

Indoor air pollution of polycyclic aromatic hydrocarbons emitted by computers

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ABSTRACT

We investigated the polycyclic aromatic hydrocarbons (PAHs) emitted from computers to evaluate the characteristics of indoor air pollution and the potential levels of human exposure. Gaseous naphthalene and phenanthrene were the dominant compounds in the indoor environment. The levels of Σ_{16} PAHs in an office, a computer room, and a server room were 2–25 times higher than those in outdoor air or in a room without a computer. The PAH profiles inside the computers were similar to those in the indoor air, suggesting that the PAHs released from computers contribute to indoor air pollution. The PAH emissions from computers had a positive relationship with the age of the computer, the operating time, the internal temperature, and the size of the computer. Moreover, this study is the first to identify that computers are an indoor PAH source by confirming PAH emissions from computer components in an enclosed space. Adults are potentially exposed via inhalation to PAHs at a concentration of 1.9 ng TEQ/day in their office. The non-carcinogenic and carcinogenic risks of PAHs inhaled in an office were lower than the risk threshold set by the US EPA. However, in addition to PAHs, other volatile compounds that may be emitted from computers need to be considered.

1. Introduction

In the current high-tech era, the computer has become indispensable for various uses at home and at work. Since the emergence of the internet in the 1990s, the number of household computers has increased steadily, with the Organization for Economic Co-operation and Development (OECD) reporting that 75–97% of households in the OECD had a computer in 2018 [1], while the internet access rate for OECD households was 87–99% [2]. This increasing reliance on computers has been strengthened further by the recent global COVID-19 pandemic, with a significant proportion of academic, work, and administrative processes shifting online, resulting in a sharp increase in daily computer usage.

A computer is composed of various materials, including metal, silicon, and plastic. Previous studies have reported that harmful substances (e.g., ozone, volatile organic compounds [VOCs], and brominated flame retardants) can be emitted from the electronic components of a computer [3,4]. In particular, when a computer is used for a long time or in an enclosed space, the pollutants emitted from the computer may affect the indoor air quality. In specific places where computers are used to study and work (e.g., offices, classrooms, and server rooms), the indoor

air pollution may be more serious because the computer usage time is longer than at home and more computers are in use [5–7].

Because the time spent indoors has increased in recent decades, health problems arising from poor indoor air quality have received more attention. Americans and Koreans are reported to stay indoors for 95% [8] and 97% [9] of the day, respectively. The World Health Organization reported that 2.8 million people die annually due to indoor air pollution and that exposure rates to indoor air pollutants are 1000 times higher than those to ambient air pollutants [10]. In addition, the United States Environmental Protection Agency (US EPA) reported that indoor air pollution is one of the five most serious environmental problems [11]. As such, various studies related to indoor air pollution have been conducted; however, research on indoor air pollution caused by electronic products such as computers has been limited.

Polycyclic aromatic hydrocarbons (PAHs) are organic pollutants containing two or more aromatic rings. They are mainly generated by the incomplete combustion of organic substances from the burning of biomass, exhaust from vehicles, and industrial activities and are primarily discharged into the atmosphere [12,13]. Sixteen PAHs have been listed as priority pollutants by the US EPA [14], and the PAH content in

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diesel fuel, plastics, and rubber materials is regulated in the European Union (EU) [12,15]. Also, National Institute for Occupational Safety and Health and Occupational Safety and Health Administration have established recommended an exposure limit (0.1 mg/m^3) and a permissible exposure level (0.2 mg/m^3) for PAH in the workplace, respectively. PAHs can accumulate in body tissues such as the ovaries, spleen, kidneys, liver, and fat [16], and long-term exposure to PAHs can cause adverse health effects, with many previous studies reporting a correlation between PAH exposure and carcinogenesis [17–19]. PAHs are also known to cause cardiovascular disease and developmental disturbances of the fetus [20]. Despite the harmful effects of PAHs, most previous studies on indoor air quality have been conducted for the criteria air pollutants listed in the EPA guidelines for indoor air quality (e.g., PM_{10} , $\text{PM}_{2.5}$, NO_2 , and CO), VOCs, and formaldehyde (<http://www.epa.gov/indoor-air-quality-iaq>), and research on indoor PAH pollution has been insufficient.

Previous studies have reported PAH levels in indoor air and dust samples collected from home [21], comparison results for PAHs in indoor and outdoor air [22], PAHs in the indoor air of home and public places (e.g., schools, shopping centers, and subways) [20,21,23], and indoor air pollution by PAHs emitted from tobacco smoke [24], cooking [25], heating systems [26], and decorative candles [27]. In addition, the PAHs in the indoor air of internet cafés with a relatively large number of computers in a single room has been investigated [5]; however, the cause of PAH pollution in the internet café was proposed to be smoking, and the PAH emissions from computers were not considered. Furthermore, PAH levels in dust samples collected inside computers have been reported [28,29], but potential factors affecting the changes in level and composition of PAHs in the dust samples over time (e.g., the age of computer, characteristics of the area in which the computer is located, and PAH emissions from the electronic components of the computer) were not investigated. Given that gaseous PAHs can also be released, previous research has been insufficient to understand the influence of PAHs emitted from computers.

