential sources of PM10 in Beijing. Atmospheric Environment, 45, 594-604.

Figures and Tables



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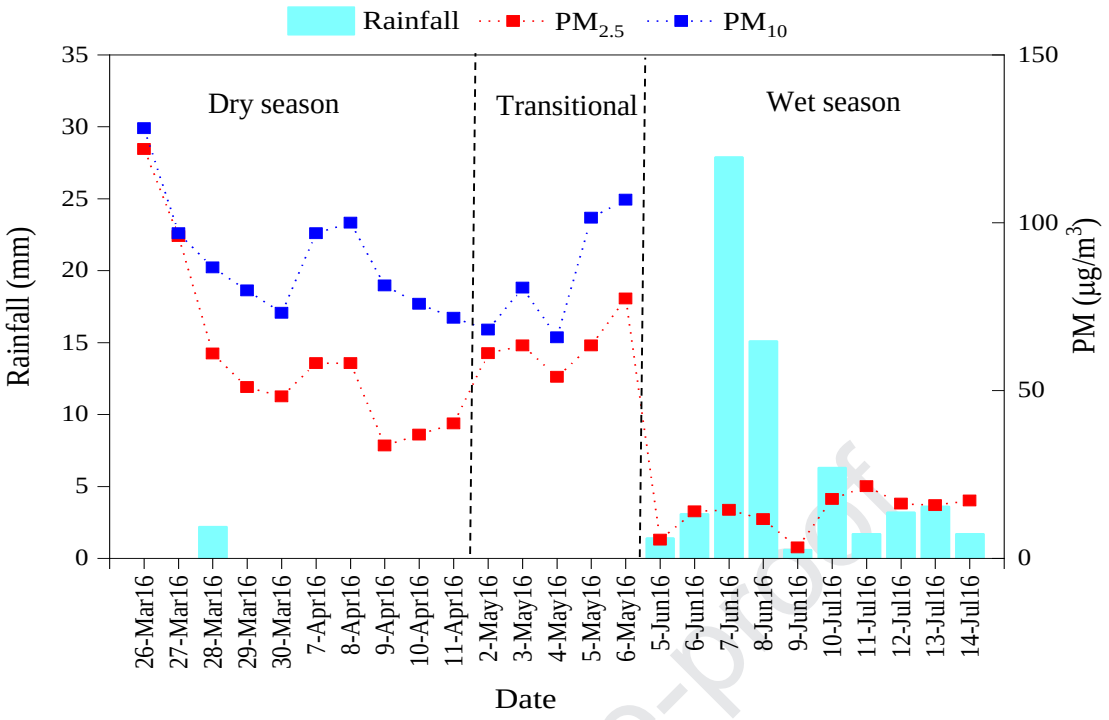

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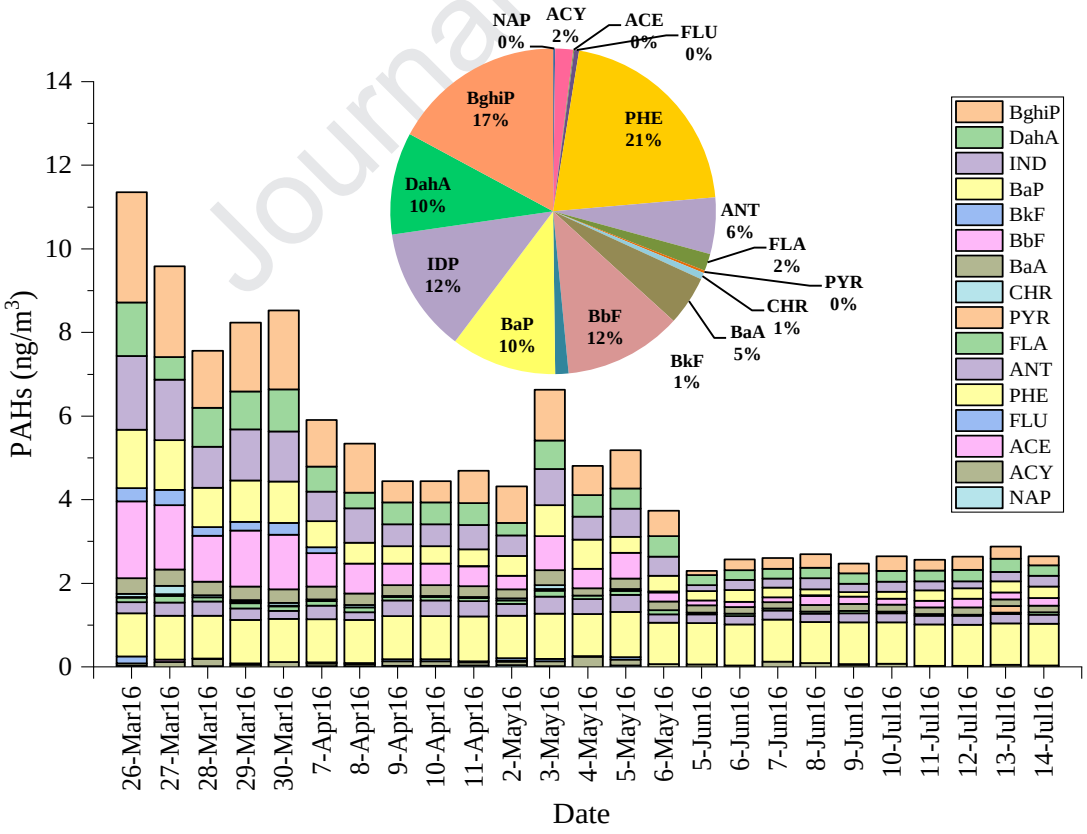


Fig. 3 Percentage contributions (pie graph) and temporal variations (bar graph) of $PM_{2.5}$ bound PAHs at sampling site.

