Indoor and outdoor air quality: a university cafeteria as a case study

Célia A. Alves[1*], Estela D. Vicente[1], Margarita Evtyugina[1], Ana M. Vicente[1], Teresa Nunes[1],

Franco Lucarelli[2], Giulia Calzolai[2], Silvia Nava[2], Ana I. Calvo[3], Carlos del Blanco Alegre[3],

Fernanda Oduber[3], Amaya Castro[3] and Roberto Fraile[3]

[1]Centre for Environmental and Marine Studies, Department of Environment, University of Aveiro,

3810-193 Aveiro, Portugal

[2]Department of Physics and Astronomy, Università di Firenze and INFN-Firenze, 50019 Sesto

Fiorentino, Italy

[3]Department of Physics, IMARENAB University of León, 24071 León, Spain

*Correspondence to: celia.alves@ua.pt

Abstract: A short but exhaustive air sampling campaign was conducted in a university cafeteria, an occupational environmental not yet studied. Carbonyls and volatile organic compounds were collected by passive diffusion samplers. Temperature, relative humidity, CO_2, CO and particulate matter were continuously monitored indoors and outdoors. Simultaneous PM_{10} sampling with high and low volume instruments, equipped with quartz and Teflon filters, respectively, was performed during working hours and at night. The quartz filters were analysed for their carbonaceous content by a thermo-optical technique and organic constituents by GC-MS. Water-soluble ions and elements were analysed in the Teflon filters by ion chromatography and PIXE, respectively. Low air change rates (0.31-1.5 h^{-1}) and infiltration factors of 0.14, for both $PM_{2.5}$ and PM_{10}, indicate poor ventilation conditions. Concentrations of both gaseous pollutants and particulate matter were much higher in the cafeteria than outdoors, showing strong variations throughout the day depending on occupancy and activities. The average concentration of indoor-generated PM_{10} was estimated to be 32 µg m^{-3}. Organic compounds in PM_{10} included alkanes, PAHs, saccharides, phenolics, alcohols, acids, alkyl esters, triterpenoids, sterols, among others. The complex particle composition reveals the multiplicity of sources, formation reactions and removal processes, not yet fully known, and suggests the contribution from dust resuspension, abrasion and off-gassing of building materials, cooking emissions, tobacco smoke, and several consumer products. Many compounds are in the list of ingredients of personal care products, pesticides, plasticisers, flame retardants and psychoactive drugs. The inhalation cancer risks of metals and PAHs were found to be negligible.

Keywords: Cafeteria, Indoor air quality, PM_{10}, Metals, Organic compounds, Sources

1. Introduction

In recent decades, Indoor Air Quality (IAQ) has been the focus of attention by the scientific community, governments and international organisations (Abdul-Wahab et al., 2015). It is a known fact that we spend more than 90% of our time indoors and that the levels observed are often higher than those recorded outdoors. Thus, the health risks from exposure to indoor air pollution may be larger than those associated with outdoor pollution. Poor IAQ can be especially detrimental to vulnerable groups such as children, young adults, the elderly, or those suffering from chronic respiratory and/or cardiovascular diseases (Cincinelli and Martellini, 2017). Furthermore, IAQ in the workplace is important because employees are occupationally exposed on a regular basis to a variety of pollutants with potential impact on comfort, work-related health problems, sickness absence, and productivity (Al horr et al., 2016; Carrer and Wolkoff 2018; Śmiełowska et al., 2017). IAQ in elderly care centres (e.g. Almeida et al., 2016; Bentayeb et al., 2015; Buczyńska et al., 2014; Mendes et al., 2015; Mui et al., 2008; Simoni et al., 2003), office buildings (e.g. Mandin et al., 2017; Śmiełowska et al., 2017; and references therein) and schools (e.g. Alves et al., 2013; Barmparesos et al., 2018; Bennet et al., 2019; Blondeau et al., 2005; Chatzidiakou et al., 2015; Fuoco et al., 2015; Kalimeri et al., 2016; Madureira et al., 2016; Yang et al., 2015) has been extensively investigated. Some studies were also conducted to evaluate the IAQ in restaurants with different cooking styles (Dai et al., 2018; Lee et al., 2001; Lee et al., 2011; Elsharkawy and Javed, 2018; Mugica et al., 2001; Sofuoglu et al., 2015; Taner et al., 2013; Wilson et al., 2011).

Restaurants not only have many of the same threats generally observed in office buildings, but they also encompass several other pollutants given off by cookware, open flames, cooking appliances, refrigeration units, strong chemical degreasers and aggressive cleaning solvents. Most of the published studies report the monitoring of comfort parameters, ventilation and/or some regulated pollutants. Gaseous compounds (CO, CO_2, NOx and some volatile organic compounds), particles less than 2.5 and 10 µm in diameter ($PM_{2.5}$ and PM_{10}, respectively), as well as some organic compounds in the particulate phase, normally from the group of polycyclic aromatic compounds (PAHs) with proven carcinogenic properties, are the most commonly monitored pollutants (Abdullahi et al., 2013; Marć et al., 2018; and references therein).

University cafeterias, a specific type of eating establishment, are popular meeting places for students and staff, providing space for leisure, relaxation and a room to have a quick lunch or snack. For this category of restaurant, as far as we know, no exhaustive evaluation has been made. Using optical counters, Zhao et al. (2010) measured PM concentrations in three different cafeterias located at Tsinghua University, Beijing, China. However, no information is given on the chemical composition of the particles and on the concentration of other pollutants. Hsu and Huang (2013) conducted a study in the student cafeteria at Chia-Nan University in southern Taiwan. Measurements of relative humidity, temperature, airflow velocity, CO_2 and total suspended particles were made, although the focus of the

75　work was not centered on IAQ but rather on the efficiency of two disinfection treatments in removal of

76　microorganisms.

77　　　This paper describes and discusses a short case study on a comprehensive air quality monitoring

78　programme carried out in a university cafeteria. Besides the measurement of traditional parameters,

79　such as regulated gaseous pollutants, the work had as main objective the detailed characterisation of

80　PM_{10}, including organic and inorganic components, in order to understand which sources and processes

81　contribute to the measured levels and to estimate the carcinogenic and non-carcinogenic risks associated

82　with inhalation of elements and PAHs by employees and customers. This preliminary study can serve

83　as a basis for improving future monitoring campaigns, especially as regards the definition of source-

84　marking chemical species, and for defining mitigation strategies to reduce exposure to pollutants in

85　similar facilities.

86

87　**2. Methodology**

88

89　***2.1. Sampling***

90

91　　　A one-week extensive sampling campaign was carried out from 27th November to 1st December 2017

92　at the León (Spain) university cafeteria. The commercial establishment is continuously open for 12

93　hours (7:00-19:00, UTC) from Monday to Friday. The cafeteria is one large open room (410 m^2) with

94　fourteen tables, each capable of seating four to eight people, three standing round tables and stools along

95　the 21 m long bar (Fig. 1). A separate kitchen provides the space to prepare the daily special menus and

96　several dishes of fast food, such as burgers, sandwiches and French fries. The kitchen is equipped with

97　a diesel fired range cooker, electric grills and toasters. The extractor hood of the kitchen has 4.8×2.4

98　m. There is no information regarding the brand, model or power. The cafeteria has two entrance doors,

99　one opening to an adjacent alley and the other opening to an outdoor patio. A heat pump works as heater

100　during the colder months and as air conditioner during the warmer months. Attendance indoors was

101　highly variable, depending on the time of the day. Officially, the maximum occupancy allowed is 140

102　persons. People occupancy was logged manually throughout the campaign. In general, in the early

103　morning, there was a simultaneous presence of 10 to 20 people, reaching a peak at lunch time (60 to

104　100 people). During the afternoon, the frequency of attendance varied from 10 to 40 clients, with a

105　maximum around 16:00.

106　　　Temperature, relative humidity, CO_2 and CO were continuously monitored with air quality probes

107　inside (TSI, model 7545) and outside (Gray Wolf®, WolfSense IQ-610) the cafeteria. Real-time laser

108　photometric instruments (TSI, DustTrak DRX 8533) were used to record particulate matter (PM_1, $PM_{2.5}$

109　and PM_{10}) concentration profiles over time in the indoor and outdoor environments, simultaneously.

110　All these continuous monitoring instruments were operated with 1-min resolution and were factory-

111　calibrated before use. Simultaneous sampling with two PM_{10} high volume air MCV (model CAV-A/mb)

112   instruments for gravimetric quantification was carried out on pre-weighed 150 mm quartz fibre filters
113   (Pallflex®). The equipment was operated at a flow of 30 m^3 h^{-1}. Samples were collected during the
114   opening hours, simultaneously indoors and outdoors. Night time samples were also taken in order to
115   compare the PM levels during occupancy and non-occupancy periods. Furthermore, PM_{10} samples were
116   collected on Teflon membrane filters (47 mm diameter, Pall Corporation) with a TCR TECORA (model
117   2.004.01) operated at a flow of 2.3 m^3 h^{-1}. The gravimetric quantification was performed with an
118   analytical balance (Mettler Toledo, XPE105 DeltaRange®). Due to the possibility of vandalism and
119   unfavourable weather conditions, with freezing temperatures, the outdoor equipment was shut down
120   and sheltered at night. Thus, outdoor measurements correspond to the occupancy period in the cafeteria.
121   All the data were registered in UTC.

122   Passive sampling of formaldehyde and acetaldehyde was done using Radiello™ diffusion tubes
123   filled with 2,4-dinitrophenylhydrazine (2,4-DNPH) coated FLORISIL® over one week of continuous
124   exposure, simultaneously indoors and outdoors. Aldehydes react with 2,4-DNPH to give the
125   corresponding 2,4-dinitrophenylhydrazones. Volatile organic compounds (VOCs) were also collected
126   with passive samplers from Radiello™. To verify if there were large variations in the carbonyl and
127   VOC levels, passive sampling was repeated two more times in the weeks following the campaign
128   described above.

129

130   ***2.2. Chemical analyses***

131

132   After gravimetric determinations, thermal-optical analysis of PM_{10} quartz filters was performed to
133   obtain the carbonaceous content (organic and elemental carbon, OC and EC). This method is based on
134   the CO_2 quantification, by a non-dispersive infrared (NDIR) analyser, which is released from the
135   volatilisation and oxidation of different carbon fractions under controlled heating. A laser beam and a
136   photodetector that measure the filter light transmittance allow separating the EC formed by OC
137   pyrolysis from the one that was initially in the filter (Alves et al., 2011).

138   Two circular punches of 47 mm from the quartz filters were sequentially extracted with
139   dichloromethane and methanol. The resulting extract was separated into five different organic fractions
140   by flash chromatography with silica gel and various solvents of increasing polarity. To meet the
141   detection limits required by the speciated organic compounds, the night sampling filters were combined
142   and extracted together. Thus, an average composition was obtained for non-occupancy periods at night.
143   After each elution, the different fractions were concentrated in a TurboVap system from Biotage and
144   evaporated to dryness by a nitrogen stream. Before chromatographic analysis, the fractionated extracts
145   containing polar compounds were derivatised to trimethylsilyl ethers. All the extracts were analysed by
146   gas chromatography-mass spectrometry (GC-MS). A detailed description of the whole procedure for
147   the speciation of organic compounds was previously reported by Alves et al. (2011).

148   Half of each Teflon filter was extracted with ultrapure Milli-Q water for ion chromatography

149   analysis. A detailed description of the method can be found elsewhere (Vicente et al., 2018). All the

150   elements with $Z > 10$ were determined in the other half of the filters by proton-induced X-ray emission

151   (PIXE) (Lucarelli et al., 2014).

152   The carbonyl-DNPH derivatives were extracted from the Radiello tubes with 2 mL of acetonitrile

153   for 30 min, filtered through a micropore filter membranes with porosity of 0.45 μm, and analysed by

154   high performance liquid chromatography. HPLC measurements were carried out with a Jasco PU-980

155   pump, a Rheodyne 7125 six-port valve with a 20 μL loop and a Jasco MD-1510 diode array detector.

156   An isocratic mixture of acetonitrile and water (60:40), at a flow rate of 1.5 mL min^{-1}, was used to

157   separate the 2,4-DNPH on a Supelcosil™ LC-18 column (5 μm particle size; L × I.D.: 250 × 4.6 mm).

158   The carbonyl concentrations were quantified with external calibrations curves constructed from

159   standard solutions of carbonyl-DNPH derivatives in acetonitrile. The UV wavelength for detection was

160   360 nm. VOCs in activated charcoal cartridges were chemically desorbed with 2 mL of carbon

161   disulphide (CS_2) and 100 μL of internal standard (2-fluorotoluene) for 30 min and analysed by gas

162   chromatography coupled to flame ionisation detector (GC-FID), following the conditions described in

163   Pegas et al. (2010).

164

165   ***2.3. Air change rates***

166

167   In the present study, the air change rates per hour (ACR, h^{-1}) were estimated by the CO_2

168   concentration decay method (Alves et al., 2013). For a well-mixed space, the change in CO_2

169   concentration with time is given by:

170

171
$$C_\tau = C_{ext} + \frac{q_{CO_2} \times 10^6}{Q} - \left(C_{ext} - C_0 + \frac{q_{CO_2} \times 10^6}{Q} \right) e^{\left(-Q_t/_v \right)} \tag{1}$$

172

173   where C_t is the indoor concentration of CO_2 at time t (ppm), C_{ext} is the outdoor concentration of CO_2

174   (ppm), C_0 is the concentration of CO_2 in the indoor air at time 0 (ppm), Q is the volume flow rate of air

175   entering the space (m^3 s^{-1}), q_{CO_2} is the volumetric indoor emission rate of CO_2 (m^3 s^{-1}), V is the volume

176   of the room (m^3) and t is the interval since $t = 0$ (s). When the room is unoccupied there is no CO_2

177   emission from the occupants, and $q_{CO_2} = 0$. Thus, Eq. (1) can be rearranged to give the following

178   expression, which allows the ventilation rate (Q) to be estimated from measured concentration values

179   time t apart:

180

181
$$Q = -\frac{V}{t} \times \ln \left(\frac{C_t - C_{ext}}{C_0 - C_{ext}} \right) \tag{2}$$

5

183 The air change rate is defined as:

184

185 $$ACR = Q/V \qquad (3)$$

186

187 **3. Results and discussion**

188

189 *3.1. Comfort parameters and gaseous pollutants*

190

191   The indoor thermal environment is not only related to energy use in buildings, but is also linked to

192 human satisfaction and wellbeing, health outcomes, and productivity. While temperature is the most

193 influential factor in generating a phenomenological sense of thermal comfort, relative humidity (RH)

194 contributes to a combined effect on the occupants' perception. Relative humidity levels below 25% are

195 associated with increased discomfort and drying of the mucous membranes and skin, which can lead to

196 chapping and irritation. Low relative humidity also increases static electricity, which can hinder the

197 operation of computers and other equipment. High humidity levels can result in condensation within

198 the building structure, providing the conditions for dust mite, mould and bacteria populations to grow.

