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**Anoushka Chatterjee, László Pál, Szabolcs Lovas, Martin McKee, Judit Diószegi, Nóra Kovács & Sándor Szűcs**

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# Assessment of health risks from exposure to indoor volatile organic compounds in European educational buildings

Anoushka Chatterjee[a*], László Pál[a*], Szabolcs Lovas[a], Martin McKee[b], Judit Diószegi[a], Nóra Kovács[a], Sándor Szűcs[a]

[a]Department of Public Health and Epidemiology, Faculty of Medicine, University of Debrecen, Debrecen, Hungary; [b]Department of Health Services Research and Policy, London School of Hygiene and Tropical Medicine, London, United Kingdom

*These authors are equal contributors.

E-mail addresses: anoushkachatt@gmail.com (A. Chatterjee), pal.laszlo@med.unideb.hu (L. Pál), lovas.szabolcs@med.unideb.hu (Sz. Lovas), martin.mckee@lshtm.ac.uk (M. McKee), dioszegi.judit@med.unideb.hu (J. Diószegi) kovacs.nora@med.unideb.hu (N. Kovács), szucs.sandor@med.unideb.hu (S. Szűcs)

*Please address all correspondence to: Dr. László Pál, Department of Public Health and Epidemiology, Faculty of Medicine, University of Debrecen, H–4012 Debrecen, P.O. Box 9, Hungary. (T) 36 52 512 765 (F) 36 52 417 267 (E) pal.laszlo@med.unideb.hu

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## Abstract

Volatile organic compounds (VOCs) comprise an important group of indoor air pollutants, commonly found in building materials and consumer products. Due to their low boiling points, VOCs are prevalent in indoor environments, with concentrations in homes, schools, and offices often two to five times higher than outdoors. Chronic exposure to VOCs is linked to a range of adverse health outcomes, including respiratory, neurological, cardiovascular damage, and an increased cancer risk. Children and adolescents who spend a significant amount of time in educational buildings are particularly vulnerable to these effects. Therefore, this study aimed to estimate the related health risks to those in day care centres, schools, high-schools and universities across 17 member states of the European Union (EU) by utilizing a previously published dataset and collecting data on levels of 9 VOCs. Health risks were assessed using the World Health Organization's Indoor Air Quality Risk Calculator. Point of departure indices (PODIs) and cancer risks were calculated, with values above 1.0 and 10 cases/1 million population respectively indicating an increased health risk. Our results showed that formaldehyde exposure poses an increased risk of respiratory, neurological and carcinogenic effects in educational buildings in 14 of the countries studied. Additionally, neurological risks from exposure to benzene were above the limit in 4 EU countries. These findings indicate the need to reduce formaldehyde and benzene concentrations in European educational buildings to protect the health of the next generation.

## 1. Introduction

Indoor air pollution has been identified as a significant public health problem worldwide, with a non-negligible contribution to the overall burden of disease [1,2]. Survey data indicate that people spend approximately 90% of their time indoors, although where they spend their time, in homes, offices, schools, and other public and private buildings varying by age, employment, and other characteristics [3,4]. Globally, the World Health Organization (WHO) estimates that 3.2 million people die prematurely each year from diseases related to exposure to a range of indoor air pollutants [5]. To inform measures to reduce these risks, the WHO has established guideline values for indoor air quality (IAQ) for individual chemical air pollutants [6,7]. Most of the substances listed in the guideline are organic compounds, an epidemiologically important class of hazardous indoor air pollutants that can cause both acute and chronic health effects [1,6-8]. The WHO classifies these chemicals as very volatile organic compounds (VVOCs), volatile organic compounds (VOCs) and semi-volatile organic compounds (SVOCs) characterised by boiling points ranging from below 0 °C to 50–100 °C, from 50–100 °C to 240–260 °C, and from 240–260 °C to 380–400 °C, respectively [9,10]. Aldehydes (formaldehyde, acetaldehyde), aromatic hydrocarbons (benzene, ethylbenzene, toluene, xylene), chlorinated hydrocarbons (trichloroethylene, tetrachloroethylene) and esters (n-butyl acetate) are among the VOCs most frequently detected indoors [6,7]. Their low boiling point and ubiquitous presence, in many industrial and consumer products including building materials, furniture, carpets, dyes, air fresheners, paint solvents, adhesives, household cleaning products, cosmetics, and electronic equipment [11-13], give rise to concentrations in homes, schools, and offices that are 2 to 5 times higher than outdoors [4,14,15]. In addition, some of the VOCs found indoors come from outside, especially from road traffic emissions [13–15]. This wide variety of sources and differences in building characteristics mean that people are often exposed indoors to a complex mixture of harmful VOCs [16–18].

