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Effectiveness of wearing face masks against traffic particles on the streets of Ho Chi Minh City, Vietnam†

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Motorbikes are by far the dominant mode of transportation in Ho Chi Minh City (HCMC). They solve mobility problems but represent a health risk since riders are directly exposed to noxious exhaust fumes. Hence, face masks emerge as a solution to reduce exposure to harmful particles. The manufacturers of these masks report that they can significantly reduce particle exposure on roads with vehicular traffic. Such reports are usually based on laboratory assessments, with limited data from field experiments. To evaluate the performance of the masks commonly worn by HCMC commuters under quasi-real exposure conditions, we tested the total inward leakage of particles (*i.e.*, including penetration through the filter media, and leaks from the face seal and exhalation valve if the mask is equipped with one) of six representative masks mounted on manikins at the curbside of two busy roads during high traffic time periods. Several particle metrics, including mass and number concentrations, active surface area, and abundances of equivalent black carbon and particle-bound polycyclic aromatic hydrocarbons, were measured to determine the protection level provided by masks against distinct types of particles. As part of this study, through a set of measurements using the same instrumentation we found that commuters are exposed to a mix of freshly emitted particles and aged particles, including contributions from sources other than motorbike exhaust, such as trash burning and street food stalls. Ultrafine particles, especially those in the nucleation mode (<50 nm), turned out to be the dominant fraction in terms of number concentration. This study focused its evaluation on these particles. We found that no mask can completely remove all particles under practical conditions. It is largely due to inappropriate mask fitting. Performance efficiency of 60–80% was achieved by an N95 respirator, a reusable valved filtering mask, and a locally manufactured carbon-layer sandwiched mask. Surgical and cloth masks achieved efficiencies of 25–60%. The results show that any face mask provides some level of protection. Efforts should be made to provide end users with practical information on the effectiveness of masks under real conditions, and informing on how to best fit each mask to increase effectiveness.

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Environmental significance

Cities like Ho Chi Minh Cit (Vietnam) in Southeast Asia are known for their dense crowds of motorbikes. Most motorbike users wear long sleeves, gloves, and face masks, even in hot weather, to protect their skin from the sun, and reduce the intake of traffic particles and resuspended dust. Face masks do remove particles with varying efficiencies under real wearing conditions on the streets that may differ from the efficiencies reported by their manufacturers. It is important to study the characteristics of the particles to which commuters are exposed to and to conduct *in situ* tests on the efficiency performance of such masks.

1. Introduction

Even before the onset of COVID-19, face masks have been the *de facto* of daily life in Vietnam and in many other countries of Asia due to air pollution problems.[1–3] Residents of large cities in Southeast Asia like Ho Chi Minh City (HCMC), Bangkok and Jakarta regularly wear masks when commuting as a protection against traffic particles, especially when traveling by motorbike. Motorbikes are by far the dominant transport mode in these cities. They are swift, affordable, and appropriate for the urban

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landscape and road infrastructure of these cities. For the case of HCMC, the city studied in this paper, the share of trips made by motorbike exceeds 85% of all transport modes.[4] Almost every person aged 16 years or older owns a motorbike. From 2014 to 2019 the number of motorbikes increased 25%. Eight and half million motorbikes are in use by HCMC's 9-million inhabitants.[5,6]

Multiple studies have showed the threat to public health that traffic pollution represents.[7–9] For the case of HCMC, two studies have found nitrogen dioxide (NO_2) and fine particles (≤2.5 μm in size, $PM_{2.5}$), two air pollutants largely attributed to traffic emissions, to be associated with cardiorespiratory hospitalizations. These studies are in agreement with studies elsewhere.[10,11] Due to the lack of effective emission controls, especially on motorcycles throughout the developing world, exhaust emissions from motorbikes tend to exceed those from passenger cars.[12–15] The enormous number of motorbikes exacerbates the adverse health effects triggered by traffic pollution.

In this context, the effectiveness of wearing face masks is relevant to assess the exposure of motorbike commuters and pedestrians to traffic pollution, since they are directly exposed to exhaust fumes and dust resuspension. The evidence from laboratory-based studies suggests that distinct types of face masks provide varying levels of protection depending on the materials used for their design, the load of pollutants, the size and characteristics of the particles, and the face fit (*i.e.*, edge-seal leakage).[16–29]

The COVID-19 pandemic triggered a spate of studies on the performance and filtering efficiency of face masks. Before the pandemic, limited studies were available, and the majority focused on medical masks and high efficiency respirators for occupational exposure. The mandatory wearing of masks in many countries to reduce the spread of the Severe Acute Respiratory Syndrome Coronavirus 2 (SARS-CoV-2) limited the supply of medical masks and respirators, forcing the community to evaluate the filtering capacity of other types of masks. Most of these and previous studies have followed test protocols (somewhat) similar to the US National Institute for Occupational Safety and Health (NIOSH) testing procedures for air-purifying particulate respirator certification. NIOSH protocols test respirator filters sealed onto a test fixture at 85 L min^{-1} using relatively small (0.3 μm) sodium chloride (NaCl) aerosols as test agent.[30] Some studies have deviated from NIOSH protocols and included instrumentation to assess filtration efficiencies against particles of a given size and chemical composition, as well as used custom-built exposure chambers, manikins, and even personal sampling systems to test inward protection efficiencies.[16–20,24,26,28,29] Only a few studies have included special set ups to test the masks efficiency against diluted exhaust fumes from diesel engines.[22,23,25,27]

Studies that have used diesel aerosols as test agent provide better insight than those that have used NaCl aerosols to evaluate the protection provided by face masks and respirators in microenvironments affected by traffic emissions. But their results cannot be directly compared since they used different test protocols and the metrics evaluated were not the same. For example, Cherrie *et al.* evaluated both filtration and performance efficiencies for a range of face masks available in China. Filtration efficiency was tested by drawing diesel aerosols through a section of the filtering medium and measuring concentrations of $PM_{2.5}$ and equivalent black carbon (eBC) upstream and downstream. Performance efficiency was assessed by measuring the particle penetration factor using eBC as test agent on human subjects in a chamber.[22] Shakya *et al.* tested the performance efficiency of cloth and surgical masks by measuring their ability to remove diesel particles of three specific sizes. The masks were mounted on a manikin inside a chamber and the particles were measured with a particle counter.[23] Burton *et al.* tested the filtration efficiency of three Australian certified respirators against elemental carbon, total carbon, and total suspended particles produced by a diesel generator. A section of the filtering medium was mounted on a fixture inside a chamber, and particles were collected on filters before and after passing through the respirator material. Particle samples were analyzed offline.[25] Finally, Penconek *et al.* tested the filtration efficiency of two European certified respirators sealed to a manikin head inside a chamber. The number size distributions of diesel aerosols inside and outside the mask were measured with a differential mobility analyzer.[27] What these studies have in common is that all four were carried out in laboratories under controlled conditions, and that only evaluated the masks efficiency against aerosols from a single source of diesel exhaust.

To our knowledge, no study has evaluated the masks performance efficiency considering the entire mix of particles in microenvironments affected by vehicular traffic. Only a handful of studies have directly evaluated the effectiveness of wearing masks on health outcomes under real exposure conditions focusing on changes in blood pressure, heart rate, and oxidative stress.[31–34]

The degree of protection the commonly worn masks offer to motorbike commuters in cities like HCMC is the main question to answer in this paper. We tested the performance efficiency of six types of face masks using manikins and a set of portable air quality monitoring and measuring instruments. However, one may wonder if wearing a face mask is really necessary to reduce exposure concentrations on HCMC streets. If particle pollution is not severe, there would be no point in wearing a mask in the first place. To answer this second question, the same particle metrics that were measured to test the masks efficiency were evaluated at street level during the morning and evening rush hour.

2. Methodology

The measurements to test the inward protection efficiency of the face masks and to characterize air pollution at street level were performed on the curbside along a set of representative roads. We tested the ability of face masks to reduce the mass concentration of $PM_{2.5}$, eBC and particle-bound polycyclic aromatic hydrocarbons (pPAHs), and the number concentration of particles (PN) and associated active surface area (ASA) as proxies of ultrafine particles (≤100 nm, UFP). Carbon monoxide

Fig. 1 Measurement sites. Except for site #3 that was only affected by motorbike emissions, all sites were also affected by emissions from passenger cars to a lesser extent. Large diesel trucks were only observed at site #2, and public buses at sites #4 and #7. Contributions from street food stalls and trash burning were experienced at all sites, to a greater extent at sites #1 and #8. The ESI includes a map (Fig. ESI1†) indicating the location of the measurement sites within HCMC urban sprawl.

(CO) mixing ratios were also measured as a tracer of traffic emissions during the street level measurements. We used relationships between pPAHs concentration and ASA as fingerprints of particles originated from combustion sources.[35,36] In addition to traffic particles, we saw that pedestrians and commuters were also exposed to cooking and biomass burning particles from street food stalls and trash burning, which were inherently included in the measurements. Similarly, assuming spherical particles, the concurrent and independent measurements of PN concentration and ASA were used to estimate the size of the particles by the diameter of average surface ($D_{\mathrm{Aver,S}}$) as proposed by Kittelson *et al.*[37]

The monitors were placed on a table at the breathing height of the motorcyclists (~1.20 m). Each set of measurements had a duration of 2–3 hours. Fig. 1 supplies the locations, dates, and periods of the measurements that were made to initially characterize traffic pollution. The masks efficiency tests were then conducted on later dates at sites #2 and #8.

The eight sites that were selected for this study were based on information provided by local students who took part to assist. These sites depicted distinctive characteristics in terms of traffic flow and urban morphology, but minor variation in terms of fleet composition. All sites were affected by motorbike emissions, and to a lesser extent by emissions from passenger cars, except site #3 which corresponded to an underground parking lot restricted to motorbikes. Heavy duty vehicles (>2.5 metric tons) are banned to enter the inner districts of the city during daytime, thus large diesel trucks were only observed in a major ring road (site #2) where they can circulate the entire day. Public buses were only observed at sites #4 and #7. Although all sites were affected by both trash burning smoke and fumes from street food stalls, ubiquitous along HCMC streets, the impact of both emission sources was most acute at sites #1 and #5. Site #1 was also affected by emissions from an informal motorbike workshop on the street while site #4 was affected by emissions from a busy gasoline station.

Fig. 2 shows the fleet composition at each measurement site. At all sites, motorbikes dominated (>80%) the traffic flow. Only sites #4 and #6 saw slightly more passenger cars. Diesel trucks accounted for 6% at site #2, while small diesel trucks (<2.5 metric tons) carrying construction materials and consumer goods accounted less than 2% at the other sites. From nearly

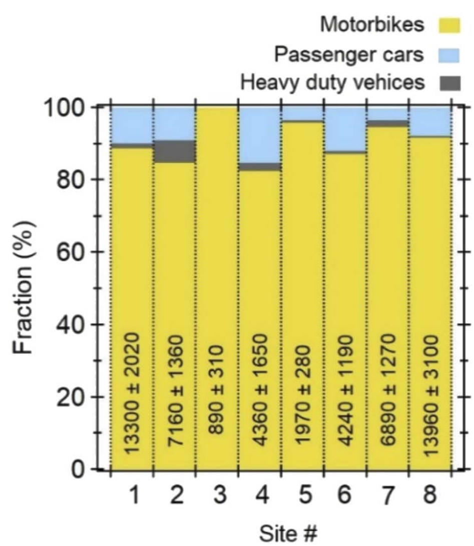

Fig. 2 Composition of the vehicular fleet at each measurement site. Figures inside the bars indicate mean and one standard deviation of the total traffic flow per hour.

2000 to 14 000 vehicles per hour were manually counted onsite during the measurements. Sites #1 and #8 recorded the highest traffic flow. Severe traffic congestion was experienced at site #2 (chronic flowing congestion) and #1 (acute standstill congestion). The fleet composition reported here is in line with data obtained from a traffic survey conducted across the city in 2017.[38] This survey found that 63% and 32% of motorbikes met Euro III and Euro II emission standards, respectively, while the rest Euro I or other. Regarding to passenger cars, 85% met Euro V or Euro IV standards, and the rest Euro III standards. While most diesel vehicles met Euro II standards.

