Characterizing the patterns and predictors of exposure to urban air pollution and environmental noise among retired adults in Bucaramanga, Colombia

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Abstract

Background: Air pollution and environmental noise are major environmental pollutants, though most studies have been conducted in European and North American cities. By comparison, there are very few studies of exposure to air pollution and environmental noise in Latin American cities, which are rapidly urbanizing. Setting-specific exposure assessment studies are needed to understand the sources and determinants of air pollution and environmental noise exposures, which vary within and between cities and regions.

Objective: The aim of my thesis was to characterize levels and determinants of exposure to fine particulate matter ($PM_{2.5}$), black carbon (BC), and indoor noise among retired adults living in Bucaramanga, a medium-sized city in Colombia.

Methods: We enrolled 78 retired adults from four neighbourhoods in Bucaramanga that represented a range of traffic settings including high traffic/high diesel, low traffic/high diesel, high traffic/low diesel, and low traffic/low diesel/high braking. We measured 48-hr personal exposure to $PM_{2.5}$ and BC; 48-hr indoor levels of $PM_{2.5}$, BC, and noise; 5-day outdoor levels of $PM_{2.5}$ and BC; and administered detailed questionnaires related to potential housing and socio-demographic determinants of these pollutants. To explore potential determinants of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor equivalent sound pressure level (Leq), we built multivariable random effects regression models with a neighborhood-specific random-intercept.

Results: Mean (± SD) personal exposures to $PM_{2.5}$ and BC were 13.5 ± 5.9 $\mu g/m^3$ and 2.4 ± 0.5 $\mu g/m^3$, respectively, and were very similar to indoor $PM_{2.5}$ (12.7 ± 5.6 $\mu g/m^3$) and BC (2.5 ± 0.5 $\mu g/m^3$) concentrations. Traffic or diesel levels did not correlate with levels of $PM_{2.5}$ or BC. However, the neighbourhood that was high traffic/high diesel had the highest $PM_{2.5}$ levels. Mean indoor Leq in the four neighbourhoods ranged from 53.0-57.1 dB(A)) and indoor Leq was only weakly correlated with indoor air pollution (range of Pearson's r: -0.34-0.20). Higher level of indoor (household) air pollution and higher socioeconomic position (SEP) were determinants of higher personal exposure to $PM_{2.5}$ and BC, whereas natural household ventilation predicted lower levels. Women, compared to men, tended to also have notably higher exposure to $PM_{2.5}$, but this was not the case for BC. Higher SEP and household mechanical ventilation (i.e., use of a fan) predicted higher levels of indoor Leq.

Conclusion: Exposures to $PM_{2.5}$ among retired adult participants in urban Colombia were well-below the WHO's 24-hr guideline, yet their exposures to BC and noise were high. These results indicate that considering only exposures to $PM_{2.5}$ may underestimate the health risks of environmental pollution in our study setting. Efforts to reduce environmental exposures for this vulnerable population should target indoor (household) air pollution concentrations, indoor noise sources, and household ventilation factors.

Résumé

Contexte: La pollution atmosphérique et le bruit ambiant sont des polluants environnementaux majeurs, bien que la plupart des études aient été menées dans des villes européennes et nord-américaines. En comparaison, il existe très peu d'études sur l'exposition à la pollution atmosphérique et au bruit environnemental dans les villes d'Amérique latine, qui s'urbanisent rapidement. Des études d'évaluation de l'exposition spécifiques au milieu sont nécessaires pour comprendre les sources et les déterminants de la pollution atmosphérique et des expositions au bruit environnemental, qui varient à l'intérieur des villes et des régions et entre elles.

Objectif: Le but de ma thèse était de caractériser les niveaux et les déterminants de l'exposition aux particules fines ($PM_{2.5}$), au carbone noir (BC) et au bruit intérieur chez les retraités vivant à Bucaramanga, une ville de taille moyenne en Colombie.

Méthodes: Nous avons inscrit 78 adultes retraités de quatre quartiers de Bucaramanga qui représentaient une gamme de paramètres de circulation, y compris un trafic élevé/diesel élevé, un trafic faible/diesel élevé, un trafic élevé/diesel faible et un trafic faible/diesel faible/freinage élevé. Nous avons mesuré l'exposition personnelle de 48 heures aux $PM_{2.5}$ et BC; niveaux intérieurs de $PM_{2.5}$, de BC et de bruit durant 48 heures; niveaux extérieurs de $PM_{2.5}$ et de BC durant 5 jours; et administré des questionnaires détaillés sur le logement et facteurs sociodémographiques, des déterminants potentiels de ces polluants. Pour explorer les déterminants potentiels de l'exposition personnelle à $PM_{2.5}$, l'exposition personnelle à BC et le niveau de pression acoustique équivalent intérieur (Leq), nous avons construit des modèles de régression à effets aléatoires multivariables avec une interception aléatoire spécifique au quartier.

Résultats: Les expositions personnelles moyennes (± ET) aux $PM_{2.5}$ et BC étaient de 13,5 ± 5,9 $\mu g/m^3$ et 2,4 ± 0,5 $\mu g/m^3$, respectivement, et étaient très similaires aux $PM_{2.5}$ d'intérieur (12,7 ± 5,6 $\mu g/m^3$) et BC (2,5 ± 0,5 $\mu g/m^3$). Les niveaux de trafic ou de diesel ne correspondaient pas aux niveaux de $PM_{2.5}$ ou de BC. Cependant, le quartier à trafic élevé/diesel élevé présentait les niveaux de $PM_{2.5}$ les plus élevés. Le Leq intérieur moyen dans les quatre quartiers variait de 53,0 à 57,1 dB (A)) et Leq intérieur n'était que faiblement corrélé à la pollution de l'air intérieur (plage de r de Pearson: -0,34-0,20). Un niveau plus élevé de pollution de l'air intérieur (des ménages) et une position socio-économique (SEP) plus élevée ont été des déterminants d'exposition personnelle plus élevée aux $PM_{2.5}$ et BC, tandis que la ventilation naturelle des ménages prévoyait des niveaux plus bas. Les femmes, comparativement aux hommes, avaient également tendance à être considérablement plus exposées à $PM_{2.5}$, mais ce n'était pas le cas pour BC. Un SEP plus élevé et une ventilation mécanique domestique (c'est-à-dire l'utilisation d'un ventilateur) prédisaient des niveaux plus élevés de Leq intérieur.

Conclusion: Les expositions à $PM_{2.5}$ chez les participants adultes à la retraite dans notre étude à Bucaramanga en Colombie étaient bien-dessous la ligne directrice quotidienne recommandé par l'OMS, mais leur exposition au BC et au bruit était élevée. Ces résultats indiquent que la prise en compte uniquement de l'exposition à $PM_{2.5}$ peut sous-estimer les risques pour la santé de la pollution de l'environnement dans notre cadre d'étude. Les efforts visant à réduire les expositions environnementales de cette population vulnérable devraient viser les concentrations de pollution de l'air intérieur (domestique), les sources de bruit intérieur et les facteurs de ventilation domestique.

Preface

My thesis organization is as follows. I start with an introduction (Chapter 1) and then transition into a description of my three main objectives (Chapter 2). Following this is a literature review (Chapter 3), where I situate my research aims into the existing research on urban air pollution and environmental noise in low- and middle-income countries. Chapter 4 is a manuscript on levels and determinants of personal exposure to urban air pollutants (fine particulate matter ($PM_{2.5}$) and black carbon (BC)) and indoor noise among retired adults living in the city of Bucaramanga, Colombia. This manuscript contains a detailed methods section, results, and the main contents of my discussion. I plan to submit this manuscript to the journal *Indoor Air*. Chapter 5 consists of a discussion and conclusion for my thesis. I also provide a discuss the strengths and limitations of my study, in addition to proposing future steps in this section.

Contributions of authors

I, Kabisha Velauthapillai, am the author of this thesis, including the manuscript that will be submitted to the journal *Indoor Air*. As my supervisor and co-supervisor, Dr. Jill Baumgartner and Dr. Ellison Carter provided extensive and critical revisions on the manuscript and larger thesis. Dr. Scott Weichenthal is on my committee and also provided critical revisions on the contents of this thesis. Dr. Mauricio Victor Herrera, Dr. Kento Magara, and Ms. Skarlet Vasquez, along with Dr. Jill Baumgartner, Dr. Ellison Carter, and Dr. Scott Weichenthal designed the larger study.

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I want to start by acknowledging my family who, in both explicit and implicit ways, remind me of why I do the work that I do, hold the frameworks that I do, and stay committed as much as I do.

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Dedication

I dedicate my thesis to all the folks out there who have several barriers up against them in academic institutions. To people who feel like this place is not for them and not for people like them. To all the people who have been told, who have been shown, both explicitly and implicitly and repetitively, that they do not belong.

You have a place here. You make a difference here, whether you feel it, or not. Your very presence challenges the status quo. You are paving a path for the generations to come. I know it is tough but go back to the seed that got you to where you are. Remember why you are here and why you need to keep going. Collectively, we'll shift mountains. Keep your spirit alive.

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List of abbreviations or acronyms

BC: black carbon

CO: carbon monoxide

CO_2: carbon dioxide

dB(A): A-weighted decibel levels

HIC: high-income countries

LMIC: low- and middle-income countries

PM: particulate matter

$PM_{2.5}$: fine particulate matter

ppm: parts per million

L10: sound pressure level surpassed for 10% of the measurement period

L50: sound pressure level surpassed for 50% of the measurement period

L90: sound pressure level surpassed for 90% of the measurement period and represents the background noise level

Lday: sound pressure level from 7:00 – 19:00

Lden: weighted sound pressure level that includes day measurements, evening measurements with a 5 dB(A) penalty, and night measurements with a 10 dB(A) penalty

Leq: equivalent sound pressure level

Levening: sound pressure level from 19:00 – 23:00

Lmax: maximum sound pressure level

Lmin: minimum sound pressure level

Lnight: sound pressure level from 23:00 – 7:00

SEP: socioeconomic position

$\mu g/m^3$: micrograms per meter cubed

Chapter 1: Introduction

1.1 Low- and middle-income countries experience disproportionately high levels of urban environmental pollutants

Urban air pollution is associated with a range of health outcomes over the life-course, including adverse birth and pregnancy outcomes [1,2], childhood asthma [3], adult cardio-respiratory diseases, and early mortality [4]. Increasing evidence [5-8], largely from studies in Europe and North America, indicates that exposure to noise may also pose health risks, including the development of cardiovascular diseases and poorer mental health [9,10].

Increasing traffic is a major global contributor to these two urban environmental pollutants [11-16]. Urban population growth, strengthening economies, and the expansion of urban settings all contribute to increases in the number of motor vehicles in use [17-19]. While the average number of vehicles per 1000 residents has only slightly increased in cities located in high-income countries (HICs), there has been a prominent increase in vehicles in low- and middle- income countries **(**LMICs) [15,20-24]. In comparing 27 HIC cities with 5 LMIC cities, the Union Internationale des Transports Publics found that from 1995 to 2012, on average, there was a 12% increase in motorization (number of cars per 1000 residents) in HICs compared with an increase of 117% in LMICs during the same period [22]. We expect that increases in the number of vehicles on the road will impact urban air pollution and environmental noise, but there is currently very little research on co-exposures to urban air pollution and environmental noise in LMICs, a research gap that this thesis aims to address.

1.2 Air pollution

Global levels

Outdoor and urban air pollution levels are substantially higher in LMICs compared with HICs [17-19,25,26]. In most HIC countries, annual average population exposure to $PM_{2.5}$ falls below the WHO's guideline of 10 $\mu g/m^3$ [19]. Between different LMICs annual $PM_{2.5}$ levels can be as low as a mean of 10 $\mu g/m^3$ but as high as 100 $\mu g/m^3$ in certain settings. **Figure 1** shows annual outdoor $PM_{2.5}$ levels in various countries in 2017 [27]. Most air pollution studies conducted in

LMICs are based in Asia, partly due to the high levels of pollutants and the availability of resources to conduct this monitoring [28]. Data on exposure to air pollution in non-Asian LMIC urban areas, such as Latin American regions, are still needed.

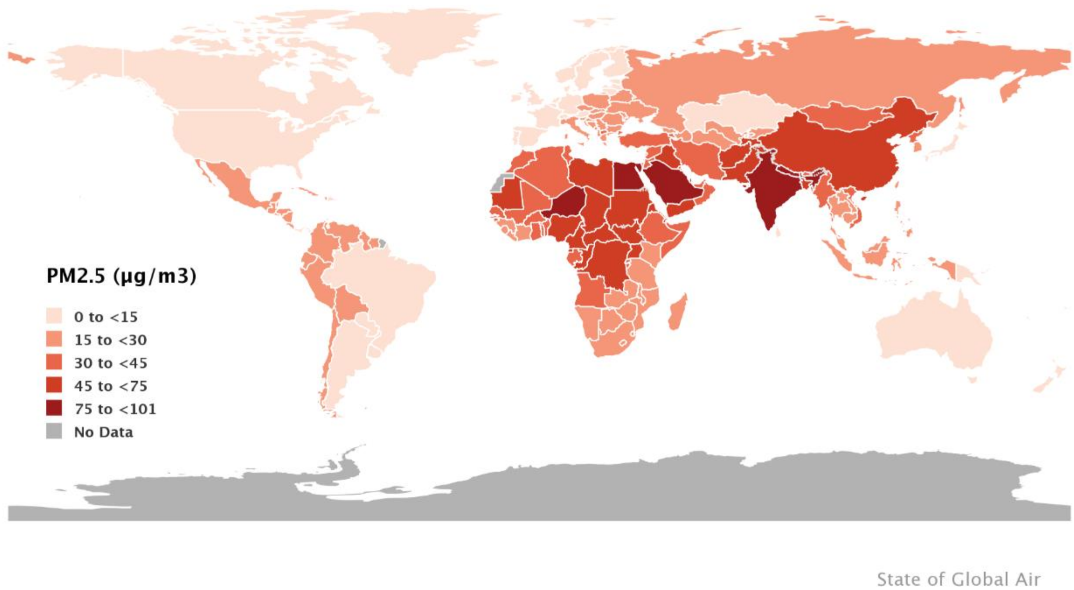

Figure 1: Average annual population-weighted $PM_{2.5}$ concentrations in 2017 [27]

Sources of urban air pollution

Globally, the major anthropogenic source contributors to urban $PM_{2.5}$ include traffic, household solid fuel combustion for cooking and heating (primarily), agricultural burning, and industry [16]. The chemical composition of air pollution in settings with different sources of air pollution can vary drastically between HIC and LMIC regions as well as between countries in the same region [19]. In South Asia and Latin America, for example, traffic contributes to >34% and > 30% of $PM_{2.5}$ mass, respectively. By comparison, traffic contributes up to an estimated 17% of outdoor $PM_{2.5}$ in Africa (based on 8 studies)[16] and 15-25% of outdoor $PM_{2.5}$ in North America and Europe (based on 65 studies) [16]. These findings demonstrate that traffic is an important source of $PM_{2.5}$ and that there are variations in traffic contributions between LMICs and HICs and within these regions. Exposure assessment in different geographic areas is required to develop localized understandings of air pollution in areas less-frequently studied.

Urban air pollution from traffic sources

Vehicle combustion (engine fuel combustion) and non-combustion sources (brake and tire wear or resuspended road dust) are both sources of air pollution [15]. Many air quality studies have focused on tailpipe emissions (combustion sources), in part due to the focus of regulatory standards and policies on tailpipe emissions [29,30]. Resuspended road dust, tire wear, and brake wear (non-combustion sources) are increasingly important sources of air pollution, as tail-pipe emissions are decreasing with pollution abatement policies or interventions [21,29,31,32]. Differences in vehicle fleet (vehicle age, vehicle types, and emissions) and maintenance [13,33,34], the quality of roads and fuel, and urban form are factors that can differentially impact exposures across urban settings [15,21,35]. These factors can be very different between and within urban settings in LMICs and HICs [23]. Thus, to avoid extrapolating findings from one setting to other dissimilar settings, what is needed is more targeted research on urban air pollution in LMICs.

1.3 Environmental noise

Global levels

Recognition of environmental noise as a significant human health risk is growing, particularly in European cities [36]. The WHO regional office in Europe considers noise to be an environmental stressor of growing concern [37]. A 1999 WHO report recommends a 55 dB(A) limit guideline for noise exposure in outdoor living environments and a 35 dB(A) limit for indoor environments [38]. A-weighted sound pressure levels (dB(A)) use the A logarithmic scale to provide a relative measure of sound [36]. Mean estimated road traffic noise levels in European cities ranged from 40-75 dB(A) [39]. Levels in Europe remained relatively constant since 2012, but over 20% of the European population lives in places where noise exceeds safe levels [40]. In 2018, the WHO Europe identified noise as a leading environmental stressor based on increasing epidemiological findings on the adverse health effects of noise, including hearing impairments and poor mental health [11,41]. The WHO Europe subsequently established guidelines to reduce road traffic, railway, and aircraft noise to below 53, 54, and 45 dB(A), respectively [41].

