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How to protect school children from the neurodevelopmental harms of air pollution by interventions in the school environment in the urban context

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A R T I C L E I N F O

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A B S T R A C T

Recently, there has been a flurry of publications assessing the effect of air pollution on neurodevelopment. Here we present a summary of the results obtained within the BRain dEvelopment and Air polluTion ultrafine particles in scHool childrEn (BREATHE) Project, which aimed to evaluate the effects of the exposure to traffic related air pollutants in schoolchildren in Barcelona. To this end, we comprehensively characterised air quality in 39 urban schools from Barcelona and identified the main determinants of children's increased exposure. We propose a series of measures to be implemented to improve air quality in schools within the urban context and, consequently, minimise the negative effects on children's neurodevelopment that we found to be associated with the exposure to air pollution. We also aimed to list some of the actions pushed by governments and the society (including school managers, parents, and children) that have been taking place around Europe for promoting better high quality in the school and its surroundings.

1. Introduction

Over 80% of world's population lives in urban areas that have higher levels of air pollution than the guidelines set by the WHO (2006). Particulate air pollution is the main environmental contributor to the global burden of disease and is one of the top preventable causes of disease over time (Cohen et al., 2017). Air pollution effects on the respiratory (such as asthma and reduced lung function) and cardiovascular system are well established but, because of the inadequacy of the available evidence, the potential effects of air pollution on brain development (and cognitive decline) have not been considered to date when estimating the burden associated with air pollution (Cohen et al., 2017). Pioneering studies on brain tissue from autopsies in dogs and children living in highly polluted areas of Mexico City showed inflammation in several brain areas (Calderón-Garcidueñas et al., 2008) and this work led to a long series of experiments in mice exposed to fine, ultrafine, and diesel particles (Costa et al., 2014). In mice, the central nervous system could be a direct or indirect target (via the olfactory or lung pathway, respectively) of particles that elicit a neuroinflammatory response in various brain regions. In humans, exposure to air pollution in utero is associated with increased risk of neurodevelopmental delay and autism (Lam et al., 2016).

Children are particularly vulnerable to environmental exposures since they are still under development. Moreover, due to their physiological (e.g. high breathing rates) and behavioural distinctions (e.g. high physical activity), children may receive higher doses of air pollutants than adults. As they spend long time in a shared location such as the school, it is important to ensure a good air quality in this environment for the benefit of the children and public health. Schools are a setting where children are aimed to expand their knowledge and manage behavioural responses, among other skills. Therefore, a proper characterisation of air pollutants in the schools and their associated health effects on cognition are needed to identify and target preventive actions to minimise the impact of air pollution.

BREATHE (BRain dEvelopment and Air polluTion ultrafine particles in scHool childrEn) is the largest epidemiological study in the general population assessing whether exposure of children to traffic related air pollutants (TRAPs) in schools adversely affects cognitive development (Sunyer et al., 2015) of urban children. The key strengths of BREATHE were the direct assessment of exposure in school classrooms and the school playgrounds, the study of cognitive function trajectories using repeated exams and the inclusion of neuroimaging. Here, we briefly

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Table 1
List of all BREATHE publications summarised in this article a by main topic.

