

This is the peer reviewed version of the following article: Lyu, X., Huo, Y., Yang, J., Yao, D., Li, K., Lu, H., ... & Guo, H. (2021). Real-time molecular characterization of air pollutants in a Hong Kong residence: Implication of indoor source emissions and heterogeneous chemistry. *Indoor air*, 31(5), 1340-1352, which has been published in final form at <https://doi.org/10.1111/ina.12826>. This article may be used for non-commercial purposes in accordance with Wiley Terms and Conditions for Use of Self-Archived Versions. This article may not be enhanced, enriched or otherwise transformed into a derivative work, without express permission from Wiley or by statutory rights under applicable legislation. Copyright notices must not be removed, obscured or modified. The article must be linked to Wiley's version of record on Wiley Online Library and any embedding, framing or otherwise making available the article or pages thereof by third parties from platforms, services and websites other than Wiley Online Library must be prohibited.

1 Real-time molecular characterization of air pollutants in a Hong Kong residence: implication of  
2 indoor source emissions and heterogeneous chemistry

3 Xiaopu Lyu, Yunxi Huo, Jin Yang, Dawen Yao, Kaimin Li, Haoxian Lu, Yangzong Zeren, Hai  
4 Guo\*

5 Air Quality Studies, Department of Civil and Environmental Engineering, The Hong Kong  
6 Polytechnic University, Hong Kong, China

7 \* Correspondence to Hai Guo ([hai.guo@polyu.edu.hk](mailto:hai.guo@polyu.edu.hk))

8  
9 **Abstract:** Due to the high health risks associated with indoor air pollutants and long-term exposure,  
10 indoor air quality has received increasing attention. In this study, we put emphasis on the molecular  
11 composition, source emissions and chemical aging of air pollutants in a residence with designed  
12 activities mimicking ordinary Hong Kong homes. More than 150 air pollutants were detected at  
13 molecular level, 87 of which were quantified at a time resolution of not less than 1 hour. The indoor  
14 to outdoor ratios were higher than 1 for most of the primary air pollutants, due to emissions of  
15 indoor activities and indoor backgrounds (especially for aldehydes). In contrast, many secondary  
16 air pollutants exhibited higher concentrations in outdoor air. Painting ranked first in aldehyde  
17 emissions, which also caused great enhancement of aromatics. Incense burning had the highest  
18 emissions of particle-phase organics, with vanillic acid and syringic acid as markers. The other  
19 noteworthy fingerprints enabled by online measurements included linoleic acid, cholesterol and  
20 oleic acid for cooking, 2,5-dimethylfuran, stigmasterol, iso-/anteiso- alkanes and fructose isomers  
21 for smoking, C<sub>28</sub>–C<sub>34</sub> even *n*-alkanes for candle burning, and monoterpenes for the use of air  
22 freshener, cleaning agents and camphor oil. We showed clear evidence of chemical aging of  
23 cooking emissions, giving a hint of indoor heterogeneous chemistry. This study highlights the  
24 value of organic molecules measured at high time resolutions in enhancing our knowledge on  
25 indoor air quality.

## 26 **Practical Implications**

- 27 • Online mass spectrometry technology enabled definite and fast-response detection of a  
28 large number of air pollutants in a residence with implications on source emissions and  
29 indoor chemistry.
- 30 • Incense burning caused intensive emissions of organic matters in both gas and particulate  
31 phases.
- 32 • Distinct tracers identified for nine individual activities facilitated the understanding of  
33 emission characteristics of indoor sources.
- 34 • Chemical aging of cooking emissions was clearly shown, indicating the importance of  
35 indoor heterogeneous chemistry.

36 **Keywords:** Indoor air quality, volatile organic compounds, particulate matters, organic aerosol,  
37 indoor chemistry

38

## 39 1. Introduction

40 People spend up to 90% of their time indoors (Spengler and Sexton, 1983). Indoor air pollution  
41 causes heart disease, pneumonia, stroke, diabetes and lung cancer, resulting in 1.6 million  
42 premature deaths globally in year 2017 (IHME, 2018). Some indoor sources, such as cooking  
43 emissions, also play significant roles in influencing ambient air quality (Lee et al., 2015; Wang et  
44 al., 2020a).

45 Though no consensus has been reached on which air pollutants are most responsible for the health  
46 effects of indoor air pollution, many air pollutants have been confirmed to be harmful to human  
47 health (Salthammer et al., 2010; Chi et al., 2019), such as the gaseous pollutant formaldehyde and  
48 fine particulate matters (PM<sub>2.5</sub>). Therefore, great efforts have been made to understand the  
49 chemical compositions of indoor air pollution as comprehensively as possible (Lee et al., 2002;  
50 Weisel et al., 2008; Krugly et al., 2014; Zhao et al., 2019), plus the other characteristics of these  
51 air pollutants (e.g. size distributions of ultrafine particles) (Morawska et al., 2003). Indoor air  
52 pollutants are generated from a wide range of human activities (cooking, cigarette smoking,  
53 cleaning, human breath, etc.), furnishings, and building & decoration materials (Samet et al., 1987;  
54 Habre et al., 2014; Liu et al., 2016). It is well documented that combustion of solid fuels for  
55 cooking and heating is the leading source of indoor air pollution with most serious health  
56 consequence in developing countries (Ezzati, 2005). Additionally, outdoor air pollution presents  
57 to be a universal source of air pollutants indoors due to air exchange (Blondeau et al., 2005; Chen  
58 et al., 2012). The chemical evolutions of indoor air pollutants have not been studied as extensively  
59 as those in ambient air (Weschler and Carslaw, 2018; Gligorovski and Abbatt, 2018). However,  
60 recent studies discovered the higher-than-expected concentrations of hydroxyl radical (OH) in  
61 special indoor settings, due primarily to the photolysis of combustion-related nitrous acid (HONO)  
62 (Alvarez et al., 2013) and/or terpene-ozone reactions during cleaning (Carslaw et al., 2017).  
63 Besides, the high surface-to-volume ratios indoors enhance the possibility and significance of  
64 surface reactions (Ault et al., 2020; Wang et al., 2020b), accelerating heterogeneous formation of  
65 HONO for example (Gandolfo et al., 2017; Liu et al., 2020). Recently, by applying novel tools for  
66 studying atmospheric chemistry to indoor chemistry, researchers have made compelling  
67 breakthroughs in revealing indoor dynamic source emissions and ozone chemistry (Kristensen et  
68 al., 2019; Liu et al., 2019, 2021). Last but not the least, the occurrence and transformation of indoor  
69 air pollutants are not independent from the ventilation conditions (Godish and Spengler, 1996).

