

1 Online characterization of primary and secondary emissions of 2 particulate matter and acidic molecules from a modern fleet of city 3 buses

4 Liyuan Zhou^{1,2#}, Qianyun Liu^{2,a#}, Christian M. Salvador^{3,b}, Michael Le Breton^{3,c}, Mattias Hallquist³, Jian
5 Zhen Yu⁴ and Chak K. Chan^{1,2*}, Åsa M. Hallquist^{5*}

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7 ¹ Division of Physical Sciences and Engineering, King Abdullah University of Science and Technology, Thuwal, Saudi Arabia

8 ² School of Energy and Environment, City University of Hong Kong, Hong Kong SAR, China

9 ³ Department of Chemistry and Molecular Biology, University of Gothenburg, Gothenburg, Sweden

10 ⁴ Division of Environment and Sustainability, Hong Kong University of Science and Technology, Hong Kong, China

11 ⁵ IVL Swedish Environmental Research Institute, Gothenburg, Sweden

12 ^anow at: RELX Science Center, Shenzhen RELX Tech. Co., Ltd., Shenzhen, China

13 ^bnow at: Environmental Sciences Division, Oak Ridge National Laboratory, Oak Ridge, TN 37830, USA

14 ^cnow at: FEV Sverige AB, Gothenburg, Sweden

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17 #The authors contribute equally.

18 *Correspondence to:* Åsa M Hallquist (asa.hallquist@ivl.se); Chak K. Chan (chak.chan@kaust.edu.sa)

19

20 **Abstract.** The potential impact of transitioning from conventional fossil fuel to a non-fossil fuel vehicle fleet was investigated
21 by measuring primary emissions via extractive sampling of bus plumes and assessing secondary mass formation using a
22 Gothenburg Potential Aerosol Mass (Go:PAM) reactor from 76 in-use transit buses. Online chemical characterization of
23 gaseous and particulate emissions from these buses was conducted using a chemical ionization mass spectrometry (CIMS)
24 with acetate as the reagent ion, coupled with a filter inlet for gases and aerosols (FIGAERO). Acetate reagent ion chemistry
25 selectively ionizes acidic compounds, including organic and inorganic acids, as well as nitrated and sulfated organics. A
26 significant reduction (48-98%) in fresh particle emissions was observed in buses utilizing compressed natural gas (CNG),
27 biodiesels like rapeseed methyl ester (RME) and hydrotreated vegetable oil (HVO), as well as hybrid-electric HVO (HVO_{HEV}),
28 compared to diesel (DSL) buses. However, secondary particle formation from photooxidation of emissions was substantial
29 across all fuel types. The median ratio of particle mass emission factors of aged to fresh emissions increased in the following
30 order: DSL buses at 4.0, HVO buses at 6.7, HVO_{HEV} buses at 10.5, RME buses at 10.8, and CNG buses at 84. Of the compounds
31 that can be identified by CIMS, fresh gaseous emissions from all Euro V/EEV buses, regardless of fuel type, were dominated
32 by nitrogen-containing compounds such as nitrous acid (HONO), nitric acid (HNO₃), and isocyanic acid (HNCO), alongside
33 small monoacids (C₁-C₃). Notably, the emission of nitrogen-containing compounds was notably lower in Euro VI buses
34 equipped with more advanced emission control technologies. Secondary gaseous organic acids correlated strongly with

35 gaseous HNO₃ signals ($R^2= 0.85-0.99$) in Go:PAM, but their moderate to weak correlations with post-photooxidation
36 secondary particle mass suggest they are not reliable tracers for secondary organic aerosol formation from bus exhaust. Our
37 study highlights that non-regulated compounds and secondary pollutant formation, not currently addressed in legislation, are
38 crucial considerations in the evaluation of environmental impacts of future fuel and engine technology shifts.

39 **1. Introduction**

40 Air pollution remains a critical global issue, posing significant threats to both human health and the environment. Despite
41 substantial progress in reducing emissions from major sources like industry, energy production, households, transportation,
42 and agriculture, the worldwide achievement of air quality targets continues to be a daunting challenge. Notably, the road
43 transport sector, particularly in urban environments, significantly contributes to the emissions of nitrogen oxides (NO_x) and
44 particulate matter (PM), impacting the health of individuals in densely populated regions. In tandem with these concerns,
45 efforts to combat climate change have spurred an increase in the adoption of renewable energy sources within the transportation
46 sector. Biodiesel has risen as the most prevalent renewable fuel, followed by biogas and ED95 ethanol (Guerreiro et al., 2014).
47 Moreover, numerous cities are progressively integrating hybrid-electric and electric vehicles into their public transport fleets,
48 aiming to reduce emissions.

49

50 Emissions from vehicles, especially buses, exhibit considerable variability. They are influenced by fuel type, engine design,
51 operational conditions, emission after-treatment technologies and maintenance (Pirjola et al., 2016; Zhao et al., 2018; Watne
52 et al., 2018; Liu et al., 2019a; Zhou et al., 2020). While diesel (DSL) buses are common, there is an increasing trend towards
53 the use of alternative fuels such as compressed natural gas (CNG), rapeseed methyl ester (RME), and hydrotreated vegetable
54 oil (HVO). These alternative fuels offer several benefits, including reduced PM emissions, particularly soot, and lower levels
55 of carbon monoxide (CO) and total hydrocarbons (THC) (Pflaum et al., 2010; Hassaneen et al., 2012; Liu et al., 2019a).
56 However, the efficacy of RME and HVO in diminishing NO_x emissions can be inconsistent (Pirjola et al., 2016; Liu et al.,
57 2019a); and CNG buses exhibit considerable variability in particle number (PN) emissions (Watne et al., 2018). In Sweden,
58 approximately 23% of the fuel mix of the transport sector in 2020 comprised renewable fuels, with HVO accounting for over
59 half of this proportion (Vourliotakis and Platsakis, 2022; Energimyndigheten, 2021). Emission control strategies, such as
60 aftertreatment systems including diesel particulate filters (DPFs) and selective catalytic reduction (SCR) systems, have been
61 implemented to mitigate pollutant emissions from vehicles. These systems have shown significant efficacy in reducing PM
62 and NO_x emissions respectively, though their performance can vary under different operational conditions.

63

64 Accurately determining vehicle emission factors (EFs) is crucial for devising and implementing effective air quality
65 policies (Fitzmaurice and Cohen, 2022). Methods such as chassis dynamometer tests, on-board measurements with portable
66 emission measurement systems (PEMS), and on-road vehicle chasing experiments have been employed to assess emissions

67 from various types of vehicles (Kwak et al., 2014; Jezek et al., 2015; Pirjola et al., 2016). Chassis dynamometer tests offer
68 high repeatability over standard driving cycles but may not reflect real-world driving conditions or fleet maintenance levels.
69 There are also challenges in accurately replicating real-world dilution effects (Vogt et al., 2003; Kuittinen et al., 2021). On-
70 board measurements with PEMS provide data under a wide range of operating conditions, yet like dynamometers, they may
71 not realistically mimic ambient dilution processes (Giechaskiel et al., 2015; Wang et al., 2020). On-road vehicle chasing
72 experiments involve following individual vehicles with a mobile laboratory to capture the exhaust plumes, providing insights
73 into realistic dilution processes from the tailpipe to ambient air, though these experiments often require a test track to ensure
74 traffic safety (Wang et al., 2020; Tong et al., 2022). All three methods are limited by small sample sizes, which constrain
75 understanding of the real emission characteristics of vehicle fleets. Alternatively, roadside or near-road measurements provide
76 the ability to monitor emissions from a large number of vehicles under actual driving conditions within a short
77 timeframe (Hallquist et al., 2013; Watne et al., 2018; Liu et al., 2019a), which is particularly important for assessing exposure
78 risks to pedestrians and bus passengers. However, this method is limited by its inability to monitor specific engines or
79 operational conditions, such as varying engine speeds and loads. Integrating results from diverse methodologies would ideally
80 yield a comprehensive understanding of emissions from vehicle transport systems.

81

82 In a prior study, we conducted roadside point measurements and reported EFs for general air pollutants such as PM, NO_x, CO,
83 and THC from individual buses during stop-and-go operations at a bus stop in Gothenburg, Sweden (Liu et al., 2019a). Our
84 findings showed that hybrid buses, when using their combustion engines to accelerate from a standstill at bus stops, tended to
85 emit higher particle numbers (PN) than traditional DSL buses, likely due to their relatively smaller engines. Expanding on our
86 prior findings, it is important to acknowledge that primary emissions are not the only way in which engine emissions impact
87 air quality. Emissions from engine exhaust can contribute to secondary particles through oxidation of gas-phase species,
88 primarily via functionalization reactions, yielding lower-volatility products (Hallquist et al., 2009; Kroll et al., 2009).
89 Laboratory studies have demonstrated that secondary organic aerosols (SOA) produced from diluted vehicle exhaust frequently
90 exceed the levels of primary organic aerosols (POA) in less than one day of atmospheric equivalent aging (Chirico et al., 2010;
91 Nordin et al., 2013; Platt et al., 2013; Gordon et al., 2014b; Liu et al., 2015). Oxidation flow reactors (OFRs) enable the
92 simulation of several days of atmospheric aging in a few minutes, with minimized wall effects compared to traditional smog
93 chamber experiments (Palm et al., 2016; Bruns et al., 2015). OFRs have been extensively employed to assess the SOA
94 formation potential of ambient air and emissions from diverse sources, including motor exhausts (Tkacik et al., 2014; Bruns
95 et al., 2015; Simonen et al., 2017; Watne et al., 2018; Liu et al., 2019b; Kuittinen et al., 2021; Zhou et al., 2021; Liao et al.,
96 2021; Yao et al., 2022). In real-world traffic scenarios, the rapid response capabilities and convenient deployment of OFRs,
97 coupled with roadside point measurements, provide a robust method for evaluating emissions from a significant number of
98 vehicles. This approach effectively captures the considerable variability among individual vehicles within a fleet, offering a
99 comprehensive view of emissions under actual driving conditions (Watne et al., 2018; Zhou et al., 2021), although it may not

100 encompass an extensive range of engine operations as setups that integrate OFRs with chassis dynamometer tests (Kuittinen
101 et al., 2021).