Studies on levels and exposures to environmental noise are concentrated in HICs [42,43], though measurements in different settings show that exposure to noise is typically higher in urban areas of LMICs [38]. With increasing recognition of environmental noise as a prominent health risk, exposure assessment studies in LMICs are also increasing. More studies on noise mapping are being conducted in Latin America, Africa, and Asia [37,42,44-48]. However, most noise measurements are conducted outdoors [48-51] rather than indoors where urban residents spend at least half their time [52-54].

Noise Sources

Common sources of outdoor environmental noise in urban areas include transport, industry, construction, schools where children play outside, entertainment, and people [55-58]. Transportation (airplanes, trains, and traffic) is a major source of environmental noise in urban settings [11,26,40,43], with traffic being a notable source in urban settings within both HICs and LMICs [40,59]. Vehicle engines; the tire friction against road surfaces or surrounding air; and vehicle speed and type all contribute to noise [11,59,60]. Features of the urban built environment, such as building heights, may also influence noise propagation [44,45,61]. As with air pollution, more location-specific research, especially in LMICs, is required to understand the impacts of the built environment on environmental noise, since sources and determinants of pollutants (e.g. vehicle fleet, building sound insulation) can vary between settings. Studies on indoor noise sources and the relative contributions of these sources are limited [62], making it more challenging to target indoor noise abatement policies.

1.4 Traffic as a major source of urban air pollution and environmental noise

Research on traffic-related co-exposures to environmental hazards has found that both air pollution and noise are independently associated with negative health outcomes, highlighting that joint exposure may result in a joint additive or multiplicative effect on health [63-65]. Studies based typically in cities of HICs (e.g., London, England; Thessaloniki, Greece; and Vancouver, Canada) have assessed co-exposures to air pollution and environmental noise among adults and children [7,63,64,66-70]. Correlations between exposures to air pollution and environmental noise ranged from 0.05-0.83, with lower correlations for PM and higher correlations for nitrogen oxides (NO_x) [12,63,70-73]. Higher correlations for NO_x may be due to traffic being a larger source of NO_x compared to PM [74]. Whether the strength in associations between air pollution and environmental noise are similar for urban settings in LMICs is unknown and requires further inquiry. The varying sources of air pollution and environmental noise, as well as the varying composition of air pollution in different settings may lead to varying effects on population health

[64]. Setting-specific research for urban air pollution and environmental noise, and the relationships between these two urban health risks, are necessary to better understand these exposures in LMICs.

1.5 Environmental exposure assessment

Exposure studies provide insight into the levels, types, sources, and determinants of pollutants. These studies are vital for estimating exposure-response relationships in epidemiological studies, identifying vulnerable populations, and understanding sources of exposure [67,75-77]. Additionally, exposure assessment, such as assessing levels and determinants of pollutants, can direct policy-making and clarify the impacts of interventions or policies aimed at reducing environmental exposures [75,78,79]. Exposure assessments of urban air pollution and environmental noise in rapidly expanding cities of LMICs are vital for targeting health and transport-related policies in those areas.

Chapter 2: Objectives

Urban air pollution and environmental noise are prominent health risks that affect most of the world's population. These risks are of increasing concern in cities in LMICs amidst rapid urbanization and increasing traffic volumes, yet most studies of personal exposure to air pollution and indoor noise were conducted in Europe and North America. There is relatively little knowledge of the levels and determinants of urban environmental pollutants in LMICs, which are likely to be different from those in HICs. Localized research can more accurately inform health risk assessments and policy development. Detailed exposure assessment studies in urban settings within LMICs can fill this gap. Such studies can further inform epidemiologic and environmental health research, as well as the design and implementation of policies and interventions aimed at reducing exposures to environmental risks.

Geographic scope of research

Over 80% of the population in Latin America already lives in urban areas, and urbanization continues to increase [80,81]. Megacities tend to receive the most attention for urban health risks, such as air pollution, whereas characterization of these risks is more limited in rapidly growing mid-sized cities, which is where urbanization is most rapid in LMICs [82]. Municipal governments in Latin America introduced a number of policies and interventions to curb air pollution from transport sources over the past several decades [81,83]. For example, in certain Colombian cities, such as Bogotá, vehicles with certain license plates cannot be driven on specific days [83,84]. Exposure assessments of environmental pollution in Latin American cities are scarce, but such studies are instrumental to informing effective policies on air pollution and environmental noise abatement.

2.1 Main objectives

- To characterize the levels of personal exposure to fine particulate matter ($PM_{2.5}$) and black carbon among retired adults living in neighborhoods with different features of traffic in a Colombian city.
- To quantify levels of indoor noise in households of retired adults living in neighborhoods with different features of traffic.
- To evaluate the environmental, housing, and socio-demographic determinants of personal exposures to air pollutants ($PM_{2.5}$ and black carbon) and indoor noise.

Chapter 3: Literature review

I used the Scopus database to search for articles. Search queries included the following terms in combination “air pollution”, “personal exposure”, (“fine particulate matter” OR PM2.5), (“black carbon” OR BC), (indoor AND noise OR “environmental noise” OR “indoor noise” OR “residential noise” OR “noise pollution”), (determinant* OR predictor* OR predict*), (“South America” OR “Latin America” OR brazil OR mexico OR colombia OR argentina OR peru OR venezuela OR chile OR guatemala OR ecuador OR bolivia OR honduras OR paraguay OR "el salvador" OR nicaragua OR "costa rica" OR panama OR uruguay), (“developing countr*” OR “LMIC*” OR “low-income countr*” OR “middle-income countr*” OR “low* and middle*income countr*”), (urban OR city OR cities), (window* OR ventilation OR gender OR socioeconomic OR “socioeconomic status” OR SES OR income OR animal* OR pet* OR crowd* OR traffic OR vehicle*), (outdoor OR ambient OR indoor OR exposure OR personal OR infiltration), (elder* OR retire* OR “older adult*”). A total of 1363 studies were identified. Additional search criteria, such as looking specifically at reviews and omitting conference abstracts, were used in certain searches. I excluded studies if they were not relevant to the specific search that was conducted. 198 studies were identified based on abstracts and only relevant studies were kept for the literature review. Further studies were identified by consulting references or consulting the “cited by” list.

3.1 Urban air pollution

Air pollution is ubiquitous in urban atmospheres and levels are highest in LMICs [17-19,25,26]. Fine particulate matter (particles with an aerodynamic diameter of 2.5 μm or less, $PM_{2.5}$) is a major air pollutant found in cities [85]. Black carbon, a by-product of incomplete combustion, water soluble ions (e.g. SO_4^{2-}, NO_3^-, and NH_4^+), and various trace elements (e.g. Al, Cu, Fe, Mg, Pb) are among the components of $PM_{2.5}$ [15,86,87]. BC is especially relevant for the urban setting, since it is emitted from traffic sources [85] and has been shown to be associated with adverse health outcomes, independent of exposure to $PM_{2.5}$ mass. However, individual pollutant levels and their relative contributions to $PM_{2.5}$ mass can vary depending on sources and determinants of these pollutants.

The World Health Organization (WHO) set an annual guideline of 10 μg/m^3 for ambient $PM_{2.5}$ and a daily guideline of 25 μg/m^3. Similar guidelines do not exist for BC. In LMICs, levels of air pollutants tend to be higher than the WHO's annual guideline [17-19,25,26]. East Asia, South Asia –mainly China and India-, the Middle-East, and West, Central, and North Africa have the highest levels of $PM_{2.5}$ (> 35 μg/m^3), with levels in Nepal that can go as high as 100 μg/m^3 [19]. South-East Asian, Caribbean, and Latin American regions have comparatively lower levels of exposure to $PM_{2.5}$, typically within the 10-15 μg/m^3 range, and sometimes reaching the 15-25 μg/m^3 range or higher [19], which is closer to $PM_{2.5}$ levels in HICs.

3.2 Urban noise

Environmental noise, defined as noise that is generated by non-workplace environments, is a growing concern in urban settings [36,88]. Regional, national, and municipal bodies have set guidelines for environmental noise in cities. Based on epidemiological studies of noise and health outcomes, such as hearing impairment, sleep disturbances, and cardiovascular effects, the WHO recommended maximum nighttime outdoor levels of 55 dB(A) over a decade ago [89]. More recently, the WHO developed a new set of guidelines for Europe, recommending that road traffic, railway, and aircraft noise remain below 53, 54, and 45 dB, respectively [41]. Furthermore, noise limits or guidelines for indoor environments are less common. In 1999, WHO Europe established indoor noise level recommendations of 35 dB(A) [38].

With growing recognition of the negative impacts of excess environmental noise, several cities in Latin American also developed guidelines for outdoor noise. Santiago, Chile has a recommended outdoor daytime noise limit of 65 dB(A) [90]. In Curitiba and Rio de Janeiro, Brazil, residential areas have outdoor average noise limits of 55 dB(A) during the day and 45-50 dB(A) during the night [51,91,92]. Mixed-use areas with both residences and urban services, such as shops, have slightly higher noise limits: 55-60 dB(A) during the day and 55 dB(A) at night [51,91,92]. The maximum allowable noise limit in residential areas of Mexico is slightly higher: 67 dB(A) [93]. In Colombia, maximum allowable outdoor residential levels of noise during the day and night are 65 and 55 dB(A), respectively [94,95]. While noise regulations are present at national and international levels, the degree to which these regulations and policies are enforced has not been studied.

3.3 Health impacts of air pollution and environmental noise

The WHO recognizes both air pollution and environmental noise as leading environmental health risks [26,41]. Acute and long-term, chronic exposures to air pollution and environmental noise have been associated with negative health impacts across the life-course [96], which I summarize in the sections below.

3.3.1 Air pollution

In pregnant women, a 5 μg/m^3 increase in outdoor $PM_{2.5}$ was associated with an increased risk of preeclampsia during the first trimester (relative risk (RR): 1.07, 95% CI: 0.95-1.00), while throughout the pregnancy, it was associated with increased hypertensive disorders (RR: 1.18, 95% CI: 1.02–1.34) [97]. Assessing nine studies, Chun et al. (2020)[98] found that maternal exposure to $PM_{2.5}$ was associated with a higher odds of children developing the autism spectrum disorder (odds ratio (OR): 1.06, 95% CI: 1.01-1.11 or 1.13). Furthermore, a systematic review on cohort studies found positive associations between levels of outdoor $PM_{2.5}$ during pregnancy and the following birth outcomes: preterm delivery and low or decreased birth weight [99].

In children, air pollution has been associated with attention deficit, otitis media, rheumatic disease (systemic autoimmune rheumatic diseases and rheumatoid arthritis), and an increased risk of developing pneumonia [100-103]. Furthermore, in a systematic review of over 40 studies on exposure to traffic-related air pollution and asthma development in children, a 0.5 x 10^{-5}/m increase in BC and a 1 μg/m^3 increase in $PM_{2.5}$ were associated with an increased risk of asthma development of 1.08 (95% CI: 1.03 - 1.14) and 1.03 (95% CI: 1.01 - 1.05), respectively [3]. According to the Health Effects Institute (2010)[104], there is enough epidemiological evidence to go beyond an association and conclude a causal relationship between asthma exacerbation and traffic-related air pollution.

In adults, an increase of 10 μg/m^3 in long-term $PM_{2.5}$ was associated with increased risk of depression (OR 1.12, 95% CI: 0.97–1.29) [105]. Furthermore, chronic exposure to outdoor air pollution was also positively associated with type II diabetes, cognitive decline, and dementia [106]. Most epidemiological research on outdoor air pollution and health outcomes, such as the findings above, is generally based in HICs, highlighting the need for more studies and reviews that focus on LMICs [101,107].

A meta-analysis found that in urban areas in LMICs, every 10 µg/m^3 increase in long-term or same-day $PM_{2.5}$ was associated with 0.38 - 0.57% increases in respiratory, cardiovascular, or total mortality [4,108]. In 2017, air pollution was estimated to be the eighth largest contributor to premature mortality and the 10th largest contributor to disability-adjusted life years, globally [109]. Outdoor PM contributions to global disability-adjusted life years was 19.3% for chronic obstructive pulmonary disease was, 17.4% for all lower respiratory infections, and 12.6% for type 2 diabetes [109]. In 2015, $PM_{2.5}$-attributable deaths in LMICs ranged from 4.9-9.0% (Latin America 3.0-7.3%), while in HICs they were 4.3% [110]. Many of these studies assess outdoor air pollution levels, which may not accurately estimate the actual exposure of air pollution that people are experiencing.

3.3.2 Environmental noise

Acute and chronic exposures to environmental noise have also been associated with negative health impacts across the life-course [6,36]. In a systematic review, pregnant women and infants in Japan, North America, and Europe, transport-related noise has been associated with preterm birth, low birthweight, small for gestational age, and congenital abnormalities [111]. However, only 14 studies were assessed and all six studies on aircraft-related noise had a high likelihood of bias [111]. Over 20 studies on acute or chronic exposure to high noise levels found associations with decreased motivation, reading deficits, poorer memory, and higher blood pressure among children [36,112,113].

In adults, epidemiologic studies found that long-term exposure to noise was associated with increased blood pressure, glucose, lipid, adiposity, and blood viscosity, which are independently associated with the development of cardiovascular disease [36,114]. Among adults in Europe, chronic exposure to nighttime aircraft and daily traffic noise were associated with increased odds of hypertension (OR: 1.10-1.14) [115,116] or myocardial infarction [5]. Acute exposure to high-intensity noises (i.e., gunshot) and chronic exposure to noise levels of 75-85 dB(A) can damage hearing [36,112]. As with air pollution, the majority of studies on environmental noise and health were based in HICs, with limited research in LMICs. Furthermore, studies primarily use outdoor noise levels as the exposure of interest, despite findings that people in urban settings tend to spend at least half their time in indoor settings [52-54]

3.3.3 Epidemiologic studies of co-exposures to air pollution and environmental noise

Studies in Canada, Europe, and China have considered the potential associations of both urban air pollution and environmental noise with health outcomes [65]. Since traffic is a common major source of both pollutants, the association between one pollutant and a designated outcome may be confounded by the other pollutant [63,64]. However, by measuring both exposures, a study can address potential confounding [64]. Air pollution and noise have been independently associated with hospital admissions in children, heart rate variability, cardiovascular disease, and diabetes [63,66,70,73,117,118]. One study based in Canada studied potential interaction effects of air pollution and noise on mortality due to coronary heart disease but did not find any such effect [73]. However, this study was limited to black carbon, one health outcome, and one urban Canadian setting. Therefore, findings from this study may not be generalizable to other settings with differing sources and determinants of these urban environmental pollutants. These findings demonstrate the importance of studying multiple urban environmental exposures at the same time to better understand their joint and independent influences on health.

3.4 Levels of exposure to $PM_{2.5}$, exposure to BC, and indoor noise

3.4.1 Air pollution

There are relatively few studies of personal exposures to $PM_{2.5}$ or BC in urban areas of LMICs (**Table 1**). The majority of existing studies are based in Asian settings, where concentrations of $PM_{2.5}$ and BC range from 40-489 μg/m^3 and 3.7-14.7 μg/m^3, respectively. Studies assessing $PM_{2.5}$ in other settings fell within the above-mentioned range, while BC concentrations in Londrina, Brazil were lower than the range found in Asian cities. Furthermore, most studies in China focus on university students, whose daily exposures may differ from other age or occupation groups, potentially limiting the generalizability of these studies to other settings or populations.

Table 1: Studies on levels of personal exposure to air pollution in urban settings in LMICs.

