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REVIEW

Indoor air quality in schools

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Abstract In recent years, the use of synthetic materials in building and furnishing, the adoption of new lifestyles, the extensive use of products for environmental cleaning and personal hygiene have contributed to the deterioration of indoor air quality and introduced new sources of risk to humans. Indoor environments include home, workplaces such as offices, public buildings such as hospitals, schools, kindergartens, sports halls, libraries, restaurants and bars, theaters and cinemas and finally cabins of vehicles. Indoor environments in schools have been of particular public concern. According to recent studies, children aged between 3 and 14 spend 90 % of the day indoors both in winter and summer. Moreover, children have greater susceptibility to some environmental pollutants than adults, because they breathe higher volumes of air relative to their body weights, and their tissues and organs are actively growing. In this review, the authors explore the methodological approaches used for the assessment of air quality in schools: monitoring strategies, sampling and analysis techniques and summarizing an overview of main findings from scientific literature concerning the most common pollutants found in school environments.

Keywords Carbon dioxide (CO_2) · Formaldehyde and carbonyl compounds · Indoor air quality (IAQ) · Inorganic gases · Monitoring strategies · Ozone (O_3) · Particulate matter (PM) · School environments sources · Volatile organic compounds (VOCs)

Introduction

In recent years, numerous scientific studies highlighted that citizens spend most of their time in indoor environments, e.g., home, offices, schools, hospitals, kindergartens, sports halls, libraries, restaurants, bars, theaters and vehicles. Citizens are more exposed to indoor pollution than outdoor

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(Blondeau et al. 2005; Bruno et al. 2008; Pegas et al. 2010). Indoor air quality has a considerable impact on public health because indoor exposure may pose harmful health effects such as respiratory and cardiopulmonary pathologies and asthma, especially for children (Yang et al. 2009; Sohn et al. 2012). There is a considerable interest in the assessment of the association between air pollution exposure and health effects in school environments, as shown by more than 70 epidemiological publications currently available (e.g., Guo et al. 1999; Venn et al. 2000). Indoor air pollution is characterized by a large variability in pollutants' concentration among different indoor environments and may also vary within a specific environment as a function of location and time. The extent of these variations depends on factors such as the emission characteristics of the sources, the occupants' behavior and the microclimatic and ventilation conditions (Report EUR 16051 EN 1994; UNI EN ISO 16000-1: 2006). Thus, indoor air pollution and human exposure are highly dynamic processes rather than static phenomena.

In this review, the attention will be focused on air quality in school buildings. Children spend large amount of time in these environments and are more sensitive subjects to indoor pollutants (Faustman et al. 2000; Mendell and Health 2005; WHO 2006a, b; Chithra and Shiva Nagendra 2012). Several studies reported that indoor air pollution can increase the chance of long- and short-term health problems for students and teachers in terms of comfort, productivity and academic performance (Daisey et al. 2003; Shendell et al. 2004; Dijken et al. 2005; Mendell and Health 2005; Wargocki et al. 2005; Mi et al. 2006; Shaughnessy et al. 2006; Croome et al. 2008). The indoor pollution observed inside school buildings can be traced back to a variety of causes, such as the use of high emitting materials for building construction and furnishing, minimal landscaping with poor drainage, heating, ventilation and air conditioning units, the lack of preventative maintenance, crowded conditions (Godwin and Batterman 2007) and cleaning products that release chemicals into the air (UBA 2008). Each school environment is uniquely characterized, and thus, each personal exposure is determined by a combination of the outdoor and indoor pollutant levels (Stranger et al. 2007, 2008). In fact, age and location of school buildings, pollutants transport from outdoor, chemical reactions in indoor air and heterogeneous processes at the air–solid interfaces are the other factors that influence the pollutant concentrations (Poupard et al. 2005). In developed countries, many studies were conducted during the past decade in order to assess Air quality in school environments (Seppanen et al. 1999; Daisey et al. 2003; Bartlett et al. 2004; Shendell et al. 2004; Ramachandran et al. 2005; Shaughnessy et al. 2006; Godwin and Batterman 2007) and concentration of a large number of indoor air pollutants were measured including Carbon dioxide (CO_2), Ozone (O_3), Nitrogen oxides (NO_x), Carbon oxide (CO), Sulfur dioxide (SO_2) (Lee and Chang1999; Scheff et al. 2000a, b; Bartlett et al. 2004; Shendell et al. 2004; Blondeau et al. 2005; Ramachandran et al. 2005; Godwin and Batterman 2007) and Volatile Organic Compounds (VOCs) (Kotzias 2005; Godwin and Batterman 2007; Pegas et al. 2010, 2012) and Particulate Matter (PM) (Koutrakis et al. 1992; Ozkaynak et al. 1996; Daisey et al. 2003). The aim of the present review was to describe the methodological approaches used for the assessment of air quality in schools, according to the main characteristics of school environments. Chemical pollutants and their sources and the monitoring strategies and an overview of the main scientific findings are discussed. This article is an abridged version of the chapter by Dambruoso et al. (2013) published in the book series Environmental Chemistry for a Sustainable Word (http://www.springer.com/series/11480).

Indoor environments and pollutants

The wide range of school building designs leads to large variations in indoor pollutants levels and hence personal exposure (Ashmore and Dimitroulopoulou 2009). Children spend their school hours in different environments: classrooms, laboratories where available, playgrounds and other locations within the school. As a result, individual exposure changes related according to the variation in pollutants levels inside the several school locations (Mejía et al. 2011).

Pollutants emission can occur in many school settings where different activities take place. Surely the most important ones with respect to the time spent by children are the classrooms (Lee and Chang 2000; Hulin et al. 2011; Bertoni et al. 2002; Blondeau et al. 2005; Mi et al. 2006; Ekmekcioglu and Keskin 2007; Fromme et al. 2007; Godwin and Batterman 2007; Diapouli et al. 2008; Weichenthal et al. 2008; Yang et al. 2009; Sofuoglu et al. 2010; Wu et al. 2010; Goyal and Khare 2011; Gul et al. 2011; Mejía et al. 2011; Mullen et al. 2011; Park et al. 2011; Smedje et al. 2011; Szoboszlai et al. 2011; Sohn et al. 2012; Zhang and Zhu 2012); the gyms (Godwin and Batterman 2007; Branis et al. 2009; Branis and Safránek 2011; Hochstetler et al. 2011; Szoboszlai et al. 2011); the science labs (often without fume hoods) (Godwin and Batterman 2007; Yang et al. 2009; Jo and Kim 2010; Goyal and Khare 2011; Park et al. 2011; Szoboszlai et al. 2011); the computer rooms (Yang et al. 2009; Wu et al. 2010; Szoboszlai et al. 2011; Sohn et al. 2012); and the dining halls (Gul et al. 2011). In addition, the exposure that may occur in other school environments such as the arts and crafts labs (Blondeau et al. 2005; Godwin and Batterman 2007); the

