sustainability

MDPI

Review

Airborne Nanoparticles ($PM_{0.1}$) in Southeast Asian Cities: A Review

Worradorn Phairuang [1,2,*], Muhammad Amin [1], Mitsuhiko Hata [1] and Masami Furuuchi [1,3]

[1] Faculty of Geosciences and Civil Engineering, Institute of Science and Engineering, Kanazawa University, Kanazawa 920-1192, Ishikawa, Japan
[2] Department of Geography, Faculty of Social Sciences, Chiang Mai University, Muang, Chiang Mai 50200, Thailand
[3] Faculty of Environmental Management, Prince of Songkla University, Hat Yai 90112, Thailand
* Correspondence: pworradorn@se.kanazawa-u.ac.jp or worradorn@gmail.com

Abstract: $PM_{0.1}$ (particles with a diameter $\leq$ 0.1 μm), nanoparticles (NPs), or ultrafine particles (UFPs) were interchangeably used in the scientific communities. $PM_{0.1}$ originated from both natural and human sources; however, $PM_{0.1}$ and its effects on the environment, visibility, and human health to understanding air pollution levels, sources, and impacts in Southeast Asia (SEA) countries continue to be challenging. The concentrations of $PM_{0.1}$ in most SEA countries are much worse than in western countries' environments. A further motivation of this reviewed article is to provide a critical synthesis of the current knowledge and study of ambient $PM_{0.1}$ in SEA cities. The primary influence of characteristics of $PM_{0.1}$ appears to be local sources, including biomass burning and motor vehicles. Continuous monitoring of $PM_{0.1}$ in mass and number concentration should be further understood. A critical review is of great importance to facilitating air pollution control policies and predicting the behavior of $PM_{0.1}$ in SEA.

Keywords: Asia; air pollution; biomass burning; health risk; motor vehicles; nanoparticles; $PM_{0.1}$; ultrafine particles

Citation: Phairuang, W.; Amin, M.; Hata, M.; Furuuchi, M. Airborne Nanoparticles ($PM_{0.1}$) in Southeast Asian Cities: A Review. *Sustainability* **2022**, *14*, 10074. https://doi.org/10.3390/su141610074

Academic Editors: Xinfeng Wang and Lan Yao

Received: 13 July 2022
Accepted: 12 August 2022
Published: 15 August 2022

1. Introduction

Particulate matters (PM) have a complex chemical composition, including acids (nitrates and sulfates), organic chemicals, and heavy metals. Some of these components are hazardous to human health [1–3]. Particularly, smaller particles down to the nano-size range mainly come from human sources and contain highly hazardous components such as heavy metals, carbon components, and polycyclic aromatic hydrocarbons (PAHs) [4,5]. PM has a wide range of particle sizes and can categorize into three modes: coarse particles ($PM_{10\text{-}2.5}$; aerodynamic diameter between 2.5 and 10 μm), fine particles ($PM_{2.5}$; predominantly in accumulation mode, aerodynamic diameter range from 0 to 2.5 μm), and ultrafine particles ($PM_{0.1}$; nucleation mode, aerodynamic diameter smaller than 0.1 μm, or 100 nm) [6]. $PM_{0.1}$, ultrafine particles (UFPs), and nanoparticles (NPs) are interchangeably used in scientific societies [5,7]. NPs are used to represent particles from engineering material released into the environment. Concurrently, other scientist groups use various names. For example, toxicologists typically use ultrafine, fine, and coarse modes, and monitoring organizations, i.e., the United States Environmental Protection Agency (US-EPA), use $PM_{0.1}$, $PM_{2.5}$, and PM_{10} to refer to the ambient PMs [8]. In the recent decade, it has been found that $PM_{0.1}$ poses the most risks to human health [9]; it is not enough information on the status of $PM_{0.1}$, and its emission sources remain incomplete because $PM_{0.1}$ is challenging to study due to its tiny size, high chemical reactivity, and rapid changes [10,11].

The data on physical, optical, and chemical characteristics of $PM_{0.1}$ are scarce worldwide, including in all Southeast Asian (SEA) countries, where in the past decades have been the dominant contributors of PMs into the atmosphere [12]. Several research publications

have investigated the airborne PM-bound chemicals in SEA due to ecological damage and health effects [13,14]. Atmospheric PMs in SEA countries are presently based on PM_{10} and $PM_{2.5}$ and a small expanse on PM_1 [15,16]. Quite a lot of studies based on satellite imaging, ground-based detecting, and mathematical modeling techniques exposed that aerosol plumes during smoke-haze exceeded the PM national standard of SEA, which has a very high value compared to the World Health Organization (WHO) or other countries' standard [13,14,17]. A recent study confirmed that $PM_{0.1}$ occupied 15–20% of the total suspended particulates (TSP) in the SEA atmosphere [18–21].

All these available data for $PM_{0.1}$ in western countries, i.e., Europe and the United States of America (USA), are more progressive [11,22]. Venecek et al. (2019) [3] reported that the $PM_{0.1}$ concentration across many cities in the USA is higher than 2 $\mu g/m^3$ during episodes of summer pollution. On the contrary, the yearly averaged $PM_{0.1}$ mass concentration is less than 1 $\mu g/m^3$. Furthermore, studies show that in the United Kingdom (UK), the $PM_{0.1}$ level based on the estimated fraction of PM_{10} in each emission source is as follows; production process, non-road transport, agriculture, industrial off-road mobile machinery, transformation industries, energy combustion, combustion in industries as well as waste incineration (15%, 14%, 9%, 9%, 8%, 7%, and 4%, respectively) [8]. The $PM_{0.1}$ level is typically measured in terms of particle number concentrations (PNC) due to the very low mass concentrations in all cities around the western part of the world [5,7,8]; however, the data for SEA city environments are still separate in detail. So, the motivation of this review is to recognize the recent data, sources, and knowledge gap in $PM_{0.1}$ emission and exposure levels. The authors discuss the current situation of $PM_{0.1}$ study in SEA to understand better and future perspective of ambient $PM_{0.1}$; this review collects the recent research papers on all aspects of $PM_{0.1}$ in SEA. Over 60 peer-reviewed journals in Scopus and ISI databases were used to analyze and integrate synthesis and group. The search terms and keywords included $PM_{0.1}$, biomass burning, ultrafine particles, haze pollution, health effects, emission inventory, and clean air policies. The synthesis-reviewed article prefers to use the recent publication in 5 years, from 2017 to 2021.