199 These, in turn, can cause respiratory problems and/or allergic reactions. Increasing RH is also associated

200 with enhanced off-gassing of VOCs from building materials (Huang et al., 2016). The American Society

201 of Heating, Refrigerating, and Air-Conditioning Engineers (ASHRAE Standard 55–2004) proposed

202 indoor temperatures from 20 to 23 °C in the autumn/winter season, while the recommended RH values

203 are in the 30–60% range for optimum comfort. The inter-daily variations (mean±SD) of temperature

204 during the occupancy periods were weak, ranging from 19.2±1.7 to 20.0±1.0 ºC, while RH fluctuated

205 between 31.2±2.8 and 36.1±5.3% (Fig. 2). Outside, the daytime temperatures and RH oscillated

206 between -5.8 and 6.5 ºC and from 46.2 to 90.9%, respectively.

207   CO_2 at the concentrations usually observed in buildings is not a direct health risk, but its levels can

208 be used as an indicator of occupant odours (bioeffluents). CO_2, primarily generated through human

209 metabolism, provides a good indication of air change rates. Its build up in indoor environments is

210 attributed to inefficient ventilation. ASHRAE (Standard 62.1-2016) reports that while not necessarily

211 hazardous, the amount of CO_2 in an indoor environment should remain under 1,000 ppm. The mean

212 daytime carbon dioxide concentration indoors was 891±226 ppm, decaying to values around 400 ppm

213 at night. A twofold difference in indoor concentration was observed between occupancy and non-

214 occupancy periods. Concentrations recorded throughout the weekend were similar to those observed in

215 nocturnal periods. The inter-day fluctuations depended on the occupancy rates, reaching a maximum

216 concentration of 3430 ppm in the early evening of Thursday (Fig. 2), the day the cafeteria closed later

217 due to a party attended by about 150 people.

ACR ranged between 0.31 and 1.5 h^{-1}, indicating poor ventilation conditions. For all spaces in general, a minimum ACR of 4 h^{-1} is recommended, while specific values of 8-12 h^{-1} and 12-15 h^{-1} are suggested for restaurants and cafeterias, respectively (The Engineering ToolBox, 2005). Natural ventilation can be challenging during the wintertime in cooler climates. While open windows and doors can bring fresh air inside, this action obviously allows heat to escape. The installation of heating, ventilation, and air conditioning (HVAC) technologies may be the only solution to assure ventilation requirements for acceptable IAQ and thermal comfort.

Carbon monoxide levels were very low indoors. The maximum concentration ever recorded was 1.1 ppm. All the doors and windows were kept closed, due to the cold weather outside, preventing CO infiltration into the cafeteria from the nearby low traffic street and doorstep smokers. Low concentrations, always < 3 ppm, were also observed outdoors. ASHRAE (Standard 62.1-2016) lists a maximum allowable short-term limit of 9 ppm for CO in indoor air, while the WHO (2010) recommends CO concentrations based on exposure time ranging from 90 ppm (15-minute exposure) to 10 ppm (8-hour exposure). The American Conference of Governmental Industrial Hygienists (ACGIH) has assigned CO a threshold limit value (TLV) of 25 ppm as a time weighted average (TWA) for a normal 8-hour workday and a 40-hour workweek. The ACGIH limit is based on the risk of elevated carboxyhemoglobin levels. The National Institute for Occupational Safety and Health (NIOSH) has established a recommended exposure limit of 35 ppm as an 8-hour TWA and 200 ppm as a ceiling, based on the risk of cardiovascular effects.

Formaldehyde and acetaldehyde, two ubiquitous compounds emitted by combustion processes, meat cooking operations and consumer products, presented indoor mean concentrations of 8.8 and 17.8 µg m^{-3}, respectively, while the corresponding outdoor values were 1.3 µg m^{-3} and of the same order of the blanks. The inter-week variations were always <10%. Formaldehyde was below the air quality guideline of 100 µg m^{-3} (as a 30-min average value) recommended by the World Health Organisation (WHO) in 2010. However, short-time exceedance of this guideline level might have occurred in the indoor environment. Acetaldehyde was included by the WHO in Group 2, which encompasses pollutants of potential interest, but further investigation would be needed before it is clear whether there is enough evidence to warrant their inclusion in the guidelines. However, based on studies of short- and long-term exposure, countries such as Canada have set a maximum daily limit of 280 µg m^{-3}.

The indoor levels of VOCs (Table 1) were higher than those observed outdoors. Among these, ambient concentrations of benzene are regulated by law due to its toxicity (European Commission, 2009). In contrast, indoor levels of VOCs are not subject to any legislation. Of all the VOCs found in buildings, BTEX (benzene, toluene, ethyl benzene, o-, m-, and p-xylene) are among the most common. Indoors, these compounds can originate from multiple sources, including building materials (paints, varnishes), consumer products (cleaning agents, fingernail polish, lacquers, adhesives), and human activities (smoking and combustion). In outdoor air, traffic is pointed out as a major source. The outdoor concentrations of benzene were far below the annual ambient air quality standard of 5 µg m^{-3} set by the

European Commission. Previous studies have reported the indoor BTEX levels at homes (Dai et al., 2017; Esplugues et al., 2010; Fan et al., 2018; Liu et al., 2013), libraries (Cincinelli et al., 2016), schools (Demirel et al., 2014; Pegas et al., 2011; Raysoni et al., 2017), printing shops and photocopy centres (El-Hashemy and Ali, 2018), and a confined parking area (Castro et al., 2015). Highly variable concentrations, up to hundreds of $\mu g\ m^{-3}$, have been obtained, depending on the source strengths. Benzene levels of 18.4, 10.0, 9.3 and 3.7 $\mu g\ m^{-3}$ were measured in the dining areas of a Korean barbecue style restaurant, a Chinese hot pot restaurant, a Chinese dim sum restaurant and a Western canteen, respectively (Lee et al., 2001). The corresponding values reported for toluene were 156.1, 93.8, 81.6 and 17.6 $\mu g\ m^{-3}$. In the present study, *n*-butanol and toluene were the VOCs with the highest indoor air concentrations. Butanol (or butyl alcohol) is present in many foods and beverages and is also used in a wide range of consumer products, such as cleaning products or cosmetics. Toluene is commonly added to gasoline and can enter the indoor spaces from outside. Toluene is also often used as solvent and can be released from a variety of products found in homes, including paints and other finishes, adhesives, and some personal care products. It is also found in tobacco smoke. Although detected at lower concentrations compared to these two VOCs, *n*-hexane showed very high I/O ratios (up to 13), suggesting the presence of active sources in the cafeteria. This aliphatic compound is primarily used as a solvent and as a component of certain glues and adhesives, cleaning agents, biocidal products, ink and toners, textile dyes, perfumes, fragrances and cosmetics (Federal Institute for Occupational Safety and Health, 2017). It is also present in volatile fractions of various plant species, such as apples and orange juice (Toxno, 2018). Moreover, Ahearn et al. (1996) reported that a variety of fungi found in buildings are capable of releasing gases that include *n*-hexane.

3.2. Particulate matter

3.2.1. Mass concentrations

From the gravimetric measurements, average PM_{10} concentrations of 35.6 ± 5.6, 12.4 ± 3.4 and 23.3 ± 10.0 and $\mu g\ m^{-3}$ were recorded indoors during the occupancy and non-occupancy hours, and in the outdoor air, respectively, not exceeding the 24-h guideline of 50 $\mu g\ m^{-3}$ set by the WHO. However, the real-time data collected by the photometric monitors show strong variations throughout the day. The highest values were reached at morning snack and lunch time, when the influx of students and staff was typically higher. Throughout the week, the maximum $PM_{2.5}$ and PM_{10} values were in the ranges 44-92 and 62-136 $\mu g\ m^{-3}$, but peaks as high as 2560 and 2710 $\mu g\ m^{-3}$, respectively, were registered in late afternoon and early evening of the day of a student party, during which even some firecrackers were launched Fig. 3). Throughout the week, a peak was observed between 19:00 and 20:00, time at which cleaning activities of the premises took place. Fine particles represented an overwhelming proportion of PM_{10} indoors: $PM_{2.5}/PM_{10}$=0.82-0.91 (avg=0.88) and PM_1/PM_{10}=0.81-0.91 (avg=0.87). The

292   PM$_{2.5}$/PM$_{10}$ and PM$_1$/PM$_{10}$ ratios were even higher outdoors, denoting the dominance of fine aerosols:

293   0.92-0.97 (avg=0.95) and 0.91-0.97 (avg=0.94). Regardless of particle size, indoor concentrations were

294   1.5-2 times greater than those found in outside air. During the night, in the cafeteria, the concentrations

295   drop approximately 3 times.

296   The infiltration factor, which represents the equilibrium fraction of the outdoor PM that penetrates

297   indoors and remains suspended (Othman et al., 2019), was calculated as follows:

298

299   $C_{in} = F_{inf} \times C_{out} + C_{ig}$ (4)

300

301   where C_{in} and C_{out} are the indoor and outdoor PM$_{2.5}$ or PM$_{10}$ concentrations, F_{inf} is the infiltration factor,

302   and C_{ig} represent the concentrations of indoor-generated particles. The slopes of the correlations

303   between indoor and outdoor concentrations for the occupancy periods, i.e. the infiltration factors, were

304   0.14 for both PM$_{2.5}$ and PM$_{10}$. As inferred from the CO$_2$ measurements, the low particle infiltration

305   factors denote air tightness. On average, concentrations of indoor-generated particles were 27 µg PM$_{2.5}$

306   m^{-3} and 32 µg PM$_{10}$ m^{-3}. To evaluate the influence of outdoor particulate pollutants on indoor

307   environment, the concept of contribution rate (ρ) was applied, as suggested by Lv et al. (2017):

308

309   $\rho = (F_{inf} \times C_{out})/C_{in} \times 100\ \%$ (5)

310

311   The contribution rates of outdoor sources were estimated to be, on average, 17.7 and 16.1 % for PM$_{2.5}$

312   and PM$_{10}$, respectively, whereas a major input from indoor sources was observed (82.3 and 83.9 %).

313

314   *3.2.2. Chemical composition*

315

316   To obtain a PM$_{10}$ mass balance (Fig. 4), the measured element concentrations were converted into

317   the respective mass concentrations of the most common oxides (SiO$_2$, Al$_2$O$_3$, CaO, MgO, MnO, Fe$_2$O$_3$,

318   TiO$_2$, K$_2$O, etc.). Because the GC-MS analyses indicated the presence of highly oxygenated compounds,

319   to derive the organic matter content in PM$_{10}$, a total organic mass to organic carbon ratio (OM/OC) of

320   1.8 was adopted (Polidori et al., 2008). Total carbon (TC = OC + EC) accounted for 36.0 ± 5.8, 42.8 ±

321   7.9 and 27.6 ± 12.6 %wt. of the PM$_{10}$ mass indoors during the occupancy and non-occupancy hours,

322   and in the outdoor air, respectively (Fig. 5). Besides sampling and analysis artefacts affecting the

323   attainment of chemical mass balance, the unaccounted mass can partly be explained by the presence of

324   unanalysed constituents. Furthermore, part of the unaccounted PM mass is usually assigned to particle-

325   bound water (Tsyro, 2005). On average, I/O ratios of 1.5 and 0.7 were obtained for OC and EC,

326   respectively, suggesting the presence of organic matter emitting sources in the cafeteria, while

327   elemental carbon is predominantly generated from combustion processes in the outdoor environment.

328   Mean OC/EC ratios of 7.5, 3.5 and 3.6 were observed for the occupancy and night-time periods in the

cafeteria and for the outdoor air, respectively. A much higher ratio (21.0) was registered for the weekend daytime period. Lower EC infiltration rates over the weekend due to the fact that the building is closed, absence of indoor EC sources (e.g. toasting bread) and off-gassing of VOCs from surfaces leading to secondary organic aerosol formation, may justify the higher proportion of OC compared to EC. It should be noted that OC and EC concentrations decreased from 10.7 and 1.1 $\mu g\, m^{-3}$ in the periods of occupation to 6.1 and 0.29 $\mu g\, m^{-3}$ in the weekend, respectively.

On average, water-soluble ions accounted for PM_{10} mass fractions of 6.0, 15.8, 13.4, and 1.3 % during the occupancy of the cafeteria, at night-time indoors, in the daytime period on the weekend inside the building, and at regular working hours outdoors, respectively. Expect for Ca^{2+} and K^+, the I/O ratios for all water-soluble ions were less than 1. The higher I/O ratios for Ca^{2+} and K^+ may be related to resuspension of fine dust from human movement. Concentration values in daytime were higher than at night-time (Fig. 6). Together with chloride, the secondary ions sulphate, nitrate, and ammonium were the dominant species. In addition to sea salt and biomass burning, chlorinated water and cleaning products containing bleachers can release significant amounts of chloride. Ammonium may also originate from cleaning products. For unknown reasons, in the outdoor sample of Tuesday, concentrations increased approximately 4-fold compared to other days, contributing to a high standard deviation. A good correlation between concentrations of total anions and total cations in PM_{10} was observed, but an anion deficit was registered. Non-measured organic ions, such as formate and acetate, could contribute to the ionic balance. However, the anion deficit is most likely due to the presence of non-determined carbonates (CO_3^{2-}/HCO_3^-) from resuspended local soil. Outdoors, the molar ratio of NH_4^+ to SO_4^{2-} was found to be greater than 2. This higher molar ratio suggests that in addition to $(NH_4)_2SO_4$, NH_4NO_3 was formed because of excess quantity of NH_4^+. Ammonium nitrate is produced in the presence of high NH_4^+ and HNO_3 concentrations, low temperatures and high relative humidity. A mean NH_4^+/SO_4^{2-} molar ratio of 1.6 was obtained indoors, indicating aerosols in the form of $(NH_4)_2SO_4$ and NH_4HSO_4. Due to the higher temperatures indoors, NH_4NO_3 may decompose to form NH_3 and HNO_3, which are then partially lost by deposition and sorption processes to indoor surfaces, especially those with higher roughness, while another part reacts with NaCl and Ca-containing particles. $NaNO_3$ and $Ca(NO_3)_2$ have low volatility compared to NH_4NO_3.

Both indoors and outdoors, the elements with highest concentrations were Na, Mg, Al, Si, S, Cl, K, Ca and Fe (Table 2). Elements detected at I/O much higher than 1, suggesting indoor sources, were Mg, P, K, Ca, Cr, Ni and Zn. Some of these elements (e.g. Ca, Mg, K) are present in mineral matter and can be associated with soil dust that is brought in from outdoors on shoes. It has been shown that resuspended dust contributes to the airborne particulate matter in the "personal cloud" (i.e. personal breathing zone). Resuspension rates of settled dust depend on factors such as amounts accumulated, floor characteristics, walking style, shoe types, humidity and cleaning activities (Rasmussen et al., 2018). Some elements may result from abrasion of metallic components from furniture and other building materials. Cr, for example, is an important alloying element in stainless steel and is also present

as chromated copper arsenate (CCA), a preservative for furniture and wood building materials against insect, bacterial, and fungal decline (Rivas et al., 2015; and references therein). P is likely associated with chlorinated phosphorus-based flame retardants and plasticisers, which are ubiquitous in the indoor environment (Araki et al., 2014). Many metals (e.g. Ni and Zn) are also in the list of ingredients of numerous personal care products (Borowska and Brzóska, 2015; Omenka and Adeyi, 2016).

