Ambient and household exposure to particulate matter in a low income settlement

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PREFACE

I Simon Dlangamandla Moletsane (27770826) declare that the information presented in this dissertation was written using my own words and understanding. I further declare that, in instances where other people's ideas and phrases were used in this document, such people were acknowledged and cited using the North-West University Harvard referencing style.

This dissertation is made up of seven chapters. **Chapter 1: Research Background** introduces the problem statement, justifies the relevance of the current research, and outlines the aim and objectives that this study intends to achieve. **Chapter 2: Literature Review** provides the existing body of knowledge relating to the subject matter under investigation and defines important concepts that will later be used throughout this document. **Chapter 3: Research Methods** outlines and discusses the study location and methodology applied to achieve the aim of this study. **Chapter 4: Characterising Ambient PM Concentrations in Lebohang** presents and discusses the results relating to the ambient PM concentrations in Lebohang. **Chapter 5: Indoor PM_4 Concentrations in Lebohang** presents the results and discussion of indoor PM_4 concentrations in Lebohang. **Chapter 6: PM Exposure by Subgroups in Lebohang** discusses and presents the results of the simulated residential exposure to PM concentrations in Lebohang estimated respectively for different subgroups. **Chapter 7: Summary and Conclusion** presents the main findings and limitations of the present study.

A portion of the results presented in this dissertation was peer-reviewed, presented, and published as a conference paper at the *National Association for Clean Air Conference* in 2020. The article derived from this dissertation is outlined below:

S.D. Moletsane., L. Tshisi., J.A. Adesina., R.P. Burger., B. Language. & S.J. Piketh. 2020. Assessment of indoor particulate matter concentrations in coal and noncoal reliant formal and informal households in Mpumalanga Highveld. Paper presented at 2020 Conference of the National Association for Clean Air, North-West, 20 Nov 2020.

ABSTRACT

Ambient and indoor exposure to particulate matter (PM) is a worldwide health problem, which is most prevalent in developing nations, and typically affects vulnerable groups (elderly, children, and sick people). In South Africa, where most low income communities rely on dirty solid fuels to satisfy their daily energy demands, the situation is no different. Although many houses are now electrified, the use of electricity as a primary energy carrier poses a huge economic burden to most impoverished households in the townships. Hence, people still rely on domestic combustion of dirty energy sources such as coal and wood to meet their household duties (cooking and heating) since are cheaper and easily accessible. Despite these benefits, little is known about the exposure that susceptible people, who live in such residential areas, experience because of the domestic burning of coal, in particular. This study was aimed at assessing the exposure of residential members that are subjected to ambient and indoor PM concentrations in Lebohang. Lebohang is a low income community located at the Mpumalanga Highveld, a region well known for having particulate concentrations, which frequently surpass the permissible NAAQS. To achieve the aim of this study, the outdoor $PM_{2.5}$ and PM_{10} concentrations were firstly characterised using the ambient data collected from the monitoring station in Lebohang. Secondly, the indoor PM_4 from 22 households that varied in structure and energy carrier use in the area; and which were sampled as part of the Sasol baseline monitoring campaign using DustTrak II Model 8530 monitors, was also characterised. Thirdly, the indirect exposure assessment method was used as well as the Monte Carlo simulation to estimate the PM exposure experienced by community members in Lebohang. These simulated estimates were based on outdoor and household PM measurements, the time activity data from the literature, and the 2011 Census community survey demographic information. It was found that ambient $PM_{2.5}$ and PM_{10} in Lebohang were mostly at unhealthy levels, particularly during the cold period (May-August 2018) as both pollutants were not compliant with NAAQS. Meanwhile, the relatively clean ambient air quality was observed during the months that constitute the warm period (October 2017-February 2018). In terms of the indoor PM_4, the 24h indoor concentrations of all houses enrolled in the study, regardless of their household type and primary energy carrier ranged between 24 and 482 $\mu g.m^{-3}$. Surprisingly, based on the structural types, on average, formal homes (109 ± 67 $\mu g.m^{-3}$) and informally structured households (105 ± 53 $\mu g.m^{-3}$) measured similar indoor PM_4 levels throughout the entire sampling period. Regarding PM exposure assessment, it was found that there are exposure inequalities in Lebohang and elderly people are the most susceptible group in this community. Finally, the study revealed that household exposure to PM contributes greatly to the total IPWE to PM concentrations residents experience in Lebohang.

Keywords: Indoor PM_{4}, ambient PM, formal households, informal households, integrated population-weighted exposure, vulnerable groups

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ABBREVIATIONS

AAP	Ambient Air Pollution
AEML	Atmospheric Emission Licenses
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CFH	Coal Formal Household
CIH	Coal Informal Household
CF	Correction Factor
DEAT	Department of Environmental Affairs and Tourism
DBW	Domestic Burning of Waste
ESP	Electrostatic Precipitators
FFP	Fabric Filter Plants
FGD	Flue Gas Desulphurization
H_2S	Hydrogen Sulfide
HPA	Highveld Priority Area
HAP	Household Air Pollution
IPWE	Integrated Population Weighted Exposure
MEs	Microenvironments
MES	Minimum Emission Standards
NAAQS	National Ambient Air Quality Standards
NEMAQA	National Environmental Management: Air Quality Act no. 39 of 2004
NFH	Non-coal Formal Household

NIH	Non-coal Informal Household
NO_x	Oxides of Nitrogen
NRF	National Research Foundation
PM	Particulate Matter
$PM_{2.5}$	Fine Particulate Matter
PM_4	Respirable Particulate Matter
PM_{10}	Coarse Particulate Matter
RDP	Reconstruction and Development Programme
SAS	Sasol Ambient Station
SAAQIS	South African Air Quality Information System
SANAS	South African National Accreditation System
SAWS	South African Weather Services
Stats SA	Statistics South Africa
SO_2	Sulphur Dioxide
SO_X	Sulphur Oxide
USEPA	United States Environmental Protection Agency
VTAPA	Vaal Triangle Airshed Priority Area
WHO	World Health Organisation

CHAPTER 1: RESEARCH BACKGROUND

This chapter provides a general overview of the problem of ambient and indoor particulate pollution, not only globally and nationally but as well as on a local scale. It also highlights some of the major behavioural and legislative challenges that complicate air pollution control, particularly in South African low income communities. Furthermore, it justifies the scientific significance and relevance of the present exposure assessment study, as well as the aim driving this research. Ultimately, it clearly outlines the three main objectives which this work intends to achieve.

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Ambient and indoor particulate pollution are health challenges, and both are ranked among the leading risk factors by the global burden of disease study (Lim *et al.*, 2012). Even at lower concentrations, exposure to PM harms human well-being (WHO, 2013). Worldwide, health risks associated with poor outdoor and household air quality attributes to over 4 million premature deaths (Cohen *et al.*, 2017). Such a burden is also felt in areas of Sub-Sahara Africa (Brauer *et al.*, 2012) and mainly impacts susceptible populations, i.e. women, infants, disabled and elderly people (Smith *et al.*, 2004). Similarly, in South Africa, illnesses triggered by air pollution are of significant concern, especially within low income settlements (Adesina *et al.*, 2019; Friedl *et al.*, 2008; Norman *et al.*, 2007; Wernecke, 2018). Mabuya and Scholes (2020) reported that annually, about 2000 early loss of children’s lives are attributable to respiratory infections caused by exposure to household air pollutants. Therefore, the air people inhale within poor communities violates their right to a clean and harmless environment as stipulated by section 24 of the Republic of South African constitution (Republic of South Africa, 1996).

South African low income areas are mostly situated close to industries (Adesina *et al.*, 2020; Scorgie, 2012; Moletsane *et al.*, 2021) and have diverse local PM sources (e.g. vehicle emissions, domestic combustion of solid fuels, resuspension of dust, waste, and biomass burning). These sources contribute to both ambient air pollution (AAP) and household air pollution (HAP). Human exposure to PM concentrations, however, is predominantly associated with the latter as a result of the combustion of solid fuels in low income communities (Desai *et al.*, 2004; Language *et al.*, 2016; Norman *et al.*, 2007). Households in low income settlements often rely on solid fuels for their primary energy requirements since they are more economical and can fulfil multiple functions, i.e. cooking and space heating simultaneously (Buthelezi *et al.*, 2019; Langerman &

Pauw, 2018). Such energy carriers are usually combusted using cracked inefficient stoves (Nkosi *et al.*, 2017b; Pauw, 2017), thus, emitting substantial particulate concentrations within indoor environments, where more than 80% of people's time is spent (Arvanitis *et al.*, 2010; Desai *et al.*, 2004; Language *et al.*, 2015).

The government has tested numerous strategic interventions to resolve the issue of human exposure to ambient and indoor air pollution in low income settlements. These include but are not limited to increasing households' accessibility to electricity, subsidising cleaner fuels, introducing less smoke-emitting burning methods, and providing properly constructed formal households (Langerman *et al.*, 2018). By far, Mugabo (2011) has shown that electrification has effectively reduced outdoor air pollutants; however, for HAP, the opposite is the case. Enhancing electrification was aimed at shifting residential reliance mainly from dirty solid fuels to cleaner energy carriers (Nkosi *et al.*, 2017a). However, despite electricity subsidies, traditional energy sources are still used in impoverished communities as most people are unemployed or only dependent on social grants for income (Qhekwana, 2019). Moreover, for some households, especially informal ones, the burning of solid fuels is their only energy option since most of them are yet still unelectrified (Langerman & Pauw, 2018). Therefore, this complicates HAP management because prohibiting solid fuel use in such households will be synonymous with criminalizing poverty.

It is estimated that over 4 million people in poor communities are still reliant on either animal dung, wood, or coal to meet their daily energy demands (Kapwata *et al.*, 2018). This is one of the reasons why human exposure to PM concentrations in townships is more severe than in urban places (Hersey *et al.*, 2014). The residential reliance on traditional fuels is motivated by several factors, for example, consistent power cuts, high cost of electricity, and high poverty levels (Buthelezi *et al.*, 2019; Nkosi *et al.*, 2017b; Qhekwana, 2019). Typically, the use of electricity in these settlements is associated with activities that include refrigeration, lighting, and entertainment (Balmer, 2007). Meanwhile, intense energy-consuming duties such as cooking and winter season space heating are often performed by burning wood and coal to avoid the cost of electricity (Nkosi *et al.*, 2017b). Emissions produced by these solid fuels lead to human exposure to both ambient and indoor particulate concentrations. Consequently, increasing the vulnerability of susceptible population groups to air pollution-related health risks such as tuberculosis, lung cancer, acute respiratory infections, and asthma (WHO, 2013).

Coal is among the frequently used dirty solid fuels in South African low income areas on the South African Highveld, especially during the winter season, as it is easily accessible and affordable (Buthelezi *et al.*, 2019; Naidoo *et al.*, 2015). Over 950 000 households are coal-reliant and emit about 3% of the total coal emissions generated in the country (Van den Berg, 2015). These houses are mainly concentrated in places surrounded by mines, coal-fired power plants, and train railways supporting coal transportation (Buthelezi *et al.*, 2019; Qhekwana, 2019). Such areas include, for instance, low income settlements in Gauteng, North Eastern Free State, and the Mpumalanga Highveld region (Langerman & Pauw, 2018). The use of coal in these places is not only driven by its affordability, however, also by the resident's cultural preferences. This is an overlooked factor when managing and mitigating indoor and ambient exposure to air pollutants in impoverished communities. For instance, inhabitants prefer to cook food such as tripe and samp using coal-reliant traditional methods because if not, the authentic taste of these meals changes (Nkosi *et al.*, 2017b). As such, electric stoves are ineffective in replacing the role of coal as far as taste preservation of particular cultural food is concerned (Nansaior *et al.*, 2011). Therefore, the resident's cultural behaviours make it challenging to address the persistent issue of human exposure to PM because of coal use in low income settlements.

The National Ambient Air Quality Standards (NAAQS) stipulate the allowable PM concentrations to regulate human exposure to harmful particulate loads. Table 1-1 below shows the South African NAAQS for $PM_{2.5}$ and PM_{10} concentrations. Even though other pollutants such as oxides of nitrogen (NO_x), sulphur dioxide (SO_2), and carbon monoxide (CO) are also regulated by NAAQS, in terms of PM management, the NAAQS is exclusively designed to control $PM_{2.5}$ and PM_{10} emissions. PM_4 is one of the significant pollutants released during the combustion of coal for domestic purposes (i.e. cooking and space heating) (Kapwata *et al.*, 2018). In recent years, researchers have shown that household members are typically exposed to high indoor PM_4 concentrations, especially within low income settlements (Adesina *et al.*, 2019; Language *et al.*, 2015; Langerman *et al.*, 2018; Qhekwana, 2019; Letsholo, 2021). The need for assessing the total exposure of people in these areas to both ambient and indoor air pollution is therefore justified.

Table 1-1: National Ambient Air Quality Standards (NAAQS) for PM_{10} and $PM_{2.5}$ concentrations ($\mu g.m^{-3}$) in South Africa (Republic of South Africa, 2009).

Priority pollutants	**Averaging period**	**Standard concentration ($\mu g/m^{-3}$)**	**Annual frequency of permissible exceedances**
PM_{10}	24 hours	75	4
$PM_{2.5}$	24 hours	40	4

Accurate estimations of exposure to PM concentrations that people in low income areas experience outdoors and indoors are critical as they identify the main micro-environments that are mostly health-threatening (Park *et al*., 2020). Moreover, they also outline places that need to be prioritised when conducting PM exposure health assessments and drafting air quality management interventions for impoverished residential places. It is, however, complex to accurately estimate population exposure to PM concentrations in a given community as residential exposure varies in time and space due to different movement patterns of various subgroups (Jafta *et al.*, 2017; Wernecke, 2018).

The residential exposure PM concentrations estimates, drawn particularly in dense low income settlements, are typically solely based on the neighbourhood ambient concentrations (Faria *et al.*, 2020; Language *et al.*, 2015). However, many argue that this method is an unreliable proxy for assessing personal exposure to particulate pollution (Bo *et al.*, 2017; Language *et al.*, 2015; Wernecke, 2018). The reason is such an approach introduces three main limitations that somewhat disregard some of the most critical elements of exposure. Firstly, it assumes that residential exposure to PM concentrations on a neighbourhood scale is uniform (Hswen *et al.*, 2019; Krasnov *et al.*, 2016). Consequently, it ignores the complexity of spatial and temporal variation of PM (Moletsane *et al.*, 2021; Wernecke, 2018; Wilson *et al.*, 2006), which influences personal exposure to similarly vary in space and time amongst different individual residents in the community. Secondly, given that ambient PM concentrations vary from those in the household environment (i.e. where people spend most of their time), the indoor exposure calculated from outdoor measurements has uncertainties. Thirdly, estimates of household exposure derived solely from the ambient concentrations neglect the time activity patterns of various demographic groups, leading to erroneous estimates of personal exposure to particulate pollution (Faria *et al.*, 2020). This can result in incorrect conclusions, especially when assessing the effectiveness of an intervention in mitigating poor air quality.

Therefore, there is a need for an appropriate population exposure assessment that considers not only the ambient and household PM concentrations, however, also the portion of time people spend in each of these environments. The indirect exposure method is a good measure for assessing the exposure levels of people in low income communities since it is less economical and can be applied to a widespread population sample. However, this approach is also associated with limitations that might cause uncertainties in the results (Ezzati & Kammen, 2001). Notwithstanding, provided that such shortfalls are accounted for, the indirect method can provide a clear picture of exposure profiles of different demographic groups in low income areas.

Understanding the population exposure to PM concentrations based on residents' different population subgroups in low income communities is important for 1) designing and developing effective mitigation strategies for AAP and HAP; 2) assessing exposure inequalities at different microenvironments in low income communities; 3) estimating the health risks associated with exposure profile for each demographic group at different locations; and 4) assessing the effectiveness of the intervention strategies in reducing the burden of exposure amongst vulnerable groups in impoverished communities. Therefore, this study aims to assess the exposure profiles of residential members in Lebohang low income township to determine the burden of exposure associated with different subgroups at various microenvironments in the community. Lebohang was chosen because 1) it is situated close to industries, 2) it is within the Highveld Priority Area, 3) it has environmental and socio-economic components that facilitate air pollution, and 4) it is representative of typical low income areas on the Highveld.

1.1 Scientific relevance of the study

Numerous studies have observed that ambient and household PM concentrations in low income communities, particularly those in the South African Highveld region are often beyond NAAQS (Adesina *et al.*, 2019; Chidhindi *et al.*, 2019; Friedl *et al.*, 2008; Jafta *et al.*, 2017; Kapwata *et al.*, 2018; Language *et al.*, 2015; 2016; Moletsane *et al.*, 2021; Mugabo, 2011; Nkosi *et al.*, 2017a; Qhekwana, 2019; Wernecke, 2018). Given that the NAAQS was formulated to protect vulnerable groups from harmful levels of criteria pollutants, understanding the exposure levels resident experiences daily is important. Although several studies were conducted on the Highveld, a limited number have estimated population exposure by considering both the ambient and household PM measurements, as well as the time activity pattern of different population groups in the township. The results of this study will:

(i) Contribute to the global studies about suitable approaches which can be used to measure population exposure to air pollutants in urban settlements

(ii) Provide insight into the state of HAP in different structural households, which use different energy sources in South African low income communities

(iii) Contribute to the township geography by enhancing our understanding of the outcomes of different time activity patterns on the estimates of residential exposure

(iv) Provide an insight into the extent of exposure inequalities in low income communities

1.2 Research aim and objectives

This research aims to assess the exposure of residential members that are subjected to ambient and indoor PM concentrations in Lebohang's low income settlement. The specific objectives that are set to achieve the aforementioned aim of the study are to:

1. Characterise the ambient PM concentrations in Lebohang low income settlement,
2. Assess the indoor PM concentrations in Lebohang low income settlement, and
3. Calculate the exposure by population subgroups for community members that are subjected to ambient and household PM concentrations in Lebohang low income settlement.

CHAPTER 2: LITERATURE REVIEW

Chapter 2 provides a reader with an overview of information about the physical characteristics of PM as well as how different particles are classified based on their fraction sizes. Moreover, this chapter outlines and discusses the main emission sources of particulate pollution that are commonly identified in the literature as the driving forces of poor air quality within low come communities. The chapter further discusses and defines the concept of exposure and dose and makes a clear distinction between the two. Ultimately, this chapter focuses on the methods of exposure assessment, health impacts of PM, and PM legislative control in South Africa.

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PM is typically indicative of poor air quality and amongst other toxic atmospheric air pollutants, has by far, severe health impacts (Lim *et al.*, 2012). It is a compound pollutant that is mainly made up of species such as water, mineral dust, sulphate, black carbon, sodium chloride, ammonia, and nitrates (Yang *et al*., 2018). PM, therefore, can be thought of as a complex combination of particles (both organic and inorganic) that are either in liquid or solid forms and are suspended in the atmosphere (Bilal *et al.*, 2019; Kim *et al.*, 2015; Language, 2020; Lü *et al.*, 2019; Tshehla & Wright, 2019). Particulate pollution can be formed from emissions suspended straight into the atmosphere (called primary particles) and from the transformation of other gaseous pollutants such as SO_2 and NO_x (called secondary particles) (Kim *et al.*, 2015). Atmospheric PM is formed by both emission sources from anthropogenic and natural activities (Amaral *et al.*, 2015; Language, 2020). Anthropogenic sources include petrochemical industries, coal-fired power stations, the burning of solid fuels, and industrial activities. Meanwhile, PM emissions that are naturally suspended in the air originate from processes such as sea spray, forest fires, dust resuspension, and volcanic eruptions (Kim *et al.*, 2015; Yatkin & Bayram, 2008).

Airborne PM varies in physical characteristics (e.g. density, size, and shape), and chemical composition (Hu *et al.*, 2015; Tshehla & Wright, 2019; Van Dingenen *et al.*, 2004). This is because atmospheric aerosols are suspended by different emission sources, formation processes, and chemical properties. Although the chemical composition is important for understanding the toxicity of PM and its sources (Araújo *et al.*, 2014), only PM's physical properties will be discussed. PM can be suspended in the atmosphere in four (4) different morphologies, either as 1) "*dust contaminant*", 2) "*biological contaminants*", 3) "*particulate contaminants*", or 4) "*gaseous contaminants*" (Tawabini *et al.*, 2017). Likewise, PM is emitted in numerous shapes by different types of particles. For example, when suspended from clay particles it is platy, irregular when emitted from dust particles, or spherical when generated from pollen particles. The geographical location where particles are emitted influences the nature of their physical properties, thus, the place where PM is emitted will determine its size and shape (Tawabini *et al.*, 2017).

2.1 Classification of particulate matter

There are several ways in which PM can be classified, but for convenience, it is commonly distinguished based on its "*diameter*" (aerosols' unique fraction size) (Kim *et al.*, 2015). Although diameters exist in different forms, which range from an aerodynamic, stroke, to an equivalent diameter, the latter is the most frequently used type in air pollution studies (Chow *et al.*, 2002; Language, 2020). The aerodynamic diameter is "*the diameter of a unit density sphere of the same settling velocity as particles in the question*". This is one of the major criteria for PM classification as it determines how aerosols behave in a particular environment (Araújo *et al.*, 2014; Kim *et al.*, 2015; Milford & Davidson, 1985). For instance, it influences particles' life span and transportation in the atmosphere as well as the depth of their deposition in the respiratory system when inhaled (Araújo *et al.*, 2014; Milford & Davidson, 1985). Furthermore, the reduction of visibility due to PM is found to be, amongst other reasons, influenced by the diameter of particles. Therefore, size is one of the major influential aspects that determine the behaviour and fate of particles (i.e. either in the body or atmosphere); and classifies PM into different categories.

The atmospheric particles can range from a considerable aerodynamic diameter that exceeds 100 micrometers (µm); to the smallest fraction size of 1 nanometer (nm) (Chow *et al.*, 2002; Vallius, 2005; Van Dingenen *et al.*, 2004). Thus, for convenience, PM is annotated as PM_X (where X is aerodynamic diameters in µm), to usually show that its constituent particles are either below 1, 2.5, or 10 µm in size (Vallius, 2005). Classification of PM based on size fraction is conducted either using regulatory, cut point, occupational health, or mode sizes. Each of these four (4) conventional ways of PM classification is based on a different method for classifying airborne particles (Chow *et al.*, 2002). However, the common convention often used for classifying PM is the mode sizes, which define PM into four (4) different size distributions (Taiwo *et al.*, 2014). These are coarse (diameter between 2.5-10 µm), fine (diameter between 0 and 2.5 µm), accumulation (diameter between 0.08-2 µm), and nucleation (diameter ≤ 0.1 µm) PM. Figure 2-1 below is a diagram that shows different modal size distributions of atmospheric PM. Language (2020) emphasises the significance of differentiating between fine ($PM_{2.5}$) and coarse (PM_{10}) PM size fractions as they both vary in physical nature. Table 2-1 below summarises and outlines some of the most fundamental characteristics that make $PM_{2.5}$ and PM_{10} different.

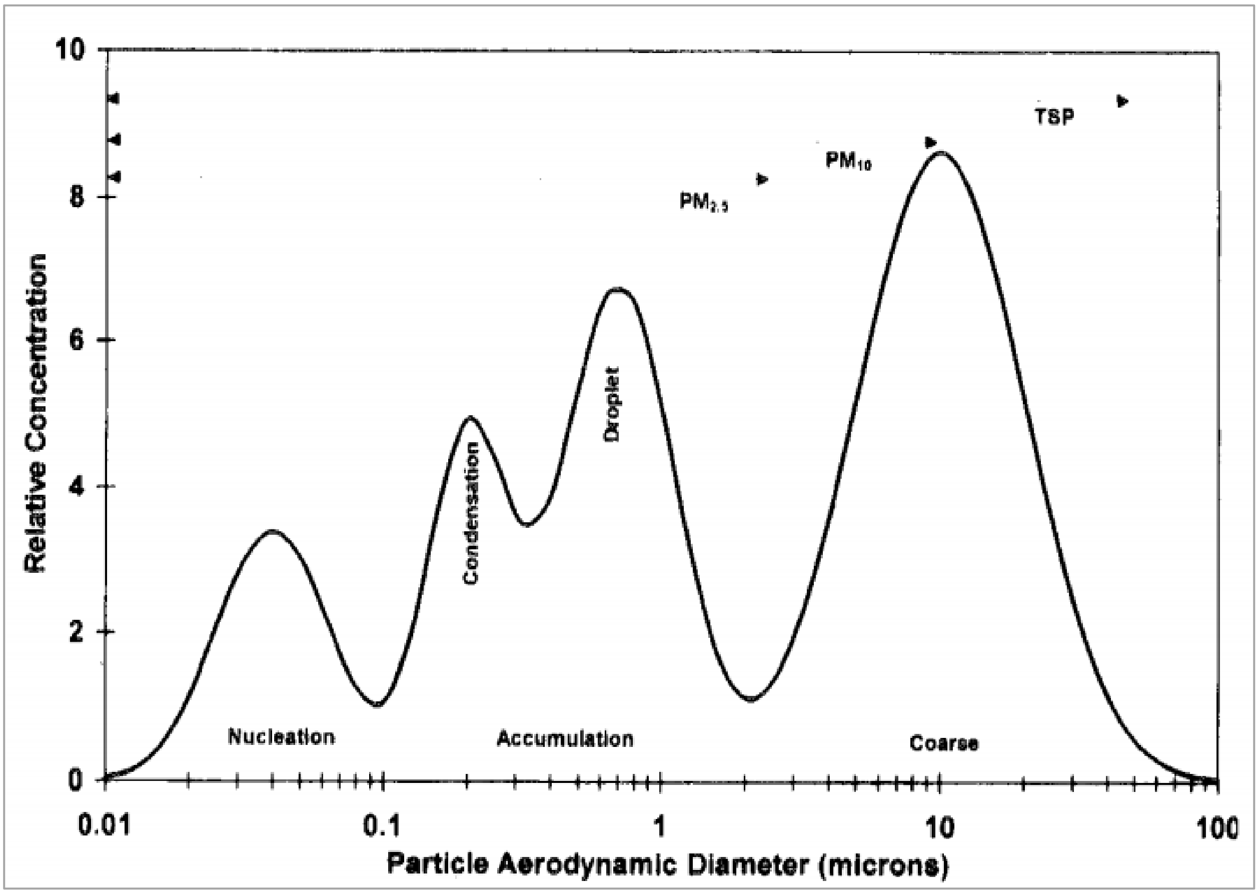

Figure 2-1: Different modal size distributions (in µm) of atmospheric PM ranging from nucleation, accumulation, and fine to coarse particle mode (Chow, 1995).

Table 2-1: Comparative analysis of the properties that make fine and coarse mode particle matter different.

Characteristics of particles	$PM_{2.5}$	PM_{10}	Sources
Aerodynamic diameter (µm)	≤ 2.5 µm	≤ 10 µm	(Kim *et al.*, 2015; Language, 2020)
Emission sources	Combustion of solid fuels (wood, coal); precursor gases (SO_2, NO_x).	Resuspension dust (e.g. from unpaved streets, mining, industries, farming); sea pray; burning of solid fuels.	(Adesina *et al.*, 2019; Kim *et al.*, 2015; Taiwo *et al.*, 2014)
Duration of particles' existence in the atmosphere	Ranges from a minimum of a few days to a maximum of weeks	Ranges from a minimum of 1 day to a maximum of 10 days	(Araújo *et al.*, 2014)
The distance particles can travel in kilometers (km)	100 - 1000	1 - 10	(Qhekwana, 2019)

2.2 Emission sources of particulate matter in low income settlements

Emission sources of PM in South African low income settlements are diverse and there is often inconsistency in how they are mentioned in previous studies (Adesina *et al.*, 2019; Altieri & Keen, 2019; Engelbrecht *et al.*, 2002; Feig *et al.*, 2016; Fuzzi *et al.*, 2015; Maritz *et al.*, 2020; Mugabo, 2011; Nkosi *et al.*, 2017c; Qhekwana, 2019; Scorgie, 2012; Tshehla & Wright, 2019; Venter *et al.*, 2012). However, some of the major PM sources which are commonly identified in low income communities include vehicle emissions, industrial emissions, coal power plants, aeolian dust, waste, biomass burning, and domestic combusion of solid fuels. Each of the aforementioned PM sources will be fully discussed in detail in the following subsections below:

2.2.1 Vehicle emissions

Traffic emission is one of the main sources of particulate air pollution within South African low income areas (Altieri & Keen, 2019; Mugabo, 2011). The temporal profile of vehicle emissions within these communities is often characterized by variable diurnal patterns (Lourens *et al.*, 2016). Previous studies have shown that the traffic flow in low income communities displays two diurnal circles; the first one between 06:00 and 09:00; and the second one between 16:00 and 18:30 (Feig *et al.*, 2016; Wernecke *et al.*, 2015). Furthermore, it was found that vehicles in low income residential areas commonly emit high PM concentrations during the weekdays as opposed to weekends. This is because, during the week, there are increased timing activities as people travel back and forth from home to either work or school (Wernecke, 2018). Traffic emissions mostly impact low income residential areas that are near highways, as well as township microenvironments that are either close to taxi ranks or main entrance roads (Engelbrecht *et al.*, 2002; Van den Berg, 2015).

The frequency of the PM concentrations emitted in a particular township, because of emissions generated from the vehicles' exhaust pipes, can be influenced by several factors (Qhekwana, 2019). It depends, for instance, on vehicles' annual mileage, driving speed, regulatory emission technology, age, and fuel type (i.e. it uses either gasoline or diesel) (Mbandi *et al.*, 2019). Despite exhaust pipes, vehicles also have other ways of emitting particulate concentrations. For instance, PM is emitted when the car brakes due to the frictional force occurring between the tires and surface, as well as when the vehicle passes over the road as it suspends dust particles (Kim *et al.*, 2015; Zhao *et al.*, 2018). The magnitude which vehicles emit PM concentrations into the airshed of a given community thus, depends on several factors, which are neighbourhood specific and vary across different microenvironments.

2.2.2 Industrial emissions

Industries are the heart of the South African economy, and it is estimated that they emit at least about 30% of the overall PM emissions suspended in the country (Jury, 2017; Morakinyo *et al.*, 2017). The Highveld Priority Area (HPA), as well as the Vaal Triangle Airshed Priority Area (VTAPA), are both considered to be the largest industrial hubs in South Africa. These regions are home to PM sources such as open-cast coal mining, oil refinery operations, smelters, and metallurgical processes, to mention a few. As low income settlements are often spatially located within the vicinity of industries (Hersey *et al*., 2014; Okello & Allan, 2015), the industrially emitted PM, either from stack or vent emissions thus, degrade the air quality of such communities (Khumalo *et al.*, 2020; Okello *et al.*, 2018). Sources of industrial emissions vary from one industry to another and are often influenced by the nature of the production processes involved. For instance, nonpoint industries such as opencast mines suspend PM through surface explosions, which emit fugitive dust; tailing dams, mine dumps; the movement of heavy diesel-engined machinery; coal transportation; ground drilling; and wind erosion (Jaggernath, 2013). However, point sources such as oil refineries emit PM through activities that include heating of oil fluids, combustion of fuels at extreme temperatures, or even through devices or equipment leaks.

PM emissions emitted by industries (either directly or indirectly) into the atmosphere are characterised by less annual variability (Josipovic *et al.*, 2010; Mathuthu *et al.*, 2018; Piketh *et al.*, 1999). This is because the production of industries is somewhat constant throughout the year thus, suspended pollutants follow a similar pattern. However, during the day, these emissions, particularly those emitted by tall stack sources, are generally identified by a midday spike if suspended into the adjacent township airshed (Chidhindi *et al.*, 2019). Because of the heights of the installed stacks (frequently ranging between 200m-300m), industrial emissions are deposited above the inversion layer during nightshift production. Consequently, these pollutants are blocked within the lower troposphere thus, cannot subside into the ground surface (Collett *et al.*, 2010). However, due to the presence of solar radiation around midday, such pollutants are released into the ground as the boundary layer dismantles (Venter *et al.*, 2012). Therefore, this results in low income residents being exposed to industrial emissions during the middle of the day, typically between 10:00 and 13:00 (Collett *et al.*, 2010).

2.2.3 Coal power plants

A large fraction of electricity that is produced in South Africa is generated through coal combustion by power plants (Winkler, 2005). Most of these power stations are widely spread over the Highveld and are the major point sources of pollutants within the region, however, with an exception for primary PM (Venter *et al.*, 2012). Over 99% of the primary PM emissions that are suspended as ash after the combustion process are effectively trapped before being emitted into the atmosphere (Scorgie, 2012). This is done by either using Fabric Filter Plants (FFP) or Electrostatic Precipitators (ESP) (Langerman & Pauw, 2018; Qhekwana, 2019). These are emission control technologies that are connected to the flue gas stream and responsible for the removal of PM before it escapes from the tall stacks (Pretorius, 2015).

Notwithstanding, power plants contribute to the formation of secondary PM since nearly none of them control precursor gases suspended during electricity production (Langerman & Pauw, 2018; Maritz *et al.*, 2020). Gaseous pollutants that are suspended through the stacks of power stations during electricity production include NO_X, and sulphur oxide (SO_X) (Langerman & Pauw, 2018; Piketh *et al.*, 1999). Coal-fired power stations are reported as being the main point source of SO_X (emitting up to 61% of the overall SO_X) in the country (Ross *et al.*, 2003). This is because several coal-fired power plants in South Africa use ESP, an abatement technology that relies on high SO_X composition in coal to perform efficiently and effectively (Pretorius, 2015). As such, there are precursor gases that are produced during electricity generation such as SO_2 and NO_X, which are then chemically converted into secondary PM in the atmosphere (Piketh *et al.*, 1999; Ross *et al.*, 2003). It must be noted that coal-fired power plants do not only emit PM through the combustion processes and the production of precursor gases; however, they also emit it from sources such as coal storage and flue ash as fugitive dust. Therefore, although power plants invest in emission reduction technologies, they remain the sources of secondary PM (Qhekwana, 2019), which impacts the airshed of the adjacent townships.

