

Chemical Properties of the Southeast Asian Haze from Indonesian Peatland Fires

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Abstract

Anthropogenic activities such as forest degradation; the expansion of commodity crops, including oil palm; and drainage of fire-prone peatlands have resulted in extensive peatland and forest fires in Indonesia. Uncontrollable and intense peatland fires have frequently occurred in the Indonesian islands of Sumatra and Kalimantan, particularly during the hot and dry seasons of the El Niño-Southern Oscillation. Air pollution in the form of transboundary haze is an urgent issue in Southeast Asia. The physical and chemical properties of the aerosols from peatland fires are fundamental to know when assessing their impact on human health and the environment, conducting source apportionment of haze events, and unraveling their dynamic state in the environment (e.g., secondary aerosol formation). In this paper, we conduct a holistic review of source profiles, key indicators of Indonesian peatland fires at source and receptor sites, and transformation of haze during its long-range transport (aging, secondary aerosol formation) as well as source apportionment in Southeast Asia, using a comprehensive chemical component dataset. Knowledge of the chemical characteristics of particulate matter at Indonesian peatland fire sources and receptor sites is still limited, especially regarding controlling factors (e.g., combustion conditions, peat composition and the effects of vegetative burning on peatland) that determine the Indonesian peatland fire source profile of particulate matter. Additionally, the process of secondary organic aerosol formation derived from Indonesian peatland fires during transport to receptor sites remains largely unresolved.

Key words: chemical composition, haze, Indonesia, peatland fire, Southeast Asia

1. Introduction

Tropical peatlands are globally and regionally important as carbon pools and carbon soil-atmosphere exchange process, and in provisioning ecosystem services of peat swamp forests (Page *et al.*, 2011). The peatland area of Southeast Asia (below, “SEA”) covers 23.7 Mha, of which 20.7 Mha is in Indonesia (87.3%), followed by 2.59 Mha in Malaysia (Omar *et al.*, 2022). Indonesia has either the world’s largest tropical peatland area (Joosten, 2009) or the world’s second largest tropical peatland area (UNEP, 2022a), and small-scale land management fires have been used for centuries in Indonesia (Simorangkir, 2007). However, anthropogenic activities such as forest degradation, the expansion of commodity crops including oil palm, and drainage of fire-prone peatlands have resulted in extensive peatland and forest fires in Indonesia. Uncontrollable, intense peatland fires have frequently

occurred in the Indonesian islands of Sumatra and Kalimantan, particularly during the hot and dry seasons of the El Niño-Southern Oscillation, since the early 1980s (Field *et al.*, 2016; Yulianti *et al.*, 2020; UNEP, 2022b). The first catastrophic fire was in 1997 and subsequently, intense fires have occurred in 2006, 2013 (a non-drought year), 2015 and 2019. Four million six hundred thousand hectares of land in Indonesia burned in 2015 (Lohberger *et al.*, 2018), and CO₂ emissions from the peatland fire exceeded the total annual emissions of a developed nation as big as Japan for that year (World Bank, 2015; Field *et al.*, 2016; Crump, 2017). In addition to vast carbon emissions affecting the global climate, large-scale smoke generation (transboundary haze pollution) degrades the regional air quality significantly in SEA, resulting in adverse impacts on both ecosystems and human health (Crippa *et al.*, 2016; Koplitz *et al.*, 2016; Cheong *et al.*, 2019; Phung *et al.*, 2022; Yin, 2023). Furthermore, Hunag

et al. (2023) have discovered an intricate fire-weather feedback loop that increases air pollution exposure through the radiative effects of smoke aerosols (aerosol-cloud interaction) in Mediterranean and monsoon climate regimes. The extensive fires have resulted in serious economic damage through its impact on various environmental and social services. For example, the economic loss from the extensive peatland fires and haze in 2015 was estimated at 16.1 million USD in Indonesia (Glauber *et al.*, 2016).

Air pollution in the form of transboundary haze is an urgent issue in SEA. Gaining information on the physical and chemical properties of the aerosols from peatland fires is fundamental to assessing their impact on human health and the environment, conducting source apportionment of haze events, and unraveling their dynamic state in the environment (e.g., secondary aerosol formation). Aerosols from biomass burning, including from peatland fires, are mostly in fine size range (Reid *et al.*, 2005). Human mortality due to exposure to ambient fine particulate matter (PM) with an aerodynamic diameter less than 2.5 μm (PM_{2.5}) is generally calculated by an exposure-response function (Burnett *et al.*, 2014) based on a multitude of epidemiological studies that have focused on long-term exposure effects (Pozzer *et al.*, 2023). This function describes the relative risk dependency on PM_{2.5} mass concentration, independent of source or composition. An increasing number of studies, however, have suggested that sources and compositions of fine particles may significantly influence the health effects (Zhang *et al.*, 2023). Unequal toxicity of individual PM_{2.5} components has been reported (Wyzga and Rohr, 2015; Park *et al.*, 2018), and chemical composition is considered to be a major determinant of the toxicity of airborne PM (Shiraiwa *et al.*, 2017). Chemical components of transboundary haze such as heavy metals and polycyclic aromatic hydrocarbons (PAHs) are critically important species that are relevant in health risk assessment^{71, 88} (Each superscript corresponds to a particular reference in the literature list in the separate dataset that can be found in the Supplementary Information. There is also a list of abbreviations in the “Appendix” sheet in the dataset. The dataset is described at the end of this section and in the next section).

Aerosol radiative effects include direct aerosol-radiation through direct scattering and absorption of incoming solar and outgoing terrestrial (infrared) radiation in the atmosphere, and indirect aerosol-cloud interactions through alteration of cloud optical properties and the formation of clouds and precipitation as cloud condensation nuclei (CCN) (Penner *et al.*, 2001). Evaluating aerosol radiative forcing involves examining the physical (e.g., size distribution, optical properties, hygroscopicity) and chemical (e.g., water-solubility of inorganic components and organic carbon as CCN) properties of the aerosols. Therefore, the chemical

composition of haze (e.g., water-soluble ions and light-absorbing carbonaceous components) is one of the key parameters for evaluating its radiative effects.

Knowing the physical and chemical properties of the haze from Indonesian peatland fires (below, “IPFs”) is indeed indispensable to assessing its human health and environmental impacts. It should be noted, however, that there exist complex interactions among environmental and socio-economic factors hidden behind these peatland fires (Sze *et al.*, 2019; Shigetomi *et al.*, 2020; Girkin *et al.*, 2022; UNEP, 2022b). To manage and mitigate the fires’ risk to human health and livelihoods, biodiversity and the global climate, visualization of the interactions through a systemic lens is crucial. A “drivers, pressures, state, impact and response” (DPSIR) framework (Smeets and Weterings, 1999) has been used in modelling the relationship between environmental and human (socio-economic) systems. We have mapped the DPSIR framework into the cause-effect chains of environmental problems associated with IPFs, as shown in Fig. 1. In the “state” element of the DPSIR framework, we have emphasized aerosols (haze) from IPFs, particularly their physical and chemical properties to ascertain their position and interrelations with other components of the framework.