The problem has been exacerbated in recent years as efforts to improve energy efficiency have led to buildings becoming increasingly sealed from the external environment, thereby reducing heating and cooling costs [19,20]. As a result, many buildings now rely solely on mechanical ventilation systems that recirculate indoor air with minimal addition exchange of fresh air [19,20], thereby contributing to their accumulation.

Acute exposure to VOCs can result in several adverse effects, including irritation of the eyes, nose, and throat, due to their interaction with mucous membranes and sensory receptors [21,22]. They can also react with mucin glycoproteins, causing IgE-mediated inflammation and epithelial damage to the respiratory tract [23]. This can lead to painful breathing and can exacerbate asthma, and cause damage to the cardiovascular and central nervous systems [23,24]. Associated symptoms include headache, dizziness, and fatigue [23,25–28].

Long-term exposure to VOCs has been shown to damage the respiratory system resulting in chronic bronchitis, reduced lung function, and progression of asthma [11,21,25,29]. As VOCs are lipid soluble, they cross the blood-brain barrier [23,30] and can influence neurotransmitter functions and induce oxidative stress in neural tissues. In these ways, chronic VOC exposure can impair cognitive functions, memory, and peripheral nerve signalling [21,23,30]. Metabolites of VOCs have been reported to cause cardiovascular injury [24,27], promoting endothelial dysfunction, hypertension, and atherosclerosis, and increasing the risk of cardiovascular diseases [24,27]. Chronic exposure to specific VOCs has also been identified as a risk factor for various cancers [21–23]. For example, benzene, formaldehyde and trichloroethylene have been classified by the International Agency for Research on Cancer (IARC) as carcinogenic to humans (Group 1), associated with increased incidence of myeloid leukaemia, nasopharyngeal and kidney cancer, respectively [31-33].

Children and adolescents are particularly vulnerable to the health hazards associated with chronic exposure to VOCs [34,35]. Their respiratory and immune systems are still developing, making them less able to metabolise and eliminate toxic substances [34,35]. In addition, they have a higher respiratory rate relative to their bodyweight, which can lead to increased inhalation of air pollutants [34,35]. The amount of time they spend in educational buildings, typically 6 hours a day, 5 days a week for many months of the year, places them at increased risk from indoor air pollutants such as VOCs, potentially leading to respiratory problems, allergic diseases, and long-term developmental complications [34–36]. Yet despite this growing body of evidence on health risks associated with both short- and long-term exposure these pollutants [21,22], exposure indoors has attracted less attention than their presence in the ambient air [21].

The potential risks are especially great in educational facilities. Like any workplace, day care centre, elementary school, high-school and university buildings contain VOC-emitting items, such as carpets and furniture. However, they have an increased likelihood of having products like adhesives, paints, and cleaning agents, all important sources of VOCs [37]. Inadequate ventilation in classrooms can exacerbate this problem, allowing pollutants to accumulate over time [37]. For these reasons, this study investigates and quantifies the health risk in these settings by collecting data on VOC exposure levels in European day care centres, elementary schools, high-schools and university buildings. To our knowledge, this is the first study that used the Indoor Air Quality (IAQ) Risk Calculator, a screening tool developed by the WHO Regional Office for Europe and the European Centre for Environment and Health, to estimate the health risks associated with exposure to indoor air pollutants.

## 2. Data and methods

We extracted data on VOC levels in educational buildings in EU member states from our previously published dataset on VOC concentrations in indoor environments of offices, educational buildings and residential buildings in the European Union between 2010 and 2023 [38]. Records with data on levels of VOCs in day-care centres, kindergartens, elementary and high schools, and universities were selected to prepare a separate database for health risk assessment.

### 2.1 Development of a database for health risk assessment

Data were extracted from 28 articles and used to generate a comprehensive database of VOC concentrations in educational buildings. The database included information on the authors, the titles of the articles, the concentrations of VOCs, and the type and number of buildings for countries in the European Union. Our database contained data from 17 European countries: Croatia, Cyprus, the Czech Republic, Finland, France, Germany, Greece, Hungary, Ireland, Italy, the Netherlands, Poland, Portugal, Romania, Slovenia, Spain, and Sweden. The 9 VOCs, included formaldehyde, acetaldehyde, benzene, ethylbenzene, o, m-, and p-xylenes, styrene, toluene, 1,4-dichlorobenzene, and trichloroethylene. When data on the level of VOCs was reported from multiple buildings within the same study, the mean value was calculated and entered into the database.