2. $PM_{0.1}$ Mass Concentration in Southeast Asia Atmosphere

The $PM_{0.1}$ fractions in the ambient PM have a very low particle mass concentration but a huge number of particles [7]. Most particles by numbers lie below 0.1 µm (100 nm), and they are in the $PM_{0.1}$ range; however, their concentration in terms of mass per volume is very low. No standards for ambient $PM_{0.1}$ have been adopted in Asian countries. The European Committee for standardization announced that the Condensation Particle Counter (CPC) is a standard protocol to measure UFPs [23]; however, only the emission standard for diesel and gasoline direct injection engine road vehicles must meet a type-approval of UFPs for non-volatile particles of >23 nm diameter (The Solid Particle Number >23 nm method; SPN23) [24]. The most widely used measure UFPs is particle number concentration (PNC) due to their tiny volume and mass concentration. The estimated concentration based on 10 $\mu g/m^3$ in $PM_{2.5}$ found that $PM_{2.5}$ 1 particle/cm^3 equal to $PM_{0.02}$ (<20 nm) 2.4×10^6 particle/cm^3, or PM_{10} 1 particle/cm^3 similar to UFPs 1.0×10^6 particle/cm^3 [25]. The particle number concentration and surface area are suitable for measuring UFPs' small mass concentration in the past decades [7].

The average $PM_{0.1}$ mass concentration in SEA has shown in the past decade (Table 1). The initial study of ambient $PM_{0.1}$ in SEA and published in an international peer-review journal is based on a survey in Thailand from 2014 to 2105 [20]. The $PM_{0.1}$ in Bangkok and Chiang Mai, Thailand, was 14.80 ± 1.99 and 25.21 ± 4.73 $\mu g/m^3$, respectively. Bangkok is the capital city in Thailand and one of the densest populated cities in SEA. The high episode of $PM_{0.1}$ in Chiang Mai arises from biomass fires in the dry season (February–April).

In Thailand, $PM_{0.1}$ in Pathumthani, in Bangkok Metropolitan Region areas, also elevated mass concentrations in wet and dry seasons [26]. In Hat Yai, southern Thailand [27], $PM_{0.1}$ was 10.17 ± 2.23 $\mu g/m^3$ representing PM concentrations lower than in other parts of Thailand; moreover, Zhao et al. (2016) [28] reported that they compared $PM_{0.1}$ during

the dry season (Mar-Apr 2016) in many cities, including Chiang Mai, Bangkok, Songkhla, Riau, Ho Chi Minh City, Phnom Penh, and Kuala Lumpur. The study found that mass concentration ($\mu g/m^3$) decreased in the order, Phnom Penh (18.9) > Chiang Mai (16.5) > Hanoi (15.4) > Ho Chi Minh City (13.1) > Riau (12.4) > North Bangkok (11.9) > Hat Yai (10.9) > Kuala Lumpur (9.3) > Bangkok (7.7).

The higher mass concentration in Upper SEA countries than in lower SEA countries due to the dry season during that period that the open biomass burning has been reported by several researchers [13,20,29]. PMs increased every dry season (February–April), and they started to rise around February and peaked in March before decreasing by mid-April [20]. The primary emission source of PMs worsening during the dry season in these areas was identified as open biomass burning, including forest fire and crop residue burning [20,26]. The above-mentioned corresponded with the accumulated fire hotspot locations that were high in the dry season and low in the wet season. Each active fire location represents the center, approximately decreasing by mid-of 1 km pixel flagged as one or more actively burning hotspots/fires [13]. Generally, $PM_{0.1}$ is derived from diesel exhaust, and it was sensitive to biomass burning in this area [20]. In addition, ambient $PM_{0.1}$ in Vietnam and Indonesia is very progressive. In Indonesia, the study by Amin et al. (2021a) [18] and Putri et al. (2021) [21] suggested that the $PM_{0.1}$ mass concentration in the dry season is higher than in the wet season in all monitoring sites. In addition, the mass concentrations by different monitoring sites have attention as follows, urban > suburban > rural. In Vietnam, many studies of $PM_{0.1}$ in Hanoi, the capital city in Vietnam, are more progressive than in other SEA cities [28,30–35]. The results show that mass concentrations in different environments in Hanoi ranged from 1 to 17 $\mu g/m^3$.

Table 1. Mass concentration of $PM_{0.1}$ at different locations in the Asian Environment.