office rooms (Godwin and Batterman 2007; Goyal and Khare 2011; Zhang and Zhu 2012), the kitchen (MacIntosh et al. 2012); the cafeterias (Godwin and Batterman 2007; Hochstetler et al. 2011; Zhang and Zhu 2012); other miscellaneous use rooms (e.g., music room, library); the swimming pools or stairwells should be taken into account, as demonstrated by numerous scientific papers (Godwin and Batterman 2007; Goyal and Khare 2011; Gul et al. 2011; Zhang and Zhu 2012, Sohn et al. 2012).

In order to give an as complete as possible assessment of air quality in schools, many authors have considered in their experimental activities several aspects that may affect the air quality and so the levels of people exposure (Stranger et al. 2008; Pegas et al. 2012). Among these, the most relevant ones appear on the school sites such as industrial (Scheepers et al. 2010; Tran et al. 2012), rural (Blondeau et al. 2005; Fromme et al. 2007; Hulin et al. 2011; Tran et al. 2012; Zhang and Zhu 2012), traffic (Blondeau et al. 2005; Hochstetler et al. 2011; Raysoni et al. 2011; Szoboszlai et al. 2011; Chithra and Shiva Nagendra 2012), suburban (Branis and Safránek 2011), urban (Fromme et al. 2007; Hulin et al. 2011; Mullen et al. 2011; Tran et al. 2012; Zhang and Zhu 2012) or background site because of the proximity of outdoor relevant sources (Janssen et al. 1997, 2001; Green et al. 2004; Wu and Batterman 2006; Van Roosbroeck et al. 2007; Appatova et al. 2008; Branis and Safránek 2011; Hochstetler et al. 2011; Mejía et al. 2011; De Giuli et al. 2012); the age of the buildings (Godwin and Batterman 2007; Ashmore and Dimitroulopoulou 2009; Yang et al. 2009; Hochstetler et al. 2011; Mullen et al. 2011; Zhang and Zhu 2012) in respect of the type of heating systems (MacIntosh et al. 2012; Park et al. 2011; De Giuli et al. 2012; Corgnati et al. 2007); the quality of the used materials, the capacity to accumulate or disperse pollutants; the room design (floor area and room volume) and the level of occupancy (Daisey et al. 2003; Godwin and Batterman 2007; Theodosiou and Ordoumpozanis 2008; Weichenthal et al. 2008; Mumovic et al. 2009; Mejía et al. 2011; Goyal and Khare 2011; Mullen et al. 2011; Chithra and Shiva Nagendra 2012), measured by indoor CO_2 levels used as a surrogate of the rate of outside supply air per occupant (Daisey et al. 2003); the type and quality of ventilation in terms of number of doors and windows or the presence of natural or mechanical ventilation systems (Ashmore and Dimitroulopoulou 2009; Goyal and Khare 2011; Mullen et al. 2011, Mejía et al. 2011, Grimsrud et al. 2006; Lee and Chang 2000; Mumovic et al. 2009; Wåhlinder et al. 1997; Theodosiou and Ordoumpozanis 2008; Blondeau et al. 2005), very important for the removal of pollutants (Sohn et al. 2012; UBA 2008; Yang et al. 2009).

Many authors emphasized also the decisive role played by the micrometeorological parameters such as mean temperature and relative air humidity (Godwin and Batterman 2007, Park et al. 2011; Smedje et al. 2011; Weichenthal et al. 2008; Zhang and Zhu 2012; Fraga et al. 2008; Yang et al. 2009; De Giuli et al. 2012), fundamental in the emissive process of indoor pollutants, by the choice of materials of board, desks, chairs, floor, because of their different emission capacity (Pegas et al. 2010; Yang et al. 2009; Goyal and Khare 2011; Chithra and Shiva Nagendra 2012) and by the activities carried out by the occupants like the use of cleaning products or collage and painting activities (Chithra and Shiva Nagendra 2012).

The most common pollutants found in schools and childcare facilities are the following: PM, VOCs, Formaldehyde and Carbonyl compounds, other Inorganic Gases: NO_x, CO, SO_2, CO_2 and O_3, deeply described in following paragraphs. Their sources can be classified as: continuous (with a uniform or irregular pattern) and intermittent sources (with a periodic or variable pattern) respect to the duration of their emission activity (UNI EN ISO 16000-1:2006).

Particulate matter

Among the indoor air pollutants, nowadays there is a growing interest in PM. The aerosol exposure via the inhalation route represents a major potential source of hazard for human health, depending on the duration of exposure and concentrations, size and chemical composition of airborne particles (Abdel-Salam 2006). In several papers, in fact, the exposure to high PM10 concentrations has been associated to increased risk of death for cardiovascular or respiratory causes (Englert 2004; Zanobetti and Schwartz 2005; Forbes et al. 2009; Pope et al. 2009). These effects may be largely caused also by finer particles that, as a consequence of their greater surface area, could be an effective media to transport different kinds of pollutants (PAHs, heavy metals, asbestos, etc.) deeply into the lung (Nadadur et al. 2007; Sager and Castranova 2009; Reich et al. 2009). In particular, the exposure to these finer particles can cause short- and long-term effects such as increased respiratory symptoms, decreased lung function, alterations in tissue and structure lung, in respiratory tract and premature death (Prieditis and Adamson 2002; Damek-Poprawa and Sawicka-Kapusta 2003; Wahab and Basma 2004; Huang and Ghio 2006; Hong et al. 2007; Wild et al. 2009; Daresta et al. 2010; Liuzzi et al. 2011).