Author & Year	Location	Study Population	Duration	Mean (μg/m^3)
Du et al. (2010)[119]	Beijing, China	114 school children and office workers	24-hr	$PM_{2.5}$: 102.5 ± 68.4 BC: 14.7 ± 6.8
Fu et al. (2019)[120]	Beijing, China	113 university students, aged 18-25	12-hr	$PM_{2.5}$: 156.2 ± 89.6
Chen et al. (2020)[121]	Guangzhou, Nansha, or Luogang districts in China (includes central urban, peri-urban, and rural)	16 people, of which 14 were college students, aged 18-30	24-hr	$PM_{2.5}$: 67.2 ± 28.8
Ren et al. (2019)[122]	Guangzhou, China	41 undergraduate students	3-day	$PM_{2.5}$: 60.3 ± 52.1
H. Zhou et al. (2020)[123]	Jinan, China	67 retired adults, aged 60-69	24-hr	BC: 4.1 ± 2.0
Chen et al. (2018)[124]	Shanghai, China	36 college students	3-day	$PM_{2.5}$: 39.9 ± 32.1 BC: 6.1 ± 2.8
Lei et al. (2016)[125]	Shanghai, China	51 graduate students	3-day	$PM_{2.5}$: 110 BC: 5.3
Y. Zhou et al. (2020)[126]	Wuhan and Zhuhai, China	158 participants with a total of 394 measurements	24-hr	$PM_{2.5}$: 89-168.5 §
Mehta et al. (2014)[127]	Ho Chi Minh, Vietnam (urban and peri-urban)	64 primary caregivers of young children (<5 years)	9 x 24-hr for each person	$PM_{2.5}$: 54.7-73.6 †
Habil et al. (2016)[128]	Agra, India	6 participants	24-hr	$PM_{2.5}$: 125.8-140.2 †
Pant et al. (2017)[129]	New Delhi, India	18 adults	48-hr	$PM_{2.5}$: 53.9-489.2 † BC: 3.7-23.3 †
Sanchez et al. (2019)[130]	Hyderabad, India (peri-urban)	402 adults, 639 measurements	24-hr	$PM_{2.5}$: 55.1-58.5 † BC: 4.6-6.2 †
Carvalho et al. (2018)[54]	Londrina, Brazil	12 people between ages 20-50	48-hr	BC: 2.4-2.5 †
Riojas-Rodríguez et al. (2006)[131]	Mexico City, Mexico	33 adults with ischemic heart disease	11-hr	$PM_{2.5}$: 46.8 ± 1.82
Wylie et al. (2017)[132]	Dar es Salaam, Tanzania	Pregnant girls and women aged 15 and older	2 x 24-hr	$PM_{2.5}$: 40.5 ± 21.2

‡ Median, not mean, † range of means, § range of medians

3.4.2 Environmental noise

Studies on environmental noise are fewer than studies on air pollution. Most studies on the subject are on outdoor noise [9] . Indoor Leq was below 60 dB(A) in all studies that assessed this measure, while Lnight levels were below 46 dB(A) (**Table 2**). The limited evidence on indoor noise exposure assessment suggest that levels of indoor noise tend to exceed the WHO's 35 dB(A) guideline [38]. Of the seven studies we found on indoor noise in urban areas, six were in HIC areas and usually in European countries.

Table 2: Studies on indoor noise levels

Author & Year	Location	Location of measurement	Mean (dB(A))
Pirrera et al. (2011)[133]	Brussels, Belgium	24 homes (71% being apartments), measurements in bedrooms	Leq: 41.1
Pujol et al. (2012)[134]	Besançon, France	44 households	Lday: bedroom 51.3 ± 5.0, main room 57.7 ± 5.0 Levening: bedroom 53.6 ± 5.8, main room 61.0 ± 6.0 Lnight: bedroom 36.9 ± 5.9, main room 45.7 ± 7.0
Kraus et al. (2015)[135]	Augsburg, Germany	109 households with 305 measurements that were valid	Leq: 59.6 ‡, †
Foraster et al. (2014)[136]	Girona, Spain	Indoor noise levels in bedrooms obtained by applying an insulation factor to modeled outdoor noise ●	Lnight: traffic-related 27.1 (16.2), railway-related 10.5 (21.6) §
Neitzel et al. (2016)[137]	Rockville, US	9 homes	Leq: 48.2-51.3 ¶
Park et al. (2017)[62]	Seoul, Korea	Apartment living rooms with no people inside and closed windows	Leq: 30.3 †
Meng et al. (2020)[138]	Harbin, China	45 bedrooms in university residence hall	Leq: 21-51 ¶

‡ personal exposure in an indoor setting: the home, † Median, § Median (interquartile range), ¶ Range, ● modeled indoor noise level, rather than measured

3.5 Determinants of exposure to urban air pollution and indoor environmental noise

3.5.1 Indoor and outdoor source determinants of exposure to $PM_{2.5}$ and environmental noise

Air pollution

Common outdoor sources of $PM_{2.5}$ are traffic-related emissions, fuel combustion, industrial sources, secondary aerosols, and crustal matter [139-142]. Traffic emissions contributed 7-49% in Europe [139], 17-22% in China [143], and 24-30% in Brazil and Argentina [141,144] to $PM_{2.5}$. Furthermore, fuel combustion or industrial emissions contributed to more than 30% of $PM_{2.5}$ [139] in Europe [139], but ranged from 7-21% in China, Brazil, and Argentina [141,143,144]. These studies highlight variations in $PM_{2.5}$ source contributions within regions and countries. Furthermore, they demonstrate that traffic may contribute more to $PM_{2.5}$ in LMICs than fuel combustion or industrial sources. While source apportionment findings in China covered all seasons [143], the literature from Europe and Latin America only covered a subset of seasons [139,141,144], which may limit comparability between findings in these studies.

Indoor sources of $PM_{2.5}$ and their contributions to personal $PM_{2.5}$ exposure relative to contributions from outdoor $PM_{2.5}$ can vary, depending on a range of factors, including the outdoor setting. For instance, indoor, household $PM_{2.5}$ sources in urban Chile were cooking and environmental tobacco smoke (ETS) (28%), cooking and cleaning (11%), and dust (7%), while outdoor sources that contributed to indoor $PM_{2.5}$ were traffic (26%), secondary sulfates (16%), and resuspended road dust (13%) [145]. This study in Santiago, Chile also found that traffic-related air pollution was a large contributor to indoor $PM_{2.5}$, which is a common finding in both HIC and LMIC studies on contributors to outdoor $PM_{2.5}$, Other sources of indoor $PM_{2.5}$ include biomass or solid fuel combustion due to cooking or heating [16,52,146] or incense and candle burning can also be sources of indoor air pollutants [146,147].

Environmental noise

Studies on environmental noise have generally discussed outdoor sources of noise more than indoor sources. Relative source contributions to noise in different settings has not been a focus of the existing body of literature on environmental noise. The primary sources of outdoor environmental noise in urban areas include transportation sources, such as aircrafts, trains, and road traffic [55], with road traffic recognized as the most prominent [11,88,148]. In studies based in urban Chile and Brazil, measured and predicted outdoor noise levels decreased with decreasing traffic flow [11,51,90,149-151]. Other sources of outdoor noise include industries, construction [58], schools [56], and entertainment, such as nightclubs and fireworks [91,152,153], and people, especially crowds [55,58]. In indoor settings, ventilation systems, heating systems, and appliances can be sources of noise [55,58,151]. Furthermore, speaker systems and, like in outdoors environments, people can also be sources of noise [55,58,151]. However, there appears to be a gap in research on indoor noise, and particularly, in LMICs. While traffic is a known contributor to outdoor noise, to my knowledge, there is no study exploring the relationship between traffic volume and measured indoor noise levels. Studies on outdoor noise levels may still inform indoor noise levels, but there are limitations. Factors such as building insulation and indoor noise sources can play significant roles in determining indoor noise levels, as described in **Section 3.5.3**.

3.5.2 Traffic as a source determinant of air pollution and environmental noise

Exhaust and non-exhaust-related sources and determinants of air pollution

Traffic can be separated into two different sources of air pollution: tailpipe emissions and non-exhaust-related emissions [32,154]. Exhaust emissions includes combustion of fuels, with diesel being a main source of BC [32]. Global mean BC emission factors from 1960-2006 were more than eight times greater for diesel fuel than for gasoline [155]. Non-exhaust emissions include brake wear, tire wear, road wear, and re-suspended road dust [32]. In a large review of 99 articles, most of which were based in North America, Europe, and Asia, non-exhaust emissions contributed to approximately 33% of $PM_{2.5}$ [156]. However, this review did not include studies in Latin America or Africa, where there is limited research on relative contributions of exhaust and non-exhaust-related sources to air pollution [156]. This is important because setting-specific factors such as climate and tire types (e.g. winter vs summer tires) can contribute to variability in the relative contributions of exhaust and non-exhaust sources to total $PM_{2.5}$ mass [157].

Exhaust and non-exhaust-related sources and determinants of environmental noise

Engines are also a source of noise and the intensity of this noise can depend on the engine type [91]. Tire against pavement friction is another source of urban noise, particularly for vehicles travelling at high speeds and for heavy-duty vehicles [158,159]. However, the relative contribution of these traffic-related sources to noise have not been studied extensively in non-laboratory settings, therefore, little is known about differences that may exist between high-income and low-income urban settings, which may vary with respect to the composition of vehicle fleet, a determinant in outdoor noise levels [94,160]. Again, localized research could clarify the determinants of noise in urban settings in LMICs.

3.5.2 Environmental, housing, and socio-demographic determinants of personal exposure to air pollution or indoor noise

3.5.2.1 Indoor or outdoor settings and air pollution

Different outdoor and indoor environments are associated with different levels and compositions of pollutants, and time spent in these environments may depend on varying socio-demographic characteristics including gender, age, and socio-economic status. These microenvironments can be outdoor or indoor, including commuting environments and specific places in the home or the workplace, such as a kitchen.

Research, based in both LMICs and HICs, has examined potential associations between outdoor concentrations of $PM_{2.5}$ or BC and personal exposure [121,123,124,127,130]. Studies on personal exposure to $PM_{2.5}$ or BC in urban LMIC areas found outdoor concentrations to be significant predictors of these pollutants [121,123,124,127]. A 1 $\mu g/m^3$ increase in outdoor BC concentrations has been associated with a 8.8 – 16.1% increase in BC [123,124], and a 0.3 – 1.3% [121,124] or 0.66 $\mu g/m^3$ increase in $PM_{2.5}$ [127]. While these studies agree that outdoor air pollution contributes to higher personal PM and BC exposures, the relative increase in exposure to $PM_{2.5}$ is lower than for BC. Furthermore, these studies were based in China or Vietnam, and therefore may not be representative of urban settings in Latin American countries.

Studies have also found that indoor concentrations of air pollutants can also be major determinants of personal exposure to $PM_{2.5}$ or BC. In HIC studies in urban South Korea and urban and rural Australia, residential contributions to personal exposure to BC were 51 – 64% [52,161,162]. Furthermore, research in HICs found that people tend to spend 80 - 98% of time indoors, while in LMICs, it was 50-78% [52-54], meaning that the findings from the above-mentioned studies may overestimate contributions of indoor concentrations to personal exposure in LMICs. Indoor contributions to personal exposure to $PM_{2.5}$ or BC are less well understood in urban areas in LMICs. Furthermore, while there are indoor sources of pollutants, outdoor concentrations also contribute to indoor concentrations of pollutants [52,53,146,163-166]. In brief, outdoor concentrations, indoor concentrations, and outdoor pollutants that contribute to indoor concentrations may all affect personal exposure to air pollutants.

3.5.2.2 Associations between ventilation-related variables and air pollution or environmental noise

Air pollution

Ventilation methods can affect levels of indoor air pollutants, influencing exposure. Indoor spaces can be ventilated through natural or mechanical means. Infiltration of outdoor air pollutants and exfiltration of indoor air pollutants can occur by travelling through open windows or cracks in buildings [53]. Fans, air conditioning units, air purifiers, and other household ventilation systems mechanically ventilate homes [123,127,146,167].

Poor ventilation can lead to higher concentrations of indoor pollutants if there are pollutants generated by indoor sources [146]. Some studies based in LMICs found positive correlations between $PM_{2.5}$ and air change rates (a measure of how well-ventilated a space is, with higher rates of air exchange generally representing better ventilation), while others reported positive relationships between indoor air pollutant concentrations and poor ventilation (e.g. low air exchange rates), particularly in settings with higher second-hand smoke levels [146,168]. Although research on these topics have been limited, the prevalence of mechanical ventilation in Latin American countries is generally lower than in HICs. Mechanical ventilation is designed to manage air supply and exhaust and may include filtration within the ventilation system or be timed to supply outdoor air at times when outdoor air pollution levels are lower. Thus, in studies that examine relationships between indoor air pollution levels and mechanical ventilation, we may find negative correlations. For example, in urban China, where households that used mechanical ventilation systems had an average of 4.89 $\mu g/m^3$ lower indoor concentrations of $PM_{2.5}$ [167]. Similarly, in urban China and Vietnam, mechanical ventilation was also associated with lower air pollution exposure (8.91% decrease in BC exposure or 0.01 ug/m^3 decrease in $PM_{2.5}$) [123,127].

However, among older adults in urban China, each hour with open windows was associated to a 0.013 $\mu g/m^3$ (0.010 – 0.017 $\mu g/m^3$) increase in personal exposure to BC [123]. Similarly, studies on commuters in urban China, Brazil, and Australia found that when windows were open in cars, buses, or taxis, BC exposure levels were 2.2 – 4.3 times higher [54,162,169]. Higher exposure to BC associated with open windows may be due to the significant sources of outdoor BC exposure: motor vehicles.

Environmental noise

Open versus closed windows can also make a difference in indoor noise levels [170]. Tilted or open windows are a primary way through which sound can enter buildings [170]. In Switzerland, median noise levels within homes for open, tilted, and closed windows were 10.0 ± 2.9 dB(A), 15.8 ± 2.7 dB(A), and 27.8 ± 4.4 dB(A), respectively [170]. In their prediction model for open/tilted windows, the strongest predictor of difference between indoor-outdoor Leq was outdoor noise level, which explained 55% of noise variability [170]. In the closed windows model, decreased indoor-outdoor Leq differences were associated with a higher number of windows [170].

3.5.2.3 Associations between housing features and environmental noise

Building-related factors can affect indoor noise levels. In the case of open/tilted windows, newer buildings were associated with decreased noise level differences between indoor and outdoor settings [170]. Building sound insulation (e.g. using window glazing and certain pane types) is a well-known determinant of noise [171]. Improved building façade sound insulation and, for closed windows, no window gaskets were associated with lower indoor noise levels in Italy and Switzerland [170,171]. However, in urban France, window type (glazing type or double window) was not associated with indoor noise levels in children's bedrooms [134].

Room type and size may also affect indoor noise levels. For open/tilted windows, Locher et al. (2018)[170] found that compared to bedrooms, kitchen/dining rooms and living rooms were associated with a 5 dB(A) and 1 dB(A) lower indoor-outdoor noise level difference, respectively. This difference may be due to sound-absorbing materials typically found in bedroom and living rooms (e.g. carpets, curtains, beds, sofas), whereas kitchen/dining rooms often have more surfaces that reflect sound [170]. Furthermore, when windows were open/tilted, bigger rooms (60 – 150 m^3) were associated with lower noise levels, leading to a larger difference between outdoor and indoor levels (+1.1 dB(A)) [170]. This may also contribute to the difference observed between room types, since living and kitchen/dining rooms tend to be larger than bedrooms. Similarly, Pujol et al. (2012)[134] also found main room indoor noise levels were significantly higher than levels in bedrooms.

In urban France, crowding was associated with outdoor noise levels, while increased number of people in the home, number of children in bedrooms, musical instruments, and televisions were associated with higher indoor noise [134]. De Noronha Castro Pinto & Moreno Mardones (2009)[91] and Souza & Giunta (2011)[152] also mention that human density in buildings can affect noise levels. The above-mentioned findings provide insight on associations between household-related factors and indoor noise or outdoor-indoor noise level differences, however, none are based in LMICs.

3.5.2.4 Gender and exposure to air pollution

Studies across many settings show that women, on average, tend to spend more time indoors than men [52,172-174] due to household-related responsibilities that can entail differential exposures to air pollutants. As a result, exposure to domestic environments (e.g. cooking and cleaning) may be different between men and women, which was the case in an urban Chinese setting [173]. While these latter studies focus on LMICs, they are all based in Asian, primarily rural, settings, where time spent indoors for different genders or sexes may differ from time-activity patterns in other urban regions or countries, such as those in Latin America.

In their $PM_{2.5}$ and BC exposure prediction models based in peri-urban India, Sanchez et al. (2019)[130] found that predictors for these pollutants varied by gender. For women, the predictors were cooking and household SEP indicators, with a 62% increase in BC exposure if biomass was used for the primary stove [130]. For men, predictors were more varied: occupation, time spent cycling (for $PM_{2.5}$), smoking (for $PM_{2.5}$), and presence of biomass cooking in home (BC) [130]. In contrast, studies in urban China and Brazil did not find differences in air pollution exposures by gender [54,124]. Setting-based differences may be related to the absence of biomass as a fuel source in these latter studies. Differences may also be due to varying gender roles in different urban settings, since in Londrina, Brazil, both women and men tended to commute for work. Additionally, the study in India was in a peri-urban setting, whereas the other two were in urban settings, where gender roles may vary.

Choice of transport mode can also vary by gender and the degree of variation can differ between urban settings within LMICs as well. In Suzhou, a city in China, women tended to commute more by bicycle, while men tended more toward cars [175]. While public transport use was low for both genders, women tended to use it more than men [175]. In a larger study on travel modes among women in LMICs, women were found to rely more on walking and informal or formal public transport [176]. Variation in choice of transport between genders may affect personal exposure to air pollution, as more time spent in commuting-related environments have been associated with high exposures, especially BC, since diesel vehicles are a source of this pollutant [52,146,177].