In several studies, high PAH levels have been detected in the air, soil, and agricultural products near electronic waste recycling sites because PAHs are produced and released from electronics via pyrolysis at $200\text{--}300 \text{ }^\circ\text{C}$ [30–32]. PAHs were detected in plastics and rubber, which are major components of computers [15,33], and carbon black and extender oils in consumer products unintentionally contain PAHs [34, 35]. Carbon black is a reinforcing filler in pigment in plastics or rubber, and extender oils are used as softeners and plasticizer oils [36]. The levels of carcinogenic PAHs in toys, childcare products, and other consumer goods are regulated by Registration, Evaluation, Authorization, and Restriction of Chemicals (REACH) [37]. In addition, the emission of fine particles and carcinogenic PAHs from printers and copiers used in an office have been reported [38], and in our previous study that investigated PAHs in a library, the PAH levels in the indoor air of a server room were significantly higher than those in other places [7], suggesting that PAHs are released from computers. However, few studies have evaluated PAH emissions, indoor air pollution, and human exposure risks resulting from the use of electronic products in daily life.

In the present study, the levels and composition of PAHs in the indoor air of an office, a computer room, and a server room were monitored, and gaseous and particulate PAHs inside the computers were analyzed. The computer components were also heated in an oven, and the subsequent release of PAHs was monitored. Finally, the risk to human health of inhaling PAHs in the office was evaluated. To the best of our knowledge, this is the first study demonstrating that computers are an indoor air pollution source of PAHs by confirming PAH emissions from computer components in an enclosed space.

2. Materials and methods

2.1. Sample collection

Sampling was conducted in a university office, a computer room, and a computer server room, where the density and usage of computers were relatively high. Key information was collected from these areas, including computer operating time per day, age of the computer, characteristics of the sampling location, and the number of computers per room. Images of the sampling methods employed and information on the sampling location are presented in Fig. S1 and Table S1, respectively, in the Supplementary Information.

A high-volume air sampler (HiVol, HV-700F, Sibata, Japan) was used to collect gaseous and particulate PAHs indoors (Fig. S1a). The sampler operating time was 24 h, and the airflow rate was adjusted based on the size of the sampling space: $144 \text{ m}^3/\text{day}$ in the office, $216 \text{ m}^3/\text{day}$ in the computer room, and $126.72 \text{ m}^3/\text{day}$ in the server room. Polyurethane foam (PUF) plugs (5.0 cm diameter \times 4.5 cm thick, Ziemer Chromatographie, Germany) and glass fiber filters (GFFs, $25.4 \text{ cm} \times 20.3 \text{ cm}$, Advantec, Japan) were used for gas and total suspended particle sampling, respectively. Before sampling, the PUF plugs were sonicated with n-hexane and acetone for 30 min, and the GFFs were heated at $400 \text{ }^\circ\text{C}$ for 4 h. After sampling, they were wrapped in aluminum foil and stored in polyethylene zipper bags at $-4 \text{ }^\circ\text{C}$.

To collect gaseous and particulate PAHs inside the computers, two PUF disks (14 cm in diameter \times 1.35 cm thick, Ziemer Chromatographie, Germany) were placed in each computer for 50 days as passive air samplers for gaseous compounds [39]. In addition, GFFs were used to collect dust at the bottom of the computer as dry deposition samplers (Fig. S1b). In the previous studies, PUF disks and GFFs were placed in passive air samplers to collect gaseous and particulate samples, respectively [40,41]. The passive air sampler has two stainless steel domes, and their role is to protect the passive sampling media (PUF disks and GFFs) from rainfall and wind. Therefore, in this study, only the sampling media were placed inside the computers. The dust inside a computer enters from the indoor air via the cooling fan. Samples were collected from 10 computers in the office (S1–S5), computer room (S6–S9), and server room (S10).

In addition, one computer was disassembled, and 3 g samples of each of the transistor, polyvinyl chloride (PVC) cable, and printed circuit board were taken to measure the PAHs emitted from the electronic components by heating. The components were individually placed in a pre-cleaned brown glass bottle with a PUF plug (Fig. S2). After sealing the bottle cap with Teflon tape, the bottles were placed in an oven at $60 \text{ }^\circ\text{C}$, which is the approximate temperature of the central processing unit (CPU) in a computer when in use. The sampling periods for each type of computer components were 1, 3, 5, and 6 days to examine the change in the PAH levels collected in the PUF plug; a total of 12 samples were prepared. To minimize the volatilization of PAHs from the PUF plug when the bottle cap was opened, the bottle was cooled at room temperature. The samples were then stored in polyethylene zipper bags at $-4 \text{ }^\circ\text{C}$.