### 2.2 Health risk assessment

The IAQ Risk Calculator software (Indoor Air Quality Risk Calculator, version 1.0.0.0; https://www.who.int/europe/tools-and-toolkits/indoor-air-quality-(iaq)-riskcalculator--assessing-risks-for-children-s-health-from-chemical-indoor-air-pollution) was used to estimate the health risks associated with indoor exposure to the selected VOCs [39]. This is a user-friendly tool that assesses health risks from combined exposure to indoor air pollutants in public places, such as schools, for children [39]. To achieve this, the tool incorporates tiers modified from those of the WHO framework for assessing exposure to multiple chemicals [39]. It includes risk calculation spreadsheets and a supporting database of guideline values containing information on points of departure (PODs) for inhalation for selected effects [39,40]. The terminology and approach of the screening tool are consistent with the International Programme on Chemical Safety (IPCS) and the WHO IPCS framework [41,42]. In assessing hazards, the tool uses modified early stages (stages 0 and 1) of the WHO IPCS framework [41]. The concentrations of various individual VOCs in EU countries, as reported in the database, were entered into the IAQ Risk Calculator, yielding article-specific results. The substances included in the supporting toxicological database for the screening tool are those for which co-exposure in indoor air in public settings is most likely, particularly among children [39,40]. The PODs included in this database involve the respiratory, nervous and cardiovascular systems, as well as carcinogenicity of IARC group 1 carcinogens [39,40].

The WHO IPCS risk assessment approach is based on a tiered assessment strategy [39,40]. Assessments start with relatively simple and often conservative evaluations in the early tiers and progress to more complex and refined estimations in subsequent tiers as deemed necessary [39,40]. This hierarchical system is designed to guide risk assessors through a systematic evaluation process [39,40]. Each tier represents an increasing level of complexity and refinement in both exposure and hazard assessment [39,40]. Tier 0 is the initial screening stage [39,40]. At this point, the assessment is typically based on limited information about the specific chemicals under consideration [39,40]. Exposure estimates in Tier 0 are primarily derived from the fundamental physicochemical properties of the substances and their potential exposure routes [39,40]. This provides an initial, rapid filter to identify substances that could plausibly give rise to concern [39,40]. At this stage, the software calculates a hazard index (HI) [39,40], defined as the sum of the hazard quotients (HQs), i.e., the level of exposure (the concentration of the substance in question) to each of the constituents in an assessment group (AG) divided by its respective reference concentration (RC). An AG is a set of substances that are evaluated together because they might affect the same organ or system in the body. The RC is the maximum safe level of that substance, based on the most sensitive health effect it can cause [39,40]. HI can be calculated using the following equation:

$$\mathrm{HI} = \sum_{x=1}^{n} \frac{\text{measured concentration}_x}{\mathrm{RC}^x}$$

*where*

*x = each substance included in the AG, irrespective of its health effects,*

*RC = reference concentration for inhalation based on critical effect.*

To calculate HI, we used the most recent reference concentrations from the WHO and the Agency for Toxic Substances and Disease Registry

(ATSDR) [39,40]. A HI greater than 1 requires a refined risk assessment or consideration of corrective measures to reduce exposure [39,40]. Tier 1, Level 1 is applied to those exposures that pass the Tier 0 screening and involves a more detailed assessment of exposure and hazard [39,40]. At this stage, the evaluation considers information on chemicals that goes beyond their basic physicochemical properties [39,40]. Tier 1 exposure estimates are usually derived from exposure modelling results [39,40]. This information may include details on the exposed population, exposure routes, the environmental fate of the substances, production volumes, and average values from air quality monitoring databases [39,40]. In this case, the software calculates the hazard index assessment group ($HI_{ag}$) in the same way as for Tier 0 but categorises chemicals based on five selected adverse effects of indoor air pollution, including respiratory, cardiovascular, neurological, irritative, and carcinogenic effects [39,40].

$HI_{ag}$ can be calculated according to the following equation:

$$HI_{ag} = \sum_{x=1}^{n} \frac{\text{measured concentration}_x}{RC^x}$$

*where*

*ag = substances grouped for evaluation for one of the designated priority health effects,*

*x = each substance included in the assessment group,*

*RC = reference concentration for inhalation for the relevant substance, for example, WHO guidance values.*

The $HI_{ag}$ was calculated using the same reference concentrations as in Tier 0. At Tier 2, the software calculates an adjusted point of departure index

($PODI_{adj}$) for the selected effects of interest, which include respiratory, cardiovascular, neurological, and irritative effects [39,40]. The $PODI_{adj}$ is based on the lowest point of departure (POD) [39,40]. The POD is defined as the dose or concentration chosen as the point of comparison for exposure estimates and serves as a basis for risk assessment [39,40]. The POD can be the "no observed adverse effect level" (NOAEL), the "lowest observed adverse effect level" (LOAEL), or the benchmark dose or benchmark concentration [39,40]. When data are available from both human studies and animal experiments, the lowest dose or concentration is used to calculate the $PODI_{adj}$ [39,40]. If data are only available from animal studies, the POD values are divided by the conventional default uncertainty factor (UF) of 10 to account for possible differences between humans and other species [39,40]. The NOAEL is used in calculations wherever possible [39,40]. If only the LOAEL is available, the screening tool divides this value by three [39,40]. Human variability is accounted for by dividing the POD by ten [39,40]. For systemic effects, PODs determined for chronic exposure are preferred [39,40]. If these values are unavailable, the shorter exposure duration is considered by adjusting the POD values by a factor of two [39,40].