The study was conducted on weekdays during the last week of February and first week of March 2016 during the dry season in southern Vietnam (December to April), when air pollution gets worse. Ho Chi Minh City has a hot and humid climate all year round with a distinctive rainy season and dry season. Heavy rains during the former season clean the air a bit. In terms of $PM_{2.5}$, monthly mean concentrations reach 40 μg m^{-3} during the dry season, and 30 μg m^{-3} during the rainy season.[39–41] Throughout the day, ambient concentrations of $PM_{2.5}$ vary relatively little, except during the morning rush hour when they peak. At the time of our measurements during the morning (7–9 h) and evening (17–19 h) rush hour, the air quality monitoring station at the US consulate located in the city center reported concentrations of 34–74 μg m^{-3} and 24–36 μg m^{-3} (percentiles 25th and 75th), respectively.

2.1 Instrumentation

Table 1 lists the instruments used in this study and summarizes their main characteristics. All sensors were programmed for 1 second readings, with the exception of those measuring pPAHs and ASA which were programmed for 10 second readings. These frequencies are needed to capture the variability that characterizes air pollution in microenvironments affected by vehicular traffic according to our previous studies.[42,43] The instruments' configuration and preparation, as well as corrections applied and data postprocessing are described in those articles. Details are also provided in the accompanying ESI† to this article. In those studies, we evaluated the responses of the instruments for measuring concentrations of $PM_{2.5}$, eBC and PN against reference or scientific grade instrumentation, as a means of indirect calibrations.

The optical aerosol monitor used for measuring $PM_{2.5}$ measures size-segregated mass fraction particle concentration with a laser photometer, whose readings depend on the particle properties, such as size distribution, morphology, and refractive index. To account for local particle properties, the manufacturer recommends adjusting the readings to those of a reference instrument based on gravimetry. Unfortunately, no reference monitor was available for this study, and $PM_{2.5}$ readings were adjusted to previously obtained calibration factors for Singapore's urban atmosphere.[42]

Table 1 Instruments used and their measurement characteristics

Parameter	Instrument	Measuring range (lower threshold)	Accuracy^a	Logging interval (s)	Sampling flow rate (mL min^{-1})	Model	Manufacturer
Size segregated mass-fraction concentration for $PM_{2.5}$	Handheld DustTrak DRX aerosol monitor	0.001–150 mg m^{-3} (1 μg m^{-3})	±0.1% of reading or 0.001 mg m^{-3b}	1	3000	TSI 8534	TSI, Shoreview, MN, USA
Particle number concentration (particles <1 μm diameter)	Handheld condensation particle counter (CPC)	0–1 × 10^5 # cm^{-3} (1 # cm^{-3})	±20% of reading	1	700	TSI 3007	TSI, Shoreview, MN, USA
Active surface area	Handheld diffusion Charger (DC)	0–1000 mm^2 m^{-3} (1 mm^2 m^{-3})	±15% of reading or ±2 mm^2 m^{-3b}	10	1000	DC 2000CE	EcoChem Analytics, League City, TX, USA
Equivalent black carbon	Personal exposure monitor for black carbon	0–1 mg m^{-3} (0.001 μg m^{-3})	±0.1 μg m^{-3}	1	100	AE51	AethLabs, San Francisco, CA, USA
Total pPAHs concentration (particles <1 μm)	Handheld photoelectric aerosol sensor (PAS)	0–4000 ng m^{-3} (1 ng m^{-3})	±15% of reading or ±3 ng m^{-3b}	10	1000	PAS 2000CE	EcoChem Analytics, League City, TX, USA
Carbon monoxide	CO measurer	0–200 ppm (0.50 ppm)	50 ppb	1	NA	T15n	Langan Products Inc., San Francisco, CA, USA
Temperature & relative humidity	HOBO ProV2 logger	−40–70 °C, 0–100% (0.01 °C, 0.05%)	±0.2 °C (0–50 °C), ±2.5% (10–90%) to max ±3.5%	1	NA	U23-001	Onset Computer Corp., Bourne, MA, USA

^{a} As reported by the manufacturer. ^{b} Whichever is greater.

2.2 Combustion origin and average size of the particles

To determine the presence of particles originated by combustion sources other than motorbikes, the ratio obtained from the concurrent measurements of pPAHs and ASA was used as a fingerprint of the type of combustion particles.[35] Following the acronyms of the instruments used to measure both metrics, Photoelectric Charger (PC) for measuring pPAHs and Diffusion Charger (DC) for measuring ASA, the fingerprint is known as PC/DC ratio. It depicts distinctive values according to the type of fuel and combustion. For instance, diesel exhaust particles depict ratios of $\sim$1 ng mm^{-2}, while those from gasoline cars portray ratios of <0.6 ng mm^{-2}.[36] Ratios of >1 ng mm^{-2} are related to high emissions of pPAHs during periods of hard acceleration due to incomplete combustion.[44] Biomass burning particles produce ratios of $\sim$0.30 ng mm^{-2}, while particles from non-combustion sources or those that are already coated by condensable species such as semi-volatile hydrocarbons or molecules of water yield a PC/DC $\approx$ 0.

Similarly, the mean size of the particles in microenvironments heavily impacted by traffic emissions can be estimated as the $D_{\text{Aver},\text{S}}$ computed from the simultaneous but independent measurements of PN and ASA.[35,37] Because >90% of the number of particles within exhaust plumes fall in the nucleation mode (<50 nm) and are formed as the hot exhaust gases cool and condense after passing the emission control devices, the assumption of spherical particles of this approach can be used as a fair approximation to estimate their average size. $D_{\text{Aver},\text{S}}$ represents the diameter of a hypothetical monodisperse particle that has the same ASA as the measured polydisperse particle.

2.3 Masks inward protection efficiency test

The level of protection provided by six distinct types of face masks (see Table 2) was tested at the curbside of two busy roads (sites #2 and #8) by measuring sequentially with and without a mask mounted on a manikin the same particle metrics used to characterize the particle burden at street level (see Fig. 3). In both cases all instruments ran simultaneously. This methodology evaluates the total inward leakage defined as the combination of contaminated air that leaks through a mask from various sources, including face seal, exhalation valve, and penetration through the filter media.[45]

Cloth and surgical masks commonly worn by HCMC commuters were tested including a mask with a layer of charcoal fiber that was introduced into the local market at the time of the study. The test also included an N95 respirator and a reusable valved filtering mask. The N95 respirator has an established filtration ability of at least 95% against particles down to 0.3 μm according to NIOSH testing protocols. Likewise, a 99% filtration efficiency can be expected from the reusable valved filtering mask based on laboratory tests presented by the manufacturer; but as the manufacturer itself clarifies, the mask is not certified by NIOSH. NIOSH does not certify respirators for the public. NIOSH and OSHA (Occupational Safety and Health Administration, US Department of Labor) regulate respirators in workplaces only. Looking for a device for maximum protection we also tested the performance efficiency of a reusable half facepiece air-purifying respirator designed for industrial purposes equipped with both P95 filters against certain oil and non-oil based particles and cartridges against organic vapors approved by NIOSH. For practicality we will refer to it as reusable industrial respirator with the understanding that any type of respirator can be worn in industrial settings.

The masks were mounted on a manikin. Tygon Environmental Sampling Tubing were passed through holes from the back of the manakin to the front of its nose as shown in Fig. 3. Students fitted the masks to the manikin as they do themselves when they are wearing them, tightening the straps and forming the nose clip appropriately. Beyond that, the good fit of the masks on the manakin was not evaluated.

Three sampling sessions were conducted, each lasted for 3 hours. During the total 9 hours of sampling, 27 sets of mask efficiency tests were completed. Each set included a 10 min test period in which the manikin was with a mask, and two periods of the same length before and after without a mask. The data collected during the two mask-free periods was combined in the analysis as a baseline yielding a total of 1200 readings. Likewise, the test period mounting a mask yielded 600 readings. The number of readings in both sampling cases was set to minimize the variability inherent in the particles load in microenvironments affected by vehicular traffic. The average concentrations obtained during the two periods without having the mask mounted and the period with the mask mounted were used to calculate the total inward leakage as described by the formulae included in an inset of Fig. 3.

Particle adsorption on tubing was evaluated by a series of tests in which the instruments ran for 5 min with the tubes mounted on the manikin without a mask and then 5 min unplugging the tubes. These tests did not show statistically significant variations in the readings.

The Anderson–Darling normality test was performed on the data collected. The test showed that the observed variables were not normally distributed but positively skewed. Simple nonparametric tests were used to investigate the equality of results obtained from periods with a mask and periods without a mask. The Mood's median test and Kruskal–Wallis test were conducted ($p \leq 0.05$) to evaluate and verify the significance of the reduction on particles with a mask.

During the measurements, we estimated the percentage of commuters who wore a mask, 67 ± 7% (median ± 1 standard deviation). This percentage is within the statistical range seen for commuters in Hanoi, Vietnam's capital city, where 71% of them worn a cloth mask, 21% a surgical mask, and 8% an N95 respirator or any other high efficiency mask.[46]

3. Results

Heavy loads of particles were seen at all sites. All particle metrics and mixing ratios of CO depicted relatively higher values than anticipated with observed variability as shown in Fig. 4. Table ESI1† provides statistical details of all measured and computed metrics for each monitored site. The few hours of measurement at each site provided only a cross sectional sample of particle abundance. More samples are needed for a complete exposure assessment. This study sheds light on face masks as a means of partial self-protection against particles on

Table 2 Face masks and respirators tested in this study

Type	Manufacturer and model	Material	Expected filtering efficiency[a]	Photograph[b]
Air purifying (reusable industrial) respirator	3M half facepiece respirator series 6000, particulate filter P95 5P71, gas cartridge 6001, NIOSH approved, USA	Silicone, particle fliter, and cartridges against organic vapors	95%	
Reusable valved filtering mask	Vogmask CV with exhalation valve, not NIOSH certified, USA	Four layers: woven microfiber inner layer, highly efficient particle filtering textile, coconut shell derived carbon filter, cotton outer layer	99%	
N95 respirator	3M N95 8210, NIOSH approved TC-84A-007, USA	Filter: propylene; shell: polyester; cover web polyester	95%	
Carbon layer mask	Hoang Thanh GP Extreme activated carbon mask, not NIOSH certified, Vietnam	3 mm activated charcoal fiber and non-woven fabric within a honeycomb cover	95%	
Cloth mask	Generic, Vietnam	Two cotton layers	—	
Surgical mask	Famapro medical face mask, Vietnam	Nonwoven fabric and a melt-blown filtration layer	—	

[a] Against particles around 0.3 μm at a constant flow rate of 85 L min^{-1}, following NIOSH guidelines for testing respirators. Such flow is considered a 'worst-case' scenario since most people breathe at 15–40 L min^{-1}. [b] Photographs of each mask and respirator in a manner that illustrate how they held onto the face are included in the ESI, Fig. ESI2.

the streets. The results of the measurements to characterize HCMC traffic pollution are presented first, followed by the results obtained from masks efficiency tests.