Topic	References	Main findings
Air quality in school: levels, sources, pollutant infiltration, and greenness	Amato et al. (2014)	Identification of 7 outdoor and 2 children-activity-related $PM_{2.5}$ sources at schools.
	Dadvand et al. (2015b)	A reduction of indoor and outdoor air pollution was associated with greenness within and around schools.
	Minguillón et al. (2015)	The sands from playgrounds are fine enough to be resuspended and increase PM concentrations.
	Moreno et al. (2014)	Air quality in schools has notable spatial and temporal variations. High concentrations of traffic-carbon and metal PM into the classroom.
	Reche et al. (2015)	Indoor and outdoor BC levels depend on the distance to traffic.
	Reche et al. (2014)	Schools near traffic showed 40% higher indoor and outdoor UFP concentrations. High indoor UFP contributions from cooking, cleaning, and surface chemistry reactions mediated by O_3.
	Rivas et al. (2015)	High infiltration of air pollutants, with maximum infiltration observed for BC and Cd.
	Rivas et al. (2014)	School concentrations of BC, NO_2, UFP and, partially, $PM_{2.5}$ where the influenced by traffic emissions. Intermediate levels between UB and traffic stations were observed in schools.
Children's personal exposure	Nieuwenhuijsen et al. (2015)	The correlation between modelled (LUR) and measured personal black carbon levels was generally good, except for commuting times.
	Rivas et al. (2016)	School contributes to 37% of children's daily dose. Commuting periods have the highest dose:time intensity.
Aerosol instrumentation	Viana et al. (2015)	Good performance of three portable monitors for BC, UFP, and PM mass concentrations when compared with reference stationary monitors.
Air pollution and cognitive development	Alvarez-Pedrerol et al. (2017)	Exposure to $PM_{2.5}$ and BC during commuting by foot was associated with a reduced growth of working memory
	Basagaña et al. (2016)	From 9 different $PM_{2.5}$ sources, traffic was the only one associated with a reduction in cognitive development.
	Forns et al. (2016)	TRAPs at school were associated with increased behavioural problems and noise with more ADHD symptoms.
	Sunyer et al. (2015)	Children attending schools with higher TRAPs had a reduced improvement in cognitive development.
	Sunyer et al. (2017)	Short-term exposures to TRAPs were negatively associated with attention.
Gene-environment interactions	Alemany et al. (2016)	Involvement of the *PID1* gene, mTOR signalling and Alzheimer disease-amyloid secretase pathways in attention functions.
	Alemany et al. (2018)	For *APOE ε4* allele carriers, TRAPs were associated with higher behaviour problems and smaller reductions in inattentiveness, while no or weak associations were observed in *APOE ε4* noncarriers.
Air pollution and brain (MRI)	Mortamais et al. (2017)	Exposure to PAHs is associated with reduction in the caudate nucleus volume. No significant associations between PAH and ADHD symptoms.
	Pujol et al. (2016)	TRAPs were associated with brain changes of a functional nature, with no evident effect on brain anatomy, structure, or membrane metabolites.
Greenness and cognitive development	Dadvand et al. (2015a)	There was a beneficial association between exposure to green spaces in school and cognitive development, partly mediated by a reduction in exposure to air pollution.
Greenness and brain (MRI)	Dadvand et al. (2018)	Lifelong exposure to greenness was positively associated with grey and white matter volume in different regions of the brain.

MRI: Magnetic Resonance Imaging; BC: Black Carbon; UFP: ultrafine particles; PM: Particulate Matter; LUR: Land use Regression Models; TRAPs: Traffic-related air pollutants.

summarise the findings of the subprojects across the BREATHE Project (listed in Table 1) with the aim to discuss potential interventions at urban schools to lessen the negative effects of air pollution on children's neurodevelopment.

2. Data collected

Participants were recruited through cluster sampling by first selecting 39 schools in Barcelona (Catalonia, Spain) and then inviting all students without special needs in grades 2 through 4 (7–10 years of age) to participate (Sunyer et al., 2015). Most of the participants lived in Barcelona city, with some of them residing in suburban areas from the Barcelona Metropolitan Area. Participating children (n = 2897) from the 39 high and low TRAPs schools, paired by socio-economic status, were tested via a series of four computerized tests from January 2012 to March 2013 to evaluate working memory development, executive attention, impulsivity, and selective attention (Sunyer et al., 2015). Behavioural problems (Strengths and Difficulties Questionnaire) were reported by parents. Teachers reported Attention Deficit and Hyperactivity Disorder (ADHD) symptoms of each child using the ADHD Criteria of Diagnostic and Statistical Manual of Mental Disorders, fourth edition (ADHD-DSM-IV) list. From teacher ratings, we classified the children as having ADHD if 6 or more symptoms were present (López-Vicente et al., 2016). MRI (T2, flair, spectroscopy, and DTI) and fMRI (resting, visual and audition stimuli) were conducted in 265 children (Pujol et al., 2016). To assess gene-environment interaction, DNA samples were obtained from saliva samples from 2492 children, from which a subset of 1778 was selected for Genome Wide Association study (GWAs) (Alemany et al., 2016). A similar protocol to assess working memory and attention was applied to the 9-year follow-up of the INMA -Infancia y Medio Ambiente (Environment and Childhood) - birth cohort children (Gascon et al., 2017) to replicate the results in the near future.

Air pollution (nitrogen dioxide (NO_2; Gradko dosimeters), ultrafine particle number (UFP; DiSCmini, Matter Aerosol), Black Carbon (BC; MicroAethAE51, Aethlabs), and particulate matter (PM) $\leq 0.25\,\mu m$ (quasi-ultrafines), 0.25 to 2.5 μm (accumulation mode), 2.5 to 10 μm (coarse mode; all the previous fractions with a Sioutas Personal Cascade Impactor), $\leq 2.5\,\mu m$ ($PM_{2.5}$; with a MCV high volume sampler)) was measured during two one-week campaigns simultaneously inside the classroom and on the playground in each school pair during 2012 (Rivas et al., 2014). A total of 1092 PM filters were collected and more than 50 inorganic and organic compounds and elements were analysed (including organic carbon (OC), elemental carbon (EC), Al_2O_3, Ca, Sr, Fe, Mg, Cu, Sb, Sn, As, Co, Pb, Cr, and Polycyclic Aromatic Hydrocarbons (PAHs)). The same pollutants were also monitored in a reference urban background station in Barcelona (UB-PR). Note that for UFP, instruments with different size range were used and therefore the

Table 2
$PM_{2.5}$, NO_2, BC, UFP, and PAH concentrations for school hours (except for NO_2, which is for 24 h) of the 39 schools (indoor and outdoor), and the urban reference station of UB-PR. Modified from Rivas et al. (2014) and Mortamais et al. (2017).