$PODI_{adjag}$ can be calculated using the following equation:

$$\mathrm{PODI}_{adjag} = \sum_{x=1}^{n} \frac{\text{measured concentration}_x}{\mathrm{PODxadj}}$$

*where*

*$PODI_{adjag}$ = the PODI adjusted for an assessment group for each of the priority effects,*

*PODx adjusted = the POD for the relevant priority health effect for substance x adjusted by period of exposure (e.g., intermediate vs. chronic) and an acceptable margin to account for uncertainty.*

If the $\mathrm{PODI}_{adj}$ is greater than 1, the assessment should be refined or corrective measures should be considered to reduce exposure [39,40]. In our risk assessment, the $\mathrm{PODI}_{adj}$ was calculated using the lowest POD value obtained from epidemiological or animal studies [39,40].

Some of the chemicals in the IAQ database are classified as IARC Group 1 carcinogens [39,40]. For these compounds, the software also calculates the $\mathrm{PODI}_{adjcancer}$ value based on the tumorigenic concentration of 50 (TC50), which is the concentration associated with a 50% increase in cancer risk [39,40]. If the TC50 values are from animal experiments or human studies, they are divided by 50,000 or 5,000, respectively [39,40]. This gives a cancer risk of 10 in $10^6$ (ten cancer case per 1 million population). These values align with the limits established by the European Food Safety Authority as low-priority for the risk management of carcinogenic and genotoxic substances [43].

$\mathrm{PODI}_{adjcancer}$ can be calculated using the following equation:

$$\mathrm{PODI}_{adjcancer} = \sum_{x=1}^{n} \frac{\text{measured concentration}_x}{\text{TC50xadj}}$$

*where*

*$\mathrm{PODI}_{adjcancer}$ = the PODI adjusted for an assessment group for cancer,*

*TC50x = the concentration associated with a 50% increase in cancer risk, adjusted by an acceptable margin.*

If the $\text{PODI}_{\text{adjcancer}}$ is more than 10 cases per 1 million population, the assessment should be refined or remedial measures should be taken to reduce exposure [39,40]. Using the separate database prepared from our previously published dataset, the VOC concentrations measured in EU countries were entered into the IAQ Risk Calculator and used to calculate the PODI values for irritation, as well as for respiratory, neurological, cardiovascular and carcinogenic effects [39,40]. The results are presented in Tables 1 and 2.

## 3. Results

For 7 of the 9 VOCs examined, the $\text{PODI}_{\text{adj}}$ values for respiratory, cardiovascular, neurological, and carcinogenic effects were less than one (see data in Supplement 1). The remaining two were formaldehyde and benzene and, in their cases, the calculated $\text{PODI}_{\text{adj}}$ values for respiratory and neurological effects exceeded one in several cases. Furthermore, in multiple instances, the cancer risk associated with exposure to formaldehyde equalled or exceeded 10 cases per 1 million population. As exposure to formaldehyde and benzene posed the highest health risk, the results obtained for these VOCs are presented in Tables 1 and 2.

### 3.1 Health risks attributable to indoor benzene exposure

The PODIadj values for respiratory, neurological, and irritation effects due to indoor exposure to benzene are presented in Table 1. As can be seen, the PODIadj values for respiratory and irritation effects were low, ranging from 0.0001 to 0.0031 in both categories. In addition, the PODIadj values for neurological effects from benzene exposure varied between 0.0747 and 2.6265, with values greater than 1 indicating an increased health risk in 3, 6, 21 and 11 school buildings in Germany, Greece, Hungary, and Italy, respectively. In all the studies included in our research, the cancer risk from exposure to benzene in educational buildings was below the maximum acceptable level of 10 cases of cancer per 1 million population.

**Table 1: $PODI_{adj}$ values for respiratory, neurological, and irritation effects due to indoor benzene exposure in educational buildings in the member states of European Union**