2.2. QA/QC and instrumental analysis

The 16 priority PAHs prescribed by the US EPA were analyzed; a list of the target compounds and their abbreviations is provided in Table S2 in the Supplementary Information. The PAHs were Soxhlet extracted from the PUFs and GFFs with 350 mL of n-hexane/acetone (9:1 v/v) [42]. Before extraction, surrogate standards (naphthalene- d_8 , acenaphthene- d_{10} , phenanthrene- d_{10} , chrysene- d_{12} , and perylene- d_{12}) were injected into the samples. After extraction, the samples were concentrated using a Turbo Vap II (Caliper Life Sciences, MA, USA). Silica gel columns consisting of 2 g of Na_2SO_4 , 2 g of alumina, 5 g of activated silica gel, and 2 g of Na_2SO_4 were used to clean up the extracts. The samples were eluted with 70 mL of n-hexane/dichloromethane (9:1

v/v). The final solution was evaporated until 0.5 mL using a nitrogen gas evaporator (MG-2200, Eyela, Tokyo, Japan) and transferred into 1.5 mL vials, and the internal standard (p-terphenyl-d₁₄) was spiked before analysis. The final solution (1 µL) was injected into a gas chromatograph/mass spectrometer (GC 7890/MS 5975C, Agilent, USA) with a DB-5MS column (30 m × 0.25 mm ID × 0.25 µm film thickness, Agilent J&W Scientific, CA, USA). The temperature program of the GC was as follows: the temperature was increased from 60 °C to 320 °C at a rate of 10 °C/min and then maintained for 5 min. Helium was used as the carrier gas at a flow rate of 1.0 mL/min.

Method blanks were analyzed for every sample batch to check for potential contamination. The mean PAH levels in the blank samples ranged from not detected to 0.03 ng/m³, and all PAH levels were adjusted accordingly. The recovery rates ranged from 64 to 115% (mean: 75%), and the coefficients of determination (R²) for the calibration curves for the individual PAHs were above 0.99. For accuracy testing, representative samples were analyzed repeatedly, and the relative standard deviation of the detected target compounds was less than 30%. Seven replicates of spiked blank samples were used to measure the method detection limits, which were calculated by multiplying the Student's t-value (3.14) for a 99% confidence level by the standard deviation of the seven replicates (Table S2). SPSS Statistics 20 (IBM Corp., USA) was used for statistical analysis of the data. The three most volatile compounds (i.e., Nap, Acy, and Ace) in the atmosphere are not frequently reported due to potential contamination, low recovery rates, and sampling artifacts (e.g., breakthrough) using a HiVol [43]. Nonetheless, we reported the concentration of these three compounds because Nap is a dominant compound both in indoor air and in computers.

2.3. Health risk assessment

PAHs can be divided into non-carcinogens and carcinogens [14]. In the present study, the hazard quotient (HQ) for the non-carcinogen Nap and the lifetime cancer risk for the carcinogens, BaA, Chr, BbF, BkF, BaP, IcdP, and DahA, were calculated. Of the non-carcinogenic compounds, Nap is detected at relatively high concentrations in the atmosphere and is considered a representative low-molecular-weight (LMW) PAH. Therefore, Nap was selected as a target compound for non-carcinogenic risk assessment in the present study [44]. BaP also has an inhalation reference concentration (RfC) for effects other than cancer [45], but its HQ could not be calculated because BaP was not detected from the indoor air samples. Human exposure to PAHs through inhalation was assessed using the US EPA method [45], and toxicity equivalency quantities (TEQs) based on the toxicity equivalency factors (TEFs) listed in Table S2 [46] were used for the carcinogenic risk assessment [47]. As a conservative approach, all PAHs inhaled were assumed to be absorbed through lungs.

The exposure concentration (EC, µg/m³) for PAHs was calculated using Equation (1) under the assumption that all of the gaseous and particulate PAHs enter the lungs through inhalation and are absorbed by the body [48,49]. CA (mg/m³) is the ambient air concentration, which is the total (i.e., gas + particle) mean concentration of each PAH in the office. ET (8 h/day) is the exposure time, which is the average indoor dwelling time for graduate students each day. EF (247 days/year) is the exposure frequency for graduate students, which was calculated based on five working days per week, excluding holidays. ED (40 years) is the exposure duration for office workers. The averaging time (AT) for non-carcinogenic risk is ED × 365 days/year, and the AT for carcinogenic risk is the average lifespan of Koreans (82.7 years × 365 days/year). The AT was referenced from the Korean Exposure Factor Handbook [50], and the ET, EF, and ED were collected using a questionnaire. The HQ for Nap was calculated using Equation (2). If HQ is less than 1, the risk is acceptable. The reference concentration for inhalation exposure (RfC) for Nap is 0.003 µg/m³ [45].

$$EC = \frac{CA \times ET \times EF \times ED}{AT} \quad (1)$$

$$HQ = \frac{EC}{RfC \times 1000 \mu\text{g}/\text{mg}} \quad (2)$$

The carcinogenic Risk for BaP was calculated using Equation (3). The inhalation unit risk (IUR) for BaP is 6.0 × 10⁻⁴ per µg/m³ [45].

$$Risk = IUR \times EC \quad (3)$$

3. Results and discussion

3.1. PAHs in indoor air

The Σ₁₆ PAH levels in the indoor air (n = 2) collected in the office (34.0 ng/m³), computer room (39.0 ng/m³), and server room (50.4 ng/m³) are presented in Fig. 1a, while the concentrations of the individual compounds from the HiVol samples are listed in Table S3. Carcinogenic PAHs (Σ_{7carc} PAHs: BaA, Chr, BbF, BkF, BaP, IcdP, and DahA) in the office (1.9 ng/m³), computer room (1.0 ng/m³), and server room (1.7 ng/m³) accounted for 3–5% of the concentration of Σ₁₆ PAHs, and the concentration of particulate PAHs and the proportion of high-molecular-weight (HMW) PAHs in the office were higher than those in the other rooms. LMW PAHs with two or three rings, such as Nap, Phe, Flu, and Ace, were detected at the highest concentrations, whereas HMW PAHs with four to six rings were detected at low concentrations (Fig. 1b). In particular, Nap, which has the highest volatility (octanol–air partition coefficient [log K_{OA}] at 25 °C: 5.13) [51], was the PAH with the highest concentration.