3.5.2.5 Neighborhood, household, and individual-level socioeconomic position (SEP)

Air pollution

Extensive environmental justice literature in North America highlights the inequities in exposure to air pollution, with higher exposures in lower SEP areas [178,179]. Higher levels of urban air pollutants and environmental noise were associated with lower SEP at primarily the individual level, but also at the neighbourhood level [178]. In contrast, a recent systematic review, found that the relationship between SEP and air pollution was less consistent in urban European settings. In London, for example, there was no trend between income and neighborhood PM or BC, and some of the highest income neighborhoods also had the highest levels of air pollution [180]. These findings demonstrate that trends between SEP and air pollution vary between settings, but this could also depend on whether the SEP indicator is at the individual, household, or neighbourhood/area-level and the specific type of indicator (e.g. income, education, or occupation).

Findings in LMICs are also inconsistent. In urban China and India, lower neighbourhood-level income, household-level SEP (income, occupation status of household head, and lack of dwelling or vehicle ownership), and individual-level SEP (education level) were associated with lower indoor concentrations or personal exposure to air pollutants [123,146,166,181]. However, prediction models in peri-urban India, urban Vietnam, or urban China, concluded a lack of association between exposure to $PM_{2.5}$ and BC and individual or household-level income indicators [123,127,130]. Little is known about the relationships between neighbourhood, individual, or household-level indicators and urban air pollution in LMIC regions, such as Latin America [178].

Environmental noise

Positive associations between lower SEP and higher levels of outdoor noise from traffic, aviation, and railway sources were observed in cities in Canada [182,183], Germany [184], England [185,186], France [187], Spain [188], Hong Kong [189,190], and the U.S. [112,191-194]. Other studies found varying associations between noise levels and composite SEP indicators. In urban France, noise levels were highest in census blocks in the intermediate categories of deprivation [195], while in the US, lower and higher deprivation were associated with higher levels of outdoor noise [196]. In urban England, some higher SEP populations experienced higher outdoor levels of transport-related noise, which may have been due to the benefits of living in locations with good transport [185]. These findings show the variability in associations between SEP indicators and noise levels, even in high-income settings.

One study in Besançon, France did assess indoor noise but found no association with household-level SEP indicators [134]. However, lower outdoor noise was associated with higher SEP of the more privileged parent [134]. The lack of associations between household-level SEP indicators and indoor noise levels may be due to the household-level SEP indicators being related to location of residence in a potentially higher traffic area, compared to a lower traffic area, while building insulation may be higher for residences in higher outdoor noise areas.

3.6 Motivation behind air pollution exposure and indoor noise prediction models

This literature review assesses the state of findings on two prominent urban environmental pollutants: air pollution and environmental noise. I described the variability in levels and potential determinants of air pollution or environmental noise, highlighting that findings between and within urban settings in HICs and LMICs can vary. A key finding of this literature review highlights the need for more field-based research on air pollution and noise in urban areas in LMICs, which has otherwise been lacking.

Several research gaps were identified. Firstly, there is a lack of empirical data on outdoor, indoor, and personal exposure to $PM_{2.5}$ and BC in urban settings in mid-sized Latin American cities. Second, while studies on outdoor noise levels are on the rise, there is a major empirical gap in indoor noise level-specific studies, especially in cities based in LMICs. Furthermore, studies on determinants of $PM_{2.5}$ and BC exposure are limited in urban areas in LMICs, with existing studies focused primarily in Asian settings. Prediction studies on indoor noise are few, both in HICs and LMICs. This study seeks to address the empirical gaps on $PM_{2.5}$ and BC exposure and indoor noise in a rapidly growing mid-sized city in Latin America.

Chapter 4: Levels and determinants of personal exposure to air pollution and indoor noise among retired adults in urban Colombia

There is a dearth of research on both air pollution and noise in urban Latin American settings where traffic is a major source of these pollutants. This is especially the case for rapidly growing mid-sized cities, such as Bucaramanga, Colombia. This chapter consists of a manuscript on the concentrations and predictors of personal exposures to air pollution ($PM_{2.5}$ and BC) and indoor noise in four neighbourhoods with varying traffic characteristics in Bucaramanga, Colombia. First, I present a background and the motivation behind the study objectives. I then present the detailed methods that were used for the study, followed by key results. I follow with a critical discussion and conclusion, contextualizing findings within the existing literature. The manuscript has been prepared for submission to the journal *Indoor Air*.

Levels and determinants of personal exposure to air pollution and indoor noise among retired adults in urban Colombia

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4.1 Abstract

Air pollution and environmental noise are major environmental pollutants in urban low- and middle-income country (LMIC) settings. However, there is an absence of research on co-exposures to air pollution and environmental noise in rapidly growing mid-sized Latin American cities. We aimed to characterize the levels and determinants of exposure to fine particulate matter ($PM_{2.5}$), black carbon (BC), and indoor noise among retired adults in Bucaramanga, Colombia. We enrolled 78 retired adults from four neighbourhoods in Bucaramanga that represented a range of traffic settings, and measured their 48-hr integrated gravimetric personal exposures to $PM_{2.5}$ and BC; 48-hr indoor levels of $PM_{2.5}$, BC, and noise; 5-day outdoor levels of $PM_{2.5}$ and BC. We also administered detailed questionnaires that obtained housing (e.g. ventilation characteristics) and socio-demographic information of households and participants potential determinants of these pollutants. Mean (± SD) personal exposures to $PM_{2.5}$ and BC were 13.5 ± 5.9 μg/m^3 and 2.4 ± 0.5 μg/m^3, respectively, while mean indoor equivalent sound pressure (Leq) in the four neighbourhoods ranged from 53.0-57.1 dB(A)). Traffic or diesel levels did not correlate with levels of $PM_{2.5}$ or BC. However, the neighbourhood that was high traffic/high diesel had the highest $PM_{2.5}$ levels. We also found that indoor concentration of $PM_{2.5}$ or BC was a major contributor to personal exposure, while household ventilation characteristics were protective against personal exposure. In contrast, household ventilation predicted higher indoor Leq. Furthermore, socioeconomic position predicted higher levels of all three pollutants. Our findings indicate that exposures to $PM_{2.5}$ among retired adult participants in urban Colombia were well-below the WHO's 24-hr guideline, yet their exposures to BC and noise were high. Considering only exposures to $PM_{2.5}$ may underestimate the health risks of environmental pollution in our study setting. Efforts to reduce environmental exposures for this vulnerable population should target indoor (household) air pollution concentrations, indoor noise sources, and household ventilation factors.

4.2 Background

Air pollution and environmental noise are prominent environmental pollutants in urban settings around the world. Strong evidence associates exposure to urban air pollution (fine particulate matter ($PM_{2.5}$) and (BC)) and environmental noise with a range of health outcomes over the life-course, including adverse birth and pregnancy outcomes [1,2], childhood asthma [3], adult cardio-respiratory diseases, poorer mental health [9,10], and early mortality [4,197]. Motor vehicles are a major source of both pollutants [11-16]. With rapid growth in traffic due to increasing urban populations, strengthening economies, and urban expansion, the risk of these pollutants to human health is also increasing [17-19]. Exposure studies provide insight into the levels and determinants of pollutants, which is vital to estimating exposure-response relationships in epidemiological studies, identifying vulnerable populations, and informing policy-making on pollution abatement [67,75-79].

Exposures to air pollution and environmental noise tend to be higher in low- and middle-income countries (LMIC) than in high-income settings [19,38], yet exposure studies on both pollutants are limited in LMICs, particularly for noise [63,68-73]. Furthermore, research on patterns of personal exposure levels and determinants of $PM_{2.5}$ and BC in urban settings in LMICs are predominantly based in Asian countries [121,123,127,146,181,198]. Environmental (e.g. traffic, indoor or outdoor microenvironments such as specific commuting environments), housing (e.g. natural or mechanical ventilation practices), and socio-demographic (e.g. gender or age) determinants have been found to affect personal exposure to $PM_{2.5}$ or BC in China, Vietnam, and India [121,123,127,146,181,198]. Personal exposure studies in Latin America have focused on commute-related exposures [54,199-203]. As for environmental noise, outdoor noise from traffic sources is the primary focus of the literature in LMICs [37,42,44,45], with several noise mapping studies conducted in urban Latin American settings [11,51,90,204,205]. The few studies on indoor noise exposure assessment are mostly based in high income areas and have largely focused on building features (e.g. open/closed windows, window type, televisions) when considering determinants of noise pollution [62,134,138,170].

Our study aimed to address these research gaps by: (1) characterizing the levels of exposure to air pollutants among retired adults living in neighborhoods with different features of traffic, (2) quantifying levels of indoor noise in households of retired adults living in these neighborhoods, and (3) evaluating the potential environmental, housing, and socio-demographic determinants of personal exposures to urban air pollutants ($PM_{2.5}$ and BC) and indoor noise in a Colombian city.

4.3 Methods

4.1 Study location and participant recruitment

We conducted a cross-sectional study in Bucaramanga, a rapidly growing mid-sized city in northeastern Colombia (7.1193° N, 73.1227° W) with a population of approximately 580 000 [206]. We identified four neighbourhoods with the following traffic features to include in our investigation: high traffic/high diesel, low traffic/high diesel, high traffic/low diesel, and low traffic/low diesel/high braking.

We enrolled retired adults from households in each study neighbourhood. Participants were eligible if they were over 60 years old, retired, had lived in their current home for at least 5 years, and were not currently a smoker. We limited participation to retired adults, who are more likely to remain in their neighbourhood environments during measurements and are also more vulnerable to the health impacts of air pollution [207]. We identified eligible participants by first selecting a random household in the neighbourhood that was within 50 m from the highest traffic road. If the household did not have an eligible or interested participant, staff moved to the neighbouring household. Staff introduced the study to eligible adults, and those interested in participating provided written consent and were enrolled into the study. Of the eligible residents identified in the high traffic/high diesel and low traffic/high diesel neighbourhoods, 37% and 27%, respectively, agreed to participate [*]. We obtained ethical approval for the study protocols from La Universidad Autónoma de Bucaramanga (Bucaramanga, Colombia) and McGill University (Montreal, Canada).

4.2 Data collection

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[*] Staff were not able to access the participation rates for the other two neighbourhoods due to COVID-19 building closures, but these other participation rates will be added prior to submitting the manuscript for publication.

Measurements were conducted during weekdays and on one weekend day per week in June and July of 2018. Following the initial recruitment visit, staff visited each participant on two occasions for data collection. During the first data collection visit, we administered a questionnaire and installed equipment to measure indoor (household) levels of air pollution and noise as well as personal exposure to air pollution. During the second visit, approximately 48-hr later, we retrieved the air pollution and indoor noise monitoring equipment and administered the household characteristics portion of the questionnaire, which inquired about recent behaviors including ventilation over the previous 48 hours. Staff visited 5-6 participant homes per day.

4.3 Outdoor (community) air pollution measurements

In each of the four study neighbourhoods, two environmental enclosures with outdoor air pollution monitors simultaneously measured outdoor concentrations over a period of 5 days (Monday morning through Saturday morning). Each enclosure included a set of air samplers that measured real-time and integrated gravimetric fine particulate matter ($PM_{2.5}$) and real-time black carbon (BC). We positioned the enclosures on tripods that were approximately 1 m off the ground on the second or third floor of an enrolled household. The enclosures were covered with sunshades to protect the equipment from rain and direct sunlight (**SI Fig. 1**).

We used an Ultrasonic Personal Aerosol Sampler (UPAS, Access Sensor Technologies, Fort Collins, CO, USA) to measure integrated gravimetric $PM_{2.5}$ [208]. The UPAS collected $PM_{2.5}$ mass on a 37 mm PTFE filter, with a pore size of 2.0 μm. The PTFE filters were first removed from sealed Petri-dishes and then loaded into clean UPAS cartridges, which were to be deployed the next day. Up until the point of transport, UPAS samplers were left face down on a clean surface to avoid contamination of loaded filters.

We transported loaded samples to the site in clean anti-static bags within plastic containers to reduce the possibility of contamination. Filter-based $PM_{2.5}$ mass samples were collected at a flow rate of 1 L/min, running on a duty cycle of 50% (to extend battery life). Flow rates were calibrated immediately before deployment, and measured following deployment, using a mini-Buck Calibrator M-5 (A-P Buck Inc., Orlando, Florida, USA). All sample flow rates were within ± 3% of the target flow rate following sample collection. In each neighbourhood, 2 x 48-hr outdoor $PM_{2.5}$ samples (Monday morning to Wednesday morning and Wednesday morning to Friday morning) and a 1 x 24-hr outdoor $PM_{2.5}$ sample (Friday morning to Saturday morning) were collected. At the end of the first 48-hr measurement period, we transported a clean UPAS cartridge with a new filter in an anti-static bag and exchanged the used filter and cartridge with the new one. We collected at least one field blank per neighbourhood, where a UPAS filter and its cartridge were brought to and from the outdoor air pollution field sampling site using the same protocol as for samples, but the monitors were not turned on. Post-measurement, all UPAS cartridges were unloaded, and each PTFE filter was placed into its initial Petri-dish. The Petri-dish was then sealed and kept at room temperature over the course of the study.

Real-time outdoor $PM_{2.5}$ concentrations were measured using laser photometers (DustTrak Model 8520, TSI Inc., Shoreview, MN, USA) that were zeroed and calibrated prior to each use of the device. We co-located the laser photometers with the filter-based samplers (i.e., UPAS) to correct the real-time $PM_{2.5}$ data for known, systematic bias associated with the measurements made by the laser photometer. We determined correction factors for each real-time $PM_{2.5}$ instrument (n = 3 total devices deployed) by dividing the time weighted average $PM_{2.5}$ concentration from the laser photometer by the $PM_{2.5}$ mass determined from the integrated gravimetric measurements (discussed below, in **Section 4.7**). The time-weighted average $PM_{2.5}$ concentration was evaluated for the same sampling duration over which the filter-based $PM_{2.5}$ measurement was collected. Device-specific correction factors were then applied to all real-time $PM_{2.5}$ measurements that were conducted by the given laser photometer.

Real-time BC was measured at a 5-min resolution using a microaetholometer (microAeth®, Model AE51, San Francisco, CA, USA) (**SI Fig. 2**). To reduce noise in the measurements, outliers in the dataset were corrected using the US Environmental Protection Agency's Optimized Noise-Reduction Algorithm (ONA) [209]. All outdoor air pollution instruments were powered by an electrical outlet.

4.4 Indoor air pollution

We measured indoor 48-hr integrated, gravimetric $PM_{2.5}$ mass using the same UPAS samplers and field procedures as for outdoor measurements. We placed the monitor on a table or shelf in living rooms (**SI Fig. 3**), at a height of approximately 0.5-1.5 m off the ground and away from direct emission sources, including kitchen windows/entryways and cleaning products. Participants provided guidance on placing the monitors in locations that would not disturb daily activities. One household was undergoing renovations in the living room, which was not in use, and the sampler was instead placed in the participant's bedroom. We collected 16 field blanks (17% of all indoor and personal samples collected, combined). In a random subsample of 16 households (21%), we also measured CO (carbon monoxide) and CO_2 (carbon dioxide) for the same 48-hr at a 1-min resolution using a Q-trak Indoor Air Quality Monitor (Model 7575, TSI Inc., Shoreview, MN, USA) (**SI Fig. 4**).

4.5 Indoor noise

We measured indoor Leq (equivalent sound pressure level) using a sound level meter/data logger (Noise-Sentry, Convergence Instruments, Sherbrooke, QC, Canada) (**SI Fig. 3**). The instrument provided A-weighted maximum, average, and minimum sound levels every 5 minutes over the 48-hr measurement period. Due to its small size (< 5 cm in height and width) we used a cable to connect the noise sensor to the indoor UPAS (**SI Fig. 3**) to prevent loss. The two monitors were kept at a distance of approximately 15 cm or more from each other to minimize potential noise from the UPAS that could affect recorded noise levels.

The noise metrics derived from the 48-hr real-time data in each household included: equivalent sound pressure level (Leq); maximum sound pressure level (Lmax); minimum sound pressure level (Lmin); sound pressure level from 7:00 – 19:00 (Lday); sound pressure level from 19:00 – 23:00 (Levening); sound pressure level from 23:00 – 7:00 (Lnight); weighted sound pressure level that includes day measurements, evening measurements with a 5 dB(A) penalty, and night measurements with a 10 dB(A) penalty (Lden); sound pressure level surpassed for 10% of the measurement period (L10); sound pressure level surpassed for 50% of the measurement period (L50); and sound pressure level surpassed for 90% of the measurement period and represents the background noise level (L90) [36,210].

4.6 Personal exposure to $PM_{2.5}$

We measured 48-hr integrated, gravimetric exposures to $PM_{2.5}$ in all participants with the UPAS samplers and using the same field procedures as for the other gravimetric measurements. Participants carried UPAS samplers in either a case on their upper arm (**SI Fig. 5a**) or in a waist pack (**SI Fig. 5b**) and were instructed to go about their usual daily activities, wearing the samplers at all times but could place them within a 2 m distance while sleeping, resting, or when not possible (e.g. bathing). A pedometer was co-located with the UPAS to count the number of steps taken by participants and evaluate compliance.