70 Hong Kong, a home of 7.5 million people with a land area of 1,100 km<sup>2</sup>, is one of the cities with  
71 world's smallest living space per capita (<16 m<sup>2</sup> per capita for all housing and <5 for subdivided  
72 apartments) (NYT, 2019). The city is also known for its high densities of population, buildings  
73 and vehicles. Catering is well developed in Hong Kong, forming unique mix of Chinese and  
74 western foods. At a roadside site in vicinity of residential areas in this city, cooking emissions  
75 were found to be responsible for 38.6% (by mass) of the ambient organic aerosols (Lee et al.,  
76 2015). Besides, people in South China including Hong Kong succeed the custom of burning  
77 candles and/or incenses when worshiping ancestors, ghosts and gods in house at daily to monthly  
78 frequency depending on individuals. Other activities, such as smoking, cleaning and laundry, are  
79 also common in Hong Kong homes. The outdoor air pollution is mainly manifested as regional

80 photochemical smog and street-level air pollution resulted from vehicle emissions (Wang et al.,  
81 2017; Zhu et al., 2018). Lastly, warm and humid climate prevails in Hong Kong for more than half  
82 of a year, making it unlikely to stay indoors without air conditioning, which is another factor  
83 influencing indoor air quality in Hong Kong.

84 The concentrations, compositions and sources of regular air pollutants, e.g. volatile organic  
85 compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs) and criteria air pollutants, were  
86 studied in Hong Kong (Lee et al., 2001, 2002; Guo et al., 2003, 2009; Ai et al., 2015). The health  
87 implications of indoor air pollution or environmental factors were also investigated (Yu et al.,  
88 2004). In particular, Yu et al. (2004) and Li et al. (2007) indicated the airborne transmission of  
89 SARS virus in residential building and hospital, and its relationship with ventilation. However, the  
90 previous studies outline the gaps in understanding indoor air pollution in Hong Kong. Firstly, the  
91 molecular compositions of condensed organics, a dominant constituent of indoor particular matters  
92 (Polidori A et al., 2006), were seldom analyzed except for PAHs (Tong et al., 2019). The  
93 measurements lacked either high time resolution or chemical completeness. Secondly, it was  
94 difficult to recognize the emission characteristics of individual indoor sources due to the  
95 cumulative sampling methods. Lastly, we have little knowledge on indoor chemical evolutions in  
96 Hong Kong.

97 In this context, we performed a real-time molecular characterization of indoor air pollutants in a  
98 residence with designed activities mimicking ordinary Hong Kong homes, using a set of online  
99 instruments. This study provides an overview of the sampling campaign, focusing on the  
100 concentrations and origins of air pollutants measured in the residence, emission characteristics of  
101 indoor activities, and the chemical evolution of cooking emissions. The real-time measurements  
102 of indoor air pollutants at the molecular level, especially organics, allowed an unprecedented  
103 insight on indoor air quality in Hong Kong homes.

## 104 2. Methodology

105 As shown in Figure 1, an apartment, consisting of a living room, 2 bedrooms, a kitchen and a  
106 bathroom, was rented for the study. The apartment was located on the 7<sup>th</sup> floor of a 22-year-old  
107 building with 17 floors in a residential area of Kowloon, Hong Kong. In addition to common  
108 sources of air pollutants in cities, e.g. vehicles, restaurants and gas station, two funeral parlors  
109 were located ~220 m south. The floor area of the apartment was 25.4 m<sup>2</sup>, a typical space for an  
110 ordinary Hong Kong family with 3–4 members. The ventilation system comprised 5 windows, 3  
111 air conditioners (ACs), an exhaust fan in the bathroom, and a range hood in the kitchen. The air  
112 exchange rate (AER) of the apartment was measured in a few scenarios, based on the decay of  
113 carbon dioxide (CO<sub>2</sub>) when there was no occupant in the apartment. To maintain comfort and  
114 ensure normal operation of the instruments, at least 1 AC was on all the time. Irrespective of  
115 which AC(s) was (were) on and which door(s) within the suite was (were) open, the AER remained  
116 stable (1.1 – 1.2 h<sup>-1</sup>). It did not change much, unless the range hood was on and the kitchen door  
117 was open, which brought the AER to 2.9 h<sup>-1</sup>.

118

119 Common activities in Hong Kong homes were designed and carried out following the schedules  
120 shown in [Figure 2](#). To mimic the real situations, the activities were conducted at time and  
121 frequencies close to actual since the normal operation of all instruments on 18 November 2019,  
122 while the activity frequencies were lower in the earlier phase of the sampling campaign. The  
123 intensity of the same activity was not necessarily consistent, e.g. simple breakfast *vs.* big dinner,  
124 usually 1 cigarette *vs.* occasionally 2 cigarettes for smoking, and the burning of scented candle *vs.*  
125 ritual candle. The consequent differences in indoor air quality are discussed where necessary.  
126 Natural gas was the main fuel for cooking, with electricity as an alternative when induction cooker  
127 was occasionally used. The types of cigarettes, incenses, and air fresheners were consistent  
128 throughout the study. In most of the daytime, there were 2 occupants in the apartment. The  
129 recorded temperature and relative humidity in the apartment were in the range of 20.0–30.2 °C and  
130 22.1%–57.4%, with an average of 25.5 °C and 33.7%, respectively.

131 As shown in [Figure 1](#), all the instruments were deployed in the living room. The indoor and outdoor  
132 air was sampled through a ~2 m and ~3 m pre-cleaned copper tubing (Restek Cat.# 22629, ID:  
133 0.19") at the flow rate of 16.7 L m<sup>-1</sup>, respectively. Due to the short residence time (<1 s), low wall  
134 loss of charged particles, and relatively low delay on transfer of organic vapors (Deming et al.,  
135 2019), the copper tubing was suitable for simultaneous gas and particle sampling. However, the  
136 partitioning and wall loss could not be completely ruled out. A High Resolution–Time of Flight–  
137 Aerosol Mass Spectrometer (HR-ToF-AMS, hereinafter referred to as AMS) was used to measure  
138 the non-refractory compositions of submicron particulate matters (PM<sub>1</sub>), including organic matters  
139 (PM<sub>1</sub>-OM), sulfate, nitrate, ammonium and chloride. The concentrations were recorded every 2  
140 minutes. Speciated organic aerosol (OA) markers in PM<sub>2.5</sub> were analyzed by a Thermal desorption  
141 Aerosol Gas chromatograph–Time of Flight–Mass Spectrometer (TAG-ToF-MS, hereinafter  
142 referred to as TAG). We identified 92 OA markers according to their retention time and mass  
143 spectra, and quantified the concentrations of 64 species at the time resolution of 1 hour. The  
144 operation principles, measurement configurations, calibrations and data quality control of AMS  
145 and TAG were elaborated in [Lyu et al. \(2020\)](#). The accuracy and precision for the measurements  
146 were within the range of 1.5%–21.2% and 4.1%–33.9%, respectively.