3.1 Particle pollution at curbside

Examining only data from primary roads with severe traffic (*i.e.*, except sites #3 and #5), mean concentrations of $PM_{2.5}$ ranged from 35 to 60 μg m^{-3} (54, 43–63 μg m^{-3}, median, 25th to 75th percentiles considering the full set of data). These values were similar to those reported by the monitoring station at the US consulate during the morning rush hour, but higher than in the evening rush hour. Similarly, Hien *et al.* reported $PM_{2.5}$ concentrations in the same range during the morning rush hour, but lower in the evening rush hour for a road site in the city center.[40] Spikes over 100 μg m^{-3} were frequent, but short in duration. However, the fifth higher percentile recorded at these six road sites exceeded 80 μg m^{-3}. On average, eBC contributed 21% (15–33%, 25th to 75th percentile) to $PM_{2.5}$. Heavy traffic at site #2 led a higher eBC contribution (45% on average). Frequent braking and accelerating increase the emission of carbonaceous aerosols.[47]

Regarding the number of particles, none of the sites significantly exceeded the others, without considering sites #5 and #6, which registered clearly lower concentrations. The low concentration at site #5 was consistent with low readings of other metrics as a consequence of less traffic. Site #6, located in a wide boulevard, also showed lower values, except for $PM_{2.5}$ despite a relative intense traffic flow. Disregarding again sites #3 and #5, mean PN concentrations between 95 and 115 × 10^3 #

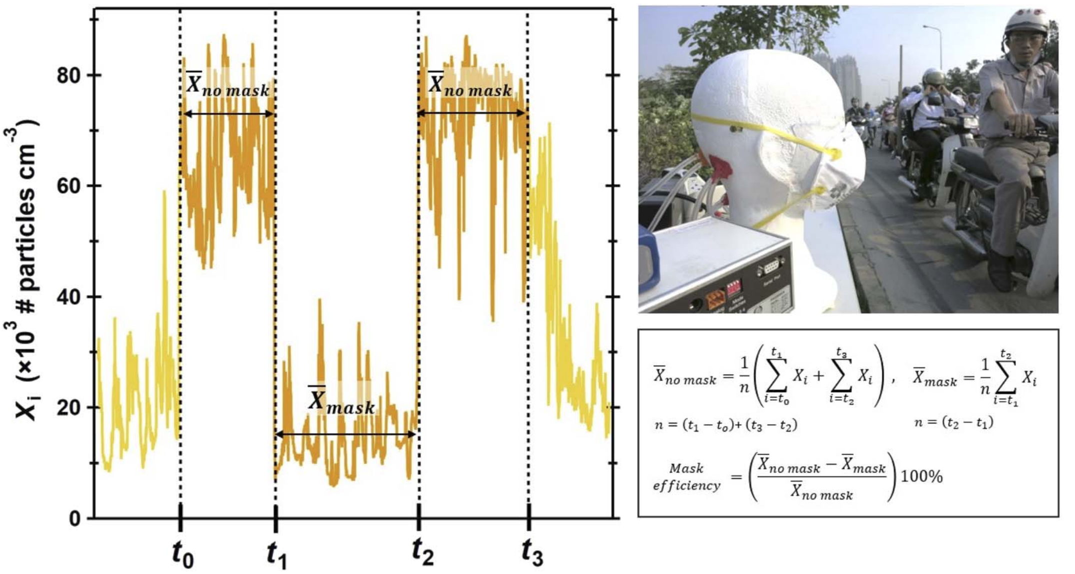

Fig. 3 Mask efficiency tests. The particle instruments were connected by sampling tubes passing from the back to the front (nose area) of the manikin. Each test consisted of three ~10 min sampling periods. The particle concentration without a mask ($X_{no\ mask}$) was obtained in the first and third periods, and the concentration with a mask (X_{mask}) during the second period. The total inward leakages of the masks were calculated based on the formulae outlined.

particles per cm^3 (66–114 × 10^3 # particles per cm^3, 25th to 75th percentiles) were observed in roads with severe traffic.

The levels of pPAHs and ASA followed the patterns observed for $PM_{2.5}$ and eBC, but with higher variability. Both metrics recorded mean concentrations of 114 ng m^{-3} (42–236 ng m^{-3}, 25th to 75th percentiles) and 399 mm^2 m^{-3} (282–549 mm^2 m^{-3}, 25th to 75th percentile), respectively, considering the full set of data collected in major roads.

Carbon monoxide followed in general the pattern observed for particles at each site (*i.e.*, sites with high concentration of particles also showed high concentrations of CO). Considering only roads with heavy traffic, mean concentrations of 7–20 ppm (15, 8–22 ppm, 50th, 25th to 75th percentiles) of CO were observed.

The highest concentrations of $PM_{2.5}$ and eBC were recorded at the underground parking lot of the university (site #3) for 30 min after the end of the evening class session when students pick up their motorbikes. Poor ventilation and an agglomeration of idling motorbikes trying to exit triggered a mean $PM_{2.5}$ concentration of 103 µg m^{-3} (86–122 µg m^{-3}, 25th to 75th percentiles). Braking and acceleration was less intense than in congested roads, thus the fraction of eBC in $PM_{2.5}$ was lower than on those roads, but higher than on roads with fluid traffic, it averaged 31%. The highest levels of pPAHs, ASA and CO were also observed at this site, but not in the number of particles, metric representative of UFP. After ignition, the incomplete combustion and low temperature of catalytic converters cause high emissions of precursor gases (CO, volatile organic compounds, and nitrogen-containing species) and eBC, which yield higher concentrations of accumulation mode particles (>50 nm).[47] The highest emissions of nucleation mode particles are associated with high-speed traffic.[48] Accumulation mode particles are produced in the engines, while an important fraction of nucleation mode particles is formed within the exhaust plume as combustion gases cool and condense, and consist mainly of low volatile organic and sulfur containing compounds. These particles have a short lifetime since they coagulate with other particles, a process that contributes to their rapid decrease in number concentration, but a corresponding increase in the growth of larger particles.[49] The larger size of the particles (57 nm on average as calculated as $D_{Aver,S}$) at this site in comparison to particle sizes observed at the other sites confirms this process.

Particles of 30–40 nm (25th to 75th percentile) were observed on HCMC streets. All particles were essentially larger than 20 nm. This range falls into the nucleation mode and indicates that commuters are exposed to freshly emitted particles (*i.e.*, particles produced in the engine and particles formed in cooling dilution of exhaust) from gasoline engines which are typically in the 20–60 nm range.[50] Contributions from aged particles from sources other than traffic, such as trash burning, industry and regional background, as well as resulting from photochemical reactions cannot be neglected, but they have apparently a minor contribution at street level, since they tend to fall in the accumulation mode.[49]

3.2 Combustion particles fingerprint

The PC/DC ratio of 1.13 obtained at the underground parking lot can be considered as the fingerprint of motorbike exhaust emissions during cold start and slow acceleration, since the readings were not affected by contributions from any other emission source. At cruise speed and fast acceleration, a higher ratio can be expected since engine load, exhaust temperature and exhaust flow increase, resulting in higher emission of pPAHs but not of UFP (*i.e.*, ASA does not increase substantially).[51] In a previous study in Mexico City, we observed higher PC/DC ratios on roads with smooth traffic than on roads with

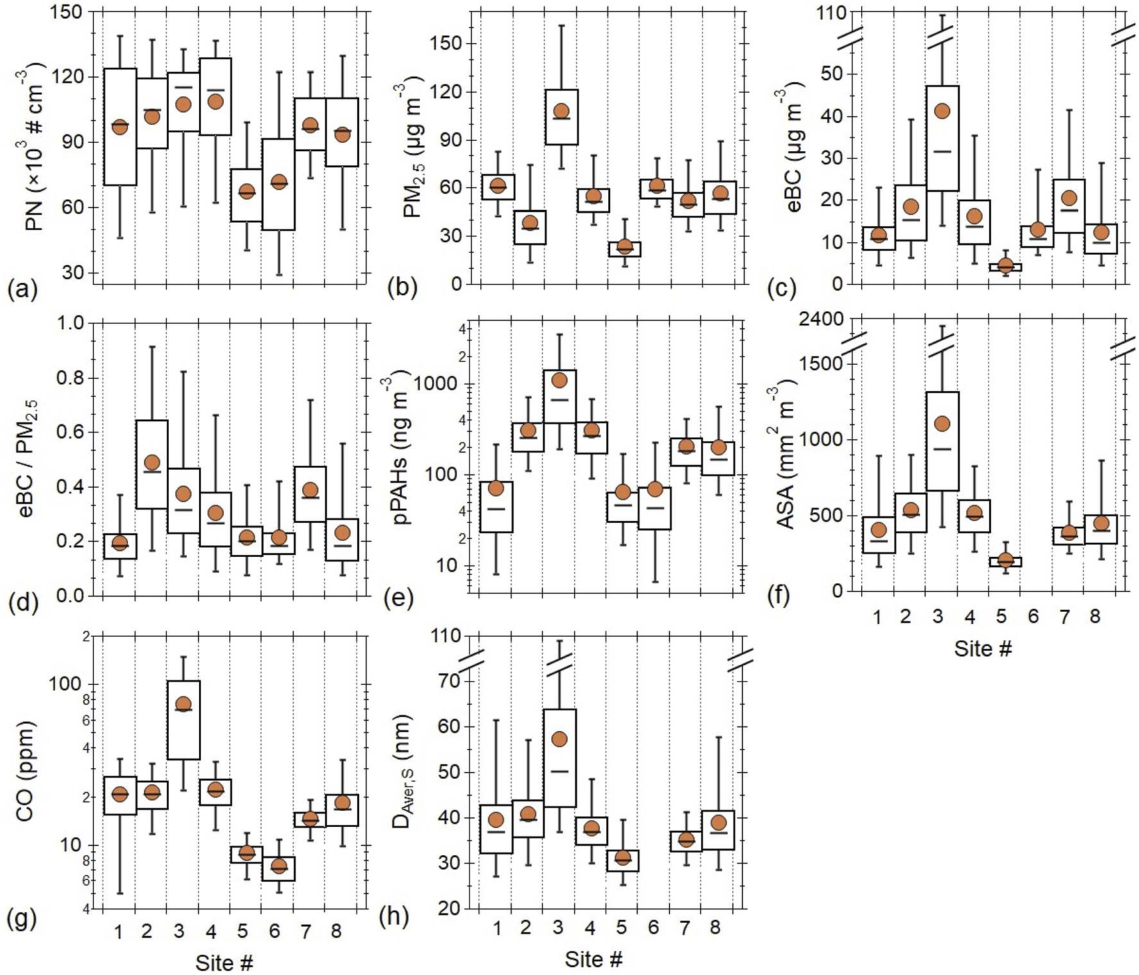

Fig. 4 Box plots for each measured/computed metric during each set of measurements at the eight different sites assessed in this study (see Fig. 1). In each box the middle, top and bottom lines are median value, upper and lower quartile (75th and 25th percentile), respectively, whiskers extend to the 95th and 5th percentiles, and colored dots are arithmetic means.

congested traffic as a consequence of a 4–6 fold increase in pPAHs, but a null or small increase in ASA.[52]

Fig. 5 shows the PC/DC ratios obtained at all sites. Ratios well below the motorbike exhaust fingerprint were observed at all roads, ranging from 0.16 to 0.63, but most often in the 0.5–0.6 range. These lower ratios suggest an important presence of atmospherically aged exhaust particles and particles from sources other than motorbike exhaust. Gasoline cars made minor contributions at all sites, while heavy duty vehicles only at site #2.

Plumes rich in combustion aerosols from other emission sources were evident at sites #1 and #5. The scattering plot of pPAHs *versus* ASA for site #1 depicts two distinctive branches (Fig. 5b), the branch with the steeper slope yields a PC/DC ratio similar to that observed at the other road sites and evidences the presence of motorbike aerosols, while the flatter branch responds to large fumes full of non-photoemitting particles emitted by street food on the grill in stalls using charcoal or firewood as fuel. Ott and Siegmann reported PC/DC ratios of 0.01–0.02 ng mm^{-2} resulting from burning toast and cooking hamburger patties, and 0.20–0.30 ng mm^{-2} for firewood smoke.[36] Depending on the type of firewood and charcoal, their burning may generate particles with high ASA and mass, but low concentration of pPAHs. Similarly, measurements at site #5 yielded three branches (Fig. 5c). A first group of readings yielded a PC/DC ratio close to the ratio obtained at the underground parking lot. The urban canyon formed in this narrow street packed with tall buildings seems to prevent a vigorous ventilation, which helps to accumulate freshly emitted particles. A second group of readings temporally related yielded a low ratio that can be explained by particles related to burning household garbage and gardening waste; while a few but well-defined and consecutive readings formed a third group with a slope as high as that observed for incomplete combustion processes.[53] We hypothesize that in this case it was caused by the burning of plastic waste. Burning of plastic generates abundant amounts of pPAHs, especially of those with 4–6 rings of benzene, which are extremely hazardous due to their carcinogenic and mutagenic nature.[54]

Carbon monoxide can be used as a tracer of traffic pollution.[55] However, the diversity of combustion sources on HCMC