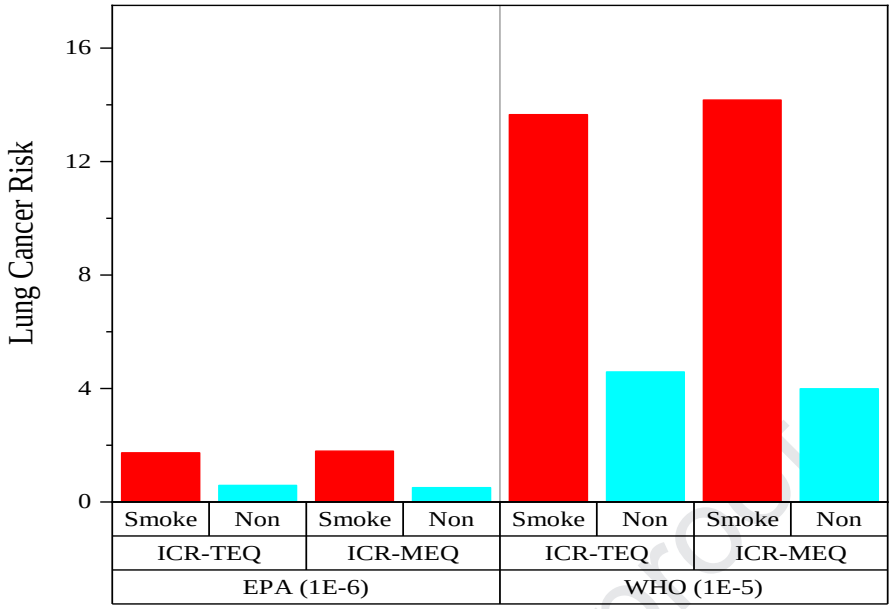
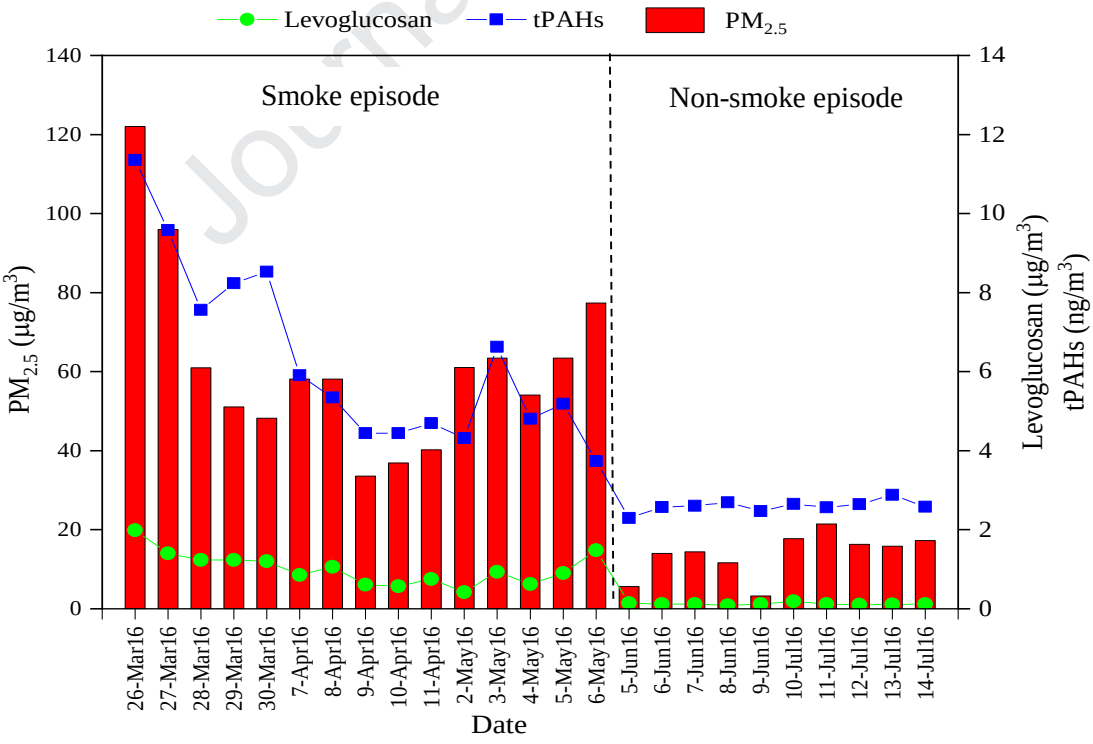


Fig. 4 Assessment of lifetime lung cancer risk from the TEQ_{BaP} and MEQ_{BaP} during smoke episode and non-smoke episode in Mae Sot district. The US-EPA and WHO Unit Risk were also remarked with 1.1×10^{-6} ng/m^3 and 8.7×10^{-5} ng/m^3, respectively.



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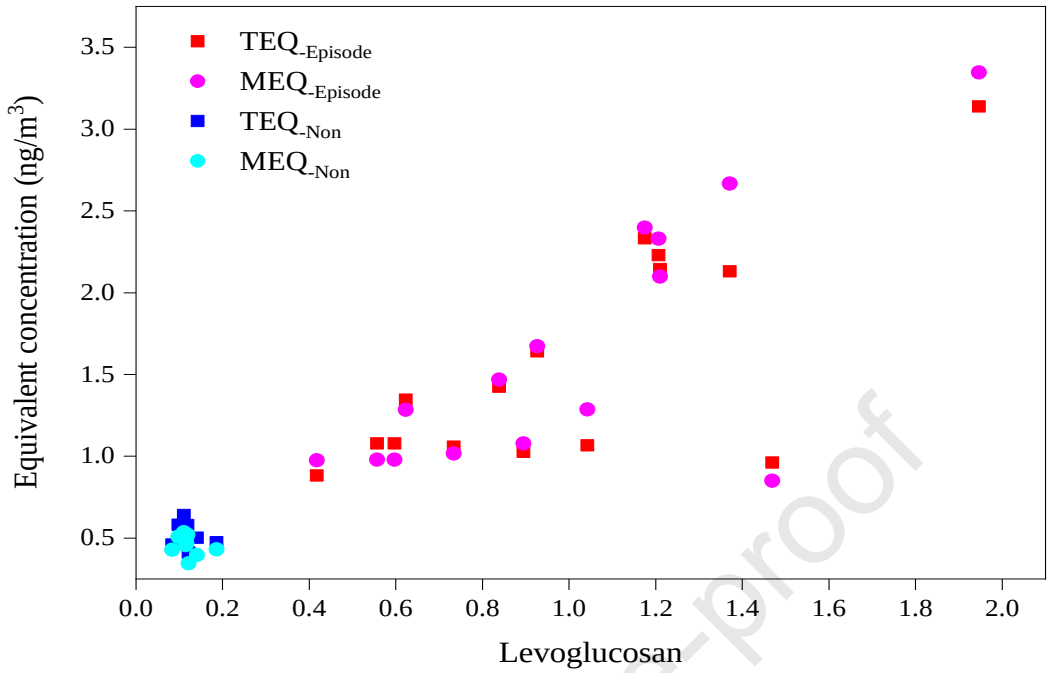
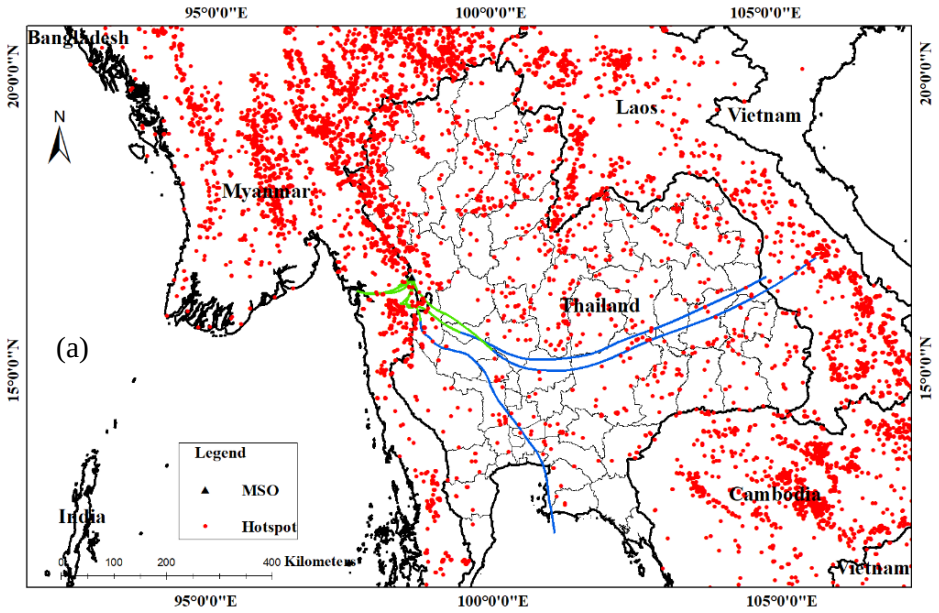