4.7 Filter preparation, $PM_{2.5}$ mass gravimetric analysis, and BC mass optical scans

The PTFE filters were analyzed for $PM_{2.5}$ mass before and after data collection at the University of Colorado in Fort Collins, USA. Filters were weighed in triplicate using a microbalance housed in a temperature (21-22 °C)- and humidity (30-34%)- controlled room. The average of the three weights was used for statistical analysis. Initial filter weights were subtracted from the post-sampling weights to obtain the $PM_{2.5}$ mass. To control for potential filter contamination, we corrected sample filters by subtracting the median field blank mass for each neighbourhood. Field blanks with a measure below the limit of detection (LOD) were replaced with ½ LOD (= ½ x 3 x standard deviation). The standard deviation was based on field blank masses that were > LOD. To obtain concentrations, we divided the final mass by the sampled volume.

Following gravimetric analyses, the PTFE filters were shipped to McGill University in Montreal, Canada for analysis of BC. Using an Optical Transmissometer Data Acquisition System (SootScan™ OT21 Transmissometer, Magee Scientific, Berkeley, CA, USA) we scanned the filters in triplicate and obtained the average infrared attenuation factor (IR ATN) of the three measures, which was then converted into mass loading (micrograms per cubic centimeter of filter surface area). We multiplied the mass surface loading by the total filter surface area and then divided that mass by the sampled volume to obtain concentrations. We adjusted filter-based BC concentrations using the outdoor real-time BC concentrations measured with the microaetholometer (microAeth®, Model AE51, San Francisco, CA, USA). By time-matching the outdoor time-integrated BC concentrations with the corresponding outdoor real-time BC data, we constructed a univariable linear regression model (**Eq. 1**). This regression model was used to estimate the BC concentration for each scanned filter.

$\boldsymbol{Eq.\,1}$: $BC_{aeth}\ (\mu g/m^3\ ) = 1.92 + 0.16(BC_{optical};\ \mu g/m^3\ )$

4.9 Neighborhood traffic counts

We recorded traffic at the busiest intersection in each study neighborhood for 5 days using a video camera (EZHusky Outdoor Bullet Camera, EZVIZ Inc., City of Industry, CA, USA). Staff identified households that were < 50 m from the intersection and went door-to-door to ask permission to mount the camera on a household's roof or balcony. If a household gave permission, we verified that the view of the intersection was unobstructed by trees, buildings, and other urban features. The camera was mounted on the railing of a balcony or roof and recorded video of the intersection during the same time period as outdoor air pollution measurements. A trained technician randomly selected one full weekday of video from recordings in each neighborhood and manually recorded counts of the different vehicle types: motorcycles, passenger cars, light trucks, and heavy trucks (including public transport buses). The counts were summed at the end of every 5 minutes, for a total of 24 hours.

4.10 Questionnaires

Trained study staff administered a detailed questionnaire to participants. The relevant sections of the questionnaire were based off Colombia's Demographic and Health Survey and existing studies of indoor air quality and personal exposure that seek to investigate the role of physical housing conditions [211,212]. Study staff read the questionnaire to participants in Spanish and participants' answers were recorded on paper. The questionnaire inquired into participants' social and demographic characteristics, socioeconomic indicators at the individual and household level, and household characteristics, including presence of mechanical ventilation, ventilation patterns, housing material type, housing structure, and the presence of pets. We did not collect data on cooking-related variables beyond primary fuel use and ventilation.

4.11 Statistical analyses

We characterized the means and standard deviations of indoor noise and outdoor, indoor, and personal exposures to $PM_{2.5}$ and BC for all observations and separately by neighbourhood. Additionally, a time series with hourly Leq, Lmax, and Lmin averaged for each neighbourhood was plotted to assess daily patterns of indoor noise levels within and between neighbourhoods.

We used Multiple Correspondence Analysis (MCA) to combine household and individual-level SEP indicator variables (i.e., housing/neighbourhood quality, health insurance subsidies, monthly household income, monthly personal income, and education level) into a composite SEP metric. Given that indoor Leq was a household-level measure, only household-level SEP indicators (i.e., housing/neighbourhood quality, health insurance subsidies, monthly household income) were used in the MCA analyses for this model. Briefly, this variable reduction process analyzes patterns in multiple categorical variables [213]. The MCA output generated 5 synthetic variables. We selected the synthetic variable that explained the most variability in the SEP indicator variables. The relative contributions of the SEP indicator variable categories to each synthetic variable are shown in **SI Fig. 6** and **SI Fig. 7**.

We constructed multivariable linear regression models with random intercepts at the neighborhood level to assess the environmental, housing, and socio-demographic determinants of exposure to environmental pollutants ($PM_{2.5}$, BC, and indoor noise). Prior to conducting the multivariable regression models, we assessed collinearity between continuous independent variables using Pearson's r. If two variables had a $r > 0.70$, high correlation [214], the variable that was more weakly correlated with the outcome was excluded. This was determined by assessing correlations between the independent variable and the pollutant. The generalized variance inflation factor (GVIF) was used to assess collinearity between two or more categorical variables or a combination of categorical and continuous variables. We used a GVIF cutoff of 10 for models, similar to other air pollution modeling studies, since multicollinearity is likely when GVIF is > 10 [215-217]. Indoor/outdoor $PM_{2.5}$ and BC ratios were excluded from the personal exposure models due to their high correlation with indoor $PM_{2.5}$ or BC. We did not identify any correlated variables using the GVIF. The pollutant data were normally distributed and thus not transformed.

For the multivariable models evaluating determinants of pollutants, we evaluated the following variables and their associations with personal exposure to air pollution: indoor $PM_{2.5}$ or BC (continuous), outdoor $PM_{2.5}$ or BC (continuous), distance of home to nearest major road (continuous), type of nearest major road (primary, secondary, or tertiary), presence of a garage (yes/no), number of fans per room (continuous), number of windows per room (continuous), frequency of fan use over the last 48 hours (no fan/never/rarely versus daily/frequently), frequency of window use over the last 48 hours (never/rarely versus daily/frequently), indoor temperature (continuous), age (continuous), gender (male/female), number of steps taken in the last 48 hours (continuous), and SEP (continuous, based on the composite variable analysis).

For the models examining determinants of indoor noise levels (Leq), we evaluated the distance of home to nearest major road (continuous), type of nearest major road (primary, secondary, or tertiary), presence of a garage (yes/no), number of fans per room (continuous), number of windows per room (continuous), frequency of fan use over the last 48 hours (no fan/never/rarely versus daily/frequently), frequency of window use over the last 48 hours (never/rarely versus daily/frequently), SEP (continuous, based on the composite variable analysis), presence of a dog or bird (yes/no), number of floors in home (1-story home/2 or more-story home), and number of rooms in home (3 rooms or less/4 or more rooms). Number of floors or rooms in the home were considered indicators of household size, which may be associated with indoor noise levels [170]. All plausible variables were included to minimize bias in the effect estimates explaining the pollutant levels [218]. We used R program version 3.6.1 for the descriptive analyses and for the regression analyses, and the FactorMineR version 2.3 package for the MCA [219].

4.3 Results

4.3.1 Participant and household characteristics

We enrolled 18-21 participants (total n = 78) in each neighborhood. Participants had a mean age of 72.9 ± 9.0 years and over half (63%) were women (**Table 1**). Half of the participants had no formal education or some level of primary school education. Participant homes were similar in type and material, with all floors made of permanent materials (e.g. cement, tile, or ceramic) (**Table 2**). Most participants (85%) lived in single-family homes. Nearly all households used a natural gas cookstoves, except for three that used propane. Less than a quarter of households (23%) reported having no fan in their home, and 15% (10-21%, by neighborhood) reported at least one fan per room. All households in the high traffic/low diesel neighbourhood opened their windows daily or frequently over the 48-hr measurement period, compared with 74-86% in the other three neighbourhoods.

4.3.2 Outdoor, indoor, and personal exposures to $PM_{2.5}$, BC, or CO_2

Figure 1 shows mean concentrations of outdoor, indoor, and personal exposure to $PM_{2.5}$ and BC in the four neighbourhoods included in this study. Mean daily outdoor $PM_{2.5}$ was below the WHO's 24-hr guideline of 25 µg/m^3 for all homes (13.1 ± 5.2 µg/m^3) (**SI Table 1**), but above the WHO's annual yearly average guideline of 10 µg/m^3 [26] (**Fig. 1**). Compared to mean outdoor $PM_{2.5}$ concentrations, mean indoor $PM_{2.5}$ concentration was slightly lower (12.7 ± 5.6 µg/m^3), while mean personal exposure to $PM_{2.5}$ among participants (13.5 ± 5.9 µg/m^3) was similar. The high traffic/high diesel neighbourhood had the highest outdoor, indoor, and exposure to $PM_{2.5}$. In the low traffic/high diesel neighbourhood, mean indoor $PM_{2.5}$ concentrations and personal $PM_{2.5}$ exposure were the lowest among the four neighbourhoods, but outdoor $PM_{2.5}$ concentrations were the second highest of the four neighbourhoods.

For all households, mean indoor BC concentration (2.5 ± 0.5 µg/m^3) was slightly lower than the outdoor BC concentrations (2.8 ± 0.7 µg/m^3). Mean personal exposure to BC among participants (2.4 ± 0.5 µg/m^3) was 0.4 µg/m^3 lower than mean outdoor concentration, and similar to mean indoor concentration (0.1 µg/m^3 difference). The low traffic/high diesel neighbourhood had the lowest mean outdoor BC concentration, but the highest indoor concentration (2.9 ± 0.6 µg/m^3) and personal exposure (2.9 ± 0.6 µg/m^3). BC comprised 12-29% of $PM_{2.5}$ (based on neighbourhood-specific means), with outdoor settings usually having the highest percentage of BC. In the low traffic/high diesel neighbourhood, BC instead made its largest contributions to indoor concentrations and personal exposure levels of $PM_{2.5}$.

Mean indoor concentration of CO_2 was 484.1 ppm (**SI Table 1**). The low traffic/low diesel/high braking neighbourhood had the highest CO_2 concentration among the neighbourhoods (503.8 ppm), and the high traffic/low diesel neighbourhood had the lowest (464.5 ppm).

4.3.3 Levels of indoor noise

Mean indoor Leq varied little among the four neighbourhoods (range: 55.2 ± 3.7 dB(A)) Indoor noise levels followed a diurnal trend, where daytime noise levels (mean Lday 59.8 ± 4.8 dB(A)) were higher than evening (mean Levening 57.6 ± 5.9) and nighttime levels (mean Lnight 46.5 ± 3.6) (**Fig 2**). Daytime variability in indoor noise levels within the high traffic neighbourhoods was greater than for lower traffic neighbourhoods (high traffic SD: 5.8 and 5.1 dB(A) versus low traffic SD: 2.8 and 4.0 dB(A)). Nighttime variability in noise was similar across neighbourhoods (range of SDs: 3.1-3.8 dB(A)). Indoor noise was highest in the high traffic/high diesel neighbourhood and lowest in the high traffic/low diesel neighbourhood.

4.3.4 Traffic counts at the intersection with the highest traffic volume in each neighbourhood

Neighborhoods with the highest to lowest 24-hr intersection vehicle counts were high traffic/low diesel, high traffic/high diesel, low traffic/low diesel/high braking, and low traffic/high diesel (**SI Table 2**). The percentage of heavy trucks in the vehicle fleet ranged from 0.07-4% among the four neighbourhoods, with the low traffic/high diesel intersection having the highest percentage and higher count of heavy trucks than in the high traffic/low diesel and low traffic/low diesel/high braking neighbourhoods. As for light trucks, the percent composition in high traffic neighbourhoods was 5% (high traffic/low diesel) and 7% (high traffic/high diesel), while both low traffic neighbourhoods had 4% each of light trucks. The low traffic neighbourhoods had higher proportions of motorcycles (64 and 67%), compared with the high traffic neighbourhoods (48 and 51%).

4.3.6 Correlations between $PM_{2.5}$, BC, and indoor noise

We observed the strongest Pearson correlation between personal exposures to $PM_{2.5}$ and BC (r = 0.68) (**Fig. 3**). Correlations between indoor and personal exposure to $PM_{2.5}$ were moderate (Pearson's r = 0.58), with the low traffic/high diesel neighbourhood having the strongest correlation (r = 0.92) among the four neighbourhoods, while the low traffic/low diesel/high braking neighbourhood had the weakest correlation (r = 0.36). We observed a moderate Pearson correlation between indoor and personal exposure to BC (r = 0.50). The high traffic/low diesel neighbourhood had a strong correlation (Pearson's r: 0.69), but other neighbourhoods had weak correlations (range of Pearson's r for other neighbourhoods: -0.009-0.18). We did not observe strong or consistent Pearson's correlations between air pollution and indoor noise levels. Across all neighbourhoods, Pearson correlations between air pollutants and indoor Leq ranged from -0.34 to 0.20 for $PM_{2.5}$ and -0.29 to 0.13 for BC, with the highest and lowest correlations in the low traffic/high diesel neighbourhood.

4.3.7 Determinants of personal exposure to air pollution

Higher indoor $PM_{2.5}$, higher indoor temperature, and higher SEP were strong determinants of higher personal exposure to $PM_{2.5}$ among our participants (0.5-1.4 µg/m^3 per unit increase in indoor $PM_{2.5}$ (µg/m^3), indoor temperature (°C), or SEP) in the multivariable models (**Table 2)**. Women (compared with men) and participants living nearest to a tertiary road (compared with primary and secondary road types) also had higher personal exposure to $PM_{2.5}$ (1.2 and 1.8 µg/m^3 higher, respectively). Additionally, each 5-year increase in age was associated with 0.3 μg/m^3 (95% CI: -0.4-1.1) higher personal exposure to $PM_{2.5}$. Daily or frequent household window use was most protective against exposure to $PM_{2.5}$ (-3.3 μg/m^3, 95% CI: -6.6-0.01). Higher outdoor $PM_{2.5}$ and more steps taken by participants were not associated with higher exposure to $PM_{2.5}$.

In the multivariable models assessing determinants of personal exposure to BC, higher indoor BC (0.23 μg/m^3, 95% CI: 0.07-0.60 per 1 μg/m^3 increase in indoor BC), higher SEP (0.19 μg/m^3, 95% CI: 0.04-0.41, higher exposure per unit increase in SEP), and higher outdoor BC (0.12 μg/m^3, 95% CI: -0.20-0.36, higher exposure per 1 μg/m^3) were all associated with higher exposure to BC (**Table 3)**. Living near a tertiary road was also associated with 0.14 μg/m^3 (95% CI: -0.16, 0.31) higher exposure to BC. Greater natural household ventilation (more windows per room and higher frequency of use) was associated with 0.13-0.28 μg/m^3 lower BC exposure. Number of steps and age were not associated with exposure to BC.

4.3.9 Determinants of indoor Leq

In the models assessing determinants of indoor noise, higher household-level SEP and a higher number of fans per room had the strongest associations with higher indoor Leq (1.5-1.7 dB(A)), while the presence of at least one dog or bird in the home and a 2 or more-story home were most strongly associated with lower Leq (-2.2 to -1.8 dB(A)) (**Table 4)**. Further distance to the nearest major road or daily or frequent household window use did not appear to be associated with indoor Leq.

4.4 Discussion

To our knowledge this is the first study to characterize the patterns and determinants of personal exposures to $PM_{2.5}$, BC, and indoor noise in a mid-sized Latin American city. We found that personal exposures to $PM_{2.5}$ and BC were relatively low compared with other recent studies of personal exposure conducted in urban areas in Europe, North American, Asia, and Latin America. Additionally, indoor noise levels were high relative to the WHO's recommended guideline of 35 dB(A) [38]. Correlations between indoor Leq and indoor $PM_{2.5}$ or BC were weak and inconsistent across neighbourhoods. Increased household ventilation was associated with lower exposure to $PM_{2.5}$ and BC but higher indoor Leq. Higher SEP was also associated with higher levels of all pollutants.

Mean personal exposures to $PM_{2.5}$ (14 µg/m^3) and BC (2.4 µg/m^3) among retired adults in our study were considerably lower than those observed in exposure studies conducted primarily among adults in urban China, India, Mexico, and Tanzania ($PM_{2.5}$ range: 17-169 µg/m^3; BC range: 3.7-14.7 µg/m^3) [119-123,125,126,129,131,132,181,198,203]. Relative to Hong Kong and urban settings in Europe and North America (range of means/medians of $PM_{2.5}$: 19-31 µg/m^3; of BC: 1.3-2.9 µg/m^3)[161,165,220-225], personal exposure to $PM_{2.5}$ in our study was slightly lower, but personal exposure to BC was slightly higher, though comparable to a mid-sized city in Brazil (2.5 µg/m^3) [54]. Through these comparisons, it appears that $PM_{2.5}$ and BC personal exposure levels in Bucaramanga, Colombia tended to be lower than levels that have been observed in studies in other LMICs and even Europe or North America. This could reflect differences in air pollution sources and source activity patterns, as well as differences in determinants of exposure.