The Σ₁₃ PAH levels in outdoor air collected on the UNIST campus in Ulsan [52] were compared to the indoor Σ₁₃ PAH levels (except Nap, Acy, and Ace). The indoor-to-outdoor (I/O) ratio of the Σ₁₃ PAHs was 3.2, which suggests that the indoor PAH concentrations were strongly affected by indoor sources. A previous study on PAHs in indoor air at home also reported higher levels of Σ₁₂ PAHs than in outdoor air, but the I/O ratio was only 1.2–1.4 due to the presence of strong ventilation [22]. In the present study, the samples were collected from enclosed spaces with poor ventilation, while the samples in the previous study were collected from spaces in which the windows were kept open during the summer and fall. In other enclosed places such as supermarkets and shopping centers, a mean I/O Σ₁₅ PAH ratio of 2.35 has been reported [23]. Therefore, the I/O ratio can be much higher in an enclosed space with a poor ventilation system and many electronic devices, such as computers.

The TEQ of the Σ₁₆ PAHs in the office, computer room, and server room was 0.40, 0.26, and 0.52 ng TEQ/m³, respectively, and the TEQ of Σ_{7carc} PAHs in the office (0.34 ng TEQ/m³), computer room (0.20 ng TEQ/m³), and server room (0.46 ng TEQ/m³) is similar to those reported for carcinogenic PAHs in outdoor air in previous studies. For example, the range of the total (gas + particle) TEQ of Σ₁₃ PAHs and Σ_{7carc} PAHs at outdoor sites in rural, urban, and industrial areas in Ulsan was 0.34–0.77 ng TEQ/m³ and 0.31–0.72 ng TEQ/m³, respectively [53]. In addition, the Korea Environment Corporation (KECO) analyzed carcinogenic PAHs collected at 31 outdoor sites across Korea (<https://www.airkorea.or.kr>) and reported a mean concentration and TEQ for Σ_{7carc} PAHs in Ulsan in 2012 of 2.1 ng/m³ and 0.50 ng TEQ/m³, respectively. This similarity in the carcinogenic PAH concentration in outdoor and indoor air suggests that most of the carcinogenic PAHs present indoors may be introduced from outside.

The levels of gaseous and particulate Σ₁₆ PAHs were 29.3–46.9 ng/m³ and 2.48–4.34 ng/m³, respectively, accounting for 87–94% and 6–13% of the total concentration. The concentration of gaseous PAHs was thus 11 times higher than that of particulate PAHs, a result that differs from previous studies. The gaseous PAH levels in smoking-affected (49.6 ng/m³) and non-smoking (26.9 ng/m³) homes (Porto,

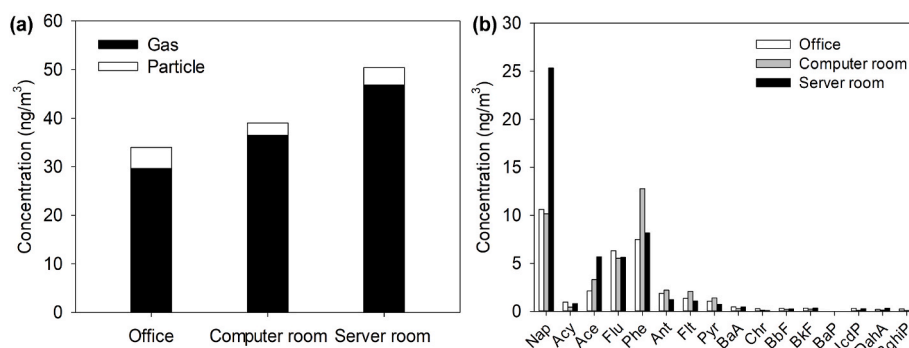


Fig. 1. Concentrations and profiles of PAHs in the indoor air collected from an office ($n = 2$), a computer room ($n = 2$), and a server room ($n = 2$): (a) concentrations of gaseous and particulate Σ_{16} PAHs and (b) profiles of 16 PAHs.

Portugal) were reported to be 1.5 and 1.8 times higher than particulate levels (smoking home: 33.7 ng/m³; non-smoking home: 14.8 ng/m³), respectively [54]. In addition, the gas/particle (G/P) partitioning ratio of 22 PAHs in a subway station (gas: 59.4 ng/m³; particle: 46.9 ng/m³) and exhibition facility (gas: 46.1 ng/m³; particle: 40.6 ng/m³) in six cities (Seoul, Daejeon, Gwangju, Suncheon, Gwangyang, and Yeosu) in Korea was 1.3 and 1.1, respectively, and the gaseous PAH concentration was higher than the particulate concentration [55]. In addition, the G/P partitioning ratio in internet cafés (gas: 102 ng/m³; particle: 108 ng/m³), terminal waiting rooms (gas: 39.2 ng/m³; particle: 54.2 ng/m³), indoor parking lots (gas: 44.3 ng/m³; particle: 84.2 ng/m³), supermarkets (gas: 44.2 ng/m³; particle: 47.4 ng/m³), child care facilities (gas: 47.9 ng/m³; particle: 51.3 ng/m³), and funeral halls (gas: 27.2 ng/m³; particle: 57.8 ng/m³) was 0.9, 0.7, 0.5, 0.9, 0.9, and 0.5 [55], respectively. Similarly, the G/P partitioning ratio for 17 PAHs (gas: 8.89–14.2 ng/m³; particle: 8.63–11.8 ng/m³) in outdoor air in residential areas in Seoul, Korea was 1.0–1.2 [56], while the G/P partitioning ratio of 16 PAHs (gas: 11.5–170 ng/m³; particle: 3.04–23.1 ng/m³) in industrial areas in Pohang, Korea was 2.0–7.4, which was higher than those in other regions [57].