Enrichment factors of each element with respect to crustal material (Wedepohl, 1995) were calculated using Fe as reference element:

$$EF = (X/Fe)_{air}/(X/Fe)_{crust} \tag{3}$$

where EF is the enrichment factor of element X, $(X/Fe)_{air}$ is the concentration ratio of X to Fe in the PM_{10} samples, and $(X/Fe)_{crust}$ is the average concentration ratio of X to Fe in the continental crust. Whether for samples of the cafeteria or for outdoor PM_{10} filters, EFs lower than 5 were obtained for Na, Mg, Al, Si, K, Ca, Ti, V, Mn, Rb, Sr, Y and Zr, indicating a dominant soil dust origin. Although the sampling campaign was carried out in winter, when the residential biomass combustion is a common practice, the EFs for potassium do not reveal the dominance of wood or coal burning. However, it should be borne in mind that residential combustion equipment is relatively inefficient, reaching temperatures generally below 700-800 °C. It has been extensively reported that, before condensing to form particles, potassium is released into the gas phase at temperatures above 727 °C and that only the full combustion of biomass fuels at elevated temperatures (>1327 °C) is expected to result in most of the potassium entering the gas phase (Mason et al., 2016). Compared to the dust-derived elements, the EFs for S, Cl, Cu, Zn, As, Br, Mo and Pb were larger than 10, for both indoor and outdoor samples, illustrating the influence of anthropogenic sources. Se was a particularly enriched element with EF values > 1100. In addition to As and S, coal combustion is also a strong emitter of Se. More than 20% of the total energy consumption in the region of León come from coal combustion.

The emissions from coal burning increase during coldest months due to the use of domestic heating and cooking devices (Blanco-Alegre., 2019). In outdoor environments, many of these elements, in particular Cu, Zn, Pb and Mo, are mostly associated with traffic emissions (Pant and Harrison, 2013). In dwellings and in working and leisure places, in addition to infiltration from outdoors, these constituents can originate from the multiplicity of materials and consumer products used indoors. Attention is drawn to the fact that the concentration of some elements has increased sharply on the day of the student party. Potassium reached a level of 2140 ng m^{-3}, while in other occupancy periods concentrations ranged from 305 to 367 ng m^{-3}. Cl, Zn and Cu increased from usual levels in the ranges 214-396, 21-24 and 2.7-5.7 ng m^{-3} to values of 1870, 61 and 15 ng m^{-3}, respectively, on the party day. These elements are abundant constituents of tobacco smoke (Misha et al., 1986), reason why there are suspicions that, although prohibited, this practice may have occurred during the event.

The particulate matter organic extracts encompassed several aliphatics, polycyclic aromatic hydrocarbons (PAHs), alcohols, acids, sterols, glycerol derivatives, phenolic compounds, saccharides, among others. Aliphatic compounds included the homologous series of n-alkanes from C_{11} to C_{32} and some alkenes. These are ubiquitous compounds in the environment with both natural and anthropogenic origin. Alkane homologues were detected at I/O ratios ranging from 1.0 (dodecane) to 54 (dotriacontane). The sum of their concentrations was as follows: 49.4 ng m^{-3} (indoor, occupancy), 15.9 ng m^{-3} (indoor, night-time), 16.5 ng m^{-3} (outdoor, daytime), and 6.0 ng m^{-3} (indoor, weekend, daytime). The maximum indoor concentration (13.4 ng m^{-3}) was observed for dotriacontane. Alkenes (C_{14}, C_{16}, C_{18}, C_{20} and C_{23}) were present at I/O ratios from 3 to 5 and very low concentrations, peaking in the cafeteria for hexadecene (0.47 ng m^{-3}). Excepting hexadecene, alkenes were not detected in samples from non-occupancy periods.

Figure 7 depicts the mean concentrations of individual PAHs in the cafeteria and outdoors. Excluding naphthalene and acenaphthene, two volatile PAHs consisting of two benzene rings, the I/O ratios for all other polyaromatics were always < 1. Contrary to many other compounds, the average outdoor concentrations (7.8 ng m^{-3}) was 3.3 times higher than those measured indoors (2.4 ng m^{-3}). Levels in the cafeteria dropped to 0.49 ng m^{-3} on the weekend, whereas the night-time value (4.3 ng m^{-3}) was between the one measured during working hours in the cafeteria and that registered outdoors. Although nocturnal measurements in the ambient air were not carried out in the present study, outdoor concentrations usually higher at night may have contributed to the build-up of PAHs in the cafeteria through infiltration processes. Due to residential heating emissions, decreased atmospheric mixing height and nocturnal atmospheric stability, an appreciable increase in concentrations in the evening is likely to occur. On the other hand, at daytime, after emission by combustion sources (e.g. traffic), PAHs undergo chemical transformations. Better dispersion conditions and atmospheric reactivity during daytime, in general, lead to lower diurnal concentrations compared to nocturnal values. PAH oxidative transformations in ambient air are supposed to be started by gas-phase OH and NO_3 radicals, followed by reactions with NO_2 and O_3 leading to ring-retaining quinones, nitro-PAHs and ring-opened phthalic acids and their degradation products (Alam et al., 2015).

The homologous series of fatty alcohols from C_{10} to C_{30} was observed in the PM_{10} samples (Table 3). 1-Hexacosanol and 1-octadecanol were the most abundant homologues. Their indoor concentrations were more than 600 times higher than those measured outdoors. These n-alkanols are constituents of many household and personal care products. Furthermore, octadecanol was identified as one of the dominant compounds in fine organic aerosols from charbroilers and meat cooking operations (Rogge et al., 1991). The homologous series of alkanoic acids (C_6-C_{26}) and some alkenoic acids were also present. In general, the concentrations recorded during the occupancy period in the cafeteria far exceeded those observed outside. Palmitic, stearic and oleic were the dominant acids, in agreement with previous studies describing cooking emissions (Abdullahi et al., 2013; He et al., 2004; Rogge et al., 1991; Zhao et al., 2007).

439   Some dicarboxylic acids from C_4 to C_{16}, at I/O ratios generally higher than 1, were present in PM_{10},

440   but at lower concentrations compared to those of fatty acids. He et al. (2004), who also detected diacids

441   in emissions from cooking processes at lower levels in comparison with alkanoic and alkenoic acids,

442   suggested that these compounds are formed by oxidation of dialdehydes during the autoxidation process

443   of unsaturated lipids. Among diacids, azelaic (C_9) was the most abundant in the cafeteria, reaching

444   concentrations up to 67 ng m^{-3}. The lab-engineered form of this acid is typically used in skincare

445   products. It is also an abundant compound in emissions from electronic cigarettes, heat-not-burn

446   tobacco products, and conventional cigarettes (Ruprecht et al., 2017). Biomass combustion is also a

447   source of diacids, in addition to emitting keto- (e.g. levulinic) and hydroxy-acids (e.g. glycolic, glyceric

448   and lactic) (Alves et al., 2011). All these acids were not detected in the outdoor samples or were present

449   in lower concentrations when compared to those observed inside the cafeteria. Liu et al. (2017)

450   measured a wide variety of carboxylic acids in a university classroom, indoors and outdoors, including

451   monoacids, diacids, hydroxy acids, carbonyl acids, and aromatic acids. Besides indoor sources such as

452   off-gassing of building materials, evidence for acid formation from indoor chemical reactions with

453   ozone was noticed. It was also suggested that human occupants may contribute to the abundance of

454   carboxylic acids either through direct emissions or surface reactions. Although generally present at very

455   low concentrations indoors, ozone may react with unsaturated organic compounds present in skin and

456   other materials. As observed in the university classroom, the elevated indoor concentrations of lactic

457   acid in the cafeteria during the occupancy period was likely human perspiration. A recent study

458   conducted in an art museum revealed the unexpected build-up of lactic acid from sweat (Pagonis et al.,

459   2019). It was found that this acid leaves the skin, travels through the air, and sticks to the walls and

460   other surfaces at high rates. It should be also noted that lactic and other short chain acids (levulinic,

461   glycolic, etc.) are frequently used in exfoliants, anti-wrinkle products and other cosmetics. Thus, both

462   the emission and partitioning mechanisms between the gas and the particulate phases of these and many

463   other compounds are complex and still far from being fully understood.

464   Some resin acids were detected as PM_{10}-bound organic components: dehydroabietic, isopimaric,

465   abietic and podocarpic. Among these, dehydroabietic acid dominated. Its concentrations outside the

466   cafeteria exceeded those observed indoors, pointing to the residential burning of coniferous wood

467   (Vicente and Alves, 2018). Although it is an unlikely source in the cafeteria, it should be noted that

468   very high concentrations of free resin acids were detected in depilatory wax strips (Nilsson et al., 2008).

469   Levoglucosan and its stereoisomers, mannosan and galactosan, were one of the dominant groups of

470   organic compounds. Levoglucosan has been recurrently used as a biomass burning tracer, since it is a

471   thermal degradation product of cellulose (Vicente and Alves, 2018). However, it was also detected in

472   particulate matter samples emitted from Sichuan, Dongbei, Hunan and Cantonese cooking styles,

473   suggesting that this anhydrosugar can also be generated during the heating process of vegetables (He et

474   al., 2004; Zhao et al., 2007). Mean I/O ratios lower than 1 reveal the strong impact of residential biomass

475   burning on the ambient air of the surroundings due to the low temperatures.

476   Many other monosaccharides, disaccharides, polysaccharides and polyols, were detected in PM_{10}.
477   However, due to structural isomerism and identical fragmentation patterns in the mass spectra, their
478   individual quantification was not possible. Thus, a mean response factor of the calibrated compounds
479   with the same functional groups was applied. A much higher indoor concentration for these unidentified
480   saccharides and a strong nocturnal decay was observed, suggesting that many of these compounds were
481   produced by the day-to-day activities of the cafeteria.

482   Many phenolic compounds and derivatives were detected in both indoor and outdoor samples. These
483   constituents are used in commercial or consumer products and in building materials. Methoxy- and
484   dimethoxyphenols, such as vanillic acid, sinapyl alcohol and syringic acid, are also found in smoke
485   particles from biomass burning (Vicente and Alves, 2018). Among aromatic compounds, benzyl alcohol
486   is worth mentioning because of the high concentrations. It is used as a solvent, and a preservative. It is
487   also employed in inks, textiles and sheet plastics, and as a fragrance in perfumes and in flavouring.

488   Glycerol and many oxygenated derivatives were present at much higher concentrations in the
489   cafeteria compared to the outside. During the cooking process, triglycerides (i.e. fatty acids esterified
490   to a glycerol backbone) of fat in uncooked meat are hydrolysed or thermally oxidised, forming free
491   glycerol, free fatty acids and mono- and diglycerides (Abdullahi et al., 2013).

492   Cholesterol and phytosterols were abundant compounds in PM_{10}. Concentrations in the cafeteria
493   during the occupancy period far exceeded those recorded outdoors. Cholesterol (C_{27}) is biosynthesised
494   by higher animals and found in body tissues, especially in oils and fats. Phytosterols, such as β-sitosterol
495   and stigmasterol (C_{29}), are present in plant lipid waxes and membranes. All these sterols were previously
496   observed in $PM_{2.5}$ emitted from Chinese cooking (He et al., 2004; Zhao et al, 2007). However,
497   cholesterol is the only sterol that have been detected in meat cooking fumes (Rogge et al., 1991).

498   In the present study, many unusual compounds were detected in indoor aerosols, as far as we know,
499   for the first time. Some of the substances are part of the list of ingredients of various cosmetics. Ethylene
500   brassylate, for example, which was only present in indoor samples, is a macrocyclic compound mainly
501   used as a fragrance ingredient for its musk-like odour. Parsol MCX (ethylhexyl methoxycinnamate) is
502   used in the formulation of a wide variety of brands, including sunscreens and makeup products that
503   contain ingredients to protect the skin from the sun. A mean I/O ratio of 122 suggests its emission from
504   personal care products and concentration build-up in confined spaces, such as the cafeteria. Dibutyl
505   adipate is the diester of butyl alcohol and adipic acid. In cosmetics and personal care products, it is used
506   in nail polish and skin care products. Once again, much higher concentrations indoors than outdoors
507   (I/O=23) indicate a possible origin in the products used by the occupants. Because of its floral fragrance,
508   methyl dihydrojasmonate is an additive in a wide range of cleaning and personal care products. It was
509   absent from outdoor samples, but the indoor concentrations did not show appreciable differences
510   between occupancy and non-occupancy periods, suggesting the presence of a permanent source. Acetyl
511   tributyl citrate is an organic compound that is employed as a solvent in paints, inks, and nail enamel. It
512   may also be found in eye makeup and aerosol hair sprays. Moreover, it is used to make plastics more

flexible (e.g. plasticisers in packaging films for food), added as a flavour ingredient in non-alcoholic beverages and applied in the manufacture of many pharmaceutical drugs. Acetyl tributyl citrate was only detected in indoor aerosols. Pyroglutamic acid, known either as pyrrolidone-5-carboxylic acid, 5-oxoproline, PCA or pidolic acid, is a ubiquitous natural amino acid derivative. It is used for dry skin and hair products, owing to its humectant characteristics. Pidolic acid is found in large quantities in skin and brain tissues. It is sold online as a nootropic dietary supplement to improve cognitive and executive functions, and memory. The presence of this acid was only observed indoors, and at much higher concentrations during the occupancy periods.

2-Propanol-1-chloro-phosphate, also known as Amgard TMCP or Hostaflam OP 820, is a chlorine containing alkyl phosphate ester. It is mainly employed as an additive flame retardant, especially in polyurethane foam, with some minor use in other plastics and as a back coating in textiles. It was only present in indoor samples collected during daytime, including weekend. This suggests that the emission of this compound can be promoted by sunlight, although its formation processes and detection in the particulate phase deserve further investigation.

Another compound that was only detected indoors was piperonyl butoxide (PBO). It is a man-made organic compound used as a component of pesticide formulations, particularly insecticides, and classified as a synergist. By itself, PBO does not harm insects. Instead, it acts with insect killers to intensify their effectiveness. Boric acid, which was present in both indoor and outdoor samples, is also a pesticide that can be used to control a wide variety of pests, including insects, spiders, mites, algae, moulds, fungi, and weeds.

Some psychoactive substances, including drugs of abuse (e.g. tetrahydrocannabinol, THC), were found in the particulate matter samples. THC is the principal psychoactive constituent of cannabis. THC was only present in two indoor samples collected during the occupancy periods, reaching 21.7 ng m^{-3} on Thursday, the day of the student party. THC was previously detected as a constituent of outdoor airborne particles in 6 Spanish urban locations at concentrations ranging from 23 to 44 pg m^{-3} (Viana et al., 2012). Another psychoactive substance in airborne particles was caffeine. However, this compound was only detected in outdoor samples and during non-occupancy periods. Since caffeine is expected to partition between the gas and the particulate matter (Viana et al., 2012), its absence during working hours may be related to the higher temperatures in the cafeteria because of the use of heating. In the outdoor air across Italy, caffeine reached gross average concentrations of 15±8 ng m^{-3} in winter and 1.2±0.6 ng m^{-3} in summer, while in the metropolitan area of Rome the corresponding levels were 1.5±0.5 ng m^{-3} and 0.3±0.1 ng m^{-3} (Viana et al., 2012; and references therein).