Many field and laboratory studies have been conducted to investigate the chemical characteristics of smoke haze from IPFs at source and receptor sites in SEA. Table S1 gives a summary of review papers relating to physical (not including optical properties) and chemical properties of haze from IPFs and other biomass burning, and source apportionment of PM (including contribution of IPFs, or IPF and other sources to PM concentrations) according to field observations in this region and laboratory studies based on combustion of Indonesian or Malaysian peat. Some papers also cover the climate, human health and economic impacts of the haze^{78, 99, 116}. They do not refer, however, to laboratory studies and cutting-edge analysis, nor do they cover sufficient chemical species for impact assessment, source profiling or ascertaining the dynamic state, and limit their target area to a certain country. Fujii and Tohno¹⁰⁴ carried out a narrative review focusing on the chemical compositions of IPF-derived aerosols based on field and laboratory studies and source apportionment of PM. Their review contained information on various species but not from a systematic perspective.

Our research questions were fourfold: 1) What differences exist in chemical compositions between haze from IPFs and non-haze samples in SEA (source profiles and contributions of other sources, particularly at receptor sites), 2) What differences exist in chemical compositions between haze sampled at source and at receptor sites (unique source profiles if available, key source indicators and aging process of the haze), 3) Ways in which advanced analytical techniques are applied in

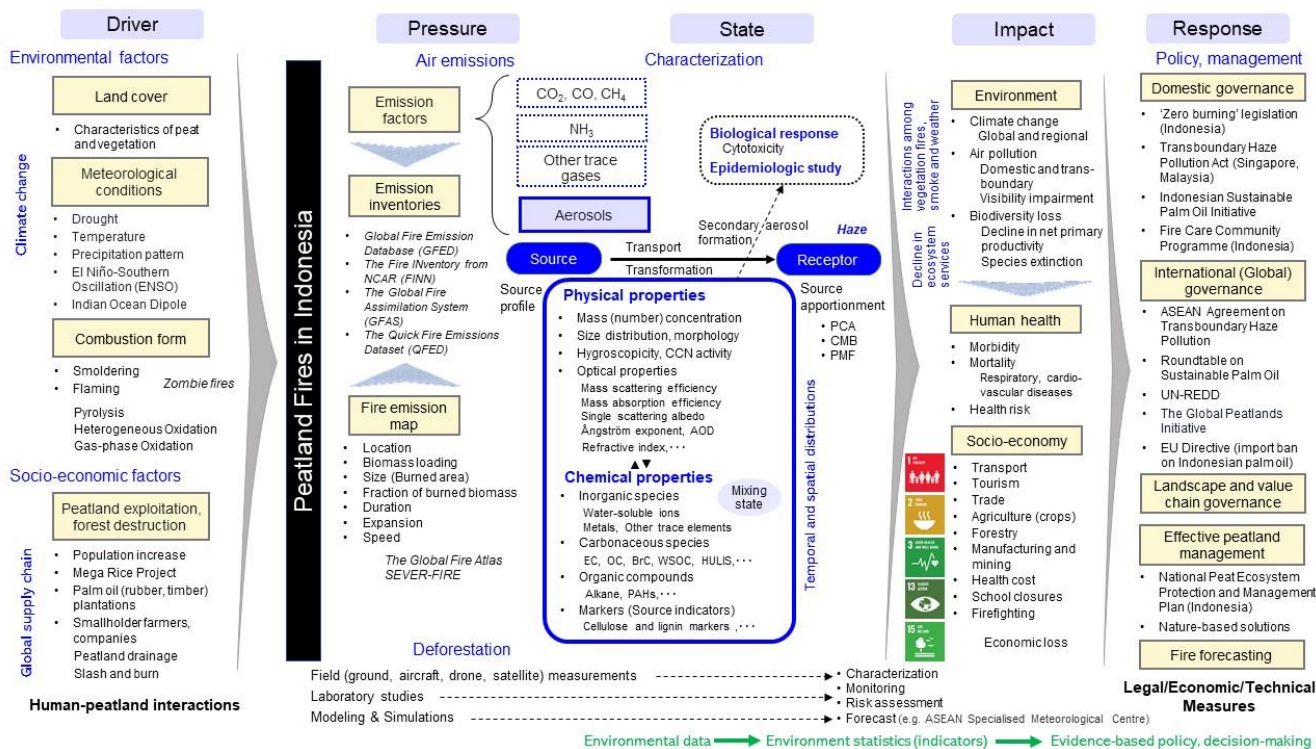


Fig. 1 Schematic representation of the cause-effect chains of environmental problems associated with Indonesian peatland fires according to a DPSIR framework. Emphasis is placed on aerosols from the fires in the “state” element of the framework. (Revised from Fig. 1 in Fujii and Tohno¹⁰⁴)

investigating the chemical properties of haze, and 4) The extent to which smoke haze degrades the air quality in SEA (the potential of peatland fires to contribute to and elevate ambient PM concentrations). Therefore, we conducted a holistic review of source profiles, key indicators of IPFs at source and receptor sites, and transformation of haze during its long-range transport (aging, secondary aerosol formation) as well as source apportionment in SEA, using a comprehensive chemical component dataset.

Van *et al.*¹¹⁶ discussed the definition of haze from these fires and proposed to use $PM_{2.5} > 50 \mu g m^{-3}$, visibility < 5 km and relative humidity (RH) $< 90\%$ as criteria for a “haze day,” but we used whatever definition of haze each respective article on field studies used as it was. The origin of the haze in each article was confined to IPFs using air mass backward trajectory analysis, meteorological factors and other factors. That meant clear differences in total PM mass concentration or visibility, etc. were found between haze events from IPFs and non-haze periods in each field study. Firstly, we created a detailed dataset of as many chemical compositions as possible and size distributions of the haze as well as possible indicators of peatland fires by collecting and reviewing the data from field and laboratory studies. Contained therein is also the contribution of IPFs or biomass burning to PM concentrations (annual or long-term average) in SEA. Our dataset creation methods and review process are described in the next section. Our dataset enables a subsequent holistic review of the chemical compositions of haze from IPFs in SEA.

2. Methods

Our dataset is based on a literature search and review to identify and extract relevant articles from scientific databases under appropriate inclusion and exclusion criteria. We aimed to find comprehensive chemical properties of IPF-derived PM (haze) at source and receptor sites in SEA with size distributions as well as laboratory-generated PM from burning of Indonesian or Malaysian peats. Hence, studies were included when they were in peer-reviewed publications, written in English with a full text article available, and 1) included mass concentration or mass fraction of chemical species in PM (all available species, carbonaceous components, water-soluble ions, elements, organic compounds, isotopes, aerosol mass spectra, etc.) with the PM samples having been collected during a haze period or at (near) fire sources in SEA, 2) included mass concentration, emission factor (below, “EF”), or emission ratio of generated PM species in laboratory studies, or 3) presented the contribution of IPFs or biomass burning to PM using samples collected during haze periods in SEA. Studies were excluded when they 1) focused on a specific purpose (e.g., optical properties, simulation models) and included no observed chemical properties, or 2) were review articles, book chapters, or reports, except for one pioneering US-EPA report⁵.