Although the school environment normally lacks typical indoor PM sources such as smoking and cooking, many children are present in a limited space over a period of several hours. The use of cleaning products and floor polish can also temporarily affect the air quality determining an increase in chemical pollutants in school environments. On

the other hand, the floor surface type and level of cleaning are important factors in maintaining low dust levels. The presence of PM can be related to: (1) insufficient ventilation in schools (especially in winter), (2) infrequently and unthoroughly cleaned indoor surfaces, (3) a large number of pupils in relation to room area and volume, (4) low class level related to floor numbers of school buildings and (5) resuspension of particles from room surfaces (Sexton and Ryan 1988), which is related to physical activity of the pupils. Moreover, numerous studies showed that gas-phase reactions between O_3 and terpenes (for example used in cleaning products) can contribute significantly to the growth of indoor secondary organic aerosols (Weschler and Shields 1999; Long et al. 2000; Wainman et al. 2000; Li et al. 2002; Fan et al. 2003; Sarwar et al. 2003).

Recent studies report levels, behaviors and chemical composition of PM in different indoor environment (e.g., Chao and Wong 2002; Gemenetzis et al. 2006; Martuzevicius et al. 2008; Olson et al. 2008; Smolík et al. 2008 Lai et al. 2010; Saraga et al. 2010; Zhu et al. 2010; Huang et al. 2012) and, in particular, in elementary schools (Fromme et al. 2008; Almeida et al. 2011; Oeder et al. 2012; Pegas et al. 2012; Smolík et al. 2008).

Chithra and Shiva Nagendra (2012) monitored the PM10, PM2.5 and PM1 concentrations by means of an environmental dust monitor in order to study the relationship between outdoor and indoor air quality in eight French schools. The indoor–outdoor (I/O) ratios of PM were higher than two for coarse fraction and minor than one for finer fraction. The high I/O value of PM10 concentration and its behavior indicated significant contribution from the activities of occupants inside classroom and thus from dust resuspension. On the contrary, the lower I/O values for PM1 and CO suggested that no indoor source of finer particles were in classrooms and confirmed their intrusion from the nearby road and due to vehicular emissions (Fig. 1). This evidence was confirmed by a strong seasonal variability of finer PM fraction. Moreover, investigating the influence of classroom occupancy, the authors found that higher particulate matter concentrations were detected for classroom during the periods when the classroom was occupied.

In same way Yang et al. (2009), evaluating indoor air quality inside three different school environments in Korea found that the mean I/O PM10 ratios (gravimetric measurements) were higher in the classrooms than in laboratories and computers rooms, respectively. In addition, Diapouli et al. (2008) showed higher I/O ratio for PM10 and PM2.5 inside gymnasium, where intense activity took place, smoking office and classrooms and the I/O ratio smaller than one for ultrafine particles (UFP) in all investigated indoor environments (Fig. 2). These evidences confirmed that the most important contribution to PM concentrations in school classroom is the resuspension of particles due to pupil's activity.

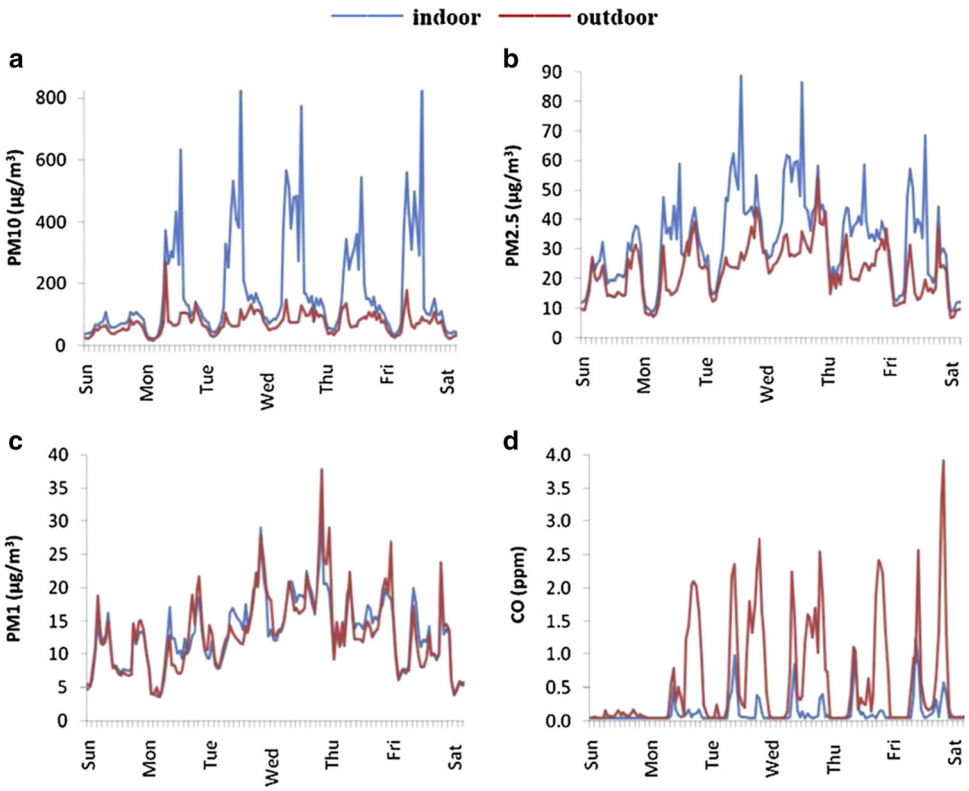

Fig. 1 Weekly variations in indoor–outdoor **a** PM10, **b** PM2.5, **c** PM1 and **d** CO concentrations inside classroom (Chithra and Shiva Nagendra 2012)

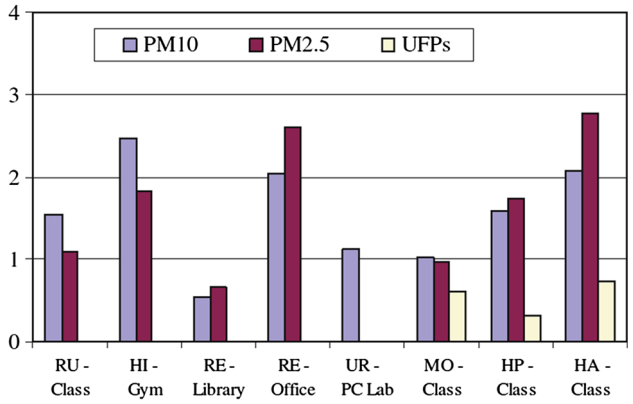

Fig. 2 Mean indoor/outdoor ratio of PM10, PM2.5 and UFPs in a rural area (RU), two blocks away from a major highway (HI), a residential area (RE), a heavy-trafficked neighborhood in the center of Athens (UR), a residential area close to a major motorway (MO), a densely populated area close to a major motorway (HP) and at the harbor of Athens (HA) (Diapouli et al. 2008)

The presence of carpets in schools building also contributed to poor indoor air quality. Stranger et al. (2007), in a study regarding Belgium schools, found a significant difference between I/O ratios calculated for the classrooms with and without the presence of carpets, and in particular, the authors reported a mean I/O ratio equal to 2.63 in the classrooms where carpets were present and mean I/O ratio equal to 1.03 in the classroom where tiles or linoleum floor coverage were present.