Location	Site Description	Period	Concentration ($\mu g/m^3$)	Reference
Pathumtani, Thailand	Suburban	October 2019	13.47 ± 0.79	[26]
	Suburban	January–February 2020	18.88 ± 3.99	
Hat Yai, Thailand	Mixed	January–December 2018	10.17 ± 2.23	[27]
Hat Yai, Thailand	Mixed	March–April 2016	10.90	[28]
North Bangkok, Thailand	Urban-traffic	July 2014–June 2015	14.80 ± 1.99	[20]
North Bangkok, Thailand	Urban-traffic	March–April 2016	11.90	[28]
Bangkok, Thailand	Urban-traffic	March–April 2016	7.70	[28]
Chiang Mai, Thailand	Suburban	September 2014–June 2015	25.21 ± 4.73	[20]
Chiang Mai, Thailand	Suburban	March–April 2016	16.50	[28]
Riau, Indonesia	Urban	March–April 2016	12.40	[28]
North Sumatra, Indonesia	Urban-traffic	February 2019	13.10 ± 3.80	[21]
	Rural-volcano	March 2019	7.10 ± 2.50	
	Industry Area	February–March 2019	16.80 ± 4.00	
	School Environment	February 2019	15.90 ± 1.60	
Padang, Indonesia	Rural	March 2018	5.36	[18]
		August 2018	5.57	
Muaro Jambi, Indonesia	Suburban	March 2018	9.20	[18]
		August 2018	9.61	
Pekanbaru, Indonesia	Urban	March 2018	10.92	[18]
		August 2018	15.16	
Hanoi, Vietnam	Mixed	August–December 2015	5.36–5.79	[30]
	Urban-traffic	August–December 2015	6.06–11.90	
Hanoi, Vietnam	Mixed	November–December 2015	5.44 ± 2.03	[31]
Hanoi, Vietnam	Mixed	July–August 2015	1.47 ± 0.54	[32]
	Mixed	March 2016	1.71 ± 0.61	
Hanoi, Vietnam	Mixed	March–April 2016	15.40	[28]
Hanoi, Vietnam	Residential Area	January 2019	8.74	[33]
		April–May 2019	5.28	
Hanoi, Vietnam	Suburban 1 (Rice burning)	November 2019	6.50 ± 2.20	[34]
	Suburban 2 (Rice burning)	November 2019	11.50 ± 3.90	

Table 1. *Cont.*

Location	Site Description	Period	Concentration (μg/m^3)	Reference
Hanoi, Vietnam	School Environment	November 2019–January 2020	17.07 ± 3.70	[35]
	Urban-traffic	March–April 2016	13.10	[28]
Phnom Penh, Cambodia	Urban	March–April 2016	18.90	[28]
Kuala Lumpur, Malaysia	Suburban	March–April 2016	9.30	[28]

3. Sources and Characteristics of $PM_{0.1}$ in Southeast Asia

Regarding the natural sources, primary $PM_{0.1}$ is mainly generated by forest fires, while lesser fractions come from maritime aerosols and volcanic eruptions [5,19]. On the other hand, anthropogenic sources of $PM_{0.1}$ include transportation (on- and off-road automobiles, diesel engines, airplanes, and shipping), industrial combustion processes including biomass combustion and waste incineration, cigarette smoking, and meat cooking [7]. So, the primary emission sources of $PM_{0.1}$ are both natural and anthropogenic sources. Most studies of emission sources of $PM_{0.1}$ in Asia are related to road vehicles [36–38]. In an urban area, traffic is the primary source of $PM_{0.1}$ emissions. Diesel engines dominate $PM_{0.1}$ in megacities, including Shanghai, China [39], Hanoi, Vietnam [30], and Bangkok, Thailand [20]. Diesel engines have about two times higher emission factors than gasoline engines [40]. On the other hand, forest fires emit particles that dominate traffic emissions, and the size distribution peaks at approximately 120 nm for a fresh aerosol plume [9]; moreover, Phairuang et al. (2019) [20] stated that open biomass burning in central and northern Thailand dominated the release of carbon components into the atmosphere. The $PM_{0.1}$ particle, primarily derived from motor vehicle emissions, is also strongly affected by open biomass burning in the upper part of Thailand. Hence, this activity significantly affects air quality during the dry season. Similarly, an ambient $PM_{0.1}$ study in Hanoi, Vietnam, showed high mass concentrations of $PM_{0.1}$ during rice straw burning periods [30]; however, some possibly essential sources, such as domestic wood burning, are poorly quantified in SEA [41].

The chemical composition study in NPs is still limited, especially in ambient $PM_{0.1}$ particles; however, a few publications have been from NPs and chemical composition sources. For instance, polycyclic aromatic hydrocarbons (PAHs) dominate PM emissions [4,41]. Most PAHs emissions are generated from incomplete combustion of natural and human sources, including vehicle exhaust, biomass burning, industrial activities, and coal combustion. Hata et al. (2014) [4] studied the characteristics of NPs emitted from biomass fuels burning in Asia. The result demonstrated that approximately 30% of the biomass fuels burning smoke had a mass of <100 nm. Additionally, PM smaller than 0.43 μm significantly contributed to the toxicity of PAHs and the fraction of Water-soluble Organic Carbon (WSOC). Similarly, Chomanee et al. (2018) [42] considered the PAHs in smoke plumes released from the para-rubber fuel-wood combustion; this result displays that the ultrafine ($PM_{0.07}$) smoke particles comprised the highest number of PAHs and Benzo[a]pyrene-Toxic Equivalence Quotient (BaP-TEQ). The important fraction of NPs had the most considerable emission of toxicity per unit PM mass compared to fine and coarse PM; this is a critical point to concentrate on smaller particles, especially NPs, on any emission sources due to a lack of reliable information on the origins and magnitudes in SEA countries [20].

Secondary $PM_{0.1}$ aerosol is mainly generated from atmospheric photochemical of gaseous precursors and by condensation of semi-volatile vapors. Such new particle formation can occur during low relative humidity and wind speed at quiet pre-existing particle surface areas and global radiation [43]. Reche et al. (2011) [44] described that new particle formation in an urban area in Europe during warmer and sunny climates is essential to air pollution. In SEA, Thuy et al. (2018) [30] stated that Secondary Organic Carbon (SOC) is more dominant in smaller particles than in larger particles; and the SOC in $PM_{0.1}$ contributes up to 42.7% of the OC level in Hanoi, Vietnam. The secondary atmospheric $PM_{0.1}$ in the SEA environment remains poorly understood. Based on the chemical, optical, and

thermal properties of such samples, these carbon components can be broadly classified into two main fractions, i.e., organic carbon (OC) and elemental carbon (EC). The source of OC can be primary (POC) and secondary emissions (SOC), while EC is mainly emitted from primary sources [27,30]. The importance of carbonaceous particulate matter (OC and EC) in the Earth's climate system is becoming more broadly recognized due to its ability to induce climate changes by directly disrupting the Earth's radiative balance via the extinction of incoming solar radiation and indirectly by serving as cloud condensation nuclei (CCN). Carbonaceous particulate matter was also demonstrated to significantly influence air quality and human health [18–20].