147 The VOCs were detected utilizing a quadruple Proton Transfer Reaction–Mass Spectrometer  
148 (PTR-MS, hereinafter referred to as PTR). The procedures for PTR calibration are described in  
149 [Text S1](#) and [Figure S1](#). 50 VOC species were identified, 12 of which were quantifiable with the  
150 accuracy of 3.0%–19.9% and precision of 1.6%–10.0%. The time resolution of PTR measurement  
151 was 1 minute. Besides, 6 inorganic trace gases, including sulfur dioxide (SO<sub>2</sub>), carbon monoxide  
152 (CO), nitric oxide (NO), nitrogen dioxide (NO<sub>2</sub>), ozone (O<sub>3</sub>) and carbon dioxide (CO<sub>2</sub>), were  
153 monitored by a set of trace gas analyzers at the resolution of 1 minute ([Table S1](#)). All the online  
154 instruments measured the indoor and outdoor air in turn every other hour, which was achieved  
155 through automatic switching of 2 solenoid valves installed on the sampling lines ([Figure 1](#)).  
156 Exhausts of all the instruments were pumped out of the apartment. Due to malfunctions of some  
157 instruments (especially TAG), the periods with valid data for the species measured online were  
158 not exactly the same ([Table S1](#)).

159 In total, we obtained the mixing ratios (or concentrations) for 87 out of 153 air pollutants detected  
160 at the time resolution of 1 minute to 1 hour. While we mainly focus on the quantifiable air  
161 pollutants, the signal intensities of some air pollutants which were not quantified are also adopted  
162 for discussions. The sensitivity of PTR was relatively stable, based on the small changes in K  
163 values between every two calibrations (<7%). For TAG data, the signal intensities were the peak  
164 areas of target compounds scaled by those of corresponding internal standards (ISs), i.e. IS-scaled  
165 peak areas. The ISs, comprising 41 deuterated compounds, were used to track and correct the  
166 changes in instrument sensitivity (Lyu et al., 2020). Therefore, it is reasonable to use the signal  
167 intensities for ratio analyses.

168 To facilitate discussions, the 64 quantifiable OA markers are classified as even *n*-alkanes, odd *n*-  
169 alkanes, PAHs, phenolic acids, phthalic acids, saccharides, sugar alcohols, fatty acids, oleic acid,  
170 dicarboxylic acids (DCA), hydroxyl dicarboxylic acids (OHDCA), Secondary Organic Aerosol  
171 (SOA) tracers of isoprene (Isop-SOA-T), monoterpenes SOA tracers (MT-SOA-T), and 2,3-  
172 dihydroxy-4-oxopentanoic acid (DHOPA). The species in every group are listed in Table S2.

### 173 3. Results and Discussion

#### 174 3.1. Indoor versus outdoor concentrations

175 Figures S2–S5 show the time series of gas-phase and particle-phase air pollutants that were  
176 quantified in this study. The average mixing ratios (or concentrations) in 3 scenarios, i.e. with  
177 indoor activities, without indoor activity and outdoors, are compared in Table 1.

178 Most of the air pollutants with predominantly primary sources exhibited higher levels indoors,  
179 resulting in the indoor-to-outdoor (I/O) ratios of higher than 1. This did not necessarily mean the  
180 more intensive emissions of interior sources but demonstrated the great impacts of interior sources  
181 on indoor air quality, due to proximity to emissions and worse dispersion conditions indoors than  
182 outdoors. Exceptions were NO, NO<sub>2</sub>, SO<sub>2</sub>, chloride, PAHs, phenolic acids and phthalic acids,  
183 whose I/O ratios were close to or even lower than 1. Because of intensive emissions indoors, the  
184 low I/O ratios indicated significant outside emissions, such as NO, NO<sub>2</sub> and PAHs in vehicle  
185 exhausts (Guo et al., 2003), and phenolic acids emitted from burning of worship supplies (mainly  
186 biomass) in nearby funeral parlors (Simoneit et al., 1999). The indoor mixing ratios (or  
187 concentrations) of all the primary air pollutants were considerably elevated in presence of activities,  
188 which partially accounted for the I/O ratios of higher than 1. In absence of activity, the  
189 concentrations of the primary particle compositions were lower than those in ambient air. However,  
190 it was interesting that the mixing ratios of CO<sub>2</sub> and aldehydes indoors without activity still  
191 overwhelmed those outdoors. While the higher CO<sub>2</sub> levels were explained by the breath of  
192 occupants, the more abundant aldehydes in the apartment even in absence of activity were likely  
193 caused by emissions from furnishings and building & decoration materials (Baumann et al., 2000,  
194 Uhde and Salthammer, 2007).

195 In contrast, the I/O ratios were lower than 1 for a majority of secondary air pollutants, excluding  
196 those having primary emissions as alternative sources (e.g. formaldehyde) and oxidation products  
197 of indoor emissions (e.g. 9-oxononanoic acid produced by ozonolysis of oleic acid, see section  
198 3.3). O<sub>3</sub> is a typical photochemical air pollutant in ambient air (Lin et al., 1988), which presented



199 indoors mainly due to air exchange. Sulfate, nitrate and ammonium are well-documented  
200 compositions of secondary inorganic aerosols (Huang et al., 2014). Oxidations of anthropogenic  
201 and biogenic VOCs lead to the formations of DCA, OHDCA, Isop-SOA-T, MT-SOA-T and  
202 DHOPA (Blando and Turpin, 2000, Hu et al., 2008), while DCA is also derived from primary  
203 emissions. According to the PTR data that were not quantified, the signal intensities of some VOC  
204 precursors of these SOA markers indoors were higher than those outdoors, such as isoprene (Isop-  
205 SOA-T precursor), monoterpenes (MT-SOA-T precursors), and toluene (one of DHOPA  
206 precursors). Therefore, the lower mixing ratios (or concentrations) of the secondary air pollutants  
207 indoors most likely indicated weaker oxidative capacity of the indoor air than ambient air.

### 208 3.2. Emission characteristics of indoor activities

209 As discussed, indoor activities caused obvious enhancements of primary air pollutants. Figure 3  
210 shows the variations of some selected air pollutants on one day with intensive activities as an  
211 example. Increases of NO, NO<sub>2</sub> and PM<sub>1</sub>-OM were observed during cooking noodles in the  
212 morning with range hood off. While all of them could be emitted from natural gas combustion  
213 (Singer et al., 2017), the increase of PM<sub>1</sub>-OM was also associated with the stir fry of side dishes,  
214 in the light of rising fractions of saccharides and oleic acid. Differently, making lunch and dinner  
215 resulted in much less pronounced increases of these air pollutants, though more dishes were made.  
216 A plausible explanation was the effect of the range hood, which was turned on during both cooking  
217 periods. We compared the indoor mixing ratios (or concentrations) of some representative air  
218 pollutants with and without range hood exhaust during 2 cooking practices, when the range hood  
219 was turned on intermittently. As shown in Table S3, the indoor levels of these species were reduced  
220 by 4% – 81% with the range hood exhaust, confirming the effectiveness of the range hood in  
221 removing cooking emissions from the apartment. Besides, the fraction of oleic acid increased  
222 markedly during making lunch and dinner, due to the use of larger amount of oil than in making  
223 breakfast.