The difference between the gaseous and particulate PAH levels in the present study was thus much larger than that reported in other indoor and outdoor studies. The reason for the much higher levels of gaseous PAHs is the presence of two- and three-ring PAHs, such as Nap and Phe, with a relatively low K_{OA} [51] and a high volatility, suggesting that the emission source for gaseous PAHs may be located in the office, computer room, and server room. In other studies, Ace and Nap were reported to be the dominant compounds in the indoor air of public places [55], and Phe, Flu, and Pyr were dominant in the outdoor air of residential and industrial complexes [56,57]. This difference in PAH composition may be due to differences in the emission sources; in particular, the indoor air in the present study is likely to have been strongly affected by an emission source that releases high levels of Nap.

The PAH levels at each sampling location were compared to assess the distribution of PAHs based on the characteristics of the surrounding space. The PAH levels were highest in the server room, followed in order by the computer room and office. The size of the office, computer room, and server room was 165.0 m³, 250.3 m³, and 83.3 m³, respectively. In addition, the office and server room had no windows and, while air conditioners were installed in each room, the ventilation was not effective. The number of computers used in the office and computer room was 5 and 40, respectively, while there were two server rack cabinets with 14 nodes containing 336 CPUs in the server room. The mean computer operating time was 21.4 h/day, 18.7 h/day, and 24 h/day in the office, computer room, and server room, respectively. Due to their large power consumption and high heat output associated with all-day use, servers may emit higher levels of PAHs than regular desktop computers. Thus, the highest PAH levels in the server room were due to the small room size, the large number of computing devices, the poor

ventilation, and the long computer operating time. In our previous study, a server room in a library had the highest concentration of Σ_{16} PAHs (94.7 ng/m³) [7]. In the present study, the computer room (40 computers) had a higher concentration of PAHs than did the office (5 computers), suggesting that the number of computers was positively correlated with the PAH levels. The PAH levels in the present study were 1.7–2.6 times higher than those in a large monograph and multimedia collection room with few computers [7], suggesting that computers are an important indoor source of PAHs, particularly in small, enclosed indoor environments.

The concentration of Σ_{16} PAHs in the office was lower than that in the computer room and the server room. However, the Σ_{15} PAH levels (mean except Nap: 23 ng/m³; mean except Ace: 33 ng/m³) in the office were higher than the mean Σ_{15} PAH levels except Nap in indoor air at home in Kuwait (6.5 ng/m³) [21] and the mean Σ_{15} PAH levels except Ace in public places in Hangzhou, China [23] characterized by the presence of many people and a relatively enclosed area, including train stations (waiting room: 2.49 ng/m³), shopping centers (electronic apparatus area, clothes area, and resting area: 21 ng/m³), supermarkets (food processing area and electronic apparatus area: 15.5 ng/m³), and a hotel (dining hall and kitchen: 17.3 ng/m³). The indoor PAH levels in the present study were thus higher than those of other indoor spaces, suggesting that small indoor places with many electronic devices suffer more PAH pollution.

3.2. PAHs sampled from inside computers

To confirm the influence of computers on indoor PAH pollution, gaseous and particulate samples from inside the computers in the office (S1–S5), computer room (S6–S9), and server room (S10) were passively collected. The amount of gaseous and particulate PAHs collected from each computer and the PAH fraction by the ring number are presented in Fig. 2. The data for the individual target compounds are also listed in Table S4. The PAHs sampled from an individual computer may not only be derived from the computer itself but could also come from indoor air that has been contaminated by other indoor sources (e.g., other computers and other electronic devices) and the ambient air outside the building. The amount of total Σ_{16} PAHs collected from inside the computers had a range of 4296–8406 ng/computer (mean: 6466 ng/computer). Note that these values do not represent the total amount emitted from the computers themselves but only represent the amount of PAHs collected by the PUF disks and GFFs inside the computers. The PAHs collected by the GFFs accounted for 5–10% of the total amount of Σ_{16} PAHs collected using both the PUFs and GFFs. Because the PAH sampling rate of a PUF disk as a passive air sampler [13] and a GFF as a deposition sampler [58] differ, a direct comparison of the amount of gaseous and particulate PAHs is not possible. Nevertheless, the fraction of Σ_{16} PAHs from the GFFs was similar to the results from the HiVol sampling (6–13%). The mean amount of PAHs collected from inside the