Article identification and selection were conducted using three databases, Web of Science, Scopus and ScienceDirect, searching for publications that included the queries in Table S2. The time range was from 1990 to 31

March 2023. We identified 1,769 suitable articles in the databases. After removing duplicates and screening the titles and abstracts, we had 232 articles remaining. We reviewed the full-text articles using the study selection criteria mentioned in the paragraph above. Ultimately, we included 112 in our review, with 13 articles added through snowballing and nine review papers excluded. A comparison of the nine review articles is shown in Table S1 as described in Section 1, and the articles are included only in the literature list of the dataset in the Supplementary Information. We found one newly published article after completing our screening process, and included it in our dataset. Lastly we created a dataset of the 113 articles using a Microsoft Excel file with 13 sheets, consisting of: 1) a cover sheet, 2) carbonaceous components, 3) water-soluble ions, 4) elements, 5) organic compounds excluding PAHs, 6) PAHs, 7) aerosol mass spectra analyses, 8) isotopes, 9) lab. experiments, 10) size distributions, 11) source apportionments, 12) a list of references and 13) an appendix. PM sampling information (during haze periods, and partly including non-haze periods), total PM mass concentration (or EF) and mass concentration of each species (EF or emission ratio) are contained in sheets 2)–6). The contents of the sheets 2)–13) are listed and described in the cover sheet. Species concentrations (or EF) are available for five countries: Indonesia, Brunei, Malaysia, Singapore and Thailand, but are missing for other countries in SEA.

3. Chemical Compositions of Peatland Fire Aerosols

3.1 Carbonaceous Components

When haze was induced by IPFs, a significant increase in the concentrations of total carbon (TC), organic carbon (OC), water-soluble OC (WSOC) and carbon content of humic-like substances (HULIS-C) was typically observed at receptor sites as shown in the Supplementary Information (“Carbonaceous Components” sheet in the dataset). For example, the OC concentrations in the total suspended particulates (TSP) during a haze period in 1991 and a non-haze period in 1992 at Petaling Jaya in Malaysia were 74 and 14 $\mu\text{g m}^{-3}$, respectively². In Singapore, significantly higher OC concentrations were observed¹⁰⁷ in the $\text{PM}_{2.5}$ of smoke dominant samples ($25 \pm 7 \mu\text{g m}^{-3}$) compared to those of non-smoke dominant samples ($5 \pm 1 \mu\text{g m}^{-3}$) in 2012–2013 and 2015. In Thailand, the OC concentrations in $\text{PM}_{2.5}$ during haze and non-haze periods in 2019–2020 were 10.1 and $2.1 \pm 0.2 \mu\text{g m}^{-3}$, respectively¹²⁰. Although the WSOC data are limited, there are some reports of an increase in WSOC concentrations during haze events in Malaysia¹⁰⁵, Singapore⁹¹ and Thailand^{120,122}. For HULIS-C, Fujii *et al.*¹⁰⁵ reported that the HULIS-C concentrations during a strong haze event ($17.5 \mu\text{g m}^{-3}$) at Bangi, Malaysia were higher than those during a non-haze period ($1.3 -$

$4.4 \mu\text{g m}^{-3}$). In contrast to some of the above species such as OC and WSOC, the elemental carbon (EC) and black carbon (BC) concentrations showed no significant increases during haze events in Malaysia^{2, 50, 55, 56, 105}, Singapore^{25, 43, 89, 91, 107} or Thailand^{120, 122}. IPFs are smoldering fires undergoing a slow, low-temperature, flameless form of burning⁴⁹, which produces much smaller EC (BC) emissions than OC emissions, as seen in the laboratory experiments in the dataset. Therefore, IPF-derived EC contributes less to non-haze levels of EC at receptor sites according to dilution during transportation. The chemical speciation of brown carbon (BrC) constituents is described in Section S4.3 of the Supplementary Information.

Based on laboratory chamber combustion experiments using Indonesian or Malaysian peat, the EFs of OC, WSOC and EC were determined as 6.02–18.5, 3.1–3.84, and 0.04–0.57 g kg-fuel^{-1} , respectively^{17, 26, 90, 98}. Iinuma *et al.*²⁶ also reported EFs of water-insoluble OC (WIOC) as 4.9 g kg-fuel^{-1} . Stockwell *et al.*⁶³ and Jayarathne *et al.*⁷⁷ presented EFs of OC and EC based on peatland fire source sampling in Central Kalimantan, Indonesia. Their EFs of OC ($12.4-16.0 \text{ g kg-fuel}^{-1}$) and EC ($0.24 \text{ g kg-fuel}^{-1}$) were in the range of those based on peat burning experiments^{17, 26, 90, 98}. Chow *et al.*⁸³ and Watson *et al.*⁹⁰, respectively, reported the species abundances in $\text{PM}_{2.5}$ mass and EFs for carbonaceous components (OC, EC and WSOC) based on aerosol data from laboratory chamber combustion of fresh and aged Malaysian peat, separated by two- and seven-day photochemical aging times simulated with an oxidation flow reactor (below, “OFR”). The EF of OC for fresh particles was 17.9–18.2 g kg-fuel^{-1} and that for aged particles decreased to 14.0–14.7 g kg-fuel^{-1} ⁹⁰. In particular, reduction of OC1 (the lowest temperature OC fraction obtained from thermal-optical method, see the “Appendix” sheet in the dataset.) was revealed, due to the loss of high-vapor-pressure semi-volatile organic compounds upon aging. In contrast, the EF of WSOC, as well as mass percentage of WSOC in $\text{PM}_{2.5}$ ⁸³, increased after oxidation in the OFR⁹⁰. Based on the field observations in Malaysia, Fujii *et al.*¹⁰⁶ mentioned a high possibility of secondary WSOC formation during transport from the IPF source to receptor sites.

3.2 Water-Soluble Ions

During haze events induced by IPFs, a significant increase in the concentrations of sulfate (SO_4^{2-}) and ammonium (NH_4^+) was typically observed at receptor sites. For example, significant differences in the total concentrations of water-soluble ions were observed between haze ($33 \pm 6.5 \mu\text{g m}^{-3}$) and non-haze ($5.5 \pm 2.6 \mu\text{g m}^{-3}$) periods in Pekanbaru, Indonesia⁸⁵. They then showed the average concentrations of SO_4^{2-} and NH_4^+ during haze events to have been 7.3 and 7.8 times higher than those during non-haze periods. Highly elevated

concentrations of SO_4^{2-} ($20.4 \pm 18.4 \mu\text{g m}^{-3}$) and NH_4^+ ($7.16 \pm 6.66 \mu\text{g m}^{-3}$) were observed during haze events compared to those during pre-haze periods (SO_4^{2-} : $2.24 \pm 1.22 \mu\text{g m}^{-3}$, NH_4^+ : $0.41 \pm 0.35 \mu\text{g m}^{-3}$) and post-haze periods (SO_4^{2-} : $1.79 \pm 0.98 \mu\text{g m}^{-3}$, NH_4^+ : $0.16 \pm 0.12 \mu\text{g m}^{-3}$) in Kuala Lumpur, Malaysia⁷¹. Similar results were also obtained at other locations in Malaysia such as Petaling Jaya⁵⁵ and Bangi⁵⁶. In Singapore, significantly higher concentrations of SO_4^{2-} ($15 \pm 3 \mu\text{g m}^{-3}$) and NH_4^+ ($6 \pm 2 \mu\text{g m}^{-3}$) were observed¹⁰⁷ in smoke-dominant samples compared to those in non-smoke dominant samples (SO_4^{2-} : $4 \pm 4 \mu\text{g m}^{-3}$, NH_4^+ : $1.6 \pm 1.3 \mu\text{g m}^{-3}$). At other locations in Singapore, similar results were also obtained^{16, 25, 30, 41, 43, 48, 53, 89}. In Thailand, Chaisongkaew *et al.*¹²² reported that the concentrations of SO_4^{2-} and NH_4^+ during non-haze periods, which represented clean background air, at Thepha beach were 0.658 and $0.198 \mu\text{g m}^{-3}$, respectively, whereas during a haze period, those species' concentrations increased (SO_4^{2-} : $2.168 \mu\text{g m}^{-3}$ (partial haze) and $2.248 \mu\text{g m}^{-3}$ (strong haze); NH_4^+ : $0.825 \mu\text{g m}^{-3}$ (partial haze) and $0.834 \mu\text{g m}^{-3}$ (strong haze)). Similar results were also obtained in Hat Yai, Thailand¹²⁰.