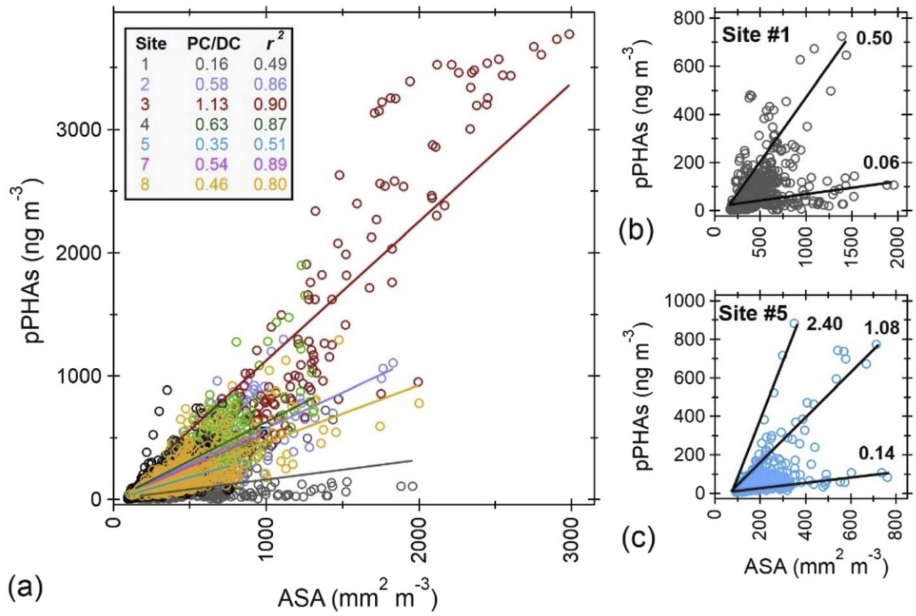

Fig. 5 Relationships between pPAHs concentration and ASA as measured by the PC and DC monitors (*i.e.*, PC/DC ratio) for each set of measurements. Solid lines indicate the best average ratio for all samples collected at each site with their lengths representing the range of observed values. These ratios exclude outlier readings as explained below for panels (b) and (c). The ratios in the box in the upper left corner in panel (a) correspond to slopes of the linear-least-square fits to the measurements, with *y*-intercepts of the lines forced through zero. The coefficients of determination (r^2) indicate the strength of each individual correlation. Panels (b) and (c) show the PC/DC ratios yielded by aerosols from emission sources other than motorbike exhaust, see text for details.

streets reduces its ability to determine contributions from vehicular traffic. To reduce the variability triggered by spiky readings, we evaluated the relationship between each particle metric and CO in 5 min averaging periods. In general, weak correlations ($r^2 < 0.30$) were observed in terms of particle mass (*i.e.*, $PM_{2.5}$, eBC, pPAHs). The number of particles showed moderate correlations ($r^2 = 0.50$–0.70) in six of the eight evaluated sites, while five of the sites showed relatively strong correlations for ASA ($r^2 = 0.60$–0.90). The better correlations for PN and ASA suggest a major contribution of UFP from exhaust emissions. It was not surprising to find high correlations ($r^2 > 0.60$) for all particle metrics in the underground parking lot due to emissions only from motorbikes. However, the highest correlations ($r^2 = 0.70$–0.90) were observed at site #7. The correlation was also strong for the PC/DC ratio ($r^2 = 0.89$), even though its value was lower than the fingerprint determined for motorbikes exhaust. This suggests important contributions from other motorized vehicles at this site. A nearby public bus terminal probably had a significant contribution to the particle load.

3.3 Masks performance efficiency

Fig. 6 shows the effectiveness of six distinct types of face masks for mitigating exposure to traffic particles on HCMC streets. The particle metrics evaluated were $PM_{2.5}$, PN, eBC, pPAHs, and ASA. The results yielded by each individual test are shown in Fig. ESI5.† The difference between using and not using a mask was statistically significant ($p \leq 0.05$) in each one of the experiments for all metrics, except for one experiment in which a surgical mask was tested against PN.

All masks showed a level of effectiveness in protecting against particles. Although some variability was observed between experiments with the same type of mask as a consequence of inherent differences on the mask fitting to the manikin, we were able to elucidate the total inward leakage of each type of mask. For $PM_{2.5}$ the reusable industrial respirator, the reusable valved filtering mask, the N95 respirator, and the carbon-layer sandwiched mask performed similarly, reducing 60% on average the particle load (57–70%, 25th to 75th percentile), while both the cloth mask and the surgical mask showed protection efficiencies of 25% (23–26%, 25th to 75th percentile).

In terms of UFP, the reusable valved filtering mask and the carbon-layer mask showed the best performance against PN with an average efficiency of 80% (77–81%, 25th to 75th percentile). The reusable industrial respirator and the N95 respirator reduced the number of particles by 60% on average (59–65%, 25th to 75th percentile), while the cloth mask and the surgical mask by less than 25% (8–33%, 25th to 75th percentile). As for ASA, the other metric that can be used as a proxy for UFP, only the N95 respirator showed a better performance compared to PN (82%, 73–91%, median, 25th to 75th percentiles). The reusable valved filtering mask and the carbon-layer mask achieved slightly lower performances (74%, 70–75%, median, 25th to 75th percentiles), but not the reusable industrial respirator, whose mean efficiency dropped to 48% (41–50%, 25th to 75th percentile). Both the cloth mask and the surgical mask

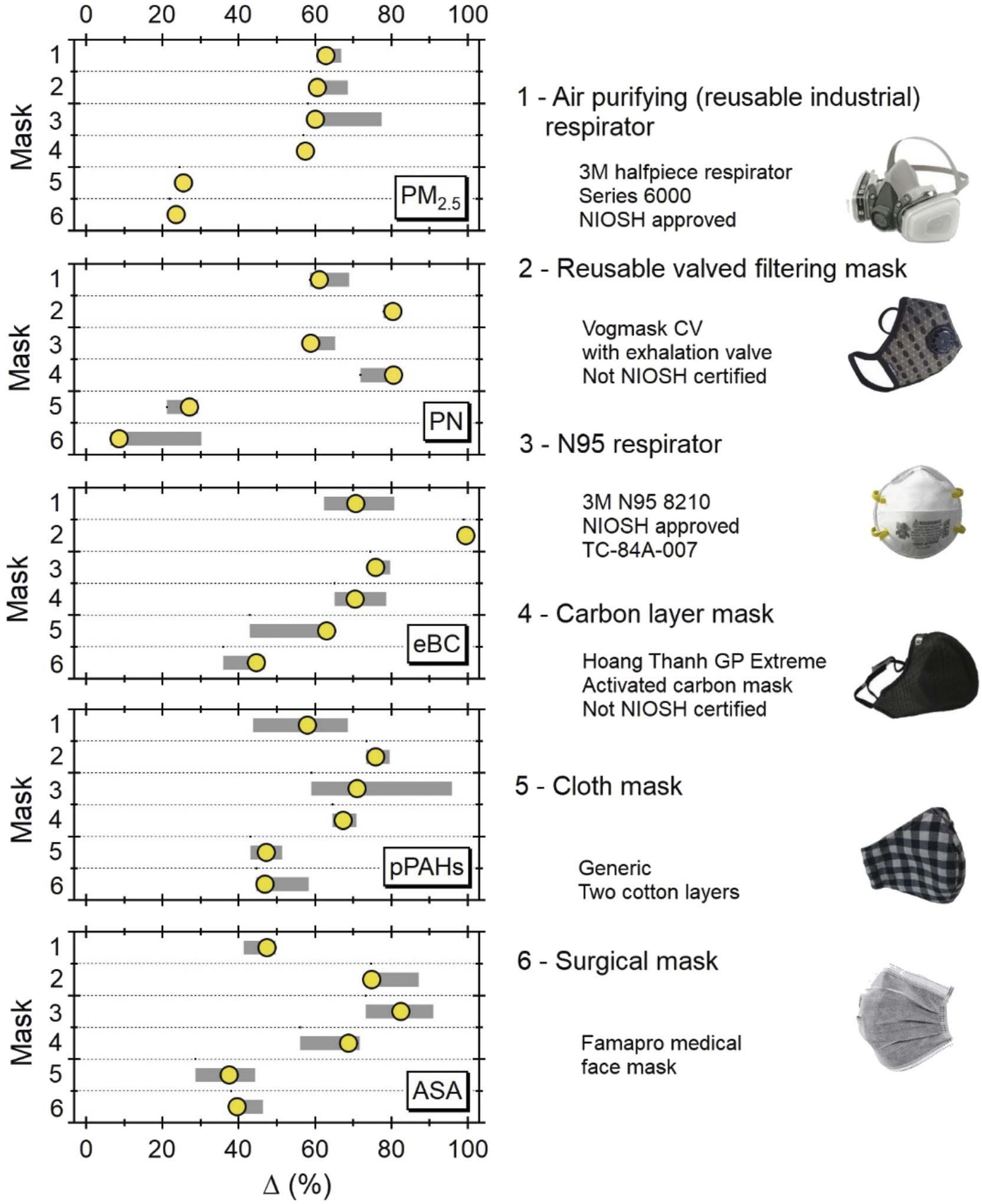

Fig. 6 Masks' performance effectiveness measured as the percentage of particles removed considering the penetration of particles through the filter media and leaks from the face seal and exhalation valve (*i.e.*, total inward leakage). The circles indicate the median and the bars the upper and lower quartile (75th to 25th percentile) of the reduction percentages.

registered mean reductions of 40% (35–46%, 25th to 75th percentile).

The reusable valved filtering mask turned out to be the most effective mask for removing eBC. All experiments showed mean reductions above 98%. The reusable industrial respirator, the N95 respirator and the carbon-layer mask showed to be effective 70–76% on average (64–82%, 25th to 75th percentile). In this case, the cloth mask and the surgical mask worked better as they did against $PM_{2.5}$ and UFP, reducing on average by 45% (36–47%, 25th to 75th percentile) and 63% (43–65%, 25th to 75th percentile) the abundance of eBC, respectively. A similar pattern was observed for pPAHs, except for the reusable valved filtering mask (76%, 73–80%, median, 25th to 75th percentiles), whose performance was closer to that of the N95 respirator and the carbon-layer mask (69%, 61–80%, median, 25th to 75th percentiles). The cloth mask and surgical mask again showed a higher efficiency close to 50% (43–52%, 25th to 75th percentiles).

In terms of particle size, no difference on $D_{Aver,S}$ was observed when using and not using any type of mask.

Differences were not statistically significant for half of the experiments, while the results of the other half were contradictory (see Fig. ESI5†).

According to standard 1910.134 for respiratory protection in workplaces promulgated by OSHA, air-purifying respirators for industrial purposes, as the one tested here must provide an Assigned Protection Factor (APF) of 10, which means they are expected to reduce pollutants concentration by 10 times.[56,57] This standard does not meet for other types of masks. As seen in Fig. 6, the mean effectiveness of the reusable industrial respirator did not meet the expected APF for any particle metric tested. The expected APF can only be meet by a perfect fit of the mask following OSHA protocols.

In two additional experiments the reusable industrial respirator and the N95 respirator were intentionally not perfectly tightened with the aim of testing how much their efficiency decreased. Although the straps of both respirators are elastic, there is some flexibility in the final adjustment. The N95 respirator tested here has a flexible metal piece over the nose bridge for a better seal, which was not given major attention in these tests. For $PM_{2.5}$ no significant difference was observed, but for UFP and eBC there were drops of at least 50% for both masks (see Fig. ESI5†).

4. Discussion

The inward protection efficiencies that we observed are comparable to those reported by laboratory studies based on protocols (somewhat) similar to those of NIOSH using NaCl aerosols as test agent, as well as studies using exposure chambers and manikins as measurement platforms[16–20,24,26,28,29] However, a detailed comparison against such studies is not feasible due to the type of aerosols used as test agent and the environmental conditions in which the measurements were carried out.

Laboratory-based studies that have used diluted exhaust fumes from diesel engines provide better insight than those using NaCl aerosols as test agent. For example, Cherrie *et al.* found performance efficiencies of 71–74% against $PM_{2.5}$, and 42–97% against eBC for non-certified commercially available face masks in the Chinese market.[22] Similar to our results, they observed higher efficiencies for eBC than for $PM_{2.5}$. Shakya *et al.* found efficiencies of 15–57% in terms of particle number for three inexpensive cloth masks, and an efficiency of 79% for a disposable surgical mask.[23] We also observed better performance for surgical masks, but less marked; in both cases the efficiency to remove UFP did not exceed 33%. Penconek *et al.* tested two European certified respirators similar to N95 and N99 respirators (FFP2 and FFP3, respectively).[27] They determined filtration efficiencies (*i.e.*, not counting leaks from the face seal) of 84–89% and 75–86% in terms of particle number for each respirator. The performance of the FFP3 respirator was similar to that of the reusable valved filtering mask we tested, whose manufacturer claims 99% filtration efficiency, but the performance of the FFP2 was superior to the performance we found for the N95 respirator. The mix of aerosols and the experiment design (filtration efficiency *versus* inward protection efficiency) explain the different results. Interestingly, Penconek *et al.* found a higher filtration efficiency for the FFP2 respirator than for the FFP3 respirator.[27] FFP3 respirators generally have a greater packing density and pressure drop in respirator filters than do FFP2 respirators, which results in a more pronounced particle penetration through face leaks than with filter materials.[26]

Probably, the closest comparison would be against the results obtained by Pacitto *et al.*, who evaluated the effectiveness of a number of commercially available masks in Spain.[58] They used instrumentation similar to us, as well as manikins to test the inward protection efficiency of the masks against urban background aerosols. Their results yielded greater variability than that observed in our measurements, and on the contrary, they showed a better performance for $PM_{2.5}$ than for UFP. Their masks showed mean efficiencies of 48% in a range of 14–96% for $PM_{2.5}$, and around 20% in a range of 5–60% for metrics associated with UFP (*i.e.*, PN, ASA and eBC). This contrasting finding can be explained by a much lower load of particles in their measurements, the particles origin and age, and differences in their physical and chemical characteristics.