Fig. 6 Correlation between levoglucosan and equivalent concentrations during smoke episode and non-smoke episode.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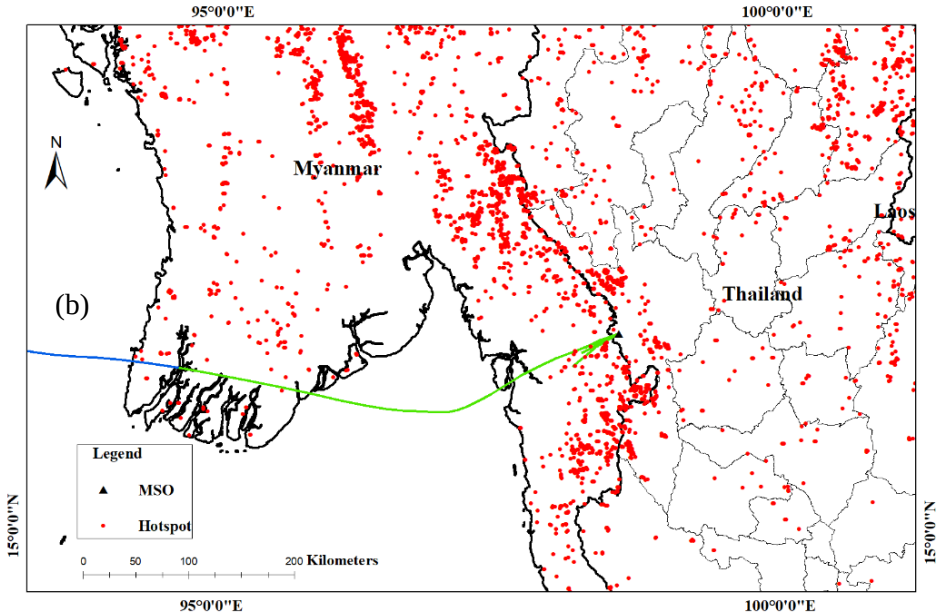
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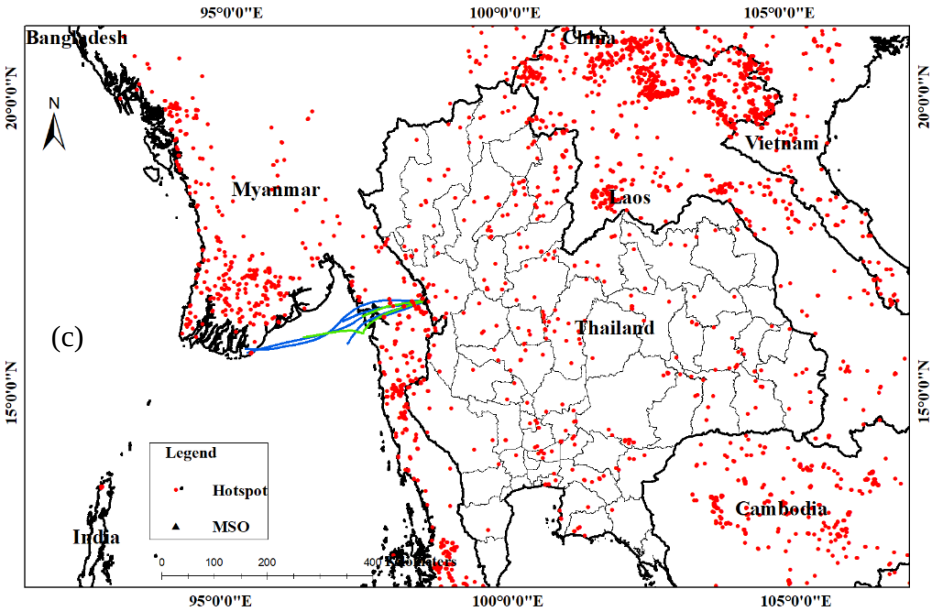
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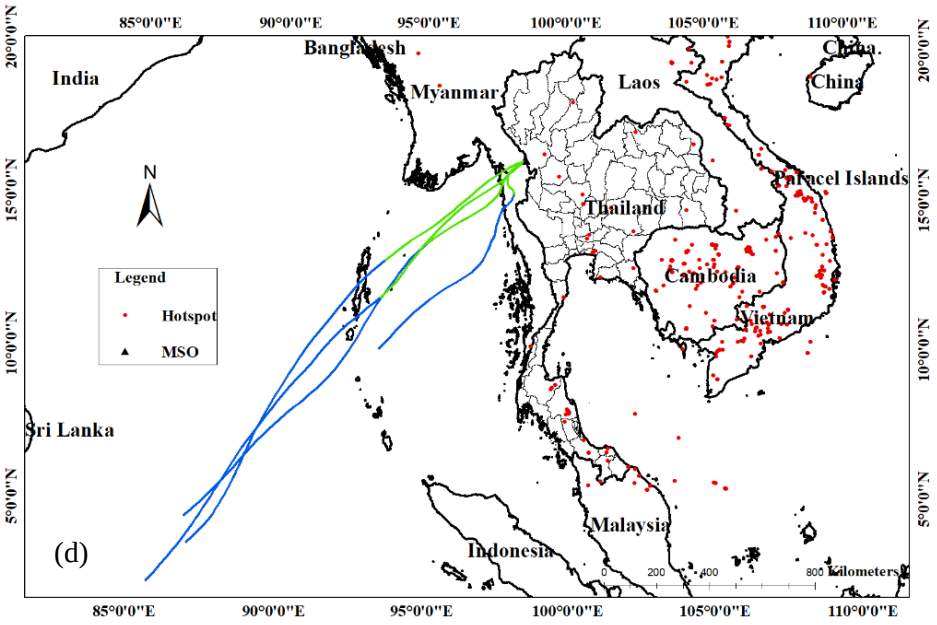
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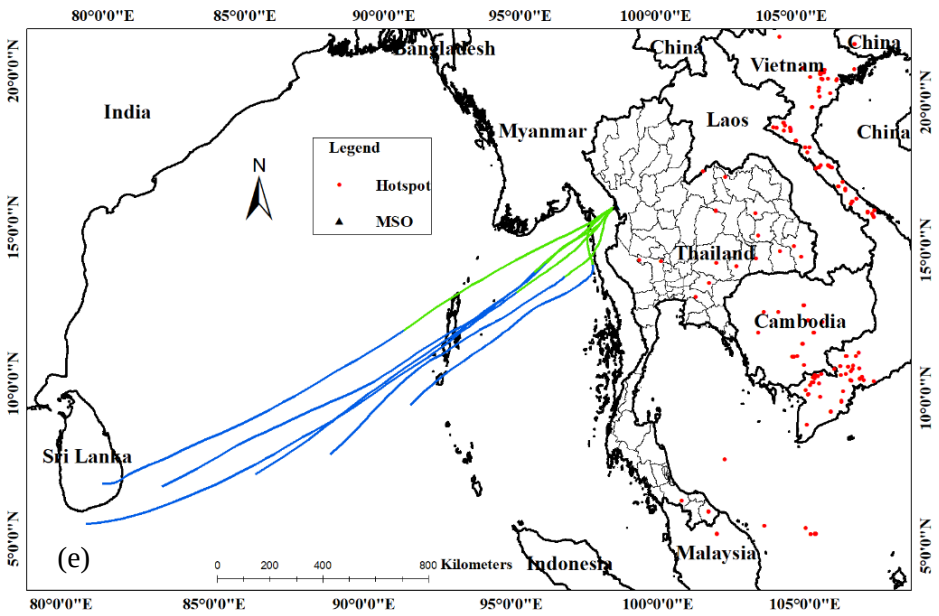


Fig. 7 Hotspots and 24 hr (green line), 72 hr (blue line) backward trajectories of air mass movements generated from the NOAA HYSPLIT model at 500 m AGL; endpoint 16 UTC for March (a), April (b), May (c), June (d) and July (e) in 2016.