The more recent study conducted by the same authors in 2008 (Stranger et al. 2008) focused the attention on the chemical composition of PM collected in 27 primary schools in the urban and suburban areas of Antwerp (Belgium). The authors showed the elemental composition of indoor particulate matter (PM2.5) collected in classroom and analyzed by the energy dispersive X-ray fluorescence (ED-XRF) was different than that evaluated in outdoor air. In particular, they found that the elements such traffic markers (V, Pb, Cr), S and Fe were the highest contributions to local outdoor PM, while high contributions to indoor PM in schools were determined by markers of crustal resuspension (Si, Ti, Al), Ca and Cl. The higher I/O ratios were determined for Cl, Ca and crustal species. Chloride could derive by detergents used for cleaning activities inside the classroom, while Ca concentrations could probably be determined by the chalk (mainly $CaSO_4$) used on the blackboards and/or the gypsum walls and plasters used as construction materials. Finally, crustal species were probably due to resuspension of dust because of room occupation. Fromme et al. (2008) also reported the elemental composition of PM collected by gravimetric sampling system at two classrooms in Munich. The scanning electron microscopy and the energy dispersive microanalysis (EDX) on PM filters showed that the indoor PM consisted mainly of earth crustal materials, detritions of the building materials and chalk ($CaSO_4$). These findings suggested that increase of PM10 concentrations in classrooms were due to a physical activity of the pupils and to resuspension of mainly indoor coarse particles, and thus, indoor-generated PM was less toxic than PM in outdoor air.

The measurements of the microclimatic parameters (ventilation, temperature and air humidity), which can influence directly or indirectly the indoor pollutant levels, result very important in the assessment of air quality in the school. Fromme et al. (2007) found that PM2.5 indoor concentrations, gravimetrically measured in several schools in Munich, increased by 1.7 $\mu g/m^3$ per 10 % increase in humidity and by 0.5 $\mu g/m^3$ per increase in CO_2 indoor concentration by 100 ppm. The higher PM concentrations in winter and their correlation with CO_2 concentrations suggested that inadequate ventilation plays a major role in the establishment of poor indoor air quality. In addition, high PM10 concentration measured in low-level classes and in rooms with high number of pupils suggested that the physical activity of pupils contribute to a constant process of resuspension of sedimented particles (Lee and Chang 1999, 2000; Blondeau et al. 2005). Furthermore, Sohn et al. (2012) evaluated the influence of mechanical ventilation systems on indoor air quality in school buildings in Korea. The results showed remarkable difference in indoor air pollutants' level according to the operation of mechanical ventilation system and in particular showed that the ventilation systems decreased the levels of indoor pollutants in the all selected classrooms. Therefore, use of mechanical ventilation system can play key roles in improving the air quality within schools.

Volatile organic compounds

Volatile organic compounds are widely present in school environments as they are emitted from multiple both internal and outdoor sources. Among the VOC, the high priority pollutants that are regulated in indoor environments and that significantly affect children health are Benzene, Naphthalene, Formaldehyde, Toluene, Xylenes, Styrene, Limonene, Alpha-pinene and Dichloromethane. Benzene, Toluene, Xylenes and Styrene can be emitted from solvent-based paints and consumer products, such as collage and painting materials, used in the art and craft rooms, from Poly Vinyl Chloride flooring and adhesive used for gyms covering and from printed materials (Kotzias 2005). Dichloromethane is found in adhesives, spray paints, while the presence of Limonene and Alpha-pinene is more related to the emission from cleaning products (aerosol and liquid) (Priscilla et al. 2010). Polymeric materials that are used for construction,

decoration and furnishing of schools are high VOC emitters due to their composition and large surface areas. Moreover, wood-based products used for construction of writing desks and cabinets are important sources of these pollutants in these environments. Some VOC are associated with a variety of serious health effects (Shendell et al. 2004) and symptoms such as asthma and allergic reactions (Sofuoglu et al. 2011). Moreover, several studies reported a strong association between mucous membrane irritation, central nervous system symptoms and total exposure to VOC; these symptoms are similar to those that are frequently attributed as a cause of sick building syndrome (Mølhave et al. 1986; Hodgson et al. 1991). In case of extreme concentrations, some VOC may result in impaired neurobehavioral function (Burton 1997). Exposure to high concentrations of several VOC commonly found in indoor air is associated with cancers in laboratory animals (Jones 1999). A preliminary screening monitoring of the sum of the VOC in a school environment can be conducted by direct measurements with automatic instruments as flame ionization detector (FID) or photo ionization detector (Hodgson 1995; Pegas et al. 2010). Short-term or long-term measurement methods allow obtaining information about the single pollutants present in the investigated indoor environment (UNI EN ISO 16000-5: 2007). Short-term measurements were conducted by active sampling on stainless steel tube packed with specific adsorbent beds using low-flow sample pumps (UNI EN ISO 16017-1: 2007; Fraga et al. 2008; Jo and Kim 2010; Pegas et al. 2010; Scheepers et al. 2010; ISO 16000-6: 2000). Diffusive sampling is the recommended method to perform long-term measurements (usually from few days to several days or weeks) (UNI EN ISO 16017-2: 2007; Bruno et al. 2005; Angiuli et al. 2003; Pennequin-Cardinal et al. 2005). VOCs collected onto adsorbent cartridges were thermally or chemically desorbed and analyzed by gas chromatography coupled to a flame ionization detector or to a mass spectrometer (Bruno et al. 2005; Angiuli et al. 2003; Pennequin-Cardinal et al. 2005). Stainless steel canisters were also used to collect VOC in indoor environments (Meininghaus et al. 2003; Guo et al. 2004).