In general, potassium (K^+) has been used as a biomass burning indicator (Chuang *et al.*, 2013). Thus, high K^+ concentrations during IPF-induced haze events are expected at the receptor sites. In most cases of haze induced by IPFs, however, partial overlapping of the variation ranges of K^+ concentrations were observed between samples collected during haze and non-haze periods as shown in the Supplementary Information (the "Water-soluble Ions" sheet in the dataset and Fig. S1). Fujii *et al.*⁴⁹ mentioned K^+ was not appropriate as an indicator of IPFs due to an extremely low K^+ fraction at source (0.0423 ± 0.0400 wt.% of $\text{PM}_{2.5}$), and a lower fraction of 0.026 wt.% was obtained at a site near another fire source⁷⁷, as shown in Fig.S2 in the Supplementary Information.

Based on laboratory chamber combustion experiments on Indonesian or Malaysian peat, chloride (Cl^-), nitrate (NO_3^-) and SO_4^{2-} were determined as the main ions in most cases^{26, 36, 83, 90}. The EFs of Cl^- , NO_3^- and SO_4^{2-} for fresh particles were reported as 25.5 – 48 , 18.6 – 28.6 and 15 – 37.2 mg kg-fuel^{-1} , respectively^{26, 90}, which were much smaller than the EFs of OC ($6,020$ – $18,500$ mg kg-fuel^{-1}) and WSOC ($3,100$ – $3,840$ mg kg-fuel^{-1}) (see Section 3.1). Although NH_4^+ was one of the main water-soluble ions in PM collected at receptor sites during haze periods, the EF of NH_4^+ based on laboratory chamber combustion experiments^{26, 90} only came to 0.56 – 0.608 mg kg-fuel^{-1} . Chow *et al.*⁸³ and Watson *et al.*⁹⁰, respectively, reported species abundances in $\text{PM}_{2.5}$ mass and EFs for water-soluble ions based on fresh and aged aerosol data from laboratory chamber combustion of Malaysian peat, separated by two- and seven-day photochemical aging times simulated with an OFR. The EFs of NO_3^- and SO_4^{2-} for fresh particles were 18.6 ± 6.63

and 37.2 ± 3.18 mg kg-fuel^{-1} , respectively, and those for particles after two days of aging increased significantly to 171 ± 14.0 and 105 ± 6.58 mg kg-fuel^{-1} , respectively. After seven days of aging, further increases in the EFs of NO_3^- (1200 ± 365 mg kg-fuel^{-1}) and SO_4^{2-} (502 ± 32.2 mg kg-fuel^{-1}) were revealed. Similar patterns were also observed for NH_4^+ (EF: 0.411 – 0.608 mg kg-fuel^{-1} in fresh particles, 159 ± 34.7 mg kg-fuel^{-1} after two days aging and $1,210 \pm 219$ mg kg-fuel^{-1} after seven days aging) and nitrite (NO_2^-) (EF: ~ 0.00 mg kg-fuel^{-1} in fresh particles, 0.148 ± 5.48 mg kg-fuel^{-1} after two days aging and 3.89 ± 7.07 mg kg-fuel^{-1} after seven days aging). In contrast to the above ions, the EF of Cl^- in fresh particles was 25.5 – 34.9 mg kg-fuel^{-1} and that in aged particles decreased to 19.1 – 26.6 mg kg-fuel^{-1} .

3.3 Elemental Components

There are many reports on elements in PM at receptor sites (especially in Singapore) during periods of IPF-induced haze. The number of detected elements in those reports, however, ranged from 3 to 48, showing considerable differences among reports (see Supplementary Information ("Elements" sheet in the dataset)). Based on field observations in Singapore, significant increases in the concentration of $\text{Al}^{25, 37, 42, 43, 58}$, $\text{K}^{9, 25, 42, 58}$, $\text{Cr}^{37, 47, 58}$, $\text{Mn}^{37, 47, 58}$, $\text{Fe}^{25, 37, 42, 43, 58}$, $\text{Co}^{37, 47, 58}$, $\text{Cu}^{25, 37, 43, 58}$, $\text{Zn}^{25, 37, 58}$, $\text{Cd}^{37, 47, 58}$ and $\text{Pb}^{37, 58, 118}$ were reported by several researchers. Although some researchers also reported higher concentrations of other elements such as $\text{Ca}^{25, 42}$ and $\text{Ni}^{37, 58}$ during haze events in Singapore, the number of such reports is quite limited. In Kuala Lumpur, Malaysia, Sulong *et al.*⁷¹ analyzed 18 elements in $\text{PM}_{2.5}$ during pre-haze, haze and post-haze periods. They then showed a significant increase of K during haze (604 ± 301 ng m^{-3}) compared to pre-haze (208 ± 118 ng m^{-3}) and post-haze (195 ± 73.7 ng m^{-3}) periods. In Hat Yai, Thailand, Promsiri *et al.*¹²⁰ analyzed 11 elements in $\text{PM}_{2.5}$ from transboundary haze samples (only two samples) and background samples. There, the highest concentrations of heavy metals (Cd, Co, Cr, Mn, Ni and Pb) were found during haze events, but neither As nor Cd were detected. They also mentioned K was about three times higher than background levels during the haze events. A similar pattern with K was also observed at Thepha Beach, Songkhla Community College, the Wang Yai Sub-district and Pattani City in Thailand¹²². At an IPF source, Betha *et al.*³⁴ showed higher concentrations of Al, Ti, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, Cd and Pb in IPF samples than in background air samples collected at the same sampling point in the absence of IPFs.

The EFs of several elements based on laboratory chamber combustion experiments of Malaysian peat were reported by Watson *et al.*⁹⁰. They showed, for example, that the EFs of Al, K and Fe in fresh $\text{PM}_{2.5}$ were 8.79 – 9.94 , 6.13 – 10.4 and 10.3 – 17.0 mg kg-fuel^{-1} , respectively. Detailed information on other elements is provided in the

Supplementary Information (“Elements” sheet in the dataset). In other experiments, Othman and Latif³⁶ analyzed eight elements (Cd, Cu, Zn, Fe, Al, Pb, Cr and Ni) in PM₁₀ from laboratory chamber combustion of Malaysian peat soil, and indicated that the dominant element was found to be Zn, followed by Al and Fe. Das *et al.*⁸⁴ analyzed 14 elements (Mg, Al, Ca, V, Cr, Fe, Co, Ni, Cu, Zn, Se, Sr, Sn and Pb) in PM from Indonesian peat burning by controlled laboratory experiment. They showed the emission ratios (mass fraction in PM) of Ca ($516 \pm 247 \mu\text{g g-PM}^{-1}$) to be highest, followed by those of crustal elements Fe ($189 \pm 117 \mu\text{g g-PM}^{-1}$) and Al ($34 \pm 18 \mu\text{g g-PM}^{-1}$). Major elemental concentrations and mass fractions in PM are shown in Figs.S3 and S4 in the Supplementary Information.

