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Polycyclic aromatic hydrocarbon (PAH) levels in indoor air and their associated health risks for children across Europe

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ABSTRACT

Polycyclic aromatic hydrocarbons (PAHs) are compounds with well-established negative health effects. In this study, the levels of the 16 Environmental Protection Agency (EPA) - PAHs were determined in air and deposited dust in different indoor environments, including schools, homes and sports halls in six metropolitan areas across Europe (Athens, Barcelona, Colchester, Copenhagen, Helsinki and Lisbon). However, naphthalene, acenaphthylene, anthracene, and dibenz[ah]anthracene were excluded from the final dataset due to analytical limitations. Results showed considerable spatial variability, suggesting that the highest median PAH concentrations (>50,000 pg/m^3) in both schools and homes are linked to emissions from intensive local residential wood burning and poor ventilation, while the lowest PAH concentrations were observed in south-western Europe (<10,000 pg/m^3). The levels of high-molecular weight (HMW) PAHs were generally higher in schools compared to homes likely due to greater occupancy and accumulation of outdoor-derived particles. Low molecular weight (LMW) PAHs, on the other hand, were more prominent in homes and associated with household practices (e.g., cleaning and ventilation frequency). In addition to active air sampling, indoor air concentrations were also estimated based on concentrations in the deposited dust samples. However, this methodology underestimated HMW-PAHs. The estimated Excess Cancer Risk for a 5-year primary education exposure period (ECR_5) across all monitored sites ranged from 10^{-8} to 10^{-5}, remaining below the unacceptable risk value of 10^{-4}, which was used as a contextual reference within the USEPA cancer risk assessment framework. Nevertheless, these findings highlight the need for continuous proactive strategies to reduce PAH exposure and improve indoor air quality, particularly in environments occupied by children.

1. Introduction

Children constitute a vulnerable population exposed to air pollution due to their higher breathing rates relative to body weight and the ongoing development of their respiratory, immune and other physiological systems (Salvi, 2007). Epidemiological evidence links indoor exposure to a range of air pollutants with adverse health outcomes including respiratory allergies and chronic conditions (e.g., asthma,

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chronic obstructive pulmonary disease) (Almeida et al., 2011; Petropoulou et al., 2023). Poor indoor air quality has also been associated with neurobehavioral and psychosocial effects such as attention deficit hyperactivity disorder (ADHD), impacts on mental illness, increased stress levels and reduced academic performance (Alemany et al., 2018; Bhui et al., 2024; Gignac et al., 2021; Mentese et al., 2015). As children spend up to 90% of their time indoors, microenvironments such as schools and homes are critical for exposure and health risk assessments (Oliveira et al., 2019). Various studies have shown that indoor air is influenced by both outdoor pollutant infiltration and indoor sources related to occupant activities, often resulting in a complex pollutant mixture (Grydaki et al., 2024; Sadrizadeh et al., 2022; Shrestha et al., 2019).

Polycyclic aromatic hydrocarbons (PAHs) are among the indoor pollutants of interest, as also addressed in guidelines issued by the World Health Organization (WHO, 2010), due to their well-documented negative health effects and their physicochemical characteristics which contribute to some persistence in indoor air (Mortamais et al., 2017; Weschler and Nazaroff, 2008). PAHs are semi-volatile organic compounds mainly formed from incomplete combustion of organic materials, making traffic emissions, residential heating, and industrial processes major sources (Howsam and Jones, 1998; Van Drooge, 2013). Several PAHs are classified as carcinogenic by the International Agency for Research on Cancer (IARC, 2010, 2012). In outdoor ambient air, benzo[a]pyrene (BaP) has a regulatory target value of 1 ng/m^3 established for its annual mean concentration within the European Union (Directive, 2004/107/EC), which is set to be a limit value in 2030 (Directive, 2024/2881, 2024). The WHO suggests an air quality guideline value of 0.12 ng/m^3 for BaP (WHO, 2021).

A growing body of research exists on indoor environments predominantly occupied by children, to evaluate the concentrations of various pollutants and associated risks (Almeida et al., 2011; Baloch et al., 2020; Pacitto et al., 2018; Rivas et al., 2014; Stranger et al., 2008; Xu et al., 2018). Numerous studies have assessed PAH levels in schools and daycare centers, considering these settings as representative environments for children's exposure (Sadrizadeh et al., 2022). However, often they consist of isolated case studies conducted within a single city, limiting the broader applicability of their findings (Fuoco et al., 2015; Krugly et al., 2014; Slezakova et al., 2017). Comparisons are usually limited to intra-city contrasts such as differences between traffic-influenced and urban background sites or between locations with differing characteristics (Alier et al., 2013; Gustafson et al., 2008; Manoli et al., 2016; van Drooge et al., 2020). Moreover, comparisons across different countries are typically based on data drawn from independent studies with different methodological approaches (Fromme et al., 2023; Ouyang et al., 2020), while others assess only either the gaseous or particulate phase PAHs (Alier et al., 2013; Alghamdi et al., 2020; Liaud et al., 2021; Oliveira et al., 2017). This heterogeneity in sample collection complicates direct comparisons between studies and introduces uncertainties and limitations in interpreting results, highlighting the need for more harmonized approaches. Additionally, although PAH concentrations have also been investigated in residential environments, most of these studies have focused on adult health risk assessments, with comparatively limited attention given to children's exposure (Maria Florencia et al., 2022; Živančev et al., 2022). Therefore, integrated studies which include both schools and homes are essential for a more accurate and comprehensive evaluation of children's total exposure to PAHs in indoor environments.

The present study conducted within the framework of the Horizon Europe project InChildHealth (https://inchildhealth.eu), aimed to provide a broader insight into PAH levels in indoor environments occupied by school-age children, including schools, sport halls and selected homes in six European metropolitan areas representing different climatic regions. The inclusion and joint assessment of multiple children-related microenvironments, rather than a single indoor setting, provides a more comprehensive characterization of children's potential exposure to PAHs. This study includes PAH concentrations in both air and dust samples and allows comparisons between the areas. It also aimed to compare active vs. passive sampling approaches and to determine the extent to which passive sampling, as a less invasive method, can provide representative or complementary results. By generating comparable multi-city data, the study enables more robust conclusions regarding current PAH levels and supports the development of evidence-based recommendations for effective mitigation measures to improve indoor air quality and protect children's health.

2. Materials and methods

2.1. Study design

This study reports results from schools, homes and sport halls obtained from a collaborative and harmonized sampling campaign (2023-2025) involving the collection of air and dust samples in six European metropolitan areas: Barcelona, Lisbon, Athens, Colchester, Copenhagen and Helsinki (Fig. 1). The participating areas represent urban environments across Southern, Western and Northern Europe differing in population size, density, traffic intensity, domestic heating methods and climatic characteristics (Eurostat, 2020). Barcelona is considered the city with the highest vehicle density in the European Union (EU), at about 6000 vehicles/km^2 (Ajuntament de Barcelona, 2020), with 3.3 million inhabitants in its wider metropolitan area. Athens and Lisbon are two of the oldest capital cities in Europe, each with a wider metropolitan population of over 3.5 million inhabitants and 2.8 million inhabitants respectively, making them highly urbanized and densely populated. Moreover, the municipality of Athens is considered one of the most densely populated urban areas in Europe with more than 19,000 inhabitants/km^2. In contrast, Colchester, located in the north-east of Essex, has a much smaller area and population of around 190,000 inhabitants (Colchester City Council, 2025), resulting in lower overall urban density and a different urban character. Lastly, Helsinki and Copenhagen are Nordic capitals with similarly growing populations in their greater metropolitan areas, with both cities having more than 1.4 million inhabitants. In Denmark, schools were also included in the smaller towns of Odense (about 200,000 inhabitants, i.e. comparable to Colchester) and Vejle (about 60,000 inhabitants). However, Copenhagen was used as the reference name for the Danish sites.

In each area five primary schools, one sports hall and five homes were included in the study. To ensure participants' privacy and compliance with data protection and ethical research standards, the precise geographic coordinates and site names are not disclosed. Each sampling location was assigned a numerical code (#1-#5) within each area, with additional letters indicating site type (S = school, H = home, SH= Sports hall). The type of setting was characterized as Urban traffic (UT), Urban background (UB), and Rural (RU) according to the definitions that the new EU Air Quality Directive (2024/2881) provides and also Suburban (SUB) that represents an intermediate category between UB and RU (Tables S1 and S2).