224 The occupant smoked in the apartment at ~12:20 and ~18:00, causing tremendous rise in the  
225 concentrations of PM<sub>1</sub>-OM and nitrate, as well as the mixing ratios of many VOCs and CO.  
226 Saccharides experienced the largest enhancements among the OA markers that were quantified,  
227 thereby an obvious increase of the proportion in the total concentration of OA makers. A ritual  
228 candle was burned during 22:21–22:40 in the kitchen. Correspondingly, we observed notable  
229 increases of NO, NO<sub>2</sub>, SO<sub>2</sub>, CO and *n*-alkanes (especially even *n*-alkanes). Spraying air freshener  
230 in the morning (~10:25) was responsible for the enhancements of formaldehyde and acetaldehyde.  
231 Lastly, makeup in the morning and eating fruit (persimmon) & drinking tea in the afternoon did  
232 not cause significant changes in the mixing ratios (or concentrations) of air pollutants discussed  
233 here. However, the use of camphor oil as a repellent in the makeup substantially increased the  
234 signal intensity of monoterpenes (see section 3.3).

235 Furthermore, we calculated the net increments (NIC) of the 12 quantifiable aldehydes during  
236 individual indoor activities, with the formula of  $NIC = C_{w/A} - C_{w/oA}$ .  $C_{w/A}$  and  $C_{w/oA}$  represent  
237 the mixing ratios with and without the presence of indoor activities, respectively.  $C_{w/A}$  were the  
238 averages over the rising, plateau (if any) and decline periods of the mixing ratios, and the time  
239 frames for calculating  $C_{w/oA}$  were the periods before the activities within the same hours the

240 activities were conducted. Since the sampling switched to outdoor air in odd hours, the decline  
241 periods were not complete in some cases. Therefore, the NICs were somewhat overestimated.  
242 However, we do not think this had a great impact on the fractions of individual aldehydes in the  
243 total NIC of the 12 aldehydes, bearing in mind the discrepancies in decay rates of different  
244 aldehydes.

245 [Figure 4](#) shows the total NIC of the 12 aldehydes during individual indoor activities and the  
246 apportionments. Overall, painting ranked the first in the total NIC, followed by incense burning  
247 and smoking. Ritual candle burning had the lowest NIC of aldehydes. Butanal contributed 54.6%  
248 to the total NIC during painting, which even played more predominant role (82.1%) in the total  
249 NIC attributable to cleaning. Pentanal, formaldehyde and butanal were the main aldehydes emitted  
250 from incense burning (smoking), contributing 40.9% (43.5%), 17.1% (12.1%) and 14.3% (16.6%)  
251 to the total NIC, respectively. Emissions from spraying air freshener mainly consisted of  
252 formaldehyde, butanal and propanal. Hexanal accounted for 72.2% of the total NIC of the 12  
253 aldehydes in emissions of scented candle burning. Pentanal, formaldehyde and hexanal were the  
254 main constituents of cooking emissions. While hexanal dominated the total NIC during the use of  
255 camphor oil, other aldehydes (such as heptanal, propanal and decanal) also made considerable  
256 contributions. The total NIC during ritual candle burning was mainly attributed to formaldehyde  
257 (44.6%), propanal (15.4%), decanal (14.1%) and acrolein (10.3%). It should be noted that the pie  
258 areas in [Figure 4](#) are proportional to the total emissions of the 12 aldehydes instead of all VOCs.  
259 Many VOCs with great enhancements in some activities were not quantified, such as ethanol in  
260 spraying air freshener and monoterpenes in cleaning and camphor oil use (see section 3.3).

261 Similarly, the NICs of particle-phase air pollutants were calculated, as shown in [Figure 5](#).  
262 Activities that did not cause significant changes in PM<sub>1</sub> compositions are not discussed. It is  
263 striking that the PM<sub>1</sub>-OM contributed more than 95% of the total NIC of PM<sub>1</sub> during cooking and  
264 all the activities involving combustion except scented candle burning. The burning of scented  
265 candles only increased PM<sub>1</sub> concentration by  $5.2 \pm 1.7 \mu\text{g m}^{-3}$  on average, which was 6.6–37 times  
266 lower than the NICs of PM<sub>1</sub> caused by the other activities. During scented candle burning,  $76 \pm 0.4\%$   
267 of the NIC of PM<sub>1</sub> was attributed to PM<sub>1</sub>-OM (NIC:  $4.0 \pm 1.7 \mu\text{g m}^{-3}$ ), with the rest contributed by  
268 sulfate (~16%) and ammonium (~7%). In contrast, incense burning caused the largest NIC of PM<sub>1</sub>-  
269 OM ( $189 \pm 45 \mu\text{g m}^{-3}$ ), followed by smoking ( $119 \pm 15 \mu\text{g m}^{-3}$ ), cooking ( $41 \pm 7.3 \mu\text{g m}^{-3}$ ) and  
270 ritual candle burning ( $33 \pm 7 \mu\text{g m}^{-3}$ ). While continuous emissions were observed during the other  
271 combustion activities, scented candle burning mainly led to enhancement of PM<sub>1</sub>-OM in the  
272 ignition and extinction states, with much higher concentrations in the extinction state. This was  
273 due to the emissions of larger size particles in the smoldering phase immediately after the candle  
274 was extinguished ([Manoukian et al., 2013](#)). For ritual candle burning, smoke plumes were  
275 discernable even in the flaming phase, which might explain the different emission patterns.  
276 Furthermore, we investigated the elemental compositions of PM<sub>1</sub>-OM detected by AMS in indoor  
277 activities. The hydrogen to carbon atomic ratios (H:C) and oxygen to carbon atomic ratios (O:C)  
278 are plotted in the right panel of [Figure 5](#). Highest H:C ratios ( $1.94 \pm 0.02$ ) and lowest O:C ratios  
279 ( $0.23 \pm 0.02$ ) were observed during ritual candle burning. The features, especially the proximity of  
280 average H:C ratio to that in long acyclic alkanes (~2), indicated strong emissions of high saturation  
281 molecules (such as alkanes) from ritual candle burning ([Heald et al., 2010](#)). In contrast, incense

282 burning witnessed the lowest H:C ratios ( $1.76\pm 0.01$ ) and highest O:C ratios ( $0.42\pm 0.01$ ), implying  
283 multiple oxygen-containing functional groups in the molecules emitted from incense burning  
284 (Heald et al., 2010). These inferences are verified by the enhancement ratios (ERs) of speciated  
285 OA markers below, while the NICs of them will be presented in a subsequent paper.

### 286 3.3. Enhancement ratios

287 Since many air pollutants (especially VOCs) that were not quantified experienced significant  
288 enhancements in signal intensities during some indoor activities, neglecting them might lead to  
289 incomplete or even biased understandings on emission characteristics of indoor activities. To make  
290 use of the unquantified species, the ERs are calculated for all the detected air pollutants following  
291 the formula of  $ER = C_{w/A} / C_{w/o A}$ . While  $C_{w/A}$  and  $C_{w/o A}$  have been defined for the quantified  
292 species, they represent the signal intensities for the unquantified species with and without the  
293 indoor activities, respectively. Limited by the time resolution of 1 hour for TAG data, we  
294 calculated  $C_{w/o A}$  on the days without activity and in the same hours as  $C_{w/A}$  was calculated.