102

103 Primary emissions can also be oxidized to higher-volatility products through fragmentation reactions, potentially producing
104 carboxylic acids (Friedman et al., 2017). Engine exhaust is a recognized primary source of organic and inorganic acids in urban
105 environments (Kawamura et al., 1985; Kawamura and Kaplan, 1987; Kirchstetter et al., 1996; Wentzell et al., 2013; Friedman
106 et al., 2017). Monocarboxylic acids are produced by both diesel and spark-ignited engines (Kawamura et al., 1985; Zervas et
107 al., 2001a; Zervas et al., 2001b; Crisp et al., 2014). Recent studies have identified gaseous dicarboxylic acids in diesel
108 exhaust (Arnold et al., 2012), compounds likely linked to the nucleation and growth of particles (Zhang et al., 2004; Pirjola et
109 al., 2015). Additionally, inorganic acids such as nitric (HNO_3) and nitrous (HONO) acids, along with isocyanic acid (HNCO)—
110 implicated in serious health issues like atherosclerosis, cataracts, and rheumatoid arthritis through carbamylation
111 reactions (Fullerton et al., 2008; Roberts et al., 2011)—have been identified in both diesel and gasoline exhausts (Wang et al.,
112 2007; Roberts et al., 2011; Wentzell et al., 2013; Brady et al., 2014; Link et al., 2016; Li et al., 2021). However, the secondary
113 production of organic acids from engine exhaust remains poorly characterized; and it may significantly contribute to the overall
114 organic acid budget and help explain discrepancies between models and measurements (Paulot et al., 2011; Millet et al., 2015;
115 Yuan et al., 2015). Furthermore, the impacts of evolving fuel and engine technologies on emissions have not been
116 comprehensively assessed. Recent advancements in analytical techniques now enable simultaneous, high-resolution online
117 measurements of both gas and particle phase acidic species. This is facilitated by high-resolution time-of-flight chemical
118 ionization mass spectrometry (HR-ToF-CIMS) using acetate as the reagent ion, coupled with a filter inlet for gases and aerosols
119 (FIGAERO) (Le Breton et al., 2019; Friedman et al., 2017; Lopez-Hilfiker et al., 2014).

120

121 In this study, we employed the OFR Gothenburg Potential Aerosol Mass Reactor (Go:PAM) along with roadside point
122 measurements to capture emissions from a diverse array of fuel types and engine technologies in in-use transit buses. We
123 present findings on the photochemical aging of emissions from a modern fleet operating on diesel (DSL) and the latest
124 generation of alternative fuels, including compressed natural gas (CNG), rapeseed methyl ester (RME), and hydrotreated
125 vegetable oil (HVO). Our study aims to compare the secondary production of PM from individual buses in real traffic scenarios
126 to their primary PM emissions, examining the impact of fuel type, engine technology, and photochemical age. Furthermore,
127 both fresh and aged emissions of gas and particle phases are characterized using HR-ToF-CIMS, providing a comprehensive
128 understanding of the emissions profile and their environmental implications.

129

130 **2. Methods**

131 **2.1 Emission measurements**

132 Roadside measurements were conducted at a designated urban bus stop, featuring a bus-only lane, in Gothenburg, Sweden.
133 (Supporting information (SI), Figure S1). The sampling occurred from March 2nd to 12th, 2016, with the average temperature
134 during this period recorded at approximately 3.9°C. Extractive sampling of individual bus plumes in real traffic was used to
135 characterize emissions, adhering to the method outlined by Hallquist et al. (2013). Air was continuously drawn through a cord-
136 reinforced flexible conductive hose to the instruments housed within a nearby container. Additional details of the experimental
137 conditions are available in our prior publication by Liu et al. (2019a). The primary focus of this study was to utilize the OFR
138 Go:PAM and the HR-ToF-CIMS to explore the potential for secondary pollutant formation and to conduct a detailed chemical
139 characterization of both gas and particle phase compounds. An experimental schematic of the roadside sampling is shown in
140 Figure S2. Briefly, the emissions from passing bus plumes were characterized as they accelerated from standstill at the bus
141 stop. A camera was positioned at the roadside to capture bus plate numbers, facilitating bus identification and enabling the
142 collection of specific information on each bus, including fuel type, engine technology, and exhaust after-treatment systems.
143 The effective identification of emissions from individual buses was achieved by employing CO₂ as a tracer, as delineated by
144 Hak et al. (2009). The concentration of CO₂ was measured with a non-dispersive infrared gas analyzer (LI-840A, time
145 resolution 1 Hz). NO and NO_x were measured with two separate chemiluminescent analyzers (Thermo Scientific™ Model 42i
146 NO-NO₂-NO_x Analyzer). In addition, specific gaseous compounds like CO, NO, and THC, were measured using a remote
147 sensing device (AccuScan RSD 3000, Environmental System Products Inc.). Particle emissions were characterized using a
148 high time resolution engine exhaust particle sizer spectrometer (EEPS, Model 3090 TSI Inc., time resolution 10 Hz) across a
149 size range of 5.6-560 nm. Due to the lack of detailed knowledge about the chemical composition of the emitted particles,
150 particle mass calculations were based on the assumption of spherical particles of unit density.

151

152 The HR-ToF-CIMS coupled with a FIGAERO was used to derive chemical information of both gas and particle phase species.
153 A detailed description of the configuration of the instrument can be found elsewhere (Aljawhary et al., 2013; Lopez-Hilfiker
154 et al., 2014; Le Breton et al., 2018; Le Breton et al., 2019). Acetate, employed as the reagent ion, was generated using an acetic
155 anhydride permeation source through a ²¹⁰Po ion source (²¹⁰Po inline ionizer, NRD inc, Static Solutions Limited). In the ion-
156 molecular reaction (IMR) chamber, the gaseous sampling flow interacted with the reagent ions, leading to the ionization of
157 target molecules. The dual inlets of the FIGAERO enable simultaneous gas phase sampling directly into the IMR and particle
158 sample collection on a PTFE filter for the duration of the plume via a separate inlet. The duration of the target plume for
159 particle collection was indicated by particle number (PN) concentration measured by the EEPS. Once the PN concentration
160 reduced to undistinguishable at background levels, the filter was automatically positioned to allow the collected particles to be
161 evaporated into the IMR. The nitrogen flow over the filter was incrementally heated from room temperature to 200°C within
162 5 minutes and then maintained at this maximum temperature for 8 minutes, ensuring complete desorption of mass from the
163 filter, followed by analysis via HR-ToF-CIMS. Perfluoropentanoic acid (PFPA), a reliable high mass calibrant, was injected
164 into the CIMS inlet during the sampling period (Le Breton et al., 2019). Mass spectra were calibrated using known masses
165 (m/z), accurate within 4 ppm: O₂⁻, CNO⁻, C₃H₅O₃⁻, C₂F₃O₃⁻, C₅F₉O₂⁻, C₁₀F₁₈O₄⁻, covering a range of 32-526 m/z (more details

166 can be found in SI). The data were acquired at 1 s time resolution. To estimate absolute EFs, a conversion of the CIMS signal
167 to concentration using a sensitivity factor is necessary. Based on the method of Lopez-Hilfiker et al. (2015), the maximum
168 sensitivity was determined to be 20 Hz ppt⁻¹, which falls within previously reported ranges (Mohr et al., 2017). Using this
169 maximum sensitivity provides a lower-limit estimate of EFs for all oxygenated volatile organic compounds (Zhou et al., 2021).
170 The assumption on sensitivity did not affect the comparative analysis of EFs with respect to different fuel types.

171

172 The EFs of constituents per kilogram of fuel burnt were calculated by relating the concentration change of a specific compound
173 in the diluted exhaust plume to the change in CO₂ concentration. CO₂ served as a tracer for exhaust gas dilution, relative to
174 background concentration (Janhäll and Hallquist, 2005; Hak et al., 2009; Hallquist et al., 2013; Watne et al., 2018).
175 Assumptions were made for complete combustion and carbon contents of 86.1, 77.3, 70.5, and 69.2% for DSL, RME, HVO,
176 and CNG, respectively, were assumed (Edwards et al., 2004). Further methodological details are elaborated in Liu et al.
177 (2019a). A more comprehensive description of the EF calculations is provided in the Supporting Information.