2.2.4 Aeolian dust

Annually, the aeolian dust contributes significantly to the total PM emissions emitted in South Africa (Piketh *et al.*, 1999). Aeolian dust can be suspended in the atmosphere by both local as well as regional sources of emissions. Coarse fraction dust particles are often emitted by local sources while fine fraction dust particles are mainly suspended by regional emission sources. Typical local sources of aeolian dust in low income settlements include untarred roads, unpaved roads, surface sand, soil, and crustal material (Van den Berg, 2015). On the other hand, regional dust emissions sources include semi-desert and desert areas, particularly those in the western parts of South Africa, mining tailings storage facilities, and fly ash (Nkosi *et al.*, 2017c; Piketh *et al.*, 1999). The frequency of dust particles emitted into the atmosphere is influenced by the prevailing meteorology and climate (Oguntoke *et al.*, 2013; Van den Berg, 2015). In South Africa, extreme aeolian dust events are typically observed during the dry winter season; meanwhile, the least episodes are commonly experienced in the wet summer period (Piketh *et al.*, 1999). However, in residential areas located on the southwestern coast of the country, aeolian dust events are extreme during dry summer seasons as opposed to wet winter periods. This is because such places experience substantial rainfall during winter rather than in summer periods (Piketh *et al.*, 1999). Therefore, relative humidity experienced in a township influences the total aeolian dust particles emitted into the ambient environment.

2.2.5 Waste burning

It is estimated that over 5 million tonnes of household-generated waste are burned each year in South African residential areas (Wiedinmyer *et al.*, 2014). Domestic burning of waste (DBW), particularly within low income settlements, is a daily challenge thus, considered one of the most significant sources of PM emissions in such areas (Van den Berg, 2015). Because of the poor supply of oxygen and low temperature involved in the process of DBW, waste is often inefficiently combusted hence, substantial PM concentrations are emitted into the environment (Krecl *et al.*, 2021). The concentration of PM suspended during DBW is often high because residents do not control or minimize the emissions suspended during the burning process (Wiedinmyer *et al.*, 2014). Typical waste materials that are often combusted during DBW vary from papers, plastics, and discarded furniture (e.g. beds and mattresses), to organic waste. Although local municipalities are responsible for household waste collection in low income communities, their services are often either inconsistent or ineffective (Ngeleka, 2010). This is frequently driven by factors such as poor governance, limited resources (e.g. trucks), and an excessive amount of township-generated waste that usually surpasses municipal collection efficiency. Consequently, residents usually resort to DBW as an alternative technique for:

- disposing of uncollected waste (Wiedinmyer *et al.*, 2014);
- avoiding the accumulation of waste as household layouts in low ncome areas do not have enough space for supporting the storage of waste (Krecl *et al.*, 2021);
- preventing uncollected waste from breeding and attracting rodents, rats, worms, and insects (Qhekwana, 2019); and
- avoiding openly disposed waste from generating unpleasant odours near their households (Krecl *et al.*, 2020).

2.2.6 Biomass burning

The burning of biomass (i.e. vegetation and grass) is a source of PM emissions and it is estimated that it contributes about 5% of aerosol loading in South Africa (Piketh *et al.*, 1999). The fraction of particulate pollution emitted from biomass burning is relatively smaller in comparison to that of other pollutants suspended during the combustion process. PM is formed by only 5% of the carbon suspended during biomass burning; meanwhile, most of this carbon (up to 90%) is converted to either form CO or carbon dioxide (CO_2) (Reid *et al.*, 2005). Notwithstanding, biomass burning can also be a major source of PM emissions in low income settlements, provided it occurs close to these communities, especially during the peak season of burning events (Piketh *et al.*, 1999). Typically, extreme biomass burning cases occur during the dry season, particularly between May to October (Malaza, 2017). This is because the moisture content of bush vegetation during this time of the year is minimal thus, making it easier for a significant portion of the biomass to be consumed by the fire (Otter *et al.*, 2001).

Biomass burning can occur as a result of both anthropogenic activities (e.g. slash and burn) and natural processes (e.g. lightning) however, the former is considered to be the main cause of wildfires (McKechnie, 2010). PM emissions are typically emitted during the smouldering phase of biomass burning because of incomplete combustion, which is caused by the lack of temperature involved during this phase (Otter *et al.*, 2001). Although particles are also formed when there is a high temperature involved in the combustion process, particularly during the flaming phase, they are suspended as condensation nuclei. Subsequently, a significant portion of PM emissions suspended during biomass burning is in the accumulation size distribution mode (Reid *et al.*, 2005).

2.2.7 Domestic combustion of solid fuels in low income settlements

PM is emitted as the main by-product pollutant of the domestic burning of solid fuels (Adesina *et al.*, 2020). In South African low income settlements, residential combustion of solid and liquid fuels for domestic activities is a common practice (Bosman, 2019; Langerman *et al.*, 2018; Mdluli

& Vogel, 2010; Nkosi *et al.*, 2017b). Over 12% of the houses in the country use traditional fuels, particularly for space heating and cooking (Langerman *et al.*, 2018). As such, people in low income residential areas bear the greatest burden of HAP than those in other urban parts of the country (Norman *et al.*, 2007). There are different kinds of liquid and solid fuels used for domestic purposes in impoverished settlements. To mention a few, such include petroleum liquid gas, paraffin, animal dung, coal, and wood (Bosman, 2019; Language, 2020; Norman *et al.*, 2007; Qhekwana, 2019). However, to some extent, the latter two energy sources are typical fuel types that are predominantly used by most homes within impoverished residential areas (Adesina *et al.*, 2019; Nkosi *et al.*, 2017b). Although wood is used across South African low income townships, it is often relied upon in rural areas (Davis, 1998; Qhekwana, 2019). In contrast, the use of coal is widely spread in townships located within mine-concentrated provinces such as Free State, Gauteng, KwaZulu Natal, and Mpumalanga, see Figure 2-2 below.

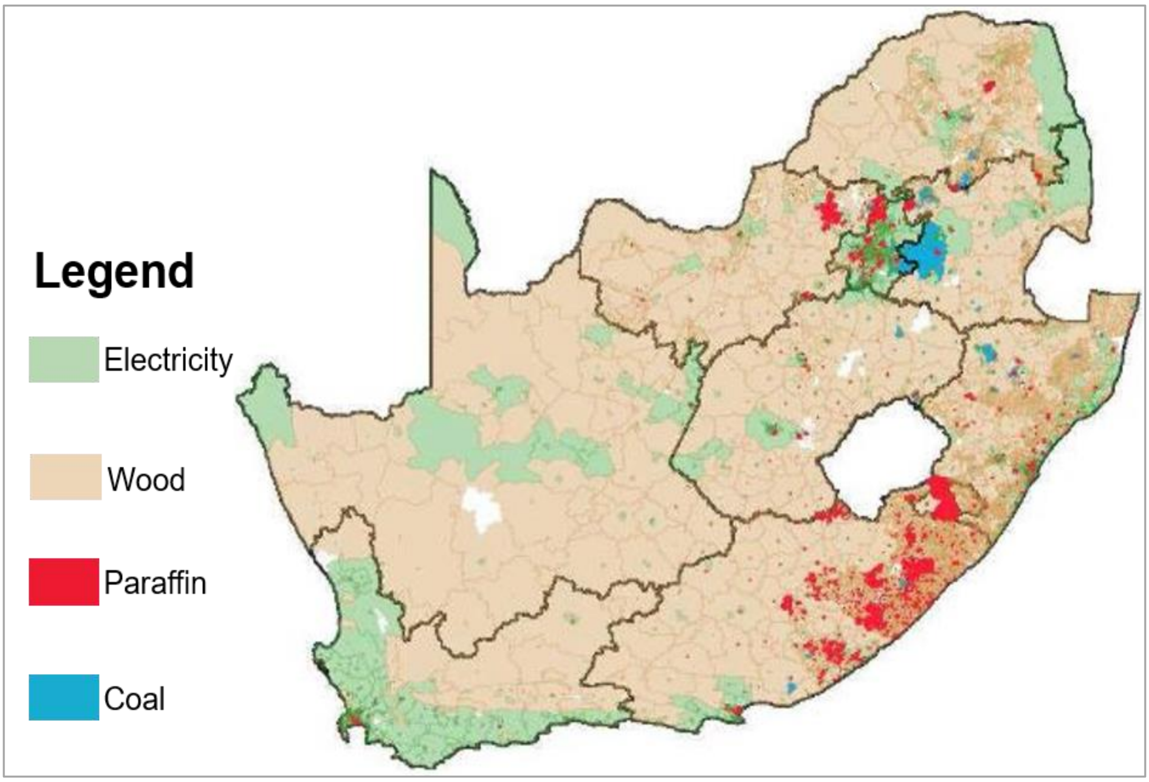

Figure 2-2: The spatial distribution of the main household energy sources used for domestic purposes across South Africa (Langerman *et al*., 2018).

Although there is over 80% of electrified households in South Africa, solid fuels are still heavily relied on, particularly in low income settlements (Adesina *et al.*, 2019; Buthelezi *et al.*, 2019; Mdluli & Vogel, 2010; Nkosi *et al.*, 2017a; Qhekwana, 2019). Based on the model of the energy ladder, income is the most influential factor which determines the type of fuel households choose to use for lighting, recreation, housewarming, and cooking (Annecke, 1999; Scorgie *et al.*, 2003). As such, electricity use in low income settlements, particularly for cooking and space heating, is expensive due to the prevalence of poverty and unemployment (Language *et al.*, 2016; Mdluli & Vogel, 2010). As a result, people often use electricity for lighting and recreational purposes; meanwhile, for cooking and space heating, they resort to affordable and easily accessible fuels such as coal and wood (Buthelezi *et al.*, 2019; Davis, 1998; Hersey *et al*., 2014; Mdluli & Vogel, 2010; Scorgie *et al.*, 2003; Winkler, 2011). This, therefore, allows them to spend most of their income (which is typically social grants) on the essentials and little on cleaner sources such as electricity.

Even though income is acknowledged as influential, many argue that the concept of the energy ladder oversimplifies the complexity of interplaying factors that encourage solid fuel use in townships (Davis, 1998; Farsi *et al.*, 2007; Hosier & Dowd, 1987; Mdluli & Vogel, 2010). The concept of the energy ladder is therefore criticized for not considering, for instance, 1) the fact that in some instances, residents use traditional fuels as a backup energy supply due to the pressing issue of power cuts in low income communities; 2) impacts of cultural preferences, which often determine residents' cooking habits, diet, and judgment of taste; 3) lack of access to electricity, especially within informal settlements; and 4) dual benefits of solid fuel use, i.e. simultaneous cooking and space heating.

Despite the incentives behind solid fuel use in low income residential areas, traditional fuel sources are commonly combusted using inefficient imbhawulas (braziers) and coal iron stoves (Balmer, 2007). Consequently, PM is emitted directly into the households and ambient environment thus, posing a threat to the well-being of residents as they inhale high concentrations of particulate pollution (Pauw, 2017). The situation is even worse for informal households since they are often poorly ventilated hence, trapping the concentrations from escaping into the outdoor environment (Zimmermann *et al.*, 2017). The combustion pattern of households in low income settlements is generally characterized by two main events, i.e. the morning (03:00-06:00) and evening (15:00-21:00) peaks of solid fuel emissions. This pattern is frequently found to be more pronounced during winter, as opposed to summer, due to the increased residential consumption of dirty fuels as a result of cold ambient temperature (Adesina *et al.*, 2019; Hersey *et al*., 2014). Low income residents are thus, often exposed to substantial PM concentrations during the peak season of domestic solid fuel burning.

2.3 Exposure to particulate matter concentrations

There is confusion about the exact meaning of the word "*exposure*" as various scientists in environmental research use the term to refer to diverse concepts (Klepeis *et al.*, 2001; Moschandreas & Saksena, 2002). Because the term has no agreed universal definition, in this study it refers to the total time when either a group of people or an individual gets in contact with a specific pollutant that has a particular concentration (Faria *et al.*, 2020). Based on this definition, exposure, therefore, occurs immediately when a person's external boundaries (e.g. skin, mouth, or nostrils) physically touch PM concentrations. The concept thus does not reflect ingestion, inhalation, or uptake of PM concentrations an individual is experiencing in a particular time and space (Hertel *et al.*, 2001; Monn, 2001; Steinle *et al.*, 2013; Wernecke, 2018).

Personal exposure to PM concentrations can occur at any time of the day in both ambient and household environments during different activities such as commuting, cooking, and exercising (Pant *et al.*, 2016). It is hence, estimated from two parameters, i.e. the time activity pattern of an exposed individual and the pollutant concentration in a predefined time and place of exposure (Bruce *et al.*, 2000; Ezzati & Kammen, 2002; Klepeis, 2006; Wang *et al.*, 2008). Estimates of exposure derived from integrating both these parameters over a specified time are referred to as "*integrated exposure*" however, are often understood as "*average exposure*" (Faria *et al.*, 2020; Monn, 2001). Figure 2-3 below graphically shows the fundamental variables and specific considerations which define the occurrence of the exposure and integrated exposure. Integrated personal exposure, as recommended by the EPA (1992), is calculated by using the outlined mathematical formula below:

$$E = \sum_{j=1}^{m} C_j t_j$$

Equation (1)

Where E = integrated exposure, C_j = the quantified magnitude of the concentration of a specified pollutant in a predefined environment j in micrograms per cubic meter ($\mu g.m^{-3}$), and t_j = the duration of time a person or a group of people spent in an air-contaminated environment j.

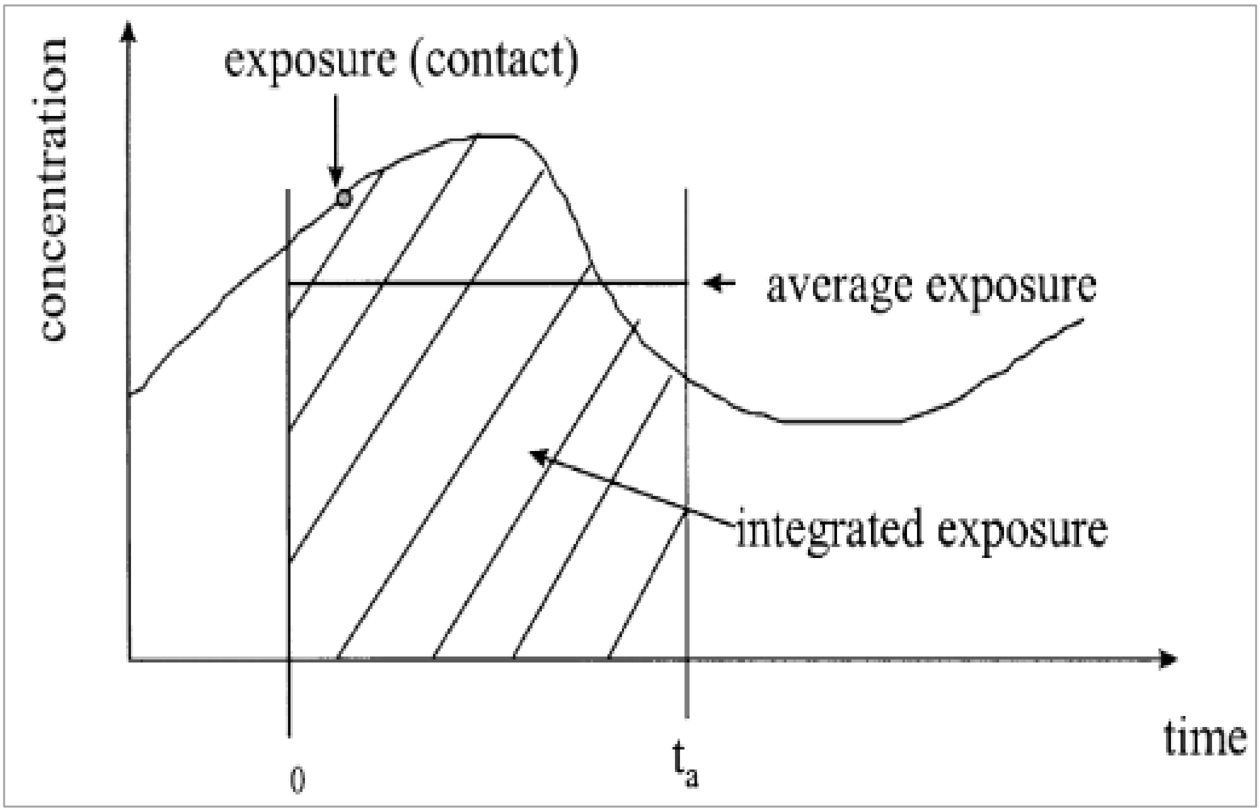

Figure 2-3: A graph showing the fundamental aspects that define the occurrence of exposure, average exposure, and integrated exposure (Monn, 2001).

Equation (1 was established in 1975 and was unofficially referred to as the "*indirect exposure assessment approach"* (Moschandreas & Saksena, 2002). It considers integrated exposure as a function of the proportion of time spent in various locations by an exposed subject and the mean PM concentration measured over time in those microenvironments. Although integrated exposure is calculated from continuous segments of time and their corresponding concentrations, equation (1 does not require these variables to be in sequential time. Instead, this equation allows for spatial and temporal discontinues between recorded concentrations in a predefined location and its associated time intervals (Klepeis, 2006).

Integrated personal exposure to PM is influenced by the time-activity budget; this, in turn, depends on the individual's gender, age, health status, and behaviour of cooking (Pant *et al.*, 2016). As such, previous studies have found that there is a heterogeneous distribution of the burden of personal exposure to PM concentrations amongst variable demographic groups. As was earlier mentioned in Chapter 1, women, children, and elderly people suffer the most from the burden of exposure to PM concentrations. Ezzati and Kammen (2002), for instance, observed that women between 15 to 50 years were exposed to higher indoor air pollutants when compared to men within the same age group. Similarly, Saksena *et al.* (2007) reported that women, particularly housewives and female workers, are mostly exposed to household air pollutants. This is because women cook more often than men, hence when cooking, they are directly exposed to PM emissions emitted by the stove. In some cases, women cook while carrying infants at their backs and as a result, children are also vulnerable to being directly exposed to high PM concentrations, especially during cooking (Bruce *et al.*, 2000).

2.3.1 Dose deposition of particulate matter concentrations in the human body

After PM passes the outer boundaries (e.g. mouth, nostrils, and skin) of an exposed person, the concentration inhaled by such an individual is referred to as the "*dose*" (Hertel *et al.*, 2001; Monn, 2001). PM concentrations cross the external boundaries of the human body through two main processes: 1) intake and 2) uptake (look at Figure 2-4 below). The process of PM intake occurs because of inhalation, and it describes the movement of particles through the mouth and nostrils (EPA, 1992). Because intake considers the particles that enter the body through inhalation, it estimates the deposited PM as a factor of the volume of air inbreathed by an exposed person over time (i.e. either in a minute or day). The amount of PM concentrations that gets into the body through the intake process is thus, influenced by the breathing rate of a subjected individual (EPA, 1992). Contrarily, PM uptake describes the actual amount of concentrations that are absorbed by the body as opposed to the volume of air inhaled into the mouth and nose. In other words, uptake provides estimates of the PM concentrations that are deposited into the body either through the skin or organ absorption. PM concentrations that are absorbed by the body through the process of uptake are influenced by the uptake rate, which in turn, depends on the permeability of the exposed target (e.g. lungs) or tissue (EPA, 1992).

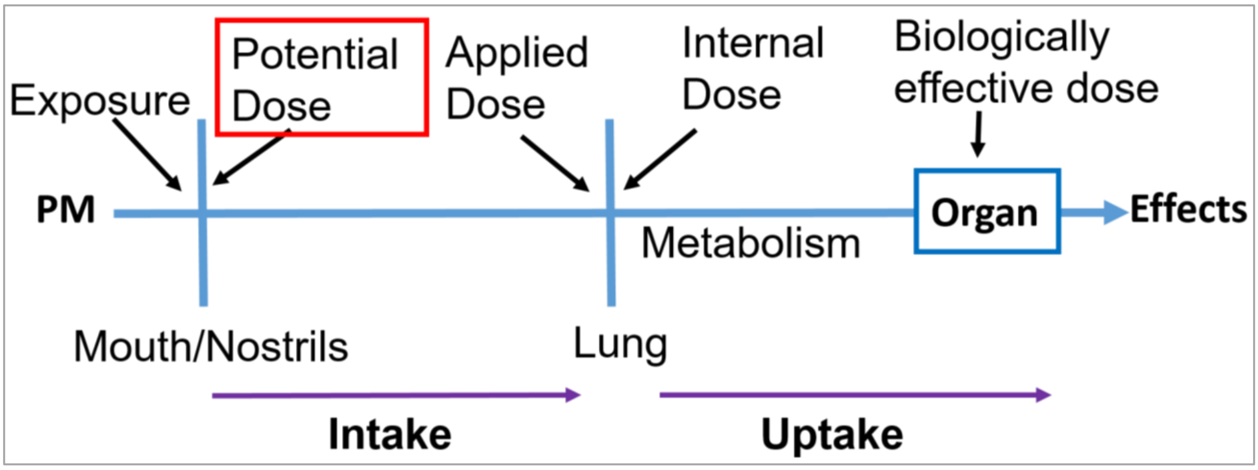

Figure 2-4: A diagram illustrating the inhalation pathway of PM concentrations from a point of human exposure to different deposition points in the human body (EPA, 1992).

PM concentrations are received in the human body in four different doses, namely, "*potential dose*", "*applied dose*", "*internal dose*", and "*biologically effective dose*" (EPA, 1992). PM is ingested in different body parts (see Figure 2-4 below) thus, the complexity of estimating the absorbed concentrations depends on the type of dose that is determined. Calculations of potential dose, for instance, only estimate PM concentrations inhaled into the nostrils and mouth of an exposed individual (i.e. PM intake). In contrast, internal dose specifically determines concentrations ingested within the respiratory tract of a person or people exposed to PM load (i.e. PM uptake) (Chalvatzaki *et al.*, 2018; 2020).

Given the nature of this study, the attention of the research is predominantly focused on potential doses hence, other types of doses will not be further discussed beyond this point. The potential dose inhaled by an exposed individual or group of people is calculated by using equation (2) outlined below:

$$D = \sum_{j=1}^{m} C_j t_j IR_j$$

Equation (2)

Where D is the potential dose, C_j = the quantified magnitude of the concentration of a specified pollutant in a predefined environment, t_j = the duration of time a person or a group of people spent in an air-contaminated environment, and IR_j = the inhalation rate.

Equation (2 above is the modification of equation (1 thus, the potential dose is calculated by multiplying the estimated integrated exposure by the breathing rate of an exposed person (Monn, 2001). A person's breathing rate is influenced by several factors, e.g. age, sex, activity level (e.g. jogging, riding a bicycle, walking, running, and relaxing), and physiology (Aleksandropoulou *et al.*, 2008; Hertel *et al.*, 2001; Mammi-Galani *et al.*, 2017). The potential dose of PM, therefore, can be highly variable amongst different individuals even when such people are simultaneously experiencing a uniform particulate concentration within the same room (Hertel *et al.*, 2001; Viegas *et al.*, 2020). Given that the activity level, behaviour, physiology, and body sizes of adults and children are different, it is found that both these population groups have variable inhalation rates. Children breathe a higher fraction of the volume of air in comparison to that inhaled by adults; this is because their lungs have a wide surface area. A study that was conducted by Almeida *et al.* (2016) found that children inhaled higher doses of PM concentrations than elderly people. Therefore, although elderly people and children are vulnerable to PM concentrations, the latter are at great risk of suffering from health impacts associated with higher PM doses. The U.S. EPA recommends standard average breathing rates for calculating the potential dose for different ages, population groups, and activity levels. Several studies have applied equation (2 when estimating the potential dose of PM concentration inhaled by exposed individuals (Almeida *et al.*, 2016; Faria *et al.*, 2020; Wernecke, 2018).

2.4 PM exposure assessment approaches

Personal exposure to PM concentrations can either be assessed directly or indirectly (Bruce *et al.*, 2000; Chow *et al.*, 2002; Devi *et al.*, 2013; Ezzati & Kammen, 2001; Fang & Lu, 2012; Hertel *et al.*, 2001; Monn, 2001; Wernecke, 2018). Direct assessment is conducted by using measurements obtained from the personal monitors carried by individuals who are being assessed. In contrast, indirect assessment is performed using stationary readings obtained from place-based monitors. Both these exposure assessment methods have different strengths and shortcomings; and require the time-activity patterns of the studied population sample as input data (Bruce *et al.*, 2000). There are a variety of instruments that can be used to collect the time-activity budget of the studied population. These include, for instance, GPS, videography, time-activity diaries, surveys, and questionnaires (Chow *et al.*, 2002).

Estimates of personal exposure derived directly are found to be more accurate since they can provide episodes of exposure patterns a person is experiencing in real-time (Fang & Lu, 2012); however, this detail can be lost, if exposure estimates are integrated over time (Ezzati & Kammen, 2002). Another strength of the direct method is that it accounts for the spatial heterogeneity of the individual's exposure to PM thus, recording the complexity of human exposure (Fang & Lu, 2012). The three main disadvantages of this approach, however, are: firstly, it is expensive, secondly, it has a limited temporal resolution (i.e. 12 hours per day) and finally, many times, the individuals are not faithful to carry the instrument wherever they go (Chow *et al.*, 2002; Fang & Lu, 2012). Despite these limitations, many previous exposure studies have utilized a direct method to assess personal exposure to PM concentration (Fang & Lu, 2012; Liang *et al.*, 2019; Wernecke, 2018).

The use of place-based stationary measurements also has several advantages that are useful for assessing human exposure to PM concentration (Chow *et al.*, 2002; Devi *et al.*, 2013; Fang & Lu, 2012). Firstly, it can assess the personal exposure of a considerable number of persons instead of just a single individual. And secondly, it has a larger spatial coverage, therefore, can provide a homogeneous concentration value for estimating individuals' exposure, for instance, in a neighbourhood-scale (Fang & Lu, 2012). Thirdly, it has a higher temporal resolution thus, can monitor the PM concentration, in a particular space that an individual or a group of people is exposed to, throughout the day (Chow *et al.*, 2002). However, the major limitation of this method is that it ignores the spatial variation of air pollutants thus, assuming that the personal exposure of different individuals in a predefined space is uniform (Ezzati & Kammen, 2002).

2.5 Health impacts of human exposure to particulate concentrations

Once PM enters the body, the toxicity and the severity of its health impacts depend on the size fraction of the particles that were inhaled (EPA, 2020). Similarly, the size of the inhaled particles determines the destination of the PM in the human respiratory system. Given that $PM_{2.5}$ is so small, it can penetrate deep within the lungs thus, it is linked to several serious health impacts than larger particles (Jimoda, 2012). In contrast, when PM_{10} is inhaled, particularly through the nasal passage, its penetration in the respiratory system does not get beyond the bronchi. Instead, it can either be deposited into the throat or bronchi per se and later be removed from the system when a person either coughs or sneezes (Kim *et al.*, 2015). PM_{10} does not penetrate deep into the respiratory tract since mucus and cilia in the human nose effectively act as the defence mechanisms hence it does not reach the lungs (look at Figure 2-5 below) (Jimoda, 2012).

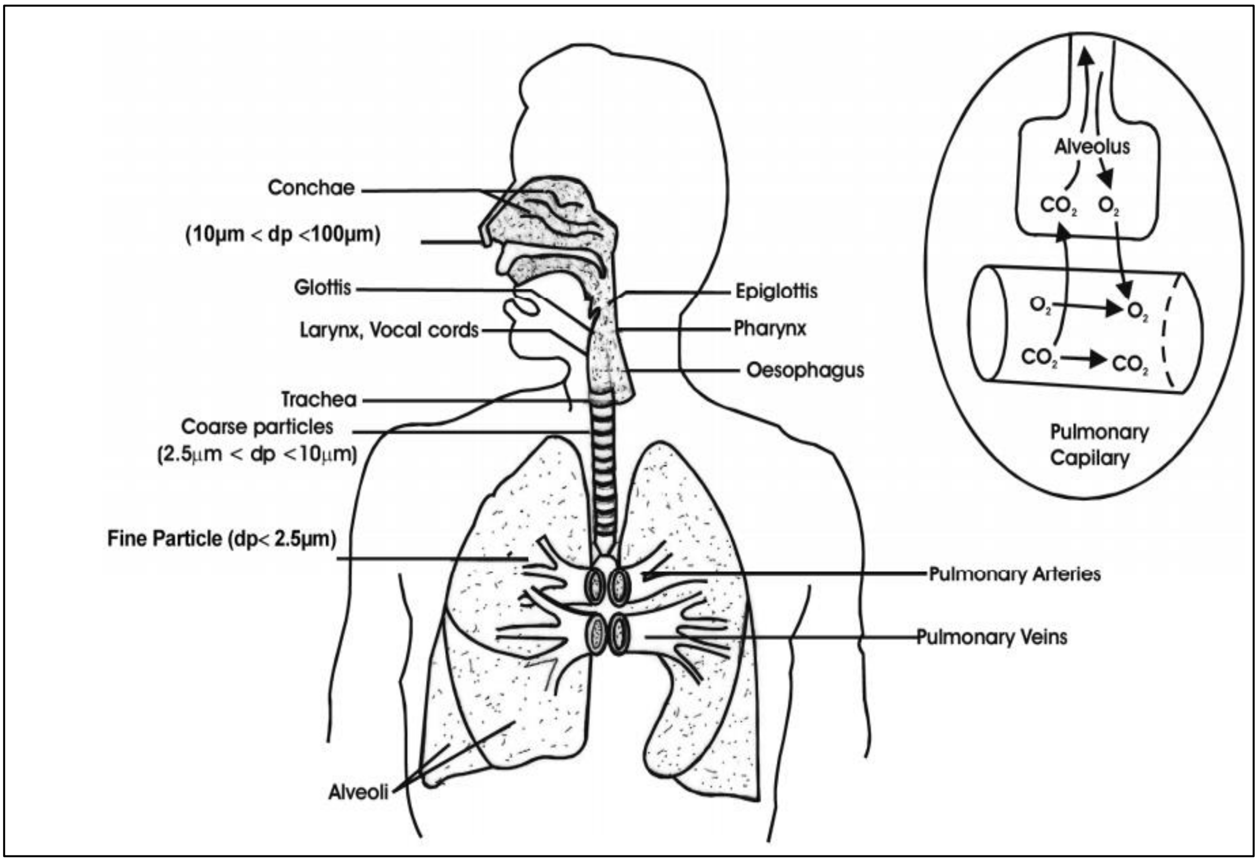

Figure 2-5: A diagram showing the human respiratory system and the deposition points of different fraction sizes of PM in the body (Jimoda, 2012).

Inhalation of PM concentrations is associated with several health impacts. To mention a few, these include premature deaths, cardiovascular diseases, reduced functionality of the lungs, and an increment in hospital admission (Kim *et al.*, 2015; Language, 2020). Furthermore, substantial particulate concentrations can cause health issues, which include infant mortalities, early childbirth, and low body weight infants after birth. Guaita *et al.* (2011) found that human exposure to PM, especially $PM_{2.5}$, can also result in less serious symptoms like coughing, runny nose, pains in the chest, and breathing difficulties. High PM loads pose greater health risks to susceptible demographic groups, such as people with pre-existing cardiac problems, children, and elderly people. For example, it has been documented that inhalation of particulate pollution by children can affect the natural growth rate of their lungs thus, resulting in malfunctioning lungs (Brauer *et al.*, 2012; Kim *et al.*, 2015). Moreover, it was reported that it is more likely for a child to suffer from either asthma or respiratory issues if exposed to fine particulate concentrations that exceed 60 $\mu g.m^{-3}$ throughout the day (Kim *et al.*, 2015).

2.6 Management of PM concentrations in South Africa

The National Environmental Management: Air Quality Act no. 39 of 2004 (NEMAQA) is the current legislation that regulates and manage air quality in South Africa. It consists of numerous air pollution regulative tools aimed at enhancing the protection of human health from industrial emissions as well as local sources. These include, for instance, 1) National Ambient Air Quality Standards (NAAQS); 2) Atmospheric Emission Licenses (AEML); 3) Air Quality Management Plans (AMP); 4) the Declaration of Priority Areas; and Minimum Emission Standards (MES) (South Africa, 2014). The latter instrument was promulgated in 2015 to reduce emissions of the criteria pollutants that are emitted specifically by the point sources. As a condition of MES, industries were required to use the abatement technologies such as Flue Gas Desulphurization (FGD) in 2015 and 2020. However, industries criticized the MES for being expensive to fully comply with; and for also disregarding the ambient air quality of most polluted communities, especially those surrounding point sources (Pretorius, 2015).

In recognition of the impacts of domestic fuel combustion on air quality in low income communities, the Department of Environmental Affairs introduced Air Quality Offsets in 2016 (South Africa, 2016). According to the guidelines of the Air Quality Offsets that were published in terms of NEMAQA offsets are "*intervention, or interventions, specifically implemented to counterbalance the adverse and residual environmental impact of atmospheric emissions in order to deliver a net ambient air benefit within, but not limited to, the affected airshed where ambient air quality standards are being or have the potential to be exceeded and whereby opportunities and need for offsetting exist*" (South Africa, 2016). Air Quality Offsets are the first instrument in

South Africa that directly addresses the issue of residential use of coal, through legal enforcement (Langerman *et al.*, 2018).

It is the responsibility of the main polluting industries, especially those in the Highveld Priority Area (HPA) and Vaal Triangle Airshed Priority Area (VTAPA), to implement offsets. Offsets are often executed by either swopping ineffective coal stoves with more efficient stoves (i.e. either improved coal stoves or liquid petroleum gas stoves) or through household insulations (Adesina *et al.*, 2019). The implementation of offset projects is a prerequisite set in the Atmospheric Emission Licenses of the specific polluting facilities which do not fully meet the MES. Thus, in a practical sense, industries execute offsets in communities where PM concentrations are often beyond NAAQS, and in turn, they receive leniency for not being completely in compliance with the MES (Langerman *et al.*, 2018). Polluting industries can only be granted leniency based on the following conditions, 1) provided they prove that there is no abatement technology of a particular activity specified in an AEL, worldwide and 2) if they will be closed after ten years, starting from when the postponement was made.