According to the dataset, the Al/Pb ratios of PM_{2.5} collected during IPF-derived haze periods at receptor sites and at a fire site were mostly much lower (3–50) than the Al/Pb ratios of TSP¹¹⁸ during haze (4,721) and non-haze (669) periods, and the ratios of the fresh laboratory-generated particles from Indonesian and Malaysian peat combustion ranged from 11 to 98 in the dataset. The discrepancy in the Al/Pb ratios between the above TSP samples and fine particles in the dataset can be explained: Because of uplifting and suspension of surface soil particles during wildfires (wind erosion), mineral dust is mainly confined in a coarse mode, while biomass burning-derived mineral particles are smaller (in a fine mode) with different average bulk compositions (Jahn *et al.*, 2021).

Atmospheric deposition is an important nutrient source for freshwater and aquatic ecosystems, and excessive loading of nutrients accelerates eutrophication (Swackhamer *et al.*, 2004). The chemical composition of nutrients (total N and P, organic N and P, and ions) in TSP (and rainwater) was determined during IPF-derived hazy and non-hazy days to quantify their dry (and wet) deposition fluxes to the coastal water of Singapore³⁰. The average concentrations of nutrients (N and P species) increased by a factor of three to eight on hazy days when compared with non-hazy days (total-N: $13 \mu\text{g m}^{-3}$, total-P: $0.5 \mu\text{g m}^{-3}$)³⁰. This finding reflects a remarkable elevation of total PM_{2.5} mass concentration on hazy days (24 to $113 \mu\text{g m}^{-3}$) because there is little difference in their PM_{2.5} mass percentages between hazy and non-hazy days. Both of the estimated mean dry atmospheric fluxes of total N and P during the hazy days were about tenfold higher compared to the non-hazy days.

3.4 Organic Compounds

In this section, the target organic compounds are analyzed by widely-used offline (conventional) analysis methods such as gas chromatography-mass spectrometry (GC-MS). The results analyzed by online aerosol mass spectrometer (AMS) are presented in Section 4. There are several reports of organic compounds aside from PAHs in

PM during IPF-induced haze events in Indonesia^{28, 89}, Malaysia^{2, 20, 21, 50, 56, 64, 76, 88, 105} and Singapore^{7, 16, 25, 27, 29, 40, 43, 65, 89, 107}. The target compounds, however, have differed considerably between researchers. More data on levoglucosan (LG) are available compared to other compounds. LG is a typical cellulose pyrolysis product and has been widely used as a useful biomass burning marker (Simoneit *et al.*, 1999). In Indonesia, Tham *et al.*⁸⁹ conducted PM_{2.5} sampling near IPF sources (approximately 2 km downwind from IPF sites) in Jambi, Sumatra Island, and a high LG concentration ($840 \pm 700 \text{ ng m}^{-3}$) in PM_{2.5} was observed. In Bangi, Malaysia, Fujii *et al.*⁵⁶ showed significant differences in LG concentrations in TSP between haze (strong haze: $1,078 \text{ ng m}^{-3}$, light haze: $352 \pm 22.9 \text{ ng m}^{-3}$) and non-haze ($207 \pm 39.3 \text{ ng m}^{-3}$) samples. Similar results have also been obtained at other locations in Malaysia such as Kuala Lumpur^{20, 88} and Petaling Jaya⁵⁰. In Singapore, Engling *et al.*⁴³ reported that the LG concentrations in TSP during haze events ($1150.9 \pm 917.2 \text{ ng m}^{-3}$) were much higher than those during the non-haze periods ($15.3 \pm 5.7 \text{ ng m}^{-3}$). Similar results were also obtained in Singapore^{29, 40, 89, 107}. Aside from LG, Fujii *et al.*⁵⁰ quantified many organic compounds derived from biomass burning and *n*-alkanes through annual observations of PM_{2.5} in Petaling Jaya, Malaysia, including IPF-induced haze periods. They then showed significant increases in the concentrations of mannosan (MN), galactosan (GL), vanillin (V), vanillic acid (VA), *p*-hydroxybenzoic acid and *n*-alkanes with carbon numbers from 23 (C₂₃) to 33 (C₃₃) for PM_{2.5} affected by IPFs at the receptor site. For *p*-hydroxybenzoic acid, through intensive observations of TSP in Bangi, Malaysia, Fujii *et al.*⁵⁶ reported even during light-haze periods, the *p*-hydroxybenzoic acid concentrations were significantly higher than those during non-haze periods, showing it to be a more useful indicator for IPFs compared to LG, MN and GL. Regarding *n*-alkanes, through comparison of molecular distributions of particulate *n*-alkanes obtained from IPFs and vehicle-related emission sources, Fujii *et al.*⁴⁹ showed particularly that C₂₇, C₂₈ and C₂₉ mass fractions in particulate total *n*-alkanes (C₂₀-C₃₃) of IPF-source samples collected in Riau, Sumatra, Indonesia were 4.57–26.6 times higher than those of other emissions. Jayarathne *et al.*⁷⁷ also agreed that *n*-alkanes could be used to distinguish IPF emissions from other types of biomass burning or other combustion sources.

The EF of LG was reported as 4.21–2,500 mg kg-fuel⁻¹ based on laboratory chamber combustion experiments of Indonesian²⁶ or Malaysian peat^{90, 98}. The EFs of other organic compounds such as MN^{26, 90}, GL²⁶, lignin pyrolysis products^{26, 98} and *n*-alkanes^{26, 98} are also provided in the Supplementary Information (“Organic Compounds” sheet in the dataset). Chow *et al.*⁸³ and Watson *et al.*⁹⁰, respectively, reported the species abundances in PM_{2.5} mass and EFs of several organic

compounds based on fresh and aged aerosol data from laboratory chamber combustion of Malaysian peat, separated by two- and seven-day photochemical aging times simulated with an OFR. Although there was no significant difference in the weight percentage of LG in the PM mass between fresh ($2.25 \pm 0.016\%$) and two-day aged ($2.28 \pm 0.99\%$) samples, such a difference was observed⁸³ between fresh ($4.38 \pm 0.50\%$) and seven-day aged ($2.53 \pm 0.19\%$) samples. For formic acid and oxalic acid, Chow *et al.*⁸³ reported the weight percentages of formic acid and oxalic acid in the PM mass of fresh particles were 0.10 ± 0.014 and $0.26 \pm 0.12\%$, respectively, and those after two days aging increased to 0.26 ± 0.049 and $1.14 \pm 0.21\%$, respectively. After seven days aging, further increases in the weight percentage of formic acid ($0.42 \pm 0.10\%$) and oxalic acid ($3.36 \pm 0.28\%$) were revealed, indicating secondary organic aerosol (SOA) formation related to these compounds.