295 Table 2 lists the top 5 ERs for indoor gaseous and particulate air pollutants. Cooking resulted in  
296 high ERs for inorganic trace gases (NO, CO and NO<sub>2</sub>), due to the combustion of natural gas (Singer  
297 et al., 2017). Ethanol also had an ER of 1.9, which was caused by the occasional use of alcohol-  
298 containing ingredients, such as fermented rice. Cyclohexane was measured in cooking emissions  
299 with the emission rates lower than many other VOCs (e.g. formaldehyde, acetaldehyde and 2-  
300 butanone) in previous studies (Alves et al., 2015; He et al., 2020). The reason for the high ER of  
301 cyclohexane in this study is unknown. The particle-phase species with high ERs in cooking  
302 included linoleic acid, cholesterol, arabinitol/xylitol,  $\beta$ -sitosterol and oleic acid. Linoleic acid,  $\beta$ -  
303 sitosterol and oleic acid were most likely derived from the oils (corn oil and peanut oil) and  
304 ingredients (Zhao et al., 2007). Cholesterol and arabinitol/xylitol were mainly derived from the  
305 ingredients, because they were enhanced when fried eggs & fishes and stir-fried vegetables were  
306 made, respectively. The most noteworthy gaseous air pollutant during smoking was 2,5-  
307 dimethylfuran whose ER was 4.3. 2,5-dimethylfuran was proposed as an excellent marker for  
308 smoking in blood and breath samples (Ashley et al., 1996; Castellanos et al., 2019). In particle  
309 phase, the iso-/anteiso-alkanes were observed with high ERs during smoking, consistent with  
310 previous studies (Cass, 1998). Stigmasterol, which had the highest ER (269), is one of the most  
311 abundant plant sterols and has often been detected with great abundance in smoking emissions  
312 (Rogge et al., 1994). Simple sugars (e.g. fructose, glucose and sucrose) are important components  
313 and common additives in tobacco, presenting at levels of up to 20% in tobacco by mass (Jansen et  
314 al., 2014). This explained the notable enhancement of fructose isomers (ER = 162) during smoking.  
315 The gaseous air pollutants differentiating incense burning from the other activities were benzene,  
316 propene and acetaldehyde. Vanillic acid and syringic acid, pyrolysis products of lignin (Simoneit  
317 et al., 1993), were unique markers of incense burning, with the ER of 108 and 80, respectively.  
318 While C<sub>28</sub>–C<sub>34</sub> even *n*-alkanes were significantly enhanced during the burning of both scented  
319 candle and ritual candle, the former witnessed enhancements of fatty acids (stearic acid and  
320 palmitic acid),  $\beta$ -sitosterol and fructose isomers, in contrast to the C<sub>27</sub>–C<sub>35</sub> odd alkanes with  
321 different structures for the latter. It should be noted that the higher ERs for C<sub>28</sub>–C<sub>34</sub> even *n*-alkanes  
322 were due to the lower  $C_{w/o A}$ . In fact, the NIC of C<sub>27</sub>–C<sub>35</sub> odd *n*-alkanes was comparable to that of



323 C<sub>28</sub>–C<sub>34</sub> even *n*-alkanes for both candles, consistent with the emissions of paraffin candles (Fine  
324 et al., 1999). The high ERs of alkanes in ritual candle burning were roughly consistent with the  
325 H:C ratio of 1.94±0.02 detected by AMS. Besides, the gaseous air pollutants with top 5 ERs were  
326 also different (except NO) for the burning of two types of candles.

327 The other indoor activities did not lead to significant increases of particle-phase air pollutants, but  
328 the signal intensities of some VOCs were obviously enhanced. For example, the ER of  
329 monoterpenes (limonene/ $\alpha$ -pinene/ $\beta$ -pinene) reached 196, 10 and 3.7, when we used camphor oil,  
330 air freshener and cleaning agents, respectively. Air freshener was also a strong source of ethanol  
331 (ER = 129). Styrene, formaldehyde and linalool/geraniol with ERs of 3.0–7.5 were widely detected  
332 in air freshener spray (Kim et al., 2015; Steinemann, 2017). Painting caused tremendous  
333 enhancements of aromatics and acrolein. Studies confirmed considerable emissions of aromatics  
334 in paint applications (Yuan et al., 2010). Acrolein, as a combustion by-product of organics  
335 (Seaman et al., 2007), however was not identified to be associated with painting. The reason for  
336 its enhancement is unknown and should be referenced with caution.

### 337 3.4. Chemical evolution of cooking emissions

338 Chemical evolution of indoor air pollutants has significant impacts on the fate and subsequent  
339 health risk of primary emissions. In this study, the aging of cooking emissions provided some  
340 insights into the indoor heterogeneous chemistry. Studies (Vesna et al., 2009; Zeng et al., 2020)  
341 indicated that unsaturated fatty acids (e.g. oleic acid and linoleic acid) were prone to be oxidized  
342 by O<sub>3</sub> and OH radical via heterogeneous reactions. The oxidation products, i.e. nonanal and  
343 nonanoic acid were too volatile to be measured by TAG. Azelaic acid and 9-oxononanoic acid, the  
344 other two oxidation products of cooking emissions, were reasonably detected. However,  
345 simultaneous enhancements of azelaic acid and oleic acid were observed during cooking,  
346 indicating the primary emissions of azelaic acid, which was confirmed by previous studies (He et  
347 al., 2004). Therefore, we select 9-oxononanoic acid as a representative of the oxidation products.  
348 Since 9-oxononanoic acid was not quantified due to lack of standard, the IS-scaled peak areas were  
349 used in the analyses below.

350 Figure 6(a)–(b) shows the diurnal patterns of 9-oxononanoic acid and O<sub>3</sub>. The average IS-scaled  
351 peak area of 9-oxononanoic acid indoors (0.97±0.11) was significantly ( $p<0.01$ ) higher than that  
352 outdoors (0.72±0.11). For both indoor and outdoor profiles, bimodal distributions were observed  
353 with one peak in the afternoon and the other peak in the evening. The daytime variations of 9-  
354 oxononanoic acid coincided fairly well with the patterns of O<sub>3</sub>. It is expected that the indoor OH  
355 had the peak at noon, when the ambient OH reached highest levels and the HONO photolysis  
356 indoors was most intensive driven by the elevated HONO levels during making lunch and strongest  
357 sunlight that penetrated windows. Therefore, the afternoon peak of 9-oxononanoic acid was likely  
358 explained by the OH-initiated oxidations of cooking emissions and the peak delay relative to OH  
359 might be due to the oxidation processes of several generations (Zeng et al., 2020). However, the  
360 possible O<sub>3</sub>-involved formation pathways could not be excluded (Vesna et al., 2009). It seemed  
361 that O<sub>3</sub> alone was not accountable for the higher levels of 9-oxononanoic acid indoors and in the  
362 evening, where and when the O<sub>3</sub> mixing ratios were relatively low. A plausible explanation was  
363 that the abundance of precursors was another influencing factor. For example, oleic acid was more

364 abundant in indoor air, and reached its highest concentrations in making dinner, which fueled the  
365 production of 9-oxononanoic acid in the evening. Besides, the high levels of indoor 9-oxononanoic  
366 acid might be partially explained by OH-initiated oxidations, especially when HONO levels were  
367 elevated after cooking. The factors influencing the yields of 9-oxononanoic acid are also worth  
368 exploring.