This chapter provided a theoretical background of what is already known about particulate matter pollution. It discussed in detail the physical properties that define particulate concentrations and the typical standards that are used to classify PM. It further outlined the typical local sources that are commonly identified in the literature as emitting particulate emissions within typical low income settings. The chapter went on to introduce the reader to the concept of "exposure" as well as "dose" and discussed the typical methods used to assess the former. Moreover, this chapter unpacked the health impacts associated with PM and lastly, it discussed relevant legislative control of PM in South Africa. In the subsequent chapter below, the methods that were followed to carry out the present research will be looked upon and discussed.

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CHAPTER 3: RESEARCH METHODS

This chapter respectively discusses the data collection procedures that were undertaken in this research as well as the methods used to analyze such data. In particular, the chapter first explains how this study was designed, the area of the study, the spatial location where measurements were taken and the population demographics of Lebohang. Secondly, it further outlines all the instrumentations that were used to collect the primary data and how secondary data was obtained.

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A descriptive research design was adapted to profile the exposure of different population groups in Lebohang low income community. According to Bloomfield and Fisher (2019), a descriptive research design is one of the four main types of quantitative research approach, which is commonly used to study a specific phenomenon within a certain sample population. This study design answers the research question by analysing the numerical data of different variables, as well as using statistical techniques and a mathematical equation model. A stochastic model and as well as statistical techniques were thus utilized to achieve the research aim of the present study, which was earlier outlined in section 1.2 above.

Ideally, the exposure assessment study relies on the concentration data that was continuously recorded within different spaces and time-activity data that were recorded concurrently with the pollutants of concern (Sidhu *et al*., 2017). Since the current study is part of the baseline monitoring conducted in 2018 by the Sasol Offset research project, the sampling design of the indoor PM, which is discussed in section 3.2.2 below, was influenced by this project. Because the Sasol Offset project was aimed at monitoring the baseline indoor concentrations in Lebohang, rather than assessing exposure, adjustments were made to meet the aim of the present study. These included the use of the corresponding outdoor PM measurements, existing time-activity data published by earlier studies, as well as the census data collected at the municipal ward levels (refer to Figure 3-1 below). Due to the lack of knowledge about the true values of the average time spent by various groups at the different places of concern in Lebohang, the time parameter was, therefore, uncertain. Because of this, all the variable inputs used to estimate exposure in Lebohang were thought of as uncertain, hence the Monte Carlo technique was applied in the study.

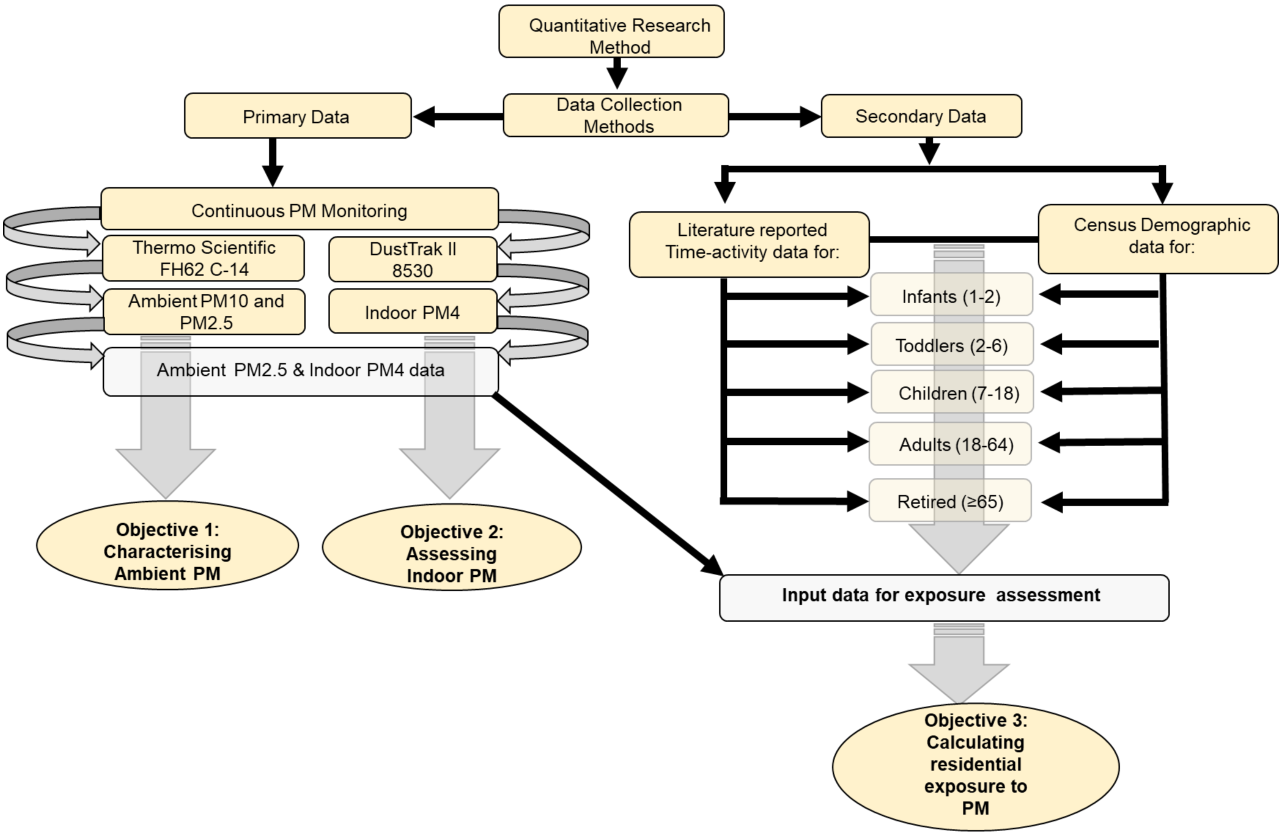

Figure 3-1: A flowchart summarising the approach that was taken to investigate ambient and household exposure to PM in Lebohang.

3.1 Study location

Lebohang (26°22'00"S 28°54'00" E) is a dense, low income residential area situated downwind from Leslie and on the south of the intersecting point of R50 and R29 (Figure 3-2). It is located within the Mpumalanga Highveld Priority Area (HPA), a region well known for substantially poor air quality. In 2007, Martinus van Schalkwyk, who was then the Minister of Environmental Affairs, declared HAP as a priority area in terms of section 18(1) NEMAQA (Act No.39 of 2004) (Department of Environmental Affairs and Tourism (DEAT), 2007). HPA has a spatial coverage of about 31 000 km^2, which does not only extend over the parts of Mpumalanga but also those of the Gauteng province (see Figure 3-2). Moreover, it is positioned within the Ekurhuleni metropolitan municipality and consists of three different district municipalities, namely: Nkangala, Sedibeng, and Gert Sibande municipality. These districts administer a total of nine local municipalities, i.e. Pixley ka Seme, Msukaligwa, Lekwa, Dipaliseng, Govan Mbeki, Steve Tshwete, Emalahleni, Delmas, and Lesedi local government (DEAT, 2007).

HPA has a flat topography and is also located within the inland plateau which is characterised by having an elevation that exceeds 1 400m but, is lower than 1 800m (Qhekwana, 2019). Places that fall within the highest and lowest elevation points of HPA are located within the southeastern and northwestern parts of the region, respectively. HPA is spatially positioned within the Grassland biome and consists of considerable places where anthropogenic activities take place (Lydia, 2010). Anthropogenic processes practiced in the region mostly include but are not limited to, commercial and subsistence farming activities such as coal mining, power generation, and crop production. The coal extracted from the mines located within this region is mainly used for two purposes, namely, to generate electricity and to provide residents with an alternative energy source (Lydia, 2010).

3.1.1 Demographic description and spatial surroundings of Lebohang

Lebohang low income community falls, particularly, within the borders of the Govan Mbeki local municipality and has an estimated population of 31 500 people with a dependency ratio of 53.6 (Statistics South Africa (Stats SA), 2011). It is estimated that 65.1% of the populace in the Lebohang residential area are between the age of 24-64; 31% are within the range of 0-14; and 3.9% are elderly people from and beyond 65 years (Stats SA, 2011). Therefore, although this community consists of different population subgroups, its population is predominantly made up of working-age people (i.e. residents between 18-64 years of age).

The settlement has over 8 900 households and 76.8% of these homes are formal; meanwhile, the remaining portion is classified as informal dwellings (Stats SA, 2011). About 40.2% of these homes in this community, irrespective of their structural classes, are headed by females thus, in such houses, women are primarily responsible for running house affairs. On average, each household in the township is estimated to have at least 3.5 household members however, the typical gender distribution of occupants for each house is unknown. Poverty is a pressing socioeconomic issue faced by a considerable portion of homes in Lebohang as more than 17% of households in this community live without any source of income (Stats SA, 2011). Over 96.5% of houses in the area have access to electricity; despite this, around 15.2% and 30.9% of households use coal for cooking and space heating, respectively, as it is cheap (Stats SA, 2011).

Despite domestic coal burning, Lebohang low income settlement is also surrounded by other PM sources, which are located within 50 km of its spatial radius. This community is located near the Kriel power station (i.e. 40 km away) and the Matla open-pit coal mine (32 km away), which are placed north and northwest of the community respectively. Furthermore, it is also positioned adjacent to the Sasol petrochemical industry, which is in Secunda, southwest, and approximately 40 km away from the township.

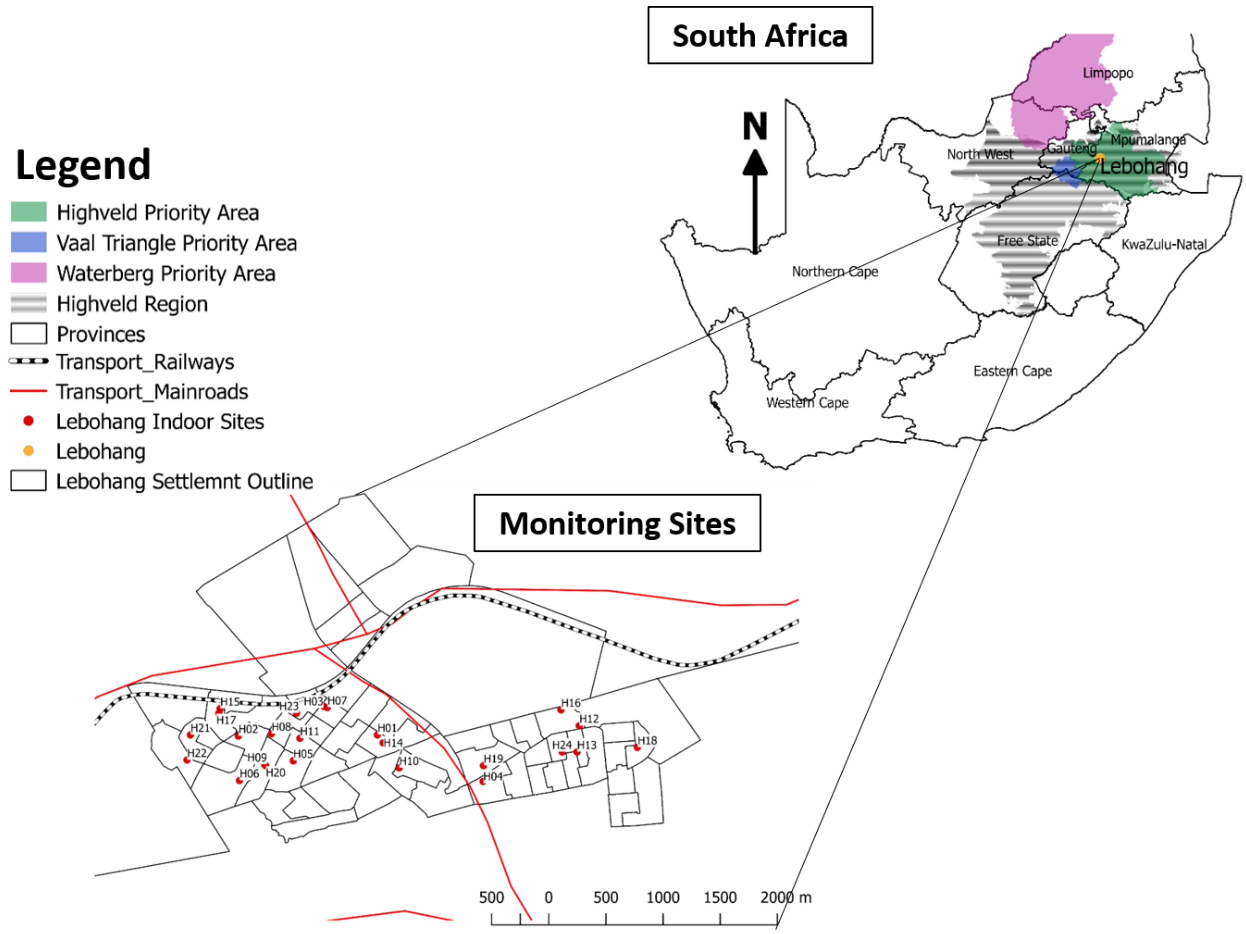

Figure 3-2: Lebohang low income settlement within the Mpumalanga Highveld in South Africa and the zoomed-in view of the spatial distribution of monitoring sites within the community.

3.1.2 Meteorological conditions of Lebohang

Although Lebohang has an ambient monitoring station, which will later be discussed in section 3.2.1 below, the meteorological parameters provided by such a station were not sufficient for describing the typical weather conditions of the community. As such, the data from the Witbank long-term monitoring station, which covered a period of 3 years (2016 to 2018), was used since this area is 30 km away from the studied area. The data was requested from the South African Weather Services (SAWS). Given that Lebohang and Witbank are positioned in the Mpumalanga Highveld region, the weather conditions experienced in both these places were deemed to be insignificantly variant. The reason is the typical weather conditions prevailing over both the aforementioned areas are influenced by the regional weather patterns experienced over the Mpumalanga Highveld.

Figure 3-3 below depicts the meteorological data for the period 2016 to 2018 collected from the Witbank permanent monitoring station. Based on Figure 3-3, it is safe to assume that Lebohang typically experiences warm and cold ambient temperatures during summer (December, January, and February) and winter seasons (June, July, and August), respectively. The range of the daily average temperature commonly experienced during the summer period changes from a minimum of 22 °C in January to a maximum of 24 °C in December. On the other hand, winter temperatures often vary from the lowest daily mean temperatures of 11 °C in June to the highest average of 14 °C. Throughout the spring period (September, October, and November) Lebohang usually records an increment in the daily mean temperatures that increases from 17 °C in September to 21 °C in November.

(a) **(b)**

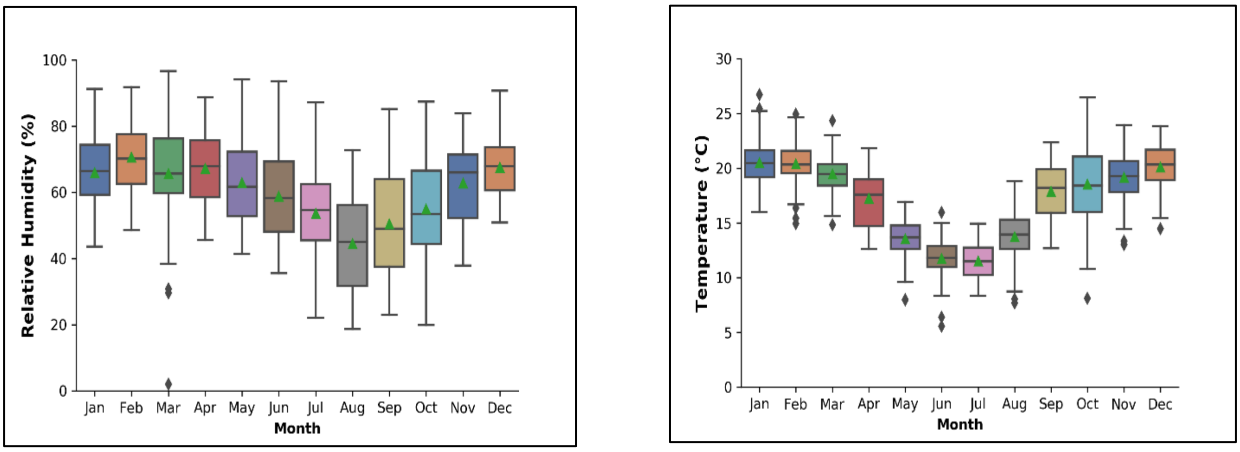

Figure 3-3: Three-year period (2016-2018) of meteorological data depicting the typical relative humidity (a) and ambient daily average temperature (b) experienced in Lebohang. The box and whisker plots show 25th, 50th and 99th percentiles; mean (green triangles inside the box); and outliers (black diamonds below and beyond the whiskers).

The lowest relative humidity is typically experienced during the winter season (June, July, and August) and ranges from a minimum of 42% in August to a maximum of 60% in June. In contrast, the highest relative humidity ranging from approximately 68% to 71% is typically experienced during the summer period (December, January, and February) (Figure 3-3 (a)).

3.2 Data collection techniques and analytical procedures

3.2.1 Characterising ambient PM concentrations

To characterise outdoor particulate concentrations in Lebohang, $PM_{2.5}$ and PM_{10} were selected as the pollutants of interest in this study since both are regulated by NAAQS. The PM data for both $PM_{2.5}$ and PM_{10} concentrations used in the present research was monitored for 11 months (October 2017 to August 2018). The date range of October 2017 to August 2018 was chosen because it was the only period that had good and available ambient PM measurements recorded, specifically in Lebohang residential area (refer to section 3.2.1.2 below). Furthermore, it also consisted of the data for different seasons namely, spring, summer, autumn, and winter thus, providing an opportunity to explore how PM varies in time in the community.

The ambient PM data was retrieved as the hourly measurements from the South African Air Quality Information System (SAAQIS). These measurements were continuously recorded at the ambient monitoring station, located in the Chief Ampie Maysa secondary school (26°22'53"S 28°55'06"E) in Lebohang low income settlement (Figure 3-4. a). This station is one of the nine (9) monitors that are operated by the Sasol petrochemical industry in the Mpumalanga Highveld region, and it reports the recorded data directly to SAAQIS (Venter & Lourens, 2020).

(a) **(b)**

Figure 3-4: The ambient monitoring station (a) and the Thermo Scientific FH62C14 (b) used to monitor PM concentrations in Lebohang low income settlement.

The Sasol ambient station (from here onwards it will be referred to as SAS) is the only permanent monitor that continuously measures PM concentrations in Lebohang. However, it is not exclusively limited to measuring $PM_{2.5}$ and PM_{10} in the area, and it also records other pollutants such as SO_2, O_3, NO_X, NO and H_2S. It further monitors the meteorological parameters. However, due to poor maintenance and constant systematic failures, the only available meteorological data at SAAQIS reported by SAS is for ambient temperature. Because Chief Ampie Maysa secondary school is positioned at the centre of the community, this site was chosen as it is somewhat representative of the air people breathe.

The site of SAS in Lebohang is immediately surrounded by grass, and a few meters away from the station, the underlying surface is open land. Approximately 10m North, 15m Southwest and 13 m West of the ambient monitoring site there are building infrastructures. Moreso, about 30m away, in all 4 cardinal directions (North, South, West, East), the ambient monitor is enclosed by the tarred residential roads.

3.2.1.1 Instrumentation

Ambient concentrations of both PM_{10} and $PM_{2.5}$ were continuously monitored in Lebohang low income settlement using the Thermo Scientific FH62C14 radiometric particulate monitor (Figure 3-4. b). Amongst other monitors, the U.S. EPA has established the FH62C14 as the equivalent method for the continuous measuring of PM_{10} concentration (Intra *et al*., 2016). This instrument provides real-time concentration measurements of particulate loads which are continuously recorded by the sensors that are operated based on the beta attenuation principle (Thermo Scientific, 2010). Table 3-1 provides a detailed summary of the technique of aerosol measuring, the lower and the higher detection range, the rate of flow and the aerosol range size sampled by the FH62C14 monitor.

Table 3-1: Principles of Thermo Scientific FH62C14 monitor operation (Thermo Scientific, 2010).

Description of the parameter	Specification of FH62C14 operation
Way of sampling	Beta attenuation
The size range of particles sampled	<10 µm
Range of the detection limits	0 – 1000 µg.m^{-3} however, if auto ranging then = 0 - 0 – 10000 µg.m^{-3}
Rate of airflow	16.67L/min
Resolution	4 µg.m^{-3}

The FH62C14 monitor measures PM concentrations by depositing the sampled air onto the filter tape using a vacuum pump. The sampled particles are sucked into the FH62C14 monitor and accumulate on the filter tape, which advances automatically just before it reaches the threshold value (Thermo Scientific, 2010). This aerosol collection into the FH62C14 monitor is operated by the principle of beta attenuation, which provides measurements by detecting the mass of the particles deposited on the filter tape. The pathway through which the sampled air enters the Thermo Scientific FH62C14 monitor is illustrated in Figure 3-5.

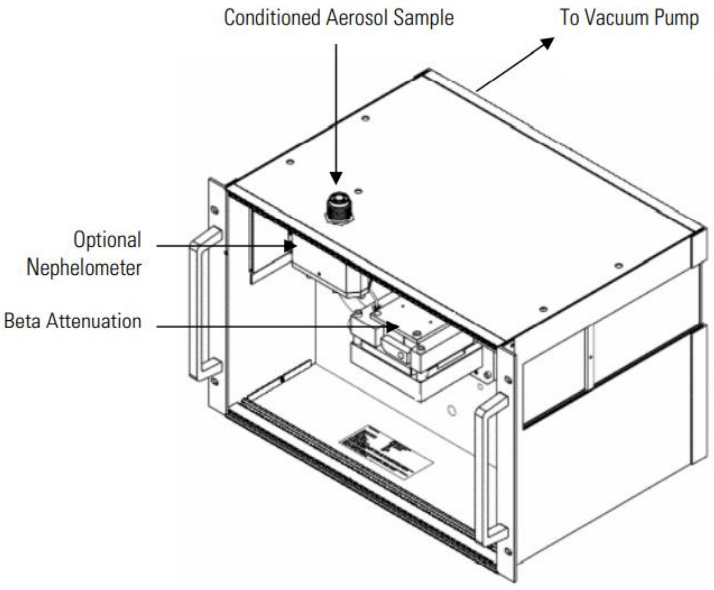

Figure 3-5: Sampling path of the FH62C14 monitor (Thermo Scientific, 2010).

3.2.1.2 Quality control and analysis of ambient PM data

The quality control procedure was conducted before analysing the ambient $PM_{2.5}$ and PM_{10} time-series data that was downloaded from SAAQIS for Lebohang low income settlement. This was to ensure that only high-quality data, which was corrected for systematic and random errors, was considered when characterising ambient PM concentrations in the study area. Several correction steps were undertaken before analysing hourly ambient particulate data. Firstly, constant values measured consecutively for more than two hours for both $PM_{2.5}$ and PM_{10} were omitted. The reason is, an instrument cannot record the same measurements constantly in a row for different time intervals due to the prevalence of noise, except if there was a systematic error (Arku *et al.*, 2008). Secondly, only PM concentration values that were within the FH62C14 monitor sampling range (i.e. 0 - 10000 $\mu g.m^{-3}$) were used for the analysis; therefore, values that were negative or below zero were removed from the dataset. This was to ensure that only valid measurements were considered when calculating the daily and monthly averages of both $PM_{2.5}$ and PM_{10} concentrations in Lebohang. Finally, to characterise PM concentration behaviour, only days that had at least 70% of the hourly concentration points were used in the analysis.

The South African National Accreditation System (SANAS) stipulates that the data monitored over three months must be at least 90% complete (SANAS, 2012). This SANAS requirement is often considered to be vague since it does not specify whether should the data meet the set threshold before or after the corrections. Conversely, Cairncross (2016) states that it appears as if the SANAS standard must be met before quality control techniques are applied. For this study, the data recovered was calculated after the data was cleaned as the percentage of the valid days for the entire sampling period and each month (see table 3-2 below). The recovered data was used to produce graphical visuals of ambient PM pollution in Lebohang, as well as to describe statistical distributions of the collected measurements.

Table 3-2: The percentage (%) of the data recovered for the entire sampling period (from October 2017 to August 2018) and each respective month.

Pollutant	Full	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	July	Aug
$PM_{2.5}$	90	98	100	90	93	100	100	77	87	100	100	100
PM_{10}	90	98	100	97	93	100	100	80	87	100	100	100

3.2.2 Monitoring of households for indoor PM concentration

To assess household PM concentrations in Lebohang, PM_4 was selected as a pollutant of interest in this study since it is inhalable, often produced during domestic coal burning and not regulated by NAAQS. Indoor PM_4 was sampled in twenty-two (22) different households in Lebohang low income settlement (see Table 3-3). Out of these 22 homes, twelve (12) houses were monitored during autumn (from 05 March to 02 April 2018) and another 12 dwellings were sampled during winter (from 18 June to 30 July 2018). Only 5 houses (i.e. H02, H03, H04, H09 and H10) out of the 22 homes selected to participate in this study were successfully monitored for both seasons meanwhile the remaining 17 households were only sampled once (i.e. either in autumn or winter). The autumn sampling duration was deemed as being representative of the warm period, and on the other hand, the winter monitoring period represented the cold season sampling.

(a) **(b)**

Figure 3-6: Examples of structural designs of formal (RDP) (a) and informal households (b) that were monitored in Lebohang low income settlement.

The criteria for selecting the households that participated in this study was set by the Sasol baseline monitoring project, which randomly chose different homes for the sampling of PM_4 load in Lebohang. As a result, the households where measurements were carried out in this community had different structural designs (i.e. some were formal and others were informal, see Figure 3-6 (a) and (b)). Furthermore, these houses also used various energy carriers, for instance, some predominantly relied on electricity and LPG stoves (non-coal-reliant) meanwhile others were using coal as their main energy source (coal-reliant). It was, therefore, anticipated that non-coal-reliant households would have better indoor air quality than those that were reliant on coal.

Due to the difference among the sampled households in terms of both structural designs and primary energy carrier usage, the 22 monitored homes in Lebohang were classified into four main groups (Table 3-3):

(i) Coal formal households (CFH)
(ii) Non-coal formal households (NFH)
(iii) Coal informal households (CIH)
(iv) Non-coal informal households (NIH).

The structural characteristics, design and building materials of the formal households were different from those of informal dwellings (see Figure 3-6). The formal homes were government-built houses that were constructed because of the Reconstruction and Development Programme (RDP), which was implemented in 1994. Each of these households had the following characteristics: 1) a total of four (4) rooms without any shack being used to extend the house; 2) a plot of land that was below 40 m^2; 3) was built using bricks as well as the cement; 4) a roof that was made up of corrugated iron (Figure 3-6 (a)), and 5) a total of four (4) windows and two (2) doors (see Figure 3-7 (a)). On the other hand, informal dwellings were two-roomed structures that were erected by formal house owners in their backyards, specifically for rental and commercial purposes. The typical structural characteristics of each of these households included: 1) land area that was below 20 m^2; 2) walls and rooftop that was constructed out of corrugated iron (Figure 3-6 (b)); 3) two windows (i.e. one in the bedroom and one in the kitchen); and one main door (see Figure 3-7 (b)).

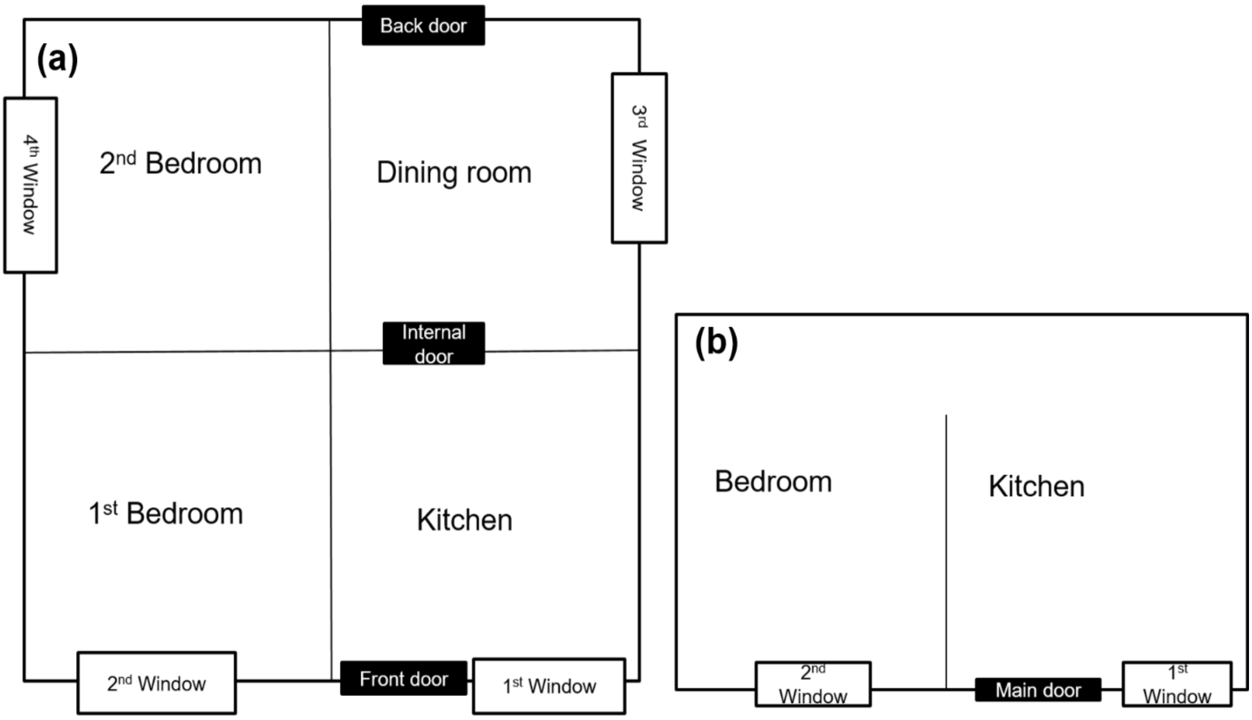

Figure 3-7: Typical structural layout of the formal (a) and informal (b) households monitored in Lebohang.

Table 3-3: A summary of the households that were monitored, including their structural characteristics, primary energy fuel, other surrounding PM sources and the number of days that were measured for PM_4 in each home during both the summer and winter sampling campaign.

Autumn Campaign (05 March to 02 April 2018)										
House id	**Housing Structure**	**Class**	**Primary fuel used**		**Other sources**	**Participatory status**		**Sampling duration**		**Sampled Days**
			Coal	**Electricity**		**Successfully sampled**	**Withdrew**	**Deployment Date**	**Removal Date**	
H01	Formal	CFH	✔		UR	✔		05/03/18	12/03/18	8
H02	Formal	NFH		✔	UR	✔		05/03/18	12/03/18	8
H03	Informal	CIH	✔		UR	✔		05/03/18	12/03/18	8
H04	Informal	NIH		✔	UR	✔		05/03/18	12/03/18	8
H05	Formal	CFH	✔		UR	✔		12/03/18	19/03/18	8
H06	Formal	NFH		✔	UR	✔		12/03/18	19/03/18	8
H07	Informal	CIH	✔		UR & WB	✔		12/03/18	19/03/18	8
H08	Informal	NIH		✔	UR	✔		12/03/18	19/03/18	8
H09	Formal	CFH	✔		UR	✔		19/03/18	02/04/18	15
H10	Formal	NFH		✔	UR	✔		19/03/18	02/04/18	15
H11	Informal	CIH	✔		UR	✔		19/03/18	26/03/18	8
H12	Informal	NIH		✔	UR	✔		19/03/18	26/03/18	8
Winter Campaign (18 June to 30 July 2018)										
H01	Formal	CFH	✔		UR		✔	-	-	-
H02	Formal	NFH		✔	UR	✔		18/06/18	25/06/18	8
H03	Informal	CIH	✔		UR	✔		18/06/18	25/06/18	8
H04	Informal	NIH		✔	UR	✔		18/06/18	25/06/18	8
H05	Formal	CFH	✔		UR		✔	-	-	-
H06	Formal	NFH		✔	UR		✔	-	-	-
H07	Informal	CIH	✔		UR & WB		✔	-	-	-
H08	Informal	NIH		✔	UR		✔	-	-	-
H09	Formal	CFH	✔		UR	✔		09/07/18	23/07/18	15

Continuation of table 3-3

H10	Formal	NFH		✓	UR	✓		09/07/18	23/07/18	15
H11	Informal	CIH	✓		UR		✓	-	-	-
H12	Informal	NIH		✓	UR		✓	-	-	-
H13	Formal	CFH	✓		UR & CS	✓		02/07/18	09/07/18	8
H14	Formal	NFH		✓	WS & UR	✓		02/07/18	09/07/18	8
H15	Informal	CIH	✓		UR	✓		02/07/18	09/07/18	8
H16	Informal	NIH		✓	UR	✓		02/07/18	09/07/18	8
H17	Informal	CIH	✓		UR & CS	✓		16/07/18	23/07/18	8
H18	Informal	NIH		✓	WB & CS	✓		16/07/18	23/07/18	8
H19	Formal	CFH	✓		UR	✓		23/07/18	30/07/18	8
H20	Formal	NFH		✓	UR	✓		23/07/18	30/07/18	8
H21	Informal	CIH	✓		UR	✓		23/07/18	30/07/18	8
H22	Informal	NIH		✓	UR	✓		23/07/18	30/07/18	8

CFH - Coal formal households
NFH – Non-coal formal households
CIH – Coal informal households
NIH – Non-coal informal households
UR – Unpaved Roads
WB – Waste Burning
CS – Cigarette Smoking
– Denote no data

3.2.2.1 Instrumentation for continuous monitoring of household PM

Indoor PM_4 concentrations in Lebohang were continuously sampled in 22 different households using DustTrak ll Model 8530 monitors (TSI, 2012). In each of these homes, PM_4 was sampled in the kitchen space using a single DustTrak instrument that was placed at least 1.6 meters away from the ground to best measure the prevailing particles in the breathing zone of the occupants (see Figure 3-8). DustTraks were suitable for indoor PM_4 monitoring in this study based on the following reasons: 1) their minimal consumption of electricity; 2) their portable size, and 3) the tolerable noise they produce during the sampling process. These criteria were important to the convenience of the occupants in monitored households and for also minimizing the risks of houses withdrawing from the study during the sampling process.