Volatile organic compounds monitoring campaigns conducted in different school environments of several cities (Michigan, Catania, Athens, Arnhem and Nijmegen, Brussels, Milan, Thessaloniki, Nicosia) highlighted that indoor sources, micrometeorological parameters and building conditions might have negative effects on indoor air quality (Kotzias 2005; Godwin and Batterman 2007; Pegas et al. 2010, 2012). Moreover, it was found that increasing ventilation rates and using low-emission materials improve indoor air quality (Pegas et al. 2010). Godwin and Batterman 2007, monitoring VOC concentrations over one workweek in 64 elementary and middle school classrooms in Michigan, found that most VOC had low concentrations (mean of individual species <4.5 $\mu g/m^3$) also if they were higher than outdoor air concentrations (mean of individual species <0.51 $\mu g/m^3$). For example, benzene and toluene concentrations in indoor air were 0.09 and 2.81 $\mu g/m^3$, respectively, while their outdoor concentrations were 0.06 and 0.52 $\mu g/m^3$, respectively; the total concentration of chlorinated compounds was 0.24 $\mu g/m^3$ in indoor air and <0.07 $\mu g/m^3$ in outdoor air. These findings suggested that none of the sampled rooms were contaminated and that no building-wide relevant contamination sources were present. Otherwise, higher indoor levels of many VOC were registered in two studies involving 14 elementary schools in Lisbon, Portugal (Pegas et al. 2010, 2012). Almost all identified VOC (up to 40 compounds) showed I/O ratios higher than one. The same results were found by Kotzias (2005) in schools and kindergartens of several cities in Southern and Central Europe: the sum of indoor concentrations ranged from a few micrograms (ca. 8) to 281 $\mu g/m^3$, while outdoor levels ranged from 7 to 153 $\mu g/m^3$. VOC concentrations two to four times higher than the outdoor concentrations were detected in kindergartens and schools of Arnhem and Nijmegen and in Izmir (Turkey) (Shendell et al. 2004; Sofuoglu et al. 2011; Stranger et al. 2008). Among monitored VOC, benzene, toluene, ethylbenzene and xylenes were most abundant compounds with I/O ratios exceeding unity.

A huge increase in indoor VOC concentrations was also observed when art works or science activities were undertaken concurrently or just prior to the measurements (Shendell et al. 2004; Godwin and Batterman 2007; Pegas et al. 2010). In particular, Pegas et al. (2010) found that there was an increase in VOC concentrations reaching 13 ppm, when glue and paints were used in pupil's art class.

Formaldehyde and carbonyl compounds

The most relevant carbonyl compounds detected in indoor environments are Formaldehyde, Acetaldehyde, Acetone, Benzaldehyde, Butyraldehyde, Capronaldehyde, 2,5-Dimethylbenzaldehyde, Isovaleraldehyde, Propionaldehyde, m-Tolualdehyde, o-Tolualdehyde, p-Tolualdehyde and Valeraldehyde. As a result of the several industrial uses in the manufacture of sheet and insulation materials, paints, cleaning agents and cosmetics, the carbonyl compounds can usually be detected in school environments. Wood-based materials made for indoor use are the following ones: (1) Particleboard (PB) used as sub-flooring and shelving and in cabinetry and furniture; (2) hardwood plywood paneling used for decorative wall covering and used in cabinets and furniture; (3) medium density fiberboard (MDF) used for drawer fronts, cabinets and furniture tops. Therefore, articles

produced from wood-based materials such as furniture, doors and paneling are still the most important sources of these compounds in schools. Formaldehyde (HCHO) is the most abundant airborne indoor carbonyl and represents an important constituent of adhesives in the sheet material industry (Urea–Formaldehyde resins, Phenol–Formaldehyde resins, Melamine–Formaldehyde resins and Melamine–Urea–Formaldehyde resins). MDF material contains a high resin-to-wood ratio and is generally recognized as being the highest formaldehyde-emitting pressed wood product. Several studies showed that indoor HCHO concentrations in schools constructed within 1 year were significantly higher, indicating that school buildings are characterized by several indoor HCHO sources such as furnishings made of PB and MDF. Several carbonyls, such as Formaldehyde, Acetaldehyde and Propionaldehyde, are included in the list of air toxics in the Clean Air Act Amendments of 1990 (USEPA 1991). More specifically, HCHO is defined as a human carcinogen on the basis of a sufficient evidence of carcinogenicity from studies in animals and humans and of supporting data on mechanisms of carcinogenesis. In recent years, scientific findings led an increasing interest in HCHO detection inside school buildings due to the high risk of children exposure (WHO 2010; NIOSH/IPCS 2004; IARC 2012). California Office of Environmental Health Hazard Assessment (OEHHA) set an 8-h chronic and acute inhalation reference exposure level (REL) for HCHO equal to 9, 9 and 55 $\mu g/m^3$, respectively (OEHHA 2008). Acetaldehyde, an abundant carbonyl in indoor air, has been classified as probable human carcinogen by USEPA (2003). Acrolein is a severe lung irritant that, in condition of high acute exposure, can induce oxidative stress and delayed-onset lung injury, including asthma, congestion and decreased pulmonary function. Because of concerns about adverse human health effects posed by Acrolein, OEHHA set an 8-h chronic and acute inhalation REL equal to 0.70, 0.35 and 2.5 $\mu g/m^3$, respectively (OEHHA 2008).

Scientific papers published during the last 10 years reported experimental results obtained from investigation of HCHO and other carbonyl compounds in school buildings (Lee and Chang 2000; Righi et al. 2002; Kotzias 2005; Mentese and Gullu 2006; Vaizoglu et al. 2003; Hanoune et al. 2006; Yang et al. 2009; Sofuoglu et al. 2011; Yamashita et al. 2012; Pegas et al. 2011a, b; Barro et al. 2009). The measurement of HCHO and other carbonyl compounds was performed according to the requirements of existing international standard (ISO 16000-3: 2011). The method is suitable for determination of these compounds in the approximate concentration range from 1 $\mu g/m^3$ to 1 mg/m^3 and involves drawing air through a cartridge containing silica gel coated with 2,4-dinitrophenylhydrazine (DNPH) reagent. The principle of the method is based on the specific reaction of the carbonyl group with DNPH in the presence of an acid, to form stable 2,4-dinitrophenylhydrazones. The DNPH derivatives are analyzed with High performance liquid chromatography and Ultraviolet (UV) absorption detector operating at 360 nm (Lee and Chang 2000; Daisey et al. 2003; Meininghaus et al. 2003; Yang et al. 2009; Pegas et al. 2010; Park et al. 2011; Sohn et al. 2012).