| benzene exposure | | | | | | |
|---|---|---|---|---|---|---|
| **author** | **country** | **number of buildings studied** | **mean concentration [mg/m$^3$]** | **$PODI_{adj}$ respiratory effects** | **$PODI_{adj}$ neurological effects** | **$PODI_{adj}$ irritation effects** |
| Brdarić et al., 2019 [44] | Croatia | 2 | 0.00104 | 0.0003 | 0.2506 | 0.0003 |
| Geiss et al., 2011 [45] | Cyprus | 3 | 0.00307 | 0.0009 | 0.7398 | 0.0009 |
| Szabados et al., 2021 [46] | Czechia | 12 | 0.00347 | 0.0010 | 0.8361 | 0.0010 |
| Geiss et al., 2011 [45] | Finland | 3 | 0.00115 | 0.0003 | 0.2771 | 0.0003 |
| Canha et al., 2015 [47] | France | 17 | 0.00210 | 0.0006 | 0.5060 | 0.0006 |
| Ramalho et al., 2015 [48] | France | 310 | 0.00232 | 0.0007 | 0.5590 | 0.0007 |
| Verriele et al., 2015 [49] | France | 10 | 0.00130 | 0.0004 | 0.3133 | 0.0004 |
| Geiss et al., 2011 [45] | Germany | 3 | 0.00622 | 0.0018 | **1.4988** | 0.0018 |
| Geiss et al., 2011 [45] | Greece | 6 | 0.00533 | 0.0015 | **1.2843** | 0.0015 |
| Geiss et al., 2011 [45] | Hungary | 6 | 0.00573 | 0.0016 | **1.3807** | 0.0016 |
| Szabados et al., 2021 [46] | Hungary | 15 | 0.00461 | 0.0013 | **1.1108** | 0.0013 |
| Geiss et al., 2011 [45] | Ireland | 2 | 0.00358 | 0.0010 | 0.8627 | 0.0010 |
| Geiss et al., 2011 [45] | Italy | 3 | 0.00320 | 0.0009 | 0.7711 | 0.0009 |
| Ielpo et al., 2021 [50] | Italy | 1 | 0.00194 | 0.0006 | 0.4675 | 0.0006 |
| Lucialli et al.,2020 [51] | Italy | 8 | 0.00153 | 0.0004 | 0.3687 | 0.0004 |
| Marzocca et al.et al., 2017 [52] | Italy | 1 | 0.00043 | 0.0001 | 0.1036 | 0.0001 |
| Romagnoli et al., 2015 [53] | Italy | 1 | 0.00092 | 0.0003 | 0.2217 | 0.0003 |
| Szabados et al, 2021 [46] | Italy | 11 | 0.01090 | 0.0031 | **2.6265** | 0.0031 |
| Geiss et al., 2011 [45] | Netherlands | 2 | 0.00170 | 0.0005 | 0.4096 | 0.0005 |
| Mainka et al., 2015 [54] | Poland | 2 | 0.00232 | 0.0007 | 0.5590 | 0.0007 |
| Szabados et al., 2021[46] | Poland | 11 | 0.00306 | 0.0009 | 0.7373 | 0.0009 |
| Pegas et al., 2011 [55] | Portugal | 14 | 0.00069 | 0.0002 | 0.1663 | 0.0002 |
| Fonseca et al., 2021 [56] | Portugal | 20 | 0.00100 | 0.0003 | 0.2410 | 0.0003 |

| | | | | | | |
|---|---|---|---|---|---|---|
| Pegas et al., 2012 [57] | Portugal | 1 | 0.00031 | 0.0001 | 0.0747 | 0.0001 |
| Szabados et al, 2021[46] | Slovenia | 12 | 0.00415 | 0.0012 | 1.0000 | 0.0012 |
| Lizana et al., 2020 [58] | Spain | 6 | 0.00123 | 0.0004 | 0.2964 | 0.0004 |
| Ninyá et al., 2022 [59] | Spain | 1 | 0.00048 | 0.0001 | 0.1157 | 0.0001 |
| Vallecillos et al., 2020 [60] | Spain | 1 | 0.00041 | 0.0001 | 0.0988 | 0.0001 |
| Villanueva et al., 2018 [61] | Spain | 18 | 0.00053 | 0.0002 | 0.1277 | 0.0002 |

The cells with yellow colour indicate studies where there was an increased health risk. PODI$_{adj}$ values greater than 1 are shown in bold.

### 3.2 Health risks attributable to indoor formaldehyde exposure

The PODIadj values for respiratory, neurological, cardiovascular, and carcinogenic effects resulting from indoor formaldehyde exposure are presented in Table 2. As shown, the PODIadj values for respiratory effects ranged from 0.7398 to 7.3133. An increased risk of respiratory effects due to formaldehyde exposure was found in a varying number of educational buildings when analysing data obtained in Cyprus (n=3), Finland (n=3), France (n=313), Germany (n=3), Greece (n=6), Hungary (n=6), Ireland (n=2), Italy (n=3), the Netherlands (n=2), Portugal (n=111), Romania (n=5), Slovenia (n=12), Spain (n=18), and Sweden (n=23). Additionally, based on the concentration data collected in 5 school buildings in Portugal by Sá et al., 2019, the PODIadj values for neurological effects due to formaldehyde exposure were greater than one [62]. The cancer risk from formaldehyde exposure ranged from 10 to 110, exceeding the maximum acceptable level of 10 cancer cases per 1 million population in educational buildings in Cyprus (n=3), Finland (n=3), France (n=313), Germany (n=3), Greece (n=6), Hungary (n=6), Ireland (n=2), Italy (n=15), the Netherlands (n=2), Portugal (n=111), Romania (n=5), Slovenia (n=12), Spain (n=18), and Sweden (n=23).