178

179 **2.2 Oxidation flow reactor setup**

180 The OFR Go:PAM was utilized for photochemical aging of emissions from individual buses to investigate the potential for
181 secondary pollutant formation. The comprehensive description and operational protocols of the Go:PAM have been detailed
182 previously (Watne et al., 2018; Zhou et al., 2021). Briefly, the Go:PAM is a 6.1 L continuous-flow quartz glass flow reactor
183 with input flows such that the median residence time is approximately 37s. The reactor is equipped with two Philips TUV 30
184 W fluorescent lamps ($\lambda=254$ nm) and enclosed by reflective and polished aluminium mirrors to ensure a homogeneous photon
185 field. The UV lamps generate OH radicals through the photolysis of O₃ in the presence of water vapor. The relative humidity
186 (RH) within the reactor was around 60 - 80%. The O₃ concentration inside the Go:PAM was measured using an ozone monitor
187 (2B technology, model 205 dual beam ozone monitor) at around 880 ppb prior to the introduction of vehicle exhaust. Particle
188 wall losses in the Go:PAM were corrected using size-dependent transmission efficiency (Watne et al., 2018). The OH exposure
189 (OH_{exp}) inside the Go:PAM was calibrated offline using sulfur dioxide (SO₂), following methodologies established in previous
190 studies (Lambe et al., 2011; Kang et al., 2007), with additional details provided in the SI. During on-road measurements, the
191 OH_{exp} may be significantly influenced by the OH reactivity (i.e., CO and HC) and titration of O₃ by NO in the plumes, which
192 varied between vehicles. Thus, the OH reactivity was estimated for each bus passage using the maximum NO_x, CO and HC
193 concentrations in the Go:PAM, along with corresponding water and ozone levels (Watne et al., 2018; Zhou et al., 2021).
194 Employing the maximum concentrations of these OH- or O₃-consuming species represents a minimum estimate of OH_{exp} in
195 our calculations. The flow-design incorporated in the Go:PAM enables investigation of transient phenomena, such as passing
196 plumes. It also works at relatively low ozone concentrations (less than 1 ppm), limiting reactions of other potential oxidants
197 such as O₃, NO₃, or O¹D (Zhou et al., 2021).

198

199 3. Results and discussion

200 3.1 Fresh and aged PM emissions from buses

201 The aged PM emissions ($EF_{PM:aged}$) of 133 plumes from a diverse set of buses, including 16 diesel (DSL), 11 compressed
202 natural gas (CNG), 20 rapeseed methyl ester (RME), 20 hydrotreated vegetable oil (HVO) and 9 hybrid-electric HVO
203 (HVO_{HEV}) buses, were investigated using Go:PAM. The corresponding average fresh PM emissions ($EF_{PM:Fresh}$) for these 76
204 buses were measured during several sequential days (Figure S2). These buses were a subset of the 234 buses described in our
205 previous study (Liu et al., 2019a), and represent data corresponding to available Go:PAM measurements. A comprehensive
206 discussion on the full data set for fresh condition is available in Liu et al. (2019a). Figure 1 shows the average $EF_{PM:Fresh}$ and
207 $EF_{PM:aged}$ with respect to fuel type. Among the buses, Euro V DSL models had the highest median $EF_{PM:Fresh}$, $MdEF_{PM:Fresh}$
208 (represented by the horizontal yellow lines), of 208 mg kg-fuel⁻¹, followed by HVO_{HEV} , RME and HVO buses with $MdEF_{PM:Fresh}$
209 of 109, 74 and 62 mg kg-fuel⁻¹ respectively. CNG buses and HVO_{HEV} buses equipped with a DPF under Euro VI standards
210 exhibited the lowest $MdEF_{PM:Fresh}$, with over half of these buses exhibiting $EF_{PM:Fresh}$ below the detection limit (<4.3 mg kg-fuel⁻¹).
211 Except for HVO_{HEV} buses with a DPF, which was limited to a small tested number, all other bus types in this subset had
212 $MdEF_{PM:Fresh}$ comparable to those of the full data set in Liu et al. (2019a), within $\pm 30\%$ and following the same rank order. The
213 average EFs of fresh and aged particle emissions and general gaseous pollutants for individual buses are given in Table 1.

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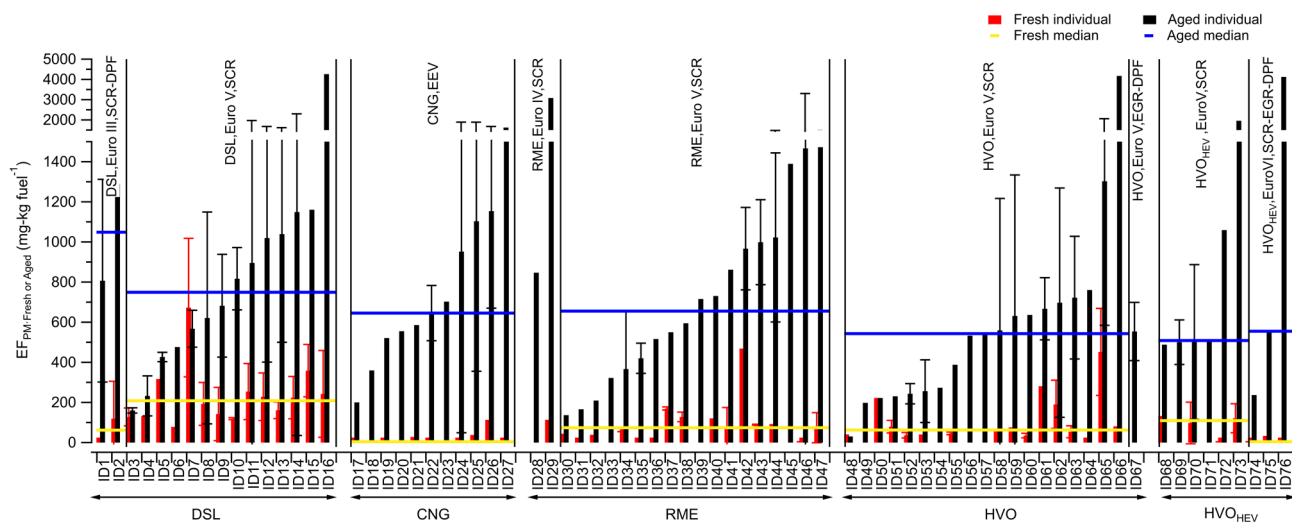
215 After photooxidation in Go:PAM, particle mass increased markedly, with half of the individual buses showing average
216 $EF_{PM:aged}$ more than eight times their average $EF_{PM:Fresh}$. For all Euro V/EEV buses, the median $EF_{PM:aged}$, $MdEF_{PM:aged}$
217 (represented by the horizontal blue lines), was highest for DSL buses of 749 mg kg-fuel⁻¹ followed by a descending order of
218 RME (655) > CNG (645) > HVO (543) > HVO_{HEV} (509). Despite low $EF_{PM:Fresh}$, CNG buses produced substantial secondary
219 particle mass. The DPF, proven effective in earlier studies (Martinet et al., 2017; Preble et al., 2015; May et al., 2014),
220 efficiently reduced primary particle emissions from DSL Euro III and HVO_{HEV} Euro VI buses. However, these bus types, even
221 with DPFs, exhibited higher $EF_{PM:aged}$ than those using the same fuels but without DPFs (Euro V), albeit the number of tested
222 buses with DPFs was limited. The variance in median $EF_{PM:aged}$ among different fuel types was less pronounced compared to
223 $EF_{PM:Fresh}$, suggesting the presence of significant non-fuel-dependent precursor sources, such as lubrication oils and/or fuel
224 additives (Watne et al., 2018; Le Breton et al., 2019).

225

226 Figure 2 shows the bus average $EF_{PM:Fresh}$ vs the corresponding $EF_{PM:aged}$ for individual bus passages, where the average
227 $EF_{PM:aged}$ for each bus is indicated by a solid horizontal line. This analysis focuses on Euro V/EEV buses to ensure a sufficient
228 number of buses in the comparison, while buses from other Euro classes were not included due to their limited numbers. The
229 median ratio of $EF_{PM:aged}$ to $EF_{PM:Fresh}$ was highest for CNG buses (84), followed by RME (10.8), HVO_{HEV} (10.5), HVO (6.7)
230 and DSL(4.0) buses. Buses equipped with DPFs, such as DSL Euro III and HVO_{HEV} Euro VI (not included in Figure 2),
231 exhibited a median ratio exceeding 50. $EF_{PM:aged}$ exhibited notable variation between passages of the same bus, likely

232 attributable to emission variability between passages and different dilution levels for plumes prior to sampling into the
 233 Go:PAM. This is illustrated in Figure 2b, where $EF_{PM:Fresh}$ and $EF_{PM:aged}$ are presented as a function of the dilution level,
 234 indicated by the integrated CO_2 area. Generally, a higher integrated CO_2 area suggests a more concentrated plume, leading to
 235 increased external OH and O_3 reactivity, which in turn reduces the concentration of OH radicals available in Go:PAM for
 236 precursor oxidation (Emanuelsson et al., 2013; Watne et al., 2018). Some buses displayed primary emissions too dilute for
 237 detection (markers located to the left in Figure 2b) but still exhibited non-negligible $EF_{PM:aged}$ after oxidation. To further
 238 examine the effects of simulated atmospheric oxidation in the Go:PAM, an estimated minimum OH_{exp} was calculated for each
 239 plume by incorporating the OH reactivities of CO and HC and the titration of O_3 with NO, following methodologies from
 240 Watne et al. (2018) and Zhou et al. (2021). For all plumes, OH_{exp} varied between 1.1×10^9 to 4.6×10^{11} molecules $cm^{-3} s$. The
 241 $EF_{PM:aged}$ for some buses, for example, the DSL and HVO located to the right in Figure 2c, increased with increasing OH_{exp} .
 242 However, due to potential large differences in the chemical composition of emissions across different passages of the same
 243 bus, where some species are more prone to forming secondary particle mass even at lower OH_{exp} , the OH_{exp} dependent $EF_{PM:aged}$
 244 for other buses was less pronounced.