4.1 Need of wearing face masks

The high load of airborne particles on the streets of HCMC makes the use of face masks a useful preventive measure against the inhalation of traffic pollution. It has been widely documented that people are likely to experience the most exposure to air pollutants during daily commutes.[59–61] When riding a motorbike, commuters are directly exposed to exhaust fumes full of toxic particles and gases (*e.g.*, UFP, eBC, pPAHs, CO), and non-exhaust particles from the abrasion of brakes and tires, dust resuspension, and wear of the road surface (which cover an important fraction of $PM_{2.5}$),[62] as well as pollutants from emission sources other than vehicular traffic, such as workshops and street food stalls placed next to or on the sidewalk. In addition, we need to account for frequent plumes from the burning of trash and gardening waste on the street, as well as of the city ambient air pollution.

The abundance of particles observed in this study is among the highest reported in the literature for traffic-influenced microenvironments.[63,64] Compared to our previous studies using the same set of instruments on streets of Singapore and Mexico City, the particle levels measured on HCMC are at least twice as high.[42,43] They compare to levels measured at busy bus stops of Singapore, where commuters are directly exposed to fumes rich in particles emitted by idling and accelerating buses.[65] To our knowledge, the burden of particles on HCMC streets had not been previously characterized. We only found data on exposure to CO and eBC while commuting by motorbike, and $PM_{2.5}$, CO, and other key pollutants at roadside in Hanoi, Vietnam's city capital. A pioneering work on personal exposure conducted in 2006 reported 16.3 ppm as the average exposure concentration of CO,[66] while a recent study looked at eBC and reported a mean concentration of 35 $\mu g\ m^{-3}$ during rush hours.[67] Tang *et al.* reported mean concentrations of 65 ± 11 $\mu g\ m^{-3}$ of $PM_{2.5}$ and 6 ± 1 ppm of CO during a three-day survey along a busy road at rush hour in 2018.[46] Using as reference data from the latter two studies, it turns out that the levels of CO on HCHC streets (12.3 ± 16.1 ppm, median ± 1 standard deviation) were twice higher, the abundances of $PM_{2.5}$ similar (53 ± 24 μg m^{-3}, median ± 1 standard deviation), and the loads of eBC three times lower (10.9 ± 12.0 μg m^{-3}, median ± 1 standard deviation) than those observed in Hanoi.

The mean concentration of UFP observed at all sites (85 ± 32 # particles per cm^3, median ± 1 standard deviation) falls in the upper range of concentrations typically observed along roads elsewhere (10^4 to 10^5 # particles per cm^3).[49] Fumes coming out from tailpipes can contain up to 10^8 # particles per cm^3 and trigger frequent spikes over 10^5 # particles per cm^3 at roadside,[49,68] as observed in our case. The high sulfur content (500 ppm) in Vietnam's gasoline and the old technology in the vast majority of motorbikes equipped at the best with low quality catalytic converters to meet Euro III emission standards, explain in part the high concentration of UFP on HCMC streets. The high content of sulfur enhances the formation of sulfuric acid which is key in the production of UFP through the exhaust cooling process.[49]

Freshly emitted traffic particles are generally less than 30 nm.[69] The slightly larger particles observed on HCMC streets (37 ± 10 nm, median ± 1 standard deviation) in concert with PC/DC ratios ranging 0.5–0.6 most of the time, below the fingerprint ratio determined for motorbike exhaust (1.13), suggest that in addition to freshly emitted motorbike aerosols, commuters are also exposed to aged particles and particles from sources other than motorbike emissions (*e.g.*, open air incineration of household garbage, and burning of charcoal and firewood to fuel grills and stoves of street food stalls). Contributions from other motorized vehicles are apparently minor.

According to previous studies in environments influenced by traffic, most of the particle mass is found in the accumulation mode (>50 nm) and over 98% of the particles are smaller than 1 μm.[68] This explain why the mass concentrations of $PM_{2.5}$ measured at street level were similar to ambient concentrations recorded by the air quality monitoring station at the US consulate. Therefore, we argue that control and mitigation measures should focus on reducing commuters exposure to UFP rather than exposure to $PM_{2.5}$. In this context, the effectiveness of face masks should focus on the ability to filter UFP, especially in nucleation mode.

The small size and chemical composition of the particles exacerbate the health risk traffic pollution poses to HCMC commuters. Ultrafine particles can be inhaled deeply into the lungs, enter the alveoli, and penetrate biological membranes, enabling them to enter into the bloodstream and reach all organ systems including the brain and nervous system, in addition to aggravate respiratory and cardiovascular diseases.[70–72] They are carriers of large loads of toxic species adsorbed or condensed on their surface. Among these species are the polycyclic aromatic hydrocarbons (PAHs), which induce the generation of free radicals and lead to systematic inflammation through oxidative stress responses, and thereby promote the progression of atherosclerosis, increase blood pressure and myocardial infarction, and worsen some respiratory symptoms.[70,73] Furthermore, PAHs are listed among the most mutagenic and carcinogenic pollutants.[74]

Concentrations of pPAHs were quite variable among the monitored streets, mean concentrations of 40 to 265 ng^{-3} were observed (184 ± 314 ng m^{-3}, median ± 1 standard deviation). These concentrations were in the upper range observed on roads affected by gasoline vehicle emissions, but lower than on roads impacted by diesel exhaust based on studies elsewhere using the same PAS 2000CE monitor.[42,75–78] The high concentration of pPAHs (660 ± 1040 ng m^{-3}, median ± 1 standard deviation) observed in the underground parking lot was similar to that observed in a busy and poorly ventilated underground parking lot in Mexico City.[52] For that parking lot, it was estimated that the amount of pPAHs inhaled under such conditions was roughly equivalent to smoking 2.2 cigarettes per hour.

In terms of health related particle exposure, the most important metric seems to be active surface area according to toxicological studies. Although particle mass and number are important parameters, studies have found that ASA is the metric that explains better the variability in pulmonary inflammation triggered by particle pollution.[79,80] This is particularly true for UFP, since they represent the highest surface area per mass, and because the molecules located on their surface (*e.g.*, PAHs) come into contact with epithelial cells and lung fluid, and trigger oxidative response, making them biologically more relevant. In this context, ASA becomes an important metric to evaluate the effectiveness of wearing face masks as a means of self-protection against traffic particles. Similar to other particle variables, the observed values of ASA on HCMC streets (340 ± 290 mm^2 m^{-3}, median ± 1 standard deviation) were among the highest reported in the literature.[78,81–83] Similar levels are common in roadways of Delhi, India,[84] and microenvironments severely impacted by emissions of heavy-duty diesel vehicles or exhaust fumes of gasoline cars with malfunctioning engines and deficient catalytic converters.[52,85]

4.2 Which mask to wear?

Our results show that the use of any type of mask provides protection against traffic particles. The level of protection depends on the material and design of the masks, and how well they fit (*i.e.*, no gaps between mask and face). The question to answer then is what type of mask is recommended to use for daily commutes.

The level of protection provided by a mask varies depending on the particle size and characteristics. No mask can completely remove all particles. Certified masks such as the N95 respirator comply only under strict fitting conditions with filtering standards developed by occupational safety and health institutions (N95 respirators ensure filtration efficiencies of 95% for NaCl aerosols of 300 nm, USA, NIOSH-42C FR84). Studies have proved that people are often unable to achieve a proper fit, even with masks well-constructed, because of personal judgement on how to wear a mask, as well as anatomical variations such as amount of subcutaneous fat under the chin, and presence of facial hair or stubble.[86,87]

Ignoring the reusable industrial respirator tested here that is not suitable for everyday use on the streets, although a number of reusable elastomeric respirators could be acceptable to the public nowadays, the masks and respirator designed to remove small particles (*i.e.*, reusable valved filtering mask, N95 respirator, and carbon-layer sandwiched mask) offer similar protection under quasi-real exposure conditions on HCMC streets. In terms of particle mass, these masks block on average 60% of particles smaller than 2.5 μm. However, this size threshold is ~70 times the average size of the particles to which commuters are exposed. Therefore, the choice of a mask should be based on metrics that better represent the abundance and characteristics of UFP on the street. The certified N95 respirator, reusable valved filtering mask and the carbon-layer mask remove 60–80% of UFP. The variability depends on the mask fitting and the metric evaluated. The metrics related to the chemical composition of particles (eBC), and the chemical and physical characteristics of their surface (pPAHs and ASA) yield efficiencies closer to 80%, and even above in some cases.

Higher performance efficiency against UFP than against $PM_{2.5}$ may sound inconsistent since aerosols size has major implications for the ability of masks to remove them. Large particles are intercepted by the filter fibers when they are within one particle radius. For small particles, Brownian motion increases the probability a particle impact a fiber, as well as when their inertia becomes sufficiently high, as it is easier for them to collide with the fibers. However, the static charge retained by some materials is the main mechanism that allows masks to filter particles that are much smaller than the pore size of the filtering media.[21,88] We must also bear in mind that masks do not seal perfectly under real wearing conditions and particles can flow through gaps at their edge (see Fig. 8 in Kähler and Hain),[20] so the infiltration of a few large particles can skew the filtering ability in terms of particle mass. Assuming hypothetically the same mass density for both $PM_{2.5}$ and UFP, the penetration of a 2.5 μm particle is equivalent to the penetration of 310 thousand particles of 37 nm (mean $D_{Aver,S}$ observed in this study). This number of particles is 3–4 times the typical number of particles per cubic meter on HCMC streets.

The microscopic structure of the filtering material helps to explain the masks ability to remove UFP. The filtration efficiency of a material depends on its porosity, filter thickness, fiber composition and diameter, and electrostatic properties. The array and distribution of fibers is also relevant. For example, the chaotic arrangement of cotton fibers provides additional opportunities to capture particles as they flow through the fabric, in contrast to polyester fibers that are highly organized and are less effective at trapping particles. However, the combination of layers, like in an N95 respirator, in which nonwoven synthetic fabrics are melted and blown into a weblike fabric enhances the ability of blocking small particles by deposition and diffusion, and electrostatic interactions.[21,89] Hydrophobic polymeric materials, such as some types of polyester and polyurethane increase the filtration ability by retaining the electrostatic charge.[90] It is therefore that masks constructed by multiple layers of different materials, including polymers, offer better protection against particle pollution.[21,90,91] However, a greater packing density in the filter media, as well as a larger number of layers may lead to pressure drop, which results in lower filtration efficiency and higher breathing resistance.[26,27]

Although cloth masks and surgical masks were less effective removing both $PM_{2.5}$ and UFP, they were still capable of providing protection against traffic particles, and they should be used in the absence of one of the other masks. These two masks reduced $PM_{2.5}$ mass concentration by a quarter, and between 25% and 60% the abundance of UFP depending on the metric. The efficiency in terms of number of particles was similar or lower than those reported by studies under controlled laboratory conditions using NaCl aerosols as testing agent.[16,28,29,92,93] As already discussed, our findings on these two types of masks are also consistent with findings obtained from laboratory studies in which diesel aerosols were used as test agent.[22,23,27]

The relative high inward protection efficiency of cloth and surgical masks, especially the latter, against eBC is attributed to the microfiber structure and low or moderate polarity of the materials used for their construction that help to retain particles by physical and chemical adsorption.[94] In agreement with our measurements Chan *et al.* demonstrated that surgical masks made of polypropylene and polyester are effective trapping pPAHs *via* van der Waals forces and electrostatic interactions.[95] We observed removal efficiencies for these toxic particles close to 50% for both surgical and cloth masks.