Figure 3-8: DustTrak ll Model 8530 monitors placed in the kitchen, approximately 1.6 metres away from the ground surface in one of the houses sampled in Lebohang low income settlement.

DustTrak ll Model 8530 monitors are photometric instruments and measure particle mass concentrations from the sampled air using a laser that operates based on a light scattering technique (TSI, 2012). This laser is positioned at a 90° angle and has a wavelength of 780nm. The particle size that can be sampled by these instruments varies from 0.1 to 10 µm and the flow rate of the air that is pumped in the DustTrak monitor varies from 0.7 and 2.4 litres per minute ($L.min^{-1}$). The specific size of the particles that are monitored by the DustTrak is determined by the type of size-selective impactors (which can either be PM_1, $PM_{2.5}$, PM_4 or PM_{10}) used when sampling (Language, 2020). The limit of the particle concentrations that can be detected by DustTrak ll Model 8530 monitors ranges from a minimum of 0.0001 $mg.m^{-3}$ to a maximum of 400 $mg.m^{-3}$.

In this study, PM_4 concentrations were exclusively sampled by attaching a 10 mm Dorr-Oliver cyclone to the inlet on each DustTrak to Cut off 50% of particles that were beyond 4.0 µm size fraction (see Figure 3-9 (a)). This is the standard method that is accepted worldwide for measuring PM_4 (TSI, 2012). The 10 mm Dorr-Oliver cyclone is specifically designed to exclusively sample PM_4 by separating particles below 4.0 µm from those with a size fraction that is equal to or beyond 10 mm (Sensidyne, 2003). At the grit pot (see Figure 3-9 (b)), the cyclone traps particles that are larger than the cut-off size and allows those that are below 4.0 µm to pass through the body and cap into the instrument.

(a) **(b)**

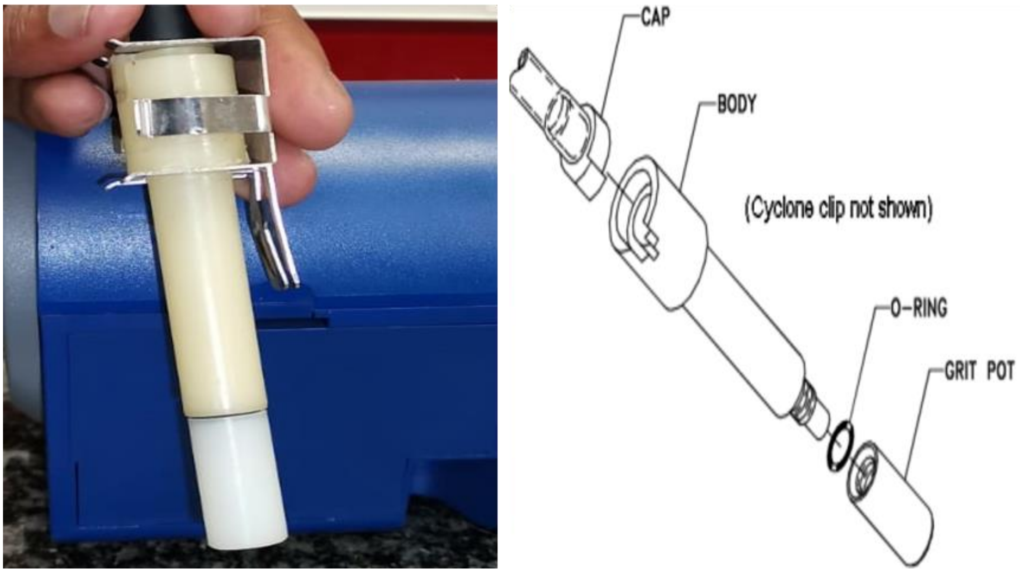

Figure 3-9: The 10 mm Dorr-Oliver cyclone (a) that was attached to the DustTrak inlet and the overview of how it looks when it is disconnected (b) (Sensidyne, 2003).

As recommended in the TSI Incorporated, to successfully obtain the cut-point size of 4.0 µm, the flow rate of each DustTrak was set to pull the sampled air into the instrument at 1.7 $L.min^{-1}$ (TSI, 2012). This is the standard flow rate that draws the sampled air in the monitor in a spiral motion, which in turn, separates PM_4 from other aerosols that have a fraction size that is larger than 4.0 µm (TSI, 2019). If the DustTrack was operated at any other flow rate except for 1.7 $L.min^{-1}$, the sampling of respirable particles would have not been possible. Hence, setting the flow rate of the instruments, specifically to 1.7 $L.min^{-1}$, made it possible for PM_4, which was a particular pollutant of interest in this study, to be monitored. Furthermore, all the DustTrak monitors were set to operate at 5-minute averaging time intervals to capture the sudden short-term variance of PM_4 concentrations in the monitored houses.

3.2.2.2 Sampling design for indoor monitoring

The aim of the household monitoring design of this study was to sample an equal number of households in the autumn and winter period to profile the spatial and temporal variation of PM_4 in Lebohang. However, due to issues such as household withdrawals, technical issues, and a limited number of DustTraks, it was not possible to meet this aim. The indoor PM_4 concentrations in Lebohang were monitored using only four DustTraks. These instruments were rotated amongst the 22 selected homes, thus making it impossible for all houses to be sampled simultaneously during each respective season. As such, for convenience, the autumn and winter sampling periods were subdivided into 4 monitoring campaigns, namely campaigns 1, 2, 3 and 4 (see table 3-4 below).

Table 3-4: Summary of the duration of the indoor PM_4 sampling campaigns that were conducted during the autumn (from 05 March to 02 April 2018) and winter monitoring period (from 18 June to 30 July 2018).

	Autumn monitoring period			**Winter monitoring period**		
Sampling reference	**House Id**	**Class**	**Sampling duration**	**House Id**	**Class**	**Sampling duration**
Campaign-1	H01	CFH	05/03/18-12/03/18	H01	CFH	Withdrew
	H02	NFH		H02	NFH	18/06/18-25/06/18
	H03	CIH		H03	CIH	
	H04	NIH		H04	NIH	
Campaign-2	H05	CFH	12/03/18-19/03/18	H13	CFH	02/07/18-09/07/18
	H06	NFH		H14	NFH	
	H07	CIH		H15	CIH	
	H08	NIH		H16	NIH	
Campaign-3	H09	CFH	19/03/18-02/04/18	H09	CFH	09/07/18-23/07/18
	H10	NFH		H10	NFH	
	H11	CIH	19/03/18-26/03/18	H17	CIH	16/07/18-23/07/18
	H12	NIH		H18	NIH	
Campaign-4	-	-	-	H19	CFH	23/07/18-30/07/18
	-	-	-	H20	NFH	
	-	-		H21	CIH	
	-	-	-	H22	NIH	

But due to systematic failures of all 4 DustTraks, the autumn sampling monitoring period only consisted of 3 campaigns rather than 4 (also refer to table 3-4). During each of the subdivided measuring campaigns, regardless of the season, four classes of households, namely CFH, NFH, CIH and NIH were simultaneously monitored for at least a minimum of 8 or a maximum of 15 days. Out of 22 households that participated in this study, only H02, H03, H04, H09 and H10, were successfully monitored during both autumn and winter. However, this was not the case for H01, H05, H06, H07, H08, H11 and H12 as they withdrew before being sampled for the winter period and were thus replaced by H13, H14, H15, H16, H17 and H18.

3.2.2.3 Data quality assurance and control

To maintain the collection of rich indoor PM_4 data in Lebohang, quality assurance procedures were conducted before and during the monitoring period of each sampled household. Such procedures were conducted during the site visits and were also accompanied by the collection of the data. Several tasks that are recommended and outlined by the manufacturer were conducted to ensure the optical performance of the DustTraks during the sampling duration. These included:

i. General settings (e.g., set calibration, logging intervals, time, date) were regularly checked
ii. Zero calibrations were done daily before the sampling
iii. The flow of the instrument was calibrated on every measuring day
iv. Leaks were frequently tested
v. Factory calibration was done once a year as specified by the manufacturer.

Zero calibration on every sampling day was important as it enabled the instrument to detect reliable and accurate measurements of the respirable particles, especially when at low levels. Before the beginning of indoor PM_4 monitoring in Lebohang, all the DustTraks that were used were already been specifically calibrated for 2018 using a photometer (Model 8587) as a reference instrument. Since the jar method meets the requirements of the Occupational Safety and Health Administration (OSHA), it was used for the flow calibration of the instruments used in this research (OSHA, 1999). Language (2020) provides a detailed discussion of the full procedure and the tools needed to conduct a jar method flow calibration of DustTrak monitors.

A thorough inspection of the cyclones and inlets was regularly conducted before each sampling day for damage checks and when found, faulty cyclones and inlets were immediately changed. The cyclones were also subjected to a cleaning process that was conducted on a daily basis before indoor PM_4 was continuously monitored in each participating house. This was done by detaching the cyclone from the instruments and disconnecting the grit pot from the rest of the body and cleaning it with isopropyl alcohol to remove the trapped particles. Leaks of the instruments were regularly assessed to check faults in either the internal pump, tubing or O-ring by utilizing the "*pump-fault leak test*". To do this, the 10 mm Dorr-Oliver cyclone was first attached to the DustTrak, then an appropriate flow rate was set and thereafter, a thumb was placed on the inlet to block the sampled air. The DustTraks were judged to have failed the test if they did not report an alert of "*flow error*" and were deemed to have passed if the opposite was the case. This approach was also used by Language (2020) and Qhekwana (2020) when sampling PM_4 concentrations within low income settlements on the Mpumalanga Highveld.

Data analysis

The DustTrak monitor is associated with erroneous measurements when used in field research to monitor ambient aerosols that prevail, for instance, in a low income residential area (Dionisio *et al*., 2010). This is because the readings of the DustTrak are affected if the instrument is used to measure particles with different physical properties (e.g. density, size, and shape) than those that were used during its factory calibration. As such, when monitoring aerosols with different characteristics, DustTrak monitors overestimate particulate pollution. As reported by previous studies, it is estimated that on average, a DustTrak monitor could overestimate PM levels by a magnitude that ranges from 2 to 3 (Chan *et al*., 2018; Wallace *et al*., 2011). To address this issue, the correction factor (CF) of 0.075 calculated by Language *et al.* (2016) was thus, used to correct raw indoor PM_4 data obtained in Lebohang during household monitoring.

The CF used in this study was calculated based on the gravimetric measurements of PM_4 that were taken at the same time as continuous DustTrak measurements obtained in KwaDela. Because gravimetric measurements are subject to fewer errors in comparison to those obtained from photometric instruments, the CF used in this study, therefore, adjusted Dustrak data against that obtained gravimetrically. A similar approach applied in this study was also used by (Adesina *et al*., 2020; Qhekwana, 2020; Chan *et al*., 2018; Language *et al*., 2016) to correct the erroneous indoor data gathered from the DustTrak monitors.

Considering the sensitivity of the mean to outliers, before processing and visualizing the household data, extreme values, which were beyond 10 000 $\mu g.m^{-3}$ were omitted from the analysis. Moreover, only homes with at least 5 days of data points were included in the analysis. Out of 22 monitored houses in Lebohang, only 20 houses met this standard and thus, specifically, H11 and H12 were removed from the analysis. The data from the remaining houses were integrated into a single dataset, which was then processed to generate descriptive statistics and box plots.

3.2.3 Exposure assessment methods

Due to the mobile nature of human beings, people experience different PM concentrations at different spaces and times. To profile the total PM exposure of different subgroups in Lebohang, the indirect assessment method was used. The accuracy of this exposure assessment technique concerning the ”*perfectly precise*” standard, ranges from moderate to good approach (Monn, 2001). Indirect, as opposed to the direct method, which evaluates an individual’s real-time contact with a pollutant, provides estimates of the total integrated exposure of well-defined groups of people or populations. Furthermore, it also relies on the use of predefined “microenvironments”

(MEs) (i.e. specified spaces or locations where pollution concentration experienced by the assessed interest group is assumed to have no significant variance) (Monn, 2001).

The MEs of interest considered in the present research were indoor home and outdoor home environments. The former was assumed to be consisting of household spaces such as living as well as bedrooms, and kitchen areas. On the other hand, the latter was assumed to be made up of places that are outside enclosed buildings, such as immediate households' yards, residential streets and roads, sporting fields, and playing grounds. It is, however, acknowledged that there are many main MEs where people, more especially children and adults also spend their time in. Monn (2001) outlined other 3 primary MEs, in addition to the ones considered in this study, where people spend their time in everyday life. The same author listed that people usually also spend their time in the MEs outlined below:

- Transport (e.g., while in taxis, vehicles, buses and/or walking)
- School and workplace
- Other indoor places, such as theatres, malls, indoor facilities of sports, bars and restaurants.

The selection of the MEs considered in the present study was, therefore, influenced by the availability of the concentration data monitored within the indoor home and outdoor home in Lebohang. Due to the lack of concentration data monitored, particularly in the transits, schools and workplaces as well as other indoor places, these MEs were thus, ignored.

Although equation (1), outlined earlier in section 2.3, is recommended by the USEPA as a basis for estimating mean integrated exposure, the equation was modified to estimate the average population-weighted total integrated exposure experienced by each subgroup in the community. Thus, to estimate the average integrated population-weighted total exposure for each demographic group in Lebohang, the following mathematical equation model outlined below was used:

$$E_{population\ group} = \sum_{j}^{n} \frac{(C_{dg,i} \ \times t_{dg,i}) + (c_{dg,a} \ \times \ t_{dg,a})}{n}$$

Equation (3)

Where $E_{population\ group}$ = total population-weighted integrated exposure of the assessed social group (µg.m^{-3}); n = total sum of the demographic group, $C_{dg,i}$ = concentration of PM experienced by the demographic group dg within indoors i; $t_{dg,i}$= time spent by the population group dg in the indoor environment i; $C_{dg,a}$ = the concentration of PM experienced by the demographic group dg in the ambient environment a; $t_{dg,a}$= time spent by the population group dg in the ambient environment a; and j = microenvironments of concern (i.e. indoor and ambient environment).

3.2.3.1 Input parameters for indirect exposure assessment

The required input parameters for the indirect exposure assessment method included indoor PM_4 data, ambient $PM_{2.5}$ measured concentrations, the approximate time (hr/day) spent by different demographic populations in MEs of interest, as well as the population size for each respective population group. The ambient $PM_{2.5}$ and household PM_4 data used in this study were obtained from the continuous monitoring that was conducted in Lebohang (see sections 3.2.1 to 3.2.2.2).

The researcher is aware that human exposure to particulate pollution should ideally be estimated using measurements of the same PM fraction size, monitored in different MEs of interest. However, since only PM_4 was sampled in houses that participated in this study during the 2018 Sasol baseline monitoring project (refer to section 3.2.2.2), only ambient $PM_{2.5}$ data obtained from SAAQIS was used to represent outdoor particulate pollution in Lebohang. Ambient $PM_{2.5}$ was selected because is a good approximation of PM_4 and has almost the same size as PM_4 (Language *et al.*, 2018). As seen in Figure 2-1, it is clear that both PM_4 and $PM_{2.5}$ falls under the coarse mode classification (≥ 2.5 µm - < 10 µm) and by shifting the $PM_{2.5}$ cut-off size a bit towards PM_4, there is no huge difference. PM_4 is, therefore, most likely represented by particles that are ≥ 2.5 µm but < 4 µm. As such, using both $PM_{2.5}$ and PM_4 to profile total PM exposure in Lebohang, did not introduce huge uncertainties in the estimates per se.

Ideally, exposure assessment relies on the concentration data that was continuously recorded within different MEs and time-activity data that was recorded concurrently with the pollutants of concern (Sidhu *et al*., 2017). However, in the current study, the exact time spent by different population groups in the various MEs of interest was unknown. Consequently, this data was obtained from the average values (hr) which were already published in the literature. With regards to the population sizes of each demographic group assessed in the present research, such information was estimated from the 2011 census community survey data of Lebohang (Stats SA, 2011).

3.2.3.1.1 Demographic data

The 2011 census community survey of Lebohang, aggregated at the three (3) ward levels, namely wards 2, 3 and 6 was used to identify the population sizes of each demographic group considered in this study. Municipal wards were the smallest geographical level, where all the demographic data of interest to this study was accessible and available. Although it is recognised that Lebohang is made of a total of 4 municipal wards (i.e. wards 1, 2, 3 and 6) (Wazimaps, 2011), only wards that consisted of at least ≥99% of Black African residents were selected. The geographical method used in this study was motivated by the absence of the demographic data (age, gender, and occupational information) of individuals who lived in homes where PM_4 was monitored in Lebohang. Several social science studies (Haal *et al*., 2018; Currie & Schwandt, 2016), have shown that aggregating demographic data at a certain specific areal level overcomes the limitation of not having the required population data, specifically at the individual level. Hence, by using the 2011 census community survey, it was possible to get the sample size of Lebohang residents, respectively according to their social traits such as age groups, gender as well as occupational status. Table **3**-**5** below summarises the social traits of different demographic groups of residents in Lebohang based on the 2011 census community survey.

Table 3-5: Social description for different population groups in Lebohang low income settlement (Stats SA, 2011).

Group Name	Age-grouping (years)	Gender	Occupational status	Population size
Infants	1 - 2	N/A	N/A	1,710[a]
Toddlers	3 - 6	N/A	N/A	1,889[a]
School going children	7 – 18	Male	N/A	2,833[b]
		Female	N/A	2,678[b]
Adults	19 - 64	Male	Employed	3,322
			Unemployed	1,598
		Female	Employed	2,062
			Unemployed	1,935
Retired people	≥65	Male	N/A	379[b]
		Female	N/A	895[b]

[a] Not gender specified
[b] Not differentiated by occupation
N/A – Not applicable

To identify different social groups which were assessed in this study, residents in Lebohang were classified into various demographic populations based on the presence of specific social characteristics. These included age (1-2 – infants, 3-6 – toddlers, 7-18 - school going children, 19-65 – adults, and ≥65 years - retired people); gender (i.e. male and female); and occupational status (i.e. either employed or unemployed). These groupings assumed that the presence of common social characteristics amongst different individuals in Lebohang, was likely to influence their time-activity budget in the same manner. In contrast, the opposite was thought to be likely the case, if people shared none of the outlined social traits. Because no evidence suggests that the activities of toddlers, infants and old people vary based on gender (Cohen Hubal *et al*., 2000), only adults and school going children were differentiated by gender. Moreover, adults were further grouped based on their occupational statuses (i.e. employed or unemployed) since working and nonworking adults have different time activity budgets (see Figure 3-10 below).

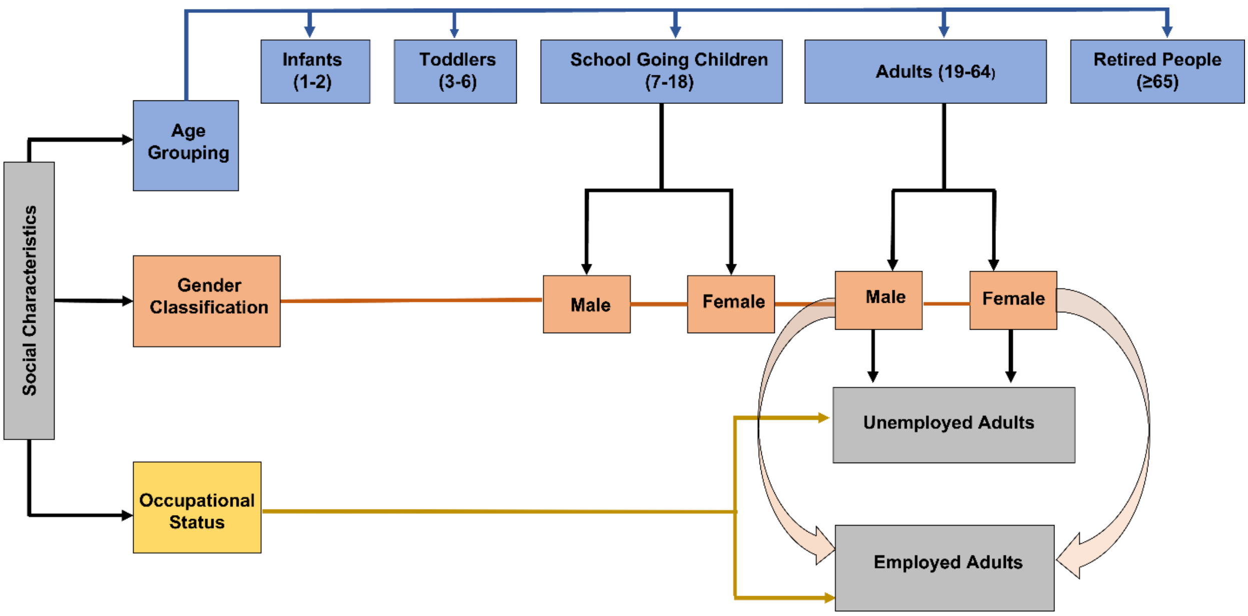

Figure 3-10: Schematic illustration of the population grouping of Lebohang residents based on the presence of common social characteristics.

3.2.3.1.2 Time activity data

Time activity data plays an important role in exposure assessment study. To determine one for various population groups assessed in Lebohang, the reported mean time spent at outdoor and indoor home (as defined in section 3.2.3) were obtained from the literature. Because time activity data documented in the literature is collected for various research purposes, not all the reported time activity data were expressed as hours or summed up to a total of 24 hours. The reason is, how the time activity data is collected is influenced by the purpose of the study in question hence, it is difficult for such data to be used in another study without making any adjustments. As such, for convenience, in instances where the average time was reported as a percentage of time spent in MEs of interest to this study, such fractions of times were converted into hours. This was done by dividing such fraction of time by 100 and multiplying it by 24h if the study in question accounted for the entire day, otherwise, the total amount of hours that the activity pattern of the subject was monitored. Since people spend most of their time indoors, when collecting the average time spent indoors at home, only studies that sampled the time activity pattern of the respondents for 24h hours were considered. This was to avoid mispresenting or underestimating the true time various population groups spent indoors in low income communities.

In real life, the time activity patterns of people are influenced by numerous factors which could either be biological (e.g. race and ethnicity), social (income and occupational status), or external (weather) (Mccurdy & Graham, 2003). These factors make exposure assessment complex as they vary from one place to another and from one person to the next. Although it is recognised that the aforementioned factors are the drivers of human time-activity patterns, in this study, age, gender, and socio-economic status were deemed to be the main determinants of how people spent their time. This decision was made based on the work done by Yang *et al*. (2010) who found that the factors considered in the present research had a significant impact on determining time spent indoors. As such, when determining time activity data for various groups in Lebohang, preference was given to studies that had primarily recorded time activity patterns of people who live in low income areas. Table 3-6 below shows the summary of the collected average time (hr) spent in the outdoor home and indoor home by different population groups as reported in the literature.

Assumptions

To use the secondary time activity data, several assumptions had to be made in this study and those already made in other published research were also adopted. Otherwise, most of the reported time values, particularly those obtained from studies that were conducted specifically in low income communities, would have been rejected. The reported average time spent by demographic groups of interests was in some instances, assumed to be the total sum of all the average time fractions spent in various MEs of interests. This, however, depended on how the cited paper defined places which constituted “indoor home” and “outdoor home” in their studies. For instance, if the cited journal article assumed that the outdoor home was made of “dirt roads” as well as “outdoor other”, and respectively reported time spent in each of these MEs, all reported values were aggregated together and included in the analysis as a single value. In situations where time spent either at outdoor home or indoor home was representative of both genders, such a value was thought of as being reflective of the dominant gender which the study in question sampled. Similarly, if the reported time spent either indoors or outdoors was not differentiated based on occupational status, then that value was assumed to be reflective of the status of the most sampled population.

The time activity patterns of each of the assessed demographic groups in Lebohang were also assumed to be independent of the days of the week and seasons. For example, in cases where the mean time spent was reported individually for each season and respectively for weekdays and weekends, all the reported values were considered and included in the analysis. The researcher is aware that some of the assumptions made here might have introduced uncertainty

Table 3-6: Summary of the mean values spent in the outdoor and indoor home by different population groups collected from the literature.

Microenvironment	**Occupational status**	**Not gender specific**	**Male**	**Female**	**Reference and study location**	**Description of the sampled population and method used**
		time (hrs)	time (hrs)	time (hrs)		
(a) Literature reported the average time (hrs) spent by infants (1-2 years)						
Outdoor home						
		1.1[d]			[a] KwaZulu Natal (South Africa)	[a] Matooane *et al*. (2003) investigated the time activity patterns of 381 adults using questionnaires. 59% of these participants were females, while the remaining 41% were males. Furthermore, 60% of the entire sampled population was unemployed. The author respectively reported the average time spent outdoors and indoor homes by unemployed and employed adults.
		1.2[e]				
		1.98[f]				
		2.9[h]				
		3.7[h]				
		4.6[k]				
		4.4[k]				
		4.7[k]			[b] Mpumalanga (South Africa)	[b] Wernecke (2018) monitored the time activity pattern of a single unemployed adult woman for 8 consecutive days (i.e. from 21-
		4.7[k]				

Continuation of table 3-6

			August to 28 August 2022), from 9 am to 6 pm using a GPS, and respectively presented the average time spent at outdoor home and indoor home for each day.
Indoor home			
	21.3[h]		
		[c] KwaZulu Natal (South Africa)	[c]. Muller *et al*. (2003) surveyed the time activity patterns of 69 people, 66 of which were females around 34 years. Of these 66 participants, 39 were unemployed and the remaining were employed. The author respectively reported the average time spent outdoors during the winter and summer seasons.
	20.3[h]		
	19.4[k]		
	19.6[k]		
	19.3[k]		
	19.3[k]		
	20.7[d]	[d] Nova Scotia, Valley-Kings County, Ontario and Haldimand-Norfolk (Canada)	[d] Matz *et al*. (2015) surveyed a total of 1460 people who lived in Canadian rural areas using a diary method. The authors reported the average time spent outdoors and indoor home by sample populations respectively following their age groupings. However, the reported average time activity values in this study were not differentiated by gender and occupational status.
	15.8[e]		
	20.93[f]		
	19.4[k]		

(b) Literature reported the average time (hrs) spent by toddlers (3-6 years)

Outdoor home

Continuation of table 3-6

	2.5[d] 10.2[j] 1.2[e] 1.98[f] 4.7[h] 5.0[h]	[e] Chongqing (China)	[e] Wang *et al.* (2008) surveyed the time activity budget of 90 rural homesteads using a questionnaire. The time-activity patterns of members of these homes were recorded for about 7 days. Wang *et al.* (2008) grouped the sampled population based on age and presented the average time spent at various MEs by each group.
	9.8[j]	[f] Delhi (India)	[f] Saksena *et al*. (2000) surveyed the time activity pattern of 1100 homes in Delhi, which ranged from low to high income households. In total, the sample population of this study was made up of 4311 participants. The authors reported the average time spent outdoors and indoors respectively by children, students (both genders), workers (both genders), non-workers (both genders) and elderly people.
Indoor home	19.4[h] 19.1[h] 17.4[d] 16.3[d] 20.93[f] 13.96[j]		

Continuation of table 3-6

	13.68[j]				
	20.3[e]				
(c) Literature reported the average time (hrs) spent by school-going children (7-18 years)					
Outdoor home				[g] Minneapolis (United States)	[g] Adgate *et al*. (2004) surveyed the time activity budget of 153 children from two schools which were mostly attended by learners from low income households. Out of these 153 children, 55.9% were male and 44 were female.The authors respectively reported the average time spent by both learners in the indoor and outdoor environments.
		2.22[f]	3.6[j]		
		0.77[g]	2.09[f]		
		8.1[h]	6.6[h]		
		9.4[h]	7.2[h]		
		7.2[h]	6.0[h]		
		5.6[d]	9.4[j]	[h] Cox's Bazar, Dhaka, Faridpur, Jessore, Rajshahi, Sylhet and Rangpur (Bangladeshi)	[h] Dasgupta *et al.* (2006) surveyed the time activity budget of 4612 participants from seven regions of Bangladeshi using a questionnaire. The sample population of this study was made up of individuals who lived in households that were in urban, peri-urban and rural areas of Bangladeshi. Dasgupta *et al.* (2006) reported the daily average hours spent indoors and outdoors respectively by different age groups
		10.2[j]	6.5[j]		
			5.6[d]		
Indoor home					
		13.59[j]	20.9[j]		
		16.9[h]	20.18[j]		

Continuation of table 3-6

		14.16[f]	15.2[d]	[i] Tamil Nadu (India)	[i] Balakrishnan *et al.* (2002) sampled the time activity patterns of 529 adults who lived in 30 different villages in Tamil Nadu, India. Of these 529 participants, 339 were women who cooks full-time in their households, 29 were women who occasionally cooks, 49 were women who only assists in cooking, 76 were unemployed men, most of which were sick and retired; and 39 were employed men who worked outdoor jobs.
		14.6[h]	17.4[h]		
		15.9[h]	17.3[j]		
		16.3[g]	14.51[f]		
		15.2[d]	16.9[h]		
			18.1[h]		

(d) Literature reported the average time (hrs) spent by adults (19-64 years)

Outdoor home					
	Employed				
		1.26[f]	1.7[a]		
		11.64[i]	1.06[f]		
	Unemployed				
		1.90[f]	6.03[b]		

Continuation of table 3-6

		6.34[i]	*4.67[b]	[j] Telangana, Rangareddy and Warangal region (India)	[j] Balakrishnan *et al.* (2004) surveyed the time activity budget of household members from three districts (i.e. Telangana, Rangareddy and Warangal) in India using a recall method. The authors-reported the average time spent indoors and outdoors by the sampled participants respectively according to their age groups and gender.
		0.76[a]	*3.69[b]		
		5.2[e]	*4.42[b]		
			*5.67[b]		
			1.61[f]		
			1.63[f]		
			*5.04[b]		
			2.02[a]	[k] South Delhi (Kusumpur Pahari Slum) and West Delhi (Kathputly Slum) (India)	[k] Malhotra *et al*. (2000) surveyed the time activity budget of infants/mothers from two slums, namely, Kusumpur Pahari slum and Kathputly slum through interviews. In total, the time activity budget of 80 infants/mothers was sampled. Of these 80 participants, 40 were from the Kusumpur Pahari slum and the remaining portion was from the Kathputly slum. The authors reported the average time spent by infants indoors and outdoors in each slum, respectively for wood and kerosene-reliant homes.
			*4.23[b]		
			4[c]		
			6.3[c]		
			5.2[e]		
			*4.23[b]		
Indoor home					
	Employed				
		12.16[f]	14[l]		

Continuation of table 3-6

	12.36[i]	13.45[a]	[l] Mumbai (India)	[l] Anand & Phuleria (2022) used a questionnaire to capture the time activity budget of residents from 500 homes in 7 various slums in Mumbai City. The authors never provided the demographics of their sampled population (i.e. the fraction of sampled males and females). However, they reported the average time spent indoors respectively for different genders and employed and unemployed participants. It should be noted that the temporal activity patterns of employed and unemployed participants were not differentiated by gender.
	14[l]	14.09[f]		
Unemployed				
	14[l]	21.67[f]		
	16[a]	22[f]		
	20.92[f]	20.45[a]		
	17.7[e]	18.86[a]		
		17.7[e]		
		18.66[i]		
		14.3[l]		
		22[l]		
		23[l]		

(e) Literature reported the average time (hrs) spent by retired people (>65 years)

Outdoor home	
	3.3[j]

Continuation of table 3-6

	2.2[d]
	3.5[e]
	2.31[f]
	2.44[f]
	2.9[h]
	6.3[j]
	7.4[h]
	6.34[i]
Indoor home	
	20.44[f]
	21.3[h]
	16.8[h]
	18.4[d]
	20.62[f]
	20.5[e]

Continuation of table 3-6

	20.49[j]	

[a] Matooane *et al*. (2003)

[b] Wernecke (2018)

[c] Muller *et al.* (2002)

[d] Matz *et al*. (2015)

[e] Wang *et al.* (2008)

[f] Saksena *et al.* (2007)

[g] Adgate *et al*. (2004)

[h] Dasgupta *et al.* (2006)

[i] Balakrishnan *et al.* (2002)

[j] Balakrishnan *et al.* (2004)

[k] Malhotra *et al*. (2000)

[l] Anand & Phuleria (2022)

*Denotes the average time value extracted from a study that never monitored the time activity pattern of the subject in question for 24h.

in the results, hence uncertainties associated with exposure estimates obtained in this research were quantified using Monte Carlo simulation. Table 3-6 shows the summary of the collected average time (hr) spent in the outdoor and indoor home by different population groups as reported in the literature.

3.2.3.2 Monte Carlo simulation

When using a deterministic model to simulate exposure, uncertainty or parameter variability tends to be propagated, which makes the resulting estimated values of exposure uncertain (Issac *et al*., 2014). The uncertainty stems from the lack of information about the true invariant value of a particular parameter that is used in the model as a variable input to assess exposure (Moschandreas and Saksena, 2002). This consequently impacts the accuracy of the calculations made by such a model and the extent to which inferences can be drawn from estimates generated from the model per se. Since the exact time spent indoors and outdoors by various groups assessed in Lebohang was unknown, the time input variables were classified as uncertain due to the variability of values obtained in the literature (refer to table 3-6). As such, to conduct an empirical analysis, exposure for each group was calculated using equation (3) and the Monte Carlo simulation to account for uncertainties associated with the estimated exposure results.