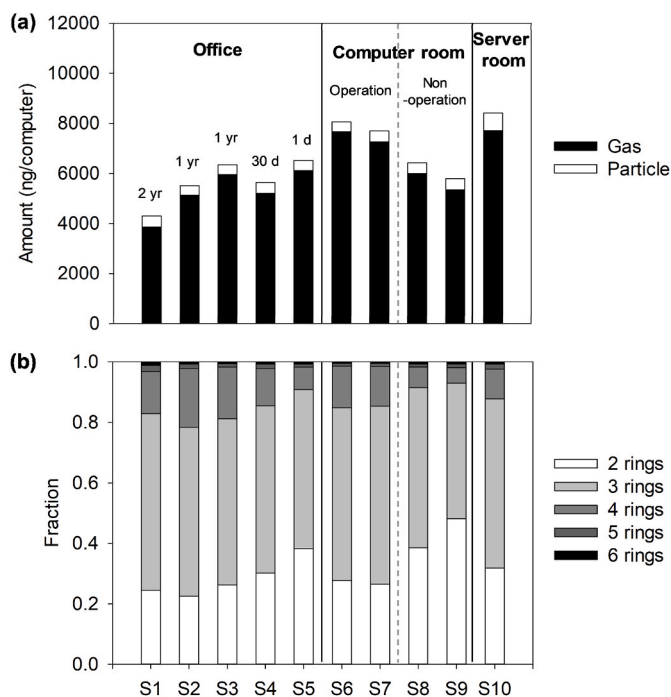


Fig. 2. Amounts of PAHs collected from inside individual computers in the office (S1–S5, $n = 5$), computer room (S6–S9, $n = 4$), and server room (S10, $n = 1$): (a) Amounts of gaseous and particulate Σ_{16} PAHs (ng/computer) and (b) PAH fractions according to the ring number.

computers was highest in the server room (8406 ng/computer), followed by the computer room (6989 ng/computer) and office (5660 ng/computer), which was the same order as the indoor air concentration of PAHs sampled using the HiVol.

The influence of the age of the computers in the office (S1: 2 years; S2: 1 year; S3: 1 year; S4: 30 days; S5: 1 day on the first day of sampling) on the amount of PAHs collected from inside computers was also evaluated. The longer the computer had been used, the lower the amount of PAHs that was collected. As with the HiVol sampling (Fig. 1), PAHs with two or three rings (Nap, Acy, Ace, Flu, Phe, and Ant) were dominant (Fig. 2b), and they were negatively correlated with the computer age. In particular, Nap ($r = -0.975$, $p < 0.01$) and Flu ($r = -0.975$, $p < 0.01$), the most abundant compounds, had a significantly negative correlation with the age of the computer, explaining the overall decrease in the contribution of two- and three-ring PAHs. This result suggests that PAH emissions are relatively high when a computer is new and gradually decrease thereafter.

In the computer room, sampling was conducted on both operating (S6 and S7) and non-operating (S8 and S9) computers (Fig. 2a). The amount of PAHs collected from inside the operating computers (mean: 7870 ng/computer) was larger than that of the non-operating computers (mean: 6109 ng/computer), possibly due to the increase in PAH emissions by the internal heat associated with operating the computer. However, the amount of Nap was higher in the non-operating computers. Nap is the PAH with the highest volatility [59], thus large amounts can be volatilized within non-operating computers even at room temperature. In addition, Nap is likely to have been strongly volatilized by the internal heat in the operating computers before the sampling took place for this study, meaning that there had already been a significant loss of Nap. In a sense, this is similar to the removal of VOCs via the baking out approach as a treatment for sick car syndrome or sick building syndrome [60].

The amounts of gaseous (7706 ng/computer) and particulate (700 ng/computer) PAHs collected from the server computer (S10) were the largest of any of the samples. The reason for this observation is that a

server rack cabinet was larger than the desktop computers, the server computers were run for the entire day in a small, enclosed room, and they generated much more heat than regular desktop computers. The server computer also had a higher fraction of three-ring PAHs than of Nap, probably due to the baking out effect and the strong ventilation of the server rack cabinet. This explanation is supported by the fact that the fraction of Nap was highest in the HiVol sample from the server room.

The profiles of individual PAHs in the gaseous and particulate phases collected from the computers are shown in Fig. 3. Nap and Phe were the dominant compounds in both the gaseous (Nap: 33%; Phe: 31%) and particulate (Nap: 15%; Phe: 13%) samples. In the gaseous phase, two- and three-ring PAHs with a relatively low $\log K_{OA}$ (5.13–7.71 at 25 °C for Nap–Ant) [51,61] and a high volatility were detected at the highest levels, with five- and six-ring PAHs rarely detected. On the other hand, both two- and three-ring PAHs (52%) and five- and six-ring PAHs (29%) were detected at high levels in the particulate samples due to their relatively high $\log K_{OA}$ values (12.1–13.4 at 25 °C for BbF–BghiP) [51], resulting in strong sorption to atmospheric particles. Similar G/P partitioning of PAHs has been observed in previous indoor air monitoring studies [21,54].