Active air sampling in primary schools took place in one classroom per school, occupied by children over one school week (Monday to Friday) during both a heating (cold) and a non-heating period (warm), defined according to the local meteorological conditions (Figs. S1 and S2). School sampling was scheduled during school hours and covered 8 h per day, depending on each country's academic timetable. The same strategy was applied in sports hall sampling. In homes, sampling was performed similarly during two periods, each lasting one week with the living room being the primary sampling location. Here, sampling generally started on Friday afternoon and lasted 30 h, while weekday measurements were carried out over 15-h intervals that covered occupied periods (usually afternoon until morning). For passive sampling, settled dust was collected on a quartz filter that was deployed at each location for a duration of approximately six months. Lastly, detailed information on specific characteristics of the sampled locations,

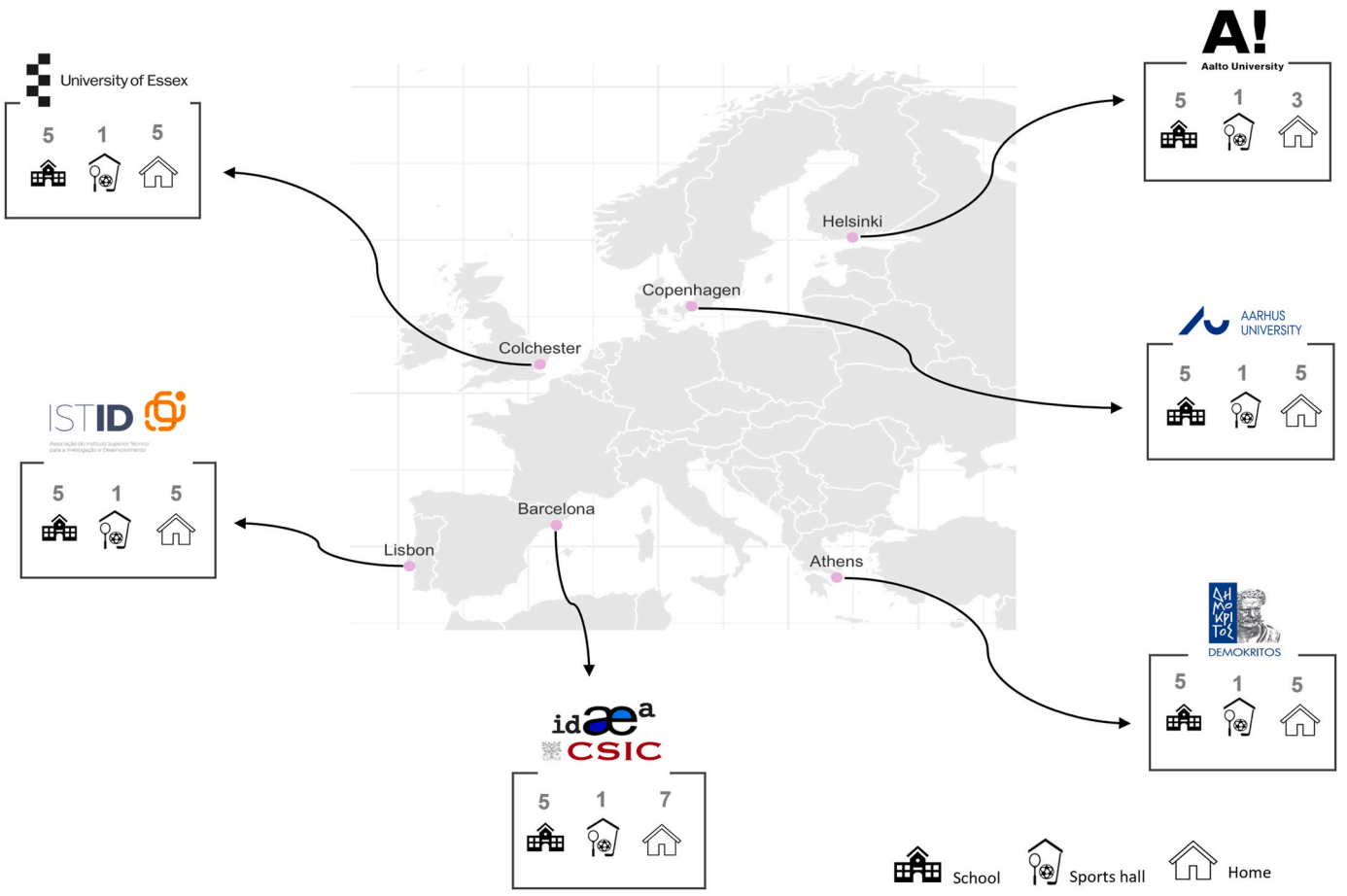

Fig. 1. Map with the participating cities and partner institutions involved in the sampling campaign, showing the number and type of monitored sites (schools, sports halls, homes) in each metropolitan area.

including cleaning and ventilation practices, as well as other factors that could influence PAH levels or act as direct indoor sources (e.g., use of fireplaces, smoking habits etc.), was collected through walkthrough surveys focusing on occupant indoor activities (Tables S1 and S2).

2.2. Sample collection

The sampling methodology for gas- and particle-phase PAHs in ambient air is described in detail elsewhere and the assignment of PAHs to the different phases reflects their expected partitioning behavior rather than representing direct phase-resolved measurements (Aretaki et al., 2024). Briefly, low-volume air sampling pumps (Leland legacy SKC, Dorset, UK) were used to collect air samples at each indoor location at a height of 1 m above the floor. To minimize disturbance in occupied environments, the pumps were placed in sound-insulated boxes. The pump was connected with two solid phase extraction (SPE) cartridges (Isolute ENV+200 mg/6 mL, Biotage, Uppsala, Sweden) via a flow splitter and the pumps had a flow rate of 4 L/min to collect PAHs in both gas and particle-phase. Former studies used this active sampling methodology for the collection of airborne semi-volatile flame retardants and plasticizers in indoor environments (Desmet et al., 2025; Esplugas et al., 2022). No filter or particle-size selection device was used upstream of the SPE cartridges but from the collected PAHs, low-molecular weight PAHs (LMW-PAHs) were considered predominantly associated with the gas phase, whereas high-molecular-weight PAHs (HMW-PAHs) were considered primarily associated with the particle phase, consistent with their well-established atmospheric partitioning behavior (Lohmann and Lammel, 2004). In this study the LMW-PAHs group included acenaphthene, phenanthrene, fluoranthene and pyrene while the HMW-PAHs group included benz[a]anthracene, chrysene, benzo[b+j]fluoranthene, benzo[k]fluoranthene, benzo[a]pyrene, indeno[1,2,3-cd]pyrene and benzo[ghi]perylene) (Howsam and Jones, 1998). One cartridge was used for PAH analysis, the other for other chemical analyses. For passive sampling of dust, prebaked (200 °C for 4h) quartz fiber filters (150 mm; PALL 2500 QAT-UP) were placed on aluminum foils or in pre-cleaned glass petri dishes at 2-m height above the floor in sites that were not affected by human activities. This method enables the collection of settled dust on the filter, which in other studies is often collected by brushes and dustpans (Wang et al., 2021). Here, the filters were weighed before and after sampling to obtain the settled dust mass. After sampling, the SPEs and filters were wrapped in aluminum foil and heat-sealable bags (Kapak SealPAK, Ampac Flexibles,USA), shipped cold to the laboratory in Barcelona where they were stored at −20 °C before further analysis. One field and laboratory blank from each sample batch were provided and analyzed alongside the air and dust samples to identify any background contamination introduced during transport, storage or analytical processing.

2.3. PAH analysis in SPE and filter samples

Detailed information regarding the analytical procedure and quality assurance for the PAH analysis is provided elsewhere (Aretaki et al., 2024). Briefly, prior to extraction, a mixture (1.25 ng) of sixteen deuterated PAHs, serving as internal standards, was added to each SPE cartridge and to the quartz filters (previously cut in 16 pieces and placed in vials). Internal standard solution contained sixteen deuterated polycyclic aromatic hydrocarbons (d8-naphthalene, d8-acenaphthylene, d10-acenaphthene, d10-fluorene, d10-phenanthrene, d10-anthracene,

d10-fluoranthene, d10-pyrene, d12-benz[a]anthracene, d12-chrysene, d12-benzo[b]fluoranthene, d12-benzo[k]fluoranthene, d12-benzo[a]pyrene, d12-indeno[123-cd]pyrene, d12-benzo[ghi]perylene, d14-dibenz[ah]anthracene) dissolved in cyclohexane (Dr.Ehrenstorfer, Augsburg, Germany). The 16 parent PAHs were analyzed in the samples. All samples were extracted with high purity acetone and *n*-hexane (1:1) (Merck, KGaA, Darmstadt, Germany). For the extraction of quartz filters in an ultrasonic bath (15 min), an initial volume of 50 mL of the solvent mixture was used. Subsequently, an additional 20 mL of solvent was added to the filter and the extraction process was repeated. The joint extract was filtered through a funnel packed with glass wool (PanReac AppliChem), and concentrated under a gentle N_2-gas stream in Turbo-Vap (20 °C) to 1.5 mL, and 0.5 mL was transferred to vials further concentration to 25 μL. The extracts were analyzed on an Agilent 7890A GC system coupled with an Agilent 7000B GC–MS/MS equipped with a fused 30 m column (HP-5MS, 0.25 mm × 25 μm film thickness) operating in selected reaction monitoring (SRM) mode for acquisition of the 16 PAHs. PAH concentrations were determined by external calibration. Field and laboratory blanks, including transport and storage tests, were included as part of the QA/QC program to assess the conservation of the samples, and blank-corrected concentrations were reported. The method performance showed analytical recoveries between 40% and 115% and detection limits of 3-10 pg/m^3.