Nicotinic acid, also denominated niacin or vitamin B3, is present in all living cells and found in whole and processed foods. It is used to make other chemicals and added to feeds and flours as a dietary supplement. It is also used as a medication to treat dyslipidaemic states. Niacin was not detected in outdoor air but reached concentrations up to 16.1 ng m^{-3} in the cafeteria during the occupancy period.

549 The mechanisms by which it is formed and remains airborne in the particulate phase are yet to be
550 clarified.

551   Irgafos 168 (tris(2,4-di-tert-butylphenyl)phosphite), as one of the common antioxidants, is
552 extensively used in polyolefin materials (e.g. polypropylene employed in food packaging), to protect
553 polymers from aging and oxidation. However, irgafos 168 may degrade into several products, such as
554 2,4-di-tert-butylphenol (DP1), and tris(2,4-di-tert-butylphenyl)phosphate (DP2), which is the oxidised
555 form of irgafos 168 (Yan et al., 2018). In the present study, whilst DP1 was detected at 5-fold higher
556 concentrations in the cafeteria than outside, DP2 was only present in two daytime indoor samples (both
557 with 19 ng m^{-3}) and in just one outdoor sample (39 ng m^{-3}).

558

559 *3.2.3. Health risk analysis*

560

561   Noncarcinogenic and carcinogenic risks associated with inhalation exposure to trace elements in
562 indoor PM$_{10}$ by the employees were estimated following the methodology proposed by the United States
563 Environmental Protection Agency (USEPA, 1989, 2009):

564

565 THQ = (EF × ED × ET× C)/(RfC × AT)   (6)
566 TR = (EF × ED × ET × C × IUR)/AT   (7)

567

568 The target hazard quotient (THQ) and target carcinogenic risk (TR) are dimensionless, EF is the
569 occupational exposure frequency (250 days per year, i.e. 5 days per 50 weeks), ED is the exposure
570 duration (25 years), ET is the exposure time (8 h per day), C is the metal concentration in the air (mg
571 m^{-3}), and AT is the averaging time (365 days per year × 24 h per day × ED for noncarcinogenic effects
572 and 612,000 h, i.e., 70 years × 365 days per year× 24 h per day, for carcinogenic effects). RfC is the
573 USEPA reference concentration (mg m^{-3}). Considering that for some elements, reference doses for oral
574 exposure (RfD, mg kg^{-1} day^{-1}) are available instead of RfC values (USEPA, 2017, 2019), these latter
575 were derived using the following equation (USEPA, 2013):

576

577 RfC = (RfD × BW) / IR   (8)

578

579 where IR and BW are the inhalation rate and body weight of an adult (20 m^{3} day^{-1} and 70 kg). Chronic
580 inhalation unit risk (IUR) values for the carcinogenic elements found in the PM$_{10}$ samples were taken
581 from USEPA (2017): arsenic 4.3×10^{-3} (μg m^{-3})$^{-1}$, lead 1.2×10^{-5} (μg m^{-3})$^{-1}$, hexavalent chromium
582 8.4×10^{-2} (μg m^{-3})$^{-1}$ and Ni oxide 2.6 × 10^{-5} (μg m^{-3})$^{-1}$. The IUR of Cr(VI) is based on a Cr(III):Cr(VI)
583 proportion of 1/6. Since in the present study total Cr was determined, one seventh of the concentration
584 obtained was used to estimate the risk.

585   The noncarcinogenic risks associated with inhalation exposure by employees to particulate trace
586   elements in the indoor air during the occupancy period are presented in Figure 8. A THQ < 1 indicates
587   no significant (acceptable) risk, a THQ > 1 suggests that the noncarcinogenic effect is likely to manifest,
588   whereas a high chronic risk is evident when THQ > 10 (Slezakova et al., 2014). The estimated mean
589   THQ for PM_{10}-bound trace elements ranged from null values for many elements 0.92 for Cl. The total
590   mean THQ of all eleven elements in PM_{10} was 1.7. Cl and Mn were the most important contributors (55
591   and 38 %, respectively) to the total noncarcinogenic risk, followed by Si (6.0 %), and Al (1.3 %). If the
592   exposure time and frequency assumed for the working hours were considered for non-occupancy
593   periods and for outdoor air, ΣTHQ values of 0.17-0.26 and around 2.0 would be obtained, respectively.
594   For carcinogenic constituents, USEPA considers that targeting a 10^{-6} risk level for individual
595   chemicals and pathways will usually lead to negligible risks. However, caution is suggested to guarantee
596   that the cumulative cancer risk for all potential carcinogenic contaminants does not have a residual
597   cancer risk above 10^{-4} (Slezakova et al., 2014). In the present study, ΣTR was always $< 2 \times 10^{-6}$,
598   suggesting that the risk associated with the exposure to PM_{10}-bound carcinogenic elements is negligible.
599   To estimate the carcinogenic risks associated with PAH inhalation, the benzo[a]pyrene equivalent
600   carcinogenicity (BaP_{eq}) was evaluated by multiplying concentration of each PAH with their toxicity
601   equivalent factor (TEF) (Bari et al., 2010, and references therein). The exposure concentration (EC) is
602   calculated as follows (CalEPA, 2005):

604    $EC = (CA \times ET \times EF \times ED) / AT$ (9)

606   where CA represent the benzo[a]pyrene equivalent concentrations (ng m^{-3}). The inhalation unit risk
607   (IUR) of respiratory cancer for BaP_{eq} is 3.9×10^{-6} (ng kg day)$^{-1}$. This value is obtained multiplying the
608   inhalation cancer risk unit factor of BaP (1.1×10^{-6} (ng kg day)$^{-1}$) by the inhalation rate (20 m^3 per day)
609   and dividing by the body weight (70 kg). The carcinogenic risk of the PAH mixture is then calculated
610   using the following equation:

612    $Risk = IUR \times EC$ (10)

614   The mean estimated excess inhalation cancer risk associated with total BaP_{eq} concentrations in the
615   cafeteria during the occupancy periods was 3×10^{-8}, which is considered negligible. Outdoor and non-
616   occupancy PAH levels also pose insignificant risks. However, it should be noted that other routes of
617   exposure (dermal and ingestion) have not been considered and that only particle-bound PAH were used
618   in the estimates. After formation and emission, these compounds partition between the gas phase and
619   atmospheric aerosols.
620

4. Conclusions

A short, but comprehensive, air quality monitoring programme was carried out in a university cafeteria, a type of occupational environment for which no inspections have been made so far. The mean values obtained for comfort parameters and regulated pollutants were, generally, within the ranges stipulated by international organisations. However, real time measurements of CO_2 and particulate matter revealed strong fluctuations, depending on activities and occupancy rates. Very low air change rates and particle infiltration factors from outdoors indicate inadequate ventilation. For most pollutants, the indoor levels during the working hours largely exceeded those measured at night and outdoors, pointing to the presence of multiple indoor sources. It was observed that more than 80 % of the particle mass concentrations were generated indoors. Several metals, either crustal or anthropogenic, were detected in the particulate matter. About 200 organic compounds were also identified, some of which, to the best of our knowledge, are described as indoor aerosol constituents for the first time. Many organic compounds detected in indoor air are components of various cosmetics and other personal care products, but also of pesticides, drugs of abuse, plastics, building materials (e.g. flame retardants), cooking emissions, sweat, among others. Some compounds can also be secondarily formed as by-products of surface reactions. However, the formation pathways of many compounds and the partitioning mechanisms between the gas and the particulate phases of some semi-volatiles are complex and still far from being well elucidated, deserving extensive investigation in the future.

The total mean hazard quotient that represents the noncarcinogenic effects due to the inhalation of particle-bound metals during the occupancy period in the cafeteria was higher than the acceptable level (>1). The cumulative cancer risk for both carcinogenic metals and PAHs was lower than the acceptable level (10^{-4}). However, exposure to metals and PAHs occurs also via ingestion and dermal contact and if these routes are considered, then the estimated risks might be higher.

To protect the restaurant workers and other public, possible strategies to control and minimise the emissions of indoor pollutants at their point sources within the cafeteria, such as confining the cooking area and equipping the kitchen with a more efficient smoke exhaust system, should be targeted. Other measures could be the installation of appropriate ventilation and air cleaner systems in the dining room, the use of eco-labelled cleaning products, the adoption of integrated pest management (IPM) alternatives to pesticides, placing footwear sanitiser mats at the entrance, inspection of wall coverings and other building materials and, if necessary, replacement by low emitting materials, minimising the use of plastics and compliance with prohibitions (e.g. smoking). At the official level, the implementation of regular inspection protocols is also essential.

674

675 **References**

676

677 Abdullahi, K.L., Delgado-Saborit, J.M., Harrison, R.M., 2013. Emissions and indoor concentrations of particulate
678   matter and its specific chemical components from cooking: a review. Atmos. Environ. 71, 260–294.

679 Abdul-Wahab S.A., En, S.C.F., Elkamel, A., Ahmadi, L., Yetilmezsoy, K., 2015. A review of standards and
680   guidelines set by international bodies for the parameters of indoor air quality Atmos. Pollut. Res. 6, 751-767.

681 Ahearn, D.G., Crow, S.A., Simmons, R.B., Price, D.L., Noble, J.A., Mishra, S.K., Pierson, D.L., 1996. Fungal
682   colonization of fiberglass insulation in the air distribution system of a multi-story office building: VOC
683   production and possible relationship to a sick building syndrome. J. Ind. Microbiol. 16, 280-285.

684 Al horr, Y., Arif, M., Katafygiotou, M., Mazroei, A., Kaushik, A., Elsarrag, E., 2016. Impact of indoor
685   environmental quality on occupant well-being and comfort: A review of the literature. Int. J. Sust. Built
686   Environ. 5, 1-11.

687 Alam, M.S., Keyte, I.J., Yin, J., Alan, C.S., Jones, A.M., Harrison, R.M., 2015. Diurnal variability of polycyclic
688   aromatic compound (PAC) concentrations: Relationship with meteorological conditions and inferred sources.
689   Atmos. Environ. 122, 427-438.

690 Almeida, S.M., Ramos, C.A., Almeida-Silva, M., 2016. Exposure and inhaled dose of susceptible population to
691   chemical elements in atmospheric particles. J. Radioanal. Nucl. Chem. 309, 309-315.

692 Alves, C., Nunes, T., Silva, J., Duarte, M., 2013. Comfort parameters and particulate matter (PM_{10} and $PM_{2.5}$) in
693   school classrooms and outdoor air. Aerosol Air Qual. Res. 13, 1521–1535.

694   Alves, C.A., Vicente, A., Monteiro, C., Gonçalves, C., Evtyugina, M., Pio, C., 2011. Emission of trace gases and
695   organic components in smoke particles from a wildfire in a mixed evergreen forest in Portugal. Sci. Total
696   Environ. 409, 1466–1475.

697   Araki, A., Saito, I., Kanazawa, A., Morimoto, K., Nakayama, K., Shibata, E., Tanaka, M., Takigawa, T.,
698   Yoshimura, T., Chikara, H., Saijo, Y., Kishi, R., 2014. Phosphorus flame retardants in indoor dust and their
699   relation to asthma and allergies of inhabitants. Indoor Air 24, 3-15.

700   Bari, M.A., Baumbach, G., Kuch, B., Scheffknecht, G., 2010. Particle-phase concentrations of polycyclic aromatic
701   hydrocarbons in ambient air of rural residential areas in southern Germany. Air Qual. Atmos. Health 3, 103–
702   116.

703   Barmparesos, N., Assimakopoulos, M.N., Assimakopoulos, V.D., Loumos, N., Sotiriou, M.A., Koukoumtzis, A.,
704   2018. Indoor air quality and thermal conditions in a primary school with a green roof system. Atmosphere 9,
705   75. https://doi.org/10.3390/atmos9020075

706   Bennett, J., Davy, P., Trompetter, B., Wang, Y., Pierse, N., Boulic, M., Phipps, R., Howden-Chapman, P., 2019.
707   Sources of indoor air pollution at a New Zealand urban primary school; a case study. Atmos. Pollut. Res. 10,
708   435-444.

709   Bentayeb, M., Norback, D., Bednarek, M., Bernard, A., Cai, G., Cerrai, S., Eleftheriou, K.K., Gratziou, C., Holst,
710   G.J., Lavaud, F., Nasilowski, J., Sestini, P., Sarno, G., Sigsgaard, T., Wieslander, G., Zielinski, J., Viegi, G.,
711   Annesi-Maesano, I. on behalf of the GERIE Study, 2015. Indoor air quality, ventilation and respiratory health
712   in elderly residents living in nursing homes in Europe. Eur. Respir. J. 45, 1228-1238.

713   Blanco-Alegre, C., Calvo, A.I., Coz, E., Castro, A., Oduber, F., Prévôt A.S.H., Močnik, G., Fraile, R., 2019.
714   Quantification of source specific black carbon scavenging using an aethalometer and a disdrometer. Environ.
715   Pollut. 246, 336-345.

716   Blondeau, P., Iordache, V., Poupard, O., Genin, D., Allard, F., 2005. Relationship between outdoor and indoor air
717   quality in eight French schools. Indoor Air 15, 2-12.

718   Borowska, S., Brzóska, M.M., 2015. Metals in cosmetics: implications for human health. J. Appl. Toxicol. 35,
719   551–572.

720   Buczyńska, A.J., Krata, A., Van Grieken, R., Brown, A., Polezer, G., De Wael, K., Potgieter-Vermaak, S., 2014.
721   Composition of $PM_{2.5}$ and PM_1 on high and low pollution event days and its relation to indoor air quality in a
722   home for the elderly. Sci. Total Environ. 490, 134-143.

723   CalEPA, 2005. Air Toxics Hot Spots Program Risk Assessment Guidelines. Part 2: Technical Support Document
724   for Describing Available Cancer Potency Factors. Office of Environmental Health Hazard Assessment,
725   Oakland, CA.

726   Carrer, P., Wolkoff, P., 2018. Assessment of indoor air quality problems in office-like environments: role of
727   occupational health services. Int. J. Environ. Res. Public Health 15, 741.
728   https://doi.org/10.3390/ijerph15040741

729   Castro, B.P., Machado, G.S., Bauerfeldt, G.F., Fortes, G.D., Martins, E.M., 2015. Assessment of the BTEX
730   concentrations and reactivity in a confined parking area in Rio de Janeiro, Brazil. Atmos. Environ. 104, 22-
731   26.

732   Chatzidiakou, E., Mumovic, D., Summerfield, A.J., Altamirano, H.M., 2015. Indoor air quality in London schools.
733   Part 1: 'performance in use'. Intell. Build. Int. 7, 101–129.