3.3. Emission of PAHs from electronic components

The results presented in Sections 3.1 and 3.2 strongly suggest that LMW PAHs are emitted from computers and affect the levels and composition of PAHs in indoor air. To confirm this, more direct evidence of PAH emissions from computers is provided in this section. The amounts and profiles of PAHs released from three types of electronic components (printed circuit board, PVC cable, and transistor) stored with PUF plugs in closed amber bottles and heated at 60 °C for 1, 3, 5, and 6 days are presented in Table S5 and Fig. 4. The amounts of PAHs collected by the PUF plugs for the three electronic components increased with the heating time, indicating that PAHs were continuously released over this period. Because the PAHs were collected by the PUF plugs in a closed space, these findings confirm that electronic components are a source of PAHs. This result strongly suggests that computers and other electronic devices are associated with indoor PAH pollution.

The largest amount of PAHs was emitted from the transistor (mean: 861 ng/PUF), followed by the PVC cable (mean: 748 ng/PUF) and the printed circuit board (mean: 313 ng/PUF). In addition, the profiles of the PAHs emitted by each component differed. For the printed circuit board, LMW PAHs with two or three rings were most common, with Nap dominating, while Phe and Nap were emitted at equally high levels by the PVC cable, and two HMW PAHs (DahA and BghiP) were also detected. For the transistor, PAHs with two to four rings (e.g., Nap, Ace, Pyr, and Phe) were most common. Nap was the most dominant

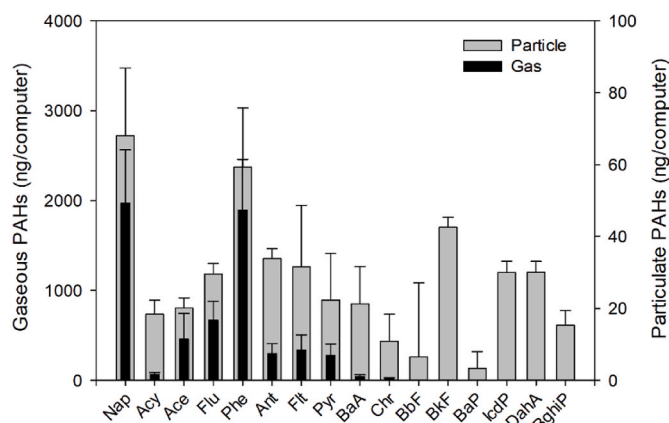


Fig. 3. Mean profiles of the amount of individual 16 PAHs in the gaseous and particulate samples collected from inside the computers (ng/computer, $n = 10$). The error bars represent the standard deviation.

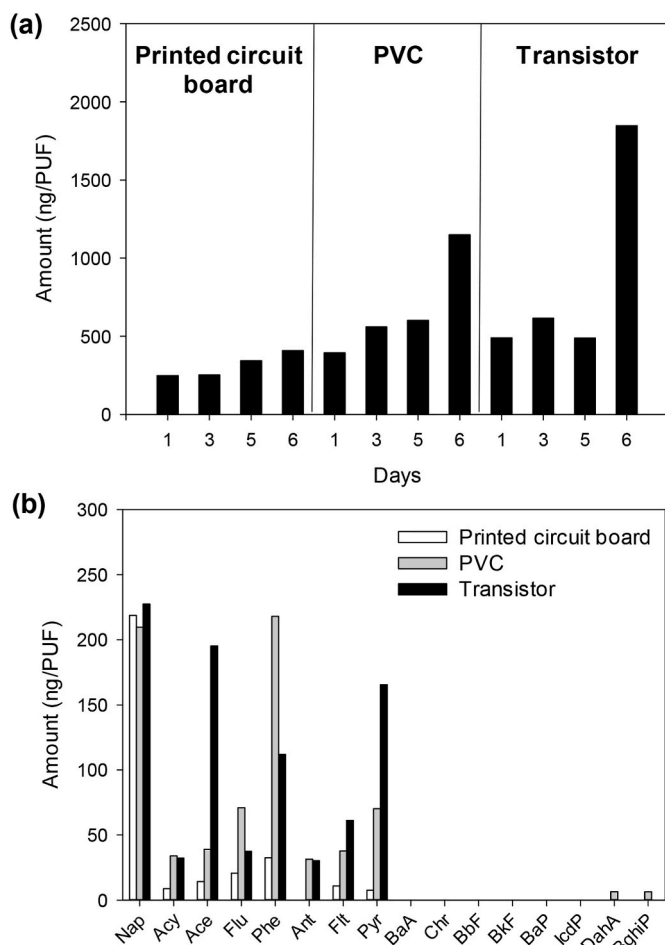


Fig. 4. PAHs released from electronic components (printed circuit board, polyvinyl chloride [PVC] cable, and transistor) stored with PUF plugs in amber bottles and heated at 60 °C for 1, 3, 5, and 6 days: (a) Amounts of Σ_{16} PAHs and (b) profiles of individual PAHs. The amount of PAHs in the PUF plug was divided by the weight of the electronic component (3 g each).

compound for all three electronic components, which is identical to the results for the HiVol sampling in the rooms and the PUF/GFF sampling inside the computers. The dominance of LMW PAHs emitted by the computer components suggests that the HMW PAHs detected in the office were mostly from outside or other indoor sources such as printers and copiers, which are known to emit fine dust associated with HMW PAHs [38]. In the present study, the concentration of particulate PAHs in the office where printers were used was higher than that in the other rooms without a printer or copier (Fig. 1a).