2.4. Estimation of PAH air concentrations from dust samples

The quartz filters collected PAHs and dust over a six-month time period, and allows to calculate the fluxes of the individual PAH mass (F_{PAHi}) and the deposited dust mass (F_{dust}), as well as the PAH concentrations in the dust samples Cp (μg/g) through the ratio of the two fluxes. Here, the passive sampling method acts as a long-term sink of PAH rather than a direct measurement of the indoor air quality only when the children are present, as was applied in the case of the active sampling method. To evaluate if the passive sampling gives representative or complementary results, the PAH dust concentrations were used to estimate the indoor air PAH concentrations. The air particle-phase PAH concentration ($Cpart_{-PAH}$ (ng/m^3)) at the studied sites was estimated by multiplying the flux ratios with the total suspended particle (TSP) concentrations (equation (1)).

$$Cpart_{-PAHi} = (F_{PAHi} \ / \ F_{dust}) \ * \ TSP \quad (1)$$

where TSP value was set at 60 μg/m^3 and also at a range from 20 μg/m^3 and 130 μg/m^3 based on previous studies in schools in Barcelona (Moreno et al., 2014; Torres-Agullo et al., 2026). In order to estimate the indoor gas-phase concentrations of individual PAH compounds in the air, a gas-particle partitioning (K_p) calculation was used (Lohmann and Lammel, 2004, equation 2), including the average and a range of fractions of organic matter ($f_{OM} = 0.3$; range from 0.1 to 0.5) and elemental carbon ($f_{EC} = 0.02$; range 0.01 to 0.05) in atmospheric particulate matter in the schools and homes in Barcelona (Aretaki et al., 2026), and the corresponding temperature corrected (20 °C) octanol-air (K_{OA}) and soot-air (K_{SA}) partitioning coefficients for the individual PAHs, and considering densities of octanol (ρ_{oct}) and EC (ρ_{EC}) of 1 (Lohmann and Lammel, 2004; Odabasi et al., 2006) (Table S3).

$$K_p = 10^{-12} \left[(f_{OM} / \rho_{oct}) \ * \ K_{OA} + (f_{EC} / \rho_{EC}) \ * \ K_{SA} \right] \quad (2)$$

The indoor gas-phase PAH concentrations (ng/m^3) of in the air were calculated by:

$$Cgas_{-PAHi} = C\ part_{-PAHi} \ / \ Kp \quad (3)$$

where the sum of the individual PAH concentrations in both gas and particle phase ($Cgas_{-PAHi} + C\ part_{-PAHi}$) represents the total indoor air PAH concentrations.

2.5. Health risk assessment

Carcinogenic risks associated with exposure to polycyclic aromatic hydrocarbons can be assessed by expressing their combined toxicity as benzo[a]pyrene equivalent concentrations (BaPeq). This approach applies toxic equivalency factors (TEFs), which relate the carcinogenic potency of individual PAHs to that of BaP. The TEFs proposed by Nisbet and LaGoy (1992) were used to calculate the BaPeq according to the following equation (4):

$$BaPeq = \sum iCi * TEFi \quad (4)$$

, where *Ci* represents the concentration of the i_{th} PAH compound. In order to perform risk assessment for the Excess Cancer Risk (ECR) for a 5-year exposure during primary education, the total PAH mixture expressed as BaPeq was used to calculate the daily inhalation dose (δBaPeq) for children in indoor spaces of schools, homes and sports halls. The parameters used for the estimation of ECR_5 in children were taken from the US EPA methodology (US. Environmental Protection Agency, 2011) and are presented in Table S4. More specifically, the ECR_5 was calculated with the following equation (5) adapted from Buonanno et al. (2015) and it represents the incremental probability of cancer associated with this defined childhood exposure period rather than a lifetime-averaged risk:

$$ECR_{5\ years} = \sum_{y=1}^{y=5} \left(\frac{\delta_{BaP\ eq} * SF_{BaP}}{BW} * R \right) \quad (5)$$

The parameters are the following:

- δBaP_{eq} is the daily inhalation dose (mg/day)
- SF_{BaP} (kg day mg $^{-1}$) is the inhalation cancer slope factor for the cancer potency of BaP (OEHHA, 2009)
- BW (kg) is the average body weight of children 6-11 years old
- R is the ratio of occupied days per year, differing in schools and homes. An exposure duration of 180 days/year was assumed for the school environment to represent the typical number of school days across European cities. For homes where it is considered that children are present almost every day, an exposure duration of 350 days/year was applied, following the recommendations from the US EPA methodology (USEPA, 2009).

2.6. Statistical analysis

Statistical comparisons between groups were conducted using the open-source software R version 4.4.1 (R Foundation for Statistical Computing, Vienna, Austria) within the RStudio environment. Differences among groups were first assessed using the Kruskal-Wallis test. When significant differences were detected, a post-hoc pairwise Mann-Whitney *U* test was performed. Statistical comparisons were based on site-level mean concentrations calculated across the two sampling periods, with each sampling site treated as an independent observation. For sites where only one valid seasonal campaign was available, the concentration from that campaign was used as the site-level value. Statistical significance was set at $p < 0.05$ and p-values were adjusted using the Holm-Bonferroni correction. Data visualization was conducted using the ggplot2 package (Wickham, 2016) (for boxplots). Boxplots were constructed using the standard Tukey method implemented in the ggplot2 package, with the central line representing the median value, while the lower and upper edges of the box correspond to the 25th (Q1) and 75th (Q3) percentiles, respectively. The box therefore represents the interquartile range (IQR). The whiskers extend to the most extreme observations within 1.5 × IQR from the lower and upper quartiles. Observations outside the whiskers are displayed individually as outliers.

3. Results and discussion

3.1. Indoor concentrations of atmospheric PAHs across European cities

Naphthalene, acenaphthylene, anthracene, and dibenz[ah]anthracene are not reported from this study as the first two exhibited elevated blank levels and low recoveries. On the other hand, anthracene, and dibenz[ah]anthracene were quantification limits in most samples (Aretaki et al., 2024). Regarding the total measured atmospheric PAH concentrations determined from cartridge extractions across all sampling sites in the six European cities, Colchester exhibited the highest median concentration as well as the greatest absolute variability (56, 915 ± 52,696 pg/m^3), whereas Lisbon showed the lowest levels (7741 ± 9070 pg/m^3). Barcelona presented higher median concentration than Lisbon (11,204 ± 8231 pg/m^3), while the Nordic cities, Helsinki and Copenhagen, showed moderate and comparable total concentrations to one another (15,003 ± 9351 pg/m^3 and 14,128 ± 32, 088 pg/m^3, respectively). Finally, Athens, the southernmost city included in the study, exhibited higher median concentrations than the Nordic cities, although lower variability (17,220 ± 8457 pg/m^3). All cities exhibited similar compositional profiles of PAHs in schools and homes, although profiles differed between these environments, and significant differences were observed in the concentrations of individual PAHs among the six cities as shown in Figs. S3 and S4. BaP, the only currently regulated PAH in ambient air with an average annual limit value of 1000 pg/m^3, exhibited considerable spatial variability across the cities. Athens showed an average value of BaP of 206 ± 301 pg/m^3 while Colchester followed with 166 ± 208 pg/m^3. Helsinki and Copenhagen displayed overall moderate mean concentrations of 113 ± 49 pg/m^3 and 66 ± 60 pg/m^3, respectively. Conversely, Lisbon and Barcelona were characterized by substantially lower mean BaP concentrations of 33 ± 25 pg/m and 38 ± 32 pg/m, respectively. Notably, none of the cities exceeded the annual EU limit value but in Colchester and Athens the annual mean BaP concentrations exceeded the WHO air quality guideline value of 120 pg/m^3.

The overall PAH concentration pattern among metropolitan areas in indoor air in the present study is in agreement with other studies. Haglund et al. (2024) examined indoor concentrations of various pollutants in different European countries, and reported a similar spatial pattern concerning PAH levels, with the highest concentrations observed in the UK, followed by Greece, while other Southern European countries showed comparatively lower PAH concentrations and Northern countries moderate levels. The differences in PAH levels among these European cities suggest differences in dominant emission sources but also site-specific characteristics and varying indoor activities (Tables S1 and S2). Although traffic emissions are generally considered a major source of PAHs, sites (schools/homes) affected by indoor combustion sources, such as fireplaces, or by nearby biomass burning exhibited the highest PAH concentrations regardless of their urban classification (Figs. S5 and S6). These results highlight the importance of combustion activities and infiltration in compromising indoor air quality (Hu et al., 2018; Lovrić et al., 2024; Manoli et al., 2016; Rivas et al., 2015). In addition, household practices varied substantially across cities. However, ventilation frequency appeared to influence PAH levels more strongly than cleaning frequency (Fig. S7). Homes in Colchester, where the highest PAH concentrations were observed, reported infrequent ventilation despite weekly cleaning, whereas homes in Lisbon generally combined daily ventilation with weekly cleaning and exhibited lower PAH concentrations. Similar trends were observed in Copenhagen and Helsinki where numerous locations were not ventilated frequently. Although daily ventilation was reported in most homes in Athens and Barcelona, relatively high PAH concentrations were observed in some homes, suggesting that the timing of ventilation should also be considered in order to avoid infiltration during high-emission periods. These findings are consistent with previous studies showing that household activities influence indoor PAH concentrations, with cleaning reducing PAH accumulation in dust and ventilation decreasing concentrations through air renewal and dilution (Li et al., 2005; Romagnoli et al., 2014; Živančev et al., 2022).