734   Cincinelli, A., Martellini, T., 2017. Indoor air quality and health. Int. J. Environ. Res. Public Health 14, 1286.
735   https://doi.org/10.3390/ijerph14111286

736   Cincinelli, A., Martellini, T., Amore, A., Dei, L., Marrazza, G., Carretti, E., Belosi, F., Ravegnani, F., Leva, P.,
737   2016. Measurement of volatile organic compounds (VOCs) in libraries and archives in Florence (Italy). Sci.
738   Total Environ. 572, 333-339.

739   Dai, H., Jing, S., Wang, H., Ma, Y., Li, L., Song, W., Kan, H., 2017. VOC characteristics and inhalation health
740   risks in newly renovated residences in Shanghai, China. Sci. Total Environ. 577, 73-83.

741   Dai, W., Zhong, H., Li, L., Cao, J., Huang, Y., Shen, M., Wang, L., Dong, J., Tie, X., Ho, S.S.H., Ho, K.F., 2018.
742   Characterization and health risk assessment of airborne pollutants in commercial restaurants in northwestern
743   China: Under a low ventilation condition in wintertime. Sci. Total Environ. 633, 308-316.

744   Demirel, G., Özden, Ö, Döğeroğlu, T., Gaga, E., 2014. Personal exposure of primary school children to BTEX,
745   NO_2 and ozone in Eskişehir, Turkey: relationship with indoor/outdoor concentrations and risk assessment. Sci.
746   Total Environ. 473–474, 537-548.

747   El-Hashemy, M.A., Ali, H.M., 2018. Characterization of BTEX group of VOCs and inhalation risks in indoor
748   microenvironments at small enterprises. Sci. Total Environ. 645, 974-983.

749   Elsharkawy, M., Javed, W. 2018. Study of indoor air quality level in various restaurants in Saudi Arabia. Environ.
750   Prog. Sustain.37, 1713-1721.

751   Esplugues, A., Ballester, F., Estarlich, M., Llop, S., Fuentes-Leonarte, V., Mantilla, E., Iñiguez, C., 2010. Indoor
752   and outdoor air concentrations of BTEX and determinants in a cohort of one-year old children in Valencia,
753   Spain. Sci. Total Environ. 409, 63-69.

754   European Commission, 2009. Directive 2008/50/EC of the European Parliament and of the Council of 21 May
755   2008 on ambient air quality and cleaner air for Europe. http://ec.europa.eu/environment/air/legis.html

756   Fan, G., Xie, J., Yoshino, H., Yanagi, U., Hasegawa, K., Kagi, N., Goto, T., Zhang, Q., Wang, C., Liu, J., 2018.
757   Indoor environmental conditions in urban and rural homes with older people during heating season: a case in
758   cold region, China. Energ. Build. 167, 334-346.

759   Federal Institute for Occupational Safety and Health, 2017. Substance Evaluation Conclusion as required by
760   REACH Article 48 and Evaluation Report for *n*-Hexane. BAUA, Dortmund, Germany.

761   Fuoco, F.C., Stabile, L., Buonanno, G., Trassiera, C.V., Massimo, A., Russi, A., Mazaheri, M., Morawska, L.,
762   Andrade, A., 2015. Indoor air quality in naturally ventilated Italian classrooms. Atmosphere 6, 1652-1675.

763   He, L.Y., Hua, M., Huang, X.F., Yu, B.D., Zhang, Y.H., Liu, D.Q., 2004. Measurement of emissions of fine
764   particulate organic matter from Chinese cooking. Atmos. Environ. 38, 6557–6564.

765   Hsu, C.S., Huang, D.J., 2013. Disinfection efficiency of chlorine dioxide gas in student cafeterias in Taiwan. J.
766   Air Waste Manage. 63, 796-805.

767   Huang, S., Xiong, J., Cai, C., Xu, W., Zhang, Y., 2016. Influence of humidity on the initial emittable concentration
768   of formaldehyde and hexaldehyde in building materials: experimental observation and correlation. Sci. Rep.
769   6, Article nº 23388. https://doi.org/10.1038/srep23388

770   Kalimeri, K.K., Saraga, D.E., Lazaridis, V.D., Legkas, N.A., Missia, D.A., Tolis, E.I., Bartzis, J.G., 2016. Indoor
771   air quality investigation of the school environment and estimated health risks: Two-season measurements in
772   primary schools in Kozani, Greece. Atmos. Pollut. Res. 7, 1128-1142

773 Lee, S.C., Li, W.M., Chan, L.Y., 2001. Indoor air quality at restaurants with different styles of cooking in
774   metropolitan Hong Kong. Sci. Total Environ. 279, 181-193.

775 Lee, H., Shuai, J., Woo, B., Hwang, M.Y., Park, C.H., Yu, S.D., Yang, W., 2011. Indoor air quality at restaurants
776   and bars in evening hours in Korea. Epidemiology 22, S219
777   https://doi.org/10.1097/01.ede.0000392356.22828.af

778 Li, A., Zhao, Y., Jiang, D., Hou, X., 2012. Measurement of temperature, relative humidity, concentration
779   distribution and flow field in four typical Chinese commercial kitchens. Build. Environ. 56, 139–150.

780 Liu, Q., Liu, Y., Zhang, M., 2013. Personal exposure and source characteristics of carbonyl compounds and
781   BTEXs within homes in Beijing, China. Build. Environ. 61, 210-216.

782 Liu, S., Thompson, S.L., Stark, H., Ziemann, P.J., Jimenez, J.L., 2017. Gas-phase carboxylic acids in a university
783   classroom: Abundance, variability, and sources. Environ. Sci. Technol. 51, 5454-5463

784 Lucarelli, F., Calzolai, G., Chiari, M., Giannoni, M., Mochi, D., Nava, S., and Carraresi, L., 2014. The upgraded
785   external-beam PIXE/PIGE set-up at LABEC for very fast measurements on aerosol samples. Nucl. Instrum.
786   Meth. Phys. Res. B 318, 55–59.

787 Lv, Y., Wang, H., Wei, S., Zhang, L., Zhao, Q., 2017. The correlation between indoor and outdoor particulate
788   matter of different building types in Daqing, China. Procedia Eng. 205, 360–367.

789 Madureira, J., Paciência, I., Pereira, C., Teixeira, J.P., Fernandes Ede, O., 2016. Indoor air quality in Portuguese
790   schools: levels and sources of pollutants. Indoor Air 26, 526–537.

791 Mandin, C., Trantallidi, M., Cattaneo, A., Canha, N., Mihucz, V.G., Szigeti, T., Mabilia, R., Perreca, E., Spinazzè,
792   A., Fossati, S., Kluizenaar, Y., Cornelissen, E., Sakellaris, I., Saraga, D., Hänninen, O., Oliveira Fernandes,
793   E., Ventura, G., Wolkoff, P., Carrer, P., Bartzis, J., 2017. Assessment of indoor air quality in office buildings
794   across Europe – The OFFICAIR study. Sci. Total Environ. 579, 169-178.

795 Marć, M., Śmiełowska, M., Namieśnik, J., Zabiegała, B., 2018. Indoor air quality of everyday use spaces dedicated
796   to specific purposes - a review. Environ. Sci. Pollut. Res. 25, 2065–2082.

797 Mason, P.E., Darvell, L.I., Jones, J.M., Williams, A., 2016. Observations on the release of gas-phase potassium
798   during the combustion of single particles of biomass. Fuel 182, 110-117.

799 Mendes, A., Bonassi, S., Aguiar, L., Pereira, C., Neves, P., Silva, S., Mendes, D., Guimarães, L., Moroni, R.,
800   Teixeira, J.P., 2015. Indoor air quality and thermal comfort in elderly care centers. Urban Clim. 14, 486–501.

801 Mishra, U.C., Shaikh, G.N., Sadasivan, S., 1986. Trace elements in tobacco and tobacco smoke by X-ray
802   fluorescence technique. J. Radioanal. Nucl. Chem. 102, 27–35.

803 Mugica, V., Vega, E., Chow, J., Reyes, E., Sánchez, G., Arriaga, J., Egami, R., Watson, J., 2001. Speciated non-
804   methane organic compounds emissions from food cooking in Mexico. Atmos. Environ. 35, 1729-1734.

805 Mui, K.W., Wong, L.T., Fong, K.N.K., Wong, W.W.Y., 2008. Evaluation of indoor environment quality of
806   elderly centers of Hong Kong. Int. J. Housing Sci. 32, 121–131.

807 Nilsson, U., Berglund, N., Lindahl, F., Axelsson, S., Redeby, T., Lassen, P., Karlberg, A.T., 2008. SPE and
808   HPLC/UV of resin acids in colophonium-containing products. J. Sep. Sci. 2008 31, 2784-2790.

809 Omenka, S.S., Adeyi, A.A., 2016. Heavy metal content of selected personal care products (PCPs) available in
810   Ibadan, Nigeria and their toxic effects. Toxicol. Rep. 3, 628-635.

811 Othman, M., Latif, M.T., Matsumi, Y., 2019. The exposure of children to $PM_{2.5}$ and dust in indoor and outdoor
812   school classrooms in Kuala Lumpur City Centre. Ecotoxicol. Environ. Saf. 170, 739-749.

813   Pagonis, D., Price, D.J., Algrim, L.B., Day, D.A., Handschy, A.V., Stark, H., Miller, S.L, de Gouw, J., Jimenez,
814   J.L., Ziemann, P.J., 2019. Time-resolved measurements of indoor chemical emissions, deposition, and
815   reactions in a university art museum. Environ. Sci. Technol. 53, 4794-4802.

816   Pant, P., Harrison, R.M., 2013. Estimation of the contribution of road traffic emissions to particulate matter
817   concentrations from field measurements: A review. Atmos. Environ. 77, 78-97.

818   Pegas, P., Alves, C.A., Evtyugina, M., Nunes, T., Cerqueira, M., Franchi, M., Pio, C., Almeida, S.M., Cabo Verde,
819   S., Freitas, M.C., 2011. Seasonal evaluation of outdoor/indoor air quality in primary schools in Lisbon. J.
820   Environ. Monit. 13, 657-667.

821   Pegas, P.N., Evtyugina, M.G., Alves, C., Nunes, T., Cerqueira, M., Franchi, M., Pio, C., Almeida, S.M., Freitas,
822   M.C., 2010. Outdoor/indoor air quality in primary schools in Lisbon: a preliminary study. Quím. Nova. 33,
823   1145-1149.

824   Polidori, A., Turpin, B.J., Davidson, C.I., Rodenburg, L.A., Maimone, F., 2008. Organic $PM_{2.5}$: fractionation by
825   polarity, FTIR spectroscopy, and OM/OC ratio for the Pittsburgh aerosol. Aerosol Sci. Technol. 42, 233-246.

826   Rasmussen, P.E., Levesque, C., Chénier, M., Gardner, H.D., 2018. Contribution of metals in resuspended dust to
827   indoor and personal inhalation exposures: Relationships between PM_{10} and settled dust. Build. Environ. 143,
828   513–522.

829   Raysoni, A.U., Stock, T.H., Sarnat, J.A., Chavez, M.C.C., Sarnat, S.E., Montoya, T., Holguin, F., Li, W., 2017.
830   Evaluation of VOC concentrations in indoor and outdoor microenvironments at near-road schools. Environ.
831   Pollut. 231, 681-693.

832   Rivas, I., Viana, M., Moreno, T., Bouso, L., Pandolfi, M., Alvarez-Pedrerol, M., Forns, J., Alastuey, A., Sunyer,
833   J., Querol, X., 2015. Outdoor infiltration and indoor contribution of UFP and BC, OC, secondary inorganic
834   ions and metals in $PM_{2.5}$ in schools. Atmos. Environ. 106, 129-138.

835   Rogge, W.F., Hildemann, L.M., Mazurek, M.A., Cass, G.R., Simoneit, B.R.T., 1991. Sources of fine organic
836   aerosol. 1. Charbroilers and meat cooking operations. Environ. Sci. Tecnol. 25, 112-1125.

837   Ruprecht, A.A., De Marco, C., Saffari, A., Pozzi, P., Mazza, R., Veronese, C., Angellotti, G., Munarini, E.,
838   Ogliari, A.C., Westerdahl, D., Hasheminassab, S., Shafer, M.M., Schauer, J.J., Repace, J., Sioutas, C., Boffi.
839   R., 2017. Environmental pollution and emission factors of electronic cigarettes, heat-not-burn tobacco
840   products, and conventional cigarettes. Aerosol Sci. Technol. 51, 674–684

841   Sofuoglu, S.C., Toprak, M., Inal, A., Cimrin, A.H., 2015. Indoor air quality in a restaurant kitchen using margarine
842   for deep-frying. Environ. Sci. Pollut. Res. 22, 15703–15711.

843   Simoni, M., Jaakkola, M.S., Carrozzi, L., Baldacci, S., Di Pede, F., Viegi, G., 2003. Indoor air pollution and
844   respiratory health in the elderly. Eur. Respir. J. 21, 15s-20s.

845   Slezakova, K., Morais, S., Pereira, M.C., 2014. Trace metals in size-fractionated particulate matter in a Portuguese
846   hospital: exposure risks assessment and comparisons with other countries. Environ. Sci. Pollut. Res. 21, 3604–
847   3620.

848   Śmiełowska, M., Marć, M., Zabiegała, B. 2017. Indoor air quality in public utility environments - a review.
849   Environ. Sci. Pollut. Res. 24, 11166–11176.

850   Taner, S., Pekey, B., Pekey, H., 2013. Fine particulate matter in the indoor air of barbeque restaurants: Elemental
851   compositions, sources and health risks. Sci. Total Environ. 454–455, 79-87.

852 The Engineering Toolbox, 2005. Air Change Rates in typical Rooms and Buildings. https://www.engineeringtoolbox.com/air-change-rate-room-d_867.html (accessed on 20 May 2019).

854 Toxno, 2018. Toxno Substance Profile: Hexane. https://www.toxno.com.au/toxins/substance_id_998.html (accessed on 26 November 2019).

856 Tsyro, S.G., 2005. To what extent can aerosol water explain the discrepancy between model calculated and gravimetric PM_{10} and $PM_{2.5}$? Atmos. Chem. Phys. 5, 515-532.

858 USEPA, 1989. Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual (Part A). EPA/540/1-89/002. US Environmental Protection Agency, Office of Emergency and Remedial Response. Washington, DC, USA. https://www.epa.gov/sites/production/files/2015-09/documents/rags_a.pdf (accessed on 3 June 2019).

862 USEPA, 2009. Risk Assessment Guidance for Superfund Volume I: Human Health Evaluation Manual (Part F, Supplemental Guidance for Inhalation Risk Assessment). EPA/540/R/070/002. US Environmental Protection Agency, Office of Emergency and Remedial Response. Washington, DC, USA. https://www.epa.gov/sites/production/files/2015-09/documents/partf_200901_final.pdf (accessed on 3 June 2019).

867 USEPA, 2013. Users' Guide and Background Technical Document for USEPA Region 9 - Preliminary Remediation Goals (PRG) Table. Available online: https://semspub.epa.gov/work/02/103453.pdf (accessed on 3 June 2019).

870 USEPA, 2017. Regional Screening Level (RSL) Summary Table. Available online: https://semspub.epa.gov/work/03/2245073.pdf (accessed on 3 June 2019).