Monte Carlo simulation is a mathematical tool that repetitively uses random data points from the distribution of each input variable to forecast all possible scenarios of human exposure (Ramanathan, 2008). As such, it is also referred to as the "*what if analysis*", as it generates exposure episodes that range from worst to best while also accounting for input parameter's uncertainties. A step-by-step detailed explanation of how to conduct a monte carlo simulation is covered by Nkosi (2018) and Ramanathan (2008) thus, it is not much discussed in this research. However, it should be noted that to conduct the monte carlo simulation, the statistics (mean and standard deviation) and distributions of all the input parameters need to be identified. Because values collected to represent the time activity budget for each population subgroup were few due to the scarcity of relevant data in literature; and were variable, bootstrapping was performed.

Bootstrapping is a statistical method that produces set a of arrays by repetitively selecting a single value randomly from the original input data to generate a new dataset that has a larger sample size (Ramanathan, 2008; Mestl *et al*., 2006). It was performed to account for uncertainties of time input variables in the model by estimating with a high confidence interval (95% CI), the mean and standard deviation of uncertain parameters. Due to the data scarcity of the indoor time activity values for employed adults, the sample size of time data collected (n = 2) for both male and female workers was too small to represent reality. As a result, bootstrap could not be performed on adults workers' indoor time activity data and because of this, exposure profiles for employed adults in Lebohang were not estimated.

Input parameter distributions

The probability distribution of each input variable outlined in section 3.2.3.1, was determined based on expert judgment and literature to select the best conventional distribution for each variable. For indoor PM_4 and ambient $PM_{2.5}$ concentration data collected in Lebohang, it was assumed that these variables followed a gamma distribution. On the other hand, the distribution of the time activity data for each subgroup in Lebohang was assumed to have a normal distribution. This decision was supported by the work of Mestl *et al*. (2006) and Park *et al*. (2020) who also simulated exposure profiles of various groups by assuming that the time activity data for each group was normally distributed. Table 3-7 summarizes all the input variables used in this study to estimate the exposure profiles of different subgroups in Lebohang as well as their associated statistics, distributions and their uncertainties.

The monte carlo simulation was conducted using Python version 3.9.16 and the simulations were run 10 000 times to be highly certain (95% confident) about the generated exposure estimates. In each trial of the simulation, an array of numbers was randomly selected from each input variable based on their selected distributions and were used to calculate exposure using equation (**3**). Hence, all possible exposure cases for each population group were generated in the simulations and the most likely scenario to occur 95% of the time, was presented in the results. The selected iteration number for the monte carlo simulation in the present research is similar to that of Allan and Richardson (1998) and Nkosi (2018).

Table 3-7: Input parameters with their associated distributions and statistics used in this study to estimate the exposure profiles of various population groups in Lebohang using the Monte Carlo simulation.

Input parameter	Count	Distribution	Mean	Std	Range
Autumn daily average ambient $PM_{2.5}$ ($\mu g.m^{-3}$)	29	Gamma	21	± 9	7 - 49
Autumn daily indoor PM_4 ($\mu g.m^{-3}$)	29	Gamma	102	± 30	62 - 199
Winter daily average ambient $PM_{2.5}$ ($\mu g.m^{-3}$)	37	Gamma	50	± 21	17 - 95
Winter daily indoor PM_4 ($\mu g.m^{-3}$)	37	Gamma	111	± 36	55 - 198
Infant outdoor time (hr)	9	Normal	3.3 (95% CI, 2.3 - 4.1)	± 1.4	(95% CI) 1.1 – 4.7
Toddlers' outdoor time (hr)	7	Normal	5.0 (95% CI, 2.7 – 7.7)	± 3.4	(95% CI) 1.2 – 10.2
Male school children's outdoor time (hr)	7	Normal	6.2 (95% CI, 3.6 – 8.5)	± 3.3	(95% CI) 0.8 – 10.2
Female school children's outdoor time (hr)	8	Normal	5.9 (95% CI, 4.4 – 7.3)	± 2.0	(95% CI) 2.1 – 9.4
Unemployed male adults' outdoor time (hr)	4	Normal	3.6 (95% CI, 1.3 – 5.8)	± 2.3	(95% CI) 0.8 – 6.3
Unemployed female adults' outdoor time (hr)	14	Normal	4.2 (95% CI, 3.4 – 4.9)	± 1.5	(95% CI) 1.6 – 6.3
Employed male adults outdoor time (hr)	-	-	-	-	-
Employed female adults outdoor time (hr)	-	-	-	-	-
Retired people's outdoor time (hr)	9	Normal	4.0 (95% CI, 2.9 – 5.4)	± 1.9	(95% CI) 2.2 – 7.4
Infant indoor time (hr)	9	Normal	19.7 (95% CI, 18.5 – 20.5)	± 1.6	(95% CI) 15.8 – 21.3
Toddlers' indoor time (hr)	8	Normal	17.7 (95% CI, 15.8 – 19.4)	± 2.6	(95% CI) 13.7 – 20.9
Male school children indoor time (hr)	7	Normal	15.2 (95% CI, 14.4 – 16.1)	± 1.1	(95% CI) 13.6 – 16.9
Female school children indoor time (hr)	8	Normal	17.5 (95% CI, 16.2 – 19.0)	± 2.0	(95% CI) 14.5 – 20.9
Unemployed male adult's indoor time (hr)	5	Normal	16.5 (95% CI, 14.4 – 18.9)	± 2.7	(95% CI) 14.0 – 20.9
Unemployed female adult's indoor time (hr)	10	Normal	19.3 (95% CI, 17.3 – 21.0)	± 3.0	(95% CI) 14.1 – 23.0
Employed male adults indoor time (hr)	-	-	-	-	-
Employed female adults indoor time (hr)	-	-	-	-	-
Retired people indoor time (hr)	4	Normal	19.8 (95% CI, 18.6 – 20.87)	± 1.5	(95% CI) 16.8 – 21.3

CHAPTER 4: CHARACTERISING AMBIENT PM CONCENTRATIONS IN LEBOHANG

The ambient $PM_{2.5}$ and PM_{10} concentration data collected in Lebohang low income settlement is presented in this chapter. The chapter answers the first object of this research by analysing the ambient $PM_{2.5}$ and PM_{10} data for the October 2017 to August 2018 monitoring period obtained from Lebohang SAS.

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To characterise ambient PM concentrations for Lebohang, the data was explored temporally to assess how PM varies across different times of the day, weekdays and year in the community. This was important as it also provided PM concentration signatures, which in turn, allowed for potential local sources that might have likely impacted the air quality in the area during the sampling period, to be traced back. When exploring the temporal variance of the ambient PM concentrations in the community, the following analyses were conducted: Firstly, a time series graph showing all the ambient PM concentration measurements that were collected during the sampling period was plotted to summarise what was sampled in the area. Secondly, the data was further analysed by evaluating the 24h mean monthly variance of the ambient $PM_{2.5}$ and PM_{10} concentrations to identify the monthly profiles of PM during the sampling period. Thereafter, to assess the impacts of different days of the week on the behaviour of ambient particles in Lebohang, the plot showing the PM variation during the weekdays was generated. Finally, this was followed by the analysis of the seasonal variance of PM concentrations, which was done by evaluating the diurnal cycles (produced by hourly mean concentrations) for each respective season. Of note, when evaluating the temporal trends of outdoor PM data in Lebohang, where possible, a comparative analysis of ambient $PM_{2.5}$ and PM_{10} concentrations was also conducted.

4.1 Time series analysis of the ambient PM concentrations in Lebohang

Figure 4-1 (a) and (b) below respectively depicts the time series of 24h mean ambient $PM_{2.5}$ and PM_{10} concentrations recorded throughout the 11-month monitoring period in Lebohang. It further compares these measurements against the 24h mean South African NAAQS of 40 $\mu g.m^{-3}$ for $PM_{2.5}$ and 75 $\mu g.m^{-3}$ for PM_{10}. Although it was not statistically confirmed, it is evident that during the entire sampling period (October 2017 – August 2018), ambient PM_{10} in Lebohang recorded higher concentrations than those of $PM_{2.5}$ (Figure 4-1 (a) and (b)). Notwithstanding, ambient $PM_{2.5}$ and PM_{10} were mostly at unhealthy levels, particularly during the cold period (from May and August 2018) as both pollutants were not compliant with NAAQS. However, the opposite was the case from October 2017 to April 2018, as the recorded ambient $PM_{2.5}$ and PM_{10} levels were seldomly beyond the NAAQS (Figure 4-1 (a) and (b)). The difference in the compliance status of particulate pollution throughout the study period reflects that the frequency of emissions sources in Lebohang varies temporally.

Furthermore, an interesting observation made from Figure 4-1 was that ambient $PM_{2.5}$ and PM_{10} levels in Lebohang had the same distribution pattern throughout the monitoring period. The $PM_{2.5}$ and PM_{10} trends were characterised by low concentrations which were below the NAAQS from October 2017 to April 2018, and high PM levels, which were often beyond the NAAQS from May 2018 to August 2018. The observed similar pattern of ambient PM concentrations suggests that $PM_{2.5}$ and PM_{10} emissions might originate from similar primary sources in Lebohang (Feig *et al*., 2016). To specifically mention these sources, however, a source apportionment study is required to chemically trace the elemental composition of PM concentrations measured in this community (Dionisio *et al*., 2010). Nevertheless, previous studies have reported that within South African low income settlements, domestic combustion of coal is a predominant source of PM, especially during the winter season (Wernecke, 2018; Nkosi *et al*., 2021). As such, the high $PM_{2.5}$ and PM_{10} levels that were seen between May 2018 to August 2018 (Figure 4-1) in Lebohang might have been reflective of the residential burning of coal practiced in the community.

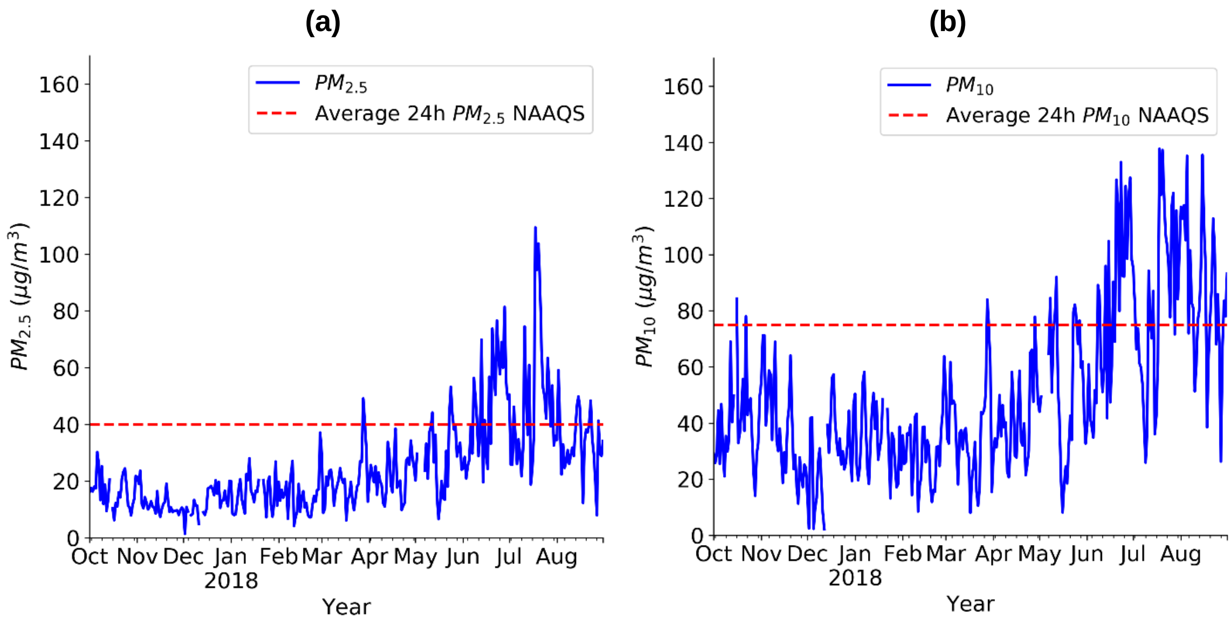

Figure 4-1: Time series data of the ambient $PM_{2.5}$ (a) and PM_{10} (b) concentrations measured in Lebohang during the entire sampling duration (October 2017 to August 2018). The red dotted line represents the 24h average South African National Ambient Air Quality Standard (NAAQS) of 40 $\mu g.m^{-3}$ and 75 $\mu g.m^{-3}$ for $PM_{2.5}$ and PM_{10}, respectively.

To further analyse the distribution of the ambient PM concentrations recorded in Lebohang, the cumulative distributions of the time series daily average measurements were plotted (Figure 4-2 (a) and (b)). Both ambient $PM_{2.5}$ and PM_{10} concentrations measured in Lebohang displayed the same frequency distribution shapes, which were skewed to the right (Figure 4-2 (a) and (b)). Due to the lognormal distribution of ambient PM measurements in the present study, the centre of the outdoor PM data was thus explored using averages and medians.

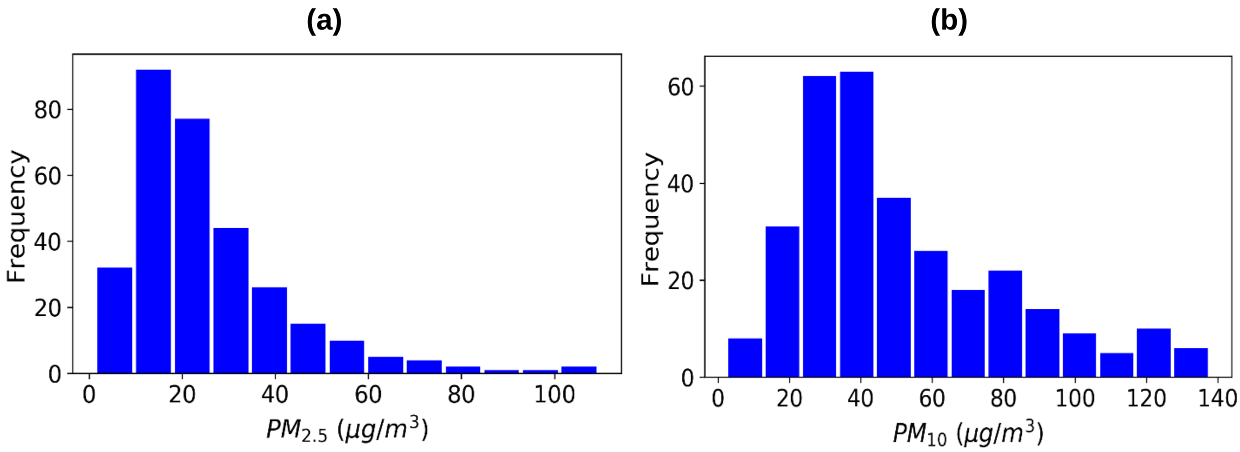

Figure 4-2: Frequency histogram showing the distribution of the 24h mean concentrations of $PM_{2.5}$ (a) and PM_{10} (b) measured in Lebohang during the monitoring period (October 2017 to August 2018). Of note, the frequency for $PM_{2.5}$ (a) and PM_{10} (b) have different scales.

Figure 4-2 further revealed that most of the 24h average PM measurements recorded in Lebohang fell between 0 µg.m^{-3} and 30 µg.m^{-3}. This observation was particularly interesting as the recorded range in Lebohang is relatively lower in comparison to those that were reported within other low income settlements by previous studies. For instance, Wernecke *et al*. (2015), reported that most of the ambient PM measurements taken in the KwaDela residential area, in Mpumalanga, South Africa, fell between >0 – 50 µg.m^{-3}. Moreover, Arku *et al.* (2008) observed that $PM_{2.5}$ and PM_{10} concentration samples, which were monitored in two impoverished communities of Accra, Ghana, were between >0 - 100 µg.m^{-3} and >0 – 200 µg.m^{-3}, respectively. The variance in the PM range recorded across all these areas might have been influenced by neighbourhood factors (e.g. population size, meteorology and socioeconomic status and geography) which varied from one community to another. However, it is also possible that it might have been influenced by the PM sampling duration and protocols that varied across all these residential areas. To explicitly explain why the range of PM concentrations measured in Lebohang was lower than that of other low income areas studied by Wernecke *et al*. (2015) and Arku *et al.* (2008) , further research is thus, required.

4.1.1 Monthly difference of the 24h average ambient PM concentrations

Table 4-1 below describes the statistics of the 24h mean PM concentrations observed in the community throughout the sampling campaign and during each monitored month. Furthermore, the averaged 24h ambient $PM_{2.5}$ and PM_{10} measurements recorded during the full study period (October 2017 to August 2018) and each sampled month in Lebohang are shown in Figure 4-3 (a) and (b), respectively. For the full period, the daily averaged particulate concentrations recorded in Lebohang varied from 1 to 109 µg.m^{-3}, as well as between 2 and 138 µg.m^{-3} for ambient $PM_{2.5}$ and PM_{10}, respectively (see Table 4-1). Ambient PM_{10} recorded the highest mean daily concentration of 51 ± 30 µg.m^{-3}, which was 2 times greater than that of $PM_{2.5}$ (25 ± 17 µg.m^{-3}). To statistically confirm this variation, a t-test was conducted and showed that the 24h average of ambient $PM_{2.5}$ and PM_{10} concentrations in Lebohang were significantly different ($p > 0.05$). Similarly, the maximum daily mean concentration of 138 µg.m^{-3} for PM_{10} recorded in the area was 1.5 times (56%) greater than that measured for $PM_{2.5}$ loads. Moreover, 99% of the time the mean daily $PM_{2.5}$ and PM_{10} values sampled in the community were respectively 88 µg.m^{-3} and 135 µg.m^{-3}. Both these values were beyond the South African 24h mean NAAQS of 40 µg.m^{-3} for $PM_{2.5}$ and 75 µg.m^{-3} for PM_{10} (Figure 4-3 (a) and (b)). Additionally, out of 325 days that were sampled for PM in this residential area, $PM_{2.5}$ and PM_{10} respectively surpassed the 24h average NAAQS 47 and 67 times, meanwhiles only 4 exceedances are allowed annually (Table 4-1). These results, therefore, indicate that the PM concentration levels in Lebohang are health threatening and people often breathe air that is legislatively declared to be unacceptable for human wellbeing.

Regarding the 24h averaged monthly $PM_{2.5}$ and PM_{10} in Lebohang, the worst PM episodes with considerably variable concentrations were recorded during the cold period (May-August 2018) (Figure 4-3 (a) and (b)). However, during the warm period (October 2017-February 2018) relatively lower particulate pollution, which also showed fewer PM concentration variance was recorded in the areas (Figure 4-3 (a) and (b)). During the cold period, PM concentrations ranged from 20 ± 9 µg.m^{-3} to 49 ± 24 µg.m^{-3} and between 37 ± 18 µg.m^{-3} and 86 ± 26 µg.m^{-3} for $PM_{2.5}$ and PM_{10}, respectively (Table 4-1). Throughout the sampling period, the highest monthly daily mean concentrations of ambient $PM_{2.5}$ (49 ± 24 µg.m^{-3}) and PM_{10} (86 ± 26 µg.m^{-3}) were recorded respectively in July 2018 and August 2018. Despite that it was earlier hypothesised that PM in Lebohang is likely to be emitted by the same sources, resuspended ground dust mostly emits PM_{10}, while the burning of coal generates $PM_{2.5}$ concentrations (Wang *et al*., 2020). Given that ground-level dust is mostly blown into the atmosphere during August rather than in July (Thabethe *et al*., 2014), this justifies why PM_{10} concentrations had higher spikes in August than in July.

Table 4-1: Statistics of the 24-hour average $PM_{2.5}$ and PM_{10} concentrations (µg.m^{-3}) measured in Lebohang during the monitoring period (October 2017 to August 2018) and each sampled month. Of note, only 4 exceedances of the permissible $PM_{2.5}$ and PM_{10} 24h average are allowed each year.

$PM_{2.5}$ concentrations									
	Count	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**	**N Exc.**	**Recovered data (%)**
Full	325	1	25	17	20	88	109	47	90
O	30	6	16	6	16	29	30	-	98
N	30	7	12	4	11	23	24	-	100
D	28	1	13	5	12	20	20	-	90
J	28	7	16	6	16	28	28	-	93
F	28	4	15	7	14	34	37	-	100
M	31	6	20	9	19	47	49	2	100
A	30	10	22	7	20	37	38	-	77
M	27	7	29	12	30	51	53	4	87
J	30	20	46	18	45	80	81	17	100
J	31	19	49	24	42	108	109	16	100
A	30	8	33	11	32	56	59	8	100
PM_{10} concentrations									
	Count	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**	**N Exc.**	**Recovered data (%)**
Full	325	2	51	30	42	135	138	67	90
O	30	14	40	16	36	83	84	2	98
N	30	11	38	17	34	71	71	-	100
D	28	2	30	14	30	57	57	-	97
J	28	13	34	12	34	57	58	-	93
F	28	8	30	12	32	58	64	-	100
M	31	8	37	18	36	81	84	1	100
A	30	21	41	15	38	74	78	1	80
M	27	8	52	24	51	90	92	7	87
J	30	35	79	31	83	131	133	16	100
J	31	26	83	31	82	138	138	18	100
A	30	26	86	26	83	135	136	21	100

During the warm months (October 2017-February 2018) most of the monthly daily mean ambient PM values recorded during this period were below the average 24h NAAQS of 40 µg.m^{-3} for $PM_{2.5}$ and 75 µg.m^{-3} for PM_{10}. However, in October 2017, 2 exceedances of the 24h mean NAAQS of 75 µg.m^{-3} for PM_{10} were recorded in this community (Figure 4-3 (b)). Notwithstanding, although some of the days sampled for PM_{10} concentrations in October were beyond the NAAQS, these exceedances were below the allowed annual number of exceeds (4) (Table 4-1). As such, 100% of the days that were sampled from October 2017 to February 2018 complied with the permissible annual exceedances of the 24h mean PM NAAQS. However, this finding was anticipated since it corresponds with observations that were made within other low income settlements by previous studies (Adesina *et al*., 2020; Wernecke, 2018; Moletsane *et al*., 2021).

Clean ambient air quality observed in Lebohang during the months that constitutes the warm period (October 2017-February 2018) can be explained as a factor of two main reasons. Firstly, reduced domestic combustion of coal in the community, particularly amongst coal-reliant households (Venter & Lourens, 2021; Wernecke, 2018). Because warm seasons are favourable to most such households, especially informal ones thus, residents burn less coal as the demand for space heating at this time of the year drops (Venter & Lourens, 2021). For instance, a study that was conducted in KwaDela found that residents in this community burn coal that is greater by a factor of 2 during winter in comparison to that burned during summer (Wernecke, 2018). Secondly, it can be justified by the atmospheric conditions that prevail during warm seasons, which allows for the dispersion and vertical mixing of pollutants in the atmosphere, as well as the rapid formation of rainfall (Lourens *et al*., 2011). Due to the vertical motion of pollutants and high rainfall, emitted PM concentrations are effectively removed from the atmosphere hence, resulting in warm months having low PM concentrations.

During the cold months (May-August 2018), the 99th percentile of the 24h averages concentrations of both $PM_{2.5}$ and PM_{10} surpassed the mean 24h NAAQS of 40 µg.m^{-3} for $PM_{2.5}$ and 75 µg.m^{-3} for PM_{10} (Table 4-1). However, this excluded the 99th percentile ambient $PM_{2.5}$ daily mean values of 38 µg.m^{-3} that was measured in April 2018, and which complied with 24h mean NAAQS (Figure 4-3 (a)). The highest number of daily averages that were beyond $PM_{2.5}$ 24h mean NAAQS (40 µg.m^{-3}) were recorded in June (17), in a downward sequence, followed by July (16), August (8), May (4) and March (2) (Table 4-1). For PM_{10}, maximum exceedances of the averaged 24h NAAQS (75 µg.m^{-3}) were measured in August (21), in a downward order, followed by July (18), June (7), and both April and May (1) (Table 4-1). All the number of exceedances recorded from March to August 2018, when combined respectively for $PM_{2.5}$ and PM_{10} were beyond the allowed annual frequency of 4 exceedances. The months that constitute the cold period (May to August) in Lebohang were, therefore, characterised by substantially poor ambient air quality.

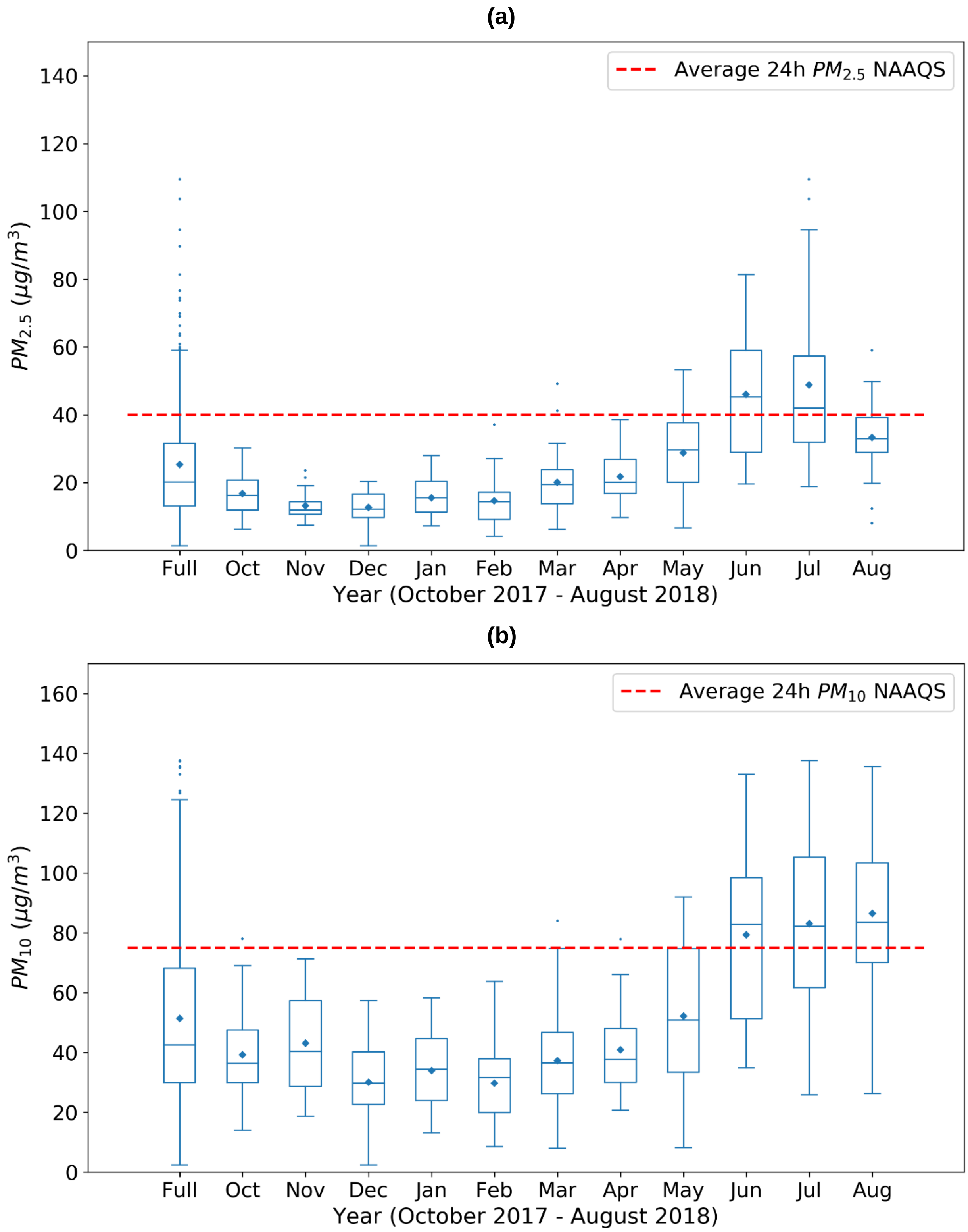

Figure 4-3: 24-hour average ambient $PM_{2.5}$ (a) and PM_{10} (b) measured in Lebohang during the entire sampling period (October 2017 - August 2018) and each of the monitored months. Here and onwards in this dissertation, the box and whisker plots show 25th, 50th and 99th percentiles; mean (blue diamond inside the box); outliers (blue stars beyond the whiskers); and the 24h average South African National Ambient Air Quality Standard (NAAQS) of 40 µg.m^{-3} and 75 µg.m^{-3} for $PM_{2.5}$ and PM_{10}, respectively.

These results agree with those of previous studies that were conducted within low income residential settings (Adesina *et al*., 2020; Feig *et al*., 2016; Lourens *et al*., 2011; Wang *et a*l., 2013; Qhekwana, 2019).

High ambient particulate concentrations measured in Lebohang during the period of May-August 2018 can be attributed to the increased residential coal burning for space heating purposes. As mentioned earlier, a considerable portion of houses (30.9%) in this community use coal to keep warm (Stats SA, 2011). Given that informal and government-subsidised households are characterised by poor thermal conditions (Language *et al*., 2016; Matandirotya *et al*., 2019), the households' demand for coal burning substantially increases during cold periods due to low ambient temperatures. This phenomenon was observed, for instance, by Nkosi *et al*. 2021 in KwaDela low income residential area. These authors found that the average number of daily events of coal burning for households in the aforementioned community ranged from 0 to 2 and between 0 to 3 during the warm and cold seasons, respectively.

Because in a township, domestic use of dirty fuels typically intensifies during the months that constitute the cold periods (May-August) (Lourens *et al*., 2011; Venter & Lourens, 2020), it thus makes sense why Lebohang showed high ambient PM at this time of year. Another reason that might well justify high PM concentrations from May-August 2018 in Lebohang is the meteorological conditions that prevail over the Highveld region during this time (Lourens *et al*., 2011). Given that winter months are associated with slow-moving winds and the inversion layer that forms over the interior of the Highveld, these restrict PM emissions from mixing in the atmosphere (Langerman *et al*., 2018; Jury, 2017). The reason is, the dispersion of particulate concentrations is driven by strong winds, which serve as a transportation mechanism for emitted pollutants in the atmosphere (Wernecke, 2018). The presence of the winter inversion layer, combined with low wind speed, therefore, disrupts the dispersion process of particles, resulting in an enhanced lifespan of PM in the lower atmosphere.

4.1.2 Diurnal distribution of ambient PM concentrations in Lebohang

4.1.2.1 Weekday variation of ambient diurnal PM concentrations

The behaviour of ambient $PM_{2.5}$ and PM_{10} concentrations during hours of the day for different days of the week captured in Lebohang during the entire monitoring period is shown in Figure 4-4. Throughout the days of the week, both ambient $PM_{2.5}$ and PM_{10} pollution revealed bimodal diurnal distributions, characterised by morning (around 06:00) and evening (around 18:00) peaks. These observed bimodal peaks in the area further confirm that there is domestic fuel combustion however, Wernecke *et al*. (2015) and Venter *et al.* (2012) argue that such could also be reflective of vehicular emissions. A detailed discussion of the attribution of the diurnal double peaks to residential use of dirty energy sources is further elaborated on in section 4.1.2.2 below. On all the weekdays, the morning spikes for ambient $PM_{2.5}$ and PM_{10} concentrations were lower by almost a magnitude of 2 in comparison to those recorded in the evening hours (Figure 4-4). This is indicative that between 15:00-21:00, irrespective of the day of the week, residents in Lebohang are mostly engaged in local timing activities (e.g. burning coal for cooking, thermal comfort, socialising and boiling water to bath) than they are in 03:00-09:00. The reason being, during the morning hours some residents in the community, particularly unemployed and school dropouts, are still asleep; meanwhile, in the evening, almost every member is awake.

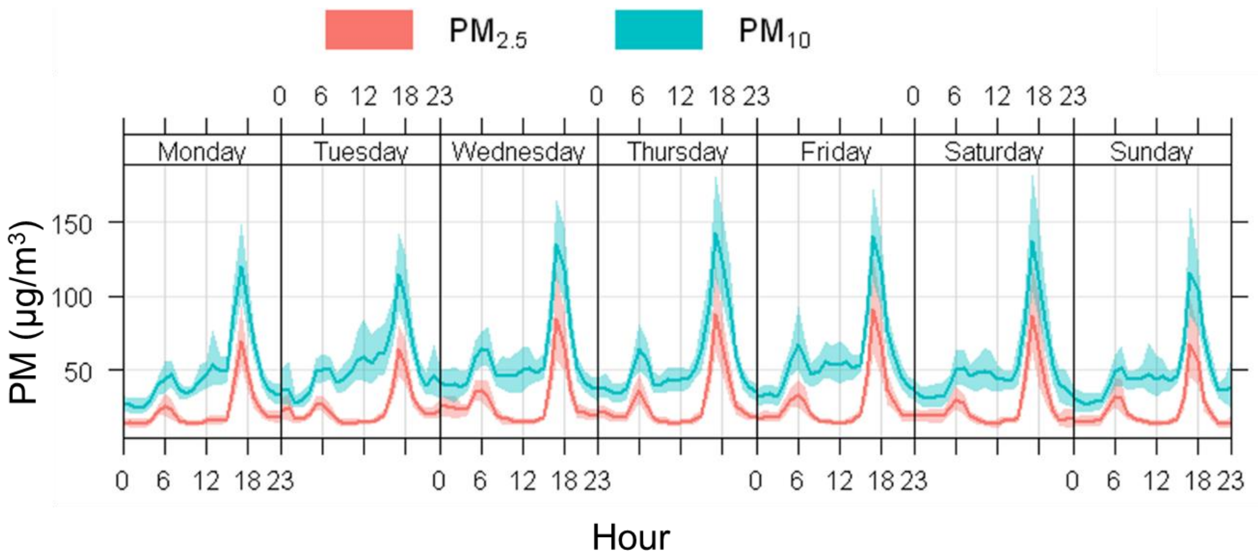

Figure 4-4: Diurnal distribution of ambient $PM_{2.5}$ and PM_{10} concentrations ($\mu g.m^{-3}$) during the days of the week in Lebohang.