Table 1 Range and mean of PAHs, PM and meteorological values measured during sampling period

Parameter	Abbreviation	IQ	IARC class*	Mean ± SD		Unit
				Episode	Non-episode	
Naphtalene	NAP	128		0.02 ± 0.01	ND	
Acenaphtylene	ACY	152		0.10 ± 0.05	0.06 ± 0.03	
Acenaphtene	ACE	154		0.02 ± 0.02	ND	
Fluorene	FLU	166	3	0.06 ± 0.04	0.02 ± 0.01	
Phenanthrene	PHE	178	3	1.04 ± 0.03	0.99 ± 0.01	
Anthracene	ANT	178	3	0.31 ± 0.07	0.21 ± 0.01	
Fluoranthene	FLA	202	3	0.11 ± 0.02	0.05 ± 0.01	
Pyrene	PYR	202	3	0.02 ± 0.01	0.16 ± 0.04	
Chrysene	CHR	228	2B	0.06 ± 0.05	ND	
Benz[*a*]anthracene	BaA	228	2B	0.29 ± 0.06	0.16 ± 0.02	ng/m^3
Benzo[*b*]fluoranthene	BbF	252	2B	0.84 ± 0.48	0.16 ± 0.03	
Benzo[*k*]fluoranthene	BkF	252	2B	0.20 ± 0.13	0.02 ± 0.03	
Benzo[*a*]pyrene	BaP	252	1	0.70 ± 0.33	0.21 ± 0.06	
Indeno[*1,2,3-cd*]pyrene	IDP	276	2B	0.85 ± 0.39	0.22 ± 0.04	
Dibenz[*a,h*]anthracene	DahA	278	2A	0.65 ± 0.27	0.26 ± 0.03	
Benzo[*ghi*]perylene	BghiP	276	3	1.21 ± 0.63	0.26 ± 0.07	
Total PAHs	tPAHs			6.32 ± 2.26	2.59 ± 0.15	
Toxicity equivalent	TEQ_{BaP}			1.57 ± 0.67	0.53 ± 0.07	
Mutagenic equivalent	MEQ_{BaP}			1.63 ± 0.76	0.46 ± 0.06	
Particulate matter (2.5)	$PM_{2.5}$			61.64 ± 22.85	13.76 ± 5.58	µg/m^3
Particulate matter (10)	PM_{10}	(Mar - May 2016)		87.55 ± 17.19	-	µg/m^3
Levoglucosan	Levo			1.00 ± 0.41	0.12 ± 0.03	µg/m^3
Temperature	Temp			30.63 ± 1.34	26.40 ± 0.74	°C
Relative humidity	RH			58.72 ± 4.89	85.19 ± 2.95	%
Rainfall	RN			0.15 ± 0.57	6.46 ± 8.63	mm
Wind speed	WS			32.84 ± 5.31	32.41 ± 7.00	km/hr

[a] QI = Quantification ion (m/z)

[b] The IARC Classified: 1= Carcinogenic to humans; 2A = Probably carcinogenic to humans; 2B = Possibly carcinogenic to humans; 3 = Not classifiable as to its carcinogenicity to humans

ND = Not detected

Table 2 Correlations between concentrations of levoglucosan, $PM_{2.5}$ and PAHs in both episodes

	Episode	Non
Levo	1	1
$PM_{2.5}$	0.795**	0.088
Nap	-0.547	a
ACY	-0.295	0.054
ACE	0.488	a
FLU	0.775**	a
PHE	-0.081	0.014
ANT	-0.482	0.473
FLA	0.514*	-0.033
PYR	-0.013	a
CHR	0.3	a
BaA	0.592*	0.124
BbF	0.736**	-0.579
BkF	0.545	a
BaP	0.723**	-0.125
IDP	0.796**	-0.041
DahA	0.698**	-0.233
BghiP	0.782**	-0.006
tPAHs	0.760**	-0.251

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

a Cannot be computed because at least one of the variables is constant.

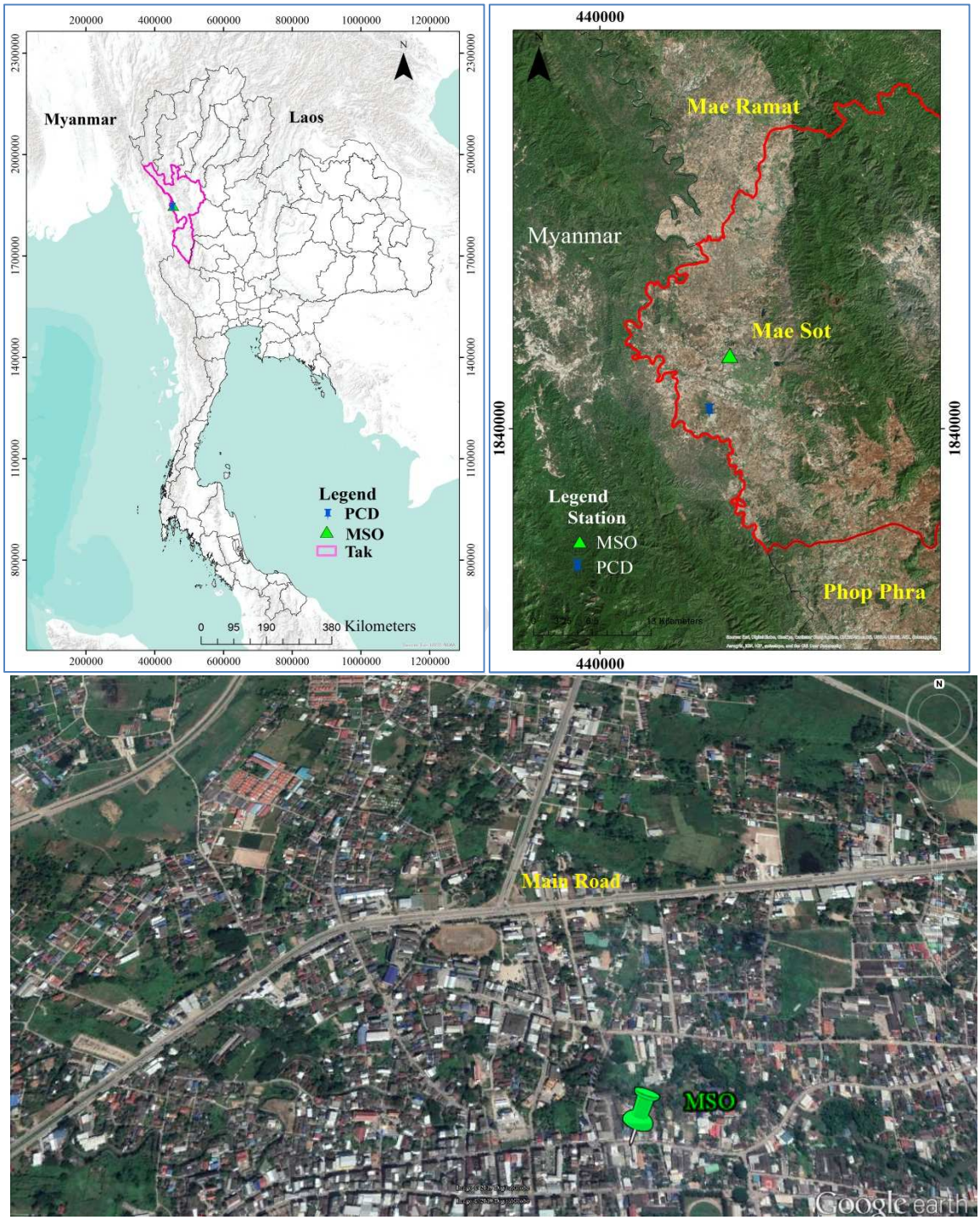
Table 3 Correlations between concentration of $PM_{2.5}$ and other parameters