The electronic components tested in the present study mainly consisted of plastics and metals. Printed circuit boards are mostly composed of epoxy and phenolic resin, which are thermosetting plastics, and printed circuits made of copper are typically coated with tin-lead to prevent oxidation [62]. In addition, contact fingers are composed of tin-lead, nickel, and gold to improve the conductivity [62]. The main component of a PVC cable is PVC [63], and transistors are primarily made of silicon, and germanium is also used [64]. Plastics such as polycarbonate (PC) and polybutylene terephthalate (PBT) are also used in computer manufacturing due to their durability, heat resistance, and moldability, making them suitable for computer covers, keyboards, cable terminals, connectors, and fans [65].

The REACH regulations limit the levels of eight carcinogenic PAHs (BaA, Chr, BbF, BkF, BbF, BeP, BaP, and DahA) to 0.5 mg/kg for the plastic and rubber used in toys and childcare products and to 1 mg/kg for all other consumer goods [37]. Previous studies have confirmed that

PAHs are present in plastic and rubber products [15,33]. For example, carcinogenic PAHs have been detected in randomly selected consumer products such as wheel covers (331.2 mg/kg), outdoor playground pads (12.1 mg/kg), house pads (11.4 mg/kg), carpets (6.6 mg/kg), and toys (0.02 mg/kg) [15], suggesting that PAHs may exist as impurities in the plastics and rubbers used in consumer products. Since PAHs are generally produced by incomplete combustion at high temperatures above 200 °C [12,13], they are not generated by operating computers. Therefore, PAH impurities in the electronic components used in computers may be released into the atmosphere by the heat generated by computer use.

3.4. Potential health risk assessment

As discussed above, PAHs are released from computers into the air, and humans can be exposed to them via inhalation in an indoor environment. Given the PAH levels in the indoor air in the present study, the mean inhalation rate of Korean adults (14.61 m³/day) [50], and the average working hours in an office (8 h/day), exposure to PAHs via inhalation is 1.9 ng TEQ/day, which is higher than that of 15 PAHs at home (0.54 ng TEQ/day) [21] but lower than that in subway stations (3.1 ng TEQ/day), child care facilities (30.2 ng TEQ/day), supermarkets (5.0 ng TEQ/day), indoor parking lots (4.2 ng TEQ/day), terminal waiting rooms (3.5 ng TEQ/day), and internet cafés (52.7 ng TEQ/day) [5]. In particular, exposure to PAHs is very high in child care facilities and internet cafés, where multiple computers are used for a long time in an enclosed space [5].

The *HQ* for Nap through the inhalation of indoor air in the office was 3.9×10^{-4} . This is much lower than 1, meaning that there is no significant non-carcinogenic risk [45]. For comparison, the *HQ* through inhalation was calculated for PAH monitoring data in a classroom without a computer in our previous study conducted on the same campus [7], with the values of the other exposure factors except for the Nap concentration assumed to be the same as those used in the present study. The *HQ* in the office was higher than the *HQ* in the classroom (1.9×10^{-4}), suggesting that the non-carcinogenic risk of PAH exposure via inhalation in an indoor environment with computers is higher than that in an indoor environment without a computer. The carcinogenic risk calculated from the TEQs of seven carcinogenic PAHs in the office was 2.6×10^{-11} , which is three times higher than that for the classroom without a computer (8.5×10^{-12}) [7]. Though this risk is considered insignificant or acceptable [45], health risks from long-term exposure to PAHs are of concern, and combining exposure through other routes, such as outdoor exposure, ingestion, and dermal contact, may have a significant impact on human health in the long term.

4. Conclusion

In this study, we investigated the potential sources for PAH pollution in indoor spaces with high computer usage. LMW PAHs were primarily detected in the gaseous phase, whereas HMW PAHs were more commonly present in the particulate phase. The concentration of PAHs in the indoor air was affected by the number and size of the computers, the age of the computers, the internal heat, the ventilation, the room size, and the operating time per day. The PAH levels in the indoor air were higher than those in the outdoor air, which highlights the impact of indoor sources of PAHs. The emission of PAHs from computer components was confirmed, with LMW PAHs detected at the highest levels, indicating that PAH impurities in computer components may be released by the heat generated by computer use. This result is direct evidence that computers are an emission source of indoor PAHs. The health risk of inhalation exposure to PAHs in indoor air polluted by the operation of computers was also evaluated. Although the health risk of PAH inhalation in indoor spaces with computers was lower than the risk threshold set by the US EPA, health risks from long-term exposure to PAHs still require attention, particularly given outdoor exposure, ingestion, and

dermal contact as other potential exposure routes.

This study has uncertainties due to the representativeness of samples and the lack of information about the composition of computer components. In the future, a large chamber study is required to determine the effect of the mode of activities of computers on PAH emissions. Despite these limitations, this is the first study to provide concrete evidence that computers are an indoor PAH source by confirming PAH emissions from computer components in an enclosed space. In addition, this study demonstrated the possibility of indoor air pollution and the potential health risks arising from the use of computers, highlighting the need for the indoor environmental monitoring of other volatile compounds in addition to PAHs that can affect the human body.

CRedit authorship contribution statement

Sung-Hee Seo: Writing – original draft. **Kuen-Sik Jung:** Formal analysis. **Min-Kyu Park:** Writing – review & editing. **Hye-Ok Kwon:** Investigation. **Sung-Deuk Choi:** Supervision, Project administration.

Declaration of competing interest

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

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.buildenv.2022.109107>.

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