During the days of the week, the highest PM_{10} evening peak of 148 $\mu g.m^{-3}$ was measured on Thursday and the lowest of 110 $\mu g.m^{-3}$ was recorded on Sunday. However, this was expected since other previous studies have reported similar results within low income residential areas (Feig *et al.*, 2016). Considering that during weekdays people typically have more obligations (e.g., children need to go to school, and adults have to be at work) than on weekends, it makes sense why PM_{10} is more problematic at this time of the week. Surprisingly, $PM_{2.5}$ displayed a reserved situation as it showed both maximum (148 $\mu g.m^{-3}$) and minimum (110 $\mu g.m^{-3}$) late afternoon concentrations during the weekdays (on Friday and Tuesday, respectively). These results are in contrast with those that were reported by Feig *et al.* (2016) which showed ambient $PM_{2.5}$ in Thabazimbi, Lephalale and Makopane displayed high concentrations during weekdays, followed by low levels at the weekend. To explain this difference, further research that includes, for instance, the understanding of the weekly cycles of weather conditions in the aforementioned areas and Lebohang is required.

4.1.2.2 Seasonal diurnal ambient PM concentrations

Spring, summer, autumn, and summer respectively start in September-November; December-February; March-May; and June-August in South Africa (Adesina *et al*., 2020). The seasonal diurnal plots presented in Figure 4-5 were generated using the mean hourly ambient PM data that corresponded with these months that constitute each respective season in South Africa. Regardless of the season, seasonal diurnal ambient PM levels in Lebohang showed double peaks, with the first one in the morning (mostly between 03:00-09:00) and the second one in the evening (mostly from 14:00-21:00). The occurrence of these spikes matches with the time of the day during which the events of domestic combustion of coal within low income communities typically takes place. A recent study by Nkosi *et al*. (2021), for instance, shows that the time when residents in typical low income communities start to burn is between 05:00-08:00 in the morning and from 16:00-17:30. Another study also found that in Tembisa informal settlement, during the morning hours, people usually burn coal to cook as well as to boil water to prepare either themselves to go to work or their children to go to school (Makonese *et al.,* 2016). In contrast, burning events during the evening hours have been previously attributed to several reasons, ranging from the need for residents to prepare food, keep warm from cold temperatures, or even socialize (Wernecke, 2018). Therefore, the double peaks reported in this research could likely be reflective of the local fuel burning behaviour of residents in the community.

Notwithstanding, despite domestic fuel burning activities, the morning PM peaks measured during all the sampled seasons in Lebohang can also be reflective of the atmospheric conditions at this time of day. There are a series of events that often occurs at the lower level of the atmosphere and well justifies why PM concentrations, irrespective of the season, peak mostly between 03:00 to 09:00 (Molepo *et al*., 2019). To mention a few, unlike the strong prevailing winds in the afternoon, the wind speed during the morning is poor, resulting in reduced vertical mixing and dispersion of pollutants. Consequently, the early hours of the day, experienced particularly within low-income settings, are often associated with high ambient $PM_{2.5}$ and PM_{10} levels (Molepo *et al*., 2019). Additionally, decreased atmospheric diffusion of PM in the morning is also enhanced by the inversion layer that forms due to the lower ambient temperature experienced at this time of the day. Another explanation can be the depth of the boundary layer in the morning, which tends to be shallow and prevents emitted $PM_{2.5}$ and PM_{10} concentrations from escaping into the atmosphere. Barmpadimos *et al.* (2011) found, for instance, the depth of the boundary layer is negatively correlated with particulate pollution, suggesting that if one of these variables increases the other decreases and vice versa. All the aforementioned factors combined could be other reasons, apart from residential time activities that justify high $PM_{2.5}$ and PM_{10} concentrations measured in the morning in Lebohang.

Similar to the weekday diurnal PM concentrations seen in Figure 4-4, the daytime seasonal plots for each respective season, were characterised by high evening and lower morning concentrations for both PM_{10} and $PM_{2.5}$ (Figure 4-5). These results agree with the observations that were made in Northern Khayelisha by Molepo *et al*. (2019) who reported that this low income community is also characterised by low morning and high evening PM concentrations. The authors attributed high night PM concentrations to the substantial domestic coal combustion, that occurs in the evenings since when at this time most households burn coal to cook dinner. It is known that the kind of food that inhabitants in impoverished communities typically prepare has different characteristics (Nkosi *et al*., 2021; Makonese *et al*., 2016; Balmer, 2007). For example, some food takes longer to cook and requires more fuel as well as heat; meanwhile, others are quick, small, and require less energy thus, resulting in fewer burning hours. Due to the limited time that residents have in the morning to prepare themselves for work and their children for school, they often cook easy and fast breakfasts (e.g. eggs and soft porridge) to save time (Makonese *et al*., 2016). The reverse, however, is the case in the evening as people have sufficient time to prepare their preferred traditional foods (such as samp and tripe), which typically takes longer to cook. Therefore, this explains why, despite different seasons, ambient $PM_{2.5}$ and PM_{10} loads in Lebohang were characterised by low morning and high evening concentrations during the day (Figure 4-5).

Although throughout all the sampled seasons, the diurnal plots of ambient particulate pollution displayed two peaks, a seasonal variation of these PM concentrations was seen in Lebohang (Figure 4-5). For instance, the winter (June-August 2018) season displayed the highest and more pronounced bimodal PM concentrations and to a lesser extent, followed by summer (December 2017-February 2018), autumn (March-May 2018), and spring (October 2017-November 2017). During the winter season, the first peak of 50 $\mu g.m^{-3}$ for $PM_{2.5}$ and 80 $\mu g.m^{-3}$ for PM_{10} occurred between 03:00-09:00, peaking respectively at 06:00 and 07:00 for $PM_{2.5}$ and PM_{10} concentrations. At this time of year, the early hours of the day in the area had ambient $PM_{2.5}$ and PM_{10} concentrations that prevailed for almost 7 hours, which were also beyond the daily average NAAQS (see Figure 4-5 (d)). When conducting a seasonal comparison, the $PM_{2.5}$ winter morning peak was found to be greater by a factor of almost 3 than that of summer and spring; and 2 times higher than the autumn morning spike. Similarly, PM_{10} had the maximum winter morning peak, which was higher by a magnitude that ranged between 2 to 3 in comparison to those recorded at the exact time of the day, during other seasons.

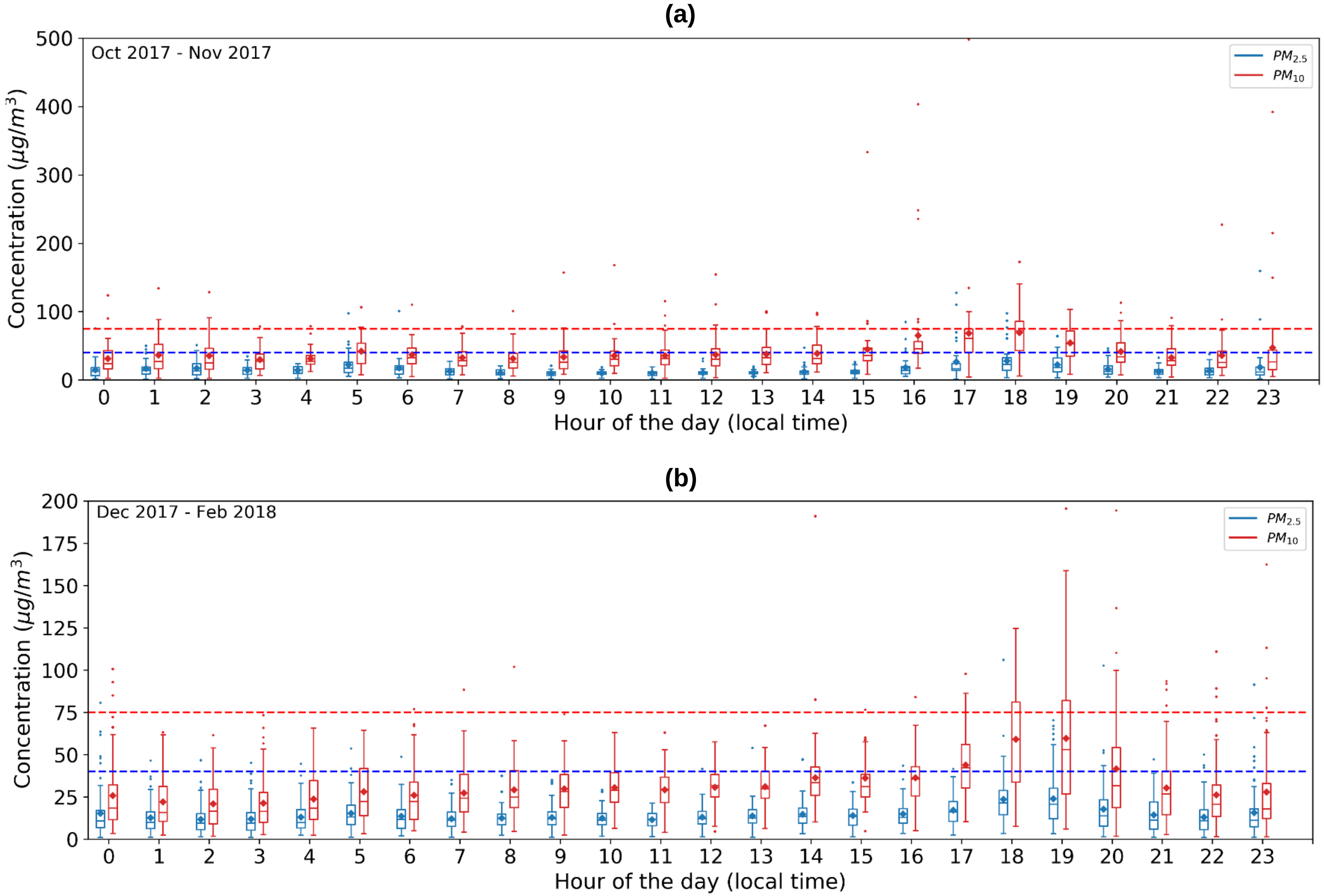

Figure 4-5: Diurnal distribution of ambient PM concentrations measured during spring (October 2017 - November 2017) (a) and summer (December 2017 - February 2018) (b). The representation of the box and whiskers is similar to that of the plots shown in Figure 4-3. The red and blue dotted lines inside the plots are 75 µg.m^{-3} (for PM_{10}) and 40 µg.m^{-3} (for $PM_{2.5}$) 24h average National Ambient Air Quality Standards, respectively.

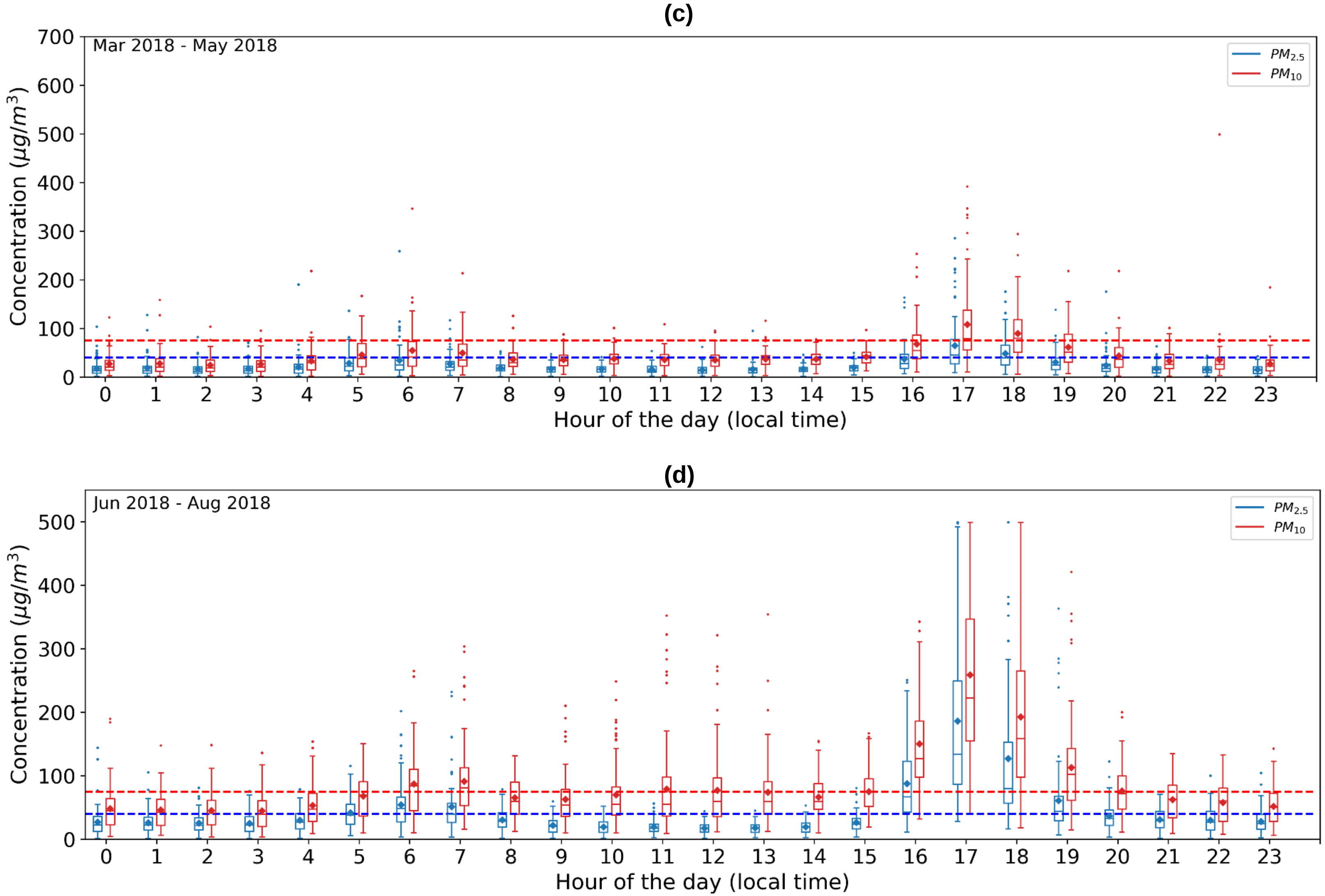

Figure 4-5: Diurnal distribution of ambient PM concentrations measured during autumn (March 2018 – May 2018) (c) and winter (June 2018 - August 2018) (d). The representation of the box and whiskers is similar to that of the plots shown in Figure 4-5 above.

In terms of the evening peaks, just as the morning spikes reported in the above paragraph, the highest concentrations for $PM_{2.5}$ and PM_{10} were both observed during the winter period. During this time of year, the late afternoon peak of 200 µg.m^{-3} for $PM_{2.5}$ and 250 µg.m^{-3} for PM_{10} occurred between 14:00-21:00 thus, lasting for approximately 8 hours. The winter duration (8 hours) of the night particulate loading was the longest in comparison to those observed during summer, autumn, and spring. This is indicative that the starting and ending hours of domestic combustion events in Lebohang change with the seasons hence, showing a considerable variance between winter and other seasons.

The maximum evening concentrations of $PM_{2.5}$ and PM_{10} measured during the winter period were both above the daily average NAAQS of 40 µg.m^{-3} for $PM_{2.5}$ and 75 µg.m^{-3} for PM_{10}. This was expected since it is known that typically, in a township setting, the concentration peaks that occur during the winter evenings, are often likely to be beyond 24h mean NAAQS (Moletsane *et al*., 2021). The incompliance during the late hours of the winter afternoons is due to an increment, either on the household or community scale, of activities that emit PM. In comparison to those measured during other seasons, the evening winter PM concentrations on average were higher by the magnitude that ranged between 3 to 8 for $PM_{2.5}$ and from 2 to 4 for PM_{10}.

4.1.3 Difference between PM concentrations in Lebohang and those reported in other urban areas

To analyse the variance of ambient PM, a comparative analysis of the particulate concentrations found in Lebohang and those reported in previous studies was done (Table 4-2). This assessment was aimed to determine whether or not, are $PM_{2.5}$ and PM_{10} values observed in Lebohang comparable to the particulate levels found in other urban low income communities. Areas that were selected for this analysis are typical hotspots of particulate pollution. For instance, such places have coal-reliant households, are spatially located close to industries, are densely populated, have unpaved roads, and are polluted by traffic emissions. Although there are local sources including residential coal use in Lebohang, $PM_{2.5}$ and PM_{10} concentrations measured in this area were relatively lower than those previously reported in international studies (see Table 4-2). For instance, in comparison to $PM_{2.5}$ concentrations recorded at Delhi, Baoshan District, Pluto District, Bafoussam, Bamenda, and Yaounde, Lebohang recorded values that were lower by a magnitude of 5, 4, 2, 3, 5, and 2, respectively.

Similarly, PM_{10} concentrations measured in Lebohang (51 ± 30 µg.m^{-3}) were lower by a factor that ranged between 2 to 4 when compared to those recorded in Delhi (222 ± 21 µg.m^{-3}), Baoshan District (149 ± 22 µg.m^{-3}), Pluto District (97 ± 44 µg.m^{-3}), Bafoussam (105 ± 29 µg.m^{-3}) and Bamenda (141 ± 107 µg.m^{-3}). The observed variance might be due to factors such as population size, meteorology and socioeconomic status, geography and PM sampling period and methods, which varied from one study area to another. In contrast, the PM_{10} concentrations measured in this study (51 ± 30 µg.m^{-3}) were somewhat the same as those recorded at Yaounde (65 ± 21 µg.m^{-3}). It is not clear what might have caused the comparable PM_{10} values recorded in Lebohang and Yaounde hence, to explain the observed similarities seen between these communities, further research is required.

Table 4-2: Comparative analysis of the PM concentration values found in Lebohang and those previously reported in other urban low income settlements

Study location	**Country**	**$PM_{2.5}$ (µg.m^{-3})**	**PM_{10} (µg.m^{-3})**	**Sampling duration**	**Authors**
Delhi	India	129.8 ± 103.4	222 ± 142.0	01/09/2010-23/08/2012	Tiwari *et al*. (2014)
Lebohang	South Africa	25 ± 17	51 ± 30	01/10/2017-31/08/2018	This study
Baoshan District	China	103.07	149.22	01/07/2009-31/10/2010	Wang *et al*. (2013)
Pluto District	China	62.25	97.44	01/07/2009-31/10/2010	Wang *et al*. (2013)
Bafoussam	Cameroon	67 ± 14	105 ± 29	08/01/12-17/01/12	Antonel & Chowdhury, (2014)
Bamenda	Cameroon	132 ± 64	141 ± 107	22/01/2012-05/02/2012	Antonel & Chowdhury, (2014)
Yaounde	Cameroon	49 ± 12	65 ± 21	09/02/2012-19/02/2012	Antonel & Chowdhury, (2014)

Chapter 4 has provided a detailed discussion of the spatial and temporal variation of ambient $PM_{2.5}$ and PM_{10} concentrations in Lebohang. It specifically covered the first objective of this study, which was outlined in Chapter 1 section 1.2. Throughout this chapter, it was argued that outdoor particulate pollution tends to be more problematic during the cold seasons due to increased domestic burning of dirty solid fuels in the area. Furthermore, it was also revealed that the weeks days in Lebohang are associated with health-threatening PM levels in comparison to the weekends. Also, the chapter showed that irrespective of the seasons, Lebohang experiences peaks of ambient particulate concentrations in the morning and evening due to local sources. Chapter 4 further showed that although PM levels in Lebohang are often beyond NAAQS compared to other urban areas, the community is characterised by relatively lower particulate pollution. The subsequent chapter below will address the second objective of this study, which is to "Assess the indoor PM concentrations in Lebohang low income settlement".

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CHAPTER 5: INDOOR PM_4 CONCENTRATIONS IN LEBOHANG

Chapter 5 presents the indoor PM_4 concentrations measured in the households sampled in Lebohang low income settlement. The chapter explores how PM_4 levels vary across different household types during different seasons. Furthermore, chapter 4 make an attempt to also assess the impacts of fuel choice in Lebohang by conducting the spatial comparison of indoor particulate concentrations between households based on their respective energy carrier.

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Although indoor PM_4 was monitored in 22 households in Lebohang, only houses with at least 5 day's worth of 24h PM_4 measurements, with each day consisting of ≥50% data points, were considered for analysis. As a result, only 20 homes, excluding H11 and H12, met this criterion and were included in the assessment of indoor PM_4 concentrations in Lebohang. Table 5-1 is a statistical summary of the different types of homes sampled in the community and their respective PM_4 concentrations measured throughout the sampling period and during different seasons. As seen in Table 5-1, 10 (50%) homes included in this analysis were formally structured, and another 10 (50%) were informally designed dwellings. Of these 10 formal homes, 5 mainly used coal (CFH) for cooking and space heating, while the remaining 5 performed these activities using LPG stoves and electricity as their primary energy source (NFH). Like formal homes, the 10 informal, structured houses comprised 50% of coal-reliant dwellings (CIH) and 50% of houses that depended primarily on LPG stoves and electricity.

The 24h indoor PM_4 concentrations of all houses enrolled in the study, regardless of their household type and primary energy carrier ranged between 24 $\mu g.m^{-3}$ and 482 $\mu g.m^{-3}$ with a mean value of 107 ± 62 $\mu g.m^{-3}$ and a median of 98 $\mu g.m^{-3}$. Based on the structural types, on average, formal homes (109 ± 67 $\mu g.m^{-3}$) and informally structured households (105 ± 53 $\mu g.m^{-3}$) sampled in Lebohang measured similar indoor PM_4 concentrations throughout the entire sampling period (Table 5-1). This was unexpected as formal households are often better insulated and sealed than informal ones, as such, indoor concentrations are typically highly trapped within formal than within informal dwellings (Wernecke, 2018). The results of this study disagree with the observations made by Savage (2007) who found higher concentrations within formal as opposed to informal homes of Msunduzi Municipality in KwaZulu-Natal. Furthermore, they also contrast with those made in eThekwini Municipality by Jafta *et al*. (2017), who found high PM concentrations within informal homes rather than within formal dwellings. This is thus, indicative that the different types of household structures have an inconsistent pattern of effects on HAP as a result of the difference in certain specific factors. These include, for instance, the building materials, geography, ventilation practices and the time of the year during which the of sampling indoor particulate concentrations was conducted (Sidhu *et al*., 2017).

Table 5-1: Household characteristics of sampled homes in Lebohang and a summary of their 24h mean indoor measurements.

	Sampled homes (n = 20) & (%)	**Mean (Std)**			**Ratio**
Household Attributes		**Full Period**	**Autumn**	**Winter**	**Winter/Autumn**
House Design	Formal 10(50)	109 ± 67	113 ± 81	105 ± 52	1
	Informal 10(50)	105 ± 53	64 ± 24	125 ± 52	2
House Group	CFH 5(25)	132 ± 83	139 ± 105	124 ± 52	1
	NFH 5(25)	82 ± 37	86 ± 41	79 ± 34	1
	CIH 5(25)	96 ± 39	65 ± 21	114 ± 36	2
	NIH 5(25)	97 ± 52	63 ± 25	111 ± 54	2

n – stands for the total number of sampled households

Overall, the worst polluted season for informal dwellings was represented by winter with a mean value of 125 ± 52 µg.m^{-3}, which was 2 times higher than the concentration measured during autumn (64 ± 24 µg.m^{-3}) (Table 5-1). The winter/autumn ratio of 2 calculated for informal homes in Lebohang, further alluded that the winter period was associated with higher indoor PM_4 concentrations than autumn (Table 5-1). Moreover, the mean indoor PM_4 measured during autumn at the informal homes, in comparison to that sampled at formal households during the same season, was lower by almost a factor of 2. The observed difference can be attributed to structural variance between formal and informal households as well as the impacts of non-household smoking, particularly within informal homes (see table 3-3).

In comparison to informal dwellings, formal houses are often well sealed thus, trap more emissions indoors, especially during the coal burning session, particularly, when all windows and doors are closed (Savage, 2007; Wernecke, 2018). This could be a possible explanation as to why on average, informal households (64 ± 24 µg.m^{-3}) had fewer concentrations than formal homes (113 ± 81 µg.m^{-3}) during autumn. Another possible reason is that indoor smoking might have resulted in increased indoor concentrations within formal homes. Considering that a substantial number of sampled formal homes in Lebohang were practicing indoor smoking of cigarettes (see Table 3-3), it is likely such behaviour might have resulted in elevated indoor PM_4 concentrations. Although the number of times residents within formal homes smoked inside their houses was not recorded, there is evidence supporting that indoor smoking degrades air quality at least by 10% or even 60% when practiced by 2 smokers in the same house (Li *et al*., 2016; Jafta *et al*., 2017).

During winter, the mean indoor PM_4 concentration sampled at informal houses was 1.2 times higher than that recorded at formal houses (refer to table 5-1) thus, contrasting with the autumn observations. The difference can be attributed to the increment in the burning patterns of the informal household for the need for thermal comfort due to cold temperatures associated with winter. Although during winter both formal and informal houses burn more coal, the weak natural ventilation rate and small volume of informal houses, in comparison to formal houses, trap more concentrations from escaping into the ambient environment. This claim is supported by the results of Jafta *et al*. (2017), who found that houses with 2 or fewer rooms were associated with higher particulate pollution than those with 4 or more rooms. Generally, it is known that within households that have higher air exchange rates, fewer indoor particulate concentrations are expected and vice versa (Zhang *et al*., 2021b). Following this logic then, it is likely that the weak natural ventilation rate, facilitated by the small volume of informal households, could have increased particulate loads in informal homes sampled in Lebohang.

Regarding formally designed houses, surprisingly, a different picture than that seen for informal houses was displayed. The indoor PM_4 concentrations for formal households peaked during autumn (113 ± 81 $\mu g.m^{-3}$) as opposed to the winter period (105 ± 52 $\mu g.m^{-3}$) (Table 5-1). This observation disagrees with the previous results of (Language 2020; Qhekwana 2019; Adesina *et al*., 2020; Letsholo, 2020) which showed that within low income communities, cold periods are typically associated with substantial indoor PM_4 concentrations due to increased household burning of coal. For this study, based on the data that was collected, the exact reasons why PM_4 concentrations were high during autumn within formal houses are inexplicable and thus, require further research. However, given that local ambient sources such as waste burning, vehicular emissions and dust, are known to typically have more influence on indoor PM concentrations during the warm period due to increased residential ventilation behaviour (Semmens *et al*., 2015; Zhang *et al*., 2021a); it is suspected that this could have also been the case for formal houses in Lebohang.

5.1 Cross-comparative analysis of different household classes

The daily mean PM_4 concentrations sampled at CFH, NFH, CIH and NIH during the entire sampling period (05 March to 30 July 2018), autumn (05 March to 02 April 2018) and winter (18 June to 30 July 2018) period are depicted in Figure 5-1. A considerable variation of indoor PM_4 concentrations was seen between the average value of CFH dwellings and that of NIH households during the entire sampling campaign, autumn and winter seasons. Interestingly, NFH dwellings on average recorded indoor PM_4 concentrations that were lower by almost a factor of 2 in comparison to those sampled within the CFH houses. This phenomenon was consistent during both autumn and winter seasons, even during the entire study sampling campaign (Figure 5-1).

The results presented in the above paragraph are in line with the theory of the energy ladder, which suggests that electricity use as the main energy source is more cleaner and efficient than coal (Annecke, 1999; Scorgie *et al.*, 2003). Reduced emissions at NFH houses are more likely to be attributed to electricity usage, thus, indicating that the use of cleaner energy sources within formal households has the potential to improve indoor air quality. Contrary, the mean PM_4 concentrations sampled in CIH dwellings vs. that recorded at NIH homes were almost the same during the entire sampling period and both the autumn and winter seasons (Figure 5-1). This was surprising considering that the kitchen area of households that practice solid fuel burning are typically exposed to greater particulate levels than those that use cleaner energy sources (Begum *et al*., 2009). These results thus highlight the complexity of solving HAP, particularly within informal homes as the use of cleaner fuel sources alone, generally does not seem to yield expected results.

For each household group, classified by the type of fuel used and household structural design, indoor PM_4 concentrations displayed considerable seasonal differences (Figure 5-1). However, the seasonality was more pronounced for CIH and NIH categories, and to a lesser extent, followed by both CFH and NFH household classes. Unexpectedly, the highest 24h mean for CFH (139 ± 105 $\mu g.m^{-3}$) and NFH (86 ± 41 $\mu g.m^{-3}$) were observed in autumn; meanwhile, winter recorded the least concentrations of 124 ± 52 $\mu g.m^{-3}$ and 79 ± 34 $\mu g.m^{-3}$ for CFH and NFH, respectively. Typically, indoor concentrations peak in winter within low income communities due to increased combustion events during this time of year as a result of low ambient temperatures (Lourens *et al*., 2011; Venter & Lourens, 2020; Adesina *et al*., 2020; Feig *et al*., 2016; Wang *et a*l., 2013; Qhekwana, 2019). However, it is evident that for CFH and NFH houses in Lebohang, indoor PM_4 concentrations were, on average, low in winter and high in autumn. Furthermore, judging by the recorded high standard deviation, it is also clear that there was a high variance of indoor PM_4 concentrations recorded, particularly amongst CFH dwellings. This variance can be attributed to the factors that differ from one CFH home to another, such as cooking habits, burning routines, the porosity of walls, and ventilation behaviour.

High overall autumn indoor concentrations seen at CFH and NFH are thought to be reflective of the outdoor sources that might have been infiltrated into formal houses with high frequency during this time of the year. As opposed to winter, it is well documented that residents within low income communities tend to often open their household windows and doors during warm than cold periods due to increased ambient temperatures (Alnes *et al*., 2014; Semmens *et al*., 2015; Savage, 2007). For instance, a study by Li *et al*. (2016) found that during the cold season residents in Lanzhou, China, opened their windows for 7.7 hours meanwhile in the warm period, they opened their windows for 15.8 hours a day. In another study done by Weaver *et al*. (2017) in a low income community of Dhaka in Bangladesh, it was similarly found that increased ventilation practices of up to ≥13 hours of opening doors and windows, were associated with a hot period. During autumn, it is thus, more likely that the increased ventilation within formal homes sampled in Lebohang could have allowed for more ambient PM concentrations to penetrate the

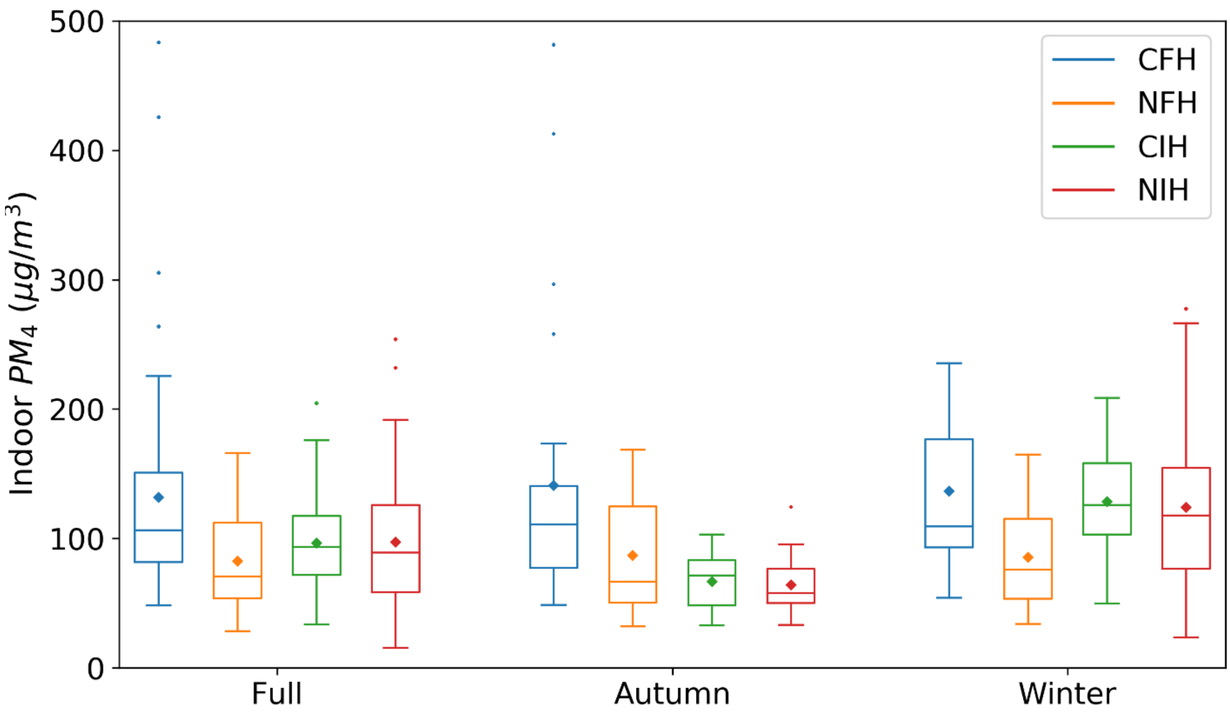

Figure 5-1: 24h mean PM_4 concentrations sampled during the full study period (05 March 2018 – 30 July 2018), autumn (05 March 2018 – 02 April 2018) and winter (18 June 2018 – 30 July 2018) seasons at CFH, NFH, CIH, and NIH. Box and whisker plots show 25th, 50th and 99th percentiles; mean (diamonds inside the boxes); and outliers (stars beyond the whiskers).

indoor environments at a high magnitude. This well justifies why elevated concentrations were more prevalent during autumn than in winter, particularly within formal homes as they have proper and adequate ventilation structures in comparison to informal homes (Jafta *et al*., 2017; Zimmermann *et al.*, 2017).