As shown in Figs. 2 and 3, both LMW and HMW-PAH concentrations differed significantly among cities, with greater variability observed in homes, suggesting that a wider range of parameters may influence indoor PAH levels in residential environments. LMW-PAH concentrations were consistently the highest in Colchester, consistent with the prominence of domestic wood burning, which remains a major contributor to local air pollution across the UK despite ongoing mitigation efforts (Kuye & Kumar, 2025, 2025; Colchester City Council, 2025). However, HMW-PAHs showed a different pattern, with Athens exhibiting the highest median concentrations in schools and homes, which may be associated with the combined influence of severe traffic congestion and residential biomass burning, particularly during the heating season as shown in previous studies (Pateraki et al., 2019; Tsiodra et al., 2021, 2024; Valavanidis et al., 2006). In contrast, Barcelona and Lisbon showed the lowest HMW-PAH concentrations, consistent with the limited contribution of residential combustion sources under milder climatic conditions and the predominance of other types of emissions derived from traffic, industrial activities, and shipping (Cerqueira and Matos, 2019; van Drooge et al., 2020; Jaén et al., 2025). In the Nordic cities, contributions from regional biomass burning and outdoor infiltration may also have influenced indoor HMW-PAH levels as supported by previous studies (Aurela et al., 2015; Hellén et al., 2017; Karlsson et al., 2015; Kukkonen et al., 2020), despite the widespread use of district heating systems and the efforts of local authorities to monitor these emissions and implement measures on proper fuel storage and the appropriate use of wood stoves (Green Transition Denmark, 2018; HSY (Helsinki Region Environmental Services, 2025). Although Helsinki schools have advanced HVAC systems, the HMW-PAH levels were comparable to those in cities without mechanical ventilation. Other studies have reported that the efficiency of mechanical removal of PAH through ventilation can be affected by factors such as filter maintenance and infiltration of outdoor combustion-derived particles indoors (Sadiktsis et al., 2016; Stasiulaitiene et al., 2019).

3.2. Intra-city comparisons and seasonal variability

Differences were observed between atmospheric PAH indoor concentrations in the school and home environments within each city. At a compound level, phenanthrene was consistently the most abundant in all samples across all cities, with concentrations ranging from 546 to 91,029 pg/m^3 followed by acenaphthene (200 - 84,736 pg/m^3) and fluorene (176-27,706 pg/m^3) (Table S5). This observation is consistent with previous studies on PAH concentrations in Europe (Garrido et al., 2014; Haglund et al., 2024; van Drooge et al., 2010). Overall, LMW-PAHs dominated all the collected air samples and defined the PAH profiles across all cities and sampling sites with concentrations approximately two orders of magnitude higher than those of HMW-PAHs. The substantially higher concentrations of LMW-PAHs compared with HMW-PAHs are consistent with findings from previous indoor air studies and are mainly attributed to their greater volatility and predominance in the gas phase as they are more readily emitted from combustion sources and can remain suspended in indoor air for longer periods, whereas HMW-PAHs preferentially adsorb onto particulate matter and hence, are more efficiently removed through deposition and ventilation processes (Lohmann and Lammel, 2004; Ravindra et al., 2008).

The two groups of PAHs also showed different patterns in their occurrence as the LMW-PAHs were generally higher in the homes and the HMW-PAHs in the schools (Fig. 4). More specifically, the higher LMW-PAH concentrations observed in homes may reflect the greater number and diversity of PAH-emitting household activities, such as cooking, candle burning, and residential heating (Lovrić et al., 2024; Orecchio, 2011; Romagnoli et al., 2014), whereas there are less

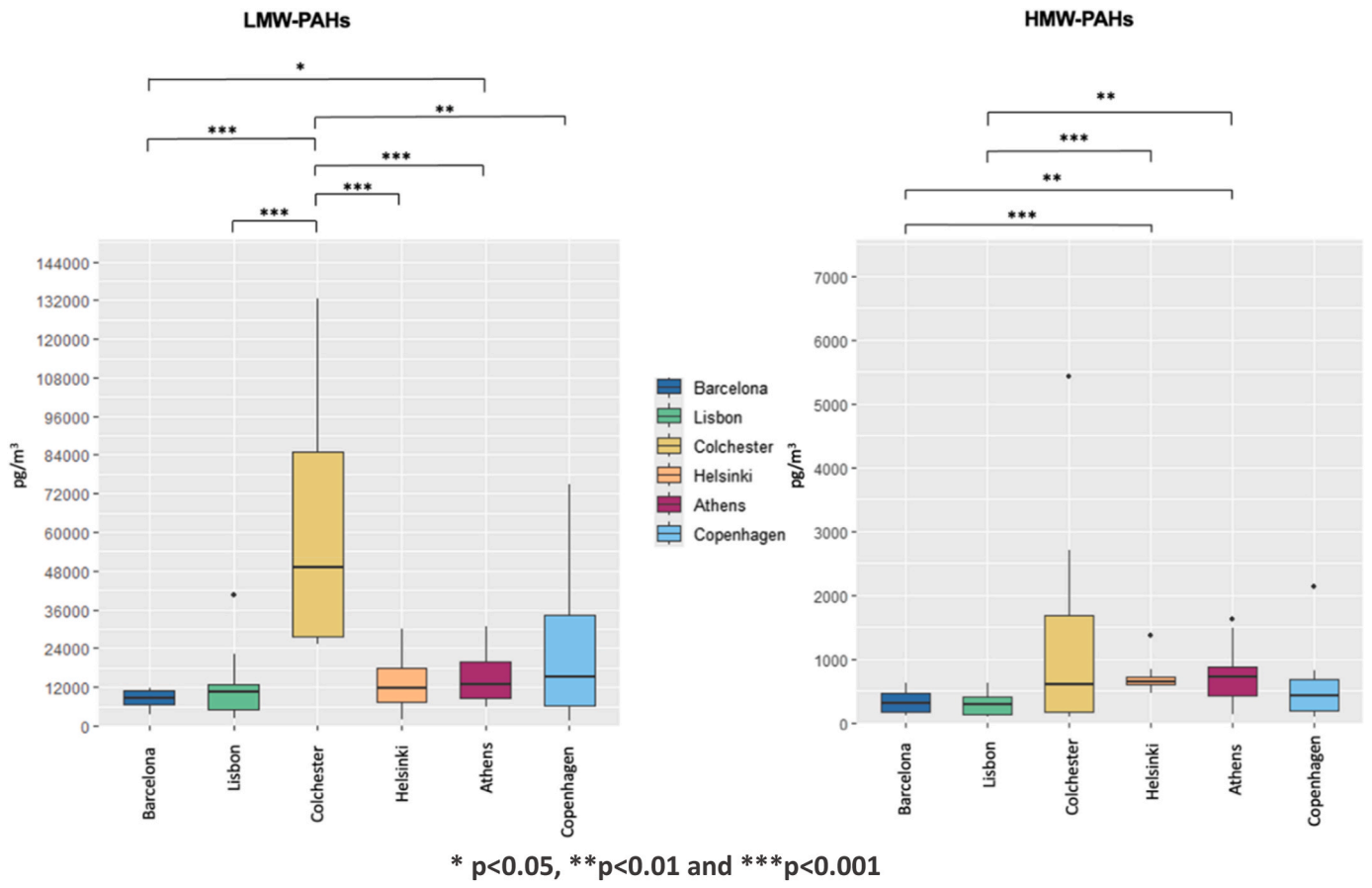

Fig. 2. Indoor concentrations of grouped low-molecular weight PAHs (LMW-PAHs) and high-molecular weight PAHs (HMW-PAHs) in pg/m^3 at schools in the different cities. Statistical comparisons were performed using the Mann-Whitney *U* test in R software. The boxplots include all seasonal measurements to illustrate the whole distribution of PAH concentrations, whereas the statistical comparisons were performed using the mean concentration for each sampling site for the two sampling periods ensuring the independence of observations.