Simple modifications to improve the fit of surgical and cloth masks, such as adding accessories like a claw-type hair clip increase the tension of the ear loops, or placing three ganged rubber bands or a nylon hosiery sleeve over the mask reduce gaps and enhance the filtering ability.[92] However, not all modifications are practical and are not suitable for daily commutes. In this context, it is important to choose comfortable masks that fit well easily to not deterrent their use. Friction caused by constant pressure and the rubbing of the mask, as well as the retention of heat and humidity from perspiration and exhaled air, may lead to acne and skin infections.[96,97] Therefore, it is important to ensure a good fit without having to overtighten the mask to cause excessive pressure, but it should not be too loose so as not lose the good seal.

Another factor to take into account when selecting a face mask is its cost considering its useful life. Prolonged use of inferior quality or disposable masks can increase the risk of inhaling microplastics that emerge from them.[98] Through an online survey we found that the N95 respirator, and the reusable valved filtering and carbon-layer masks evaluated here cost $1.00, $45.00, and $2.00 USD, respectively on Vietnam's market at the time of writing this. The suppliers of N95 respirators recommend using the same respirator for no more than 6–8 hours as long as it does not get dirty or deformed, which roughly represents one week of use considering that on average a HCMC resident makes 2.5 daily trips of ~20 min each.[4] Hence, a motorbike user will spend $52.00 USD annually on the purchase of N95 respirators. Similarly, the designer of the reusable valved filtering mask recommends replacing it every 3–5 months in extremely poor air quality conditions like those in proximity to wildfires, or every year otherwise. Since the mask is not worn all day, we can assume a yearly replacement, and thus an annual expenditure of $45.00 USD. The manufacturer of the carbon-layer mask recommends replacing it every second week, which means a total expense of $52.00 USD per year.

Since the expense associated with any of these three face masks is essentially the same, the selection of a mask should be based on how comfortable it is, its manufacturing origin, and the environmental impact its disposal might have. The perception of discomfort is related to increase of temperature, moisture and carbon dioxide inside the mask, and the breathing resistance caused by the filtering material and design, that all together are relevant variables to reduce undesirable symptoms, such as fatigue, headache and heat stress.[99–101] Although these ailments are apparently minor when wearing masks for short periods, studied subjects in those studies reported greater discomfort when wearing N95 respirators compared to other types of masks. Then, the reusable valved filtering and carbon-layer masks are more recommendable on this aspect. However, there are also N95 respirators on the market with exhalation valves, and even with attachable micro-ventilators that make them much more comfortable to wear (*e.g.*, AIR$^+$ Smart Mask, **https://www.airplus-family.com/**).[102]

Regarding the origin of the masks, priority should be given to those manufactured in the region to reduce the carbon footprint associated with transportation and distribution following the framework to implement a circular economy model in Southeast Asia.[103] In the same context the use of disposable masks should be discouraged due to the environmental impacts associated with their production and disposal.[104–106]

5. Limitations

The characterization of air pollution on the streets of HCMC had the aim of providing insight on the need to use face masks as a means of self-protection while commuting. It did not aim to provide a comprehensive characterization of the mix of pollutants associated with traffic emissions. Such characterization should be done using online instrumentation that allows to figure out the physical and chemical characteristics of pollutant gases and particles at real time.[107–109]

Regarding the performance efficiency experiments, we tested the total inward leakage of six masks of different type. The masks were selected as representative of the masks most commonly worn by HCMC residents, but they are only a small fraction of the masks available on the market. Future studies should test a larger number of masks, and evaluate the changes in their protection capacity throughout their useful life. The use of a manikin did not allow to assess the impact of heterogeneity in facial geometries on the fit and efficiency of the masks. Future studies should perform efficiency tests on volunteers wearing masks. Similarly, for a more robust assessment of which type of mask provides the best protection without impairing ease of breathing, the resistance to breathe posed by the masks should be evaluated through measurements of the differential pressure drop across them.

Another limitation was the low flow rate at which the different instruments ran (see Table 1) in comparison to common breathing rates while commuting (15–18 L min^{-1}).[110] Lower flow rates could have led to looser fits due to less negative pressure inside the masks and, in turn, have compromised a good seal, this is especially true for masks without exhalation valve.

6. Conclusions

The large load of airborne particles, especially of ultrafine particles in nucleation mode (<50 nm), on the streets of Ho Chi Minh City makes necessary that both motorbike commuters and pedestrians wear face masks as a means of self-protection. The levels of particle pollution observed in this study are among the highest reported in the literature for traffic-influenced microenvironments. Therefore, the use of masks enables to breathe cleaner air. The level of protection depends on how good the design and materials of the masks are to filter particles found on the streets, and how well the masks fit without gaps.

Masks designed to remove small particles and minimize air leakages provide better protection. No mask can completely remove all particles under practical conditions, even certified respirators cannot. People most of the times pay little attention on the proper fitting of the masks. The N95 respirator, the reusable valved filtering mask, and the carbon-layer mask tested in this study are capable of reducing 60–80% the load of particles. Surgical and cloth masks are less effective, they achieve efficiencies of 25–60% depending on the particle metric. The electrostatic properties of the materials used in some masks enhance the ability to filter certain type of particles, for example the latter two types of masks showed an improved ability to filter black carbon and particle-bound polycyclic aromatic hydrocarbons, which are highly toxic species.

In this context, authorities should inform on the need of wearing masks to reduce the health risk posed by traffic particles. The public should count with information on the protection achieved by the masks available on the market under different circumstances, and on their correct use.

The most effective approach to mitigate traffic pollution is to drive less. Though, driving a motorbike is a habit that will be hard to break in HCMC, as well as in many other large cities of Southeast Asia where the urban landscape and lack of road infrastructure make hard to introduce public bus services. It seems unlikely to make bicycles the main mode of transport again as they were in the 1980s. The economic growth in Vietnam makes it affordable nowadays to own at least two motorbikes for many households. Motorbikes have become the solution to mobility problems, they are cheap and fast, and adapt well to Vietnam's roads. Improvements in motorbike engine technology, equipping them with high quality catalytic converters, and the supply of low-sulfur content fuels will reduce exhaust emissions and particle exposure, but it will not solve the problem completely. While a massive switch to electric motorbikes does not occur, policies that encourage the use of effective and comfortable face masks will contribute to improve public health.

Author contributions

EV: Conceptualization, Methodology, Investigation, Data curation, Formal analysis, Writing-original draft. HHH: Investigation, Project Administration, Writing-review. PAD: Investigation, Project Administration, Writing-review. SR: Conceptualization, Methodology, Investigation, Writing-review and editing.

Conflicts of interest

There are no conflicts to declare.

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References

1 S. E. Kim, S. P. Harish, R. Kennedy, X. Jin and J. Urpelainen, Environmental degradation and public opinion: the case of air pollution in Vietnam, *J. Environ. Dev.*, 2020, **29**(2), 196–222.

2 T. Johnson, A. P. J. Mol and S. Yang, Living under the dome: individual strategies against air pollution in Beijing, *Habitat Int.*, 2017, **59**, 110–117.

3 L. P. Wong, H. Alias, N. Aghamohammadi, A. Ghaimi and N. M. N. Sulaiman, Control measures and health effects of air pollution: a survey among public transportation commuters in Malaysia, *Sustainability*, 2017, **9**, 1616.

4 D. T. Huynh and J. Gomez-Ibañez, *The Urban Transport Crisis in Emerging Economies – Vietnam*, ed. D. Pojani and D. Stead, The Urban Book Series, Springer International Publishing, Switzerland, 2017, vol. 13, pp. 267–282.

5 Vietnam General Statistics Office, *Completed results of the 2019 Viet Nam population and housing census*, Statistical Publishing House, 2020, https://www.gso.gov.vn/en/population/, acceded April 2021.

6 M. H. Pham, Ho Chi Minh City, Sustainable urban transport index (SUTI) – 2018. *United Nations – Economic and Social Commission for Asia and the Pacific (ESCAP)*, 2019, https://www.unescap.org/ acceded April 2021.

7 N. Brucker, S. N. do Nascimento, L. Bernardini, M. F. Charão and S. C. Garcia, Biomarkers of exposure, effect, and susceptibility in occupational exposure to traffic-related air pollution: a review, *J. Appl. Toxicol.*, 2020, **40**(6), 722–736.

8 H. Khreis, K. M. Warsow, E. Verlinghieri, A. Guzman, L. Pellecuer, A. Ferreira, I. Jones, E. Heinen, D. Rojas-Rueda, N. Mueller and P. Schepers, The health impacts of traffic-related exposures in urban areas: understanding real effects, underlying driving forces and co-producing future directions, *J. Transp. Health*, 2016, **3**, 249–267.

9 Health Effects Institute, *Systematic review and meta-analysis of selected health effects of long-term exposure to traffic-related air pollution*, Health Effects Institute, Boston, 2022, https://www.healtheffects.org/publication/systematic-review-and-meta-analysis-selected-health-effects-long-term-exposure-traffic, accessed July 2022.

10 L. T. M. Luong, T. N. Dang, N. T. T. Huong, D. Phung, L. K. Tran, D. V. Dung and P. K. Thai, Particulate air pollution in Ho Chi Minh City and risk of hospital admission for acute lower respiratory infection (ALRI) among young children, *Environ. Pollut.*, 2020, **25**, 113424.

11 D. Phung, T. T. Hien, H. N. Linh, L. M. T. Luong, L. Morawska, C. Chu, N. D. Bindh and P. K. Thai, Air pollution and risk of respiratory and cardiovascular hospitalizations in the most populous city in Vietnam, *Sci. Total Environ.*, 2016, **557–558**, 322–330.

12 B. Giechaskiel, A. A. Zardini, T. Lähde, A. Perujo, A. Kontses and L. Ntziachristos, Particulate emissions of Euro 4 motorcycles and sampling considerations, *Atmosphere*, 2019, **10**(7), 421.

13 N. Szymlet, P. Lijewski, Ł. Rymaniak, B. Sokolnicka and M. Siedlecki, Comparative analysis of exhaust emissions from passenger cars and motorcycles, *Combustion Engines*, 2019, **177**(2), 19–22.

14 J. H. Tsai, P. H. Huang and H. L. Chiang, Air pollutants and toxic emissions of various mileage motorcycles for ECE driving cycles, *Atmos. Environ.*, 2017, **153**, 126–134.

15 A. M. Vasic and M. Weilenmann, Comparison of real-world emissions from two-wheelers and passenger cars, *Environ. Sci. Technol.*, 2006, **40**(1), 149–154.

16 W. W. Su, J. Lee, J. Xi and K. Zhang, Investigation of mask efficiency for loose-fitting masks against ultrafine particles and effect of airway deposition efficiency, *Aerosol Air Qual. Res.*, 2022, **22**, 210228.

17 F. Drewnick, J. Pikmann, F. Fachinger, L. Moormann, F. Sprang and S. Borrmann, Aerosol filtration efficiency of household materials for homemade face masks: influence of material properties, particle size, particle electrical charge, face velocity, and leaks, *Aerosol Sci. Technol.*, 2021, **55**(1), 63–79.

18 F. G. Morais, V. K. Sakano, L. N. de Lima, M. A. Franco, D. C. Reis, L. M. Zanchetta, F. Jorge, E. Landulfo,

L. H. Catalani, H. M. Barbosa, V. M. John and P. Artaxo, Filtration efficiency of a large set of COVID-19 face masks commonly used in Brazil, *Aerosol Sci. Technol.*, 2021, **55**, 1028–1041.

19 J. Pan, C. Harb, W. Leng and L. C. Marr, Inward and outward effectiveness of cloth masks, a surgical mask, and a face shield, *Aerosol Sci. Technol.*, 2021, **55**(6), 718–733.

20 C. J. Kähler and R. Hain, Fundamental protective mechanisms of face masks against droplet infections, *J. Aerosol Sci.*, 2020, **148**, 105617.

21 C. D. Zangmeister, J. G. Radney, E. P. Vicenzi and J. L. Weaver, Filtration efficiencies of nanoscale aerosol by cloth mask materials used to slow the spread of SARS-CoV-2, *ACS Nano*, 2020, **14**, 9188–9200.

22 J. W. Cherrie, A. Apsley, H. Cowie, S. Steinle, W. Mueller, C. Lin, C. J. Horwell, A. Sleeuwenhoek and M. Loh, Effectiveness of face masks used to protect Beijing residents against particulate air pollution, *Occup. Environ. Med.*, 2018, **75**(6), 446–452.