4. Health Concerns of $PM_{0.1}$

$PM_{0.1}$ strongly believes in high toxic properties because a high surface area can absorb many poisonous substances. After they penetrate the human organ systems, allowing translocation and interactions to a human body organ and highly potential deeply into circular systems via respiratory mechanism [2]. The World Health Organization (WHO 2013) [17] suggested that the epidemiological data on $PM_{0.1}$ are too scarce to estimate or use in policy-making for air quality control management of $PM_{0.1}$. The Health Effects Institute (HEI 2013) [45] reflected that the ongoing evidence did not convincingly support the suggestion that $PM_{0.1}$ alone can account for actual conduct for the adverse effects that have been associated with atmospheric pollutants such as $PM_{2.5}$ and PM_{10}.

In health risk assessment, toxicity equivalent concentration (TEQs), calculations based on toxic equivalent factors (TEFs), can be used to estimate health risks associated with PAHs [41]. High concentrations of PMs containing PAHs are well-known in symptoms, i.e., eye irritation, diarrhea, vomiting, and nausea. The detrimental effects of PAHs hang on the mechanism of exposure. Benzo[a]pyrene (BaP) is the well-known PAH to cause cancer on a laboratory scale resulting long term exposure. The BaP-TEQ is a widely used indicator to estimate the exposure to PAHs to human health. On the other hand, there have been limited studies on the characteristics of $PM_{0.1}$ and toxic PAHs in the Southeast Asian environment; moreover, it is vital to note that finer particles are a more significant source of carcinogenic properties and cause more human health consequences than larger particles due to their higher surface area that can absorb many toxic elements.

Phairuang et al. (2021, 2022) [46,47] investigated the health risk of $PM_{0.1}$ and its trace elements, such as Aluminium (Al), Barium (Ba), Potassium (K), Iron (Fe), Chromium (Cr), Copper (Cu), Nickel (Ni), Sodium (Na), Manganese (Mn), Magnesium (Mg), Titanium (Ti), Lead (Pb), and Zinc (Zn), on humans in Bangkok and Hat Yai, Thailand. PM's primary sources were road traffic, industry, and biomass burning. In addition, the total cancer risk from all carcinogenic elements was high in adults, indicating that the carcinogenic risk is within a tolerable risk assessment range.

Guan et al. (2017) [48], in a study conducted in China, reported that increasing 10 $\mu g/m^3$ in $PM_{2.5}$ from any emission source was linked to a 3.1% increase in the risk of hospitalization as well as a 2.5% increase in mortality. Crippa et al. (2016) [49] reported that the short-term exposure to the burning of agricultural residues and peat-land fires in heavy haze episodes in 2015 from Indonesia might have caused 11,880 excess mortalities. Most studies have stated the adverse effect of inhaled atmospheric $PM_{0.1}$ on human health to continue lacking in SEA. Three are still limited information between $PM_{0.1}$ and disease; however, there have not become fully aware of the critical hazards of ambient $PM_{0.1}$ on human health [7].

5. Challenges Study in $PM_{0.1}$

5.1. Evaluation of $PM_{0.1}$

The present status and characteristics, comparison between cities and countries need assessments of events and long-range transportation. For instance, SEA has been a source of PM pollutants affecting countries both inside and outside this region [14]. The transport of plumes from Indonesian forest fires involves, e.g., Singapore, Malaysia, Brunei, and

southern Thailand [27]. US-EPA has mentioned that PM is one of the criteria pollutants of atmospheric pollution, the most widespread health threat. PM is the generic term used for these air pollutants, consisting of complex mixtures of solid or liquid droplets suspended in the breathing air, varying in size and composition [17]. According to the particle size criteria, most SEA countries designated $PM_{2.5}$ as a pollutant; however, $PM_{0.1}$ is more concerned about health impact than larger PM sizes. Therefore, further studies on the sources, abundance, chemical compositions, and migration of $PM_{0.1}$ between regions and countries are needed.

5.2. Information on $PM_{0.1}$ Emission Sources

The Emission Inventory (EI) of $PM_{0.1}$ is subject to very high mass and particle number emission uncertainties. There are very few emission factors (EFs) of $PM_{0.1}$ available through official EI guidebooks. EFs on $PM_{0.1}$ number, mass, and chemicals should be studied more closely and extensively. Samae et al. (2022) [50] reported the first EFs of $PM_{0.1}$ for solid biomass combustion in Thailand, including 11 types of biomasses (*Avicennia alba* Blume, *Xylocarpus moluccensis*, *Rhizophora mucronate*, bagasse, sugarcane leaves, corn residue, rice straw, rice stubble, palm fiber, palm kernel, and rubberwood). The $PM_{0.1}$ mass fraction was estimated at approximately 1–8% of total PM and the EF was shown to be in the range of 0.1–0.28 $g.kg^{-1}$. There is no completed EI of $PM_{0.1}$ in Asia due to a lack of information on EFs [20]. Knowledge of the EFs of $PM_{0.1}$ is essential to developing strategies for pollution control and air quality management.