**Table 2: $\mathrm{PODI}_{adj}$ values for respiratory, cardiovascular, neurological, and carcinogenic effects due to indoor formaldehyde exposure in educational buildings in the member states of European Union**

| formaldehyde exposure | | | | | | | |
|---|---|---|---|---|---|---|---|
| **author** | **country** | **number of buildings studied** | **mean concentration [$mg/m^3$]** | **$\mathrm{PODI}_{adj}$ respiratory effects** | **$\mathrm{PODI}_{adj}$ cardiovascular effects** | **$\mathrm{PODI}_{adj}$ neurological effects** | **cancer cases/1 million population** |
| Brdarić et al., 2019 [44] | Croatia | 2 | 0.00848 | 0.8653 | 0.0482 | 0.1381 | 10 |
| Geiss et al., 2011 [45] | Cyprus | 3 | 0.01200 | **1.2245** | 0.0682 | 0.1954 | **20** |
| Szabados et al., 2021 [46] | Czechia | 12 | 0.00779 | 0.7949 | 0.0443 | 0.1269 | 10 |
| Geiss et al., 2011 [45] | Finland | 3 | 0.01040 | **1.0612** | 0.0591 | 0.1694 | **20** |
| Hu et al., 2022 [63] | France | 3 | 0.01372 | **1.4000** | 0.078 | 0.2235 | **20** |
| Ramalho et al., 2015 [48] | France | 310 | 0.01873 | **1.9112** | 0.1064 | 0.3050 | **30** |
| Geiss et al., 2011 [45] | Germany | 3 | 0.03080 | **3.1429** | 0.175 | 0.5016 | **50** |
| Geiss et al., 2011 [45] | Greece | 6 | 0.01731 | **1.7663** | 0.0984 | 0.2819 | **30** |
| Geiss et al., 2011 [45] | Hungary | 6 | 0.01520 | **1.5510** | 0.0864 | 0.2476 | **20** |
| Szabados et al., 2021 [46] | Hungary | 16 | 0.00867 | 0.8847 | 0.0493 | 0.1412 | 10 |
| Geiss et al., 2011 [45] | Ireland | 2 | 0.01024 | **1.0449** | 0.0582 | 0.1668 | **20** |
| Szabados et al, 2021 [46] | Italy | 12 | 0.00980 | 1.0000 | 0.0557 | 0.1596 | **20** |
| Geiss et al., 2011 [45] | Italy | 3 | 0.01430 | **1.4592** | 0.0812 | 0.2329 | **20** |
| Geiss et al., 2011 [45] | Netherlands | 2 | 0.01393 | **1.4214** | 0.0791 | 0.2269 | **20** |
| Szabados et al, 2021[46] | Poland | 12 | 0.00773 | 0.7888 | 0.0439 | 0.1259 | 10 |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Branco et al., 2015 [64] | Portugal | 4 | 0.01640 | **1.6735** | 0.0932 | 0.2671 | **30** |
| Branco et al., 2019 [65] | Portugal | 25 | 0.01340 | **1.3673** | 0.0761 | 0.2182 | **20** |
| Ferreira et al., 2013 [66] | Portugal | 51 | 0.01837 | **1.8745** | 0.1044 | 0.2992 | **30** |
| Fonseca et al., 2021 [56] | Portugal | 20 | 0.01690 | **1.7245** | 0.096 | 0.2752 | **30** |
| Nunes et al., 2016 [67] | Portugal | 4 | 0.05157 | **5.2622** | 0.293 | 0.8399 | **80** |
| Oliviera et al., 2017 [68] | Portugal | 2 | 0.03500 | **3.5714** | 0.1989 | 0.5700 | **60** |
| Sá et al., 2019 [62] | Portugal | 5 | 0.07167 | **7.3133** | 0.4072 | **1.1673** | **110** |
| Neamtiu et al., 2019 [69] | Romania | 5 | 0.03416 | **3.4857** | 0.1941 | 0.5564 | **50** |
| Szabados et al, 2021[46] | Slovenia | 12 | 0.01150 | **1.1735** | 0.0653 | 0.1873 | **20** |
| Ninyá et al., 2022 [59] | Spain | 1 | 0.00883 | 0.9010 | 0.0502 | 0.1438 | 10 |
| Villanueva et al., 2018 [61] | Spain | 18 | 0.02699 | **2.7541** | 0.1534 | 0.4396 | **40** |
| Cabovská et al., 2022 [70] | Sweden | 23 | 0.01083 | **1.1051** | 0.0615 | 0.1764 | **20** |
| Wang et al., 2015 [71] | Sweden | 39 | 0.00725 | 0.7398 | 0.0412 | 0.1181 | 10 |

The cells with yellow colour indicate studies where there was an increased health risk. $\mathrm{PODI}_{adj}$ values greater than 1 are shown in bold.