	$PM_{2.5}$	PM_{10}	tPAHs	TEQ	MEQ	Levo	Temp	RH	Rain	WS
$PM_{2.5}$	1	0.630*	0.810**	0.769**	0.827**	0.864**	0.590**	-0.678**	-0.515**	-0.134
PM_{10}		1	0.263	0.077	0.204	0.671**	0.068	-0.359	a	-0.016
tPAHs			1	0.928**	0.961**	0.831**	0.445*	-0.576**	-0.471*	-0.242
TEQ_{BaP}				1	0.973**	.827(**)	0.505*	-0.578**	-0.475*	-0.314
MEQ_{BaP}					1	0.839**	0.486*	0.578**	-0.439*	-0.319
Levo						1	0.545**	-0.653**	-0.601**	-0.173
Temp							1	-0.927**	-0.387	0.357
RH								1	0.464*	-0.291
Rain									1	0.246
WS										1

** Correlation is significant at the 0.01 level (2-tailed).

* Correlation is significant at the 0.05 level (2-tailed).

Figures and Tables



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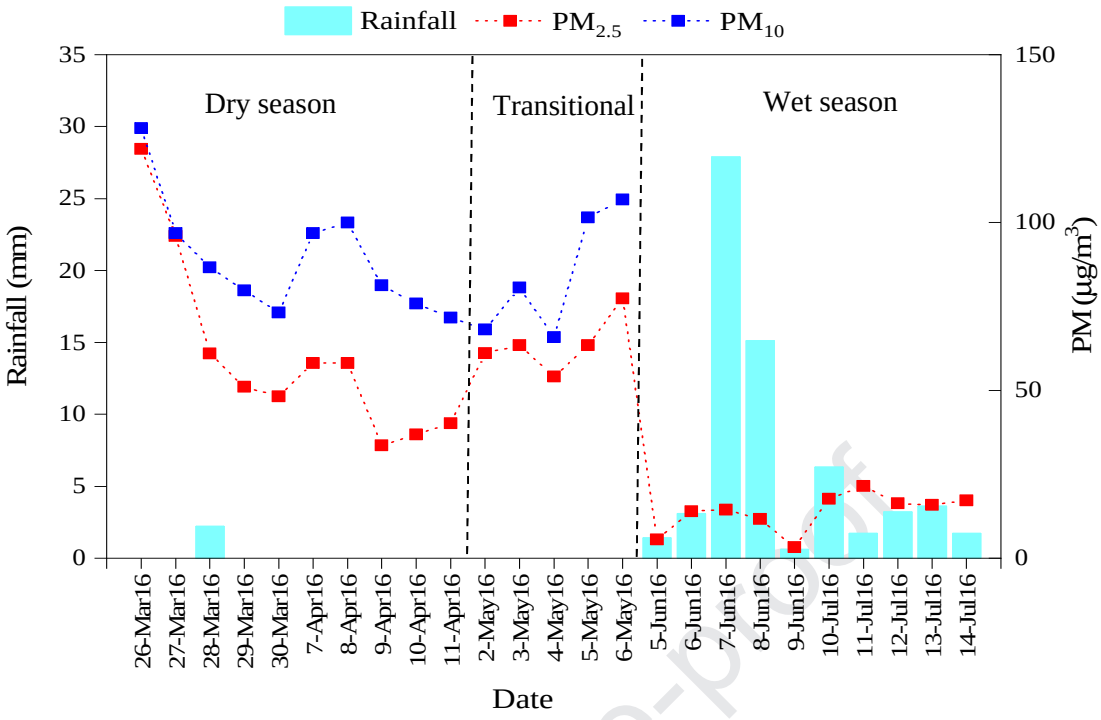

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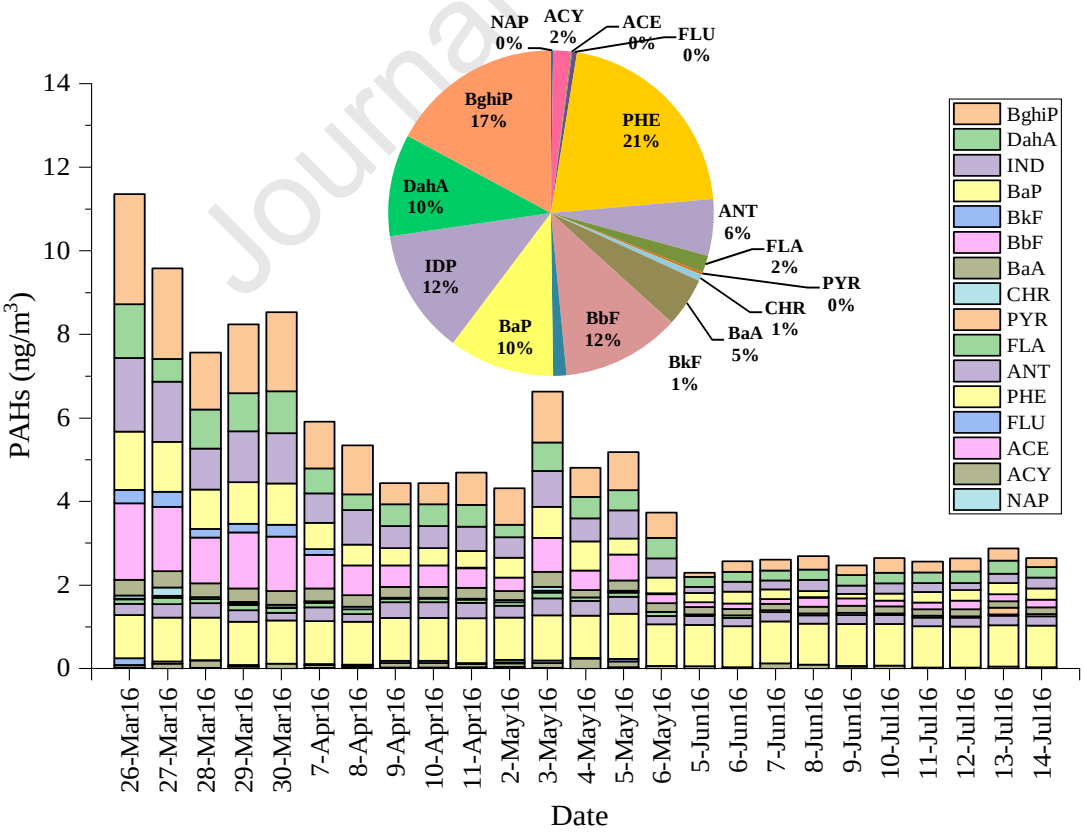