5.3. Development and Application of New $PM_{0.1}$ Tools

5.3.1. Measurements of Atmospheric $PM_{0.1}$ Particles

Measurements of atmospheric particles are inherently more complex than other gases-phase pollutants. The new technology to size-classified PMs down to $PM_{0.1}$ is also vital to studying the physical and chemical characteristics of ambient $PM_{0.1}$ [51]. For example, the inertial filter (IF) technology to collect nanoparticles in a short sampling period will be essential for gathering the ambient $PM_{0.1}$ because of its limited mass concentration [52,53]. The artifacts due to semi-volatile evaporation should be much smaller than those of conventional nanoparticle samplers, e.g., low-pressure impactors and Nano-MOUDI [54]. In SEA, two main types of cascade samplers based on the inertial filter technology by Otani et al. (2007) [55], referred to as "$PM_{0.1}$ personal sampler" and "$PM_{0.1}$ ambient sampler" were used. A $PM_{0.1}$ personal sampler was developed by Furuuchi et al. (2010) [56] and revised by Thongyen et al. (2015) [57].

5.3.2. A High-Volume $PM_{0.1}$ Air Sampler

A High-Volume $PM_{0.1}$ air sampler that can collect a large amount of $PM_{0.1}$ in a short sampling duration should be useful for toxicity evaluation that normally requires a rather large PM mass for toxicity tests. Because of their large specific surface area, the longer retention time in the sampling system causes artifacts as evaporation of semi-volatile components and degradation by oxidation of chemicals. Considering $PM_{0.1}$ is related to chemical compositions with minimal artifacts during air sampling; therefore, this is also a crucially important issue [53,54]. The High-Volume air sampler for $PM_{0.1}$ can improve the $PM_{0.1}$ instrument toward (1) understanding ambient $PM_{0.1}$ based on various chemicals in $PM_{0.1}$ collected with high time resolution, (2) risk assessment using a large amount of $PM_{0.1}$ collected by the tool [58,59].

5.3.3. The $PM_{0.1}$ Real-Time Sensor

The $PM_{0.1}$ real-time sensor in the internet of things (IoT) monitoring network (big data) will play an essential role in understanding the $PM_{0.1}$ plume migration and transportation, with temporal variations by geo-specific location [60]. The real-time and IoT sensor for PMs monitoring has been a vital tool, potentially becoming an integral part of air quality

monitoring and management, especially during the haze episode of intensive biomass burning smoke in the SEA environment [61].

5.4. Summarizing Facts on $PM_{0.1}$ for Policy-Making

Future researches on $PM_{0.1}$ need to focus on intervention, classification, and quantification of mass and number concentrations in an ambient that is influenced by mixed emission sources and also on personal levels to detrimental human health effects. The environmental quality standard regarding $PM_{0.1}$ in SEA is only focused on mass concentrations. The precise future study of particle types and sizes that govern $PM_{0.1}$ will fulfill the research gaps, perspectives, and emerging challenges for $PM_{0.1}$ policy-making in SEA.

6. Conclusions

Future studies on atmospheric nanoparticles in SEA should focus on the abundance, sources, distribution, and temporal and spatial variations of $PM_{0.1}$ in urban and rural SEA environments. The exposure to $PM_{0.1}$ should be quantified to understand the exposure of nano-size particles in humans. Continuous monitoring of $PM_{0.1}$ in mass and number concentration should be further understood. Future research on $PM_{0.1}$ needs to focus on identifying and quantifying $PM_{0.1}$ particles. Finally, better knowledge about the physicochemical characteristics of $PM_{0.1}$ generated by various emission sources in the SEA environment will help fill the gap in air quality policies and management in this region.

Author Contributions: Conceptualization, W.P.; writing—original draft preparation, W.P.; writing—review and editing, M.A., M.H., M.F. All authors have read and agreed to the published version of the manuscript.

Funding: This work was financially supported by the Office of the Permanent Secretary, Ministry Higher Education, Science, Research and Innovation (Grant No. RGNS 63-253) in Thailand.

Institutional Review Board Statement: Not applicable.

Informed Consent Statement: Not applicable.

Data Availability Statement: Not applicable.

Acknowledgments: The authors wish to thank Milton S. Feather for improving the English.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. Buseck, P.R.; Adachi, K. Nanoparticles in the atmosphere. *Elements* **2008**, *4*, 389–394. [CrossRef]
2. Lanzinger, S.; Schneider, A.; Breitner, S.; Stafoggia, M.; Erzen, I.; Dostal, M. Associations between ultrafine and fine particles and mortality in five central European cities-Results from the UFIREG study. *Environ. Int.* **2016**, *88*, 44–54. [CrossRef] [PubMed]
3. Venecek, M.A.; Yu, X.; Kleeman, M.J. Predicted ultrafine particulate matter source contribution across the continental United States during summertime air pollution events. *Atmos. Chem. Phys.* **2019**, *19*, 9399–9412. [CrossRef]
4. Hata, M.; Chomanee, J.; Thongyen, T.; Bao, L.; Tekasakul, S.; Tekasakul, P.; Otani, Y.; Furuuchi, M. Characteristics of nanoparticles emitted from burning of biomass fuels. *J. Environ. Sci.* **2014**, *26*, 1913–1920. [CrossRef] [PubMed]
5. Schraufnagel, D.E. The health effects of ultrafine particles. *Exp. Mol. Med.* **2020**, *52*, 311–317. [CrossRef]
6. Slezakova, K.; Morais, S.; do Carmo Pereira, M. Atmospheric nanoparticles and their impacts on public health. In *Current Topics in Public Health*; IntechOpen: London, UK, 2013.
7. Kwon, H.S.; Ryu, M.H.; Carlsten, C. Ultrafine particles: Unique physicochemical properties relevant to health and disease. *Exp. Mol. Med.* **2020**, *52*, 318–328. [CrossRef] [PubMed]
8. AQEG. *Particulate Matter in the United Kingdom*; PB10580; Air Quality Expert Group, UK Department for Environment, Food and Rural Affairs: London, UK, 2005; Available online: http://ukair.defra.gov.uk/library/reports?report_id=269 (accessed on 8 August 2022).
9. Kumar, P.; Pirjola, L.; Ketzel, M.; Harrison, R.M. Nanoparticle emissions from 11 non-vehicle exhaust sources—A review. *Atmos Environ.* **2013**, *67*, 252–277. [CrossRef]
10. Terzano, C.; Di Stefano, F.; Conti, V.; Graziani, E.; Petroianni, A. Air pollution ultrafine particles: Toxicity beyond the lung. *Eur. Rev. Med. Pharmacol. Sci.* **2010**, *14*, 809–821.
11. Xue, J.; Xue, W.; Sowlat, M.H.; Sioutas, C.; Lolinco, A.; Hasson, A.; Kleeman, M.J. Seasonal and annual source appointment of carbonaceous ultrafine particulate matter ($PM_{0.1}$) in Polluted California Cities. *Environ. Sci. Technol.* **2018**, *53*, 39–49. [CrossRef]