## 4. Discussion

This study reveals a concerning pattern of indoor air pollution in educational buildings in 17 EU member states, with formaldehyde and benzene emerging as the most significant threats to children's health [35,37]. Using the WHO Indoor Air Quality Risk Calculator, we found that formaldehyde levels in schools, kindergartens, and universities exceeded thresholds for respiratory and carcinogenic effects in 14 countries. Benzene exposure, while below cancer risk thresholds, posed notable neurological risks in four countries. These findings are particularly troubling given the vulnerability of children and adolescents, who spend a substantial portion of their day during term times in these environments [3,35]. Their developing respiratory and immune systems, combined with higher inhalation rates, make them more susceptible to the harmful effects of VOCs. Taken together, this evidence points to an urgent need for a comprehensive response to protect the health of future generations [72-74].

The first step is to raise awareness of the sources of each substance. Benzene comes from indoor and outdoor sources, with significant spatial and seasonal variations [6,51,75]. The main indoor sources are building materials and furnishings, including varnishes, paints, adhesives, and urea-formaldehyde resins found in pressed wood products [6,51,75]. Floor covers have also been identified as major contributors to indoor benzene pollution [6]. A multinational study performed in Central Europe has found 2.1 times higher benzene concentrations in carpet covered Italian classrooms than in schools in other countries with alternative floor covers [46]. Therefore, it can be reasonable to assume that the elevated benzene levels were partially due to its release from synthetic carpets. Cleaning products and solvents can also contribute, so that levels may be higher in buildings that have recently undergone renovations or deep cleans [6,51,75]. Consumer products such as air fresheners, glues, and personal care items can release benzene either directly or through chemical reactions with indoor air components, such as ozone. [6,51,75]. Urban schools, especially if situated near high-volume traffic or industrial sites are at additional risk from vehicle exhaust and industrial emissions [6,49,73]. The relative importance of these sources can be seen from studies of the ratio of outdoor to indoor benzene concentrations, which is often less than 1.0 [76]. Seasonal factors can exacerbate exposure , as winter months bring higher levels due to reduced ventilation and prolonged classroom occupancy [9,77]. Benzene concentrations can be further increased by emissions from car parks, garages or idling vehicles near schools [6,75,77].

Formaldehyde comes mainly from building materials and furnishings [6,75,78], with concentrations influenced by the age of the building, ventilation efficiency, and environmental factors [6,75,78]. Primary sources include pressed wood products, particleboard, and plywood made with urea-formaldehyde resins, as well as carpets and flooring, which release formaldehyde vapour, especially when they are new [6,75,78].

Other significant sources include cleaning products containing terpenes that can react with ozone in the outdoor air to produce formaldehyde, as well as residual tobacco smoke contamination that increases formaldehyde levels in classrooms and car exhaust fumes entering school buildings located next to busy roads [6,32,75,78]. This may provide a potential explanation for the findings of Sá et al. (2019), which demonstrated that formaldehyde levels in schools in the urban area of Porto exceeded threshold values [62]. This indicates that outdoor air pollution may have a substantial impact on indoor formaldehyde levels. As with benzene, there are also seasonal variations, with higher concentrations in classrooms during the summer months [6,51,75,78]. An optimal response to poor indoor air quality has four elements. The first comprises regulatory measures. These include enforcement of compliance with WHO indoor air quality guidelines,[74] mandating pre-occupancy air quality testing in new or renovated educational buildings [6,71,73,76], and introduction of VOC emission limits for building materials and furnishings used in schools. For example, institutional policies should also restrict the use of cleaning products containing formaldehyde precursors and strictly enforce smoke-free policies [6,73,76] and replace benzene-containing products [70,71]. The second element includes measures related to infrastructure and design. Here, a first step is to eliminate or reduce the number of sources, by removing building materials emitting VOCs or requiring constructors to the use of low-emitting alternatives [80]. The next step involves the development of a comprehensive indoor air quality management plan (IAQMP) to prevent, identify and resolve indoor air quality problems in school buildings [80]. Although improving ventilation has been shown to reduce VOC levels in school environments, this can only be effective if it is integrated into the IAQMP [70,71,76,80]. Mechanical ventilation with heating, ventilation, and air conditioning (HVAC) systems can significantly reduce indoor benzene and formaldehyde concentrations [70,71,76,80]. These measures have the significant added benefit of reducing exposure to airborne respiratory viruses. Activated carbon filtration in air purification devices is also effective [79]. Where high levels of formaldehyde persist, specific abatement technologies can be used, such as formaldehyde-catalysed activated carbon filtration systems [79]. However, they should not be considered as a general method for solving indoor air quality problems due to their significant limitations [82]. For example, filtration systems are often pollutant-specific and cannot address many gaseous compounds or persistent SVOCs at the same time [82]. The effectiveness of such measures can be monitored using formaldehyde-specific detectors, which will be especially useful in areas with limited air circulation [6,75,78]. The third element involves education and awareness raising. Regulations will only work if people understand why they have been adopted so specific and targeted training of educate school staff and students about sources of benzene, emphasising the risks associated with indoor smoking (which should never happen in an educational establishment anyway) and the improper storage of solvents and other chemicals containing benzene [72,73]. The final element is monitoring and evaluation. This calls for national or regional programmes for regular indoor air quality monitoring in educational facilities, coupled with public databases to track VOC levels and remediation efforts and research into long-term health effects and effectiveness of interventions to reduce exposures.