872 USEPA. 2019. IRIS Assessments. Available online: https://cfpub.epa.gov/ncea/iris2/atoz.cfm (accessed on 3 June 2019).

874 Viana, M., Postigo, C., Balducci, C., Cecinato, A., Alda, M.J.L., Barceló, D., Artiñano, B., López-Mahía, P., Alastuey, A., Quero,l X., 2012. Psychoactive substances in particles in the urban environment, in: Barceló D. (Ed.), Emerging Organic Contaminants and Human Heath, Springer-Verlag, Berlin, Heidelberg, pp. 435-460.

877 Vicente E.D., Alves C.A., 2018. An overview of particulate emissions from residential biomass combustion. Atmos. Res. 199, 159-185.

879 Vicente, E.D., Vicente, A., Evtyugina, M., Carvalho, R., Tarelho, L.A.C., Oduber, F.I., Alves, C., 2018. Particulate and gaseous emissions from charcoal combustion in barbecue grills. Fuel Process. Technol. 176, 296-306.

882 Wedepohl, K.H., 1995. The composition of the continental crust. Geochim. Cosmochim. Acta 59, 1217-1239.

883 Wilson, N., Parry, R., Jalali, J., Jalali, R., McLean, L., McKay, O., 2011. High air pollution levels in some takeaway food outlets and barbecue restaurants. Pilot study in Wellington City, New Zealand. N. Z. Med. J. 124, 81-86.

886 World Health Organization, 2010. WHO Guidelines for Indoor Air Quality: Selected Pollutants. World Health Organization, Bonn.

888 Yan, Y., Hu, S.Y., Wang, Z.W., Jiang, Z.W., 2018. Degradation of Irgafos 168 and migration of its degradation products from PP-R composite films. Packag. Technol. Sci. 31, 679–688.

890   Yang, J., Nam, I., Yun, H., Kim, J., Oh, H.J., Lee, D., Jeon, S.M., Yoo, S.H., Sohn, J.R., 2015. Characteristics of
891   indoor air quality at urban elementary schools in Seoul, Korea: Assessment of effect of surrounding
892   environments. Atmos. Pollut. Res. 6, 1113-1122.

893   Zhao, K., Zhou, X., Zhao, B., 2010. Cooking generated particles' impact on indoor air quality of university
894   cafeteria. Build. Simul. 3, 15–23.

895   Zhao, Y., Hu, M., Slanina S., Zhang, Y. 2007. Chemical of fine particulate organic matter emitted from Chinese
896   cooking. Environ. Sci. Technol. 41, 99-105.

Table 1

Weekly concentrations of volatile organic compounds (µg m^{-3}) monitored by passive sampling

	n-hexane	1-butanol	benzene	toluene	ethyl benzene	m-xylene	p-xylene	2-ethoxyethyl acetate	o-xylene	1,2,4-trimethylbenzene
Indoor	0.4-1.1	3.7-13.8	0.9-1.4	3.4-10.4	0.7-1.0	1.8-2.4	0.7-1.4	<0.3	0.8-1.1	0.8-1.2
Outdoor	0.1	<2.0	0.6-0.8	1.3-1.6	0.3-0.5	0.6-1.0	0.3-0.4	<0.3	0.3-0.5	0.4-0.5
I/O	3.8-12.6	-	1.3-1.8	2.1-6.2	1.5-3.1	1.9-3.2	1.5-3.1	-	1.4-2.9	1.7-3.0

Table 2

Minimum, maximum and average concentrations of elements (ng m^{-3}), enrichment factors and indoor/outdoor ratios for working hours

			Na	Mg	Al	Si	P	S	Cl	K	Ca	Ti	V	Cr	Mn	Fe	Ni	Cu	Zn	As	Se	Br	Rb	Sr	Y	Zr	Mo	Pb
Indoor	Occupancy Day	Min	197	69.0	173	347	20.5	100	214	305	452	11.1	-	0.418	3.93	93.4	0.356	2.73	10.3	-	-	1.24	0.426	1.26	-	0.900	-	0.603
		Max	356	776	1151	828	390	1070	1870	2140	850	33.9	0.682	5.47	8.72	271	3.91	14.9	60.9	1.41	3.89	4.31	1.25	4.94	0.304	2.03	2.05	5.98
		Avg	250	232	465	620	27.3	485	607	692	625	22.3	0.190	1.66	5.54	195	1.49	6.90	27.8	0.757	1.62	2.77	0.749	3.00	0.089	1.27	0.551	2.80
		EF	1.6	2.4	0.88	0.33	7.0	86	137	3.0	3.5	1.1	0.67	6.8	1.7	-	12	73	80	59	3094	284	1.1	1.5	0.64	0.89	68	26
Indoor	Night	Min	41.5	9.58	30.5	58.1	0.158	155	13.3	117	8.37	2.43	0.695	-	-	26.7	-	0.98	7.01	0.102	0.427	0.450	-	-	-	-	-	-
		Max	115	23.9	100	137	3.11	500	79.8	197	51.2	4.20	0.947	0.421	1.44	53.6	0.496	1.23	10.6	1.06	1.96	0.628	0.856	1.02	-	0.424	2.61	5.46
		Avg	73.2	14.8	71.8	110	1.49	305	37.8	148	28.6	3.19	0.790	0.215	0.766	39.7	0.277	1.09	8.27	0.532	1.15	0.566	0.285	0.627	-	0.141	0.944	2.51
		EF	2.3	0.86	0.74	0.29	1.7	220	47.1	3.2	0.79	0.78	11	5.1	1.2	-	13	56	115	217	9397	260	1.5	1.4		0.34	541	95
Outdoor	Day	Min	403	79.3	142	243	5.32	152	470	60.6	123	6.25	0.239	-	2.17	136	-	2.36	6.16	-	0.011	1.77	0.334	1.18	-	0.507	-	0.189
		Max	766	130	619	1363	26.3	1711	988	365	641	31.5	2.76	2.55	10.6	660	1.00	14.8	32.9	2.76	9.69	9.24	2.57	5.31	1.62	3.35	6.35	6.04
		Avg	550	102	424	802	15.3	570	722	184	356	20.7	1.23	1.11	6.09	397	0.459	8.15	17.0	0.958	2.61	3.75	1.30	3.66	0.800	1.69	2.03	2.86
		EF	2.3	0.75	0.44	0.20	1.8	32	106	0.39	0.91	0.52	1.7	1.7	0.90	-	1.5	42	24	23	1105	161	1.0	0.94	4.3	0.61	150	10
Indoor	Non-oc. Day	Concent.	224	24.7	41.0	72.6	5.45	182	114	35.5	21.0	2.43	-	-	0.621	25.7	0.450	0.867	1.13	0.585	0.924	0.819	-	-	-	-	1.72	-
		EF	11	2.2	0.64	0.29	9.8	230	214	1.3	0.86	0.93	-	-	1.4	-	29	73	26	351	13370	615	-	-	-	-	1474	-
I/O			0.46	2.3	1.1	0.77	1.8	0.85	0.84	3.8	1.8	1.1	0.16	1.5	0.91	0.49	3.3	0.85	1.6	0.79	-	0.74	0.57	0.82	0.11	0.75	-	0.98

Table 3

Concentrations (ng m^{-3}) of oxygenated organic compounds in PM$_{10}$

	Indoor occupancy	Indoor non-occupancy (night-time)	Indoor non-occupancy (Sunday, daytime)	Outdoor daytime	I/O (occupancy)
Saccharides					
Galactosan	1.60±0.61	2.05	0.422	4.5±4.0	0.4
Mannosan	3.72±2.12	4.49	0.904	8.4±6.2	0.4
Levoglucosan	39.5±11.4	41.9	bdl	59±55	0.7
Unidentified saccharides	90.6±46.6	0.84	2.00	10.6±4.9	8.5
Phenolics and alteration products					
Pyrocatechol	0.006±0.002	0.004	bdl	0.005±0.004	1.0
Resorcinol	0.019±0.012	0.073	0.003	0.10±0.12	0.2
4-Methylcatechol	0.004±0.002	0.004	bdl	0.003±0.003	1.1
5-Isopropyl-3-methylphenol	0.065±0.051	bdl	bdl	0.029±0.027	2.2
2-Isopropyl-5-methylphenol (thymol)	0.64±0.36	0.070	0.242	0.34±0.38	1.9
Eugenol	0.004±0.003	0.002	0.001	0.001±0.001	5.4
2,4-Di-tert-butylphenol	26.4±19.8	bdl	4.03	5.4±4.7	4.9
4-tert-Butylphenol + unknown	13.0±7.3	10.0	bdl	9±11	1.4
Pyrogallol	0.004±0.004	0.004	0.001	0.005±0.003	0.8
Isoeugenol	0.51±0.37	bdl	0.134	0.25±0.18	2.1
2-Methoxy-4-propylphenol	0.002±0.001	0.005	bdl	0.003±0.003	0.5
4-Phenylphenol	0.030±0.028	0.037	bdl	0.038±0.031	0.8
4-Octylphenol	0.011±0.007	0.002	bdl	0.006±0.004	1.8

Benzyl alcohol	10.0±4.5	0.050	3.24	18.0±20	0.6
Benzoic acid	0.38±0.43	0.063	bdl	0.024±0.029	15
3-Hydroxybenzoic acid	1.53±0.41	bdl	bdl	0.63±0.30	2.4
4-Hydroxybenzoic acid	0.65±0.23	0.587	0.097	0.82±0.67	0.8
Vanillic acid	0.35±0.23	0.172	0.118	0.56±0.36	0.6
Sinapyl alcohol	0.053±0.066	0.141	bdl	0.005±0.008	9.7
Syringic acid	0.20±0.22	0.452	0.073	0.24±0.25	0.8
trans-Cinnamic acid	0.22±0.23	bdl	0.071	0.080±0.065	2.8
4-Hydroxy-3-methoxycinnamic (ferulic) acid	0.032±0.028	0.004	bdl	0.013±0.009	2.4
3,4-Dihydroxycinnamic (caffeic) acid	0.013±0.017	bdl	bdl	0.001±0.001	9.0
4-Hydroxycinnamic (p-coumaric) acid	0.49±0.56	bdl	bdl	0.14±0.13	3.5
Aliphatic alcohols					
1-Decanol	0.043±0.038	0.001	0.004	0.005±0.007	9.3
1-Pentadecanol	11.8±7.6	8.63	13.2	bdl	-
iso- or anteiso-Pentadecanol	1.42±0.73	bdl	1.98	0.26±0.14	5.5
1-Hexadecanol	38.6±18.4	bld	72.2	0.06±0.13	681
2-Hexadecanol	4.8±2.1	1.60	bld	bdl	-
Iso- or anteiso-heptadecanol	0.56±0.64	bdl	0.860	bdl	-
1-Heptadecanol	2.2±2.0	1.62	bdl	bdl	-
1-Octadecanol	45±14	13.7	5.36	0.07±0.16	643
1-Eicosanol	2.05±0.80	1.27	0.399	1.2±1.1	1.7
1-Docosanol	2.25±0.87	2.14	0.320	2.0±1.3	1.2
1-Tricosanol	0.225±0.060	0.220	0.025	0.10±0.12	2.3
1-Tetracosanol	1.10±0.36	1.59	bdl	1.58±0.72	0.7

1-Pentacosanol	0.26±0.13	0.189	0.014	0.17±0.12	1.6
1-Hexacosanol	6.0±9.1	3.33	0.341	5.7±4.6	1.1
1-Heptacosanol	0.121±0.057	0.076	0.005	0.07±0.10	1.7
1-Octacosanol	1.5±0.21	1.76	0.240	2.4±3.3	0.6
1-Triacontanol	0.248±0.091	0.229	0.010	0.36±0.65	0.7
Triterpenoid and steroid compounds					
Tocopherol (Vit. E)	0.19±0.12	0.006	bdl	0.010±0.009	19
Cholesterol	20.0±8.5	0.170	0.019	0.14±0.15	139
5-Cholesten-3-ol (epicholesterol)	0.27±0.14	bdl	bdl	0.049±0.089	5.6
Stigmasterol	7.7±2.5	0.314	bdl	0.45±0.56	17
β-Sitosterol	5.7±2.1	1.37	0.008	0.8±1.4	7.0
Lupeol	0.204±0.072	1.11	0.123	0.42±0.44	1.0
Aliphatic acids					
Hexanoic	1.07±0.44	0.053	bdl	0.43±0.35	2.5
Heptanoic	4.8±3.6	0.073	0.059	0.37±0.33	13
Octanoic	2.4±1.9	0.370	0.142	0.46±0.40	5.3
Decanoic	2.7±3.0	bdl	0.106	0.37±0.31	7.4
Undecanoic	0.20±0.29	bdl	0.081	bdl	-
Nonanoic	37±37	1.13	1.35	2.4±2.1	15
Dodecanoic	9±13	1.97	0.051	3.1±5.7	3.0
Tridecanoic	2.9±2.8	1.12	3.03	bdl	-
Isotridecanoic	3.2±2.3	3.00	4.21	bdl	-
Tetradecanoic (myristic)	72±28	30.3	66.2	1.0±1.3	73
Pentadecanoic	15.7±4.6	0.286	7.29	0.06±0.11	266

Isotetradecanoic	13.4±6.5	3.780	10.9	bdl	-
Hexadecanoic (palmitic)	117±69	bdl	bdl	10.6±9.9	11
Iso-hexadecanoic	4.5±2.6	0.143	2.06	bdl	-
5-Hexadecenoic	3.10±0.86	bdl	1.13	bdl	-
9-Hexadecenoic (palmitoleic)	1.43±0.85	0.0524	0.106	0.080±0.099	18
Heptadecanoic	14.8±8.2	bdl	0.523	0.38±0.34	39
Isoheptadecanoic	7.6±2.6	0.095	1.55	bdl	-
Octadecanoic (stearic)	116±62	bdl	bdl	1.9±2.6	61
9-Octadecenoic (oleic)	103±35	1.99	0.219	1.1±1.1	93
9,12-Octadecadienoic (linoleic)	30±14	0.272	0.017	0.47±0.74	64
9,12,15-Octadecatrienoic (linolenic)	17±25	bdl	bdl	0.35±0.77	49
Nonadecanoic	0.99±0.30	0.022	0.053	0.140±0.082	6.5
Eicosanoic	8.0±2.9	0.618	0.200	2.2±2.0	3.7
Heneicosanoic	bdl	bdl	0.221	0.22±0.23	-
Docosanoic	9.0±3.2	2.46	0.171	2.2±2.1	4.1
Pentacosanoic	2.0±1.7	bdl	bdl	0.10±0.23	20
Hexacosanoic	1.36±0.66	0.509	0.046	0.54±0.57	2.5
Diacids					
Butanedioic (succinic)	2.8±2.4	bdl	0.691	2.3±1.4	1.2
1,5-Pentanedioic (glutaric)	3.9±2.6	1.52	5.25	0.07±0.14	58
Hydroxybutanedioic (malic)	0.7±1.0	bdl	bld	0.5±1.2	1.2
Hexanedioic (adipic)	1.07±0.31	bdl	0.388	bdl	-
Heptanedioic (pimelic)	1.28±0.25	0.032	0.190	0.116±0.076	11
Octanedioic (suberic)	3.0±1.1	bdl	0.338	0.175±0.098	17