3.5 PAHs

Some studies^{5, 29} have measured the concentrations of PAHs in the gaseous and particulate phases, but not all phase concentrations were available, so the data were excluded. Saha *et al.*⁷⁰ analyzed 36 PAHs, but in most cases, the concentrations of the US EPA 16 PAHs (see the “Appendix” sheet in the dataset) in PM during IPF-derived haze periods have been analyzed in field measurements. The concentrations of all 16 PAHs in PM_{2.5} (hereafter, “total PAHs level”) at receptor sites ranged from (a) 0.15 ng m^{-3} in Thailand¹²⁰ to (b) 61 ng m^{-3} in Singapore (severe haze in 2010)³⁷. However, the concentration rose to (c) 562 ng m^{-3} 100 m away from the boundary of a peat fire in Riau Province of Sumatra, Indonesia²⁸. The mass fractions of the total of the 16 PAHs in PM corresponding to cases (a), (b) and (c) were 30, 570 and 970 ppm ($\mu\text{g g-PM}_{2.5}^{-1}$), respectively. The differences in mass fraction suggest differences in haze intensity, as well as dilution and transformation, during transportation. The concentration varied significantly, depending on IPF intensity, distance from the emission source to the receptor site, meteorological conditions and local sources.

The total PAHs concentration increased during haze periods as compared with during non-haze periods. For example, the total PAHs level at Kuala Lumpur in Malaysia⁸⁸ was 3.40 ng m^{-3} during the 2015 haze period, while the annual average concentration (including the haze period) was 2.04 ng m^{-3} . In Singapore³⁷, the total PAHs level increased from 24 ng m^{-3} during a non-haze period to 61 ng m^{-3} during a severe haze event in 2010. In Thailand, the average of total PAHs levels for four sites was 0.20 ng m^{-3} during the 2019 haze period and 0.13 ng m^{-3} in the non-haze period¹²² (Fig. S5 in the Supplementary Information). High molecular weight PAHs predominated in the haze period^{24, 88, 92, 119, 122}. PAH congeners that increased in concentration during the haze

period were Acy, Ace, Flu, Phe, Flt, Pyr, BaA, Chr, BbF, BaP, Ind, BgP, and DhA (these abbreviations of congeners are spelled out in the “Appendix” sheet of the dataset) for Indonesian samples²⁸. In Singapore³⁷, an increase in congener concentrations was found for 13 congeners (Nap, Flu, Phe, Ant, Pyr, BaA, Chr, BbF, BkF, BaP, Ind, BgP and DhA) during the 2010 haze period. Among them, Pyr, Chr, BbF, BkF, BaP, InD and DBA levels were significantly higher than usual³⁷. In Kuala Lumpur⁸⁸, the dominant PAHs in PM_{2.5} were BbF, BkF, BaP, Ind and BgP during the 2015 haze event. In Thailand¹¹⁹, BkF, Act, DhA, BbF were the dominant congeners in the 2019 strong haze. Although the total PAHs concentration was specified, there were limited concentration data available on individual congeners (graphically depicted data only, or not indicated). Therefore, the difference in species between sampling sites was not significant.

In laboratory experiments, the EF of total PAHs in PM₁₀ from natural combustion of Indonesian peat was $20 \text{ mg kg-fuel}^{-1}$, and the dominant congeners were Flu ($11 \text{ mg kg-fuel}^{-1}$), Phe (2.7), Pyr (1.5), Ant (1.1) and Flt (1.1)²⁶. Sengupta *et al.*¹²¹ quantified the gas and PM EFs of 133 PAHs and their derivatives from smoldering combustion of Malaysian peat and Malaysian agricultural peat (and four other fuels), and their chemical transformations during aging in an OFR. The EF of all 113 PAHs in fresh particles was $1.80 \mu\text{g kg-fuel}^{-1}$ and that in aged particles was reduced to $0.25 \mu\text{g kg-fuel}^{-1}$ for Malaysian peat. For fresh particles from the combustion of Malaysian peat, the dominant congeners were Phe (88.6 ng g^{-1}), Flt (47.5) and BgF (20.8). In aged particles, the dominant congeners were Ace (20.2 ng g^{-1}), Flt (20.1) and Phe (14.8). After oxidation of Malaysian peat combustion emissions in the OFR, the EF of three-ring particle-phase PAHs decreased by a factor of nearly eight¹²¹.

PAH derivatives constitute a class of organic compounds derived from parent PAH structures with a variety of functional groups or elements. Typically they are nitrated (nitro) PAHs (NPAHs), oxygenated PAHs (OPAHs), halogenated PAHs (XPAHs) or alkylated PAHs (APAHs) (Peng *et al.*, 2023). In field measurements, only OPAHs were quantified, and the concentrations of the congeners were in the range of 0.03 to 4.46 ng m^{-3} during the 2006 haze period in Singapore²⁹, but the data were the sum of the gas- and TSP-phase OPAHs. Particulate OPAHs were analyzed in the two laboratory experiments mentioned in the above paragraph^{26, 121}. No OPAHs were detected in the experiment by Sengupta *et al.*¹²¹, but Iinuma *et al.*²⁶ quantified the EF of 9-fluorenone (0.75 ng g^{-1}) and 9,10-anthracenedione (0.29). In our survey, no other derivatives aside from OPAHs were found.

Diagnostic ratios were used for source apportionment of ambient PAHs in PM_{2.5} collected in Bangi, Malaysia during the southwest monsoon (including haze) in 2013⁵¹,

PM_{2.5} collected in Kuala Lumpur during different monsoon seasons⁸⁸, and PM_{0.1}, PM₁ and PM_{2.5} collected in southern Thailand during normal, partial and strong haze periods in 2019¹¹⁹. Bivariate plots of Flt/(Flt+Pyr) versus Ind/(Ind+Bgp) indicated that the major sources of PM during strong haze periods were predominantly from biomass burning in the case of Thailand¹¹⁹. Two other studies indicated different sources without specifying a definite source of the haze. Figure S6 in the Supplementary Information shows three bivariate plots of PAH ratios for haze and non-haze periods using our dataset. In Fig. S6 (A), the three blue circles in the upper right region (biomass burning region) are clearly distinguished from the other data. The data are from samples collected at three sites in Riau, Indonesia during the 2015 fire episode²⁸, and the sampling sites were 100 m, 60 km and 220 km away from the fire source. However, no definite sources of haze were identified for the other data. Moreover, contributions from local sources can dominate the contribution of IPF-derived haze¹⁵. Unfortunately, the concentration of each PAH congener is either indicated only as graphical data (Thailand data¹¹⁹ mentioned above) or not available in some reports. More quantitative data may improve source identification of IPF-derived haze.