	School classroom			School playground			Urban background (UB – PR)		
	Mean (SD)	Median	Range	Mean (SD)	Median	Range	Mean	Median	Range
$PM_{2.5}$ (μg m^{-3})	37 (13)	33	13–84	29 (20)	23	10–111	17 (7)	15	10–38
NO_2 (μg m^{-3})	30 (12)	30	5–69	47 (17)	46	14–98	41 (15)	38	23–97
BC (μg m^{-3})	1.3 (0.6)	1.2	0.4–2.7	1.4 (0.6)	1.2	0.4–2.6	1.3 (0.6)	1.2	0.6–2.7
UFP (10^3 cm^{-3})	16 (7)	15	4–31	23 (10)	21	10–56	15 (5)	13	6–33
Total PAHs (ng m^{-3})*	1.71 (1.11)	1.49	0.48–5.22	1.46 (0.70)	1.22	0.60–3.24	NA	NA	NA
B[a]P (ng m^{-3})*	0.11 (0.07)	0.10	0.02–0.43	0.10 (0.06)	0.09	0.02–0.30	NA	NA	NA

B[*a*]P: Benzo[*a*]pyrene; NA: Not available.
* Data for 35 schools.

number concentrations are not directly comparable between schools and UB-PR. The performance of online instruments was positively assessed by Viana et al. (2015). Traffic noise in the classroom and traffic intensity at school entrance was directly measured (Forns et al., 2016). Residential air pollution exposure was modelled using Land Use Regression (LUR) Models (Wang et al., 2013). In addition, we carried out personal measurements of BC during 48 h in a subsample of 45 children, who were carrying a belt bag with a MicroAeth AE51 and a GPS for tracking location. Personal measurements allow for a more accurate determination of the exposure, but it requires an intensive fieldwork and the instruments may become a burden for the participants.

3. Effects of traffic air pollution at school on neurodevelopment in the BREATHE project

Overall, school air is relevant for a healthy brain development. Children attending schools with higher TRAPs (largely diesel pollutants such as elemental carbon (EC) and UFP), had a smaller improvement with age in cognitive development in all measured cognitive functions. For instance, children attending schools with high pollution levels had a 7.4% (95% confidence interval (CI) [5.6%–8.8%]) 1-year improvement in working memory versus an improvement of 11.5% (95% CI [8.9%–12.5%]) in children in low pollution levels (Sunyer et al., 2015). Similarly, TRAPs were associated with more frequent behavioural problems (Forns et al., 2016). As an example, an interquartile range increase (IQR) of indoor EC (IQR = 1.01 μg m^{-3}) was associated with an adjusted mean ratio of 1.07 (95% CI [1.01, 1.12]) with similar results for outdoor EC and outdoor NO_2. Results did not change when adjusted for noise. From the different sources identified for fine particles, only those generated from traffic showed an association with cognitive development (Basagaña et al., 2016). An IQR increase (3.8 μg m^{-3}) in indoor traffic-related $PM_{2.5}$ was associated with reductions in cognitive growth equivalent to 30% (95% CI [6%, 54%]) of the annual change in working memory. EC and NO_2, which are traffic tracers, were associated with lower functional integration and segregation in key brain networks using neuroimaging which indicates slower brain maturation (Pujol et al., 2016), and total PAHs and Benzo[*a*]pyrene (B[*a*]P) were associated with a decrease in the caudate nucleus volume (CNV). For instance, an IQR increase in outdoor (0.067 ng m^{-3}) and indoor (0.076 ng m^{-3}) B[*a*]P concentration was significantly linked to a decrease in CNV (mm^3) ($\beta = -150.6$, 95% CI [$-259.1, -42.1$], and $\beta = -122.4$, 95% CI [$-232.9, -11.8$], respectively; Mortamais et al., 2017). These chronic relationships were independent of the acute effects, though the short-term exposures to TRAPs (the day before) were also associated with daily fluctuations in attention (e.g., an IQR increase of ambient NO_2 was associated with the responses being 14.8 ms slower (95% CI [11.2, 18.4]); Sunyer et al., 2017). Furthermore, noise inside the classroom is related to ADHD symptoms, but the effects of TRAPs were independent of noise (Forns et al., 2016; Sunyer et al., 2017). The associations between TRAPs and neurobehavioral outcomes were modified by the *APOE ε4* allele, with those children that were carriers of the *APOE ε4* allele showing the associations, while non carriers showed weak or no associations (Alemany et al., 2018).