369 Furthermore, we examined the levels of 9-oxononanoic acid and oleic acid in the indoor samples  
370 collected during cooking and two hours later. A marginal increase ( $p=0.29$ ) was identified for 9-  
371 oxononanoic acid within the 2 hours when oleic acid decreased remarkably (Figure 6(c)). As a  
372 result, the ratio of 9-oxononanoic acid/oleic acid went up (Figure 6(d)). At the same time, the  
373 decrease of oleic acid/stearic acid ratio (Figure 6(d)) was a sign of chemical aging of cooking  
374 emissions, because oleic acid is more reactive than stearic acid (Wang et al., 2020). It is noteworthy  
375 that some indoor activities were occasionally conducted 2 hours after cooking, which might  
376 interfere with the chemical compositions of TAG samples. However, the conclusion will still hold  
377 even if all the samples subject to interference of other activities are filtered out.

378 In general, there was no activity with intensive emissions after making dinner till the next morning,  
379 except for a few times of candle burning. Therefore, it provided a good opportunity to look into  
380 the continuous variations of the fingerprints of cooking emissions and the chemical evolution. As  
381 shown in Figures 6(e)–(f), the level of 9-oxononanoic acid gradually increased after cooking,  
382 reaching the maximum 4–8 hours later when oleic acid decreased to the lowest level. We noticed  
383 that the increase rate of 9-oxononanoic acid within the first 2 hours ( $9.4\% \text{ hr}^{-1}$ ) was much lower  
384 than the decrease rate of oleic acid ( $-45.8\% \text{ hr}^{-1}$ ). This might be due to the fact that 9-oxononanoic  
385 acid is not the first-generation product of the unsaturated fatty acids (Vesna et al., 2009), hence  
386 time difference between the degradation of the initial precursors and production of 9-oxononanoic  
387 acid. As expected, the ratio of 9-oxononanoic acid/oleic acid increased within the first 10 hours,  
388 and decreased subsequently due to some fresh emissions of oleic acid in making breakfast and  
389 decay of 9-oxononanoic acid. A continuous decline was observed for the ratio of oleic acid/stearic  
390 acid, till the last 2 hours. Overall, the dynamic characteristics of all the fingerprints analyzed from  
391 TAG data clearly indicated the chemical aging of cooking emissions, even at night.

#### 392 4. Conclusions

393 Indoor air quality is a big health concern in Hong Kong, where people have to endure long working  
394 hours, small living space and a wide range of indoor sources of air pollutants. This study focused  
395 on air pollution in Hong Kong homes, one of the most important indoor environments. We  
396 deployed a set of online instruments in a residence for 3–4 occupants. Air pollutants in gas and  
397 particle phases were chemically characterized at molecular level and improved time resolutions.  
398 The origins for most of the primary and secondary air pollutants were unambiguously  
399 distinguishable, based on not only the established knowledge but also the stratified I/O ratios with  
400 1 as the benchmark. While the presence of indoor activities was an essential reason for the higher  
401 concentrations of primary air pollutants indoors, aldehydes had I/O ratios higher than 1 even in  
402 absence of any activity. Emission characteristics of 9 indoor activities were thoroughly revealed.  
403 Painting and incense burning had the highest emissions of aldehydes and  $\text{PM}_{1\text{-OM}}$ , respectively.  
404 The molecular fingerprints of individual indoor activities were identified, such as linoleic acid and

405 oleic acid in cooking emissions, and 9-oxononanoic acid as one of their oxidation products. Based  
406 on the evolutions of relevant fingerprints, the chemical aging of cooking emissions indoors was  
407 clearly indicated. The high apartment rent, large sizes of the instruments and intensive labor in  
408 transferring, deploying and maintaining the instruments hamper widespread application of the  
409 online measurements in more dwellings. However, the indoor sources and chemistry examined are  
410 expected to be ubiquitous in this region. Therefore, this study gives unprecedented insights into  
411 the dynamic profiles and chemical evolutions of air pollutants that may be present in ordinary  
412 Hong Kong homes, and is also a prototype for more in-depth indoor air quality studies.

413

#### 414 **Acknowledgements**

415 This study was supported by the University Strategic Importance scheme (1-ZE1M) and the  
416 Strategic Focus Area scheme of The Research Institute for Sustainable Urban Development (1-  
417 BBW9) at The Hong Kong Polytechnic University. The advices on PTR operation and data quality  
418 control from Dr. Erik Uhde (Fraunhofer WKI) and Dr. Yingjun Liu (Peking University) are  
419 sincerely acknowledged. The authors thank Dr. Zhirong Liang, Mr. Zhe Qian and Mr. Simiao  
420 Wang for their help in the sampling campaign.

421

#### 422 **References**

423 Alvarez EG, Amedro D, Afif C, et al. Unexpectedly high indoor hydroxyl radical concentrations  
424 associated with nitrous acid. *Proc Natl Acad Sci USA*. 2013;**110**: 13294-13299.

425 Ashley DL, Bonin MA, Hamar B, McGeehin M. Using the blood concentration of 2,5-  
426 dimethylfuran as a marker for smoking. *Int Arch Occup Environ Health*. 1996; **68**:183-187.

427 Alves CA, Evtyugina M, Cerqueira M, Nunes T, Duarte M, Vicente E. Volatile organic  
428 compounds emitted by the stacks of restaurants. *Air Qual Atmos Health*. 2015; **8**: 401-412.

429 Ault AP, Grassian VH, Carslaw N, et al. Indoor Surface Chemistry: Developing a Molecular  
430 Picture of Reactions on Indoor Interfaces. *Chem*. 2020; **6**: 3203-3218.

431 Ai ZT, Mak CM, Cui DJ. On-site measurements of ventilation performance and indoor air quality  
432 in naturally ventilated high-rise residential buildings in Hong Kong. *Indoor Built Environ*. 2015;  
433 **24**: 214-224.

434 Blondeau P, Iordache V, Poupard O, Genin D, Allard F. Relationship between outdoor and indoor  
435 air quality in eight French schools. *Indoor air*. 2005; **15**: 2-12.

436 Baumann MG, Lorenz LF, Batterman SA, Zhang GZ. Aldehyde emissions from particleboard and  
437 medium density fiberboard products. *For Prod J*. 2000; **50**: 75-82.