Neighbourhood-specific outdoor, indoor, or personal exposure to BC/$PM_{2.5}$ ratios in our study (12-29%) were higher than in studies conducted in HICs, but lower than in studies that have been conducted in urban Latin American settings. In urban settings in HICs, outdoor $PM_{2.5}$/BC ratio ranged from 6-12% [226-229]. Research in urban Latin American commuting environments, where there is a higher amount of BC emissions from vehicles, found higher contributions of BC to $PM_{2.5}$ (28-100%) in outdoor and personal exposure measurements [199-201]. Compared to urban settings in HICs, there may be a higher number of diesel vehicles, a source of BC [155], in residential environments of Bucaramanga.

None of our study households met the WHO’s 35 dB(A) guideline for noise. Additionally, the mean indoor noise in our study (Leq: 55 dB(A)) was generally higher than in urban homes in Belgium, Korea, China, and the US (range of means: 30-51 dB(A)) [62,133,137,138]. Homes in Colombia tend to have more natural ventilation than homes built in colder climates [230,231]. Since natural ventilation (e.g. open windows) facilitates entry of outdoor noise into the indoor environment [134,135,170,232], it is possible that homes in Colombia experience more noise from outdoor sources compared to homes in colder climates. This could explain the high indoor noise levels observed in our study. Furthermore, indoor Lnight in the four neighbourhoods was lower than Lday and Levening. This trend in our study was consistent with that observed in a study in Besançon, France [134], which was one of the only other studies that also assessed indoor Lday and Lnight. Lower indoor noise levels at night may be due to lower traffic volume, as observed in the four neighbourhoods included in our study. Lower Lnight, compared to Lday, may also be due to reduced indoor noise sources, such as televisions and people [134].

As for associations between noise and air pollution, studies in North American cities found positive associations (range of correlations: 0.14-0.44) between outdoor noise and $PM_{2.5}$ or BC [72,73], while we found weak and negative associations between indoor noise and $PM_{2.5}$ or BC. This difference in association strength and direction may be explained by several reasons. Firstly, the sources of indoor noise, $PM_{2.5}$, and BC may be different from sources of outdoor noise, $PM_{2.5}$, and BC [134,135,164,166]. Second, determinants of noise and $PM_{2.5}$ or BC, such as traffic-related factors (e.g. traffic volume), building insulation, and meteorology, in urban settings in HICs may be different from determinants in LMICs [205,233].

While we did not find strong or consistent correlations between indoor noise and $PM_{2.5}$ or BC, we did find that the low traffic/high diesel neighbourhood had the highest mean personal exposure to BC and indoor Leq. The presence of high-diesel environments, such as the local bus terminal and the high proportion of heavy-duty vehicles (heavy trucks) in the vehicle fleet (4% compared with 1-2% in other neighbourhoods) may have driven the high personal exposure to BC and indoor noise levels in this neighbourhood [54,150,177,234]. We also noted that mean outdoor BC concentrations were lowest in the low traffic/high diesel neighbourhood. This finding may reflect the location of the two outdoor fixed-site monitors, which may not have captured the full range of variability in the outdoor BC concentrations. Furthermore, overall traffic was lowest in this neighbourhood (hence, its classification as a low traffic/high diesel neighbourhood), which may explain why mean personal exposure to $PM_{2.5}$ was also lowest.

In our study, indoor $PM_{2.5}$ and BC were important determinants of personal exposures. This result may be, in part, attributable to our study population of retired adults who were no longer commuting for work. Studies in Brazil, South Korea, Australia, and the United States found that urban residents, especially older adults, spend over 50% of their time indoors, usually in their homes [52-54,161,235-237]. Consistent with findings in Hong Kong and cities in mainland China, higher outdoor BC concentrations contributed to higher personal exposure to BC in our study [123,124]. Indoor, residential environments in Bucaramanga may contribute more to personal exposure to $PM_{2.5}$ and BC, compared to outdoor environments. However, outdoor air pollution can also contribute to personal exposure by infiltrating indoors [146]. If personal exposure measurements are not feasible in future studies in similar settings, measures of indoor, residential $PM_{2.5}$ or BC may be more indicative of personal exposures to $PM_{2.5}$ or BC compared to outdoor measures.

Another important determinant of higher exposure to $PM_{2.5}$ and BC was higher SEP, which contrasts with findings in urban areas of China, India, and many in the US [123,178,179,181], but is similar to findings in New York [238]. It could be that in Bucaramanga retired adults with higher SEP live in areas with higher traffic, a source of $PM_{2.5}$ and BC. For example, the unadjusted correlation between SEP and distance of the home to the nearest major road was low to moderate, which lends some support to this theory.

The strongest determinant of lower personal exposure to $PM_{2.5}$ was daily or frequent household window use, while lower personal exposure to BC was associated with both daily and frequent window use and a higher number of windows per room. In urban China, a higher number of hours with open windows in households was associated with 0.013 ug/m^3 (0.010 – 0.017 µg/m^3) higher personal exposure to BC among older adults [123]. It is possible that in our study, the increased ventilation achieved through the opening of windows prevented concentrations from rising over the course of the day [164], while in urban China, open windows could have led to more pollutant infiltration indoors.

Higher SEP in our study was associated with higher indoor Leq, a result which conflicts with studies in urban Europe, North America, and Hong Kong that observed negative associations between outdoor noise levels and SEP [112,182-194]. Since traffic is also source of noise, in addition to $PM_{2.5}$ and BC, the positive association between higher SEP and higher noise levels may also be due to living in higher traffic areas. Similar to our study, higher SEP groups in Birmingham, UK, experienced higher outdoor levels of transport-related noise, which may have been due to the benefits of living in locations with good transport [185]. Furthermore, a higher number of fans per room was also strongly associated with higher indoor Leq. Interestingly, we found that higher SEP households tended to have a higher frequency of fan use, which may explain why more fans were associated with greater noise. Another explanation for the association could be that higher SEP households can afford more systems that support mechanical ventilation (i.e., fans), as well as assets that emit noise, such as televisions and radios.

A key strength of this study includes the simultaneous measurement of multiple pollutants ($PM_{2.5}$, BC, and indoor noise) in multiple locations (outdoor, indoor, and personal exposure in four neighbourhoods), which allowed us to capture day-to-day variability of these pollutants in a range of settings in Bucaramanga, Colombia. The results of this study address an empirical gap on exposures to urban air pollution and noise in Latin American cities, and particularly, in mid-sized cities, where urban growth is most rapid [82]. Our study also has limitations. Our sample size, both in terms of the number of participants/households (n = 78) and the number of neighbourhoods (N = 4) was small. Small sample sizes, along with potential selection bias due to a low participation rate, could have biased effect estimates in our models that identified determinants of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor Leq.

4.5 Conclusion

Our study addresses a gap in empirical data on levels and determinants of exposure to $PM_{2.5}$, BC, and indoor noise in mid-sized, Latin American cities. Our findings suggest that $PM_{2.5}$ levels in Bucaramanga are well-below the WHO's 24-hr guideline. However, BC contributes to a substantial fraction of the $PM_{2.5}$ mass. Indoor noise was also high, potentially posing health risks to population health in Bucaramanga [41]. Important determinants for higher exposure to $PM_{2.5}$ or BC included indoor $PM_{2.5}$ or BC concentration, while natural ventilation predicted lower exposures. Higher SEP was associated with higher levels of air pollutants as well as indoor Leq, and a greater number of fans in homes was also associated with higher Leq. Targeting indoor air pollution, outdoor air pollution (a source of indoor air pollution), noise sources, and household ventilation may reduce personal exposures to PM2.5 and BC as well as indoor noise in Bucaramanga, Colombia. Subsequent research could benefit from a larger sample size, in addition to a more extensive assessment of sources of indoor $PM_{2.5}$, BC, and noise levels.

Acknowledgements

We thank field staff and participants. KV was supported by the NSERC Undergraduate Research Award and the FRQS Master's Training Award. Data collection in Colombia was supported by the Innovative Solutions for Planetary Health Seed Grants for Interdisciplinary

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Tables and Figures

Table 1: Socio-demographic characteristics of study participants by neighbourhood (number (%) and mean ± SD)

Variables	High traffic/ high diesel (n = 19)	Low traffic/ high diesel (n = 20)	High traffic/ low diesel (n = 18)	Low traffic/ low diesel/ high braking (n = 21)	All (n = 78)
Age (mean ± SD)	74.6 ± 8.8	72.2 ± 11.2	73.5 ± 8.7	71.6 ± 7.2	72.9 ± 9.0
Gender					
Female	12 (63)	13 (65)	12 (67)	12 (57)	49 (63)
Civil status					
Single	2 (11)	4 (20)	3 (17)	2 (10)	11 (14)
Married or cohabitation	11 (58)	10 (50)	9 (50)	10 (48)	40 (51)
Separated, divorced, or widowed	6 (32)	6 (30)	6 (33)	9 (43)	27 (35)
Individual indicators of socioeconomic position					
Highest level of education attained					
No formal education	1 (5)	0	1 (6)	2 (10)	4 (5)
Primary	5 (26)	12 (60)	5 (28)	13 (62)	35 (45)
Secondary	8 (42)	6 (30)	6 (33)	4 (19)	24 (31)
Post-secondary	5 (26)	2 (10)	6 (33)	2 (10)	15 (19)
Monthly personal income (in Colombian Pesos, COL$) §					
No personal income	5 (26)	13 (65)	7 (39)	7 (33)	32 (41)
≤ $781,242	10 (53)	0	4 (22)	3 (14)	17 (22)
> $781,242	4 (21)	7 (35)	7 (39)	11 (52)	29 (37)
Household indicators of socioeconomic position					
Housing and neighbourhood quality ‡					
2 = Low	0	0	0	18 (86)	18 (23)
3 = Low-middle	8 (42)	20 (100)	0	3 (14)	31 (40)
4 = Middle	11 (58)	0	18 (100)	0	29 (37)
Monthly household income (in Colombian Pesos, COL$) §					
Unknown	0	0	0	1 (5)	1 (1)
No household income	1 (5)	0	0	0	1 (1)
≤ $781,242	14 (74)	4 (20)	10 (56)	5 (24)	33 (42)
> $781,242	4 (21)	16 (80)	8 (44)	15 (71)	43 (55)
Health insurance program ¶					
No subsidy	12 (63)	9 (45)	10 (56)	8 (38)	39 (51)
Receives 1 subsidy	6 (32)	4 (20)	8 (44)	7 (33)	24 (31)

Receives 2+ subsidies	1 (5)	6 (30)	0	6 (29)	13 (17)
No insurance	0	1 (5)	0	0	1 (1)

‡ Each household is assigned a housing and neighbourhood quality status, ranging from 1 (lowest) to 6 (highest). Each status from 1 to 6 is assigned a qualifying name: 2 = "low," 3 = "low-middle," and 4 = "middle" [239,240].
§ $781 242 Pesos was the Colombia's Minimum Monthly Wage (MMW) in 2018.
¶ Residents earning less than $170 US per month are eligible for subsidized health insurance [241]. In addition, Colombia uses a beneficiary selection system to identify households that are lowest-income and most vulnerable [242] to receive additional health insurance subsidies [243]. Eligible households determined to have the most need receive the additional subsidies [243].

We enrolled 1 participant in the low traffic/high diesel neighbourhood who was retired but < 60 years of age (age = 53).

Table 2: Household characteristics of study homes (n = 78) by neighbourhood traffic characteristics (number (%))

Variables	High traffic/high diesel (n = 19)	Low traffic/high diesel (n = 20)	High traffic/low diesel (n = 18)	Low traffic/ low diesel/ high braking (n = 21)	All (n = 78)
Housing type					
Single-family home	14 (74)	20 (100)	16 (89)	16 (76)	66 (85)
Multi-family home	3 (16)	0	1 (6)	3 (14)	7 (9)
Renting a room	0	0	1 (6)	2 (10)	3 (4)
Apartment building	2 (11)	0	0	0	2 (3)
Number of floors in home					
1	4 (21)	11 (55)	2 (10)	11 (55)	28 (36)
2	8 (42)	8 (40)	14 (70)	9 (45)	39 (50)
3 +	7 (37)	1 (5)	2 (10)	1 (5)	11 (14)
Number of rooms in home					
Less than 3	4 (21)	5 (25)	4 (22)	11 (52)	24 (31)
3-5	11 (58)	14 (70)	12 (67)	8 (38)	45 (58)
6-7	4 (21)	1 (5)	2 (11)	2 (10)	9 (12)
Garage attached to home					
Yes	11 (58)	0	14 (78)	4 (19)	29 (38)
Average number of fans per room					
0	3 (16)	5 (25)	1 (6)	9 (43)	18 (23)
0.14-0.4	7 (37)	7 (35)	9 (50)	7 (33)	30 (38)
0.5-0.8	5 (26)	6 (30)	5 (28)	2 (10)	18 (23)
≥ 1	4 (21)	2 (10)	3 (17)	3 (14)	12 (15)
Frequency of fan use in last 2 days ‡					
Never	10 (63)	8 (53)	10 (59)	4 (33)	32 (53)
Rarely	3 (19)	1 (7)	3 (18)	4 (33)	11 (18)
Frequently or daily	3 (19)	6 (40)	4 (24)	4 (33)	17 (28)
Average number of windows per room					
< 1	2 (11)	4 (20)	0	4 (19)	10 (13)
1-2	16 (84)	16 (80)	17 (94)	17 (81)	66 (85)
> 2	1 (5)	0	1 (6)	0	2 (3)
Frequency of opening window in last 2 days					
Never	3 (16)	0	0	2 (10)	5 (6)
Rarely	2 (11)	3 (15)	0	1 (5)	6 (8)
Frequently or daily	14 (74)	17 (85)	18 (100)	18 (86)	67 (86)
Type of home roof and wall material †					
Permanent	17 (89)	17 (85)	18 (100)	17 (81)	69 (88)
Impermanent	2 (11)	3 (15)	0	4 (19)	9 (12)
Cooking fuel type					
Natural gas	19 (100)	18 (90)	18 (100)	20 (95)	75 (96)
Propane gas	0	2 (10)	0	1 (5)	3 (4)
Number of dogs and cats					
0	10 (53)	7 (35)	8 (44)	10 (45)	35 (45)

1	6 (32)	6 (30)	7 (39)	4 (18)	23 (29)
2	2 (11)	5 (25)	1 (6)	6 (27)	14 (18)
3 +	1 (5)	2 (10)	2 (11)	1 (5)	6 (8)
Number of birds and other pets					
0	14 (74)	13 (65)	17 (94)	15 (68)	59 (76)
1	2 (11)	4 (20)	1 (6)	2 (9)	9 (12)
2 +	3 (16)	3 (15)	0	4 (18)	10 (13)
Presence of ≥ 1 tobacco smokers in the home (past month)					
Yes	2 (11)	2 (10)	0	4 (19)	8 (10)
No	17 (89)	18 (90)	18 (100)	17 (81)	70 (90)

† Permanent was defined as tile/ceiling or concrete for roofs and brick for walls. Impermanent was defined as thatched/cane material or zinc for roofs and a wet earth-straw combination for walls.
‡ Denominator is the total number of households reported as having at least one fan in the home.

Only 3 households in the high traffic/low diesel neighbourhood had an air conditioning unit.