In addition, we proved that green space is beneficial for brain maturation (function and structure) (Dadvand et al., 2015a; Dadvand et al., 2018). For instance, we observed an improvement in the annual change in working memory associated with greenness within school boundaries (9.8, 95% CI [5.2, 14.0]). Furthermore, the exposure to $PM_{2.5}$ and BC from the commutes to schools by foot was also associated with a cognitive impairment since an IQR range increase in $PM_{2.5}$ and BC concentrations during children's commute decreased the annual growth of working memory by 5.4 (95% CI [-10.2, -0.6]) and 4.6 (95% CI [-9.0, -0.1]) points, respectively (other transport modes could not be evaluated; Alvarez-Pedrerol et al., 2017). The evidence gathered from the BREATHE project on the associations between the exposure to air pollutants in schools and impaired neurodevelopment calls for measures to abate TRAPs concentrations at schools to endorse the protection of child brain maturation.

4. Air quality at BREATHE schools

4.1. Characteristics of air quality at schools

4.1.1. Concentrations of air pollutants and infiltration of outdoor pollution

For all schools, average school-hours daily concentrations in indoor (and outdoor) school's environment ranged between 13 and 84 (10 − 111) μg m^{-3} for $PM_{2.5}$, 6–69 (14–98) μg m^{-3} for NO_2 (24 h averages), 0.4–2.7 (0.4–2.6) μg m^{-3} for BC, 4–31 (10–56)·10^3 cm^3 for UFP, and 0.48–5.22 (0.60–3.235) ng m^{-3} for total PAHs (Table 2; Rivas et al., 2014, Mortamais et al., 2017). The range of concentrations and variability for BC, NO_2 and UFP measured in the 39 schools was higher outdoors, due to the higher influence of emission sources and meteorological factors in the outdoor environments. On the contrary, although traffic is an important source of PAHs, we observed higher concentrations of PAHs indoors, probably due to the preservation of these components in the absence of sunlight, since their transformations are driven by photochemistry (data not published). Average concentrations in the school playgrounds were generally higher than in the urban background station of UB-PR: 29 and 17 μg m^{-3} of $PM_{2.5}$; 47 and 41 μg m^{-3} of NO_2; 1.4 and 1.3 μg m^{-3} for BC, and 24 and 15·10^3 cm^3 of UFP; school playground and UB-PR, respectively (Table 2).

Concentrations of BC, NO_2, and UFP in schools generally showed an increasing gradient towards the city centre, following the traffic density in the city and pointing to the high contribution of the traffic source (Fig. 1). Consequently, schools in areas with higher vehicle intensities showed 30–35% higher BC levels. Outdoor BC concentrations at schools was significantly correlated with the percentage area used for the road

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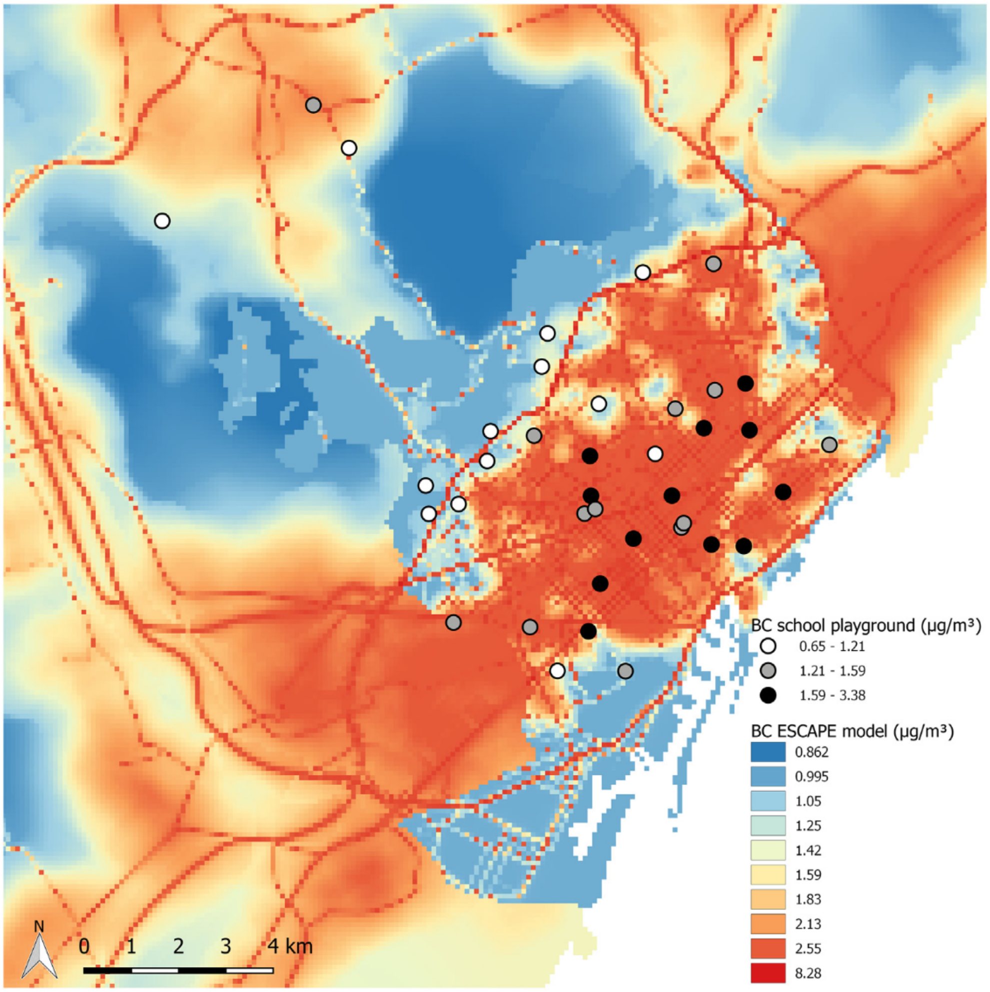