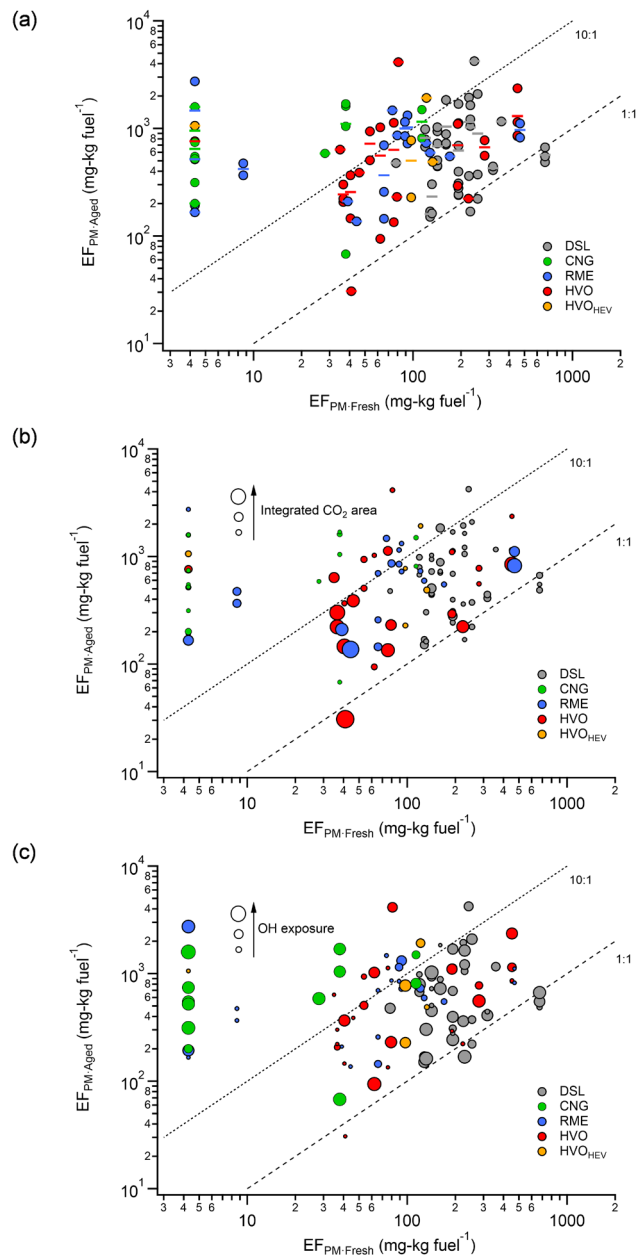
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247

248 Figure 1. $EF_{PM:Fresh}$ (red bar) and $EF_{PM:aged}$ (black bar) with respect to fuel class: DSL (diesel, ID₁-ID₁₆), CNG (compressed
 249 natural gas, ID₁₇-ID₂₇), RME (rapeseed methyl ester, ID₂₈-ID₄₇), HVO (rapeseed methyl ester, ID₄₈-ID₆₇) and HVO_{HEV} (hybrid-
 250 electric HVO, ID₆₈-ID₇₆) buses. Median values for $EF_{PM:Fresh}$ ($M^d EF_{PM:Fresh}$) and $EF_{PM:aged}$ ($M^d EF_{PM:aged}$) are indicated by
 251 horizontal yellow and blue lines, respectively. The information on engine technology and exhaust after-treatment systems is
 252 also shown. Given errors represent the standard deviation (1σ).

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256 Figure 2. $EF_{PM:aged}$ vs average $EF_{PM:Fresh}$ for all the studied bus passages (Euro V) with respect to fuel type (a) and as a function
 257 of integrated CO₂ area (b) and OH exposure (OH_{exp}) (c). The dashed lines denote the 10:1 and 1:1 $EF_{PM:aged}$: $EF_{PM:Fresh}$ ratios,
 258 and the solid lines in (a) represent bus averages. One may note that the buses with $EF_{PM:Fresh}$ values below detection limit were
 259 set to 4.3 mg kg-fuel⁻¹. Abbreviations: DSL (diesel), CNG (compressed natural gas), RME (rapeseed methyl ester), HVO
 260 (hydrotreated vegetable oil), HVO_{HEV} (hybrid-electric HVO).

261 Table 1. Average particle and gaseous EFs of individual buses for fresh emissions and average EF_{PM} for aged emissions^a.

Bus ID	Fuel ^c	Euro standard	Exhaust after-treatment system ^d	EF _{PM:Fresh} (mg kg ⁻¹ _{fuel})	EF _{PN:Fresh} (10 ¹⁴ # kg ⁻¹ _{fuel})	EF _{CO} (g kg ⁻¹ _{fuel})	EF _{THC} (g kg ⁻¹ _{fuel})	EF _{NOx} (g kg ⁻¹ _{fuel})	EF _{PM:Aged} (mg kg ⁻¹ _{fuel})
1	DSL	III	SCR, DPF	4.3	0.41	3.9±11	1.5±2.9	10±3.2	810±510
2	DSL	III	SCR, DPF	120±190	34±61	2.7±7	1.7±3.7	11±5	1300
3	DSL	V	SCR	130±45	3.3±1.3	17±18	0.35±1.3	3.9±3.7	160±13
4	DSL	V	SCR	130	3.6	20±22	1.5±3.6	4.7±7.2	230±100
5	DSL	V	SCR	320	5.9	20±28	2±3.5	9.7±7	430±23
6	DSL	V	SCR	78	1.6	20±21	2.7±5.6	13±12	480
7	DSL	V	SCR	670±350	10±6.8	42±44	2.3±3.7	6.8±5	570±92
8	DSL	V	SCR	190±110	6.5±3	14±21	0.75±1.7	12±5.1	620±530
9	DSL	V	SCR	140±130	4.3±2.6	9.8±14	1±1.5	15±13	680±260
10	DSL	V	SCR	120±4.7	3.2±0.66	16±18	2.5±4.7	12±6.9	820±160
11	DSL	V	SCR	250±140	4.7±2.7	16±23	0.8±1.4	12±8.9	900±1000
12	DSL	V	SCR	230±120	5.1±1.5	16±26	2.6±4.6	12±9.9	1000±620
13	DSL	V	SCR	160±41	3.5±0.97	27±27	1.4±2.7	17±9.8	1000±540
14	DSL	V	SCR	220±110	5.2±1.3	12±17	2.6±4.1	11±7.4	1100±1100
15	DSL	V	SCR	360±130	6.8±4.2	21±25	1.2±3.3	5.7±4.4	1200
16	DSL	V	SCR	240±220	22±11	5.5±7.5	0.74±1.6	6.8±5.6	4200
17	CNG	EEV	-	4.3±0	0.41±0	n.a.	n.a.	4.8±1.7	200
18	CNG	EEV	-	n.a.	n.a.	n.a.	n.a.	11±4.9	360
19	CNG	EEV	-	4.3±0	0.41±0	n.a.	n.a.	4±3.8	520
20	CNG	EEV	-	n.a.	n.a.	n.a.	n.a.	15±17	560
21	CNG	EEV	-	28	1.3	n.a.	n.a.	2.2±0.93	590
22	CNG	EEV	-	4.3	0.41	n.a.	n.a.	1.8±1	650±140
23	CNG	EEV	-	n.a.	n.a.	n.a.	n.a.	3.2±0.53	700
24	CNG	EEV	-	4.3	0.41	n.a.	n.a.	6.9±1.4	950±900
25	CNG	EEV	-	38	11	n.a.	n.a.	7.3±5.3	1100±750
26	CNG	EEV	-	110	200	n.a.	n.a.	8.2±4.2	1200±480
27	CNG	EEV	-	4.3±0	0.41±0	n.a.	n.a.	6±1.8	1600
28	RME	IV	SCR	n.a.	n.a.	10±8.7	3.1±3	46±20	850
29	RME	IV	SCR	110	4.1	4.2±8.4	0.19±0.38	7.2±6.8	3000
30	RME	V	SCR	44	2.2	12±14	2.2±3.6	32±32	140
31	RME	V	SCR	4.3	0.41	7.4±7.1	0.075±0.17	13±5.1	170
32	RME	V	SCR	39	6.2	5.2±4.8	0.87±1.1	18±5.4	210
33	RME	V	SCR	n.a.	n.a.	0.24±0.54	0.24±0.39	10±3.3	320
34	RME	V	SCR	66±11	2.4±1	7±7.2	1.8±2.7	23±13	370±290
35	RME	V	SCR	8.6	0.96	4.9±3.6	0.59±0.73	20±5.1	420±75
36	RME	V	SCR	4.3	0.41	22±23	1.8±2	25±16	520
37	RME	V	SCR	170±7.7	6.4±1	34±35	0.016±0.043	19±10	550
38	RME	V	SCR	130±24	11±14	17±20	2±4	16±15	590
39	RME	V	SCR	n.a.	n.a.	1.2	0.64	21	720
40	RME	V	SCR	120	5.3	12±9.4	1.8±2.6	18±8.2	730
41	RME	V	SCR	80±95	4.2±2.9	8.8±17	0.72±0.87	25±5.7	860
42	RME	V	SCR	470	5.8	4.5±5.1	0.23±0.38	18±7.8	970±210
43	RME	V	SCR	89±2.3	2.6±0.16	5.4±9.4	0.68±1.9	28±17	1000±210
44	RME	V	SCR	92	1.6	14±19	1.8±3	23±17	1000±420
45	RME	V	SCR	n.a.	n.a.	37±26	5.8±3.6	14±6.3	1400
46	RME	V	SCR	4.3±0	0.41±0	9.6±14	0.89±1.4	28±8.4	1500±1800
47	RME	V	SCR	74±75	12±6	6.1±6.3	1.1±1.4	18±5.2	1500
48	HVO	V	SCR	41	1.5	8.4±2	0.14±0.31	10±0.4	31
49	HVO	V	SCR	n.a.	n.a.	5.8±8	0.7±0.62	13±10	200
50	HVO	V	SCR	220	6.6	8.3±9.1	0.91±0.97	13±8.6	220
51	HVO	V	SCR	79±31	2.6±0.74	7.8±5.8	0.41±0.59	12±8.2	230
52	HVO	V	SCR	37±13	1.9±0.65	4.8±5.5	0.64±0.82	20±3	240±51
53	HVO	V	SCR	40	2.5	2.1±3.4	0.0083±0.019	16±4.3	260±160
54	HVO	V	SCR	n.a.	n.a.	2.1±3	0.55±0.77	22	270
55	HVO	V	SCR	46±6.6	2.6±0.52	6.2±4.1	0.79±0.55	12±8.2	390
56	HVO	V	SCR	n.a.	n.a.	11±10	0.74±0.84	5.7	530
57	HVO	V	SCR	n.a.	n.a.	14±17	0.79±1.2	11±2.6	540
58	HVO	V	SCR	62	4.1	6.8±6.7	0.22±0.31	11±6.3	560±660