438 Blando JD, Turpin BJ. Secondary organic aerosol formation in cloud and fog droplets: a literature  
439 evaluation of plausibility. *Atmospheric Environ*. 2000; **34**: 1623-1632.

440 Cass GR. Organic molecular tracers for particulate air pollution sources. *Trends Analyt Chem.*  
441 1998; **17**: 356-366.

442 Chi R, Chen C, Li H, Pan L, Zhao B, Deng F, Guo X. Different health effects of indoor - and  
443 outdoor - originated PM<sub>2.5</sub> on cardiopulmonary function in COPD patients and healthy elderly  
444 adults. *Indoor air.* 2019; **29**: 192-201.

445 Carslaw N, Fletcher L, Heard D, Ingham T, Walker H. Significant OH production under surface  
446 cleaning and air cleaning conditions: Impact on indoor air quality. *Indoor Air.* 2017; **27**: 1091-  
447 1100.

448 Castellanos M, Suñer R, Fernández-Real JM, Sanchez JM. 2,5-dimethylfuran as a validated  
449 biomarker of smoking status. *Nicotine Tob Res.* 2019; **21**: 828-834.

450 Chen C, Zhao B, Weschler CJ. Indoor Exposure to 'Outdoor PM<sub>10</sub>': Assessing Its Influence on the  
451 Relationship Between PM<sub>10</sub> and Short-term Mortality in US Cities. *Epidemiology.* 2012; **23**: 870-  
452 878.

453 Ezzati M. Indoor air pollution and health in developing countries. *Lancet.* 2005; **366**:104-106.

454 Fine PM, Cass GR, Simoneit BR. Characterization of fine particle emissions from burning church  
455 candles. *Environ Sci Technol.* 1999; **33**: 2352-2362.

456 Gligorovski S, Abbatt JP. An indoor chemical cocktail. *Science.* 2018; **359**: 632-633.

457 Guo H, Kwok NH, Cheng HR, Lee SC, Hung WT, Li YS. Formaldehyde and volatile organic  
458 compounds in Hong Kong homes: concentrations and impact factors. *Indoor air.* 2009; **19**: 206-  
459 217.

460 Guo H, Lee SC, Ho KF, Wang XM, Zou SC. Particle-associated polycyclic aromatic hydrocarbons  
461 in urban air of Hong Kong. *Atmospheric Environ.* 2003; **37**: 5307-5317.

462 Guo H, Lee SC, Li WM, Cao JJ. Source characterization of BTEX in indoor microenvironments  
463 in Hong Kong. *Atmospheric Environ.* 2003; **37**: 73-82.

464 Gandolfo A, Rouyer L, Wortham H, Gligorovski S. The influence of wall temperature on NO<sub>2</sub>  
465 removal and HONO levels released by indoor photocatalytic paints. *Appl. Catal. B-Environ.* 2017;  
466 **209**: 429-436.

467 Godish T, Spengler JD. Relationships between ventilation and indoor air quality: a review. *Indoor*  
468 *Air.* 1996; **6**: 135-145.

469 Hu D, Bian Q, Li TW, Lau AK, Yu JZ. Contributions of isoprene, monoterpenes,  $\beta$ -caryophyllene,  
470 and toluene to secondary organic aerosols in Hong Kong during the summer of 2006. *J Geophys*  
471 *Res Atmos.* 2008; **113**: doi:10.1029/2008JD010437.

472 Habre R, Coull B, Moshier E, et al. Sources of indoor air pollution in New York City residences  
473 of asthmatic children. *J Expo Sci Environ Epidemiol.* 2014; **24**: 269-278.

474 He LY, Hu M, Huang XF, Yu BD, Zhang YH, Liu DQ. Measurement of emissions of fine  
475 particulate organic matter from Chinese cooking. *Atmospheric Environ.* 2004; **38**: 6557-6564.

476 Heald CL, Kroll JH, Jimenez JL, et al. A simplified description of the evolution of organic aerosol  
477 composition in the atmosphere. *Geophys. Res. Lett.* 2010; **37**: doi:10.1029/2010GL042737.

478 He WQ, Shi AJ, Shao X, Nie L, Wang TY, Li GH. Insights into the comprehensive characteristics  
479 of volatile organic compounds from multiple cooking emissions and after treatment control  
480 technologies application. *Atmospheric Environ.* 2020; **240**: 117646.

481 Huang RJ, Zhang Y, Bozzetti C, et al. High secondary aerosol contribution to particulate pollution  
482 during haze events in China. *Nature.* 2014; **514**: 218-222.

483 Institute for Health Metrics and Evaluation (IHME). Findings from the Global Burden of Disease  
484 Study 2017. Seattle, WA: IHME, 2018.

485 Jansen E, Cremers J, Borst S, Talhout R. Simple determination of sugars in cigarettes. *J Anal*  
486 *Bioanal Tech.* 2014; **5**: 1.

487 Kim S, Hong SH, Bong CK, Cho MH. Characterization of air freshener emission: the potential  
488 health effects. *J Toxicol Sci.* 2015; **40**: 535-550.

489 Kristensen K, Lunderberg DM, Liu Y, et al. Sources and dynamics of semivolatile organic  
490 compounds in a single-family residence in northern California. *Indoor Air.* 2019; **29**: 645-655.

491 Krugly E, Martuzevicius D, Sidaraviciute R, et al. Characterization of particulate and vapor phase  
492 polycyclic aromatic hydrocarbons in indoor and outdoor air of primary schools. *Atmospheric*  
493 *Environ.* 2014; **82**: 298-306.

494 Liu J, Deng H, Lakey PS, et al. Unexpectedly High Indoor HONO Concentrations Associated with  
495 Photochemical NO<sub>2</sub> Transformation on Glass Windows. *Environ. Sci. Technol.* 2020; **54**: 15680-  
496 15688.

497 Lyu X, Guo H, Yao D, et al. In Situ Measurements of Molecular Markers Facilitate Understanding  
498 of Dynamic Sources of Atmospheric Organic Aerosols. *Environ Sci Technol.* 2020; **54**:11058-  
499 11069.

500 Li Y, Huang X, Yu IT, Wong TW, Qian H. Role of air distribution in SARS transmission during  
501 the largest nosocomial outbreak in Hong Kong. *Indoor air.* 2005; **15**: 83-95.

502 Lee SC, Li WM, Ao CH. Investigation of indoor air quality at residential homes in Hong Kong—  
503 case study. *Atmospheric Environ.* 2002; **36**: 225-237.

504 Lee SC, Li WM, Chan LY. Indoor air quality at restaurants with different styles of cooking in  
505 metropolitan Hong Kong. *Sci Total Environ.* 2001; **279**: 181-193.

506 Liu S, Li R, Wild RJ, et al. Contribution of human-related sources to indoor volatile organic  
507 compounds in a university classroom. *Indoor Air.* 2016; **26**: 925-938.