Fig. 1. Map of Barcelona and the schools by tertiles of annual Black Carbon (BC) concentration average. The background of the map is coloured according to modelled BC for Barcelona within the ESCAPE Project (Eeftens et al., 2012).

network in each district (R^2 = 0.61) (Reche et al., 2015). Higher indoor than outdoor BC levels were recorded at some schools when the indoor sampling location was relatively closer to a main road. These two facts indicate the strong dependency of the BC levels on the distance to traffic.

On the other hand, important school local sources affecting $PM_{2.5}$ concentrations prevent $PM_{2.5}$ from being a good indicator of traffic emissions in schools (Amato et al., 2014; Rivas et al., 2014). It is important to highlight that these local sources were also responsible for the bulk $PM_{2.5}$ concentrations in the schools being much higher (nearly double) than the typical concentrations recorded in the urban background of Barcelona (Rivas et al., 2014).

Different indoor-to-outdoor concentration patterns were observed for the different pollutants. On average, concentrations on the playgrounds were 1.6 times higher than indoors for NO_2 and 1.5 times higher for UFP, while BC concentrations were similar in both environments. $PM_{2.5}$ had 1.6 times higher concentration indoors because OC (the most important contributor to indoor $PM_{2.5}$), Ca, and Sr were importantly generated indoors. NO_2 showed a similar infiltration in the warm and cold seasons (50% and 56%, respectively), thus independently of the windows opening or closing (Rivas et al., 2015). Indoor-to-outdoor correlations showed low R^2 and infiltration factors for UFP because of indoor particle sources (which was indicated by high intercepts in the linear regressions). However, indoor levels of UFP were still influenced by outdoor levels as well as by average ambient temperatures (Reche et al., 2014).

4.1.2. *Sources of air pollution*

A source apportionment analysis by Positive Matrix Factorization (PMF) allowed the identification of eight factors or sources (*mineral*, *traffic*, *road dust*, *secondary sulphate and organics*, *secondary nitrate*, *sea spray*, *heavy oil combustion*, *metallurgy*) which corresponded to well-known sources of PM in the study area, plus a ninth factor named *organic/textile/chalk* which was observed for the first time (Amato et al., 2014).

The *organic/textile/chalk* source was the largest source in classrooms, contributing to 45% of indoor $PM_{2.5}$ (16.0 $\mu g\,m^{-3}$). It was characterized by OC (from cotton fibres, skin flakes, etc.), and Ca and Sr (from blackboard chalk) (Moreno et al., 2014; Rivas et al., 2014). Other studies in schools also reported very high concentrations of OC (Braniš and Šafránek, 2011; Fromme et al., 2008) as well as Ca and Sr from chalk use (Chithra and Shiva Nagendra, 2013; Dorizas et al., 2015; Fromme et al., 2008). In playgrounds, this source was still significant (16% on average; 5.3 $\mu g\,m^{-3}$), while on the contrary the contribution in the urban background station was below 1%. Therefore, this source is mostly a school-specific indoor source, characteristic of a crowded environment with an intensive use of chalkboards.