In contrast to CFH and NFH household categories, CIH and NIH homes showed a typical seasonal trend characterised by high indoor concentrations in winter rather than in autumn (Figure 5-1). NIH households and CIH homes on average recorded almost double the autumn indoor PM_4 concentration measured during the winter season. High winter indoor levels particularly at CIH homes, can likely be attributed to the increased coal burning hours at these households and the poor atmospheric dispersion, which is prevalent during this season. At NIH dwellings, such substantial winter PM_4 levels are arguably reflective of the pronounced ambient emissions produced by other surrounding solid fuel-reliant homes into the atmosphere (Langerman *et al*., 2018). Zhang and Smith (2007) refer to this phenomenon as the "*neighbourhood pollution effect*" and argue that it predominantly occurs in winter and has a reductive effect on the air quality of houses that use clean energy. This effect was likely experienced within NIH dwellings in Lebohang, hence such households generally did not show low concentrations in winter, despite that they were not burning coal.

5.2 Spatial and temporal variation of household PM_4

Indoor PM_4 sampled in Lebohang was analysed temporally and spatially to determine how it was distributed geographically amongst different individual houses based on their structural type and energy sources. The spatial comparison of indoor particulate concentrations between households based on their respective energy carrier was important to assess the impacts of fuel choice in Lebohang. To evaluate these effects within different structurally built homes, an inter-household variation analysis between coal and non-coal-reliant dwellings was conducted respectively for formal and informal houses. For each household structural design, PM_4 measurements of each house were averaged into 24h concentrations and compared against each one another.

5.2.1 Inter-household variation of PM_4 concentrations

5.2.1.1 Spatial and seasonal variation of daily mean PM_4 within formal homes

Figure 5-2 shows the inter-household variation of the mean daily indoor PM_4 concentrations for CFH and NIH sampled during autumn and the winter period in Lebohang low income settlement. Moreover, a summary of the daily mean indoor PM_4 statistics for each of the individual formal households sampled in the community is present in Table 5-2 below. Generally, regardless of the season, indoor mean 24h PM_4 concentrations had a considerable spatial difference amongst individual formal homes monitored across the community. Similarly, the house-to-house variance was also seen, even amongst the individual homes of the same energy type classification, but this was more intense across CFH dwellings, particularly in autumn. The PM_4 concentrations for CFH and NFH categories respectively ranged from 49 - 482 µg.m^{-3} and between 32 and 168 µg.m^{-3} during autumn, and from 34 - 159 µg.m^{-3} and between 54 and 236 µg.m^{-3} in winter (Table 5-2).

During autumn, the 99th percentile of the 24h mean concentration at H01, H02, H05, H06 H09, and H10 were 477 µg.m^{-3}, 56 µg.m^{-3}, 173 µg.m^{-3}, 123 µg.m^{-3}, 162 µg.m^{-3}, and 168 µg.m^{-3}, respectively. The highest autumn daily mean of 253 ± 136 µg.m^{-3} was recorded at H01, which was coal-reliant, and in a downward order, followed by H05, H10, H09, H06 and H02. When compared to the mean concentrations measured within all the NFH dwellings during autumn, the PM_4 value at H01 was 2 times greater. These results thus suggest that within some of the coal formal homes in Lebohang, particulate pollution tends to be problematic even during the warm seasons when residents are typically subjected to fewer burning hours and high ventilation practices (Adesina *et al*., 2020; Feig *et al*., 2016; Lourens *et al*., 2011; Weaver *et al*., 2017; Wang *et a*l., 2013; Qhekwana, 2019; Semmens *et al*., 2015; Savage, 2007). Conversely, the lowest autumn 24h mean indoor PM_4 of 44 ± 9 µg.m^{-3} was measured at H02 and was lower by a factor that ranged between 2 to 6 in comparison to those sampled within other formal homes in autumn. The low autumn mean indoor concentration at H02 is likely due to the impacts of non-coal reliance practiced at this particular house thus, reflecting the air quality benefits of using clean energy sources.

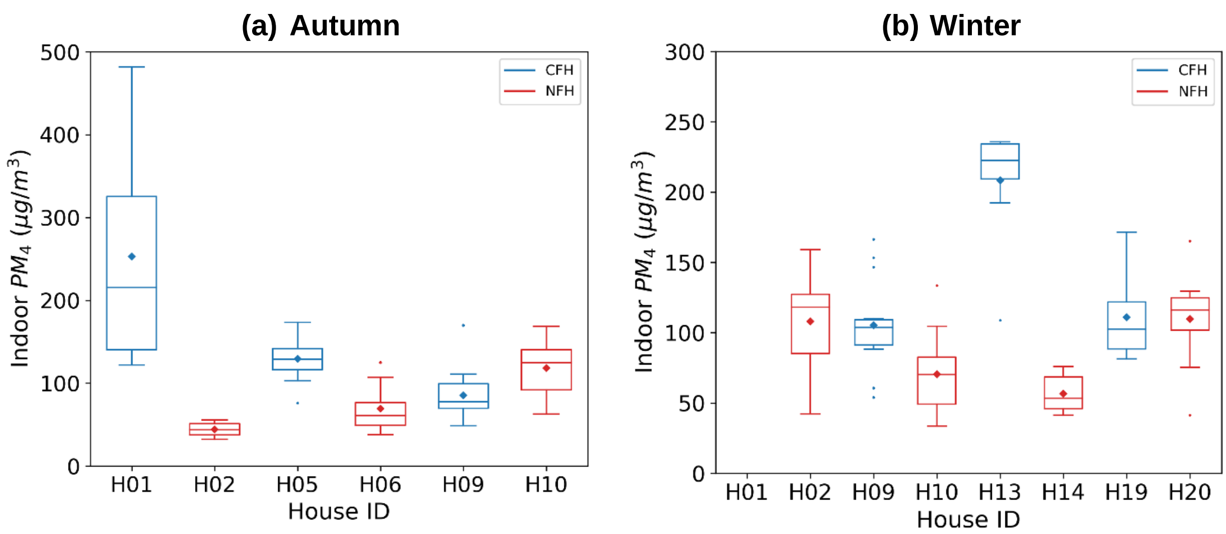

Figure 5-2: Inter-house variance of the 24h mean PM_4 concentrations measured within formally structured homes for CFH and NFH in Lebohang during autumn (a) and winter (b) seasons. Box and whisker plots

show 25th, 50th and 99th percentiles; mean (blue diamond inside the box); and outliers (blue stars beyond the whiskers). Of note: the y scale of figures (a) and (b) are not the same.

In winter, the 99th percentile of the winter daily mean PM_4 concentrations measured in formal houses flowed in the following descending order: H13 > H19 > H09 > H20 > H02 > H10 > H14. Amongst all these houses, the highest daily mean winter indoor PM_4 concentration of 236 ± 43 $\mu g.m^{-3}$ was recorded at H13, which was reliant on coal for heating and cooking. When compared to other formal homes sampled during winter in Lebohang, the 24h average PM_4 recorded at H13 was greater by an order magnitude that ranged between approximately 2 and 3. In contrast, the lowest winter daily mean PM_4 of 57 ± 13 $\mu g.m^{-3}$ was recorded at house H14, which was non-coal reliant and was almost 4 times lower than the indoor concentration measured at H13. This is indicative that despite indoor PM_4 increments typically seen in low income areas in winter, some households in Lebohang, particularly those relying on cleaner fuels, remain subjected to air that is 4 folds cleaner than that experienced in other coal-reliant homes in the area.

Table 5-2: Descriptive statistics of the daily mean indoor PM_4 concentrations ($\mu g.m^{-3}$) sampled within formal households in Lebohang low income settlement.

Autumn indoor PM_4 concentrations							
House ID	**Count**	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**
H01	8	122	253	136	216	477	482
H02	8	32	44	9	44	56	56
H05	8	76	129	33	129	173	173
H06	8	38	69	31	61	123	125
H09	15	49	85	31	77	162	170
H10	15	63	118	35	125	168	168
Winter indoor PM_4 concentrations							
House ID	**Count**	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**
H02	8	42	108	36	118	157	159
H09	15	54	105	31	194	164	166
H10	15	34	70	27	70	129	133
H13	8	109	208	43	222	236	236
H14	8	41	57	13	53	75	76
H19	7	81	111	32	103	169	172
H20	8	41	110	37	116	163	165

Notwithstanding, the air quality benefits of using cleaner energy sources were not always enjoyed by all non-coal-reliant formal homes in Lebohang. For instance, the winter 24h mean indoor PM_4 values at H02 (108 ± 36 $\mu g.m^{-3}$), H09 (105 ± 31 $\mu g.m^{-3}$), H19 (111 ± 32 $\mu g.m^{-3}$), and H20 (119 ± 37 $\mu g.m^{-3}$) showed less inter household variance. This was despite that H02 and H20 were both noncoal reliant, while H09 and H19 on the other hand were both coal-reliant households. It is, thus, clear, based on the results presented here, that the use of electricity and LPG stoves, particularly in some formal houses in Lebohang, does not always guarantee improved air quality, especially in winter. However, it is argued that this is likely due to the intensity of the impacts of the "*neighbourhood pollution effect"*, experienced especially in the winter seasons.

5.2.1.1.1 Statistical analysis of PM_4 variation measured between CFH and NFH

The effects of fuel choice in Lebohang were further assessed by determining the significance of the inter-household variation, especially between NFH and CFH by undertaking an unpaired t-test. Lesholo (2021) argues that, by only considering the mean values of sampled households as it was done in section 5.2.1.1, and comparing them against one another, such an analysis can lead to generalised conclusions. The reason is, it is not always the case that the two mean values, which seem virtually different, when statistically compared, can be significantly variable from one another. Hence, conducting a t-test in this regard, was important to reduce the generic statements, which could have been possibly made in section 5.2.1.1. When conducting the t-test, the mean indoor PM_4 concentrations measured within formal noncoal-reliant dwellings and coal-reliant households in the Lebohang were used and compared against one another. The variance of the mean indoor PM_4 concentrations sampled in different homes was judged as significant when the p-value was <0.05, otherwise, the opposite was concluded. Table 5-3 shows the matrix of the p values obtained from the statistical test of the mean comparison of different monitored formal dwellings during autumn and winter, respectively.

By separately comparing individual homes of CFH and those of NFH dwellings, most households showed a significant inter-household variation ($p<0.05$), especially in autumn. Similarly, during autumn, all the individual homes of the same category, concerning the fuel choice, also showed a statistically significant inter-variation ($p<0.05$). The difference in mean PM_4 amongst the houses that uses the same energy sources can be attributed to several factors which vary on the household level, across Lebohang. These could likely include, time-activity patterns, occupants' size, economic statuses, ventilation behaviour, and the frequency of coal-burned fuel and burning duration, particularly for CFH dwellings. Nevertheless, there were other instances during autumn, where insignificant variance ($p>0.05$) was observed, particularly when comparing the average indoor PM_4 of individual homes of CFH with those of the NFH category. This was the case in the comparison between H05 (129 ± 33 $\mu g.m^{-3}$) and H10 (118 ± 35 $\mu g.m^{-3}$); and that of H06 (69 ± 31 $\mu g.m^{-3}$) and H09 (85 ± 31 $\mu g.m^{-3}$) (refer to Table 5-3 below). Occurrences of these events, however, were seldom thus, indicating that, for a vast majority of formal coal-reliant homes in Lebohang, electricity and LPG stoves showed positive results in autumn.

Overall, during the winter season, when respectively comparing coal-reliant houses against each noncoal-reliant house, a vast majority of the compared houses had a significant spatial inter variability ($p<0.05$) (Table 5-3). This was specifically the case, for instance, on the comparison between H09 (105 ± 31 µg.m^{-3}) vs. H10 (70 ± 27 µg.m^{-3}) and H14 (57 ± 13 µg.m^{-3}); and H13 (208 ± 43 µg.m^{-3}) vs. H14 (57 ± 13 µg.m^{-3}) and H20 (110 ± 37 µg.m^{-3}). Although during winter, many formal homes, especially noncoal-reliant ones sampled in Lebohang showed significantly lower concentrations in comparison to those which relied on coal, others did not. The comparative analysis of PM_4 concentrations between some individual CFH and NFH houses, for example, that of H09 vs. H02; H19 vs. H02; and H19. vs. H20, sampled in Lebohang, were insignificantly variable ($p>0.05$). However, just as in the autumn observations, the instances where concentrations of noncoal-reliant homes showed no statistical difference to those of coal-reliant homes were also seldom even during winter. Therefore, suggesting that efforts that are specifically aimed at converting formal households from coal-reliance to electricity and LPG stoves usage are likely to reduce indoor PM_4 pollution in Lebohang.

Table 5-3: Matrix table showing the p values of unpaired t-test analysis of the autumn and winter daily mean PM_4 concentrations sampled within formal households in Lebohang.

Autumn						
	H01	**H02**	**H05**	**H06**	**H09**	**H10**
H01	1	0.00	0.02	0.00	0.00	0.00
H02		1	5.00e-06	0.04	0.00	8.88e-06
H05			1	0.00	0.00	0.46
H06				1	0.24	0.00
H09					1	0.01
H10						1

Winter							
	H02	**H09**	**H10**	**H13**	**H14**	**H19**	**H20**
H02	1	0.85	0.01	0.00	0.00	0.86	0.92
H09		1	0.00	1.30e-06	0.00	0.69	0.76
H10			1	4.89e-09	0.19	0.00	0.00
H13				1	1.53e-07	0.00	0.00
H14					1	0.00	0.00
H19						1	0.94
H20							1

Blue cells = CFH
Red cells = NFH
Yellow cells = Significantly different PM_4 mean values ($p <0.05$)

5.2.1.2 Spatial and seasonal variation of daily mean PM_4 within informal homes

The indoor PM_4 concentrations recorded within informal homes showed a considerable inter-household spatial variation, more especially in winter (Figure 5-3 (a) and (b)). The mean PM_4 concentrations measured within CIH dwellings varied from 33 to 103 µg.m^{-3} with an overall average of 65 ± 21 µg.m^{-3} in autumn and between 50 and 209 µg.m^{-3} with an average of 114 ± 36 µg.m^{-3} in winter. Meanwhile, for NIH houses, the measured concentrations ranged from 33 to

124 µg.m^{-3} with an average of 63 ± 25 µg.m^{-3} in autumn and from 8 to 278 µg.m^{-3} with a mean value of 111 ± 54 µg.m^{-3} (Table 5-5). Overall, informal households regardless of their primary energy carrier, portrayed a different picture than that of formal houses in Lebohang. Unlike the formal houses, on average indoor PM_4 concentrations within informal dwellings increased by 75% and 76% respectively within CIH and NIH households from autumn to winter. Hence, this indicates that the winter season is associated with increased PM_4 concentrations within informal dwellings in Lebohang, regardless of whether they use clean energy sources or not.

Unexpectedly, the 24h average autumn indoor PM_4 concentration sampled at H04 (64 ± 31 µg.m^{-3}) and H08 (59 ± 20 µg.m^{-3}) were somewhat similar, even though the former used a clean energy source and the latter relied on coal-reliant (Table 5-4). Amongst the informal dwellings sampled in Lebohang during autumn, the highest 24h average concentration of 80 ± 16 µg.m^{-3} was recorded at H07. By comparison, the autumn daily mean sampled at H07 was 30%, 20%, and 26% higher than that of H03, H04, and H08, respectively. The staggeringly high indoor PM_4 loads observed at H07 are thought to be reflective of emissions generated during the burning practices of coal. Surprisingly, the highest maximum indoor PM_4 value of 124 µg.m^{-3}, was, however, recorded at H04, which was reliant mainly on electricity for cooking and space heating (Table 5-4). This illustrates that, within a low income setting, where several local emission sources vary in space, magnitude, and time, some non-coal-reliant informal homes are also prone to experience extreme HAP, even during non-burning seasons. This is likely due to the penetration of outdoor emissions either through windows and doors (when opened) or visible holes in the dwelling structures of informal homes (Wernecke, 2018).

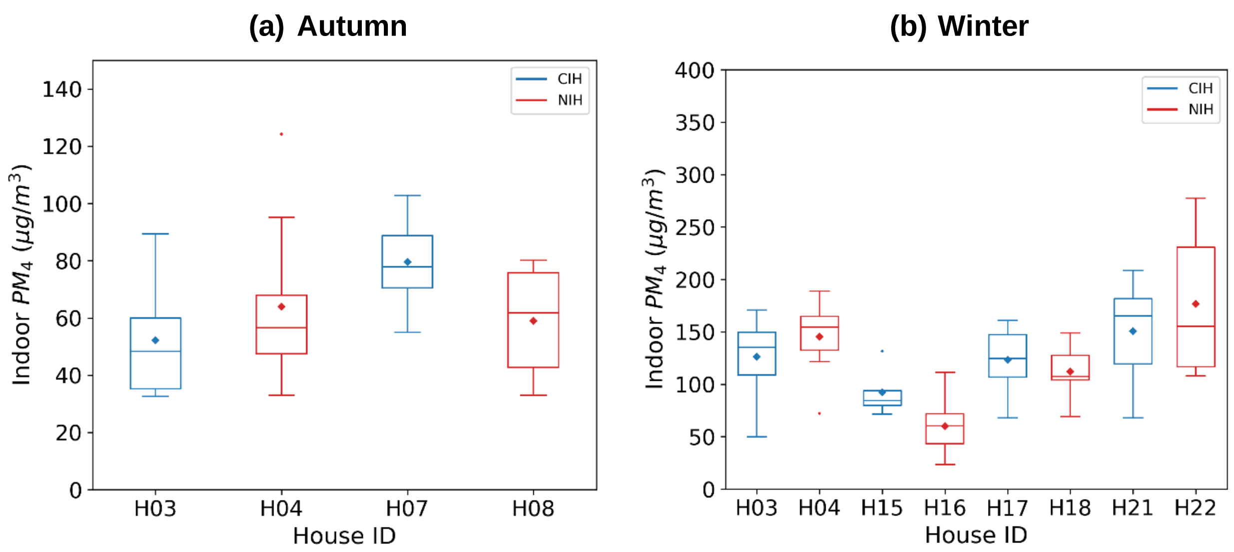

Figure 5-3: Inter-house variance of the 24h mean PM_4 concentrations measured within informally structured homes for CIH and NIH in Lebohang during autumn (a) and winter (b) seasons. Box and whisker plots show

25th, 50th and 99th percentiles; mean (blue diamond inside the box); and outliers (blue stars beyond the whiskers). Of note: the y scale of figures (a) and (b) are not the same.

During winter, the highest daily mean indoor PM_4 concentration of 177 ± 71 $\mu g.m^{-3}$ was surprisingly recorded at H22, which relied mainly on electricity for its energy demands. It is thus, apparent that for H22, the use cleaner energy carrier was not advantageous as it recorded the worst indoor air pollution episode, amongst other informal households. In comparison to the 24h average concentrations sampled at other informal homes in winter, the daily mean concentration at H22 was greater by a factor that was ranging as high as 3. This is reflective of the complexity of resolving the issue of HAP, particularly within informal homes. HAP within informal homes is a factor of various elements that interact together, including factors such as season, thermal comfort, energy carrier used, household size, ventilation arrangements, and background emissions (Jafta *et al*., 2017; Weaver *et al*., 2017). It is therefore clear in the results presented here, that some of these factors interfere with the health and air quality benefits of cleaner energy fuels, more especially within informal homes, particularly during winter. However, based on the scope of this research, the extent to which the aforementioned factors interfere is unknown and requires further investigation.

Table 5-4: Descriptive statistics of the daily mean indoor PM_4 concentrations ($\mu g.m^{-3}$) sampled within each of the sampled informal houses in Lebohang.

Autumn indoor PM_4 concentrations							
House ID	**Count**	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**
H03	8	33	52	20	48	88	89
H04	8	33	64	31	57	122	124
H07	8	55	80	16	78	103	103
H08	6	33	59	20	62	80	80
Winter indoor PM_4 concentrations							
House ID	**Count**	**Min**	**Mean**	**Std**	**Median**	**99%**	**Max**
H03	8	50	126	38	135	170	171
H04	8	72	145	36	155	188	189
H15	5	72	92	23	84	130	132
H16	8	24	60	29	60	109	112
H17	7	68	123	33	125	161	161
H18	8	69	112	24	108	148	149
H21	8	68	151	48	165	207	209
H22	8	108	177	71	155	277	278

5.2.1.2.1 Statistical analysis of PM_4 variation measured between CIH and NIH

A similar statistical test to that conducted for formal houses was also done for informal dwellings to determine the significance of the variance amongst the CIH and NIH homes sampled in Lebohang. The same method that was used for formal homes (refer to section 5.2.1.1.1) was also applied here but, in this section, the daily mean indoor PM_4 concentrations sampled at informal homes were used instead. Table 5-5 shows the matrix of the p values obtained from the statistical test of the mean comparison of different monitored informal swellings during autumn and winter.

During autumn, a vast majority of informal homes sampled in Lebohang, irrespective of their primary energy sources, were found to be insignificantly variable from one another ($p > 0.05$). Notwithstanding, only H03 and H07, which were both reliant on coal, were found to be significantly variable from each ($p < 0.05$) thus, indicating that, the frequency of burning between these homes is different. Of all the comparisons done between the PM_4 mean values of individual homes of CIH (i.e. H03 and H07) against those of NIH dwellings (i.e. H04 and H08), no significant spatial inter variability was found. It is not clear that either this observation was due to the limited number of sampled informal homes (n = 4) during autumn, or indicates the inefficiency of electricity alone to improve air quality within informally structured houses.

Table 5-5: Matrix table showing the p values of unpaired t-test analysis of the autumn and winter daily mean PM_4 concentrations sampled within informal households in Lebohang.

Autumn				
	H03	**H04**	**H07**	**H08**
H03	1	0.32	0.00	0.55
H04		1	0.22	0.73
H07			1	0.05
H08				1

Winter								
	H03	**H04**	**H15**	**H16**	**H17**	**H18**	**H21**	**H22**
H03	1	0.31	0.10	0.00	0.86	0.38	0.28	0.09
H04		1	0.01	0.00	0.23	0.4	0.81	0.28
H15			1	0.06	0.10	0.16	0.03	0.02
H16				1	0.00	0.00	0.00	0.00
H17					1	0.46	0.22	0.09
H18						1	0.06	0.02
H21							1	0.40
H22								1

Blue cells = CIH
Red cells = NIH
Yellow cells = Significantly different PM_4 mean values ($p < 0.05$)

Similarly, in winter, most of the daily indoor mean PM_4 measured within informal homes sampled in Lebohang were insignificantly variable, with p-values that ranged from 0.06 to 0.86. Out of 16 comparisons done between separate homes of CIH against those of NIH class, 11 showed insignificantly inter-household variance ($p > 0.05$) and the remaining 5 did not. Comparisons with $p > 0.05$ were between H03 vs. H04, H18 and H22; H04 vs. H17 and H21; H15 vs. H16 and H18; H17 vs. H18 and H22; H18 vs. H21; and H21 vs. H22 (Table 5-5). This suggests that, despite using electricity, a large proportion of non-coal-reliant informal homes in Lebohang, experienced the same indoor PM_4 as those which were still coal-reliant. Considering that the sampled size of the winter monitored homes was 2 times larger (n = 8) than those sampled in autumn, it is argued that the observed results, to some extent, indicate the inefficiency of only relying on cleaner energy sources to successfully reduce HAP in informal dwellings.

5.2.2 Indoor PM_4 time variation

In this section, the diurnal cycles were generated to explore the indoor PM_4 concentrations during the day across different household classes (CFH, NFH, CIH, and NIH) sampled in Lebohang. The 5-minute mean measurements sampled in each household were resampled and averaged together into hourly mean concentrations respectively for each household group. Figure 5-4 below depicts the diurnal distribution of indoor PM_4 concentrations recorded at CFH, NFH, CIH, and NIH during the summer and winter seasons, respectively.

During both autumn and winter season, double particulate peaks across all household groups were seen though, were more intense during the winter season (Figure 5-4). In the CFH group, the first autumn spike of ≈ 120 $\mu g.m^{-3}$ occurred in the morning (03:00-10:00), while the second one, which was ≈ 270 $\mu g.m^{-3}$, happened in the evening (15:00-21:00). Likewise, the morning and evening peaks at NFH, CIH, and NIH household classes, somewhat occurred during the same hours as those of CFH group. The CFH household class recorded the highest morning peak, which was greater by a factor of 2 in comparison to that recorded at NFH and NIH; however, was similar to that of CIH dwellings. Moreover, this household group had the maximum evening peak which was greater by a factor that ranged between 26% to 44% when compared to those that occurred in other household classes. The results indicate that residents in coal-burning formal households are prone to substantial indoor PM_4 concentrations, especially in the evening, more than any other household structures in Lebohang.

Similarly, during the winter season, across all the household classes sampled in this study, an increment of indoor PM_4 was recorded during the morning (03:00-10:00) and the evening hours (15:00-21:00). Surprisingly, the highest morning peak of ≈ 310 $\mu g.m^{-3}$ was recorded at NIH household class and lasted for about 8 hours. When compared to the peaks recorded at other household classes during morning hours, the NIH peak was higher by an order magnitude that ranged between 29% to 61%. During the evening, however, the highest particulate load of ≈ 420 $\mu g.m^{-3}$ was recorded both at NIH and CIF household classes and was 12% and 45% greater than that of CFH and NFH, respectively. These results are indicative that, during the winter morning hours, the "*neighbourhood pollution effect*" tends to mostly impact the indoor quality of NIH households in Lebohang.

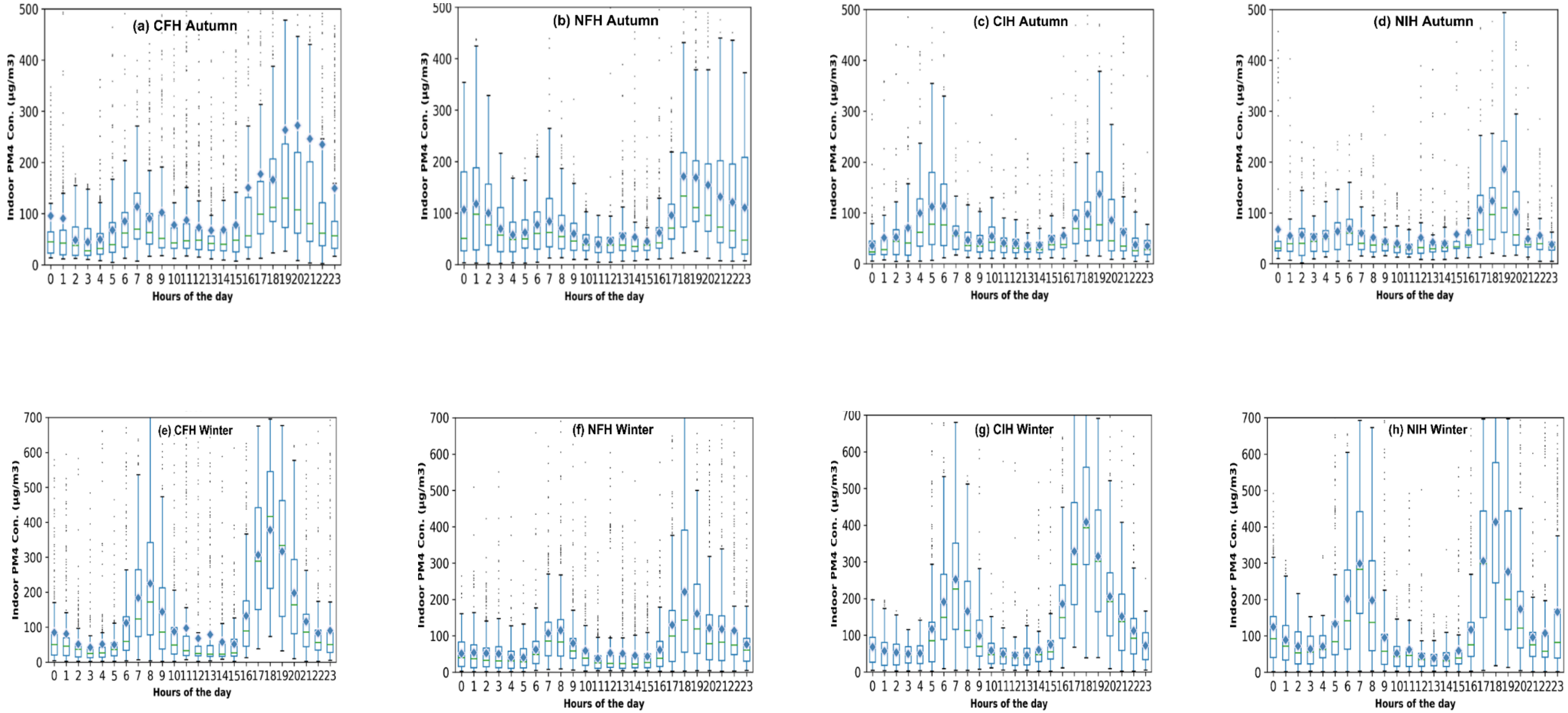

Figure 5-4: Diurnal distributions showing the mean hourly concentrations of indoor PM_4 ($\mu g.m^{-3}$) measured at CFH, NFH, CIH, and NIH during the autumn and winter season in Lebohang low income settlement. Box and whisker plots show 25th, 50th and 99th percentiles; mean (blue diamond inside the box); outliers (blue stars beyond the whiskers).

Chapter 5 presents the indoor PM_4 concentrations measured in the households sampled in Lebohang low income settlement. The chapter specifically covered the second objective of this study, which was outlined in Chapter 1 section 1.2. Throughout this chapter, the spatial and temporal variance of PM_4 across different household classes was assessed, with a specific aim of understanding whether or not the use of cleaner energy sources in this community benefits non-coal homes. The chapter showed that air quality benefits due to electricity reliance were mostly evident within formal noncoal-reliant households rather than in informal noncoal-reliant dwellings. As such, it was argued that due to the non-fuel factors such as the neighbourhood pollution effect, poor structured ventilation and small volume size of informal homes, solving HAP in Lebohang, particularly in informal dwellings, is complex, as these factors interfere with the intended outcomes of using cleaner energy sources in the community.

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CHAPTER 6: PM EXPOSURE BY SUBGROUPS IN LEBOHANG

Chapter 6 presents the total integrated population weighted exposure estimates of various subgroups in Lebohang low income community. This chapter covers the last objective of this study and provides PM exposure profiles as experienced by different demographic groups in Lebohang low income settlement during autumn and winter seasons.

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The exposure profiles of various subpopulations in Lebohang were calculated as a total integrated 24h mean population-weighted exposure yet referred to as the "*integrated population-weighted exposure*" (IPWE). The IPWE was calculated as a function of the 24h averages of the PM_4 and $PM_{2.5}$ concentrations and the time activity data (hr/day) representative of each respective group in the community. The calculations were made using equation (3) through Monte Carlo simulation. The IPWE for each assessed group was estimated for the autumn season (March 2018 – May 2018) and winter period (June 2018 – August 2018) to determine the temporal variance of exposure profiles in the area. Only PM_4 and $PM_{2.5}$ data that corresponded in time were used for calculating the IPWE of different studied subgroups for each season. Due to a lack of time activity data for employed adults' groups (i.e. both employed females and males), the IPWE was only estimated for 7 subpopulations, namely, infants, toddlers, male school children, female school children, unemployed male adults, unemployed female adults, and retired people. The results of the Monte Carlo simulation for each demographic group were presented using a frequency histogram and summarised through descriptive statistics.

6.1 Daily average integrated population-weighted exposure for different subgroups

The descriptive statistics of the 24h average IPWE estimated for various subgroups in Lebohang during the autumn and winter period are presented in Table 6-1 and Table 6-2, respectively. In both tables, the 10^{th}, 25^{th}, 50^{th}, and 99^{th} percentiles, and the mean IPWE values calculated for each group are presented; and the uncertainty of each estimated 24h average IPWE value is presented as the mean standard error. Figure 6-1 and Figure 6-2 depict the curves of probability distributions of mean IPWE for each subgroup assessed in Lebohang, respectively during the autumn and the winter seasons.

Table 6-1: Daily average total integrated population-weighted exposure ($\mu g.m^{-3}$) of various subgroups during the autumn period (March 2018 - May 2018).

Age	Gender & Occupation	Percentiles				Mean ± Std	Standard Error
		10^{th}	25^{th}	50^{th}	99^{th}		Mean
1-2	-	30	32	35	47	36 ± 5	0.04
3-6	-	23	26	29	42	29 ± 5	0.05
7-18	Male School Children	14	15	17	22	17 ± 2	0.02
	Female School Children	16	18	20	28	20 ± 3	0.03
19-64	Employed Males	-	-	-	-	-	-
	Employed Females	-	-	-	-	-	-
	Unemployed Males	20	23	26	39	26 ± 5	0.05
	Unemployed Females	24	27	31	45	31 ± 6	0.06
≥65	-	58	63	68	91	68 ± 9	0.09

Table 6-2: Daily average total integrated population-weighted exposure ($\mu g.m^{-3}$) of various subgroups during the winter period (June 2018 - August 2018).