23 K. M. Shakya, A. Noyes, R. Kallin and R. E. Peltier, Evaluating the efficacy of cloth facemasks in reducing particulate matter exposure, *J. Expo. Sci. Environ. Epidemiol.*, 2017, **27**, 352–357.

24 N. Serfozo, J. Ondráček, P. Otáhal, M. Lazaridis and Z. Vladimír, Manikin-based size-resolved penetrations of CE-marked filtering facepiece respirators, *J. Occup. Environ. Hyg.*, 2017, **14**(12), 965–974.

25 K. A. Burton, J. L. Whitelaw, A. L. Jones and B. Davies, Efficiency of respirator filter media against diesel particulate matter: a comparison study using two diesel particulate sources, *Ann. Occup. Hyg.*, 2016, **60**(6), 771–779.

26 S. A. Lee, D. C. Hwang, H. Y. Li, C. F. Tsai, C. W. Chen and J. K. Chen, Particle size-selective assessment of protection of European standard FFP respirators and surgical masks against particles-tested with human subjects, *J. Healthc. Eng.*, 2016, 8572493.

27 A. Penconek, P. Drążyk and A. Moskal, Penetration of diesel exhaust particles through commercially available dust half masks, *Ann. Occup. Hyg.*, 2013, **57**(3), 360–373.

28 S. Rengasamy, B. Eimer and R. E. Shaffer, Simple respiratory protection—evaluation of the filtration performance of cloth masks and common fabric materials against 20–1000 nm size particles, *Ann. Occup. Hyg.*, 2010, **54**, 789–798.

29 S. Rengasamy, A. Miller, B. C. Eimer and R. E. Shaffer, Filtration performance of FDA-cleared surgical masks, *J. Int. Soc. Respir. Prot.*, 2009, **26**(3), 54–70.

30 National Institute for Occupational Safety and Health (NIOSH), *Standard respirator testing procedures*, 2020, **https://www.cdc.gov/niosh/npptl/stps/respirator_testing.html**, accessed April 2021.

31 J. Shi, Z. Lin, R. Chen, C. Wang, C. Yang, J. Cai, J. Lin, X. Xu, J. A. Ross, Z. Zhao and H. Kan, Cardiovascular benefits of wearing particulate-filtering respirators: a randomized crossover trial, *Environ. Health Perspect.*, 2017, **125**(2), 175–180.

32 R. J. Laumbach, H. M. Kipen, S. Ko, K. Kelly-McNeil, C. Cepeda, A. Pettit, P. Ohman-Strickland, L. Zhang, J. Zhang, J. Gong, M. Veleeparambil and A. J. Gow, A controlled trial of acute effects of human exposure to traffic particles on pulmonary oxidative stress and heart rate variability, *Part. Fibre Toxicol.*, 2014, **11**, 45.

33 J. P. Langrish, X. Li, S. Wang, M. M. Lee, G. D. Barnes, M. R. Miller, F. R. Cassee, N. A. Boon, K. Donaldson, J. Li, L. Li, N. L. Mills, D. E. Newby and L. Jiang, Reducing personal exposure to particulate air pollution improves cardiovascular health in patients with coronary heart disease, *Environ. Health Perspect.*, 2012, **120**(3), 367–372.

34 J. P. Langrish, N. L. Mills, J. K. Chan, D. L. Leseman, R. J. Aitken, P. H. Fokkens, F. R. Cassee, J. Li, K. Donaldson, D. E. Newby and L. Jiang, Beneficial cardiovascular effects of reducing exposure to particulate air pollution with a simple facemask, *Part. Fibre Toxicol.*, 2009, **6**, 8.

35 N. Bukowiecki, D. B. Kittelson, W. F. Watts, H. Burtscher, E. Weingartner and U. Baltensperger, Real-time characterization of ultrafine and accumulation mode particles in ambient combustion aerosols, *J. Aerosol Sci.*, 2002, **33**(8), 1139–1154.

36 W. R. Ott and H. C. Siegmann, Using multiple continuous fine particle monitors to characterize tobacco, incense, candle, cooking, wood burning, and vehicular sources in indoor, outdoor, and in-transit settings, *Atmos. Environ.*, 2006, **40**(5), 821–843.

37 D. B. Kittelson, J. Johnson, W. Watts, Q. Wei, M. Drayton, D. Paulsen and N. Bukowiecki, Diesel aerosol sampling in the atmosphere, *SAE Tech. Pap.*, 2000, No. 2000-01-2212.

38 Q. B. Ho, H. N. K. Vu, T. T. Nguyen, T. T. H. Nguyen and T. T. T. Nguyen, A combination of bottom-up and top-down approaches for calculating of air emission for developing countries: a case of Ho Chi Minh City, Vietnam, *Air Qual. Atmos. Health*, 2019, **12**, 1059–1072.

39 N. T. N. Nguyen, H. A. Le, T. M. T. Mac, T. T. N. Nguyen, V. H. Pham and Q. H. Bui, Current status of $PM_{2.5}$ pollution and its mitigation in Vietnam, *Glob. Environ. Res.*, 2018, **22**, 73–83.

40 T. T. Hien, N. D. T. Chi, N. T. Nguyen, L. X. Vinh, N. Takenaka and D. H. Huy, Current status of fine particulate matter ($PM_{2.5}$) in Vietnam's most populous city, Ho Chi Minh City, *Aerosol Air Qual. Res.*, 2019, **19**, 2239–2251.

41 N. T. A. Thu, T. V. D. Hang and L. Blume, *Air Quality Report – Air Quality in Vietnam 2018*, Green Innovation and Development Center (GreenID), Hanoi, Vietnam, 2019.

42 S. H. Tan, M. Roth and E. Velasco, Particle exposure and inhaled dose during commuting in Singapore, *Atmos. Environ.*, 2017, **170**, 245–258.

43 E. Velasco, A. Retama, E. Segovia and R. Ramos, Particle exposure and inhaled dose while commuting by public transport in Mexico City, *Atmos. Environ.*, 2019, **219**, 117044.

44 S. Tang, R. Johnson, T. Lanni and W. Webster, Monitoring of PM-bound polycyclic aromatic hydrocarbons from diesel

vehicles by photoelectric aerosol sensor (PAS), *SAE Tech. Pap.,* 2001, 2001-01-3578.

45 International Organization for Standardization (ISO), *Respiratory Protective Devices – Methods of Test and Test Equipment – Part 1: Determination of Inward Leakage (ISO Standard 16900-1:2019)*, Geneva, Switzerland, 2019, **https://www.iso.org/standard/73233.html**.

46 V. T. Tang, N. T. K. Oanh, E. R. Rene and T. N. Binh, Analysis of roadside air pollutant concentrations and potential health risk of exposure in Hanoi, Vietnam, *J. Environ. Sci. Health, Part A*, 2020, **55**(8), 975–988.

47 P. Karjalainen, H. Timonen, E. Saukko, H. Kuuluvainen, S. Saarikoski, P. Aakko-Saksa, T. Murtonen, M. Bloss, M. D. Maso, P. Simonen, E. Ahlberg, B. Svenningsson, W. H. Brune, R. Hillamo, J. Keskinen and T. Rönko, Time-resolved characterization of primary particle emissions and secondary particle formation from a modern gasoline passenger car, *Atmos. Chem. Phys.*, 2016, **16**, 8559–8570.

48 D. B. Kittelson, W. F. Watts and J. P. Johnson, Nanoparticle emissions on Minnesota highways, *Atmos. Environ.*, 2004, **38**, 9–19.

49 T. Rönkkö and H. Timonen, Overview of sources and characteristics of nanoparticles in urban traffic-influenced areas, *J. Alzheimer's Dis.*, 2019, **72**, 15–28.

50 S. J. Harris and M. M. Maricq, Signature size distributions for diesel and gasoline engine exhaust particulate matter, *J. Aerosol Sci.*, 2001, **32**, 749–764.

51 M. Muñoz, R. Haag, P. Honegger, K. Zeyer, J. Mohn, P. Comte, J. Czerwinski and N. V. Heeb, Co-formation and co-release of genotoxic PAHs, alkyl-PAHs, and soot nanoparticles from gasoline direct injection vehicles, *Atmos. Environ.*, 2018, **178**, 242–254.

52 E. Velasco, P. Siegmann and H. C. Siegmann, Exploratory study of particle-bound polycyclic aromatic hydrocarbons in different environments of Mexico City, *Atmos. Environ.*, 2004, **38**, 4957–4968.

53 K. Siegmann, L. Scherrer and H. C. Siegmann, Physical and chemical properties of airborne nanoscale particles and how to measure the impact on human health, *J. Mol. Struct.*, 1999, **458**, 191–201.

54 A. Hoffer, B. Jancsek-Turóczi, Á. Tóth, G. Kiss, A. Naghiu, E. A. Levei, L. Marmureanu, A. Machon and A. Gelencsér, Emission factors for PM_{10} and polycyclic aromatic hydrocarbons (PAHs) from illegal burning of different types of municipal waste in households, *Atmos. Chem. Phys.*, 2020, **20**, 16135–16144.

55 C. Reche, X. Querol, A. Alastuey, M. Viana, J. Pey, T. Moreno, S. Rodríguez, Y. González, R. Fernández-Camacho, J. de la Rosa, J. M. Dall'Osto, A. S. H. Prevot, C. Hueglin, R. M. Harrison and P. Quincey, 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.

56 Occupational Safety and Health Administration (OSHA), *Assigned Protection Factors for the Revised Respiratory Protection Standard, OSHA 3352-02*, US Department of Labor, 2009, **https://www.osha.gov/sites/default/files/publications/3352-APF-respirators.pdf**.

57 Occupational Safety and Health Administration (OSHA), *Occupational Safety and Health Standards: Personal Protective Equipment, OSHA 1910.134*, US Department of Labor, 2019, **https://www.osha.gov/laws-regs/regulations/standardnumber/1910/1910.134**.

58 A. Pacitto, F. Amato, A. Salmatonidis, T. Moreno, A. Alastuey, C. Reche, G. Buonanno, C. Benito and X. Querol, Effectiveness of commercial face masks to reduce personal PM exposure, *Sci. Total Environ.*, 2019, **650**(1), 1582–1590.

59 A. de Nazelle, O. Bode and J. P. Orjuela, Comparison of air pollution exposures in active vs. passive travel modes in Europe: a quantitative review, *Environ. Int.*, 2017, **99**, 151–160.

60 M. Cepeda, J. Schoufour, R. Freak-Poli, C. M. Koolhaas, K. Dhana, W. M. Bramer and O. H. Franco, Levels of ambient air pollution according to mode of transport: a systematic review, *Lancet Public Health*, 2017, **2**(1), 23–34.

61 L. D. Knibbs, T. Cole-Hunter and L. Morawska, A review of commuter exposure to ultrafine particles and its health effects, *Atmos. Environ.*, 2011, **45**, 2611–2622.

62 R. M. Harrison, J. Allan, D. Carruthers, M. R. Heal, A. C. Lewis, B. Marner and A. Williams, Non-exhaust vehicle emissions of particulate matter and VOC from road traffic: a review, *Atmos. Environ.*, 2021, **262**, 118592.

63 P. Kumar, L. Morawska, W. Birmili, P. Paasonen, M. Hu, M. Kulmala, R. M. Harrison, L. Norford and R. Britter, Ultrafine particles in cities, *Environ. Int.*, 2014, **66**, 1–10.

64 P. Kumar, A. P. Patton, J. L. Durant and H. C. Frey, A review of factors impacting exposure to $PM_{2.5}$, ultrafine particles, and black carbon in Asian transport microenvironments, *Atmos. Environ.*, 2018, **187**, 301–316.

65 E. Velasco and S. H. Tan, Particles exposure while sitting at bus stops of hot and humid Singapore, *Atmos. Environ.*, 2016, **142**, 251–263.

66 S. Saksena, T. N. Quang, T. Nguyen, P. N. Dang and P. Flachsbart, Commuters' exposure to particulate matter and carbon monoxide in Hanoi, Vietnam, *Transp. Res. D: Transp. Environ.*, 2008, **13**, 206–211.

67 T. N. Quang, N. T. Hue, L. K. Tran, T. H. Phi, L. Morawska and P. K. Thai, Motorcyclists have much higher exposure to black carbon compared to other commuters in traffic of Hanoi, Vietnam, *Atmos. Environ.*, 2021, **245**, 118029.