LMW-PAH emitted during the more standardized and controlled indoor activities in schools. Lisbon was the only city which exhibited higher median concentrations of LMW-PAHs in schools (10,876 ± 11, 316 pg/m^3) compared to homes (6239 ± 3985 pg/m^3). This is likely because most school locations were situated in areas more strongly influenced by traffic compared to homes, with some also located near an airport and biomass burning activities (Tables S1 and S2). Although a similar pattern would be expected in Barcelona which is also a traffic-dominated city, additional data from the same indoor sites (Aretaki et al., 2026), identified tobacco smoke as a major indoor source of LMW-PAHs in several monitored homes. Athens and Barcelona were the only cities where smoking was noted in the homes (Table S2). However, for Athens, the influence of traffic seems to dominate. Regarding the HMW-PAHs, schools tended to exhibit higher concentrations and greater variability compared to homes which was likely associated with enhanced resuspension of settled dust due to children's activity as well as the transfer of soil and outdoor particles indoors by footwear, both of which contribute to increased HMW-PAHs levels (Xu et al., 2018). However, cities such as Colchester and Athens where biomass burning is practiced did not show differences among HMW-PAH median concentrations in schools and homes. Moreover, the sites affected by biomass burning through infiltration or through a direct indoor source (fireplace) exhibited comparatively higher log-transformed HMW-PAH concentrations relative to other sites (Figs. S5 and S6). This finding is consistent with previous studies demonstrating that these activities preferentially enhance the formation of HMW-PAHs (Han et al., 2020). Differences in sampling times between schools and homes were necessary to coincide with children's occupancy of each microenvironment and may therefore have contributed, to some extent, to the observed differences in PAH concentrations. Nevertheless, the consistent patterns observed in all the cities with respect to LMW and HMW-PAH concentrations may reflect the differences in emission sources, occupant density, and indoor activities within these microenvironments.

Seasonal variation in PAH levels was also observed across the six cities, with concentrations of both groups of PAHs generally lower during the non-heating period (Table S6). This pattern aligns with previous studies reporting increased concentrations of PAHs in colder months, primarily due to increased emissions from cold engine starts and biomass combustion for residential heating, as well as reduced photo-chemical degradation of PAHs (Cao et al., 2019; Finardi et al., 2017; Morville et al., 2011; Ravindra et al., 2008). The most evident seasonal contrast was observed in Colchester, where LMW-PAH median concentrations during the non-heating period were less than half of those measured during the heating period, while HMW-PAHs decreased by approximately one order of magnitude. A comparable pattern was observed in Athens, Helsinki and Copenhagen, where PAH concentrations decreased substantially during the non-heating period, although not for LMW-PAHs in homes in Helsinki. For Copenhagen, some outdoor temperature differences were small between the heating and non-heating period (Fig. S2), and in Helsinki, some temperatures were low (<10 °C) even during some "non-heating" sampling days. In Lisbon, seasonal differences were less noticeable, with relatively small temperature variations between the two sampling campaigns and the strong influence of traffic emissions at several sampling sites which are not season dependent. Barcelona was the only city where higher LMW-PAH

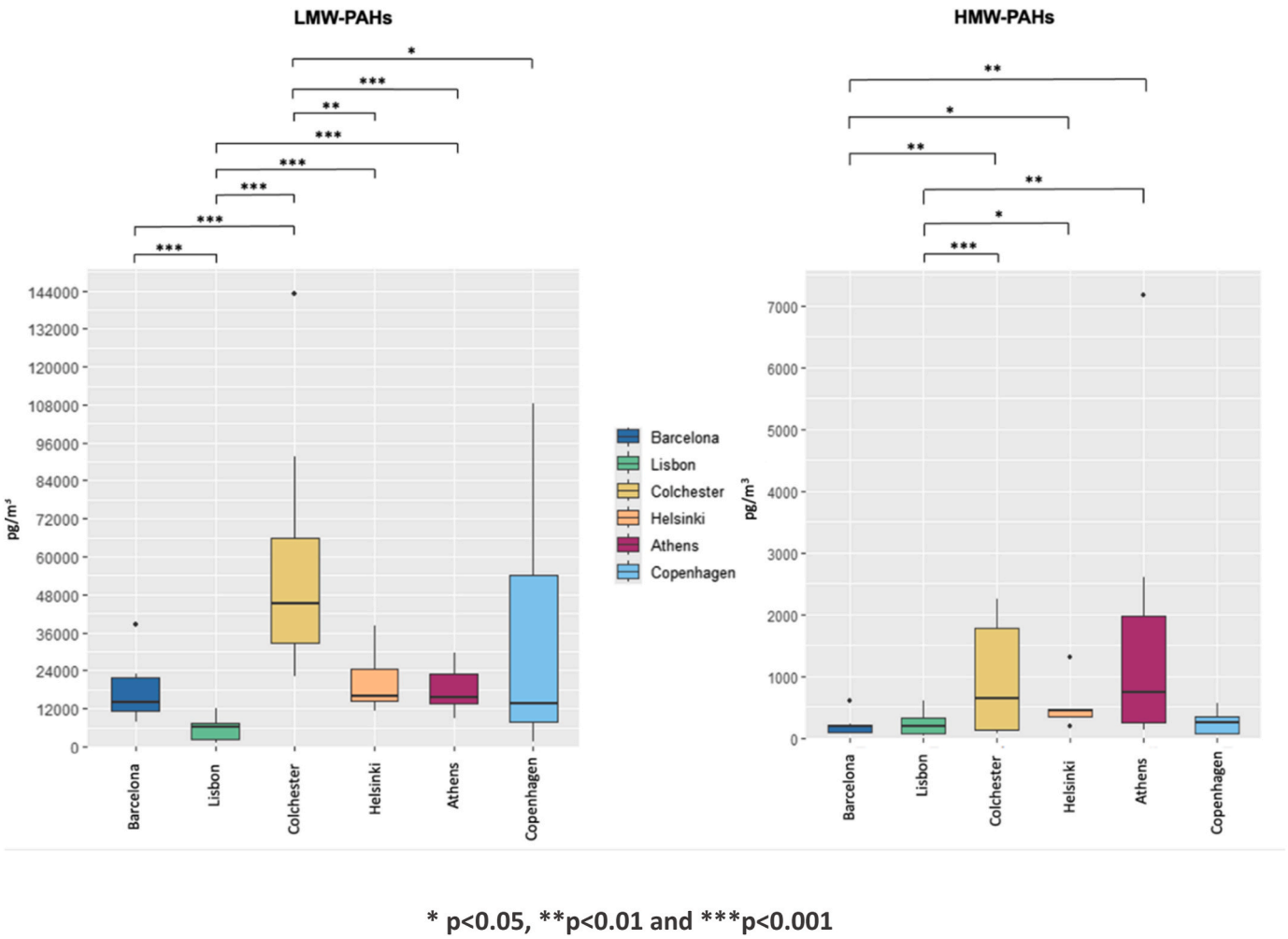

Fig. 3. Indoor concentrations of grouped low-molecular weight PAHs (LMW-PAHs) and high-molecular weight PAHs (HMW-PAHs) in pg/m^3 in homes in the different cities. Statistical comparisons were performed using the Mann-Whitney *U* test in R software. The boxplots include all seasonal measurements to illustrate the whole distribution of PAH concentrations, whereas the statistical comparisons were performed using the mean concentration for each sampling site for the two sampling periods ensuring the independence of observations.

concentrations were observed during the non-heating period compared to the heating one in homes, which could be due to the considerably higher temperatures recorded during the non-heating campaign (~25 °C), that could have enhanced the volatilization and indoor accumulation of PAHs from indoor sources and previously deposited reservoirs (Keyte et al., 2013; Ohura et al., 2004), together with contributions from additional indoor sources identified at some of these residential sites from a previous study in Barcelona (Aretaki et al., 2026).

3.3. PAH concentrations in settled dust and comparison with active sampling

The mass fluxes of settled dust particles and PAHs are provided in Tables S7 and S8, while the calculated PAH dust concentrations are given in Table S9. The average mass flux of dust particles in indoor environment are very similar in the different cities, ranging from 0.34 ± 0.08 μg/cm^2/day to 0.49 ± 0.15 μg/cm^2/day, which is consistent with other reported indoor values (Edwards et al., 1998). On the other hand, the average total PAH dust concentrations showed larger variation among the cities, ranging from 0.8 ± 0.4 μg/g to 4.7 ± 5.5 μg/g, with highest concentration in Colchester (Table S9). The dust PAH concentrations are in the range of those observed in other studies in indoor dust samples (Ali, 2019; Kang et al., 2010; Mosallaei et al., 2023). The measured PAH dust concentrations show a significant correlation with PAH air concentrations ($R^2 = 0.4$; $p < 0.05$; Fig. S8). This indicates that there is a relationship between them despite the difference between sampling techniques, with dust samples being a continuous accumulation of settled dust over six months, whereas PAH air concentrations were actively sampled in the daytime during two weeks in heating and non-heating seasons within this period.