Nonanedioic (azelaic)	44±21	1.08	1.09	7.0±4.0	6.2
Decanedioic (sebacic)	0.111±0.031	bdl	bld	0.010±0.010	12
Hexanedecanoic (thapsic)	0.052±0.022	bdl	0.006	0.023±0.031	2.2
Resin acids					
Dehydroabietic	3.0±1.4	1.98	0.757	4.6±1.7	0.6
Isopimaric	0.14±0.31	0.130	0.057	0.12±0.16	1.1
Abietic	0.119±0.061	0.054	bdl	0.076±0.088	1.6
Podocarpic	0.038±0.027	bdl	bdl	0.001±0.002	1.1
Other acids					
Boric	15±15	2.42	11.8	13±12	1.1
Propionic (lactic)	130±120	bdl	35.9	bdl	-
4-Oxopentanoic (levulinic)	13.2±9.6	5.31	1.26	0.64±0.88	15
2-Hydroxyethanoic (glycolic)	32±42	bdl	6.45	5.3±8.6	6.0
2,3-Dihydroxypropanoic (glyceric)	9.4±7.0	bdl	1.41	1.0±1.1	9.7
3-Hydroxybutanoic (3-hydroxybutyric)	0.23±0.35	bdl	0.152	0.049±0.079	4.7
3,4-Dihydroxybutanoic	3.1±2.0	bdl	0.691	0.98±0.61	3.2
Nicotinic	8.6±5.3	4.67	6.84	bdl	-
cis-Pinonic	1.03±0.87	bdl	1.05	0.53±0.45	1.9
Pinic	1.10±0.80	0.933	0.424	1.5±1.4	0.7
2-Hydroxypropane-1,2,3-tricarboxylic (citric)	1.0±1.1	bdl	0.912	0.30±0.28	3.4
Glycerol derivatives					
Glycerol	80±35	17.7	57.2	32±30	2.5
Diethylene glycol	12.2±7.0	4.22	10.8	6.8±2.7	1.8
1-Monolauroyl-rac-glycerol	0.021±0.014	bdl	bdl	0.001±0.000	23

1-Glycerol monostearate (1-monostearin)	170±130	1.29	7.52	4.0±3.3	42
2-Glycerol monostearate (2-monostearin)	16±15	bdl	bdl	0.030±0.042	550
Heptadecanoic acid glycerin-(1)-monoester	6.0±4.9	bdl	bdl	bdl	-
1-Monolinoleoylglycerol	23±18	bdl	bdl	bdl	-
1-Monopalmitin	190±110	0.519	0.902	4.3±2.7	45
1-Monomyristin	33±22	0.776	0.616	1.350±0.066	25
Other compounds					
Myrtenol	0.005±0.005	bdl	bdl	-	-
Hydroquinone	0.142±0.022	0.112	bdl	0.15±0.12	0.9
Hydroxyacetophenone	0.19±0.17	0.382	bdl	1.40±0.91	0.1
2,6-Di-tert-butyl-1,4-benzoquinone	0.44±0.40	bdl	0.088	0.36±0.21	1.2
Methyl dihydrojasmonate	46±29	55.1	55.9	-	-
2-Propanol-1-chloro-phosphate (TCPP)	1.6±1.2	bdl	4.24	-	-
Ethylene brassylate	18.4±6.2	6.77	6.24	-	-
5-oxo-L-proline	19±14	bdl	1.24	-	-
Caffeine	-	6.31	9.85	1.4±3.0	-
Parsol MCX	12.6±6.6	2.93	11.2	0.10±0.14	122
Acetyl tributyl citrate	4.0±1.6	1.78	2.82	-	-
Diisooctyl adipate	10.0±3.5	2.55	5.40	2.7±5.9	3.8
Dibutyl adipate	14.2±7.4	6.33	4.82	0.6±1.3	23
Δ-1-Tetrahydrocannabinol	4.8±9.5	bdl	bdl	-	-
Piperonyl butoxide	9.3±3.7	4.25	6.68	-	-
Oxidised Irgafos 168	7±10	bdl	bdl	8±18	1.0
Plasticisers					

Dimethyl phthalate	bld	0.002	bdl	bdl	-
Diethyl phthalate	0.064±0.092	bdl	bdl	0.24±0.27	0.26
Di-n-butyl phthalate	0.23±0.21	0.059	bdl	2.1±2.6	0.11
Benzyl butyl phthalate	0.004±0.008	bdl	bdl	0.05±0.10	0.08
Bis(2-ethylhexyl) adipate	0.006±0.009	bdl	bdl	0.09±0.19	0.06
Bis(2-ethylhexyl) phthalate	0.17±0.23	bdl	bdl	2.1±4.4	0.08
Unidentified phthalate	280±160	121	163	19±25	15

bld – below detection limit
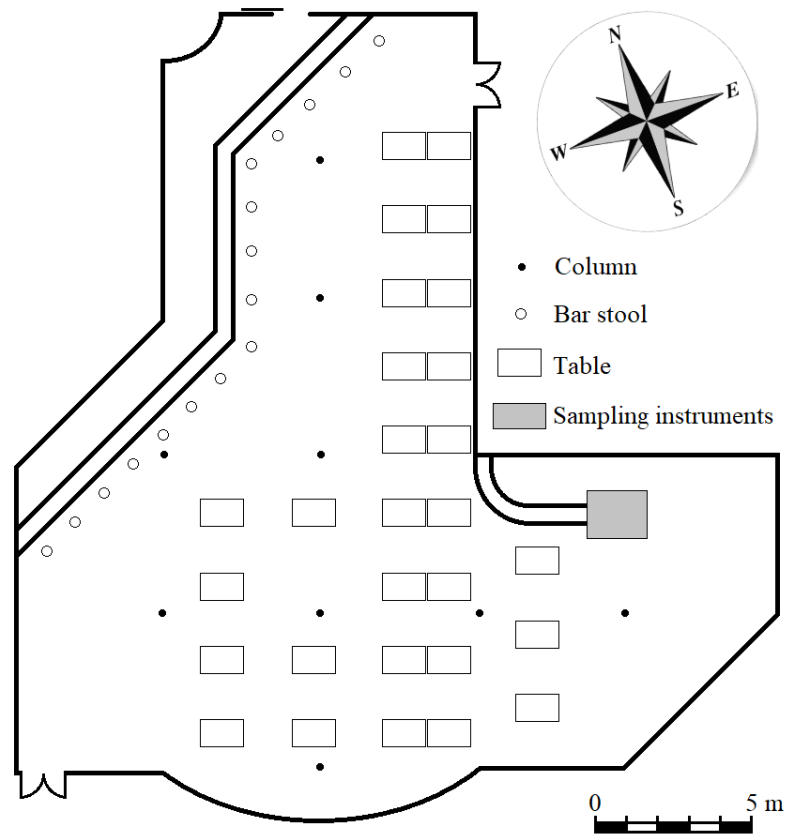
Fig 1. Floor plan of the cafeteria. Each table has four chairs.
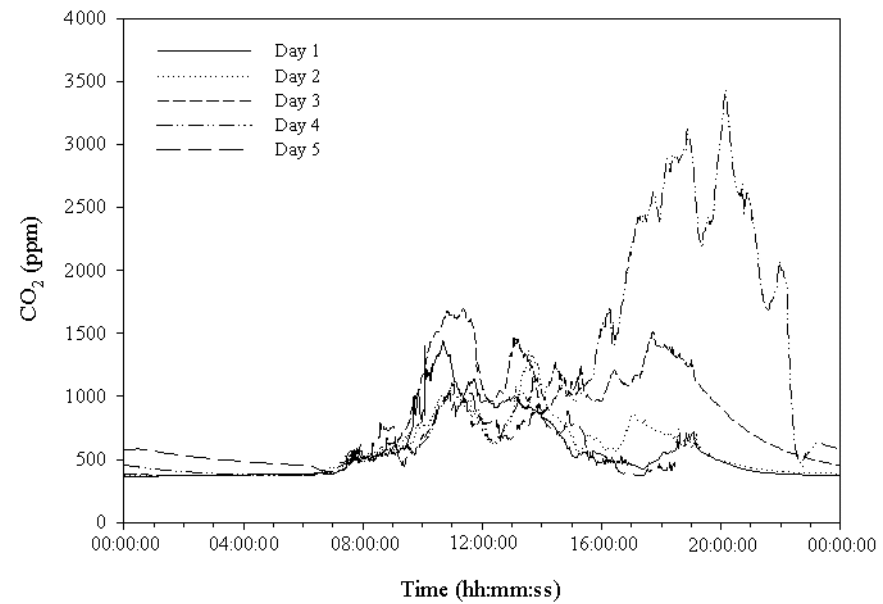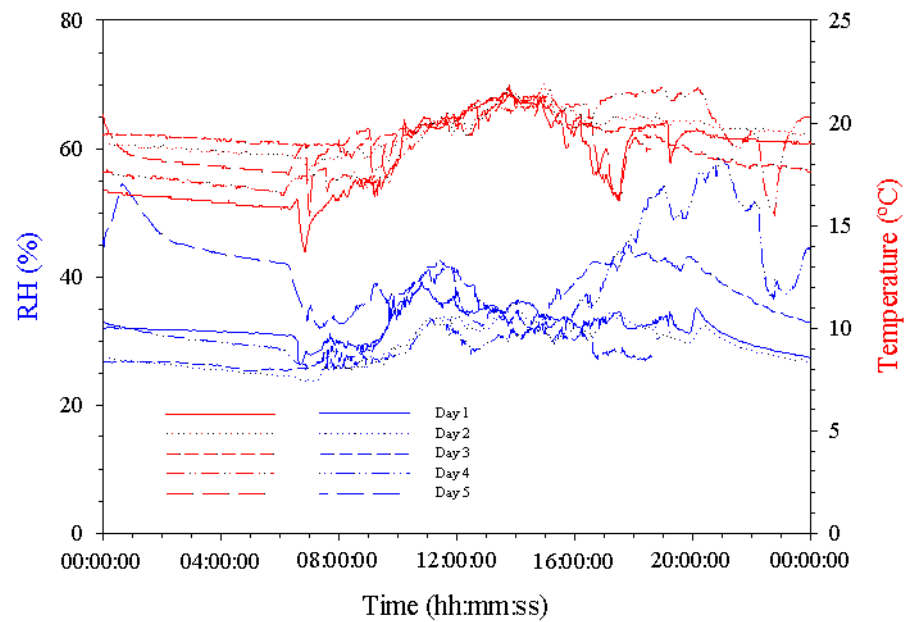
Fig 2. Variation of relative humidity, temperature and carbon dioxide levels on weekdays.
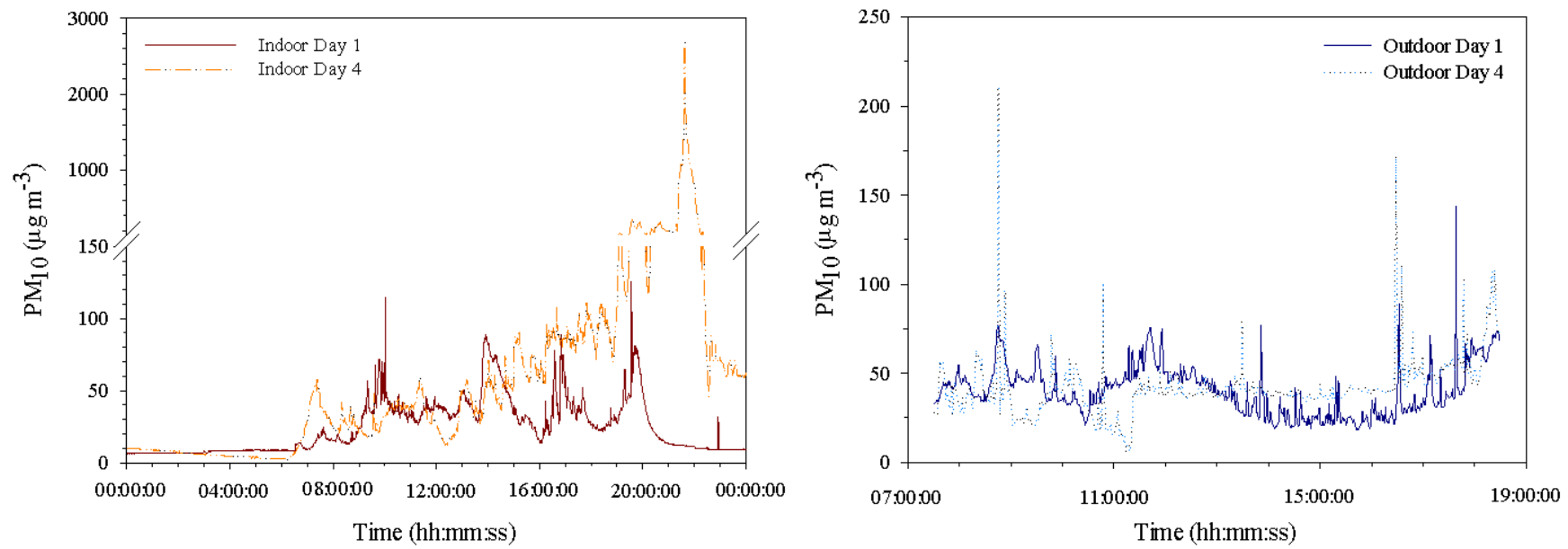
Fig. 3. Examples of PM$_{10}$ daily concentration profiles in the cafeteria and outdoors (Day 1 – Monday, Day 4 – Thursday).
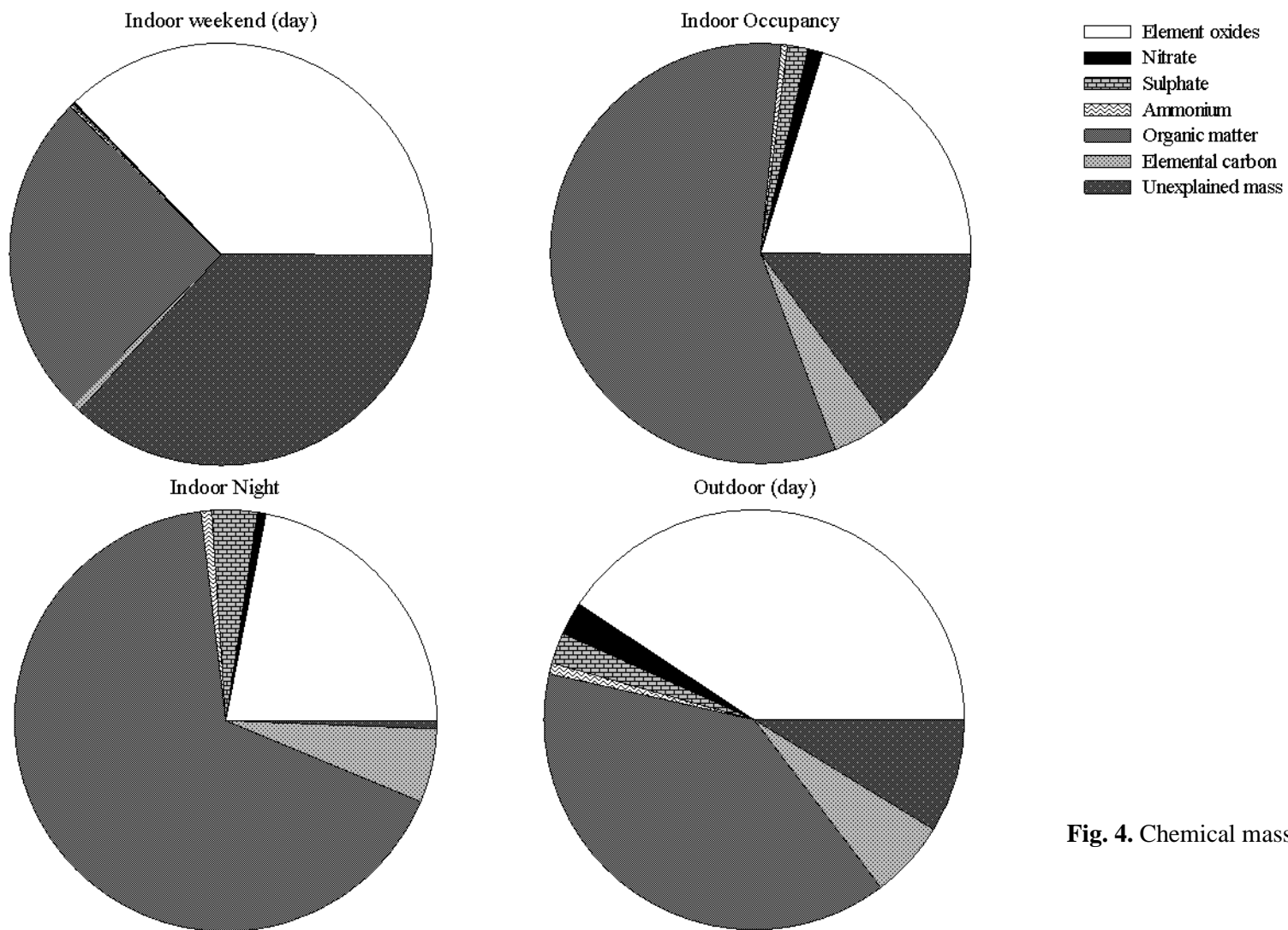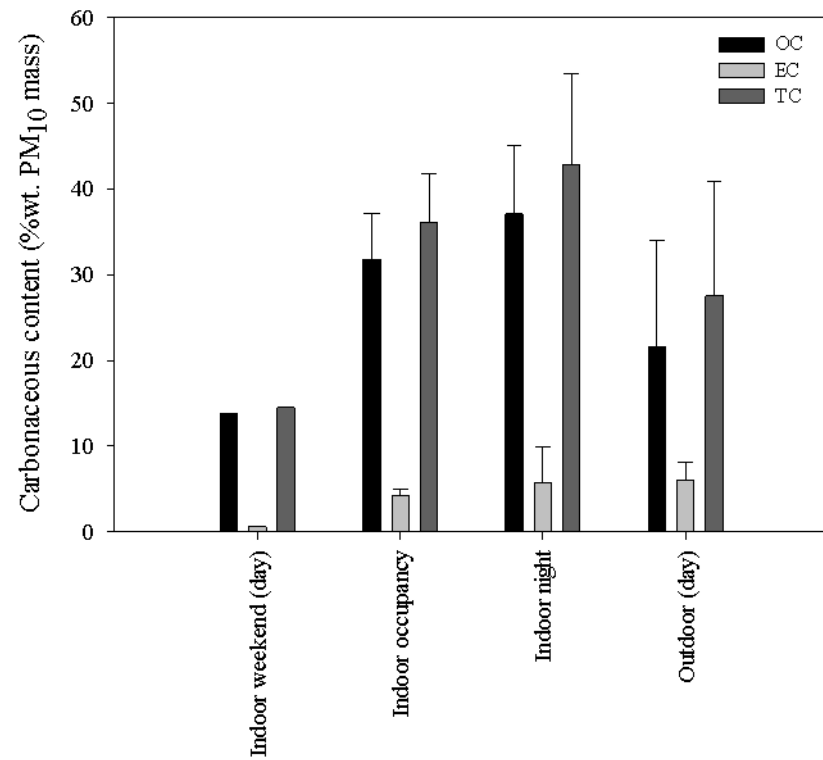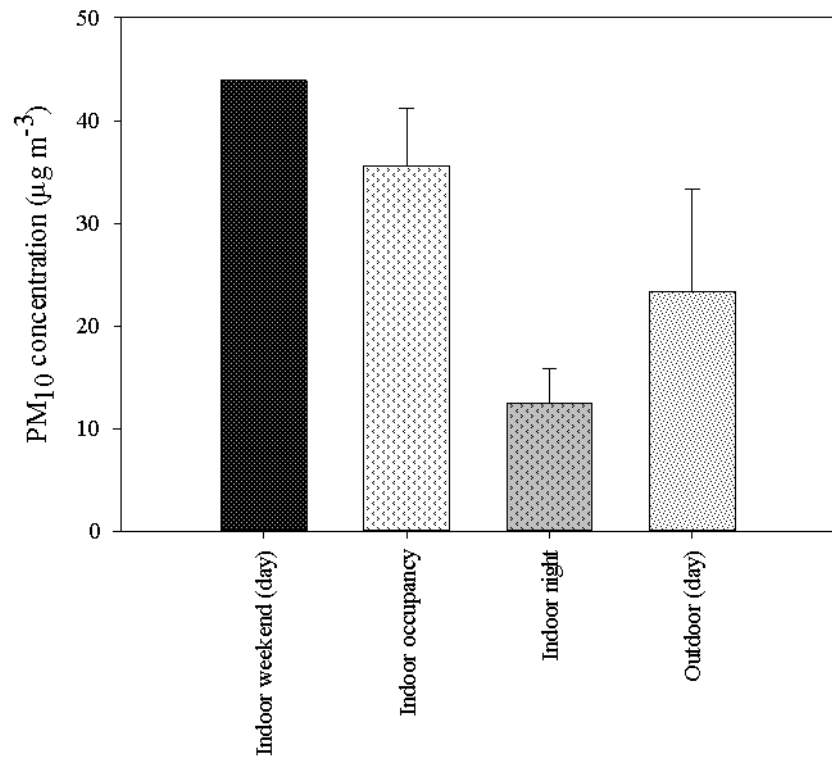
Fig. 4. Chemical mass closure of PM$_{10}$.