12. De Jesus, A.L.; Rahman, M.M.; Mazaheri, M.; Thompson, H.; Knibbs, L.D.; Jeong, C.; Evans, G.; Nei, W.; Ding, A.; Qiao, L.; et al. Ultrafine particles and $PM_{2.5}$ in the air of cities around the world: Are they representative of each other? *Environ. Int.* **2019**, *129*, 118–135. [CrossRef] [PubMed]
13. Phairuang, W.; Hata, M.; Furuuchi, M. Influence of agricultural activities, forest fires and agro-industries on air quality in Thailand. *J. Environ. Sci.* **2017**, *52*, 85–97. [CrossRef]
14. Gautam, S.; Yadav, A.; Tsai, C.J.; Kumar, P. A review on recent progress in observations, sources, classification and regulations of $PM_{2.5}$ in Asian environments. *Environ. Sci. Pollut. Res.* **2016**, *23*, 21165–21175. [CrossRef] [PubMed]
15. Cui, M.; Chen, Y.; Zheng, M.; Li, J.; Tang, J.; Han, Y.; Song, D.; Yan, C.; Zhang, F.; Tian, C.; et al. Emissions and characteristics of particulate matter from rainforest burning in the Southeast Asia. *Atmos. Environ.* **2018**, *191*, 194–204. [CrossRef]
16. Vejpongsa, I.; Suvachittanont, S.; Klinklan, N.; Thongyen, T.; Veres, M.; Szymanski, W.W. Deliberation between PM 1 and PM 2.5 as air quality indicators based on comprehensive characterization of urban aerosols in Bangkok, Thailand. *Particuology* **2017**, *35*, 1–9. [CrossRef]
17. WHO. *Review of Evidence on Health Aspects of Air Pollution-REVIHAAP Project: Technical Report*; World Health Organization: Copenhagen, Denmark, 2013.
18. Amin, M.; Putri, R.M.; Handika, R.A.; Ullah, A.; Goembira, F.; Phairuang, W.; Ikemori, F.; Hata, M.; Tekasakul, P.; Furuuchi, M. Size-Segregated particulate matter down to $PM_{0.1}$ and carbon content during the rainy and dry seasons in Sumatra island, Indonesia. *Atmosphere* **2021**, *12*, 1441. [CrossRef]
19. Amin, M.; Handika, R.A.; Putri, R.M.; Phairuang, W.; Hata, M.; Tekasakul, P.; Furuuchi, M. Size-Segregated Particulate Mass and Carbonaceous Components in Roadside and Riverside Environments. *Appl. Sci.* **2021**, *11*, 10214. [CrossRef]
20. Phairuang, W.; Suwattiga, P.; Chetiyanukornkul, T.; Hongtieab, S.; Limpaseni, W.; Ikemori, F.; Hata, M.; Furuuchi, M. The influence of the open burning of agricultural biomass and forest fires in Thailand on the carbonaceous components in size-fractionated particles. *Environ. Pollut.* **2019**, *247*, 238–247. [CrossRef]
21. Putri, R.M.; Amin, M.; Suciari, T.F.; Faisal, M.A.F.; Auliani, R.; Ikemori, F.; Wada, M.; Hata, M.; Tekasakul, P.; Furuuchi, M. Site-specific variation in mass concentration and chemical components in ambient nanoparticles ($PM_{0.1}$) in North Sumatra Province-Indonesia. *Atmos. Pollut. Res.* **2021**, *12*, 101062. [CrossRef]
22. Kumar, P.; Morawska, L.; Birmili, W.; Paasonen, P.; Hu, M.; Kulmala, M.; Harrison, R.M.; Norford, L.; Britter, R. Ultrafine particles in cities. *Environ. Int.* **2014**, *66*, 1–10. [CrossRef]
23. *CEN/TS 16976:2016*; Ambient Air-Determination of the Particle Number Concentration of Atmospheric Aerosol. European Committee for Standardization: Brussels, Belgium, 2016.
24. Giechaskiel, B.; Lahde, T.; Suarez-Bertoa, R.; Clairotte, M.; Grigoratos, T.; Zardini, A.; Perujo, A.; Martini, G. Particle number measurements in the European legislation and future JRC activities. *Combust. Engines* **2018**, *174*, 3–16. [CrossRef]
25. Li, N.; Georas, S.; Alexis, N.; Fritz, P.; Xia, T.; Williams, M.A.; Horner, E.; Nel, A. A work group report on ultrafine particles (American Academy of Allergy, Asthma & Immunology): Why ambient ultrafine and engineered nanoparticles should receive special attention for possible adverse health outcomes in human subjects. *J. Allergy Clin. Immunol.* **2016**, *138*, 386–396. [CrossRef] [PubMed]
26. Boongla, Y.; Chanonmuang, P.; Hata, M.; Furuuchi, M.; Phairuang, W. The characteristics of carbonaceous particles down to the nanoparticle range in Rangsit city in the Bangkok Metropolitan Region, Thailand. *Environ. Pollut.* **2020**, *272*, 115940. [CrossRef] [PubMed]
27. Phairuang, W.; Inerb, M.; Furuuchi, M.; Hata, M.; Tekasakul, S.; Tekasakul, P. Size-fractionated carbonaceous aerosols down to $PM_{0.1}$ in southern Thailand: Local and long-range transport effects. *Environ. Pollut.* **2020**, *260*, 114031. [CrossRef] [PubMed]
28. Zhao, T.; Hongtieab, S.; Hata, M.; Furuuchi, M.; Dong, S.; Phairuang, W.; Ge, H.; Zhang, T. Ambient nanoparticles characterization by East and Southeast Asia nanoparticle monitoring network. In Proceedings of the 9th Asian Aerosol Conference, Kanazawa, Japan, 24–26 June 2015.
29. Othman, M.; Latif, M.T.; Hamid, H.H.A.; Uning, R.; Khumsaeng, T.; Phairuang, W.; Daud, Z.; Idris, J.; Sofwan, N.M.; Lung, S.-C.C. Spatial–temporal variability and health impact of particulate matter during a 2019–2020 biomass burning event in Southeast Asia. *Sci. Rep.* **2022**, *12*, 7630. [CrossRef] [PubMed]
30. Thuy, N.T.T.; Dung, N.T.; Sekiguchi, K.; Thuy, L.B.; Hien, N.T.T.; Yamaguchi, R. Mass concentrations and carbonaceous compositions of $PM_{0.1}$, $PM_{2.5}$, and PM_{10} at urban locations in Hanoi, Vietnam. *Aerosol Air Qual. Res.* **2018**, *18*, 1591–1605. [CrossRef]
31. Nghiem, T.D.; Nguyen, T.T.T.; Nguyen, T.T.H.; Ly, B.T.; Sekiguchi, K.; Yamaguchi, R.; Pham, C.T.; Ho, Q.B.; Duong, T.N. Chemical characterization and source apportionment of ambient nanoparticles: A case study in Hanoi, Vietnam. *Environ. Sci. Pollut. Res.* **2020**, *27*, 30661–30672. [CrossRef]
32. Huyen, T.T.; Yamaguchi, R.; Kurotsuchi, Y.; Sekiguchi, K.; Dung, N.T.; Thuy, N.T.T.; Thuy, L.B. Characteristics of Chemical Components in Fine Particles ($PM_{2.5}$) and Ultrafine Particles ($PM_{0.1}$) in Hanoi, Vietnam: A Case Study in Two Seasons with Different Humidity. *Water Air Soil Pollut.* **2021**, *232*, 183. [CrossRef]
33. Ha, V.T.L.; Anh, V.D.; Hien, N.T.T.; Dung, N.T.; Shimada, Y.; Yoneda, M. Indoor and outdoor relationship of particles with different sizes at an apartment in Hanoi: Mass concentration and respiratory dose estimation. *Vietnam. J. Sci. Technol.* **2020**, *58*, 736–746. [CrossRef]