The *mineral* factor was strongly dependent on the type of playground (high concentrations for sandy playgrounds - 16 $\mu g\,m^{-3}$ outdoors - and lower for the paved ones - 2.5 $\mu g\,m^{-3}$) and showed unusually high levels of mineral matter in $PM_{2.5}$. Children's activity in sandy playgrounds may result in the grinding of mineral particles into smaller sizes, which then becomes a concern for air quality due to dust resuspension in the $PM_{2.5}$ fraction (H.Valido et al., 2018). The mineralogy determines the particle size distribution: a higher content of quartz implies a coarser size distribution. Therefore, the selection of sands with minerals of coarser size may diminish the potential impact on emissions due to resuspension. In classrooms, the highest concentrations of *mineral* were observed during the cold season, because of the accumulation of these particles due to closed windows and more indoor activities (favouring a continuous resuspension of deposited indoor particles). Other studies around the world have also observed high $PM_{2.5}$ or mineral concentrations in schools, due the high number of pupils and their high physical activity levels, which raises resuspension (Almeida et al., 2011; Blondeau et al., 2005; Chithra and Shiva Nagendra, 2013). The type of window seemed to be importantly associated with higher indoor levels of mineral components (such as Al_2O_3, Fe, Mg) and components with a very high contribution from indoor sources (OC, Ca, Sr) in those schools with aluminium or PVC windows (Rivas et al., 2015). Therefore, the presence of a more insulating window (such as the Al/PVC framed instead of wood framed) would be an important barrier for the dispersion of mineral components, which might keep resuspended in such a crowed indoor environment. Much lower *mineral* contributions were found in the urban background of Barcelona city (0.6 $\mu g\ m^{-3}$) than in schools, also indicating that this is mainly a local source at schools. Thus, *mineral* and *organic/textile/chalk* sources were responsible for the very high bulk $PM_{2.5}$ concentrations in the indoor environment (37 $\mu g\ m^{-3}$) and for almost doubling concentrations in playgrounds (29 $\mu g\ m^{-3}$) than in UB-PR (17 $\mu g\ m^{-3}$).

Motor exhaust emissions (OC, EC) and metals from brake wear (Cu, Sb, Sn and Fe) were the main components of the *traffic* factor (Amato et al., 2014). Contributions from *traffic* emissions were quite similar at the three studied environments, although a higher influence of this source is observed at schools than in the urban background: classrooms (4.8 $\mu g\ m^{-3}$), playgrounds (5.5 $\mu g\ m^{-3}$) and urban background air quality monitoring station (4.1 $\mu g\ m^{-3}$). Regarding the high concentrations of the *traffic* source indoors, BC (a traffic tracer) showed one of the highest infiltration of all $PM_{2.5}$ components, with the 92% of indoor BC coming from the outside during the warm season and 75% during the cold one (Rivas et al., 2015). This very high infiltration for BC was also observed in homes in Winsor (Canada; MacNeill et al., 2012). These results point out the necessity to locate future schools far away from trafficked streets.

Many trace elements had low or no correlation with BC (traffic tracer) and Al_2O_3 (tracer of mineral elements), which indicates a source other than traffic or crustal emissions, such as the *heavy oil combustion* (mostly from shipping emissions, with an average contribution of 0.6 and 0.7 $\mu g\ m^{-3}$ in the classroom and in the playground, respectively) and the *metallurgy* (1.0 and 1.2 $\mu g\ m^{-3}$, classroom and playground, respectively) factors identified by PMF. On the other hand, some elements such as As, Co, and Pb were quite correlated with mineral matter, suggesting that mineral matter could be polluted by dry and wet deposition of these pollutants on the playground and retained by absorption on crustal elements (Minguillón et al., 2015). Moreover, some of the trace metals were affected by significant indoor sources in a number of schools (Rivas et al., 2015). Cr should be highlighted, since it had higher levels indoors in both seasons. Further research is required in order to identify indoor sources of Cr and other trace metals, some of which are especially relevant due to their toxicity.

Besides the influence of traffic emissions, real-time measurements of UFP evidenced the contribution from cooking activities. Moreover, significant increases in indoor UFP concentrations (up to three times higher than outdoors) were observed after school hours, probably due to cleaning activities (Reche et al., 2014) that enhance the secondary particle formation by reaction between infiltrated O_3 and gaseous emissions from the cleaning products (cleaning product choice is also important; Singer et al., 2006). Cleaning activities were also identified as a significant source of UFP in schools in Brisbane (Australia; Mazaheri et al., 2016). Other O_3-reactive chemicals present in surfaces and materials, such as wood furniture and paints, can also lead to the formation of UFP (Weschler, 2006). Furthermore, midday increases of between 15 and 70% of the UFP concentrations in school playgrounds was also partly attributed to new particle formation by photochemical processes that took place all year round but with higher intensity during spring and summer, when the solar radiations is highest. Generally, indoor UFP number concentrations were lower than outdoors, and to some degree, explained by outdoor UFP concentrations as evidenced by multivariate linear regression (Reche et al., 2014).