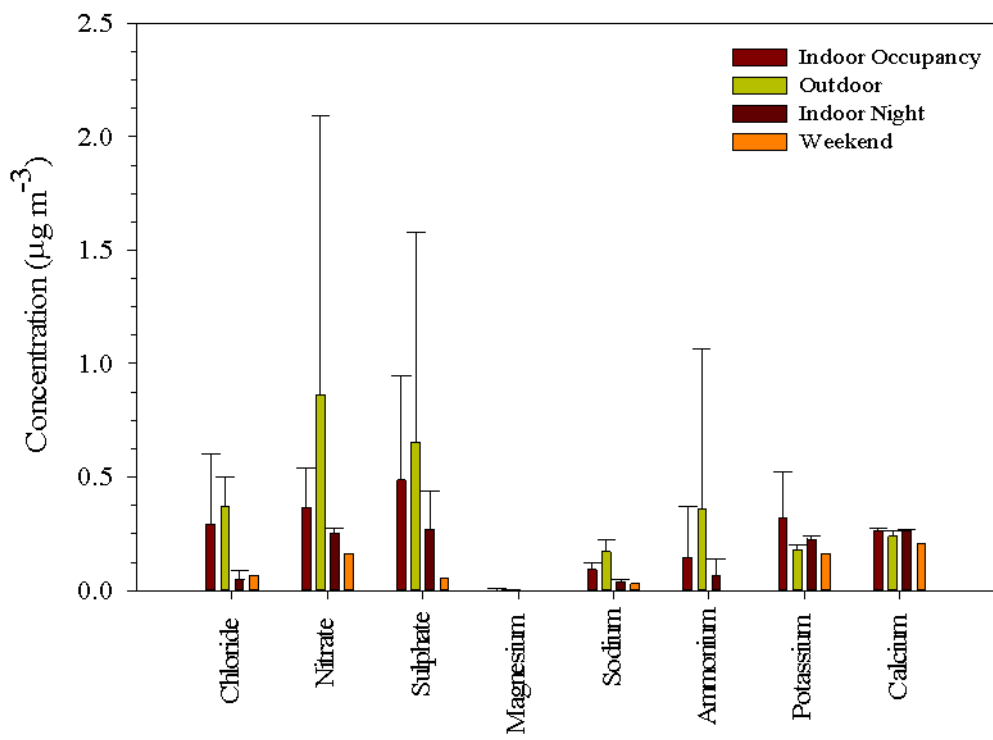**Fig 5**. Means (±SD) of PM_{10} concentrations and mass fractions of organic carbon, elemental carbon and total carbon (TC = OC + EC)

Fig. 6. Concentrations (mean±SD) of water-soluble ions in PM$_{10}$ collected indoors and outdoors.
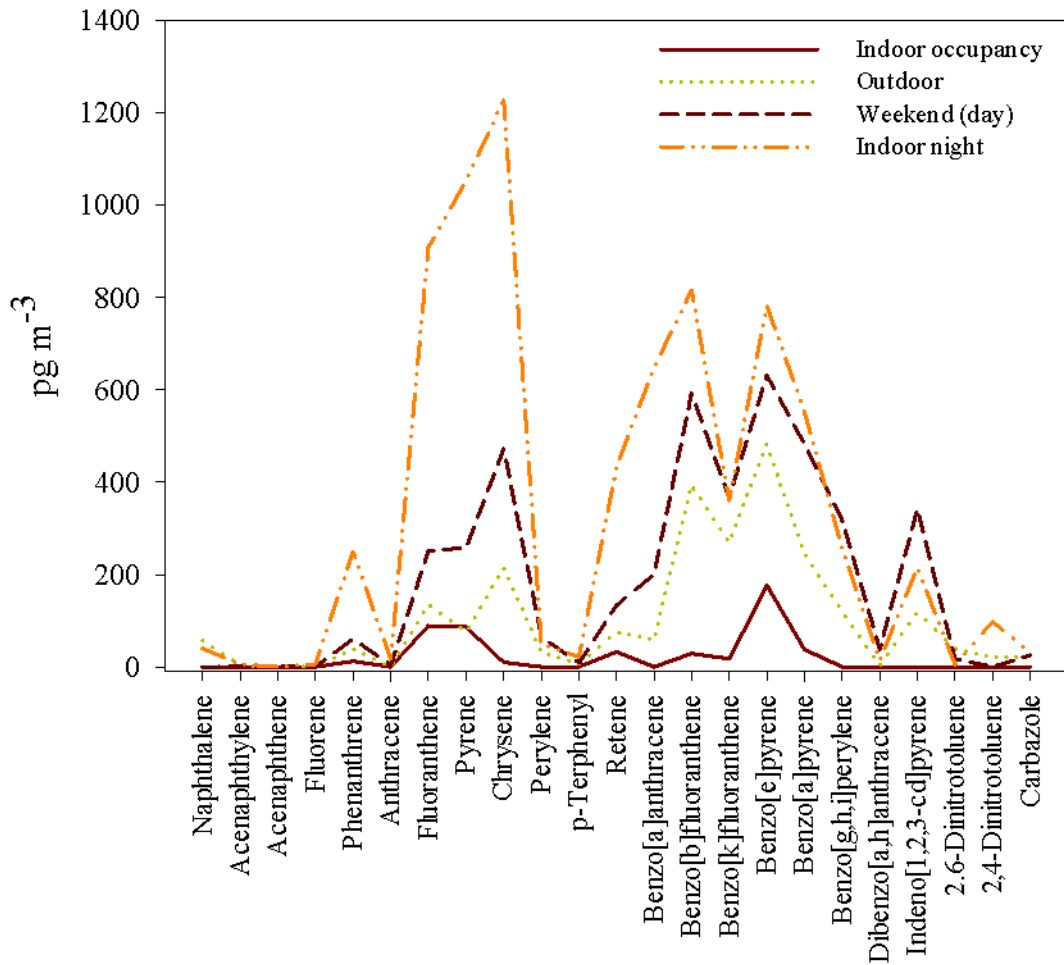
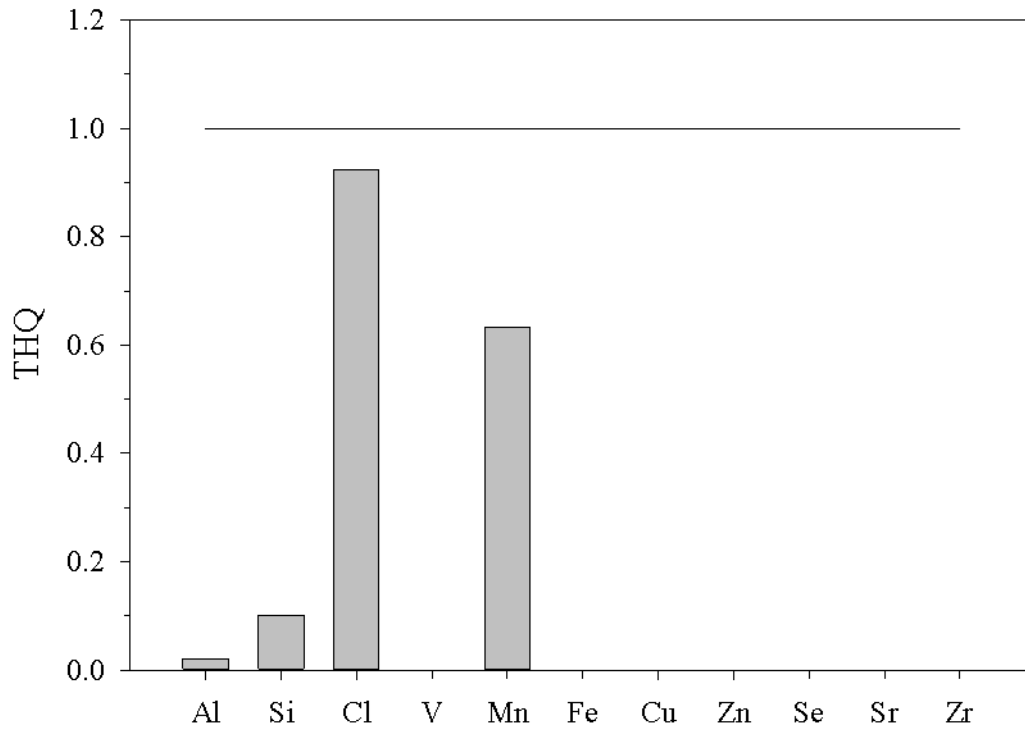**Fig. 7**. Average concentrations of polycyclic aromatic hydrocarbons.

Fig. 8. Mean risks from inhalation exposure to noncarcinogenic PM_{10}-bound metals for the cafeteria workers during the occupancy period. The horizontal line represents the USEPA health-based guideline level.