4. Aerosol Mass Spectra Analysis for Chemical Characterization

A time-of-flight aerosol chemical speciation monitor (ToF-ACSM) provides continuous online size and non-refractory (NR) chemical composition data on individual sub-micron aerosol particles without filter sampling (Fröhlich *et al.*, 2013). It produces ensemble-average mass spectra for organic and inorganic species, and organic aerosol (OA) mass spectra can be deconvolved in combination with different statistical techniques to estimate sources, composition and atmospheric processing of OAs (e.g., Chen *et al.*, 2020). In recent years, a research group based at Nanyang Technological University (hereafter “the Singapore group”) used ToF-ACSM to characterize the physical and chemical properties of organic aerosols during the 2015 haze episode in Singapore^{65, 73, 74}. However, there are no field studies of IPF aerosol using ToF-ACSM in other ASEAN countries or at peatland fire sites in our dataset. Laboratory experiments are a powerful approach for determining in-depth chemical compositions of fresh and aged peat-burning aerosols and their influencing factors using ToF-ACSM under controlled environmental conditions. Laboratory experiments on aerosols from Indonesian-peat and other biomass combustion have been investigated by the Singapore group^{65,66,69,81,82,86,102,109,112} using a flow-type reactor with ToF-ACSM, and by Ahern *et al.*⁸¹ using batch-type dual chambers with an Aerodyne high-resolution AMS and a proton transfer reaction mass

spectrometer (PTR-MS) for volatile organic compound measurement.

Detailed analyses of chemical compositions, secondary aerosol formation and the relationship between chemical compositions and physical properties are provided with additional documents in the Supplementary Information.

5. Source Apportionment of PM during Haze Episodes

5.1 Key Indicators of Peatland Fire Aerosols

Several indicators for IPFs have been suggested from the diagnostic ratios of chemical species based on field observation data (source and/or receptor sites) and laboratory burning experiments. OC/EC, WSOC/OC and HULIS-C/WSOC in PM provide a rough indication of origins or characteristics of carbonaceous aerosols (Cao *et al.*, 2005; Frka *et al.*, 2018; Shen *et al.*, 2014). The OC/EC obtained from source-dominated field observations^{28, 44, 63, 77, 106} and laboratory burning experiments^{17, 26, 66, 83, 90, 95} were 2.42–223 and 14–163, respectively. The discrepancy among these data can be attributed to differences in combustion conditions such as fuel load, moisture content (Chen *et al.*, 2010), smoldering kinetics and the reaction-zone structure of the IPF (Huang and Rein, 2014). In addition, the use of different protocols such as NIOSH and IMPROVE for OC/EC determination may also lead to uncertainties (Wu *et al.*, 2016). Thus, to show the representative OC/EC for an IPF is extremely difficult, as mentioned in Fujii *et al.*¹⁰⁶. At the receptor sites, obvious increases in OC/EC during haze periods were consistently observed compared to during non-haze periods in Malaysia^{2, 50, 56}, Singapore^{16, 25, 43, 89} and Thailand¹²⁰. WSOC/OC obtained from source-dominated field observations^{77,106} and laboratory burning experiments^{23, 26, 66, 83, 90} was 0.085–0.16 and 0.0093–0.87, respectively. WSOC/OC obtained from certain laboratory burning experiments^{23, 26} differed significantly from those obtained through field observations. These laboratory burning systems may not accurately represent real-world IPFs. Fujii *et al.*¹⁰⁵ showed WSOC/OC in TSP collected in Bangi, Malaysia during a haze period (strong haze: 0.34, light-haze: 0.39 ± 0.034) was slightly lower than that during a non-haze period (0.41 ± 0.064). Based on these data, $\Delta(\text{WSOC}/\text{OC})$ was calculated by subtracting the average concentrations of OC and WSOC during the non-haze period from those of the haze period¹⁰⁶. The resulting $\Delta(\text{WSOC}/\text{OC})$ was 0.36 ± 0.055, which is higher than the WSOC/OC obtained at the IPF source^{77, 106}, suggesting secondary WSOC formation during transport from IPF sources to receptor sites. As an indicator, WSOC/OC is not useful for IPFs. Regarding HULIS-C/WSOC, only one set of field observation data is available¹⁰⁵. Regarding that report, although HULIS-C/WSOC during the haze period in

Bangi, Malaysia tended to be higher than that during a non-haze period, some exceptions to this trend were observed.

Employing data on carbon fractions following the IMPROVE_A protocol, OP/OC4 in TSP and $(OC2 + OC3 + OP)/(EC2 + EC3)$ in PM_{2.5} were suggested as indicators of IPFs by Fujii *et al.*⁵⁶ and Tham *et al.*⁸⁹, respectively. OP indicates pyrolyzed OC. Even for PM_{2.5} samples collected in Malaysia, Fujii *et al.*^{50, 67} confirmed the effectiveness of OP/OC4 as an indicator for IPFs. However, in Singapore, there were no significant differences in OP/OC4 between IPF-dominant smoke (2.1 ± 1.2) and non-smoke-dominant samples (2.3 ± 1.6). We calculated $(OC2 + OC3 + OP)/(EC2 + EC3)$ by using data on carbon fractions in TSP collected in Bangi, Malaysia during strong haze, light haze and non-haze periods⁵⁶. The results showed significant differences between haze (strong haze: 9.07 ± 0.29 , light haze: 3.72 ± 0.61) and non-haze samples (2.04 ± 0.69) even in Malaysia. The usefulness of this indicator is demonstrated at source and receptor sites in the dataset. Although further validation of this indicator is needed, $(OC2 + OC3 + OP)/(EC2 + EC3)$ seems to be useful compared to OP/OC4.

MN/LG has been used in the source assignment of specific biofuels (e.g., Alves *et al.*, 2010). Based on a report by Fabbri *et al.* (2009), differences in MN/LG between hardwood burning (0.042–0.077) and softwood burning (0.15–0.26) can be shown¹⁰⁶. MN/LG in PM at IPF sources^{44, 106} ranged widely from 0.021 to 0.095. Whether obtained from laboratory burning of Indonesian²⁶ or Malaysian peat^{83, 90}, these ratios were less than 0.088. Thus, IPFs can be roughly categorized as hardwood burning. Based on data on LG and MN in TSP collected in Bangi, Malaysia during haze and non-haze periods reported by Fujii *et al.*⁵⁶, LG/MN was calculated. The results showed MN/LG during strong haze, light haze and non-haze periods to be 0.12, 0.113 ± 0.015 and 0.108 ± 0.014 , respectively, showing no obvious differences between haze and non-haze samples. Other IPF indicators have been suggested (e.g., SA/LG¹⁰⁶ and SA/VA⁴⁹) based on data taken at IPF sources. The effectiveness of these indicators, however, has not been fully verified at receptor sites. The above-mentioned indicators are summarized in Tables S6 and S7 in the Supplementary Information, based on the dataset.

5.2 Isotopic Compositions as a Source Signature

Some studies have been conducted to investigate the isotopic composition of haze from IPFs as a potential signature of these fires^{7, 15, 62, 80} and wind-blown mineral dust particles during IPF events¹¹⁸.

For PM_{2.1} samples collected in Singapore during the 1997 IPF and in Indonesia after the major fires had died out, the carbon isotopic ratio of TC ($\delta^{13}C_{TC}$, defined in the “Appendix” sheet in the dataset) showed a decrease from about -25.5% to -27.5% with an increase in TC

concentration ($9-81 \mu\text{gC m}^{-3}$)⁷. The results suggest an addition of OA from the combustion of C3 plants whose $\delta^{13}C$ is lower. The $\delta^{13}C$ of individual PAHs in the haze (TSP) in Malaysia from the 1997 IPF ranged from -17.7 to -27.9% ¹⁵. The isotopic ratios were similar to those of soot PAHs extracted from gasoline and diesel vehicles; therefore the $\delta^{13}C$ of the individual PAHs in the haze cannot serve as a signature of IPFs in Malaysian urban air polluted by automobile exhaust.