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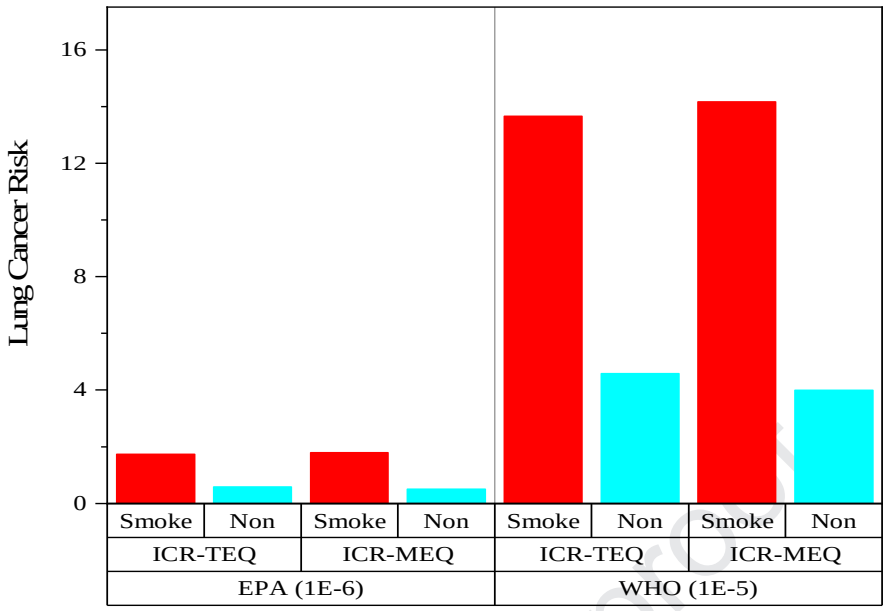

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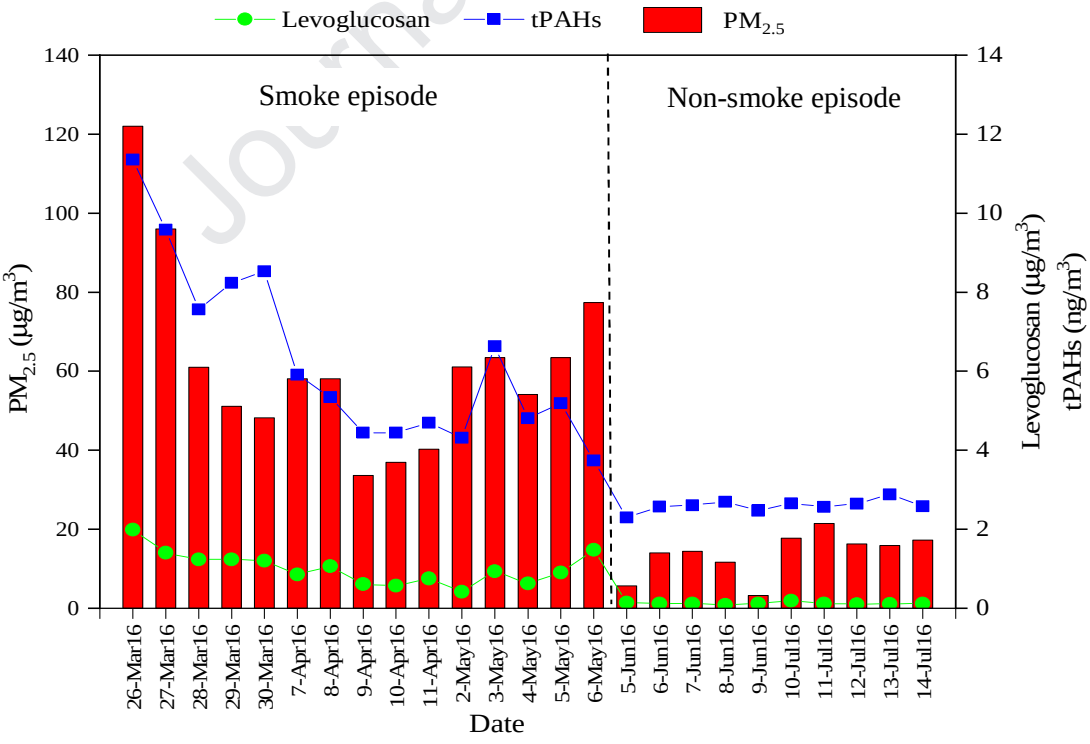

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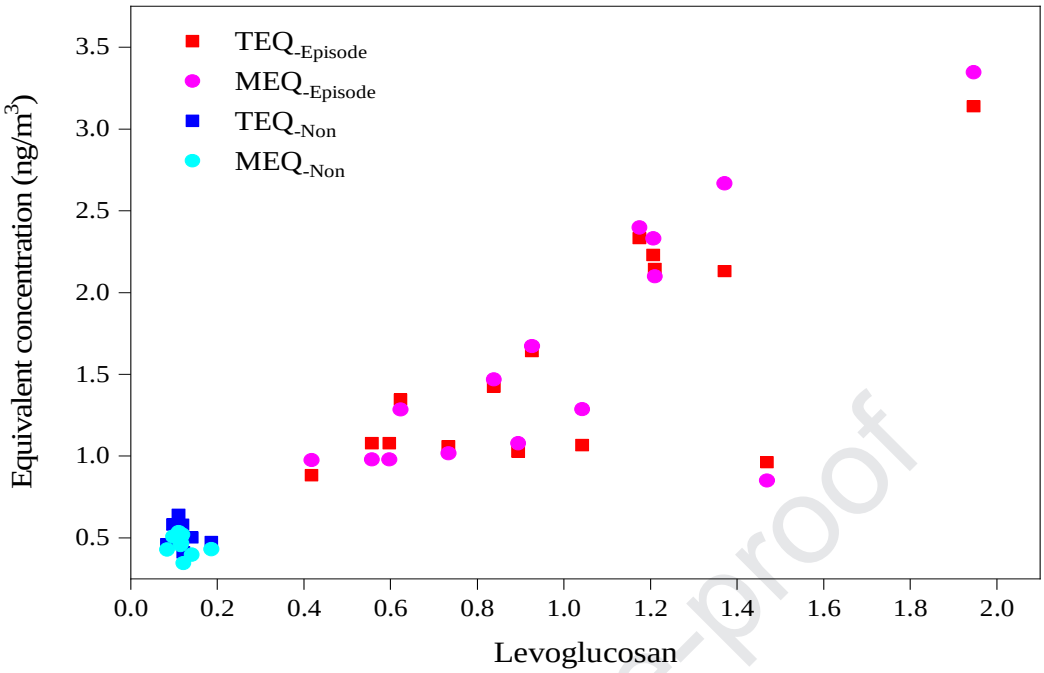