Age	Gender & Occupation	Percentiles				Mean ± Std	Standard Error
		10^{th}	25^{th}	50^{th}	99^{th}		Mean
1-2	-	41	44	48	63	48 ± 6	0.05
3-6	-	32	35	40	57	40 ± 7	0.06
7-18	Male School Children	20	22	24	31	24 ± 3	0.02
	Female School Children	23	26	28	38	28 ± 4	0.03
19-64	Employed Males	-	-	-	-	-	
	Employed Females	-	-	-	-	-	
	Unemployed Males	28	32	36	53	36 ± 7	0.06
	Unemployed Females	33	37	42	61	42 ± 7	0.07
≥65	-	79	85	93	120	93 ± 11	0.10

The mean total PM IPWE estimated for all the assessed subpopulations in Lebohang ranged from 9 to 111 µg.m^{-3} during autumn and between 3 and 149 µg.m^{-3} in the winter season. Across all the studied groups, retired people had the highest autumn average estimated IPWE value of 68 ± 9 µg.m^{-3}, followed by infants, unemployed females, toddlers, unemployed males, female school children and male school children. During winter, exposure profiles of various subgroups followed a similar order to that of autumn with elderly people experiencing the highest IPWE of 93 ± 11 µg.m^{-3}, while male schoolchildren were subjected to the least IPWE of 24 ± 3 µg.m^{-3}. Irrespective of the season, the mean estimated IPWE values for retired people were 2 times higher than the daily $PM_{2.5}$ NAAQS of 40 µg.m^{-3}, hence, showing that the burden of exposure in Lebohang is mostly felt by elderly people. These results are in line with those of Wang *et al*. (2000) and Park *et al*. (2020) who similarly reported that retired people (≥65 years) in comparison to other subgroups, experienced the highest PM exposure. Wang *et al*. (2000) argued that this was likely the case because old people spend a large fraction of their time indoors, where PM concentrations are often high.

Although during both autumn and winter season, retired people had the highest estimated IPWE in Lebohang, infants, unemployed adult females, and toddlers had higher IPWE values. Moreso, not only were these groups subjected to high PM concentrations but, they experienced almost similar PM IPWE though toddlers were often subjected to slightly lower exposure levels. For instance, in autumn the IPWE for infants, unemployed females and toddlers was 36 ± 5, 31 ± 6 and 29 ± 5 µg.m^{-3} respectively, and in winter it was 48 ± 6, 42 ± 7 and 40 ± 7 µg.m^{-3}, respectively. This was similarly the case even when comparing the winter and autumn estimated 50th and 99th percentiles of the IPWE averages across these subgroups. For example, the 99th percentile of the daily mean IPWE values for infants, unemployed females and toddlers were respectively 47, 45 and 42 µg.m^{-3} in autumn and 61, 63 and 57 µg.m^{-3} during the winter period (Table 6-1 and Table 6-2). This is indicative that, in Lebohang, infants, unemployed women and toddlers are often in contact with high PM concentrations than unemployed men and school children going children. The results found in this study were expected as they agree with the observations that were made by (Ezzati & Kammen, 2002; Saksena *et al*., 2007; Bruce *et al*., 2000; Li *et al.*, 2016).

Irrespective of the season, the school-going children in Lebohang displayed similar IPWE profiles though the male pupils were somewhat subjected to slightly lower PM exposure than female learners. Generally, amongst the subgroups assessed in the community, schoolchildren, regardless of their gender, were found to be the least exposed group thus, suggesting that they are not susceptible to high particulate exposure. Albeit the 28% to 29% PM exposure increase experienced by school going male and female children from autumn to winter, both these subgro-

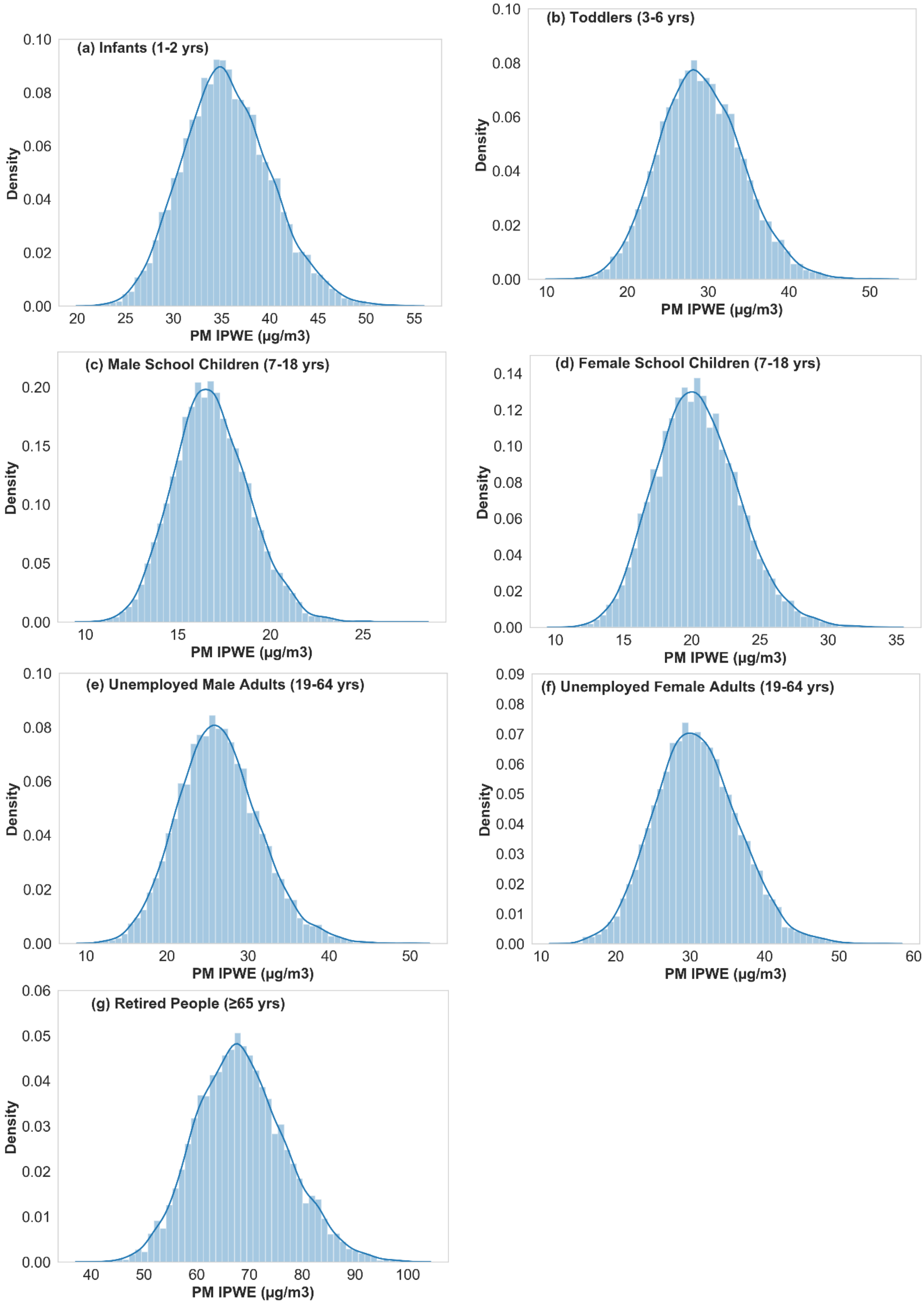


Figure 6-1: 24h average IPWE probability distribution curves for infants (a), toddlers (b), male school children (c), female school children (d), unemployed male adults (e), unemployed female adults (f) and retired people (g) in Lebohang low income community during the autumn period.

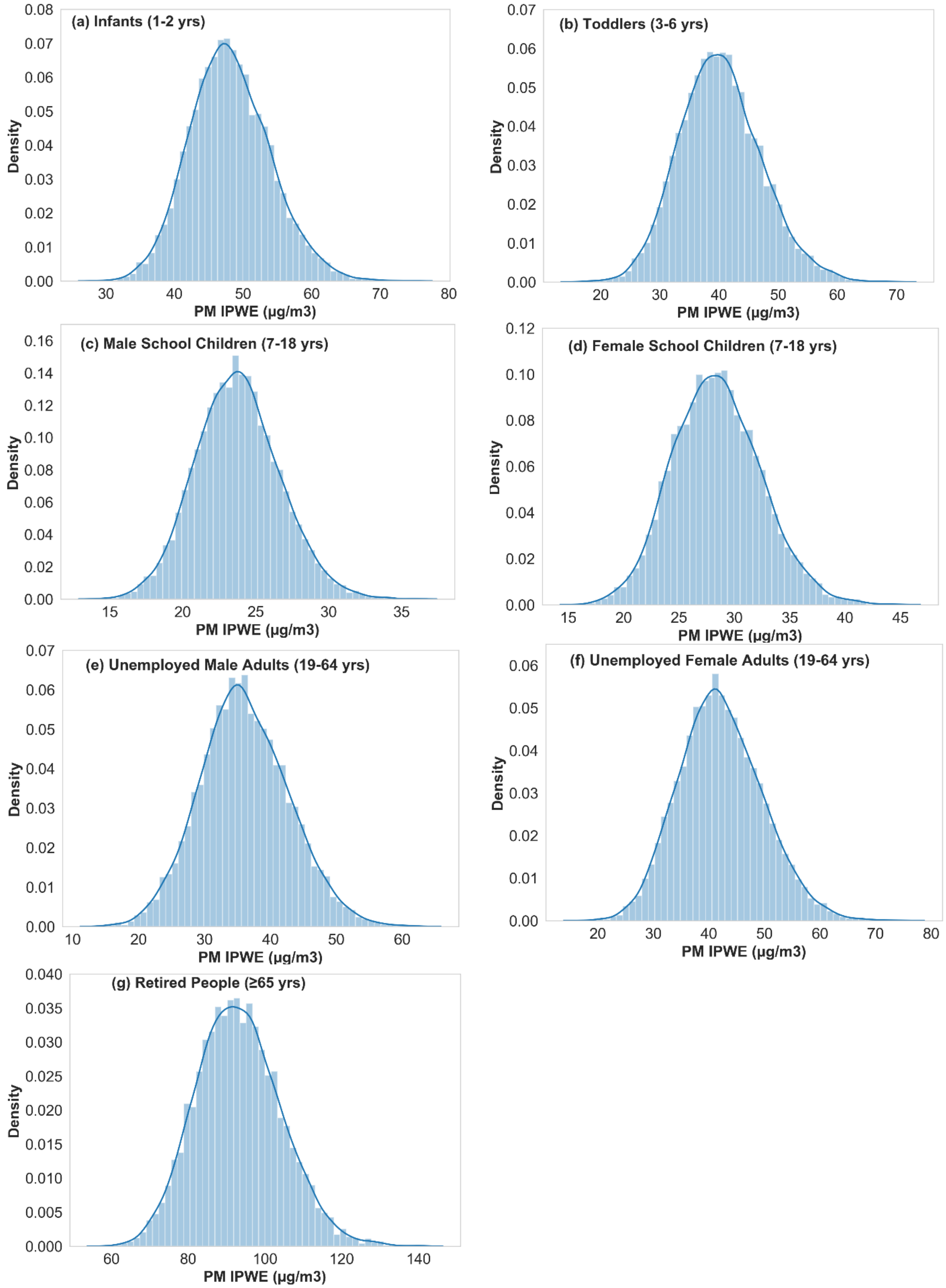


Figure 6-2: 24h average IPWE probability distribution curves for infants (a), toddlers (b), male school children (c), female school children (d), unemployed male adults (e), unemployed female adults (f) and retired people (g) in Lebohang low income community during the winter period.

ups were subjected to low PM exposure in the community. The daily mean and the 99th percentile average IPWE values for male and female school going children were even below the daily $PM_{2.5}$ NAAQS of 40 µg.m^{-3}, pointing out that these groups are not vulnerable to extreme exposure to PM concentrations (Table 6-1 and Table 6-2).

6.1.1 Contribution of microenvironments to the simulated total IPWE for various subgroups

The contribution made by each ME (i.e. indoor and outdoor) in Lebohang were evaluated to determine places where different groups in the community mostly experience PM exposure. To determine the contribution of indoor and outdoor, equations 4 and 5 outlined below were respectively used through monte carlo simulation:

$$IPWE_i = \sum_{j}^{n} \frac{C_{dg,i} \times t_{dg,i}}{n}$$

Equation (4)

Where $IPWE_i$ = integrated population-weighted exposure of the assessed social group experienced at indoor microenvironment i (µg.m^{-3}); n = total sum of the assessed demographic group, $C_{dg,i}$ = concentration of PM experienced by the studied demographic group dg within indoors i; $t_{dg,i}$= time spent by the demographic group dg in the indoor environment i.

$$IPWE_a = \sum_{j}^{n} \frac{c_{dg,a} \times t_{dg,a}}{n}$$

Equation (5)

Where $IPWE_a$ = integrated population-weighted exposure of the assessed social group experienced at outdoor microenvironment a (µg.m^{-3}); n = total sum of the assessed demographic group, $C_{dg,a}$ = concentration of PM experienced by the studied demographic group dg within outdoors a; $t_{dg,a}$= time spent by the demographic group dg in the outdoor environment a.

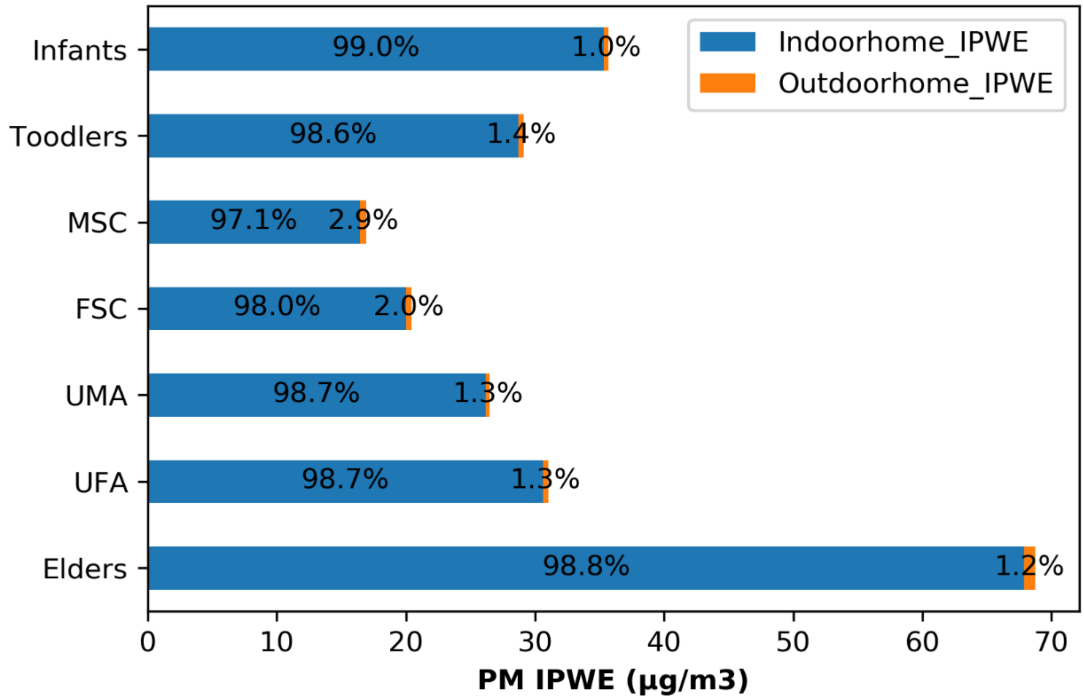
It must be noted that both equation (4) and equation (5) were derived from equation (3) but, instead of adding up the contribution of each ME just as in equation (3), the IPWE was estimated separately for each ME. The number of monte carlo trials selected for both equation (4) and equation (5) are similar to that of equation (3) as earlier explained in section 3.2.3.2.

The contribution of the indoor and outdoor environments to the total simulated daily mean PM IPWE experienced by different subgroups in Lebohang during winter and autumn is depicted in Figure 6-3. Notably, over 90% of the estimated PM total IPWE during both seasons originated mainly from the indoor environment across all the studied subpopulations in the community. The contribution of the indoor to the total IPWE to PM concentrations Lebohang residents was subjected to range from 97.1% to 99.0% in autumn and between 90.3% to 95.8% in winter. It is apparent that across all the studied subpopulations, irrespective of the season, the indoor environment greatly contributed to the total PM IPWE in the community, thus, making it a microenvironment of "*high exposure*".

During autumn, indoor contribution to PM total IPWE experienced by infants, toddlers, male school children, female school children, unemployed male adults, unemployed female adults, and elders was 99.0%, 98.6%, 97.1%, 98.0%, 98.7%,98.7%, and 98.8%, respectively. On the other hand, during the winter season, the indoor environment contributed 95.8%, 93.1%, 90.3%, 91.9%, 94.4%, 94.4%, and 94.9% to the total PM IPWE concentrations experienced by infants, toddlers, male school children, female school children, unemployed male adults, unemployed female adults, and elderly people, respectively. Interestingly, the indoor microenvironment contribution to the total PM IPWE experienced by all subpopulations in the community, was slightly lower during winter, when compared to autumn. Li *et al*. (2016) made a similar observation in Lanzhou, China, and attributed this phenomenon to the extreme particulate pollution experienced during the coal burning season, which in turn, increase the outdoor contribution to the total PM IPWE. It is, therefore, suspected that this could have likely been the case even in Lebohang as it was shown in section 4.1.1 that intense air pollution episodes in this area are experienced in winter.

The contribution from outdoors to the total PM IPWE experienced by all the studied groups was very low in comparison to that of the indoor microenvironment and ranged from 1.0% to 2.9% in autumn and between 4.2% and 9.7% in winter. During autumn, the outdoor environment in Lebohang is estimated to have contributed 1.0%, 1.4%, 2.9%, 2.0%, 1.3%, 1.3%, and 1.2% to the total IPWE experienced by infants, toddlers, male school children, female school children, unemployed male adults, unemployed female adults, and elders, respectively. For the winter period, the outdoor environment contribution to the total IPWE experienced by infants, toddlers, male school children, female school children, unemployed male adults, unemployed female adults, and elders was estimated to be 4.2%, 6.9%, 9.7%, 8.1%, 5.4%, 5.4%, and 5.1%, respectively. Despite the general increment of the outdoor contribution to the overall total PM IPWE seen in all the studied subgroups during winter, a large fraction of human exposure to PM concentration in the community was still experienced within the indoor environments. It is thus evident that residents in Lebohang are at greater risk of suffering from air pollution-related health problems when inside their homes than when they are in the ambient environment.

(a) Autumn



(b) Winter

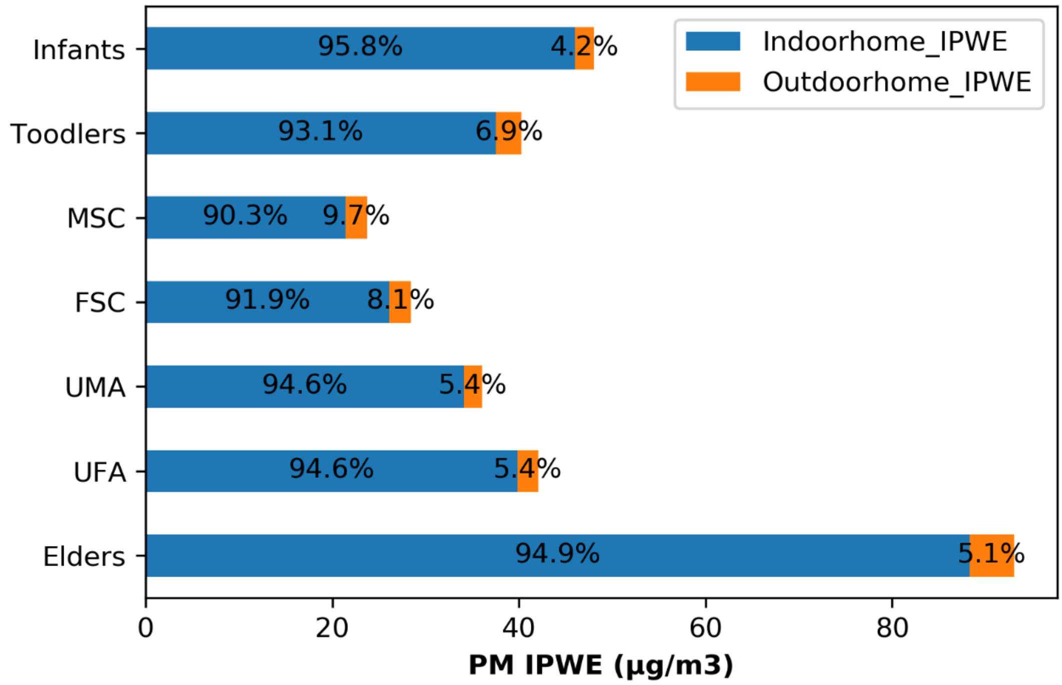


Figure 6-3: Percentage contribution of various MEs to the total simulated IPWE of each subgroup during the autumn (a) and winter (b) seasons in Lebohang.Take note that MSC refers to male school children; FSC stands for female school children, UMA stands for unemployed male adults and UFA stands for unemployed female adults.

6.1.1.1 Implications for future exposure studies and policy drafting

There are two main implications that the results presented in section 6.1.1 above have on PM exposure assessment research and air pollution mitigation plans designed for townships. The results first suggest that a comprehensive PM exposure assessment, particularly conducted in a township setting, should not be based on the assumption that residential exposure to PM only occurs in the outdoor environment. This is because by doing so, the absolute PM exposure that people are truly subject to (both from the household and ambient environments) can be largely underestimated since residential exposure mainly occurs indoors as seen in Figure 6-3. Consequently, this can further lead to the underestimation of the true health risks that residents in such areas are most likely to suffer from due to PM exposure (Saksena *et al.,* 2007). Secondly, the results further imply that mitigation strategies aimed at reducing population exposure to PM in low income communities must aim at minimising the residential exposure that occurs at household levels. This is because households are "*high exposure*" microenvironments, where different demographic groups experience substantial PM concentrations as demonstrated in Figure 6-3. Household exposure to PM should, therefore, always be considered when designing particulate pollution mitigation measures for impoverished communities; and when intending to conduct health risk assessments associated with PM exposure in such areas.

6.1.2 IPWE uncertainty analysis

Due to uncertainties associated with the time activity data in this study, it is acknowledged that this might have resulted in uncertainties in the overall estimated exposure levels in Lebohang. However, because the large sample size minimises the influence of uncertainties on the exposure estimates, running the monte carlo simulation 10 000 times, therefore, reduced the contribution of uncertainties introduced by the time activity input on the exposure estimates of each subgroup (Mestl *et al*., 2006). In each trial, the PM exposure scenario was generated and ultimately, 10 000 exposure scenarios were formulated for each subpopulation. The uncertainty of the average PM exposure estimated for each group in Lebohang was calculated based on these 10 000 exposure scenarios and expressed as mean standard error (refer to table Table 6-1 and table Table 6-2).

The uncertainty associated with the estimated autumn daily average IPWE presented in section 6.1 for infants, toddlers, female school children and retired people is 0.1% and for male school children, unemployed women, and unemployed men is 0.2% (Table Table 6-1). Furthermore, the uncertainty of the average winter estimated IPWE for infants, female school children and retired people is 0.1% and that of toddlers, unemployed men and unemployed women is 0.2%, while that of male school children is 0.08%. It is, therefore, clear that during both the autumn and winter season, the estimated daily average IPWE for all groups in Lebohang were reliable as monte carlo simulation had reduced the uncertainty of the estimated exposure, hence low standard errors were reported (Harding *et al*., 2014).

Chapter 6 has provided a detailed discussion about the daily average integrated population weighted exposure to PM concentrations that different subgroups in Lebohang are subjected to. It specifically covered the final objective of this study, which was outlined in Chapter 1 section 1.2. Throughout this chapter, it was shown that there are exposure inequalities in the community and that, the elderly, infants, toddlers, and unemployed female adults are vulnerable to PM exposure in this area. The chapter further revealed that people, irrespective of their demographic groups, mostly experience IPWE to PM while inside their households than when they are in the ambient environment. This chapter argued that this finding highlights the importance of indoor PM exposure in Lebohang, and that, any intervention aimed at reducing residential exposure to PM, must never ignore the contribution of household exposure to the total IPWE to PM concentrations.

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CHAPTER 7: SUMMARY AND CONCLUSION

Chapter 7 presents a summary of the main findings for each of the objectives outlined in chapter 1 and the limitations that forms part of this study.

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This study was aimed at assessing the exposure of residential members that are subjected to ambient and indoor PM concentrations in Lebohang. To achieve the aim of this study, the outdoor $PM_{2.5}$ and PM_{10} concentrations were firstly characterised using the ambient data collected from the monitoring station in Lebohang. Secondly, the indoor PM_4 from 22 households that varied in structure and energy carrier use in the area; and which were sampled as part of the Sasol baseline monitoring campaign using DustTrak II Model 8530 monitors, was also characterised. Thirdly, the indirect exposure assessment method was used as well as the Monte Carlo simulation to estimate the PM exposure experienced by community members in Lebohang. These simulated estimates were based on outdoor and household PM measurements, the time activity data from the literature, and the 2011 Census community survey demographic information. The main findings made in each of the objectives of this study are outlined and discussed in the subsections below:

7.1 Characterising ambient PM concentrations in Lebohang

The air quality in Lebohang low income settlement was poor during the winter season (May and August 2018) and relatively cleaner during the warm period (October 2017 to April 2018). The daily mean ambient $PM_{2.5}$ and PM_{10} concentrations followed a similar distribution trend. The trend was characterised by PM concentrations that were mostly noncompliant with the 24h mean NAAQS for $PM_{2.5}$ and PM_{10} during winter and often compliant during the warm months. The period during which ambient PM concentrations were noncompliant with NAAQS for $PM_{2.5}$ and PM_{10} was found to correlate with the time of the year when low income residents typically burn more coal.

The ambient $PM_{2.5}$ and PM_{10} concentrations measured in Lebohang were skewed to the right and mostly ranged between >0 $\mu g.m^{-3}$ and 30 $\mu g.m^{-3}$. This range was lower in comparison to that recorded in KwaDela (>0 – 50 $\mu g.m^{-3}$) by Wernecke *et al*. (2015) and in Accra ($PM_{2.5}$: >0 - 100 $\mu g.m^{-3}$; and PM_{10}: >0 – 200 $\mu g.m^{-3}$) by Arku *et al.* (2008). It is unclear what might have caused this variation across these studies despite that they were all conducted within low income areas. However, it is suspected that it could have been due to variances in neighbourhood factors that differed from one study area to another and sampling protocols that varied from one study to another.

The 24h averaged monthly ambient PM loads in Lebohang, the $PM_{2.5}$ and PM_{10} measurements ranged from 1 to 109 µg.m^{-3} and 2 and 138 µg.m^{-3}, respectively, during the study period. Out of 325 days that were sampled for ambient PM in this residential area, $PM_{2.5}$ and PM_{10} respectively surpassed the 24h average NAAQS 47 and 67 times meanwhiles only 4 exceedances are allowed annually. Residents were, therefore, found to be often breathing air that is legislatively declared to be unacceptable and health threatening in the country.

The diurnal plots of ambient PM concentrations in Lebohang were characterised by double peaks which remained consistent throughout the days of the week and different seasons of the year. The first peak occurred in the morning (around 06:00) and evening (around 18:00). The occurrence of these spikes matched with the time of the day during which domestic combustion of coal within low income communities typically takes place and traffic emissions are at their highest. Irrespective of the day of the week, the ambient PM morning peaks were 2 times lower than those recorded in the evening hours since residents are less likely to be engaged in timing activities in the morning hours. Across all seasons, the extreme bimodal peaks of the ambient PM were seen during winter (June-August 2018) further alluding that particulate pollution coincides with residential burning of coal.

7.2 Indoor PM_4 concentrations in Lebohang

The mean 24h indoor PM_4 concentrations in Lebohang for all households sampled in this study ranged between 24 µg.m^{-3} and 482 µg.m^{-3} with a mean value of 107 ± 62 µg.m^{-3}. Formally structured dwellings and informally designed households measured the same average indoor PM_4 mass concentrations during the entire sampling period. However, in autumn, the mean indoor PM_4 measured at the informal homes was two folds lower than that sampled in formal households. Meanwhile, winter mean PM_4 in informal houses was 1.2 times higher than that recorded at formal houses, thus indicating that structural designs and seasonal variance have inconsistent impacts on household air quality in Lebohang.

The average indoor PM_4 concentration of CFH homes was higher than that of NIH households during the entire sampling campaign, autumn, and winter season. Irrespective of the season, NFH dwellings, on average, recorded indoor PM_4 concentrations that were lower by almost a factor of 2 compared to those sampled within the CFH houses. This, therefore, suggested that the use of cleaner energy sources in NFH dwellings was beneficial and yielded the expected results in the community. Conversely, the mean indoor PM_4 concentration recorded in CIH dwellings and that recorded at NIH homes were the same during the entire sampling period and both the autumn and winter seasons. This demonstrates that the "*neighbourhood pollution effect*" degrades the indoor air quality of NIH dwellings, thus depriving the residents of such homes of the health and environmental benefits associated with using clean energy sources.

Indoor PM_4 measured within individual CFH homes vs. NFH dwellings in Lebohang were mostly significantly variable, irrespective of the season. Nevertheless, there were few instances in autumn and winter where comparisons between individual homes of CFH with those of the NFH category were insignificantly different. These occurrences, however, were seldom thus, suggesting that for most noncoal reliant formal homes in the community, cleaner energy sources were associated with relatively cleaner air quality. The opposite, however, was the case for informal homes as there was mostly no significant inter-house-to-house variance between the indoor PM_4 measured within individual homes of CIH vs. those of NIH dwellings. This phenomenon was consistent during both the autumn and winter seasons, further alluding that using clean energy sources alone within informal homes does not guarantee improved air quality.

The average diurnal indoor PM_4 concentrations measured within CFH, NFH, CIH, and NIH dwellings in Lebohang, all displayed bimodal distributions during autumn and winter seasons. The highest autumn average morning indoor PM_4 peak was recorded at CFH and CIH categories and was 2 times greater than that of NFH and NIH class. Meanwhile, the highest evening PM_4 peak was only recorded in the CFH category. However, during winter, the NIH group had the highest morning peak, while the evening maximum peak was recorded at NIH and CIF household categories.

7.3 PM exposure by population groups in Lebohang

Across all the studied demographic populations in Lebohang, elderly people experienced the highest levels of total IPWE to PM concentrations during the autumn and winter. The total IPWE to PM estimated for elderly people in the area was 2 folds greater than the daily $PM_{2.5}$ NAAQS of 40 $\mu g.m^{-3}$, irrespective of the season, thus suggesting that they are more susceptible to PM-related health issues. Notwithstanding, infants, unemployed adult females, and toddlers in the community were also found to be subjected to high IPWE to PM concentrations during both seasons compared to schoolchildren and unemployed males. This finding was in line with observations that were already made by previous studies that were conducted elsewhere in low income communities (Ezzati & Kammen, 2002; Saksena *et al*., 2007; Bruce *et al*., 2000; Li *et al.*, 2016). The total IPWE to PM experienced by school going children, irrespective of their gender, was low during both seasons and was below NAAQS of 40 $\mu g.m^{-3}$. This, therefore, suggested that school going children are not vulnerable to extreme exposure to PM concentrations.

The contribution of the indoor environment to the total IPWE to PM estimated for all the assessed groups in Lebohang was overall more significant during both the autumn and winter seasons. For all the assessed groups in this community, the contribution of the indoor microenvironment to the total PM IPWE ranged from 97.1% to 99.0% in autumn and between 90.3% to 95.8% in winter. Therefore, this indicated that household environments are microenvironments of "*high exposure*" in the community. The contribution of the outdoor environment to the total PM IPWE was meager compared to that of indoor and ranged from 1.0% to 9.7% during both autumn and winter seasons. There was a slight increase in the outdoor environment contribution to the total PM IPWE experienced by the studied groups in the area but the indoor environment was still a microenvironment of "*high exposure"* for the assessed groups. This, therefore, suggests that when assessing residential exposure to PM concentrations, particularly in a township setting, household microenvironments should always be considered; otherwise, the true total PM exposure people are subjected to might be underestimated. Furthermore, the results also indicate that quantifying the indoor environment is the main uncertainty in the goal to understand total residential exposure within low income areas.

7.4 Limitations of the study

Although the current research had developed a methodology to assess residential exposure to ambient and household PM concentrations in Lebohang low income settlement, several limitations formed part of this study. These limitations are outlined and discussed below:

- Firstly, the number of homes where PM_4 was monitored in Lebohang (refer to section 3.2.2.2) was too small to be representative of the total number of households in the community. However, it must be noted that the sample size of the homes monitored in this study was primarily influenced by economic constraints, hence, sampling a representative portion of homes could have been more expensive.
- The second constraint of this study is that the estimates of residential exposure to total PM concentrations were made using the secondary time activity data and an indirect exposure assessment approach. As earlier mentioned in section 2.4, the direct method is more suitable when conducting exposure assessment as it considers the spatial heterogeneity of the individual's exposure to PM thus, capturing the complex nature of human exposure. This approach could not be adopted in the current study as it was more expensive and required a substantial number of sampling equipment to be successfully conducted. However, it should be noted that the indirect approach still provides useful insight into the state of residential exposure to PM concentrations, particularly, in instances where certain required input data is unavailable (Wang *et al*. 2008).

- The third limitation of this study was that the distribution curves of the input data used to assess residential PM exposure were based on assumptions, literature, and expert judgment. Hence, the applicability of the selected distribution curves specifically in Lebohang low income settlement is unknown. However, it is thought that since such distribution curves were taken from studies that were similarly conducted in low income areas, the error that they might have been introduced to the results is not that substantial.
- Another limitation of the present study is that the fraction sizes of PM monitored in the ambient environment ($PM_{2.5}$) and the household environment (PM_4) were not the same. However, because $PM_{2.5}$ is a good approximation of PM_4 (Language *et al.*, 2018), and both PM_4 and $PM_{2.5}$ fall under the coarse mode classification (≥ 2.5 µm - < 10 µm) (refer to Figure 2-1), it was assumed that this decision did not introduce significant uncertainties in the results.

7.5 Conclusion

This study has developed a model to calculate the integrated population-weighted exposure experienced by different demographic groups in Lebohang. It applied the indirect assessment approach and demonstrated that this method can provide an insight into the exposure inequalities within low income communities. Through the use of monte carlo simulation and bootstrapping, the study further showed that these statistical tools can provide exposure estimates for residents of low income communities even in the absence of critical primary input data. The results of this study can be used by authorities to develop exposure mitigation policies in low income communities and health risk assessment studies.

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