Pegas et al. (2011a) measured indoor and outdoor concentrations of HCHO and other carbonyls in 14 elementary schools in Lisbon, Portugal. In all the investigated environments, indoor aldehydes' levels were higher than those observed outdoors, especially for HCHO. Pegas et al. (2011b) carried out a further measuring campaign in school buildings in order to evaluate seasonal variation in indoor and outdoor levels. Most of the assessed carbonyls occurred at I/O ratios above unity in all the seasons, and this evidence showed the influence of indoor sources and building conditions on indoor air quality. However, it was observed that carbonyls' levels were higher during the warm months.

Yang et al. (2009) characterized HCHO concentrations within 55 school buildings in Korea, selected on the basis of the year of construction, in order to relate indoor levels to the age of school buildings. HCHO levels were measured inside three different school building environments: classrooms, laboratories and computer rooms. Experimental results showed that mean HCHO concentrations inside classrooms and computer rooms exceeded the acute REL established by OEHHA. Moreover, HCHO concentrations inside schools constructed within 1 year were significantly higher than the Korean Indoor Air Standard, suggesting that renovated schools have important indoor HCHO sources, such as furnishings principally made of PB and MDF. Therefore, in order to improve air quality within schools, especially within renovated schools, the authors suggest the implementation of increased ventilation rates by means of mechanical systems and the use of low-emitting materials.

Kotzias (2005) reported the experimental results deriving from measuring campaigns performed in several cities in Southern and Central Europe in the frame of the AIRMEX project (Indoor Air Monitoring and Exposure Assessment Study). This study highlighted that HCHO and carbonyls' concentrations (Acetaldehyde, Propanal and Hexanal) inside the buildings/kindergartens were up to 7–8 times higher than outdoor, confirming that strong HCHO indoor sources exist.

Lee and Chang (2000) showed the results of a study carried out to characterize HCHO levels inside selected classrooms in Hong Kong in order to compare the measured concentrations with the established standards and to suggest policy interventions to improve air quality. HCHO concentrations (ranging from undetectable to 27 $\mu g/m^3$) were substantially lower than Honk Kong Indoor Air

Quality standard, indicating that there were no apparent HCHO indoor sources and that classroom furnishing did not add a remarkable contribution.

Sofuoglu et al. (2011) reported HCHO levels measured in primary school classrooms and kindergartens in Turkey. Experimental data revealed that HCHO was one of the most abundant indoor pollutants and that concentrations were related to both spatial and seasonal variability. Similar HCHO levels between urban and suburban schools, but different HCHO levels between two urban schools can be explained by the relative strength of the indoor HCHO sources compared with the outdoor ones. The HCHO concentrations measured in classrooms were in the literature range (10–400 $\mu g/m^3$) although they resulted high if compared with data related to schools in Sweden (Smedje et al. 1997a, 1997b) China (Lee et al. 2004) and Australia (Zhang et al. 2006). Furthermore, HCHO levels in classrooms were lower than the concentrations measured in homes and offices (Mentese and Gullu 2006; Vaizoglu et al. 2003) but similar to those in libraries (Righi et al. 2002; Hanoune et al. 2006). Regarding kindergartens, HCHO levels were higher than those measured in classrooms and difference in urban and suburban concentrations was not significant. It can be asserted that there were consistent sources of HCHO inside kindergartens because neither seasonal nor spatial differences were significant. The overall average of the concentrations measured in this study (85 $\mu g/m^3$) was clearly higher than the Dutch kindergartens (from ca. 6 to 11 $\mu g/m^3$) (Kotzias 2005) and in the range of the Danish and Korean kindergartens (Yang et al. 2009).

N Gases, carbon oxide and sulfur dioxide

Inorganic gases commonly found in school indoor air are CO, SO_2 and NO_2. Sometimes, high H_2S and NH_3 concentrations are determined inside school buildings near industrial plants such as water treatment plants, waste treatment, desulfurization plants.

Nitrogen oxides (NO_x sum both nitrogen monoxide (NO) and dioxide (NO_2)) enter in indoor air mainly from outside, arising from the vehicular traffic, but several studies showed that the most important factors in increased exposures to NOx, over that the position of school buildings in the city center, where the use of gas appliances for heating is more (Oie et al. 1993; Alm 1999; Coward et al. 2001; Dimitroulopoulou et al. 2005; WHO 2006a, b). In particular, long-term exposure to high NO_2 concentrations promotes the onset of diseases of the respiratory tracts: epidemiological studies suggested that NO_2 represents a modest risk factor for respiratory illnesses compared with the use of electric stoves (Basu and Samet 1999).

Suitable methods for measuring NO_x in indoor environments can be divided into short-term measurement methods and long-term measurement methods UNI EN ISO 16000-15: 2008. Short-term measurements can be performed by continuous analytical monitoring instrument and by manual methods. The continuous monitoring instruments are based on principle of Chemiluminescence and are characterized by high time resolution (10–20 s). By the manual methods, instead, NO_2 is enriched actively onto a sorbent medium by means of suction pumps, and the concentrations obtained by these methods are average concentrations for the duration of sampling. Long-term measurements are generally carried out using diffusive sampler (manual methods) since the noise produced by continuous analytical monitoring instrument could discourage their use inside confined environments (Lee and Chang 1999, 2000; Blondeau et al. 2005; Poupard et al. 2005; Pegas et al. 2010, 2012; Gul et al. 2011; Raysoni et al. 2011; Sohn et al. 2012; Stranger et al. 2008).

Nitrogen oxides determination in French schools (Blondeau et al. 2005; Poupard et al. 2005) showed that vehicular exhaust emission from nearby traffic was the most important contribute to indoor concentrations. In fact, I/O ratios of calculated NO_2 varied in a narrow range from 0.88 to 1 as shown by the positive correlation between indoor and outdoor NO_2 concentrations, since indoor concentrations reflected the outdoor ones despite varying of building air-tightness. On the contrary, I/O of NO lied in a wider range ($0.5 < I/O < 1$), and there was no apparent correlation with the airtightness of the buildings. The authors suggested that this evidence was probably related to differences in the contribution of indoor homogeneous and heterogeneous reactions that NO undergoes. Similar considerations were elaborated by Stranger et al. (2008) in Belgium, by Pegas et al. (2012) in Lisbon and by Lee and Chang (2000) in Hong Kong.

Gul et al. (2011), confirming the results reported in previous study, showed that I/O ratios for NO_2 at high schools located in Eskisehir (Turkey) were >1 in dining hall or teacher's room where cooking and smoking activities took place ($1.8 < I/O < 3$). Moreover, Sohn et al. (2012) studied the relationship between NO_2 concentrations with indoor ventilation rate and showed that a direct correlation existed.