59	HVO	V	SCR	76	5.3	2.3±2	0.24±0.47	19±3.4	630±700
60	HVO	V	SCR	35±11	1.5±0.19	3.3±5	0.45±0.86	9.2±9	640
61	HVO	V	SCR	280	14	9.9±16	0.55±0.73	11±3.6	670±160
62	HVO	V	SCR	190±120	68±86	1.1±1.9	0.3±0.49	9.3±4.9	700±570
63	HVO	V	SCR	54±30	4.6±2.2	3.5±4.6	0.49±0.48	14±3.5	720±310
64	HVO	V	SCR	4.3	0.41	2.2±3.8	0.33±0.73	12±4.8	760
65	HVO	V	SCR	450±220	18±18	1.4±1.6	0.28±0.37	12±2.6	1300±720
66	HVO	V	SCR	81	11	0.88±0.93	0.28±0.25	13±6.5	4100
67	HVO	V	EGR, DPF	n.a.	n.a.	4.6±5.9	0.64±1.2	11±8.1	550±150
68	HVO _{HEV}	V	SCR	130	52	12±19	0.97±1.4	20±15	490
69	HVO _{HEV}	V	SCR	n.a.	n.a.	4.1±8.4	0.5±1.3	18±3.3	500±110
70	HVO _{HEV}	V	SCR	97±100	25±18	3.8±6.8	1.1±1.8	17±5.7	500±390
71	HVO _{HEV}	V	SCR	n.a.	n.a.	7.6±9.9	2.9±2.4	12±2.1	520
72	HVO _{HEV}	V	SCR	4.3±0	0.41±0	3.7±5.8	1±2.4	20±10	1100
73	HVO _{HEV}	V	SCR	120±72	8.9±2.9	1.2±1.7	0.18±0.26	17±7	1900
74	HVO _{HEV}	VI	SCR,EGR,DPF	4.3±0	0.41±0	4.7±11	2.2±4.7	7.2±8.5	240
75	HVO _{HEV}	VI	SCR,EGR,DPF	33	29	1.2±2.4	0.22±0.49	6.7±3.3	550
76	HVO _{HEV}	VI	SCR,EGR,DPF	4.3	0.41	10±9.2	1.5±2.3	8.8±8.7	4100

262

263 ^aGiven errors represent the standard deviation (1σ).264 ^bn.a., abbreviation for not available.265 ^cDSL, CNG, RME, HVO and HVO_{HEV}, abbreviations for diesel, compressed natural gas, rapeseed methyl ester, hydrotreated vegetable oil, and hybrid-electric
266 hydrotreated vegetable oil.267 ^dSCR, DPF and EGR, abbreviations for selective catalytic reduction, diesel particulate filter and exhaust gas recirculation systems.

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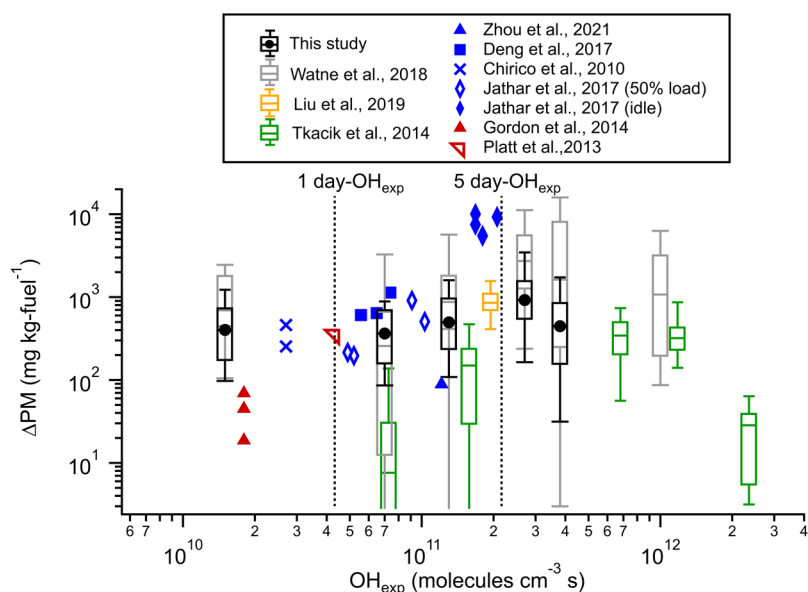
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271 The secondary particle mass formed (ΔPM) was calculated as the difference between $\text{EF}_{\text{PM:aged}}$ for a plume and the average
272 $\text{EF}_{\text{PM:Fresh}}$ for the corresponding individual bus. Figure 3 illustrates ΔPM as a function of OH_{exp} for the bus fleet in this study,
273 which includes 40% DSL, 12.2% CNG, 20% RME, 20.8% HVO, and 7% HVO_{HEV}. The results were grouped based on OH_{exp} ,
274 spanning a range from 1.1×10^9 to 4.6×10^{11} molecules cm^{-3} s. The results in this study are compared with those reported from
275 a tunnel study (Tkacik et al., 2014), an urban roadside study of a mixed fleet in Hong Kong (Liu et al., 2019b), a depot study
276 on rather modern types of city buses (Watne et al., 2018) and roadside measurements of a heavy-duty truck fleet in
277 Gothenburg (Zhou et al., 2021). Laboratory OFR and chamber studies of middle-duty and heavy-duty diesel vehicles (Deng
278 et al., 2017), diesel passenger cars (Chirico et al., 2010), a diesel engine (Jathar et al., 2017a), and gasoline vehicles (Gordon
279 et al., 2014a; Platt et al., 2013) were also included for comparison. Note that ΔPM in this study, alongside those by Watne et
280 al. (2018), Zhou et al. (2021) and Liu et al. (2019b), includes both secondary organic and inorganic aerosol, while ΔPM in
281 research by Deng et al. (2017), Chirico et al. (2010), Jathar et al. (2017a), Gordon et al. (2014a), Platt et al. (2013) and Tkacik
282 et al. (2014) pertains only to secondary organic aerosol mass.

283

284 The ΔPM from vehicle emissions is influenced by factors such as vehicle and fuel types, driving modes, and OH_{exp} during
285 experiments (Gentner et al., 2017). Considering the variability of OH reactivity among vehicles and the consequently wide
286 range of OH_{exp} , this study, along with Watne et al. (2018), categorizes ΔPM trend into OH_{exp} bins. The median ΔPM was
287 approximately 400 mg kg-fuel^{-1} at $\text{OH}_{\text{exp}} < 4.3 \times 10^{10}$ molecules cm^{-3} s (corresponding to 1 OH day, assuming an OH
288 concentration of 1×10^6 molecules cm^{-3} for 12 h per day) and was 364–495 mg kg-fuel^{-1} at 1–5 OH days, reaching a maximum

289 of around $920 \text{ mg kg-fuel}^{-1}$ at approximately 5-6 OH days for the bus fleet in this study. This peak value of ΔPM was lower
 290 than the approximately $3000 \text{ mg kg-fuel}^{-1}$ at $\sim 5\text{-}6$ OH days observed in the depot measurements by Watne et al. (2018), a
 291 difference potentially due to variations in engine technology and fuel types used in the bus fleets. Notably, HVO was not used
 292 in the depot study, while some buses switched from RME to HVO prior to this study. The ΔPM peaked and then decreased at
 293 higher OH_{exp} , likely due to the transition from functionalization-dominated reactions and condensation at lower OH_{exp} to
 294 fragmentation reactions and evaporation dominance at higher OH_{exp} (Tkacik et al., 2014; Ortega et al., 2016). The ΔPM in this
 295 study was comparable to $855 \text{ mg kg-fuel}^{-1}$ for a mixed fleet consisting of 44.1% gasoline, 41.3% diesel, and 14.6% LPG
 296 vehicles measured at an urban roadside in Hong Kong (Liu et al., 2019b). It was slightly higher than the ΔPM measured from
 297 a Euro VI dominated (more than 70%) heavy-duty truck fleet at an urban roadside in Gothenburg (Zhou et al., 2021), and
 298 from a fleet with over 80% light-duty gasoline vehicles in a Pittsburgh tunnel study (Tkacik et al., 2014). Additionally, the
 299 ΔPM in this study was consistent with that for middle-duty and heavy-duty diesel vehicles (Deng et al., 2017), diesel passenger
 300 cars (Chirico et al., 2010), and a diesel (or biodiesel)-fuelled engine under 50% load condition (Jathar et al., 2017a) (around
 301 $190\text{-}1133 \text{ mg kg-fuel}^{-1}$). However, the diesel (or biodiesel)-fuelled engine under idle conditions can produce significantly
 302 higher ΔPM (more than $5000 \text{ mg kg-fuel}^{-1}$), likely because engines at idle loads are less efficient at burning fuel, leading to
 303 higher emissions of unburnt gaseous combustion products (as precursors of secondary PM) (Nordin et al., 2013; Saliba et al.,
 304 2017; Jathar et al., 2017a). In contrast, experiments conducted for gasoline vehicles at relatively low photochemical ages (< 1
 305 OH day) typically produced ΔPM lower than $70 \text{ mg kg-fuel}^{-1}$ (Gordon et al., 2014a), except for a Euro 5 gasoline vehicle (340
 306 mg kg-fuel^{-1}) operated with a New European Driving Cycle (Platt et al., 2013).