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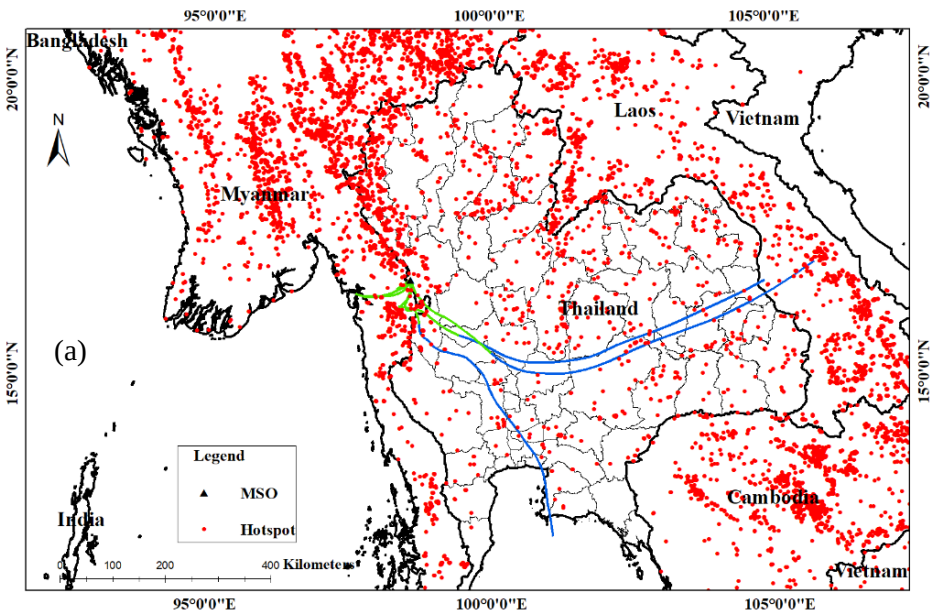
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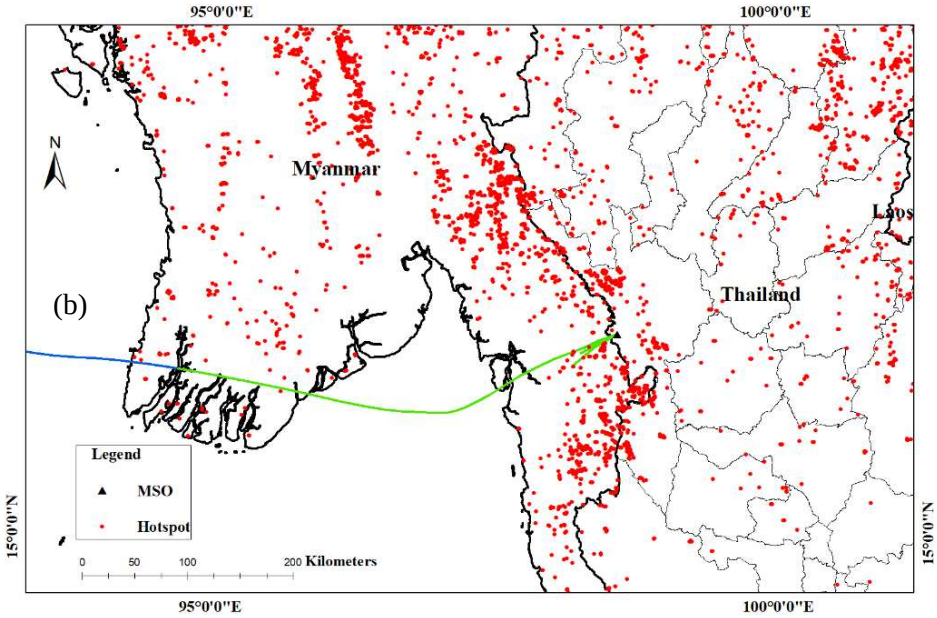
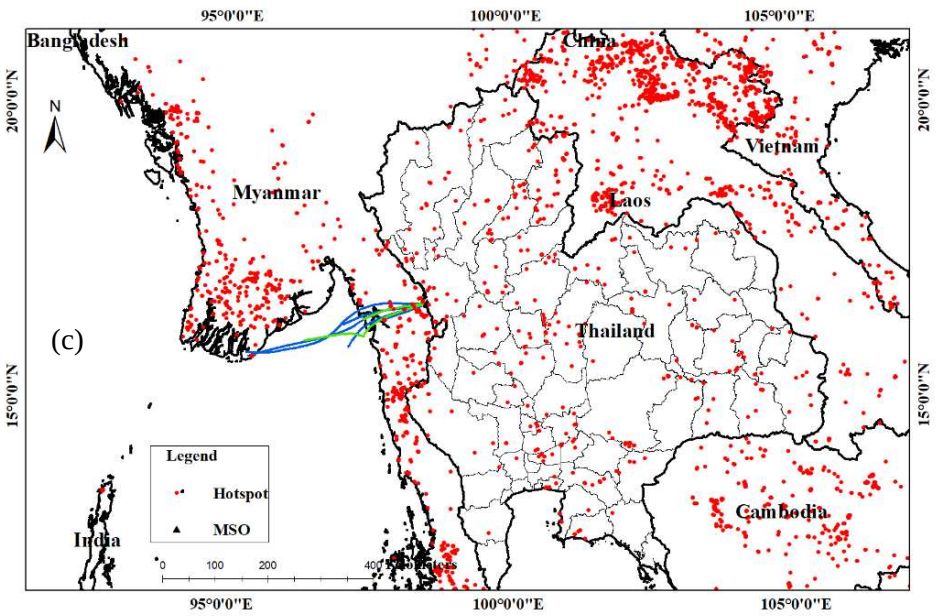
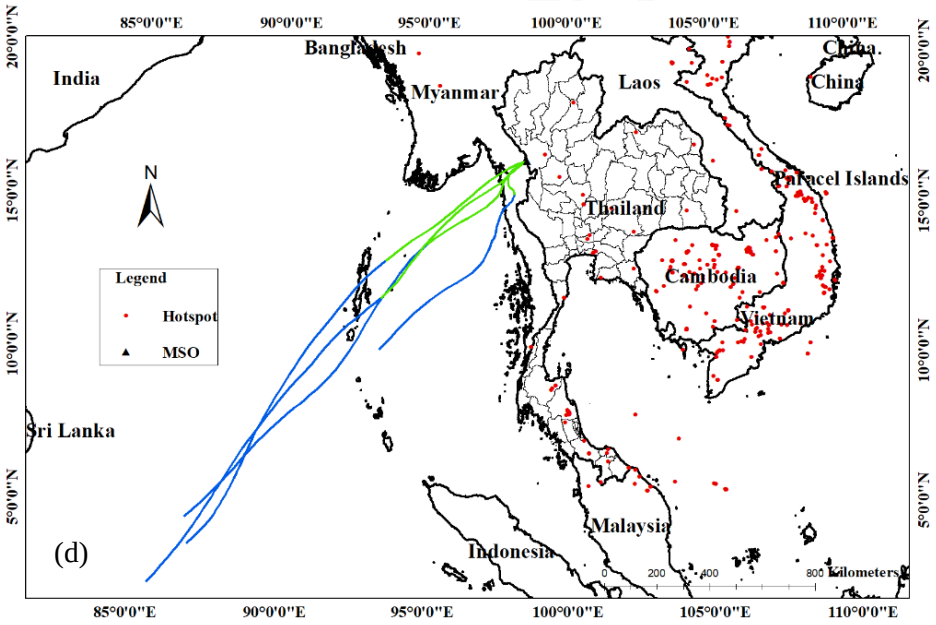

95°0'0"E
100°0'0"E
15°0'0"N
N
Myanmar
Laos
Thailand
(b)
Legend
MSO
Hotspot
0 50 100 200 Kilometers


95°0'0"E
100°0'0"E
105°0'0"E
20°0'0"N
15°0'0"N
Bangladesh
China
N
Vietnam
Myanmar
Laos
Thailand
(c)
Legend
Hotspot
MSO
India
Cambodia
0
100
200
400 Kilometers


85°0'0"E
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110°0'0"E
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15°0'0"N
10°0'0"N
5°0'0"N
India
Bangladesh
Myanmar
Laos
China
N
Paracel Islands
Thailand
Legend
Hotspot
MSO
Cambodia
Vietnam
Sri Lanka
Malaysia
Indonesia
(d)
0
200
400
800 Kilometers