²⁰⁹Po and ²¹⁰Po activities were determined for three extracted fractions of Fe and Mn oxides, organic matter and residuals in PM₁₀ (marine aerosol) sampled for two years along the east coast of Malaysia⁶². During the southwest monsoon period when strong transboundary haze affects this area, a significant amount of excess ²¹⁰Po in fractions bound to organic matter ($119 \pm 85 \mu\text{Bq m}^{-3}$, cf. 54 ± 55 of ambient values), which are regulated by seasonal changes⁶², could be an IPF signature.

Wiggins *et al.*⁸⁰ estimated the age and origin of IPF-emitted PM_{2.5}, utilizing the radiocarbon content (¹⁴C) of carbonaceous PM_{2.5} collected in Singapore during haze and non-haze periods⁸⁰. A Keeling plot (Mouteva *et al.*, 2015) was used to separate urban background contributions, and the $\Delta^{14}C$ for the haze period was significantly lower than atmospheric levels ($-76 \pm 51\%$ vs. $25 \pm 3\%$), corresponding to a carbon pool of combusted organic matter with a mean turnover time of 800 ± 420 years estimated by a radiocarbon box model⁸⁰. Under an assumption about TC EF and peat $\Delta^{14}C$ profiles, a Monte Carlo analysis indicated that the smoke plumes reaching Singapore originated primarily from peat burning ($\sim 85\%$), and not from deforestation fires or waste burning⁸⁰.

Isotopic ratios of Pb (²⁰⁶Pb/²⁰⁷Pb and ²⁰⁸Pb/²⁰⁷Pb) in TSP samples collected during the IPF-derived haze period in Singapore in 2013 were 1.169 ± 0.012 and 2.448 ± 0.013 , respectively¹¹⁸. There was a statistically significant difference between the two ratios during haze and non-haze periods (1.159 ± 0.005 and 2.438 ± 0.006). The difference in the Pb isotope ratios could be explained by mixing 10% – 70% of mineral dust (coarse) particles from suspension of crustal soil during fire events¹¹⁸. The contribution of such mineral particles to TSP during the haze period was supported by enhancement of the Al/Pb ratio (average: 4,721) comparable to the ratio in crustal material, with the average ratio during non-haze periods being 669¹¹⁸.

5.3 Application of Receptor Models

In addition to chemical characterization, quantitative knowledge of IPF source contribution to ambient PM in SEA provides valuable information to policy makers for mitigating IPF-haze pollution⁶⁷. In this section, the results of IPF source apportionment by chemical mass balance (CMB) and positive matrix factorization (PMF) models are described. Detailed information on CMB and PMF models, as well as other receptor models, are provided in

Seinfeld and Pandis (2016).

There are few reports regarding the results of PM source apportionment of IPFs by the CMB model in Indonesia²⁸, Singapore⁴³ or Thailand¹²⁰. In brief, See *et al.*²⁸ reported that IPFs contributed 51% and 18.1% of the PM_{2.5} mass during haze periods at receptor sites in Belakang Rumah and Pekanbaru, respectively, in Sumatra, Indonesia. In Singapore, Engling *et al.*⁴³ reported that IPFs were the predominant source (75.6% of TSP mass) during a haze event, followed by diesel exhaust (18.6%) and ship emissions (5.3%). In Thailand, Promsiri *et al.*¹²⁰ reported during a transboundary haze event, PM_{2.5} in Hat Yai was dominated by IPF sources (50% of the PM_{2.5} mass), followed by rubber wood burning (27%) and diesel combustion (4%).

Several sets of data on PM source apportionment by PMF model are available in Indonesia^{32,115} and Malaysia^{31, 51, 52, 55, 60, 61, 67, 71, 88, 108, 110, 111}. However, only Fujii *et al.*⁶⁷ have discussed IPF contribution clearly. They reported throughout their sampling periods (one year) that IPFs contributed, on average, 6.1–7.0 $\mu\text{g m}^{-3}$ to PM_{2.5} or ~30% of PM_{2.5} mass. In particular, PM_{2.5} was dominantly sourced from IPFs during the southwest monsoon season (51% – 55% of the PM_{2.5} mass on average). Other reports have mentioned potential source contributions of IPFs at receptor sites^{52, 55, 60, 61, 71, 88, 115}.

Information from studies on source contributions from CMB and PMF as well as principal component analyses are summarized in the Supplementary Information (“Source Apportionment” sheet in the dataset).

6. Size Distributions

As mentioned in Section 1, biomass burning aerosols, including from peatland fires, are mostly in the fine size range (Reid *et al.*, 2005). Therefore, the total mass and chemical compositions of IPF-derived aerosols have been widely determined for PM_{2.5} using a high volume air sampler or impactor as in Table S8. Simple size segregation such as coarse and fine PM sampling was conducted by dichotomous^{114, 115} or Gent stacked filter unit particle sampler^{9, 14, 31, 32, 52, 87, 97}. To obtain higher size-resolved (size distribution) mass and chemical compositions of IPF-derived haze, multi-stage cascade impactors such as an Andersen sampler⁹², MOUDI (see the “Appendix” sheet in the dataset)^{25, 47, 48}, Nano samplers^{92, 96, 100, 101, 111, 119} and others have been employed. A multi-stage impactor requires a longer sampling time, especially to determine reliable size distributions of trace elements in haze, and is time-consuming and labor-intensive. Recently, real-time characterization of organic aerosols in haze (NR-PM₁) has been conducted using an AMS as described in Section 4. There are not many, but real-time number size distributions have been measured by SMPS^{23, 25, 53, 74, 81, 102, 109}, OPC^{13, 74} or other

methods in field and laboratory experiments.

Details on the size distributions of total mass and chemical species are provided with additional documents in the Supplementary Information.

7. Conclusions

In this paper, we have conducted a holistic review of source profiles, key indicators of IPFs at source and receptor sites, and the transformation of haze during its long-range transport (aging, secondary aerosol formation) as well as source apportionment in SEA, using a comprehensive chemical component dataset. Knowledge of the chemical characteristics of PM at IPF source and receptor sites is still limited, especially regarding the controlling factors (e.g., burning conditions, peat composition and effects of vegetative burning on peatlands) that determine the IPF source profile of PM. Clarifying these factors can lead to more reliable and speciated emission inventories of IPFs, enabling both chemical transport and radiative forcing modeling and health risk assessments to advance. Additionally, the process of IPF-derived SOA formation during transport to receptor sites remains largely unresolved. Recently Lopez-Hilfiker *et al.* (2019) developed an extractive electrospray ionization time-of flight mass spectrometer (EESI-ToF) with generally insignificant decomposition of fragmentation products of OAs. Furthermore, Siemens *et al.* (2023) revealed aging characteristics of BBOA using multimodal MS techniques (combining AMS, EESI-ToF, offline HPLC-PDA-HRMS and tandem mass spectrometry). Measurements of aerosols emitted at peatland fire sites using cutting-edge techniques with precursors measurements and contributions of aqSOA to IPF aerosols would promote an understanding of the formation mechanism of SOA and OPOA, along with SOA source apportionment.

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Supplementary Information

Supplementary Information related to this article can be found on Zenodo (<https://doi.org/10.5281/zenodo.10699445>). Included are the newly created dataset and supplementary documents (for Sections 4 and 6), tables and figures.

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