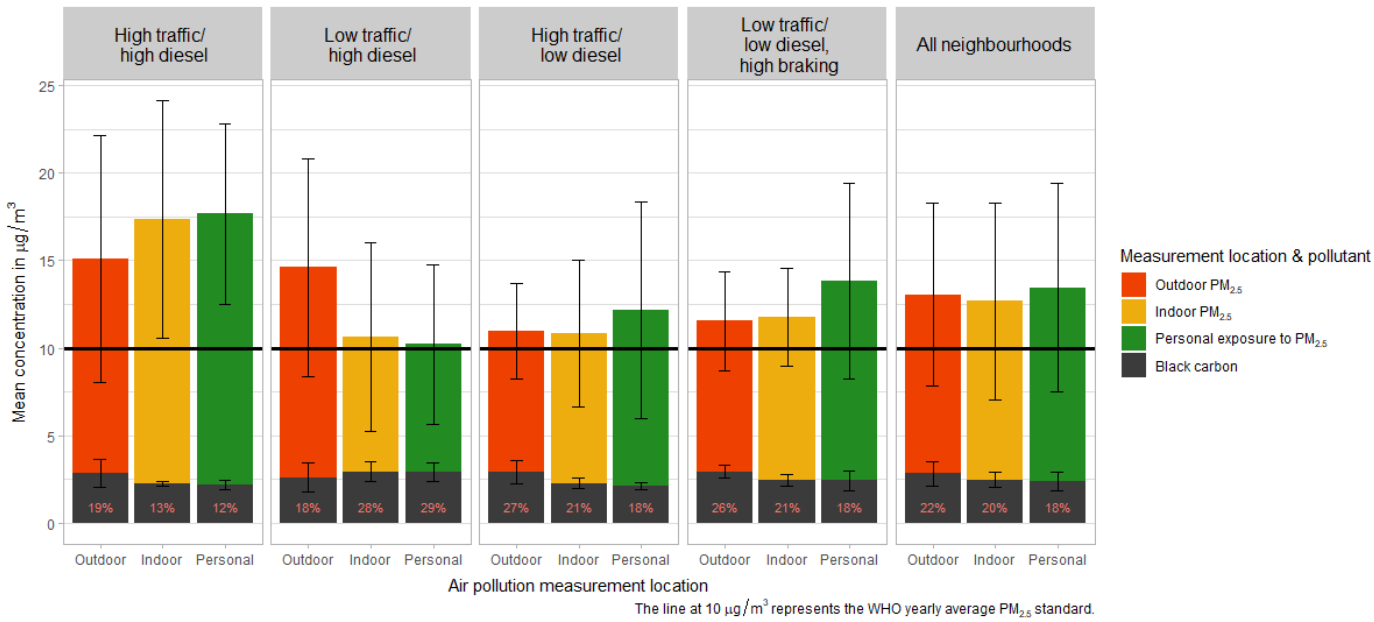


Figure 1: Concentrations of outdoor, indoor, and personal exposures to fine particulate matter and black carbon (µg/m^3) in four urban neighbourhoods in Bucaramanga, Colombia

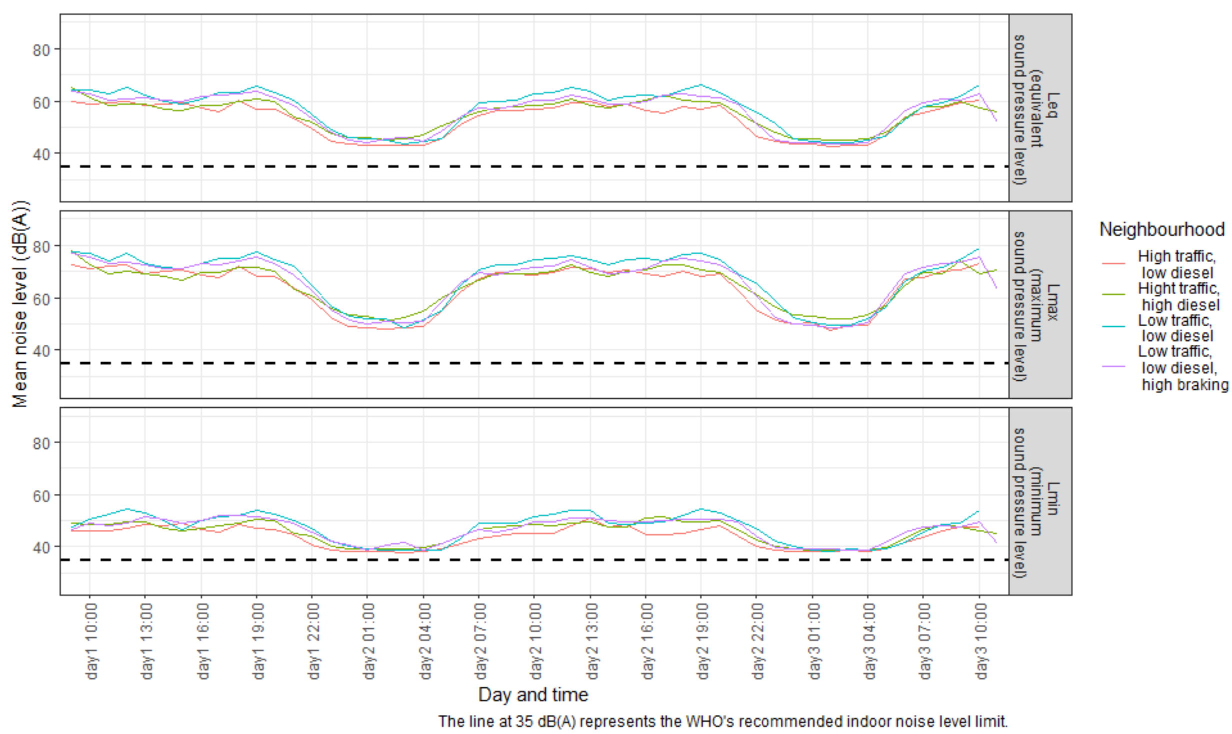


Figure 2: Time series of mean 48-hr indoor noise levels (Leq, Lmax, and Lmin) in four urban neighbourhoods in Bucaramanga, Colombia

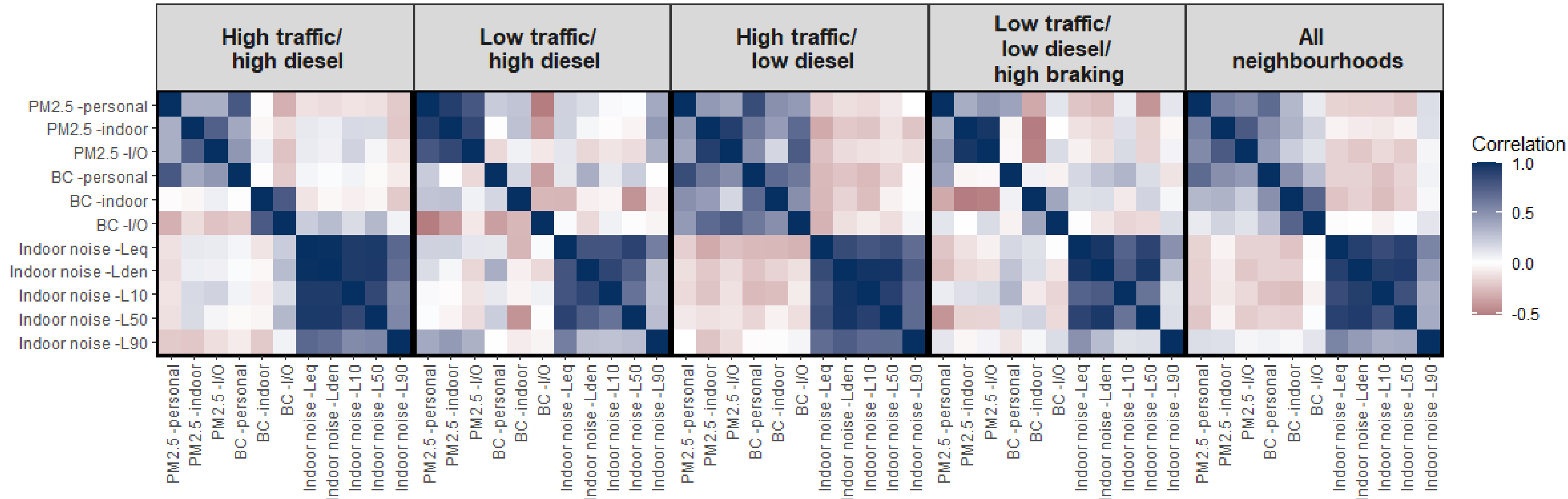


Figure 3: Pearson correlation matrix of associations between air pollution and noise pollutants by neighbourhood type in Bucaramanga, Colombia

Table 3: Results from bivariable and multivariable random-intercept regression models of environmental, housing, and socio-demographic determinants of personal exposure to fine particulate matter ($\mu g/m^3$)

Variable	Bivariable		Multivariable	
	β ($\mu g/m^3$) ‡	95% CI ‡	β ($\mu g/m^3$) ‡	95% CI ‡
Age (per 5 years)	-0.08	(-0.75, 0.62)	0.31	(-0.44, 1.05)
Woman (ref: man)	2.08	(-0.39, 4.53)	1.81	(-0.55, 4.18)
Number of steps (per 100 steps)	0.02	(-0.01, 0.05)	0.01	(-0.02, 0.04)
SEP §	1.23	(-0.94, 3.60)	1.35	(-0.67, 3.36)
Indoor fine particulate matter (per 1 $\mu g/m^3$)	0.58	(0.38, 0.81)	0.57	(0.33, 0.81)
Indoor temperature (°C)	0.25	(-0.71, 1.17)	0.52	(-0.32, 1.37)
Presence of garage (ref: none)	0.58	(-2.42, 3.76)	-1.38	(-4.15, 1.42)
Number of fans per room	0.13	(-3.42, 3.67)	-0.38	(-4.73, 3.97)
Number of windows per room	-0.15	(-3.32, 3.08)	0.92	(-2.20, 4.05)
Daily or frequent fan use (ref: no fan, never, or rarely)	1.72	(-1.25, 4.60)	0.12	(-2.98, 3.22)
Daily or frequent window use (ref: never or rarely)	-1.58	(-5.23, 1.92)	-3.29	(-6.59, 0.01)
Outdoor fine particulate matter (per 1 $\mu g/m^3$)	0.23	(-0.18, 0.64)	-0.06	(-0.44, 0.32)
Distance to nearest major road (per 50 m)	-0.52	(-1.19, 0.14)	-0.15	(-0.59, 0.28)
Nearest major road type: tertiary (ref: primary or secondary) †	0.17	(-3.93, 4.54)	1.19	(-1.32, 3.70)

‡ β is the parameter estimate, a slope representing the change in the dependent variable based on a change in the independent variable, and 95% CI is the 95% confidence interval for the corresponding estimate.

§ SEP, socioeconomic position: a synthetic variable constructed through multiple correspondence analysis by reducing household and individual-level SEP indicators

† Primary major roads have the highest traffic volume, while tertiary roads have the lowest traffic volume

Table 4: Results from bivariable and multivariable random-intercept regression models of environmental, housing, and socio-demographic determinants of personal exposure to fine black carbon (μg/m^3)

Variable	Bivariable		Multivariable	
	β (μg/m^3) ‡	95% CI ‡	β (μg/m^3) ‡	95% CI ‡
Age (per 5 years)	-0.02	(-0.07, 0.04)	0.04	(-0.02, 0.12)
Woman (ref: man)	0.07	(-0.13, 0.26)	0.03	(-0.18, 0.24)
Number of steps (per 100 steps)	0.00	(0.00, 0.00)	0.00	(0.00, 0.00)
SEP §	0.19	(0.02, 0.40)	0.19	(0.04, 0.41)
Indoor black carbon (per 1 μg/m^3)	0.31	(0.06, 0.64)	0.23	(0.07, 0.60)
Indoor temperature (°C)	-0.03	(-0.11, 0.04)	-0.02	(-0.11, 0.04)
Presence of garage (ref: none)	0.15	(-0.09, 0.40)	0.06	(-0.18, 0.34)
Number of fans per room	-0.01	(-0.29, 0.26)	-0.01	(-0.38, 0.40)
Number of windows per room	-0.05	(-0.29, 0.21)	-0.13	(-0.42, 0.13)
Daily or frequent fan use (ref: no fan, never, or rarely)	-0.04	(-0.19, 0.27)	0.05	(-0.20, 0.34)
Daily or frequent window use (ref: never or rarely)	-0.20	(-0.47, 0.08)	-0.28	(-0.63, -0.04)
Outdoor black carbon (per 1 μg/m^3)	0.25	(0.01, 0.50)	0.12	(-0.20, 0.36)
Distance to nearest major road (per 50 m)	-0.02	(-0.09, 0.05)	-0.03	(-0.06, 0.02)
Nearest major road type: tertiary (ref: primary or secondary) †	0.26	(-0.10, 0.60)	0.14	(-0.16, 0.31)

‡ β is the parameter estimate, a slope representing the change in the dependent variable based on a change in the independent variable, and 95% CI is the 95% confidence interval for the corresponding estimate.
§ SEP, socioeconomic position: a synthetic variable constructed through multiple correspondence analysis by reducing household and individual-level SEP indicators
† Primary major roads have the highest traffic volume, while tertiary roads have the lowest traffic volume

Table 5: Results from bivariable and multivariable random-intercept regression models of environmental and housing determinants of indoor Leq (dB(A))

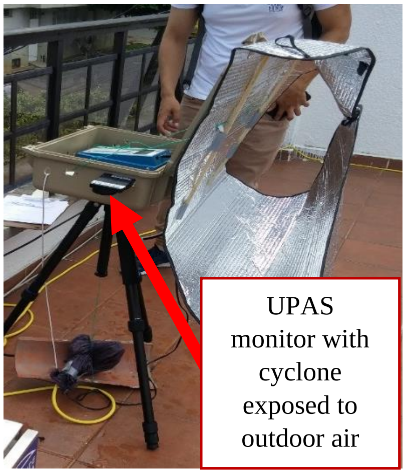
Variable	Univariable		Multivariable	
	β (dB(A)) ‡	95% CI ‡	β (dB(A)) ‡	95% CI ‡
Household-level SEP §	1.88	(0.59, 3.08)	1.52	(0.17, 2.87)
Presence of garage (ref: none)	0.32	(-2.69, 2.35)	0.85	(-1.16, 2.86)
Number of fans per room	1.90	(-0.41, 4.17)	1.69	(-1.12, 4.50)
Number of windows per room	0.43	(-1.97, 2.52)	0.42	(-2.05, 2.88)
Daily or frequent fan use (ref: no fan, never, or rarely)	0.75	(-1.21, 2.75)	-0.48	(-2.56, 1.59)
Daily or frequent window use (ref: never or rarely)	0.03	(-2.36, 2.34)	0.16	(-1.98, 2.29)
Presence of ≥1 dog and/or bird (ref: absence)	-2.50	(-4.12, -0.94)	-2.2	(-3.76, -0.63)
2 or more-story home (ref: 1 story)	-1.48	(-3.50, 0.24)	-1.76	(-3.53, 0.01)
4 or more rooms in home (ref: 3 or less)	-0.58	(-1.76, 1.85)	0.59	(-1.18, 2.35)
Distance to nearest major road (per 50 m)	0.23	(-0.17, 0.58)	0.15	(-0.14, 0.42)
Nearest major road type: tertiary (ref: primary or secondary) †	0.69	(-1.94, 3.31)	1.08	(-0.70, 2.85)

‡ β is the parameter estimate, a slope representing the change in the dependent variable based on a change in the independent variable, and 95% CI is the 95% confidence interval for the corresponding estimate.

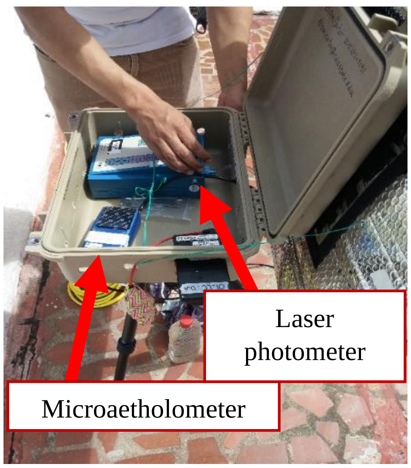
§ SEP, socioeconomic position: a synthetic variable constructed through multiple correspondence analysis by reducing household and individual-level SEP indicators

† Primary major roads have the highest traffic volume, while tertiary roads have the lowest traffic volume

APPENDIX: Supplementary Information



SI Figure 1: Outdoor air pollution monitor mounted on a tripod atop a roof



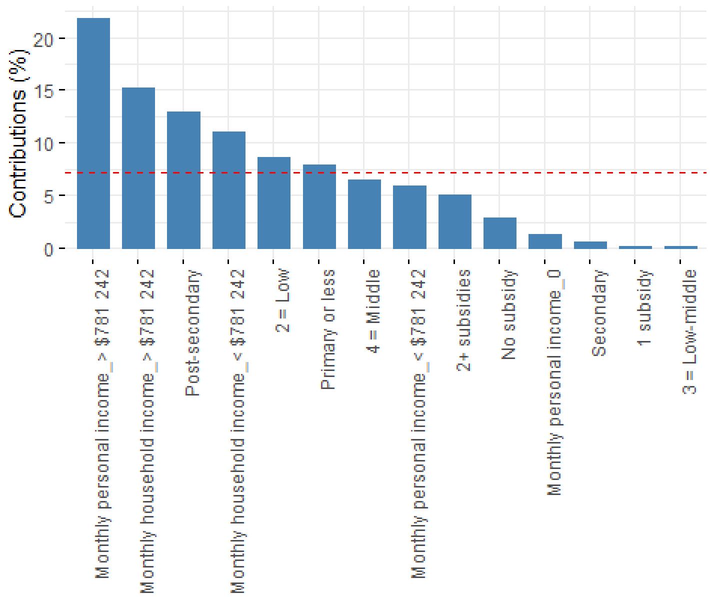
SI Figure 2: Outdoor air pollution monitor contents

During participant recruitment, enrolled households were assessed for the presence of a visible balcony or a roof. Balconies and roofs were evaluated for the following criteria: (1) safe placement of an outdoor air pollution monitor, (2) absence of significant air pollution and noise sources (e.g. a washer or dryer, a kitchen window, etc.), and (3) absence of physical obstructions that could affect measurements. If a balcony or roof met these criteria, participants were asked for permission to mount the monitor for 5 days. Outdoor air pollution monitors were then mounted on the balcony or roof of the first 2 households that accepted.