CO is a vehicular pollutant; therefore, vehicle exhaust from roads and parking areas nearby school buildings represents the most important contributor to CO indoor exposure. CO levels are generally very low inside schools since the emissive indoor sources influencing long-term CO levels can be gas cooking, unflued heaters and smoking (Alm et al. 1994; Coward et al. 2001). Exposure to high CO concentrations can cause acute intoxication since this compound combined with the hemoglobin of human blood

produces carboxy-hemoglobin and therefore disrupts the transfer of oxygen to human tissues. Various symptoms of neuropsychological impairment were associated with acute low-level exposure of CO concentration (Raub et al. 2000). Epidemiological studies reported increased relative risks of daily mortality and morbidity of the population by 0.9–4.7 % in prevailing urban air (Touloumi et al. 1994; Burnett et al. 1997, 1998).

Both indoor and outdoor measurements of carbon oxides (CO and CO_2) were conducted using a non-dispersive infrared analyzers (NDIR) (Lee and Chang 1999; Chaloulakoua and Mavroidisb 2002; Yang et al. 2009; Pegas et al. 2010, 2012; Park et al. 2011; Smedje et al. 2011; Sohn et al. 2012). Continuous measurements of CO_2 and CO can be performed with specific automatic portable sensors (Pegas et al. 2012). Diffusive portable probes reveal CO_2 concentrations based on its ability to absorb infrared radiation at a certain wavelength (2.3–4.6 μm) such as CO_2 above cited devices, whereas CO concentrations on the basis of electrochemical reactions. Moreover, CO is a purely outdoor pollutant; therefore, few studies have been conducted to evaluate the incidence of carbon monoxide on indoor environments. Chaloulakoua and Mavroidisb (2002) measured indoor and outdoor CO concentrations at a school near the center city of Athens. Authors found that the indoor and outdoor diurnal concentration cycles followed similar patterns and indoor concentrations showed a mild and slightly delayed response respect to outdoor concentration changes. In addition, they observed that CO concentrations measured during winter were higher than the respective concentrations measured during summer (3.96 and 1.92 ppm, respectively). These results were linked to the higher traffic volume and to winter meteorological conditions that favor the accumulation of pollutants. Similar results and considerations were obtained by Chithra and Shiva Nagendra (2012) in a study conducted in a school building located close to an urban roadway in India. Finally Yang et al. (2009) showed that renovation works had negative effects on the air quality, as significantly higher concentrations of CO were registered at schools constructed within 1 year (1.03 ppm) with respect to those built in previous years (0.59 ppm). These results might be caused by the new electric heating systems.

Sulfur dioxide is the main oxide of sulfur found in indoor air; however, the indoor concentrations determined inside school buildings are generally lower than those outdoors (Weschler 2009). The most important sources of SO_2 are located outdoors, and they can impact the indoor air of buildings near open coal fires, but the key problem is that SO_2 is readily absorbed onto indoor material surfaces, such as emulsion paints, the most important sink for SO_2 (Ashmore and Dimitroulopoulou 2009). Epidemiological studies on health effects by exposure to SO_2 are complicated by a paucity of representative exposure data and by confounding factors such as exposure to other indoor pollutants. Ho1wever, several studies provided some useful data concerning exposure-effect relationships showing that mortality was observed in populations exposed to 24-h pollution episodes in which SO_2 concentrations exceeded 300–400 μg/m^3 (0.12–0.15 ppm) (Health Canada 1995).

Sulfur dioxide in indoor environments is continuously measured by Electron Pulsed Fluorescence SO_2 Analyser. The operating principle of this instrument is based on measuring the fluorescence emitted consequently the absorption of ultraviolet light having wavelength in the range of 190–230 nm. The wavelength emitted in the range from 300 to 390 nm is directly proportional to the SO_2 concentration (Lee and Chang 1999; Meininghaus et al. 2003). Moreover, SO_2 concentration can also be determined using radial passive samplers (Stranger et al. 2008). Indoor O_3 concentrations can be monitored using an UV Absorption Ozone Analyzer (Blondeau et al. 2005; Poupard et al. 2005; Sohn et al. 2012) in order to give a real-time synoptic flow diagram. To perform long-term measurements, instead, it can be used specific diffusive adsorbing cartridges and the extract analyzed by UV–VIS spectrophotometry after chemical desorption (Stranger et al. 2007, 2008).

Sulfur dioxide is the less investigated pollutant for the evaluation of the indoor air quality in schools. Ashmore and Dimitroulopoulou (2009) found higher concentrations inside school buildings near open coal fires. Finally, Spedding (1974) suggested that lower SO_2 indoor concentrations could be linked with the capacity of indoor materials to absorb it. Among the wide variety of materials, the emulsion paints were identified as the most important sink for SO_2.

Carbon dioxide

Outdoor pollutant properties of CO_2 at a global scale are well documented, but in indoor school environments, it cannot be considered a pollutant, but represents an important proxy indicator of air quality. Indoor/outdoor ratio is greater than one in most of the classrooms, indicating the internal source prominent, with low level of outdoor intrusions; levels of 600–800 ppm are normally registered in the literature, indicative of inadequate ventilation rates (Seppanen et al. 1999; Apte et al. 2000), with peaks of 4,000 ppm (Daisey et al. 2003; Clements-Croome 2006). Exposure to this pollutant is associated with asthma (Mi et al. 2006) and values of 1,000 ppm are associated with a 10–20 % increase in student absences (Shendell et al. 2004), thus indicating CO_2

concentrations a primary variable in the health risk assessment of people in school (Rudnick and Milton 2003).

ASHRAE Standard 62-1989 (1989) suggested indoor CO_2 levels not exceeding 1,000 ppm s in choosing the right ventilation for acceptable air quality. With respect to worker safety, Occupational Safety and Health Administration (OSHA) has set a permissible exposure limit for CO_2 of 5,000 ppm over an 8-h work day, as also stated by the American Conference of Governmental Industrial Hygienists threshold limit value (TLV) set to 5,000 ppm for an 8-h workday, with a ceiling exposure limit of 30,000 ppm for a 10-min period based on acute inhalation data (OSHA 2014).

Occupants in school are the major sources of CO_2 with level that can vary according to occupancy levels, ventilation rate, room structure, air exchange rate (Lee and Chang 1999, 2000; Van Dijken et al. 2006; Fromme et al. 2007; Wåhlinder et al. 1997; Grimsrud et al. 2006; Theodosiou and Ordoumpozanis 2008; Mumovic et al. 2009).