68 L. Morawska, Z. Ristovski, E. R. Jayaratne, D. U. Keogh and X. Ling, Ambient nano and ultrafine particles from motor vehicle emissions: characteristics, ambient processing, and implications on human exposure, *Atmos. Environ.*, 2008, **42**, 8113–8138.

69 T. Wu and B. E. Boor, Urban aerosol size distributions: a global perspective, *Atmos. Chem. Phys.*, 2021, **21**, 8883–8914.

70 G. D. Leikauf, S. H. Kim and A. S. Jang, Mechanisms of ultrafine particle-induced respiratory health effects, *Exp. Mol. Med.*, 2020, **52**, 329–337.

71 S. Ohlwein, R. Kappeler, M. K. Joss, N. Künzli and B. Hoffmann, Health effects of ultrafine particles: a systematic literature review update of epidemiological evidence, *Int. J. Publ. Health*, 2019, **64**, 547–559.

72 R. J. Delfino, C. Sioutas and S. Malik, Potential role of ultrafine particles in associations between airborne particle mass and cardiovascular health, *Environ. Health Perspect.*, 2005, **113**, 934–946.

73 A. E. Nel, D. Diaz-Sanchez and N. Li, The role of particulate pollutants in pulmonary inflammation and asthma: evidence for the involvement of organic chemicals and oxidative stress, *Curr. Opin. Pulm. Med.*, 2001, **7**, 20–26.

74 K. Ravindra, R. Sokhi and R. Van Grieken, Atmospheric polycyclic hydrocarbons: source attribution, emission factors and regulation, *Atmos. Environ.*, 2008, **42**, 2895–2921.

75 D. Houston, D. Wu and G. Jaimes, Particle-bound polycyclic aromatic hydrocarbon concentrations on transportation microenvironments, *Atmos. Environ.*, 2013, **71**, 148–157.

76 Y. Cheng, K. F. Ho, W. J. Wu, S. S. H. Ho, S. C. Lee, Y. Huang, Y. W. Zhang, P. S. Yau, Y. Gao and C. S. Chan, Real-time characterization of particle-bound polycyclic aromatic hydrocarbons at a heavily trafficked roadside site, *Aerosol Air Qual. Res.*, 2012, **12**, 1181–1188.

77 M. V. Brachtl, J. L. Durant, C. P. Perez, J. Oviedo, F. Sempertegui, E. N. Naumova and J. K. Griffiths, Spatial and temporal variations and mobile source emissions of polycyclic aromatic hydrocarbons in Quito, Ecuador, *Environ. Pollut.*, 2009, **157**, 528–536.

78 P. Siegmann, F. J. Acevedo, K. Siegmann and S. Maldonado-Bascón, A probabilistic source attribution model for nanoparticles in air suspension applied on the main roads of Madrid and Mexico City, *Atmos. Environ.*, 2008, **42**, 3937–3948.

79 O. Schmid and T. Stoeger, Surface area is the biologically most effective dose metric for acute nanoparticle toxicity in the lung, *J. Aerosol Sci.*, 2016, **99**, 133–143.

80 G. Oberdörster, E. Oberdörster and J. Oberdörster, Nanotoxicology: an emerging discipline evolving from studies of ultrafine particles, *Environ. Health Perspect.*, 2005, **113**, 823–839.

81 P. K. Chang, S. M. Griffith, H. C. Chuang, K. J. Chuang, Y. H. Wang, K. E. Chang and T. C. Hsiao, Particulate matter in a motorcycle-dominated urban area: source apportionment and cancer risk of lung deposited surface area (LDSA) concentrations, *J. Hazard. Mater.*, 2022, 128188.

82 J. Kuula, H. Kuuluvainen, J. V. Niemi, E. Saukko, H. Portin, A. Kousa, M. Aurela, T. Rönkkö and H. Timonen, Long-term sensor measurements of lung deposited surface area of particulate matter emitted from local vehicular and residential wood combustion sources, *Aerosol Sci. Technol.*, 2019, **54**(2), 190–202.

83 C. Reche, M. Viana, M. Brines, N. Pérez, D. Beddows, A. Alastuey and X. Querol, Determinants of aerosol lung-deposited surface area variation in an urban environment, *Sci. Total Environ.*, 2015, **517**, 38–47.

84 L. Salo, A. Hyvärinen, P. Jalava, K. Teinilä, R. K. Hooda, A. Datta, S. Saarikoski, H. Lintusaari, T. Lepistö, S. Martikainen, A. Rostedt, V. P. Sharma, M. H. Rhaman, S. Subudhi, E. Asmi, J. V. Niemi, H. Lihavainen, B. Lal, J. Keskinen, H. Kuuluvainen, H. Timonen and T. Rönkkö, The characteristics and size of lung-depositing particles vary significantly between high and low pollution traffic environments, *Atmos. Environ.*, 2021, **255**, 118421.

85 L. C. Marr, L. A. Grogan, H. Wöhrnschimmel, L. T. Molina, M. J. Molina, T. J. Smith and E. Garshick, Vehicle traffic as a source of particulate polycyclic aromatic hydrocarbon exposure in the Mexico City Metropolitan Area, *Environ. Sci. Technol.*, 2004, **38**, 2584–2592.

86 E. O'Kelly, A. Arora, S. Pirog, J. Ward and P. J. Clarkson, Comparing the fit of N95, KN95, surgical, and cloth face masks and assessing the accuracy of fit checking, *PLoS One*, 2020, **16**(1), e0245688.

87 Z. Zhuang, M. Bergman, E. Brochu, A. Palmiero, G. Niezgoda, X. He, R. Roberge and R. Shaffer, Temporal changes in filtering-facepiece respirator fit, *J. Occup. Environ. Hyg.*, 2016, **13**(4), 265–274.

88 L. Brosseau and R. B. Ann, *N95 respirators and surgical masks. NIOSH Science Blog*, Centers for Disease Control and Prevention, 2009, https://blogs.cdc.gov/niosh-science-blog/2009/10/14/n95/, acceded August 2021.

89 R. Allain, The physics of the N95 face mask, *Wired*, January 2022, https://www.wired.com/, acceded February 2022.

90 M. Zhao, L. Liao, W. Xiao, X. Yu, H. Wang, Q. Wang, Y. L. Lin, F. S. Kilinc-Balci, A. Price, L. Chu and M. C. Chu, Household materials selection for homemade cloth face coverings and their filtration efficiency enhancement with triboelectric charging, *Nano Lett.*, 2020, **20**, 5544–5552.

91 A. Konda, A. Prakash, G. A. Moss, M. Schmoldt, G. D. Grant and S. Guha, Aerosol filtration efficiency of common fabrics used in respiratory cloth masks, *ACS Nano*, 2020, **14**, 6339–6347.

92 P. W. Clapp, E. E. Sickbert-Bennett, J. M. Samet, J. Berntsen, K. L. Zeman, D. J. Anderson, D. J. Weber and W. D. Bennett, Evaluation of cloth masks and modified procedure masks as personal protective equipment for the public during the COVID-19 pandemic, *JAMA Intern. Med.*, 2021, **181**(4), 463–469.

93 E. E. Sickbert-Bennett, J. M. Samet, P. W. Clapp, H. Chen, J. Berntsen, K. L. Zeman, H. Tong, D. J. Weber and W. D. Bennett, Filtration efficiency of hospital face mask alternatives available for use during the COVID-19 pandemic, *JAMA Intern. Med.*, 2020, **180**(12), 1607–1612.

94 K. Jayaraman, M. Kotaki, Y. Zhang, X. Mo and S. Ramakrishna, Recent advances in polymer nanofibers, *J. Nanosci. Nanotechnol.*, 2004, **4**, 52–65.

95 W. Chan, L. Jin, Z. Sun, S. M. Griffith and J. Z. Yu, Fabric masks as a personal dosimeter for quantifying exposure to airborne polycyclic aromatic hydrocarbons, *Environ. Sci. Technol.*, 2021, **55**, 5128–5135.

96 W. Hua, Y. Zuo, R. Wan, L. Xiong, J. Tang, L. Zou, X. Shu and L. Li, Short-term skin reactions following use of N95

respirators and medical masks, *Contact Dermatitis*, 2020, **83**, 115–121.

97 G. Damiani, L. C. Gironi, A. Grada, K. Kridin, R. Finelli, A. Buja, N. L. Bragazzi, P. D. M. Pigatto and P. Savoia, COVID-19 related masks increase severity of both acne (maskne) and rosacea (mask rosacea): multi-center, real-life, telemedical, and observational prospective study, *Dermatol. Ther.*, 2021, **34**(2), 14848.

98 L. Li, X. Zhao, Z. Li and K. Song, COVID-19: performance study of microplastic inhalation risk posed by wearing masks, *J. Hazard. Mater.*, 2021, **411**, 124955.

99 O. Geiss, Effect of wearing face masks on the carbon dioxide concentration in the breathing zone, *Aerosol Air Qual. Res.*, 2020, **21**(2), 200403.

100 E. C. H. Lim, R. C. S. Seet, K. H. Lee, E. P. V. Wilder-Smith, B. Y. S. Chuah and B. K. C. Ong, Headaches and the N95 face-mask amongst healthcare providers, *Acta Neurol. Scand.*, 2006, **113**, 199–202.

101 Y. Li, H. Tokura, Y. P. Guo, A. S. W. Wong, T. Wong, J. Chung and E. Newton, Effects of wearing N95 and surgical facemasks on heart rate, thermal stress, and subjective sensations, *Int. Arch. Occup. Environ. Health*, 2005, **78**, 501–509.

102 E. Birgersson, E. H. Tang, W. L. J. Lee and K. J. Sak, Reduction of carbon dioxide in filtering facepiece respirators with an active-venting system: a computational study, *PLoS One*, 2015, **10**(6), e0130306.

103 ASEAN Secretariat. *Framework for Circular Economy for the ASEAN Economic Community*, ASEAN Secretariat, Jakarta, Indonesia, 2021, https://asean.org/.

104 F. P. de Albuquerque, M. Dhadwal, W. Dastyar, S. M. M. Azizi, I. Karidio, H. Zaman and B. R. Dhar, Fate of disposable face masks in high-solids anaerobic digestion: experimental observations and review of potential environmental implications, *Case Stud. Chem. Environ. Eng.*, 2021, **3**, 100082.

105 M. Shen, Z. Zeng, B. Song, H. Yi, T. Hu, Y. Zhang, G. Zeng and R. Xiao, Neglected microplastics pollution in global COVID-19: disposable surgical masks, *Sci. Total Environ.*, 2021, **790**, 148130.

106 M. Schmutz, R. Hischier, T. Batt, P. Wick, B. Nowack, P. Wäger and C. Som, Cotton and surgical masks - what ecological factors are relevant for their sustainability?, *Sustainability*, 2020, **12**(24), 10245.

107 R. U. Shah, E. S. Robinson, P. Gu, A. L. Robinson, J. S. Apte and A. A. Presto, High-spatial-resolution mapping and source apportionment of aerosol composition in Oakland, California, using mobile aerosol mass spectrometry, *Atmos. Chem. Phys.*, 2018, **18**(22), 16325–16344.

108 J. Enroth, S. Saarikoski, J. Niemi, A. Kousa, I. Ježek, G. Močnik, S. Carbone, H. Kuuluvainen, T. Rönkkö, R. Hillamo and L. Pirjola, Chemical and physical characterization of traffic particles in four different highway environments in the Helsinki metropolitan area, *Atmos. Chem. Phys.*, 2016, **16**, 5497–5512.

109 C. Mohr, P. F. DeCarlo, M. F. Heringa, R. Chirico, R. Richter, M. Crippa, X. Querol, U. Baltensperger and A. S. Prévôt, Spatial variation of aerosol chemical composition and organic components identified by positive matrix factorization in the Barcelona region, *Environ. Sci. Technol.*, 2015, **9**(17), 10421–10430.

110 E. Dons, M. Laeremans, J. P. Orjuela, I. Avila-Palencia, G. Carrasco-Turigas, T. Cole-Hunter, E. Anaya-Boig, A. Standaert, P. de Boever, T. Nawrot, T. Götschi, A. de Nazelle, M. Nieuwenhuijsen and L. I. Panis, Wearable sensors for personal monitoring and estimation of inhaled traffic-related air pollution: evaluation of methods, *Environ. Sci. Technol.*, 2017, **51**(3), 1859–1867.