4.1.3. The effect of vegetation (greenness) on school air quality

Schools having more vegetation (higher greenness levels) within school boundaries and the surroundings consistently showed lower indoor and outdoor concentrations of NO_2, UFP, BC, and the $PM_{2.5}$ contribution from the traffic source (from PMF), all of them being traffic-related air pollutants (TRAPS) (Dadvand et al., 2015b). Those schools with higher number of trees around them had a stronger reduction of TRAPs concentrations. It is still not clear if this is due to a trapping effect of the vegetation or to the lower emission in areas with lower space proportion dedicated to traffic as reported by Reche et al. (2015). The reduction of the indoor concentrations was partly mediated by the reduction of concentrations on the playgrounds.

4.2. Personal BC measurements in schoolchildren

Hourly BC concentrations were higher and the range was wider in personal measurements than in schools owing to peak concentration events that took place mainly during commuting time. Children spent 6% of their time on commuting but received 20% of their daily BC dose, due to co-occurrence with road traffic rush hours and the proximity to the source. In fact, the geometric mean of personal BC concentrations were significantly higher during commuting time (2.0 $\mu g\ m^{-3}$) than during periods when children were in the classroom (1.2 $\mu g\ m^{-3}$) or in the school playground (1.0 $\mu g\ m^{-3}$). This is in accordance with Buonanno et al. (2013), where the highest dose intensity was also found during commuting time in children's personal measurements from Cassino (Italy). As an average, children received 37% of their daily-integrated BC dose at school (21% in the classrooms and 16% in the playgrounds). Indoor environments (classroom and home) were responsible for the 56% BC dose. The relationship between personal monitoring and fixed stations at schools (indoor and outdoor) and in UB-PR was also evaluated (Rivas et al., 2016). Exposure could be significantly different, even between children attending the same school, as a result of the different time-activity patterns of each child and this variability could not be taken into account only with the fixed stations. We also evaluated the relationship between modelled home and school BC estimates and personal BC exposure levels in different micro-environments (home, school, and commute) obtaining a generally good correlation, with the exception of commuting times (Nieuwenhuijsen et al., 2015).

5. Actions to demand better air quality in schools

In the last few years, there has been a boost of actions to demand or to promote a reduction of air pollution levels in the school surroundings which have started within the society, organisations, and at local/regional authorities. For instance, at a national level, the French Government has recently published a Program of measures for the improvement of indoor air quality with some actions targeting schools. The Program focuses on the reduction of indoor emissions (from cleaning products, furniture and other surfaces) but also in the protection against outdoor air pollution and promoting awareness within the school community (French Ministry of Ecological and Inclusive Transition, 2018).

At a municipal level, many localities have started to plan or introduce measures to improve air quality in the school surroundings. London is a good example, with one of the most ambitious plans to improve air quality. Besides introducing the world's first Ultra-Low Emission Zone (ULEZ) from 8 April 2019 in Central London, the Mayor of London announced that 50 primary schools located in areas exceeding legal limits of NO_2 will be assessed to identify key interventions to reduce the exposure of the children while running a pollution awareness-raising education program at each school (Mayor of London, 2017). In Spain, many cities (e.g. Barcelona, Sabadell, Granada, Zaragoza) have include specific measures to protect schools in their Program for the Improvement of Air Quality. In regions or cities where air pollution reaches extremely high levels, such as Delhi and Beijing, local and regional authorities have cancelled classes during pollution peaks to prevent the children from commuting in such a toxic air (Kausar, 2017).

Moreover, schools and communities around schools (e.g. families, teachers, school managers) are getting involved into activities to promote clean air and to make students and their parents aware of the threat of the exposure to high levels of air pollutants. For instance, in London, there is a campaign for banning parents driving their children to school, as a way to reduce emissions around schools and to promote active and public modes of transport (Taylor, 2018). In Belgium, a group of parents who wish to live in healthier cities have started an action named Filter-Café-Filtré (filter-café-filtré.be) and is calling other parents from around the country to organise street closings at the entrance of their children's school every Friday morning before the start of the school while have a coffee together in a road empty of cars.

In a direct link with the results from the BREATHE project, besides direct communications with the City Council of Barcelona, BREATHE researchers have been approach by different schools' actors (e.g. teachers, students) to ask for recommendation to improve school air quality, ask for collaboration on new air quality measurements and the assessment of the improvement by some measures taken. Moreover, there are also local and regional civil movements for better air quality that have been working bringing attention on the importance of reducing air pollution at schools. For instance, some of the BREATHE researchers have participated in the project *Enlaira't* (http://www.enlairat.org) that the Plataforma per la Qualitat del Aire in Barelona has carried out in secondary schools in which the students think about air pollution and how to improve air quality in their city.

6. Recommendations to improve air quality in schools

The previous results allow the suggestion of measures and recommendations that could be of interest for urban planners and public policymakers, as well as for school managers and families. All or some of these measures should be effective in urban areas around the world where traffic is a significant source of air pollution, although different measures might be needed depending on local sources or school characteristics of specific regions. The efficiency of these measures in reducing air pollutants in schools needs to be quantified as, to the knowledge of the authors, there are yet no studies assessing the effect of these interventions. The most important recommendations from this list are summarised in Table 3.