34. Thuy, P.C.; Le, H.A.; Tuyen, L.H.; Dung, N.T. Size distribution and contribution of particles from rice straw open burning to the atmosphere in Hanoi. *Vietnam. J. Sci. Technol.* **2020**, *58*, 94–104.
35. Tran, T.D.; Nguyen, P.M.; Nghiem, D.T.; Le, T.H.; Tu, M.B.; Alleman, L.Y.; Nguyen, V.M.; Pham, D.T.; Ha, N.M.; Dang, M.N.; et al. Assessment of air quality in school environments in Hanoi, Vietnam: A focus on Mass-Size distribution and elemental composition of indoor-outdoor Ultrafine/Fine/Coarse particles. *Atmosphere* **2020**, *11*, 519. [CrossRef]
36. Chen, S.C.; Tsai, C.J.; Chou, C.C.K.; Roam, G.D.; Cheng, S.S.; Wang, Y.N. Ultrafine particles at three different sampling locations in Taiwan. *Atmos Environ.* **2010**, *44*, 533–540. [CrossRef]
37. Zhu, C.S.; Chen, C.C.; Cao, J.J.; Tsai, C.J.; Chou Charles, C.K.; Liu, S.C.; Roam, G.D. Characterization of carbon fractions for atmospheric fine particles and nanoparticles in a highway tunnel. *Atmos Environ.* **2010**, *44*, 2668–2673. [CrossRef]
38. Hata, M.; Zhang, T.; Bao, L.; Otani, Y.; Bai, Y.; Furuuchi, M. Characteristics of the nanoparticles in a Road Tunnel. *Aerosol. Air Qual. Res.* **2012**, *13*, 194–200. [CrossRef]
39. Ding, X.; Kong, L.; Du, C.; Zhanzakova, A.; Wang, L.; Fu, H.; Chen, J.; Yang, X.; Cheng, T. Long-range and regional transported size-resolved atmospheric aerosols during summertime in urban Shanghai. *Sci. Total. Environ.* **2017**, *583*, 334–343. [CrossRef] [PubMed]
40. Beddows, D.C.; Harrison, R.M. Comparison of average particle number emission factors for heavy and light duty Vehicles derived from rolling chassis dynamometer and field studies. *Atmos Environ.* **2008**, *42*, 7954–7966. [CrossRef]
41. Phairuang, W.; Tekasakul, T.; Hata, M.; Tekasakul, S.; Chomanee, J.; Otani, Y.; Furuuchi, M. Estimation of air pollution from ribbed smoked sheet rubber in Thailand exports to Japan as a pre-product of tires. *Atmos. Pollut. Res.* **2019**, *10*, 642–650. [CrossRef]
42. Chomanee, J.; Tekasakul, S.; Tekasakul, P.; Furuuchi, M. Effect of irradiation energy and residence time on decomposition efficiency of polycyclic aromatic hydrocarbons (PAHs) from rubber wood combustion emission using soft X-rays. *Chemosphere* **2018**, *210*, 417–423. [CrossRef]
43. Rimnacova, D.; Zdímal, V.; Schwarz, J.; Smolík, J.; Rimnac, M. Atmospheric aerosols in suburb of Prague: The dynamics of particle size distributions. *Atmos. Res.* **2011**, *101*, 539–552. [CrossRef]
44. Reche, C.; Querol, X.; Alastuey, A.; Viana, M.; Pey, J.; Moreno, T.; Rodríguez, S.; González, Y.; Fernández-Camacho, R.; de la Rosa, J.; et al. New considerations for PM, Black Carbon and particle number concentration for air quality monitoring across different European cities. *Atmos. Chem. Phys.* **2011**, *11*, 6207–6227. [CrossRef]
45. HEI. Understanding the health effects of ambient ultrafine particles. In *HEI Perspectives 3. HEI Review Panel on Ultrafine Particles*; Health Effects Institute: Boston, MA, USA, 2013.
46. Phairuang, W.; Inerb, M.; Hata, M.; Furuuchi, M. Characteristics of trace elements bound to ambient nanoparticles ($PM_{0.1}$) and a health risk assessment in southern Thailand. *J. Hazard. Mater.* **2021**, *425*, 127986. [CrossRef]
47. Phairuang, W.; Suwattiga, P.; Hongtieab, S.; Inerb, M.; Furuuchi, M.; Hata, M. Characteristics, sources, and health risks of ambient nanoparticles ($PM_{0.1}$) bound metal in Bangkok, Thailand. *Atmos. Environ. X* **2021**, *12*, 100141. [CrossRef]