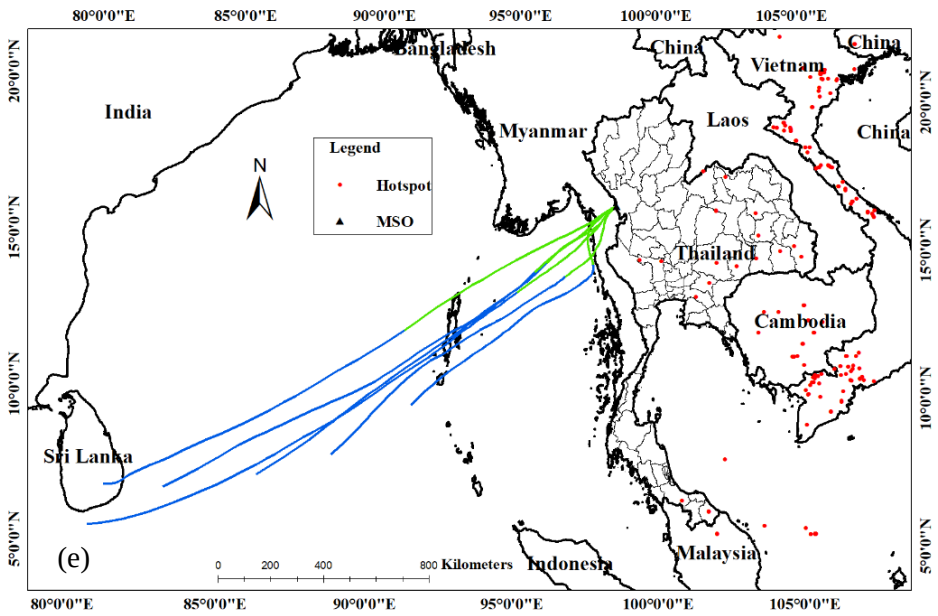

Fig. 7 Hotspots and 24 hr (green line), 72 hr (blue line) backward trajectories of air mass movements generated from the NOAA HYSPLIT model at 500 m AGL; endpoint 16 UTC for March (a), April (b), May (c), June (d) and July (e) in 2016.

Table 1 Range and mean of PAHs, PM and meteorological values measured during sampling period

Parameter	Abbreviation	IQ	IARC class*	Mean ± SD		Unit
				Episode	Non-episode	
Naphtalene	NAP	128		0.02 ± 0.01	ND	
Acenaphtylene	ACY	152		0.10 ± 0.05	0.06 ± 0.03	
Acenaphtene	ACE	154		0.02 ± 0.02	ND	
Fluorene	FLU	166	3	0.06 ± 0.04	0.02 ± 0.01	
Phenanthrene	PHE	178	3	1.04 ± 0.03	0.99 ± 0.01	
Anthracene	ANT	178	3	0.31 ± 0.07	0.21 ± 0.01	
Fluoranthene	FLA	202	3	0.11 ± 0.02	0.05 ± 0.01	
Pyrene	PYR	202	3	0.02 ± 0.01	0.16 ± 0.04	
Chrysene	CHR	228	2B	0.06 ± 0.05	ND	
Benz[*a*]anthracene	BaA	228	2B	0.29 ± 0.06	0.16 ± 0.02	ng/m^3
Benzo[*b*]fluoranthene	BbF	252	2B	0.84 ± 0.48	0.16 ± 0.03	
Benzo[*k*]fluoranthene	BkF	252	2B	0.20 ± 0.13	0.02 ± 0.03	
Benzo[*a*]pyrene	BaP	252	1	0.70 ± 0.33	0.21 ± 0.06	
Indeno[*1,2,3-cd*]pyrene	IDP	276	2B	0.85 ± 0.39	0.22 ± 0.04	
Dibenz[*a,h*]anthracene	DahA	278	2A	0.65 ± 0.27	0.26 ± 0.03	
Benzo[*ghi*]perylene	BghiP	276	3	1.21 ± 0.63	0.26 ± 0.07	
Total PAHs	tPAHs			6.32 ± 2.26	2.59 ± 0.15	
Toxicity equivalent	TEQ_{BaP}			1.57 ± 0.67	0.53 ± 0.07	
Mutagenic equivalent	MEQ_{BaP}			1.63 ± 0.76	0.46 ± 0.06	
Particulate matter (2.5)	$PM_{2.5}$			61.64 ± 22.85	13.76 ± 5.58	µg/m^3
Particulate matter (10)	PM_{10}	(Mar - May 2016)		87.55 ± 17.19	-	µg/m^3
Levoglucosan	Levo			1.00 ± 0.41	0.12 ± 0.03	µg/m^3
Temperature	Temp			30.63 ± 1.34	26.40 ± 0.74	°C
Relative humidity	RH			58.72 ± 4.89	85.19 ± 2.95	%
Rainfall	RN			0.15 ± 0.57	6.46 ± 8.63	mm
Wind speed	WS			32.84 ± 5.31	32.41 ± 7.00	km/hr

[a] QI = Quantification ion (m/z)

[b] The IARC Classified: 1= Carcinogenic to humans; 2A = Probably carcinogenic to humans; 2B = Possibly carcinogenic to humans; 3 = Not classifiable as to its carcinogenicity to humans

ND = Not detected

Table 2 Correlations between concentrations of levoglucosan, $PM_{2.5}$ and PAHs in both episodes

	Episode	Non
Levo	1	1
$PM_{2.5}$	0.795**	0.088
Nap	-0.547	a
ACY	-0.295	0.054
ACE	0.488	a
FLU	0.775**	a
PHE	-0.081	0.014
ANT	-0.482	0.473
FLA	0.514*	-0.033
PYR	-0.013	a
CHR	0.3	a
BaA	0.592*	0.124
BbF	0.736**	-0.579
BkF	0.545	a
BaP	0.723**	-0.125
IDP	0.796**	-0.041
DahA	0.698**	-0.233
BghiP	0.782**	-0.006
tPAHs	0.760**	-0.251

** Correlation is significant at the 0.01 level (2-tailed).
* Correlation is significant at the 0.05 level (2-tailed).
a Cannot be computed because at least one of the variables is constant.

Table 3 Correlations between concentration of $PM_{2.5}$ and other parameters

	$PM_{2.5}$	PM_{10}	tPAHs	TEQ	MEQ	Levo	Temp	RH	Rain	WS
$PM_{2.5}$	1	0.630*	0.810**	0.769**	0.827**	0.864**	0.590**	-0.678**	-0.515**	-0.134
PM_{10}		1	0.263	0.077	0.204	0.671**	0.068	-0.359	a	-0.016
tPAHs			1	0.928**	0.961**	0.831**	0.445*	-0.576**	-0.471*	-0.242
TEQ_{BaP}				1	0.973**	.827(**)	0.505*	-0.578**	-0.475*	-0.314
MEQ_{BaP}					1	0.839**	0.486*	0.578**	-0.439*	-0.319
Levo						1	0.545**	-0.653**	-0.601**	-0.173
Temp							1	-0.927**	-0.387	0.357
RH								1	0.464*	-0.291
Rain									1	0.246
WS										1

** Correlation is significant at the 0.01 level (2-tailed).
* Correlation is significant at the 0.05 level (2-tailed).

Hightlights:

- Mae Sot District, Tak Province, Thailand was affected with crisis of $PM_{2.5}$ concentration in burning season, the same as the northern Thailand.
- Levoglucosan indicated the emission sources of $PM_{2.5}$ from biomass burning.
- Lifetime lung cancer risk during smoke episode exceeded the acceptable cancer risk of US-EPA's recommended.
- Most air masses generated from the western region of Thailand and blew through burning areas to the study site during the smoke episode.