Real-time monitoring (Ajiboye et al. 2006) showed wide variations during the day in CO_2 levels registered, with increases in the beginning of the lessons, during physical activities (Almeida et al. 2011), peaking until time breaks started when windows are opened and adequate ventilation assured (Fromme et al. 2007; Yang et al. 2009; De Giuli et al. 2012; Pegas et al. 2012) as shown in Fig. 3.

Moreover, Park et al. (2011) showed higher CO_2 concentrations in winter because the classrooms were not well ventilated in this season with respect to summer. Since the 1950s, atmospheric CO_2 level measurements have been made on air samples by NDIR for real-time monitoring of CO and CO_2 levels with specific automatic and auto-calibrating portable instruments (Lee and Chang 1999; Chaloulakoua and Mavroidisb 2002; Yang et al. 2009; Pegas et al.2010; Park et al. 2011; Smedje et al. 2011; Pegas et al. 2012; Sohn et al. 2012). However, the precision of such real-time measurements decreases rapidly for small air samples, as in the case for air extracted from ice cores that are better analyzed with HRGC/MS.

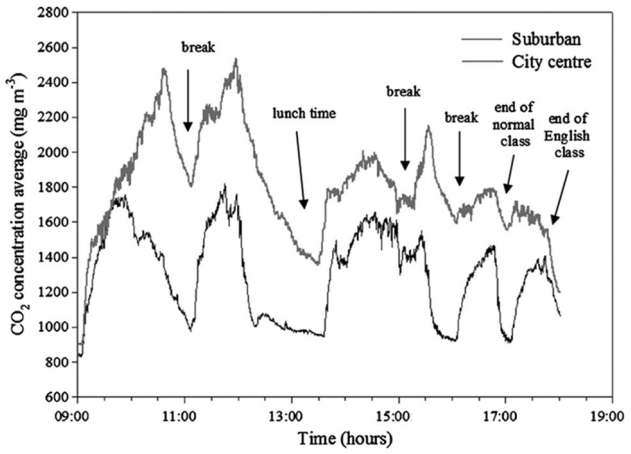

Fig. 3 Classroom CO_2 concentrations during a typical occupation period (Pegas et al. 2012)

Ozone

Also outdoor pollutant properties of O_3 at a global scale are well documented as its concentration depend on the exchange between upper and lower layers of atmosphere and on photochemical reactions involving nitrogen oxides and VOCs. Indoor/outdoor ratio is much lower than one in school (in the range 0.13–0.8) (Gold et al. 1996; Weschler 2000; Blondeau et al. 2005; Mendell and Health 2005; Poupard et al. 2005; Stranger et al. 2007, 2008; Mejía et al. 2011), for almost two reasons: O_3 reacts rapidly with indoor surfaces as well as by gas-phase reactions with some VOCs (Weschler 2006), the internal sources are insignificant, with high level of outdoor intrusions (Weschler 2000) but, generally, O_3 indoor concentrations are often below the detection limit (Grøntoft and Raychaudhuri 2004). Indoor sources of O_3 are nowadays some office equipment, primarily laser printers and copiers and electrostatic air cleaners (Leovic et al. 1996; Destaillats et al. 2008). Moreover, higher indoor O_3 concentrations were found in schools located in areas affected by industrial or urban pollution (Mi et al. 2006; Mejía et al. 2011) and an high correlation between outdoor and indoor concentration there exist as indoor concentrations increased more rapidly when windows/doors were open and outdoor O_3 concentrations increased (Gold et al. 1996). These results confirm that O_3 in indoor environments mostly comes from outdoor sources and the air exchange rate plays an important role.

Indoor O_3 levels are dependent on the generation rate, leakage, ventilation, degree of mixing and air filtration (Gold et al. 1996) and its decomposition rate is dependent on the quantity and type of materials in a building and the presence of organic chemicals characterized by highly reactive unsaturated carbon–carbon bonds VOCs coming from soft woods, carpets, linoleum, paints, polishes, cleaning products and air fresheners, soiled fabrics, soiled ventilation filters and the occupants themselves (Brown et al. 1994; Wolkoff 1995; Hodgson and Levin 2003; Weschler 2006).

Many toxicological and field studies of both adults and children (Tager 1999; Lee et al. 2004) established the short-term reversible effects of O_3 on lung function decrements, respiratory-related hospital admissions, school absence, restricted activity days, asthma-related emergency department visits and premature mortality (Gold et al. 1996; Hubbell et al. 2005; Weschler 2006). Moreover, ozone/terpene reactions (as used in cleaning agents) produce

strong airway irritants, like formaldehyde, acrolein, peroxyactyl nitrate, hydroperoxides with known adverse health effects (Wolkoff et al. 1999; Clausen et al. 2001; Wilkins et al. 2001; Rohr et al. 2002, 2003; Weschler 2006).

Real-time O_3 monitoring can be performed by an UV absorption Ozone Analyzer (Blondeau et al. 2005; Poupard et al. 2005; Sohn et al. 2012), but for long-term measurements, specific diffusive adsorbing cartridges can be used, using chemical desorption and formation of an absorbing molecule quantified by UV–VIS spectrophotometry (Stranger et al. 2007, 2008). This latter technique can achieve lower limit of detection, due to its pre-concentration capacity.

Conclusion

The main goal of this review was to summarize remarkable findings about air quality inside school buildings. More specifically, chemical pollutants, related sources and monitoring methodologies were reported. The outcomes provide suggestive evidence that certain conditions, commonly found in schools, can have adverse effects on the air quality and therefore on occupant's health. In particular, it was highlighted that the location, the age and air-tightness of school buildings, the room design, the ventilation rate, the building and furnishing materials, the occupant's activities and outdoor pollution play an important role on the indoor pollutants concentrations. Therefore, in order to safeguard the health of the occupants and in particular of children that are more sensitive to environmental pollutants some good practices should be followed. These actions include the construction of school buildings equipped with adequate ventilation systems to improve air exchange as well as the use of low-emitting building and furniture materials. Moreover, indoor concentrations of many pollutants are strongly influenced by outdoor sources so it is important that schools are not located in areas affected by high traffic or industrial pollution in order to improve air quality and reduce the impact on students' health. At this regard, several States are nowadays working to define guidelines for suggesting best practices in order to improve air quality inside school buildings, for defining reference values and for regulating the control methodologies. This need arises from the lack of available reference values for most of the pollutants monitored in indoor environments (WHO 2010).

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