- Since the exposure to traffic-related pollutants depends on distance to road traffic, future schools should be located away from trafficked roads.
- Road traffic density should be lessened around existing schools to diminish children's exposure to air pollutants, especially if schools are surrounded by canyon streets. Avoid congestion caused by bottlenecks (e.g. at the entrance or exits from a major road/highway) around the school.
- The classrooms where children spend most of their time should not be facing the busiest road, but facing an interior patio or the calmest street around the school.
- When traffic cannot be controlled, air intake for classroom ventilation should take either filtered air or fresh air from farthest away from the road traffic, both at the maximum possible height and maximum horizontal distance.
- In areas affected by high O_3 concentrations, if mechanical ventilation is turned on from April to September (when O_3 concentrations are the highest), O_3 traps should be installed on the system.
- High levels of textile, chalk and organic particles measured in $PM_{2.5}$ are due to high children density. Therefore, ventilation is advised, but only in cases when the classroom is not directly oriented to a major road. If the latter is the case, ventilation should be done during few minutes when children are not present in the classroom and avoiding traffic peak hour.
- Greening the school may help to abate exposure. Increasing the green and pedestrian spaces in the surrounding area would result in diminishing the proportion of the area used by cars and consequently would yield to lower levels of pollution.
- When selecting species for greening the schools low VOC and pollen emitting species should be selected.
- Parents and children should avoid major roads (in terms of traffic density) for commuting to and from school. Walking in the most exterior part of the pavement (furthest away from traffic) should be advised.
- Pedestrian school pathways should be implemented and designed to go through low traffic streets or at a distance to the kerbside of roads, in order to increase security and minimise children's exposure to air pollutants.
- The use of public instead of private transport for commuting would lead to the reduction of the number of cars around the school and consequently emissions would be abated.
- Periodic replacement of sand from the playgrounds (every one or two years) is advised because atmospheric scavenging of pollutants results in the accumulation of those on the playground sand. Also children activity on the playground results in the size of the mineral dust becoming finer over time which affects $PM_{2.5}$ levels.
- Sands with low clay and high feldspar or quartz content should be used in schools with sand-filled playgrounds to avoid producing fine $PM_{2.5}$. However, the emissions of $PM_{2.5}$ from the sands need to be

Table 3
Summary of the most important recommendations ordered by ease of implementation.

Recommendation	Target:	To be implemented by:
Raise awareness within and outside the school community of impacts on children's and public health of air pollution and spread measures to reduce the use of private cars	TRAPs and other pollutants	School managers, teachers, students, parents
Promote active travel or public transport to commute to school	TRAPs	School managers, teachers, students, parents, transport planners
Clean the classrooms after school hours (opening the windows) and select a ‘green’ cleaning product	VOCs and UFP	School managers, cleaning staff
Reduce traffic in school surroundings, increase greening	TRAPs	Urban designers, transport planners
Clean and replace sand from the playgrounds periodically	Mineral and traffic pollutants	School managers
Move schools or classrooms away from traffic.	Traffic emissions	School managers, urban designers

studied.

- Construction materials, paints, and furniture with a low VOC emission profile should be used to build or remodel schools to avoid high exposure to VOCs such as formaldehyde and to reduce the effect of the VOCs-O_3 reactions that may result in indoor air quality deterioration.
- Cleaning activities might help to reduce mineral matter resuspension in the indoor environments. However, since the cleaning products that are usually employed might react with O_3 to form new particles (in the range of UFP and carbonyl VOCs), cleaning works are recommended to be carried out in the afternoon after school hours (while having the windows open) to avoid children being exposed to additional concentrations of UFP.
- For ensuring better indoor air quality we also recommend the use of cleaning products with low proportion of ozone reactive constituents (e.g. use of pine oil-based instead of orange oil-based cleaning products; Singer et al., 2006) or more sustainable cleaning options (e.g. 'green' products, vinegar and baking soda).
- Raising awareness of the health impacts of air pollution in the school community (children and their parents, teachers, etc.). Monitors could be placed in school to get the students and parents involved and become an active agent of change by choosing and encouraging others to avoid using the private car when other options for commuting are available.

Our recommendations as well as the measures included in the different programs mentioned are in line with the EPA Best Practices for Reducing Near-Road Pollution Exposure at Schools (US-EPA, 2015).

It is obvious that there exists concern about the detrimental effects that air pollution may pose to schoolchildren. Hence scientists, teachers, parents, and other civil society stakeholders are raising concern, promoting, and demanding changes in the current mobility patterns based on an extensive use of private motorised vehicles in order to protect the health of children and, consequently, that of all the population. We believe that these recommendations are a useful starting point to ensure a better health of children.

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