308
 309 Figure 3. Secondary particle mass formed (ΔPM), calculated as $\text{EF}_{\text{PM:aged}}$ subtracted by the average $\text{EF}_{\text{PM:Fresh}}$, vs modeled OH
 310 exposure (OH_{exp}) for the bus fleet in this study and comparison with those reported for a tunnel study (Tkacik et al., 2014), a

311 depot study (Watne et al., 2018), roadside measurements (Liu et al., 2019b; Zhou et al., 2021), middle-duty and heavy-duty
312 diesel vehicles (Deng et al., 2017), diesel passenger cars (Chirico et al., 2010), a diesel engine (Jathar et al., 2017a), and
313 gasoline vehicles (Gordon et al., 2014a; Platt et al., 2013). Dashed lines indicate 1- and 5-day OH_{exp} assuming an OH
314 concentration of 1×10^6 molecules cm^{-3} 12 h per day (Watne et al., 2018).

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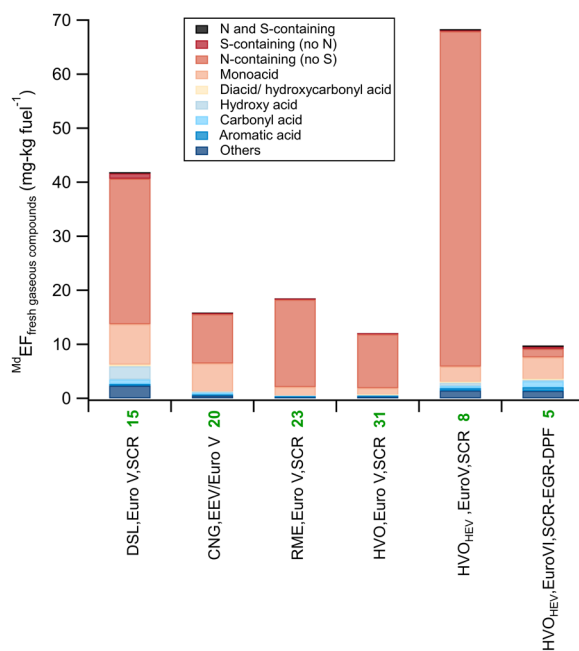
318 **3.2 Chemical characterization using CIMS**

319 **3.2.1 Fresh gaseous emissions**

320 Figure 4 presents the median emission factors ($\text{M}^{\text{d}}\text{EFs}$) of acetate CIMS measured fresh gaseous emissions with respect to fuel
321 type. The identities of the organic compounds detected by HR-ToF-CIMS are assigned based on knowledge of sensitivities of
322 the ionization scheme and the expected compounds emitted from the buses. Plausible compounds are assigned from the
323 formulae, with a caveat that other isomers might contribute to the signal. These compounds were classified into nine families
324 based on their molecular characteristics as outlined by Liu et al. (2017), with additional details provided in the SI. Among all
325 Euro V/EEV buses, hybrid-electric HVO (HVO_{HEV}) buses exhibited the highest $\text{M}^{\text{d}}\text{EF}$ of CIMS measured fresh gaseous
326 emissions ($68 \text{ mg kg-fuel}^{-1}$), followed by DSL ($42 \text{ mg kg-fuel}^{-1}$), RME ($18 \text{ mg kg-fuel}^{-1}$), and CNG ($16 \text{ mg kg-fuel}^{-1}$), while
327 HVO had the lowest $\text{M}^{\text{d}}\text{EF}$ of $12 \text{ mg kg-fuel}^{-1}$. Nitrogen (N) -containing compounds (no sulfur) and monoacid families
328 predominantly composed these fresh gaseous emissions. Compared to Euro V HVO_{HEV} buses, HVO_{HEV} buses equipped with
329 exhaust gas recirculation (EGR) and DPF systems (Euro VI) demonstrated a significant reduction in $\text{M}^{\text{d}}\text{EF}$ ($10 \text{ mg kg-fuel}^{-1}$),
330 primarily due to decreased emissions of N-containing compounds, although the $\text{M}^{\text{d}}\text{EF}$ of other compound families were higher.
331 In contrast, Zhou et al. (2021) reported significant reductions in both carboxylic acids and carbonyl compounds (by 94% on
332 average), and acidic nitrogen-containing organic and inorganic species (79%) when transitioning from Euro V to Euro VI
333 heavy-duty trucks. However, details on the types of exhaust after-treatment systems used in the trucks from such study are not
334 specified. Moreover, this study utilized acetate as a different reagent ion for CIMS compared to the iodide used by Zhou et al.
335 (2021). Table 2 lists the top 10 $\text{M}^{\text{d}}\text{EFs}$ of fresh gaseous compounds, contributing over 88% of total fresh gaseous emissions
336 measured by CIMS for most bus types, except for Euro VI HVO_{HEV} (61%). The fresh gaseous emissions from all types of Euro
337 V/EEV buses were primarily composed of nitrous acid (HONO) and nitric acid (HNO_3), with HONO being the most significant
338 acidic emission. The $\text{M}^{\text{d}}\text{EFs}$ of HONO and HNO_3 generally align with values reported in the literature, ranging from
339 approximately $7\text{-}250 \text{ mg kg-fuel}^{-1}$ for HONO (Kurtenbach et al., 2001; Wentzell et al., 2013; Liao et al., 2020; Nakashima and
340 Kondo, 2022) and around $4\text{-}14 \text{ mg kg-fuel}^{-1}$ for HNO_3 (Wentzell et al., 2013). Acetic acid ($\text{C}_2\text{H}_4\text{O}_2$), formic acid (CH_2O_2), and
341 isocyanic acid (HNCO) also exhibited relatively high $\text{M}^{\text{d}}\text{EFs}$. The $\text{M}^{\text{d}}\text{EFs}$ of formic acid for all Euro V/EEV bus types (0.02-
342 $1.97 \text{ mg kg-fuel}^{-1}$) were consistent with those from a light-duty gasoline fleet ($0.57\text{-}0.94 \text{ mg kg-fuel}^{-1}$) reported by Crisp et al.
343 (2014). The $\text{M}^{\text{d}}\text{EFs}$ of acetic acid ranged from 1.23 to $4.84 \text{ mg kg-fuel}^{-1}$, falling between values for gasoline vehicles (0.78 mg
344 kg-fuel^{-1}) and diesel buses (approximately $12\text{-}23 \text{ mg kg-fuel}^{-1}$) (Li et al., 2021). Isocyanic acid, likely an intermediate product

345 of the thermal degradation of urea in SCR systems without sufficient hydrolysis (Bernhard et al., 2012), was detected in
 346 emissions from all bus types, with $M^{d}EFs$ of 0.08-14.74 mg kg-fuel⁻¹. These values are slightly lower than those from a non-
 347 road diesel engine (31-56 mg kg-fuel⁻¹) reported by Jathar et al. (2017b), but align well with SCR-equipped diesel vehicles
 348 tested by Suarez-Bertoa and Astorga (2016) (1.3-9.7 mg kg-fuel⁻¹) and a diesel engine with a diesel oxidation catalyst
 349 (DOC) (Wentzell et al., 2013) (0.21-3.96 mg kg-fuel⁻¹). Among all Euro V/EEV buses, HVO_{HEV} buses showed the highest
 350 emissions of HNCO, potentially attributed to cold engine conditions since the combustion engine does not operate
 351 continuously. Notably, emissions of HNCO were significantly lowered and neither HONO nor HNO₃ were identified among
 352 the top 10 $M^{d}EFs$ for HVO_{HEV} buses equipped with EGR and DPF systems (Euro VI), suggesting that newer engine technologies
 353 incorporating EGR and DPF systems likely effective in reducing emissions of NO_x (Table 1) as well as HNCO, HONO and
 354 HNO₃. CH₄SO₃, potentially identified as methanesulfonic acid, was detected in the emissions from DSL and RME buses.
 355 Previous studies, such as those by Corrêa and Arbilla (2008), have shown that mercaptans, emitted from diesel and biodiesel
 356 exhausts, can transform under high NO_x conditions into products including methanesulfonic acid. The presence of sulfur-
 357 containing organic compounds in diesel fuel and lubricants, and their potential transformation upon combustion into various
 358 sulfuric derivatives, alongside the catalytic activity of engine converters, could also contribute to such findings. However, the
 359 detailed formation pathway of CH₄SO₃ in our study remains unknown.

360



361

362 Figure 4. $M^{d}EFs$ of CIMS measured fresh gaseous emissions with respect to fuel class: DSL (diesel, 15), CNG (compressed
 363 natural gas, 20), RME (rapeseed methyl ester, 23), HVO (rapeseed methyl ester, 31) and HVO_{HEV} (hybrid-electric HVO, 13)
 364 buses. The number in bold green represents the number of buses examined.

366 Table 2. Summary of top 10 ^{Md}EFs of fresh gaseous compounds measured using HR-ToF-CIMS of DSL, CNG, RME, HVO
 367 and HVO_{HEV} buses^a (color coded by different families shown in Figure 4).