To evaluate if the passive dust sampling gives representative or complementary results, the PAH dust concentrations were used to estimate the indoor air concentrations of the individual PAH compounds according to equations (1)–(3) and a sensitivity analysis was performed by applying a range of TSP concentrations as well as fEC and fOM which were observed in previous studies. The comparison between the estimated and measured indoor air PAH concentrations is shown in Fig. 5. LMW-PAHs, such as phenanthrene (Fig. S9), showed ratios in the range of one for the different cities. Good agreement between estimated and measured results were obtained for Barcelona, where measured data on organic matter and elemental carbon, as well as measured total suspended particles (TSP) concentrations were available. Although the values of these parameters in the other sites will probably be in the range of those measured in Barcelona, the application of these parameters for all sites in the different cities could lead to biases. The estimated air concentrations of LMW-PAHs are primarily controlled by the fEC, as varying fEC while keeping fOM and TSP constant, resulted in a relative

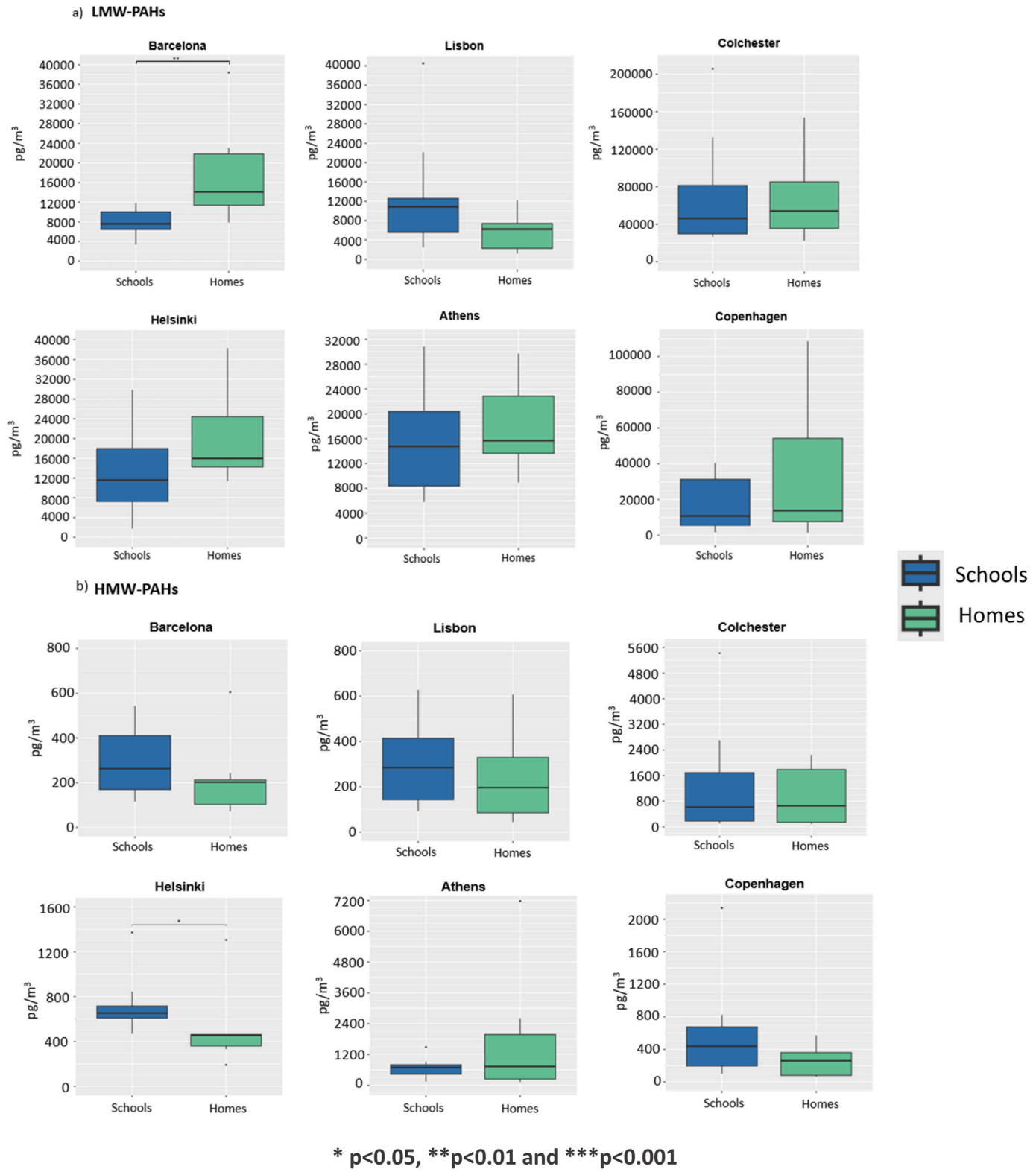

Fig. 4. Indoor concentrations of grouped a) low-molecular weight PAHs (LMW-PAHs) and b) high-molecular weight PAHs (HMW-PAHs) in pg/m^3 at the schools and homes in the different metropolitan areas. Statistical comparisons were performed using the Mann-Whitney *U* test in R software. The boxplots include all seasonal measurements to illustrate the whole distribution of PAH concentrations, whereas the statistical comparisons were performed using the mean concentration for each sampling site for the two sampling periods, ensuring the independence of observations.

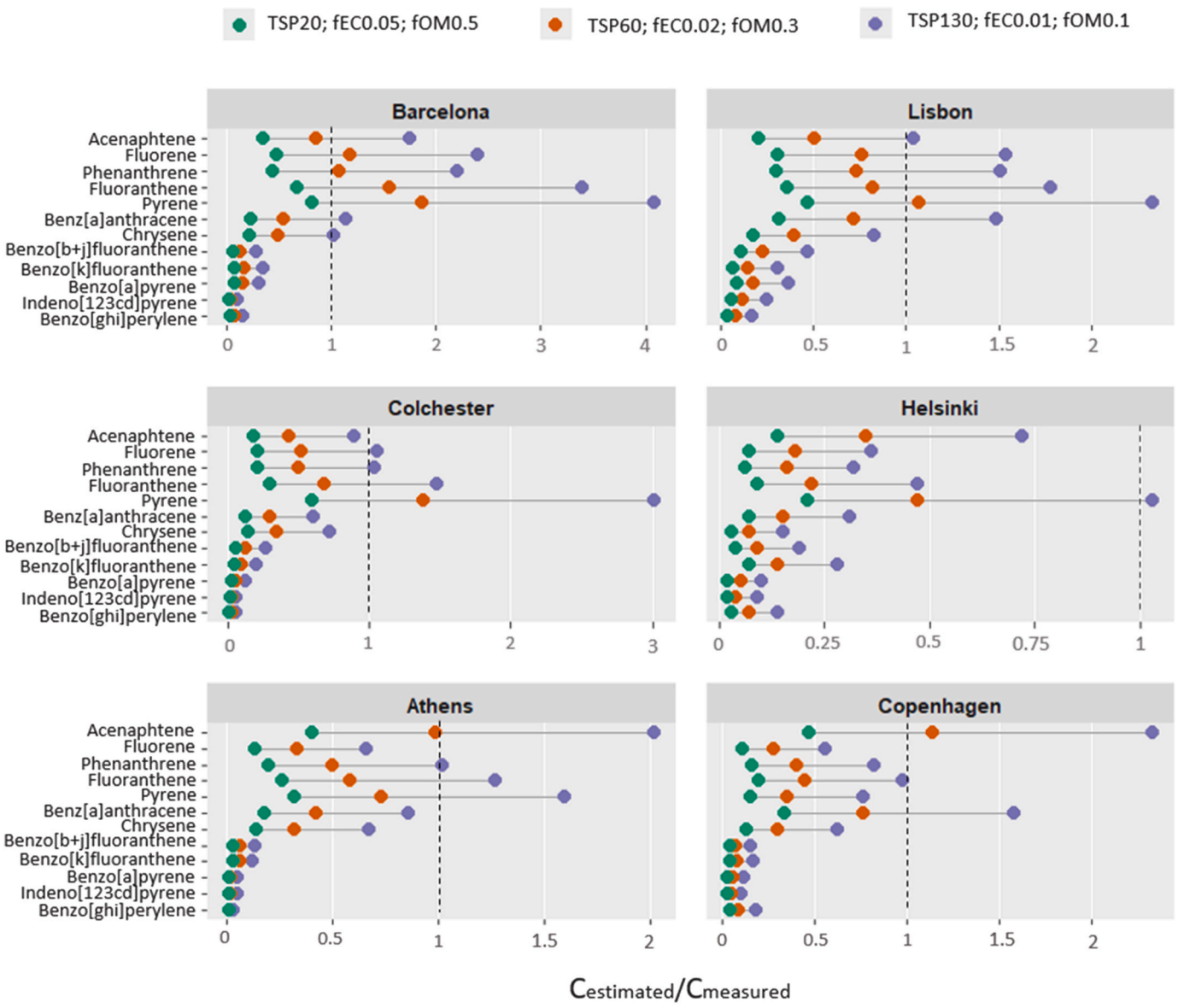

Fig. 5. Ratios between median estimated air PAH concentrations (Cestimated) and median measured air concentrations (Cmeasured) of individual PAHs in indoor air from each city and sampling sites applying a range of TSP, fEC and fOM. Estimated air concentrations are calculated from settled dust samples and model calculations (equations (1)–(3)) resulting in an intermediate ratio using TSP60; fEC0.02; fOM0.3 (orange dot), the lowest ratio using TSP20; fEC0.05; fOM0.5 (green dot), and the highest ratio using TSP130; fEC0.01; fOM0.1 (purple dot).

variation of 57 ± 10%. In comparison, varying fOM (with fixed fEC and TSP) and varying TSP (with fixed fEC and fOM) produced relative variations of 9 ± 6% and 8 ± 11%, respectively. The highest estimated concentrations were obtained using the lowest fEC values, consistent with the predominance of LMW-PAHs in the gas phase and their strong dependence on gas-particle partitioning. In contrast, the estimated air concentrations of HMW-PAHs, such as benzo[a]pyrene, were less influenced by variations in fEC (RSD = 6 ± 6%) and fOM (RSD = 4 ± 3%). Instead, these less-volatile, particle-bound compounds are predominantly controlled by the TSP variation (RSD = 70 ± 8%).