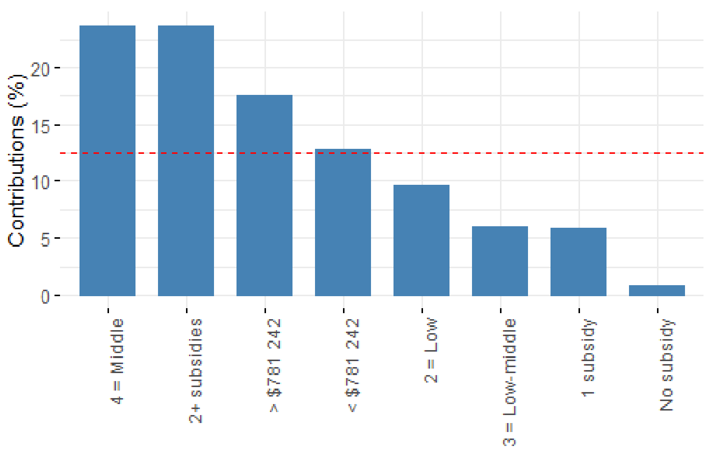
SI Figure 3: Placement of UPAS and noise sensor on a shelf for urban indoor pollutant measurements in one household

SI Figure 4: Placement of indoor air quality monitor

SI Figure 5: UPAS measuring personal exposure to PM2.5 on a) participant's upper arm, in a case and b) around the participant's waist, in a waist pack.



SI Figure 6: Percent (%) contribution of household and invidual-level variable categories to synthetic variable used in exposure to $PM_{2.5}$ and BC models



SI Figure 7: Percent (%) contribution of household-level variable categories to synthetic variable used in indoor Leq model

SI Table 1: Pollutant levels and environmental variables by neighbourhood (mean ± SD or count (%))

Variables	High traffic/ high diesel	Low traffic/ high diesel	High traffic/ low diesel	Low traffic/ low diesel/ low braking	All
Outdoor air quality (5-day average)					
$PM_{2.5}$ ($\mu g/m^3$)	15.1 ± 7.1	14.6 ± 6.2	11.0 ± 2.7	11.5 ± 2.8	13.1 ± 5.2
BC ($\mu g/m^3$)	2.9 ± 0.8	2.6 ± 0.8	2.9 ± 0.6	3.0 ± 0.4	2.8 ± 0.7
Indoor air quality (48-hr)					
$PM_{2.5}$ ($\mu g/m^3$)	17.3 ± 6.8	10.7 ± 5.4	10.8 ± 4.2	11.8 ± 2.8	12.7 ± 5.6
$PM_{2.5}$ indoor/outdoor ratio	1.2 ± 0.4	0.7 ± 0.3	1.0 ± 0.3	1.1 ± 0.3	1.0 ± 0.4
BC ($\mu g/m^3$)	2.3 ± 0.1	2.9 ± 0.6	2.3 ± 0.3	2.5 ± 0.3	2.5 ± 0.5
BC indoor/outdoor ratio	1.0 ± 0.3	0.8 ± 0.1	0.9 ± 0.1	0.8 ± 0.0	0.9 ± 0.2
CO (ppm)	0.9 ± 0.4	0.5 ± 0.2	0.7 ± 0.2	0.8 ± 0.3	0.7 ± 0.3
CO2 (ppm)	473.7 ± 45.5	497.0 ± 44.9	464.5 ± 18.2	503.8 ± 47.0	484.1 ± 41.7
Personal exposure (48-hr)					
$PM_{2.5}$ ($\mu g/m^3$)	17.7 ± 5.2	10.2 ± 4.5	12.2 ± 6.2	13.8 ± 5.6	13.5 ± 5.9
BC ($\mu g/m^3$)	2.2 ± 0.3	2.9 ± 0.6	2.2 ± 0.2	2.5 ± 0.6	2.4 ± 0.5
Indoor noise (48-hr)					
Leq (dB(A))	54.5 ± 4.2	57.1 ± 2.9	53.0 ± 3.8	55.7 ± 3.1	55.2 ± 3.7
Lmax (dB(A))	64.5 ± 4.7	67.5 ± 3.5	62.9 ± 3.9	65.7 ± 3.1	65.3 ± 4.1
Lmin (dB(A))	45.4 ± 3.9	47.3 ± 2.9	43.6 ± 2.9	46.6 ± 3.5	45.8 ± 3.5
Lday (dB(A))	58.8 ± 5.1	62.1 ± 2.8	57.0 ± 5.8	60.7 ± 4.0	59.8 ± 4.8
Levening (dB(A))	56.6 ± 5.3	61.0 ± 5.5	53.2 ± 6.0	58.8 ± 4.2	57.6 ± 5.9
Lnight (dB(A))	47.3 ± 3.2	47.1 ± 3.8	44.6 ± 3.1	46.8 ± 3.7	46.5 ± 3.6
Lden (dB(A))	59.4 ± 4.4	62.9 ± 3.5	57.1 ± 4.9	61.1 ± 3.4	60.3 ± 4.5
L10 (dB(A))	63.8 ± 5.9	68.7 ± 3.2	62.9 ± 6.5	66.7 ± 3.1	65.7 ± 5.2
L50 (dB(A))	54.6 ± 4.8	58.0 ± 3.8	51.9 ± 5.5	56.4 ± 4.9	55.4 ± 5.2
L90 (dB(A))	45.0 ± 3.4	43.5 ± 3.2	42.5 ± 2.9	43.4 ± 3.5	43.6 ± 3.3

Weekday traffic counts (24-hr)					
All vehicles	38768 †	7684 †	41620 †	8285 †	96357
Motorcycles	18616 (48)	5173 (67)	21153 (51)	5272 (64)	50214 (52)
Passenger cars	16453 (42)	1956 (25)	18405 (44)	2662 (32)	39476 (41)
Light trucks	2878 (7)	269 (4)	2033 (5)	313 (4)	5493 (6)
Heavy trucks	821 (2)	286 (4)	29 (0.07)	38 (0.46)	1174 (1)
Mean household distance to nearest major roadway (m)	202.1 ± 61.6	527.5 ± 29.5	150.5 ± 70.2	163.8 ± 52.4	264.6 ± 166.8

$PM_{2.5}$, fine particulate matter; BC, black carbon; CO, carbon monoxide; CO_2, carbon dioxide

† Measurements were conducted on a Tuesday or Wednesday on the highest-traffic road in each neighbourhood.

Sample size for each neighbourhood by measurement type: Indoor $PM_{2.5}$, $PM_{2.5}$ indoor/outdoor ratio, indoor BC, and BC infiltration factor indoor/outdoor ratio measurements (high traffic/high diesel = 19; low traffic/high diesel = 18; high traffic/low diesel = 18; low traffic/high diesel/high braking = 21). 2/78 homes did not receive an indoor $PM_{2.5}$ and BC air sampler due to equipment failures. CO and CO2 measurements (each neighbourhood = 4). Personal $PM_{2.5}$ and BC measurements (high traffic/high diesel n = 19; low traffic/high diesel n = 20; high traffic/low diesel n = 18; high traffic/low diesel/low braking n = 21). Indoor noise measurements (high traffic/high diesel n = 16; low traffic/high diesel n = 20; high traffic/low diesel n = 17; high traffic/low diesel/low braking n = 20). 5/78 homes did not receive a noise monitor due to equipment shortages. Distance to nearest major road (high traffic/high diesel n = 19; low traffic/high diesel n = 20; high traffic/low diesel n = 18; high traffic/low diesel/low braking n = 20). Leq48hr is the mean sound pressure for a set duration [36]. Lmax is the maximum sound pressure for a set duration [36]. Lmin is the minimum sound pressure for a set duration [36]. Lday is the mean sound pressure from 7:00 – 19:00 [210]. Levening is the mean sound pressure from 19:00 – 23:00 [210]. Lnight is the mean sound pressure from 23:00 – 7:00 [210]. Lden (Lday-evening-night) is the weighted sound pressure level, where evening levels receive a 5 dB(A) penalty and night levels receive a 10 dB(A) penalty [210]. L10 is the sound pressure level that is surpassed for 10% of the set time duration, while L50 is for 50% of the time duration, and L90 is for 90% of the time duration (Souza & Giunta, 2011). L90 is a measure of background noise (Souza & Giunta, 2011).

Chapter 5: Discussion and conclusion

5.1 Discussion

I met three objectives in my thesis. The first was to characterize the levels of exposure to $PM_{2.5}$ and BC among retired adults living in four neighbourhoods with different features of traffic in urban Colombia. The second objective was to quantify indoor noise levels in participants' households. My final objective was to investigate environmental, housing, and socio-demographic determinants of personal exposure to fine particulate matter ($PM_{2.5}$), personal exposure to black carbon (BC), and indoor equivalent sound pressure level (Leq).

In Chapter 3, my literature review identified traffic as a major source of $PM_{2.5}$, BC, and noise in urban areas. Evaluating levels of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor noise, I found that $PM_{2.5}$ and BC levels in urban settings in Asia tend to be higher, relative to other settings and that compared to the breadth of literature on outdoor noise, literature on indoor noise is sparse and focused in high-income country (HIC) settings. Furthermore, I identified determinants of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor noise that have been explored in cities, primarily in Asia, North America, and Europe. Determinants identified as contributors to personal exposure to $PM_{2.5}$ and BC included outdoor environments, indoor environments, commuting environments, ventilation, age, gender, and SEP. As for noise, housing features (open windows, room size, housing type, and building insulation) and SEP were identified as determinants.

In Chapter 4, I presented and discussed my findings on levels and determinants of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor noise among retired adults in Bucaramanga, Colombia. $PM_{2.5}$ levels in Bucaramanga were low compared to other urban settings. BC levels were relatively high, compared to our findings on $PM_{2.5}$. I also found that indoor noise was high when compared to the few studies assessing indoor levels in urban settings in HICs. Key determinants for personal exposure to $PM_{2.5}$ or BC among retired adult participants in Bucaramanga included indoor concentration of $PM_{2.5}$ or BC, SEP, and natural ventilation. As for indoor Leq, higher SEP and increased ventilation was associated with higher noise.

I also compared outdoor, indoor, and personal exposure levels of $PM_{2.5}$ and BC in my study. In urban European and US residential settings with similar $PM_{2.5}$ levels as in our study, mean personal exposure to $PM_{2.5}$ among children or adults was either higher or lower than indoor or outdoor $PM_{2.5}$ concentrations [165,220,221,244]. Similarly, in Bucaramanga, neighbourhood-specific personal exposures to $PM_{2.5}$ were lower than outdoor $PM_{2.5}$ levels by 1.2-2.6 $\mu g/m^3$, except for the low traffic/high diesel neighbourhood. As for BC, neighbourhood-specific outdoor concentrations were higher than personal exposure levels by 0.5-0.7 $\mu g/m^3$, except for the low traffic/high diesel neighbourhood. This contradicts findings from a study in Londrina, Brazil where outdoor BC was lower than the 48-hr mean personal exposure levels among working-age adults [54]. Findings in my study may differ from those in other studies, in part, due to differences in study populations; in this study, we enrolled retired adults, who are likely to spend more time at home than other age groups [52,235-237].

In exploring key determinants of personal exposure to $PM_{2.5}$ and BC, I found that relative to outdoor concentrations of $PM_{2.5}$ and BC, indoor concentrations contributed more to personal exposure. For BC in particular, commuting environments (e.g. inside cars, along bicycle paths, by bus stations) tended to have higher concentrations, compared to other microenvironments [177]. Since retired adults, such as those in my study, were not commuting to work, they may have been spending more time at home [52], and perhaps, their neighbourhood environments. These results could explain why mean personal exposures to $PM_{2.5}$ or BC were similar to mean indoor concentrations in our study. My findings suggest that indoor concentrations of $PM_{2.5}$ and BC may be more suitable estimates for personal exposure among retired adults in Bucaramanga, Colombia, who likely spend more time inside their homes, compared to outdoor settings. Therefore, it is possible that household microenvironments among retired adults in this study are contributing most to their personal exposure to BC. Further analyses of indoor (household) CO_2 levels can provide more insight on ventilation within households and contributions of outdoor air pollution, indoors.

Similar indoor and personal exposure levels of $PM_{2.5}$ or BC could also explain why women in our study tended to have higher personal exposure to $PM_{2.5}$ (2.0 $\mu g/m^3$), relative to men, while the difference in BC exposure (0.03 $\mu g/m^3$ higher for women) was negligible. Some studies have found that women in urban areas in HICs and LMICs tended to spend more time indoors than men [52,172-174], particularly in domestic environments (e.g. cooking and cleaning) [130,173]. These microenvironments can have relatively higher $PM_{2.5}$ levels [145]. If retired men and women in my study were spending similar amounts of time in their home, then one explanation for the higher personal exposure to $PM_{2.5}$ among women may be the type of activities they engage in, such as cooking or cleaning.

Studies in LMICs have found that women's choice of transport mode can also vary by gender [175,176]. Furthermore, different traffic environments, particularly those involving proximity to diesel vehicles and exhaust emissions, have been associated with higher BC [54,169]. Personal BC exposures were similar for women and men in our study, which could be due to two different reasons: (1) choice of transport may not vary between retired women and men in Bucaramanga or (2) compared to non-retired adults, participants in my study were spending more time inside their homes than in other environments, including commuting environments.

5.2 Strengths and limitations

Key strengths of my thesis include the simultaneous measurement of multiple pollutants ($PM_{2.5}$, BC, and indoor noise) in multiple locations (outdoor, indoor, and personal exposure in four neighbourhoods), which allowed for capturing of day-to-day variability of these pollutants in Bucaramanga, Colombia. The results of this study contribute to an empirical gap on exposures of both urban air pollution and noise in Latin American cities. My thesis also explored environmental, housing, and socio-demographic determinants of these pollutants in a geographic setting that is relatively understudied in environmental exposure assessment. Most studies of environmental pollution in urban settings in LMICs were conducted in large cities with populations of students or working-age adults. This study was the first to investigate exposures to $PM_{2.5}$ and BC as well as indoor noise in a rapidly growing mid-sized city and among retired adults, whose health is more likely to be impacted by air pollution [245].

My study also has several limitations. The sample size, both in terms of the number of participants/households (n = 78) and the number of neighbourhoods (N = 4) was relatively small. Further, the participation rate was low, potentially contributing to selection bias. If the reason for participating in the study was associated with indoor or personal exposure levels of $PM_{2.5}$, BC, or noise, then our results may have been biased. For instance, if those who had higher exposures to environmental pollutants were more likely to participate, our pollutant levels would be overestimating actual exposures to air pollutants within our target population: retired adults in Bucaramanga. However, I did not find evidence of this type of selection bias in my study. A low participation rate may have also biased the effect estimates in my model on determinants of exposure to $PM_{2.5}$. With 20% more women in my study, it is possible that more women tended to self-select. Since women tended to experience higher exposure to $PM_{2.5}$, then it is possible that the effect estimate may have been larger than if there was an equal distribution of men and women. Furthermore, we were unable to conduct repeated measurements in participants and households to assess whether measurements varied as a function of time or season. This limits the generalizability of my findings to different times and seasons, since the within-subject variance was not accounted for. Lastly, we did not collect time-activity diaries from participants, which could have provided more insight on the contribution of household environments in personal exposure to $PM_{2.5}$ or BC.

5.3 Conclusion

Little is known about the levels and determinants of $PM_{2.5}$, BC, and indoor noise levels in rapidly growing mid-sized cities in Latin America. My thesis aimed to address part of this gap by characterizing the levels and determinants of personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor noise. In doing so, I found that BC was relatively high, compared to $PM_{2.5}$, and indoor noise levels were high (mean of 55.2 ± 3.7 dB(A)) among participants/households in our study. Exposure assessment studies, like this one, can inform pollution abatement policies. My study lends support to policies aimed at reducing sources that contribute to household concentrations of, or personal exposures to, $PM_{2.5}$ or BC. Furthermore, policies that promote more natural building ventilation, associated with decreased exposure to $PM_{2.5}$ and BC, and less mechanical ventilation, associated with increased indoor Leq, may reduce levels of these prominent urban pollutants.

Future research with a larger sample size can include detailed assessments on sources of indoor air pollution and noise in Bucaramanga. Assessing time-activity patterns can also help determine where people are spending the most amount of their time, providing additional insight on the contribution of household air pollution to personal exposure. Further analyses on CO_2 decay in households can provide insight on relative ventilation levels for each, and thus, relative contributions of outdoor air pollutants, indoor. Furthermore, exploring the causal relationships between traffic volumes or vehicle fleet composition, especially heavy-duty vehicles, and personal exposure to $PM_{2.5}$, personal exposure to BC, and indoor noise, can clarify the role of traffic in determining levels of these urban environmental pollutants.

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