### 4.1 Strengths and Limitations

One strength of our study is that the database could be used to identify VOCs that pose a health risk to children and adolescents in EU member states. While it was not possible to quantify the health risks associated with non-cancer effects of VOCs, we were able to conduct a quantitative cancer risk assessment for formaldehyde and benzene. Another strength is that our estimation is based on data from the supporting toxicological database of the IAQ Risk Calculator, which was developed through extensive international collaboration [39].

The limitations of our investigation must also be considered. Firstly, the number of educational buildings from which levels of VOCs were reported and used in our analysis has varied considerably by country. In addition, when multiple investigations were available from the same country, indoor air quality problems were often identified only in specific cases, indicating local problems with indoor VOC pollution. For example, the database contains seven studies from Portugal, representing 111 buildings. In six of the studies, comprising 106 of these buildings, the PODIadj values for neurological effects due to formaldehyde exposure were less than 1. Only one study, including five buildings, had a value above 1. Consequently, the results of our analysis are only applicable to the children being in these buildings and they may not be generalised to the entire child population residing in the country where concentration data were collected. Secondly, our data were taken from research that determined the level of VOCs in day care centres, elementary schools, high-schools and universities using passive air sampling exclusively. Although passive sampling is the recommended approach for monitoring air quality in schools, it can only provide information on average VOC levels and is not suitable for measuring their peak concentrations [83]. Additionally, the IAQ Risk

Calculator does not account for differences in inhalation rate or volume of air inhaled among age groups when calculating the $\mathrm{PODI}_{\mathrm{adj}}$ values. Although the software developers considered these factors to have a negligible effect on the results, they may lead to an underestimation or overestimation of the health risks [39]. Similarly, no factor was introduced to convert intermittent exposure (five days per week during school term) to continuous exposure (all day, seven days per week), as the exposure patterns in the studies from which the reference concentrations and POD values were obtained closely resemble those experienced by schoolchildren (five days per week) [39].

Assessing indoor air quality is a complex task. While health risk assessments are important, other factors that influence indoor pollutant levels must also be considered. One key factor is the contribution of outdoor (ambient) air pollution, which can be evaluated by monitoring outdoor air quality simultaneously with indoor measurements. This helps guide decisions about further monitoring, risk assessment, and mitigation strategies such as improving ventilation or controlling pollution sources. Given these complexities, risk assessment results should be interpreted with caution, and further research is recommended.

## Conclusions

This study highlights the substantial health risks associated with exposure to volatile organic compounds (VOCs) in European educational buildings. By systematically analysing reported VOC concentrations from 18 European countries and applying the WHO’s Indoor Air Quality Risk Calculator, we identified several countries where formaldehyde and benzene exposure in schools emerged as major health concerns, particularly due to their potential respiratory and neurological effects. Given that children and adolescents spend a substantial portion of their day in educational settings, our findings highlight the need for targeted interventions to improve indoor air quality in these environments. This study adds to the growing body of evidence supporting the implementation of stricter regulations and proactive measures to reduce VOC exposure in schools. Future research should focus on the long-term health impacts of chronic, low-level VOC exposure and assess the effectiveness of mitigation strategies. Ensuring safe indoor air in educational buildings is not only a public health priority but also essential for protecting the well-being and development of future generations.

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## Statements and Declarations

**A. C.**: investigation, data curation, writing – original draft; **L. P.:** conceptualization, investigation, data curation, supervision, writing – original draft; **Sz. L.:** data curation; **M. M.**: writing – review & editing; **J. D.:** data curation; **N. K.:** data curation; **S. Sz.:** conceptualization, investigation, data curation, supervision, writing – original draft. All authors contributed to the interpretation of data, and read and approved the final manuscript.

## Competing interests

The authors declare no competing interests.

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## Data availability statement

The dataset used in this study are available on Mendeley Data: http://dx.doi.org/10.17632/x4887snd8j.1