Generally, there was an underestimation in all samples regarding the estimated PAH air concentrations based on settled dust data compared to the ones measured with active air sampling technique (Fig. 5, Fig. S9), which may, besides the parameter values mentioned before, also reflect differences in the temporal representativeness of the two sampling approaches. Consequently, active sampling may capture short-term episodic emission events that result in elevated PAH concentrations, although it may also miss these events that take place outside the sampling period. In contrast, settled dust collected on quartz filters represents the long-term integrated accumulation of particulate matter. On the other hand, there is an important difference in the ratios between LMW-PAHs, which are pre-dominantly present in the gas-phase, and the HMW-PAH, that are linked to the particle-phase. As observed in previous studies, dust consists of a heterogeneous mixture of particulate matter, including carbonaceous and mineral particles, fibers, biological material such as tissue and pollen, and consumer product particles and substances (Amato et al., 2014; Fayad-Martinez et al., 2025; Praveena et al., 2015). Notably, a recent study in schools in Barcelona, showed that fibers contributed to 84 ± 10% of the TSP (Torres-Agullo et al., 2026). Not all of these particles carry PAHs, or are formed alongside PAHs, such as elemental carbon. Especially the HMW-PAH are linked to these carbonaceous particles after combustion emissions due to their high sorptive affinity, while LMW-PAHs may undergo gas-particle partitioning, including adsorption to these other materials (Femi-Oloye et al., 2024; Jonsson et al., 2007; Lohmann and Lammel, 2004). This may lead to lower estimated HMW-PAH concentrations, due to a

dilution effect of additional dust particles, while LMW-PAH may be compensated by gas-particle partitioning. Although additional variables and parameters, such as detailed dust particle properties, as well as the photochemical stability of PAHs in dust (Chimjarn et al., 2021; Ravindra et al., 2008) are missing in this study, the use of quartz filters for passive sampling of settled dust is a promising and non-invasive method for the estimation of air PAH concentration in the gas phase.

3.4. Daily dose and ECR for school-age children exposed to indoor levels of PAHs

The average daily inhalation dose and excess cancer risk (ECR) were calculated based on benzo[a]pyrene equivalents (Section 2.5) obtained from the results of the active sampling technique. As PAH concentrations in residential environments were usually higher than those measured in schools, greater inhalation doses were calculated for the homes, nearly double those estimated for schools (Table 1). In addition, doses in homes showed greater variability, suggesting that differences in indoor activities and cleaning/ventilation practices among homes influence indoor PAH levels. Although air PAH concentrations in sports halls occasionally exceeded or were comparable to those measured in classrooms (Table S5), likely due to lower ventilation frequency and the accumulation of PAHs in indoor dust and air, the corresponding inhalation dose was the lowest in these microenvironments. This is attributable to the limited exposure duration assumed for these spaces, as children were estimated to spend no more than 1 h per day there during school activities.

The estimated ECR_5 associated with inhalation exposure to PAHs across all sampling locations for a 5-year exposure period ranged approximately between 10^{-8} and 10^{-5} (Fig. 6). According to U.S EPA guidelines, acceptable risk levels fall within the range of 10^{-6} and 10^{-4}, while risks below 10^{-6} are typically considered negligible (U.S. Environmental Protection Agency, 1990). More specifically, the latter implies that an estimated cancer risk of 10^{-6} corresponds to one cancer case per one million individuals and is considered to pose minimal concern to public health. The ECR_5 values are presented relative to the USEPA benchmark as a reference for interpreting the magnitude of the estimated risk within a well-established cancer risk assessment framework and for facilitating comparison with previous studies applying the same methodology. As the ECR_5 estimates were derived using the USEPA cancer slope factor and exposure assessment parameters, with the exposure duration adapted to represent the defined 5-year primary school exposure period, the benchmark is used here as a contextual reference, rather than as a directly applicable lifetime-risk limit for the 5-year ECR_5 estimates. In this study, all estimated cancer risks were well below the upper limit of 10^{-4}, indicating a low carcinogenic risk associated with children's exposure to PAHs in the monitored sites. Homes generally exhibited higher ECR_5 values compared to schools and sports halls in most of the participating cities, reflecting the higher exposure time in these settings combined with higher observed indoor PAH concentrations. However, variability between cities is evident with some homes in Athens and Colchester showing comparatively higher values (10^{-6}-10^{-5}). Previous studies conducted in other European cities have reported slightly higher ECR levels in indoor environments of schools and homes (Lovrić et al., 2024; Pacitto et al., 2018). Nevertheless, these values remain comparable to those observed in the present study and did not exceed the maximum target value for PAH-related cancer risk. However, PAH exposure has been associated with a range of adverse health effects beyond carcinogenicity, including respiratory, cardiovascular, and developmental outcomes, which were not evaluated in this study. Therefore, the low cancer risk estimates reported here should not be interpreted as an absence of potential health concerns associated with PAH exposure. Moreover, the ECR_5 estimates should be interpreted with caution as uncertainties associated with analytical limitations, spatial and temporal variability, and the use of default exposure parameters may influence the absolute risk estimates.

3.5. Limitations

This study required a large, coordinated effort among six research teams across Europe, which inevitably introduced practical challenges. Although the original objective was to conduct sampling during both heating and non-heating periods, this was not always possible due to site-specific climatic conditions and scheduling constraints (Fig. S1). Moreover, the sampling approach involved the placement of equipment in occupied indoor environments, particularly in school classrooms and private homes, which made recruitment challenging. Although, sampling sites were selected to capture a range of urban settings in each city, the PAH concentration dataset cannot be considered fully representative of all indoor environments within each metropolitan area due to the limited number of sampling locations that could be included. Therefore, extrapolation of the findings to city level should be done with caution. Despite these constraints, the results provide a valuable overview of children's potential exposure to PAHs across different indoor microenvironments and European cities, supported by the implementation of standardized protocols and quality control procedures across all sites to

Table 1
Daily inhalation dose of PAHs (mean, median, maximum, minimum) and standard deviation expressed as benzo[a]pyrene equivalent concentration (δBaP_{eq}) across different indoor microenvironments in the different cities.

δBaP_{eq} (ng/day)							
City	Site	Number of samples	mean	SD	median	maximum	minimum
Barcelona	School	10	0.25	0.13	0.26	0.5	0.07
	Sports Hall	2	0.1	0.05	0.1	0.13	0.06
	Home	10	0.41	0.26	0.33	1.08	0.21
Lisbon	School	10	0.32	0.21	0.28	0.68	0.09
	Sports Hall	2	0.06	0.004	0.06	0.06	0.05
	Home	10	0.39	0.35	0.32	1.17	0.07
Colchester	School	10	1.43	1.66	0.8	5.49	0.19
	Sports Hall	1	0.07	-	-	-	-
	Home	10	2.4	2.33	1.39	6.33	0.36
Helsinki	School	10	0.7	0.21	0.63	1.21	0.44
	Sports Hall	2	0.1	0.06	0.1	0.14	0.06
	Home	6	1.1	0.6	1	2.25	0.47
Athens	School	10	0.65	0.34	0.61	1.33	0.17
	Sports Hall	2	0.27	0.2	0.27	0.41	0.13
	Home	10	3.31	4.55	1.46	15.21	0.3
Copenhagen	School	10	0.64	0.5	0.54	1.76	0.09
	Sports Hall	2	0.08	0.06	0.08	0.12	0.04
	Home	10	0.72	0.61	0.53	1.76	0.13