508 Lee BP, Li YJ, Yu JZ, Louie PK, Chan CK. Characteristics of submicron particulate matter at the  
509 urban roadside in downtown Hong Kong-Overview of 4 months of continuous high-resolution  
510 aerosol mass spectrometer measurements. *J Geophys Res Atmos.* 2015; **120**: 7040-7058.

511 Liu Y, Misztal PK, Arata C, Weschler CJ, Nazaroff WW, Goldstein AH. Observing ozone  
512 chemistry in an occupied residence. *Proc. Natl. Acad. Sci. U.S.A.* 2021; **118**: e2018140118;  
513 doi.org/10.1073/pnas.2018140118.

514 Liu Y, Misztal PK, Xiong J, et al. Characterizing sources and emissions of volatile organic  
515 compounds in a northern California residence using space- and time-resolved measurements.  
516 *Indoor Air.* 2019; **29**: 630-644.

517 Lin X, Trainer M, Liu SC. On the nonlinearity of the tropospheric ozone production. *J Geophys*  
518 *Res Atmos.* 1988; **93**: 15879-15888.

519 Morawska L, He C, Hitchins J, Mengersen K, Gilbert D. Characteristics of particle number and  
520 mass concentrations in residential houses in Brisbane, Australia. *Atmospheric Environ.* 2003; **37**:  
521 4195-4203.

522 Manoukian A, Quivet E, Temime-Roussel B, Nicolas M, Maupetit F, Wortham H. Emission  
523 characteristics of air pollutants from incense and candle burning in indoor atmospheres. *Environ.*  
524 *Sci. Pollut. Res.* 2013; **20**: 4659-4670.

525 Polidori A, Turpin B, Meng QY, et al. Fine organic particulate matter dominates indoor-generated  
526 PM<sub>2.5</sub> in RIOPA homes. *J Expo Sci Environ Epidemiol.* 2006; **16**: 321-331.

527 Rogge WF, Hildemann LM, Mazurek MA, Cass GR, Simoneit BR. Sources of fine organic aerosol.  
528 6. Cigaret smoke in the urban atmosphere. *Environ Sci Technol.* 1994; **28**: 1375-1388.

529 Steinemann A. Ten questions concerning air fresheners and indoor built environments. *Build*  
530 *Environ.* 2017; **111**: 279-284.

531 Seaman VY, Bennett DH, Cahill TM. Origin, occurrence, and source emission rate of acrolein in  
532 residential indoor air. *Environ Sci Technol.* 2007; **41**: 6940-6946.

533 Salthammer T, Mentese S, Marutzky R. Formaldehyde in the indoor environment. *Chem Rev.* 2010;  
534 **110**: 2536-2572.

535 Samet JM, Marbury MC, Spengler JD. Health effects and sources of indoor air pollution. Part I.  
536 *Am J Respir Crit Care Med.* 1987; **136**: 1486-1508.

537 Singer BC, Pass RZ, Delp WW, Lorenzetti DM, Maddalena RL. Pollutant concentrations and  
538 emission rates from natural gas cooking burners without and with range hood exhaust in nine  
539 California homes. *Build Environ.* 2017; **122**: 215-229.

540 Spengler JD, Sexton K. Indoor air pollution: a public health perspective. *Science.* 1983; **221**: 9-17.

541 Simoneit BR, Schauer JJ, Nolte CG, et al. Levoglucosan, a tracer for cellulose in biomass burning  
542 and atmospheric particles. *Atmospheric Environ.* 1999; **33**: 173-182.

543 Tong X, Chen XC, Chuang HC, Cao JJ, Ho SS, Lui KH, Ho KF. Characteristics and cytotoxicity  
544 of indoor fine particulate matter (PM<sub>2.5</sub>) and PM<sub>2.5</sub>-bound polycyclic aromatic hydrocarbons  
545 (PAHs) in Hong Kong. *Air Qual Atmos Health*. 2019; **12**: 1459-1468.

546 The New York Times. Tiny Apartments and Punishing Work Hours: The Economic Roots of Hong  
547 Kong's Protests. [https://www.nytimes.com/interactive/2019/07/22/world/asia/hong-kong-](https://www.nytimes.com/interactive/2019/07/22/world/asia/hong-kong-housing-inequality.html)  
548 [housing-inequality.html](https://www.nytimes.com/interactive/2019/07/22/world/asia/hong-kong-housing-inequality.html).

549 Uhde E, Salthammer T. Impact of reaction products from building materials and furnishings on  
550 indoor air quality—a review of recent advances in indoor chemistry. *Atmospheric Environ*. 2007;  
551 **41**: 3111-3128.

552 Weisel CP, Alimokhtari S, Sanders PF. Indoor air VOC concentrations in suburban and rural New  
553 Jersey. *Environ Sci Technol*. 2008; **42**: 8231-8238.

554 Weschler CJ, Carslaw N. Indoor chemistry. *Environ Sci Technol*. 2018; **52**: 2419-2428.

555 Wang C, Collins DB, Arata C, et al. Surface reservoirs dominate dynamic gas-surface partitioning  
556 of many indoor air constituents. *Sci Adv*. 2020; **6**: eaay8973.

557 Wang Q, He X, Zhou M, et al. Hourly Measurements of Organic Molecular Markers in Urban  
558 Shanghai, China: Primary Organic Aerosol Source Identification and Observation of Cooking  
559 Aerosol Aging. *ACS Earth Space Chem*. 2020; **4**: 1670-1685.

560 Wang Y, Wang H, Guo H, et al. Long-term O<sub>3</sub>-precursor relationships in Hong Kong: field  
561 observation and model simulation. *Atmospheric Chem Phys*. 2017; **17**: 10919-10935.

562 Yu IT, Li Y, Wong TW, et al. Evidence of airborne transmission of the severe acute respiratory  
563 syndrome virus. *N Engl J Med*. 2004; **350**: 1731-1739.

564 Yuan B, Shao M, Lu S, Wang B. Source profiles of volatile organic compounds associated with  
565 solvent use in Beijing, China. *Atmospheric Environ*. 2010; **44**: 1919-1926.

566 Zhu Y, Chan KL, Lam YF, et al. Analysis of spatial and temporal patterns of on-road NO<sub>2</sub>  
567 concentrations in Hong Kong. *Atmos Meas Tech*. 2018; **11**: 6719-6734.

568 Zhao Y, Chen C, Zhao B. Emission characteristics of PM<sub>2.5</sub>-bound chemicals from residential  
569 Chinese cooking. *Build Environ*. 2019; **149**: 623-629.

570 Zhao Y, Hu M, Slanina S, Zhang Y. Chemical compositions of fine particulate organic matter  
571 emitted from Chinese cooking. *Environ Sci Technol*. 2007; **41**: 99-105.

572 Zeng J, Yu Z, Mekic M, et al. Evolution of indoor cooking emissions captured by using secondary  
573 electrospray ionization high-resolution mass spectrometry. *Environ. Sci. Technol. Lett*. 2020; **7**:  
574 76-81.