48. Guan, Y.; Chen, G.; Cheng, Z.; Yan, B.; Hou, L. Air pollutant emissions from straw open burning: A case study in Tianjin. *Atmos. Environ.* **2017**, *171*, 155–164. [CrossRef]
49. Crippa, P.; Castruccio, S.; Archer-Nicholls, S.; Lebron, G.B.; Kuwata, M.; Thota, A.; Sumin, S.; Butt, E.; Wiedinmyer, C.; Spracklen, D.V. Population exposure to hazardous air quality due to the 2015 fires in Equatorial Asia. *Sci. Rep.* **2016**, *6*, 37074. [CrossRef] [PubMed]
50. Samae, H.; Tekasakul, S.; Tekasakul, P.; Phairuang, W.; Furuuchi, M.; Hongtieab, S. Particle-bound organic and elemental carbons for source identification of PM < 0.1 μm from biomass combustion. *J. Environ. Sci.* **2022**, *113*, 385–393.
51. Hata, M.; Linfa, B.; Otani, Y.; Furuuchi, M. Performance evaluation of an Andersen cascade impactor with an additional stage for nanoparticle sampling. *Aerosol Air Qual. Res.* **2012**, *12*, 1041–1048. [CrossRef]
52. Furuuchi, M.; Eryu, K.; Nagura, M.; Hata, M.; Kato, T.; Tajima, N.; Sekiguchi, K.; Ehara, K.; Seto, T.; Otani, Y. Development and Performance Evaluation of Air Sampler with Inertial Filter for Nanoparticle Sampling. *Aerosol Air Qual. Res.* **2010**, *10*, 185–192. [CrossRef]
53. Kumsanlas, N.; Piriyakarnsakul, S.; Sok, P.; Hongtieab, S.; Ikemori, F.; Szymanski, W.W.; Hata, M.; Otani, Y.; Furuuchi, M. A Cascade Air Sampler with Multi-nozzle Inertial Filters for PM0.1. *Aerosol Air Qual. Res.* **2019**, *19*, 1666–1677. [CrossRef]
54. Liu, C.-N.; Awasthi, A.; Hung, Y.-H.; Tsai, C.-J. Collection efficiency and interstage loss of nanoparticles in micro-orifice-based cascade impactors. *Atmos. Environ.* **2013**, *69*, 325–333. [CrossRef]
55. Otani, Y.; Eryu, K.; Furuuchi, M.; Tajima, N.; Tekasakul, P. Inertial Classification of Nanoparticles with Fibrous Filters. *Aerosol Air Qual. Res.* **2007**, *7*, 343–352. [CrossRef]
56. Furuuchi, M.; Choosong, T.; Hata, M.; Otani, Y.; Tekasakul, P.; Takizawa, M.; Nagura, M. Development of a personal sampler for evaluating exposure to ultrafine particles. *Aerosol Air Qual. Res.* **2010**, *10*, 30–37. [CrossRef]
57. Thongyen, T.; Hata, M.; Toriba, A.; Ikeda, T.; Koyama, H.; Otani, Y.; Furuuchi, M. Development of PM0.1 personal sampler for evaluation of personal exposure to aerosol nanoparticles. *Aerosol Air Qual. Res.* **2015**, *15*, 180–187. [CrossRef]
58. Hata, M.; Thongyen, T.; Bao, L.; Hoshino, A.; Otani, Y.; Ikeda, T.; Furuuchi, M. Development of a high-volume air sampler for nanoparticles. *Environ. Sci. Process Impacts* **2013**, *15*, 454–462. [CrossRef]
59. Zhang, T.; Zhao, T.; Takahashi, H.; Hata, M.; Torib, A.; Ikeda, T.; Otani, Y.; Furuuchi, M. High volume air sampler for environmental nano-particles using a sharp-cut inertial filter combined with an impactor. *Meas. Sci. Technol.* **2017**, *28*, 025801. [CrossRef]

60. Kanabkaew, T.; Mekbungwan, P.; Raksakietisak, S.; Kanchanasut, K. Detection of $PM_{2.5}$ plume movement from IoTground level monitoring data. *Environ. Pollut.* **2019**, *252*, 543–552. [CrossRef]
61. Inerb, M.; Phairuang, W.; Paluang, P.; Hata, M.; Furuuchi, M.; Wangpakapattanawong, P. Carbon and Trace element compositions of Total Suspended Particles (TSP) and Nanoparticles ($PM_{0.1}$) in ambient air of Southern Thailand and Characterization of Their Sources. *Atmosphere* **2022**, *13*, 626. [CrossRef]