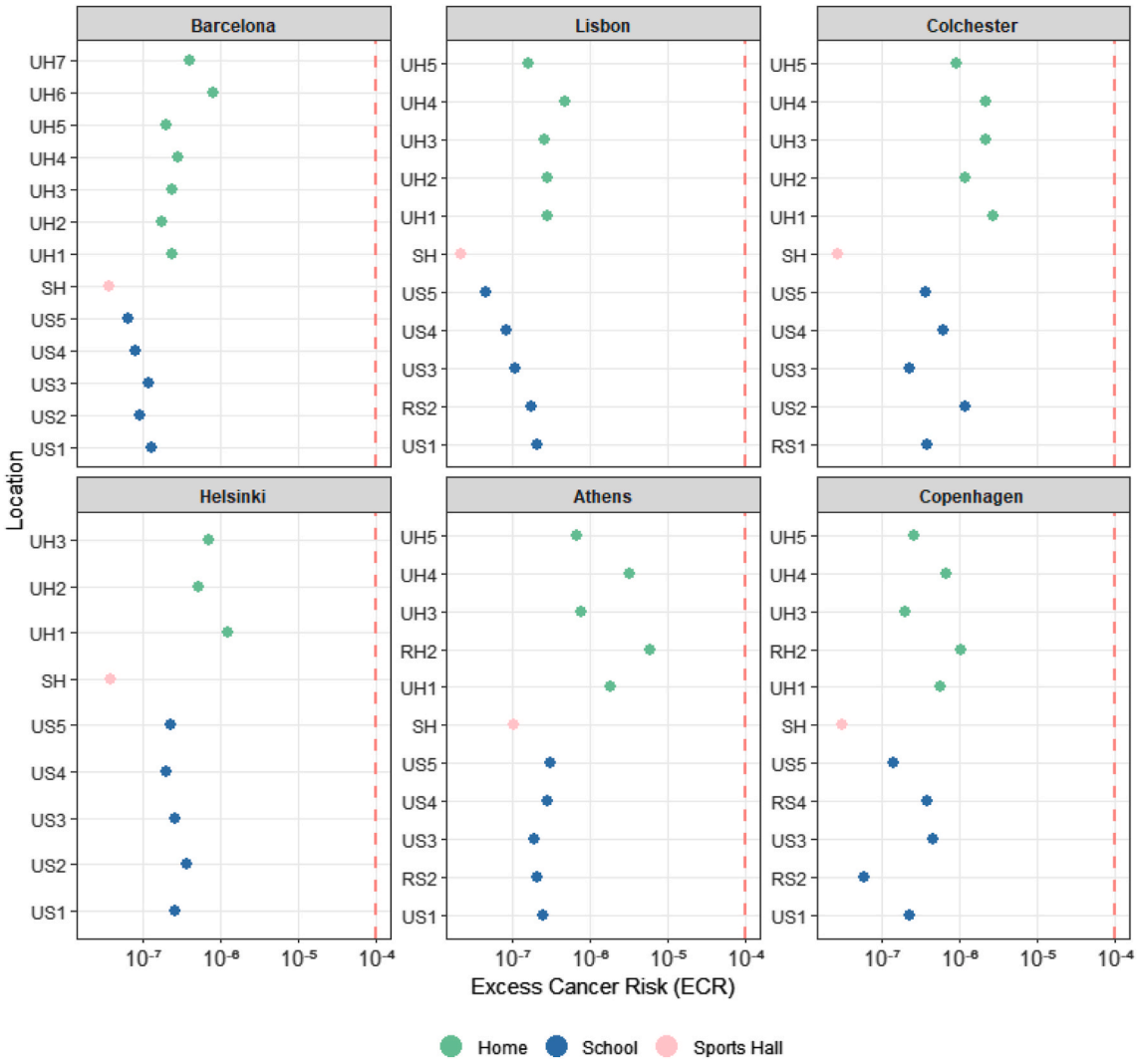

Fig. 6. Excess Cancer Risk (ECR) for a 5-year exposure period in each site across the six cities. Each point represents a specific sampling location shown on the y-axis and colors to distinguish homes (H), schools (S) and sports halls (SH). The red dashed vertical line indicates the USEPA risk level (10^{-4}), shown as a contextual reference within the USEPA cancer risk assessment framework.

ensure data comparability and reliability. Naphthalene, anthracene and acenaphthylene were omitted from the results because of blank issues in the analyses. This likely resulted in underestimations of total PAH and LMW-PAH concentrations and the health risk assessment, due to the missing contributions of these compounds.

4. Conclusions

Overall, the observed spatial variability in indoor PAH concentrations among the different European cities reflects the combined influence of emission sources, climatic conditions and indoor characteristics. Locations with indoor combustion sources or proximity to biomass burning such as the schools and homes in Colchester consistently exhibited the highest concentrations, regardless of urban classification. This suggests that outdoor infiltration and residential wood burning, might be as important as traffic emissions in influencing indoor PAH concentrations in some urban areas. A similar pattern was observed also in Athens in specific locations in the heating-period. In the other cities, this source is less remarkable, resulting in moderate PAH concentrations in Helsinki and Copenhagen, and low concentrations in Barcelona and Lisbon. LMW-PAHs consistently dominated the overall PAH profile due to their physicochemical properties which favor their presence in the gas phase and facilitate atmospheric spread. Indoor factors such as cleaning and ventilation practices also influenced indoor PAH concentrations, with ventilation emerging as a key factor in reducing pollutant accumulation indoors. Moreover, passive sampling of dust, as a low-maintenance and less intrusive alternative to active monitoring, showed some potential for calculating indoor air concentrations of more volatile, gas-phase LMW-PAHs.

Despite ongoing mitigation efforts across European cities, including restrictions and regulations on residential wood burning and vehicular traffic emissions, the present findings highlight the continued need for proactive strategies to reduce indoor PAH exposures, particularly in microenvironments where children spend most of their time. Establishing stronger and coordinated regulatory frameworks for indoor air quality in educational settings, together with improved ventilation practices and regular cleaning routines is essential for minimizing exposure and promoting healthier indoor environments for children. Lastly, as the relative contribution of PAH emission sources varies among European cities, effective mitigation strategies should integrate source-specific outdoor emission control policies with practical indoor measures, recognizing that both are complementary approaches to reducing children's overall exposure to PAHs.

CRediT authorship contribution statement

Maria A. Aretaki: Writing – review & editing, Writing – original draft, Visualization, Methodology, Investigation, Data curation, Conceptualization. **Judith Desmet:** Writing – review & editing, Methodology. **Angeliki Karanasiou:** Writing – review & editing, Project administration, Funding acquisition. **Mar Viana:** Writing – review & editing, Supervision, Funding acquisition, Conceptualization. **Raquel Pimenta:** Writing – review & editing, Investigation. **Ketlyn Oliveira:** Writing – review & editing, Investigation. **Susana Marta Almeida:** Writing – review & editing, Resources, Project administration, Investigation, Funding acquisition. **Stavroula Katsikari:** Writing – review & editing, Investigation. **Evangelia Diapouli:** Writing – review & editing, Resources, Project administration, Investigation, Funding acquisition. **Alex J. Dumbrell:** Writing – review & editing, Funding acquisition. **Ian Colbeck:** Writing – review & editing, Funding acquisition. **Corinne Whitby:** Writing – review & editing, Investigation, Funding acquisition. **Drew K. Henderson:** Writing – review & editing, Investigation. **Robert M.W. Ferguson:** Writing – review & editing, Resources, Project administration, Investigation, Funding acquisition. **Paula Guedes:** Writing – review & editing, Methodology, Investigation. **Katrin Vorkamp:** Writing – review & editing, Resources, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization. **Emmanuelle Castagnoli:** Writing – review & editing, Resources, Investigation. **Heidi Salonen:** Writing – review & editing, Project administration, Funding acquisition. **Barend L. van Drooge:** Writing – review & editing, Supervision, Resources, Methodology, Investigation, Conceptualization.

Disclaimer

Views and opinions expressed are those of the author(s) only and do not necessarily reflect those of the European Union or of the European Health and Digital Executive Agency (HaDEA). Neither the European Union nor the granting authorities can be held responsible for them.

Ethical approval

In Barcelona, approval for the sampling campaign was obtained from the Medical Research Ethics Committee of Parc de Salut Mar (CEImPSMar) (reference number: 2023/10915/I). In the UK Ethical approval was gained to conduct Sampling in homes from an institutional ethics review committee via the University of Essex Ethics Review and Management System (ERAMS) (ETH2223-1643). In Athens, ethical approval was obtained from the Research Ethics Committee of NCSR ″Demokritos″ (Protocol No. 39/10/9/2024). In Lisbon, ethical approval was obtained from Instituto Superior Técnico (reference number: 39/10/9/2024). In Denmark, the Regional Ethical Committee decided that no ethical approval was needed for the study. In Helsinki, the home sampling was carried out at volunteer staff members' homes. The data of the homes were filed under house numbers and there has been no key to link the house number to specific volunteers. Sampling in schools and sports halls (Helsinki, Finland) was permitted by the city of Helsinki under research permit number HEL 2023-016286.

Declaration of generative AI use

The authors used AI-assisted language tools to improve the coherence of the text. After using these tools, the authors carefully reviewed and edited the content as needed and take full responsibility for the content of the published article.

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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.envres.2026.125643.

Data availability

Data will be made available on request.

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