DSL, Euro V, SCR		CNG, EEV/Euro V		RME, Euro V, SCR		HVO, Euro V, SCR		HVO _{HEV} , Euro V, SCR		HVO _{HEV} , Euro VI	
Species	^{Md} EF (mg kg _{fuel} ⁻¹)	Species	^{Md} EF (mg kg _{fuel} ⁻¹)	Species	^{Md} EF (mg kg _{fuel} ⁻¹)	Species	^{Md} EF (mg kg _{fuel} ⁻¹)	Species	^{Md} EF (mg kg _{fuel} ⁻¹)	Species	^{Md} EF (mg kg _{fuel} ⁻¹)
HONO	20.64	HONO	4.92	HONO	12.72	HONO	7.62	HONO	38.96	C ₃ H ₂ O ₂	2.42
HNO ₃	5.29	C ₂ H ₄ O ₂	4.68	HNO ₃	3.24	HNO ₃	2.20	HNCO	14.74	C ₂ H ₄ O ₂	1.23
C ₂ H ₄ O ₂	4.84	HNO ₃	3.48	C ₂ H ₄ O ₂	1.23	C ₂ H ₄ O ₂	1.23	HNO ₃	7.89	C ₂ H ₂ O ₃	0.62
CH ₂ O ₂	1.97	HNCO	0.51	CH ₂ O ₂	0.48	C ₃ H ₆ O ₃	0.14	C ₂ H ₄ O ₂	1.83	C ₈ H ₆ O ₄	0.40
C ₃ H ₆ O ₃	1.79	CH ₂ O ₂	0.30	HNCO	0.15	C ₃ H ₂ O ₂	0.09	CH ₂ O ₂	0.45	C ₆ H ₅ NO ₂	0.31
CH ₄ SO ₃	0.71	C ₂ H ₂ O ₃	0.25	C ₂ H ₂ O ₃	0.05	HNCO	0.08	C ₃ H ₆ O ₃	0.43	HNCO	0.27
HNCO	0.67	C ₃ H ₂ O ₂	0.14	C ₃ H ₈ O ₃	0.03	CH ₂ O ₂	0.02	C ₃ H ₂ O ₂	0.34	C ₃ H ₄ O ₅	0.22
C ₃ H ₄ O ₅	0.37	C ₃ H ₄ O ₂	0.06	CH ₄ SO ₃	0.02	C ₂ H ₂ O ₃	0.02	C ₉ H ₁₀ O ₃	0.16	C ₇ H ₆ O ₃	0.20
C ₂ H ₂ O ₃	0.31	C ₇ H ₆ O ₃	0.05	C ₃ H ₄ O ₂	0.02	C ₃ H ₄ O ₃	0.02	C ₈ H ₆ O ₄	0.12	C ₅ H ₈ O ₃	0.17
C ₄ H ₆ O ₄	0.22	C ₅ H ₈ O ₄	0.05	C ₆ H ₆ N ₂ O ₂	0.01	C ₄ H ₆ O ₄	0.01	C ₅ H ₈ O ₃	0.10	H ₄ N ₂ O ₂ S	0.16

368 ^aDSL, CNG, RME, HVO and HVO_{HEV}, abbreviations for diesel, compressed natural gas, rapeseed methyl ester, hydrotreated vegetable oil, and hybrid-electric
 369 hydrotreated vegetable oil.

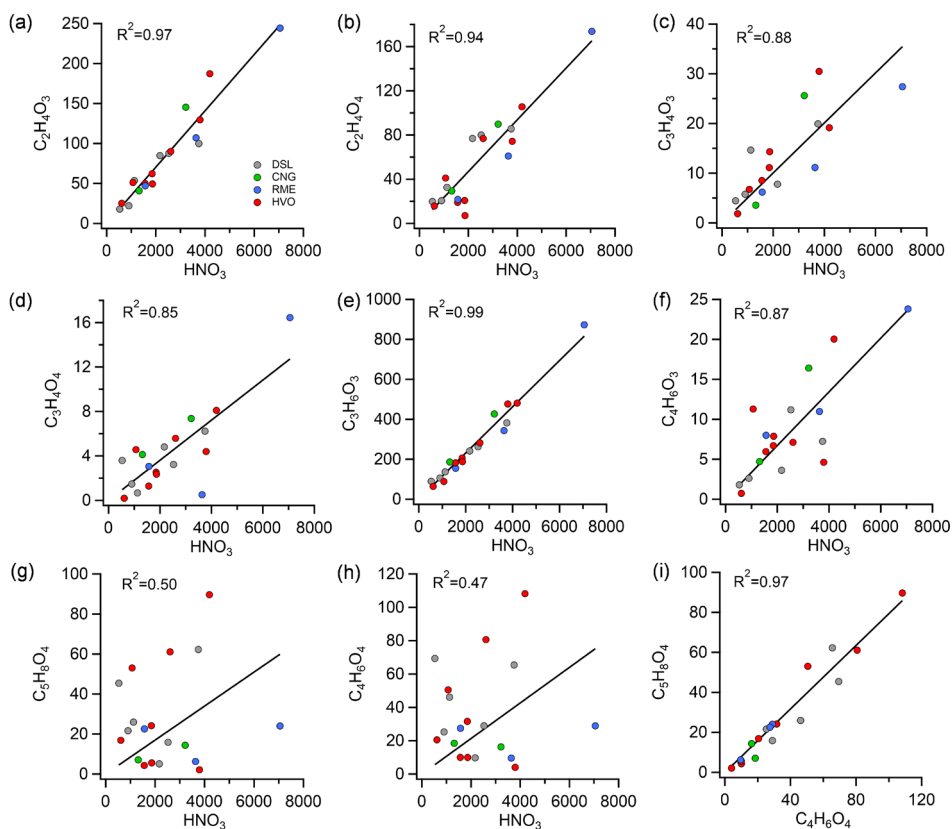
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372 3.2.2 Aged gaseous emissions

373 Secondary carboxylic acids were measured following exposure of the exhaust to OH radicals. Figure 5 shows the correlations
 374 between ion counts of the most abundant gas-phase organic acids and nitric acid (HNO₃) after oxidation in the Go:PAM. HNO₃
 375 serves as an indicator of NO_x oxidation. Most acids exhibited both primary and secondary sources, except for dihydroxyacetic
 376 acid (C₂H₄O₄), which was only identified post-aging. The chemical characterization of the aged emissions was conducted on
 377 separate occasions using HR-ToF-CIMS, capturing a limited number of buses (N=19). When these buses were categorized by
 378 fuel type, the sample size for each category became smaller, constraining statistical comparison across different bus types.
 379 Nevertheless, we analyzed the relationship between various chemical species across all buses. Glycolic acid (C₂H₄O₃),
 380 dihydroxyacetic acid (C₂H₄O₄), pyruvic acid (C₃H₄O₃), malonic acid (C₃H₄O₄), lactic acid (C₃H₆O₃) and acetoacetic acid
 381 (C₄H₆O₃) showed high correlations (R²= 0.85-0.99, Fig. 5a-f) with HNO₃ signals. In contrast, glutaric acid (C₅H₈O₄) and
 382 succinic acid (C₄H₆O₄) exhibited poorer correlations with HNO₃, suggesting different formation mechanisms for these two
 383 organic acids compared to the others mentioned. Notably, these two acids showed a strong correlation with each other (R²=
 384 0.97, Fig. 5i) and both belong to the diacid/hydroxycarbonyl acid families. It is important to note that many of these carboxylic
 385 acids can directly participate in secondary PM formation in the atmosphere in the presence of water vapor and a base such as
 386 ammonia (Chen et al., 2020; Huang et al., 2018; Hao et al., 2020). This process may significantly contribute to the overall
 387 secondary PM yield, reflecting a more complex interplay between gaseous emissions and particulate matter under atmospheric

388 conditions. While most of these small organic acids correlated well with HNO_3 , their correlations with $\text{EF}_{\text{PM:aged}}$ or ΔPM were
 389 moderate to weak ($R^2 < 0.6$, Figure S5). This possibly indicates that the OH-driven formation of these carboxylic acids occurs
 390 on a different time scale compared to the production of organic aerosol (Friedman et al., 2017), at least in this Go:PAM
 391 experiment. This could also be due to different subsets of hydrocarbon precursors driving the production of organic acids and
 392 secondary particle mass. Similarly, Friedman et al. (2017) observed a lack of correlation between organic aerosol and gaseous
 393 organic acid concentrations downstream of the flow reactor from a diesel engine, indicating that organic acids may not be
 394 reliable tracers for secondary organic aerosol formation from diesel exhaust.
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397

398 Figure 5. Correlations between ion counts of most abundant gas-phase organic acids and HNO_3 (a-h) and correlation between
 399 glutaric acid ($\text{C}_5\text{H}_8\text{O}_4$) and succinic acid ($\text{C}_4\text{H}_6\text{O}_4$) (i) from 19 buses after oxidation in the Go:PAM. Abbreviations: DSL
 400 (diesel), CNG (compressed natural gas), RME (rapeseed methyl ester), HVO (hydrotreated vegetable oil), HVO_{HEV} (hybrid-
 401 electric HVO).

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404

405 3.2.3 Particulate emissions

406 Table 3 displays the top 10 EFs of fresh particle-phase compounds (EF_{fresh}), as characterized by the FIGAERO ToF-CIMS,
 407 alongside their respective aged EFs (EF_{aged}), for Euro V DSL and RME buses. These top 10 EF_{fresh} contributed to over 82% of
 408 the total fresh particulate emissions measured by CIMS. Fresh particulate emissions from DSL buses were predominantly
 409 composed of sulfuric acid (H_2SO_4) and nitric acid (HNO_3). Benzene/toluene oxidation products ($C_7H_4O_7$, C_7H_8O , $C_6H_5NO_3$,
 410 C_6H_5O , $C_7H_7NO_3$) also had relatively high EF_{fresh} , aligning with the findings in Le Breton et al. (2019). Similarly, high EF_{fresh}
 411 of HNO_3 ($2.5 \text{ mg kg-fuel}^{-1}$) and H_2SO_4 ($0.61 \text{ kg-fuel}^{-1}$) were observed for the RME bus. Additionally, fatty acids, known as
 412 main components of unburned rapeseed oil (Usmanov et al., 2015), such as $C_{18}H_{34}O_2$, $C_{14}H_{28}O_2$, $C_{18}H_{36}O_2$, $C_{16}H_{32}O_2$, and
 413 $C_{16}H_{30}O_2$, significantly contributed to the identified mass loadings from the RME bus. When comparing the percentage mass
 414 observed by CIMS for both DSL and RME fuels in fresh and aged exhaust plumes, the total emission factors measured by
 415 CIMS (EF_{CIMS}) were notably lower than the total emission factors measured by the EEPS (EF_{total}). This difference is expected
 416 due to the sensitivity of the acetate ionization scheme of CIMS, which efficiently detects oxygenated volatile organic
 417 compounds, particularly carboxylic acids and inorganic acids, but has low sensitivity towards hydrocarbons and cannot detect
 418 metallic ions and soot. The CIMS measured EF_{fresh} accounted for 10.4% and 5.9% of the fresh EF_{total} measured by the EEPS
 419 for DSL and RME, respectively. In aged exhaust, EF_{CIMS} represented a higher percentage of EF_{total} (25.8% for DSL and 17.9%
 420 for RME), likely because of an increased proportion of organics with acid groups.

421

422 Table 3. Summary of top 10 EF_{fresh} of PM contributing species with respective EF_{aged} in Euro V DSL and RME emissions.

Species	DSL		Species	RME	
	EF_{fresh} ($\text{mg kg}_{\text{fuel}}^{-1}$)	EF_{aged} ($\text{mg kg}_{\text{fuel}}^{-1}$)		EF_{fresh} ($\text{mg kg}_{\text{fuel}}^{-1}$)	EF_{aged} ($\text{mg kg}_{\text{fuel}}^{-1}$)
H_2SO_4	4.8	6.8	HNO_3	2.5	45
HNO_3	3.2	50	$C_{18}H_{34}O_2$	1.2	0.81
$C_7H_4O_7$	1.8	3.8	H_2SO_4	0.61	0.68
$HNCO$	1.1	1.2	$C_{14}H_{28}O_2$	0.52	0.85
C_7H_8O	0.9	7.2	$HNCO$	0.45	0.089
$C_3H_6O_3$	0.6	23	$C_{18}H_{36}O_2$	0.32	0.046
$C_6H_5NO_3$	0.53	2.6	$C_{16}H_{32}O_2$	0.30	0.18
$C_4H_6O_5$	0.45	0.30	$C_6H_5O_2$	0.12	8.6
C_6H_5O	0.26	15.6	$C_4H_6O_4$	0.089	6.3
$C_7H_7NO_3$	0.15	4.6	$C_{16}H_{30}O_2$	0.081	0.012
EF_{total} measured by the EEPS	160.9	1289.8	EF_{total} measured by the EEPS	127.7	1320.6
EF_{CIMS}	16.8	320.1	EF_{CIMS}	7.5	237.2
$EF_{\text{CIMS}}/EF_{\text{total}}$ (%)	10.4	25.8	$EF_{\text{CIMS}}/EF_{\text{total}}$ (%)	5.9	17.9

423

424

425 4. Conclusion/ atmospheric implications

426 To address the challenges posed by increasing transportation needs, associated greenhouse gas emissions, and related climate
427 change impacts, biofuels have been promoted as a low-carbon alternative to fossil fuels. In 2020, for the 27 Member States of
428 the European Union, 93.2% of the total fuel supply for road transport was derived from fossil fuels, while 6.8% came from
429 biofuels, with Sweden having the highest biofuel share at 23.2% (Vourliotakis and Platsakis, 2022). This study investigated
430 renewable fuels like rapeseed methyl ester (RME), hydrotreated vegetable oil (HVO), and methane (when using biogas) in
431 terms of primary emissions of pollutants and their secondary formation after photochemical aging. DSL buses without a DPF
432 displayed the highest $EF_{PM:Fresh}$, whereas compressed natural gas (CNG) buses emitted the least, with a median $EF_{PM:Fresh}$ below
433 the detection limit. Despite more than an order of magnitude difference in $EF_{PM:Fresh}$ among buses operated with various fuel
434 types, we observed smaller variations in $EF_{PM:Aged}$, suggesting that secondary particle formation is likely influenced by
435 substantial non-fuel-dependent precursor sources such as lubrication oils and/or fuel additives. Recognizing these sources is
436 crucial for refining regulations on hydrocarbon emissions, which could notably enhance secondary PM control. The median
437 ratios of aged to fresh particle mass emission factors, listed in ascending order, were for diesel (4.0), HVO (6.7), HVO_{HEV}
438 (10.5), RME (10.8), and CNG buses (84), highlighting the significant yet often overlooked contributions of
439 aged/photochemically processed emissions to urban air quality. Furthermore, Zhao et al. (2017) revealed a strongly nonlinear
440 relationship between SOA formation from vehicle exhaust and the ratio of non-methane organic gas to NO_x (NMOG:NO_x).
441 For instance, increasing the NMOG:NO_x from 4 to 10 ppbC/ppbNO_x increased the SOA yield from dilute gasoline vehicle
442 exhaust by a factor of 8, underscoring the importance of integrated emission control policies for NO_x and organic gases for
443 better manage SOA formation. While implementing regulations for secondary particle formation presents significant
444 challenges, these are crucial for a thorough understanding of their impact on regional air quality and health. Our approach to
445 measuring the maximum secondary PM formation potential—peaking at a photochemical age of approximately 5 equivalent
446 days of atmospheric OH exposure—provides a possible semi-quantitative reference for comparing secondary PM formation
447 potential across different studies. We acknowledge the limitations of this approach for direct regulatory application and
448 emphasize the need for more precise and comprehensive research to develop a methodologically robust framework that
449 stakeholders can agree upon for systematically assessing the impacts of vehicle on air quality and informing regulatory
450 strategies.

451

452 It is important to note that the ambient temperature during this study was relatively low, which does not affect the EF
453 comparison across different buses but should be aware of when comparing these results to studies conducted at significantly
454 higher temperatures. Wang et al. (2017) noted lower particle number EFs in summer compared to winter, potentially due to
455 increased nucleation or condensation at cooler temperatures. Temperature impacts on emissions are significant during cold
456 starts when combustion is inefficient (Nam et al., 2010). Post-warm-up, soot mode particles show little temperature
457 sensitivity (Ristimäki et al., 2005). Book et al. (2015) found inconsistent trends in particle emissions from DPF-equipped

458 diesel trucks across various temperatures and driving cycles, suggesting that more research is needed to understand the
459 temperature effects on emissions from different bus types under varied operational conditions.

460

461 Non-regulated chemical species can also have serious negative impacts on air quality and human health. Organic and inorganic
462 acids influence the pH of precipitation and will potentially contribute to acid deposition, affecting ecosystem health.
463 Furthermore, there is a risk that some abatement systems might generate unintended compounds, such as HNCO from the
464 thermal degradation of urea in SCR systems without sufficient hydrolysis. Additionally, Jathar et al. (2017b) observed
465 substantial direct emissions of HNCO from diesel engines and estimated that ambient concentrations in Los Angeles could
466 vary widely, ranging from 20 to 107 ppt depending on different parameterizations of diesel engine emissions. The persistence
467 of HNCO in the atmosphere, particularly under dry conditions, poses significant health risks. It has been linked to severe
468 outcomes including respiratory and cardiovascular disorders, atherosclerosis, cataracts, and rheumatoid arthritis (Leslie et al.,
469 2019; Roberts et al., 2011). In our study, small monoacids (C₁-C₃) and nitrogen-containing compounds, such as nitrous acid
470 (HONO), nitric acid (HNO₃), and HNCO, dominated the fresh gaseous emissions measured by acetate-CIMS for all Euro
471 V/EEV buses regardless of fuel type, with HVO_{HEV} buses exhibiting the highest emissions. Notably, the emission levels of
472 nitrogen-containing compounds were significantly lowered in Euro VI buses, equipped with advanced after-treatment systems
473 that include EGR and DPF technologies in addition to SCR-only techniques. This indicates that transitioning to vehicles
474 equipped with more advanced emission control technologies can be beneficial, even though these technologies may not be
475 specifically designed to target emissions of HONO, HNO₃, and HNCO. Consequently, a detailed evaluation of the
476 environmental and health effects of emerging engine and after-treatment technologies is highly desirable for future
477 considerations. Overall, the extended online chemical characterization of in-use fleet emissions, utilizing advanced techniques
478 like HR-ToF-CIMS, enables the identification of unregulated pollutants, which is crucial for more informed policy decisions
479 and vehicle technology developments.

480

481 ***Data availability.***

482 The data used in this publication are available to the community and can be accessed by request to the corresponding author.

483

484 ***Author contributions.***

485 ÅMH, MLB and QL conducted the measurements. ÅMH designed the project, coordinated the measurements and together
486 with MH and CKC supervised the study. LZ, QL, MLB, CMS and ÅMH carried out the data analysis. LZ, QL, JZY, MH,
487 ÅMH and CKC prepared the manuscript. All co-authors contributed to the discussion and the interpretation of the results.

488

489 ***Competing interests.***

490 The authors declare that they have no known competing financial interests or personal relationships that could have appeared
491 to influence the work reported in this paper.

493 **Acknowledgments.**

494 This work was financed by VINNOVA, Sweden's Innovation Agency (2013-03058) and Formas (2020-01907) and was an
 495 initiative within the framework programme "Photochemical smog in China" financed by the Swedish Research Council (639-
 496 2013-6917). Chak K. Chan would like to acknowledge the support of the National Natural Science Foundation of China
 497 (project no. 41675117 and 41875142).

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