Characterising indoor airborne particulate matter in Sharpeville, Gauteng

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ABSTRACT

Solid fuel use for domestic purposes is a significant air quality and health challenge within low-income settlements in South Africa. Fuels such as coal, wood and other “dirty” household energy carriers (including paraffin and crop waste) add to the problematic indoor airborne concentration of pollutants detrimental to the health of the residents in low-income communities. Sharpeville is no different to these other low-income areas; although the majority of low-cost houses have access to electricity, use of household solid fuel for domestic activities is still prevalent. The purpose of this research was to characterise indoor airborne respirable particulate matter (PM_4) in the low-income community of Sharpeville in Gauteng in order to inform health assessments in terms of household energy use. To reach this aim, the study took the direction to determining the dominant fuel in the community’s household energy mix; continuously measuring the variability of PM_4 over time, and evaluating the elemental characteristics of indoor respirable PM.

A questionnaire based detailed energy use survey was carried out to determine the dominant household energy carrier used by the Sharpeville community. Following the survey, DustTrak 8530 fitted with a 10-mm Nylon Dorr-Oliver Cyclone to attain a 50% cut-off at 4 µm were installed at 16 randomly selected houses. The DustTraks were continuously measuring respirable airborne PM at 1.7 $L.min^{-1}$. Gravimetric sampling using Gilian GilAir sampling pumps with a cyclone inlet offering the same PM cut-size and connected to a 37mm cassette containing Mixed Cellulose Ester (MCE) filters was conducted simultaneous with photometric samplers. Although this was done as a reference method to compensate for overestimation that comes with the laser photometers, a correction factor of 0.715 was adopted from Language *et al* (2016) and applied to the data. Moreover, indoor temperatures were continuously measured within all sample houses for both winter and summer sampling campaigns. Additional measurements of black carbon (BC) were taken in one of the sample houses that used solid fuels as their primary household energy carrier.

The household energy mix within the community was mainly found to consist mainly of three “dirty” fuels which were wood, coal and paraffin (kerosene). Wood was found to be the predominant fuel used by the largest number of households in Sharpeville {winter: 2973 (24%); summer: 1122 (9%)}. Compared to coal and paraffin usage, wood use is higher by a factor of 1.5 in winter and 2 in summer. However, when calculated per coal-using and per wood-using household, coal was the main energy carrier in winter and wood in summer. Whereby an average 85 kg/month of coal is consumed whereas wood and paraffin usage was 47 kg/month and 14 litres/month in winter and only 8 47 kg/month, 13 47 kg/month and 9 litres/month respectively in summer. This is indicates coal as the preferred fuel for winter months whereas in summer, wood is more reliable due to the diminished need to burn for extended periods of time.

The average indoor diurnal pattern of PM_4 shows a bi-modal distribution, with an extended period of elevated PM_4 concentrations during the evening peak. This pattern was observed in all houses in which measurements were taken. Houses that do not use solid fuels (non-solid fuel burning (NSFB)) combustion as a primary source of energy experienced lower concentrations throughout the day. Nevertheless, the concentrations were still found to exceed 24hr average $PM_{2.5}$ National Ambient Air Quality Standards (NAAQS) in the winter during peak hours of the day. There are no existing NAAQS for PM_4 thus for the purpose of this study PM_{10} and $PM_{2.5}$ NAAQS are used as a proxy to measure the extent of the personal exposure to PM_4 within households. Summer concentrations in the NSFB households were mostly below both 24hr PM NAAQS. Although there was a ~47% decrease in average concentration levels of respirable PM in solid fuel burning (SFB) households, PM_4 was problematic during both seasons in these houses. In most cases, residents of SFB households were found to be exposed to PM_4 levels above the 24hr $PM_{2.5}$ and PM_{10} NAAQS throughout the year. This represents a health hazard for these communities.

Moreover, additional black carbon (BC) measurements conducted continuously in one of the SFB houses during the summer campaign. The dataset supports the assumption of continued use of solid fuels during this period as it shows a bi-modal distribution and elevated concentrations during peak hours of the day. Housing physical structures need to ensure thermal properties are suitable for keeping the indoor environment thermally satisfactory to the dwellers. Indoor temperature profiles during winter seem to be below the minimum World Health Organisation (WHO) of value 15°C for over 60% of the day (14.4 hours). Manual indoor warming (using clean or solid fuel) was observed and assumed to occur in the late evening when ambient temperatures drop below 15 C but indoor temperature remained on average above 17 C although they rise to about 19 C from 18h00 to 21h00. They then dropped again at about 22h00 but do not go below the WHO minimum value.

Gravimetric mass concentrations confirms the results obtained from the photometric measurements. Concentrations in the NSFB were lower than SFB houses. Highest concentrations were measured in SFB houses in winter. In addition to mass, the collected filters have been analysed by X-Ray Fluorescence (XRF) to obtain the elemental composition of the airborne particles in houses. The elemental species composition was found to behave differently over both daily and seasonal timescales. These species (mostly soil-related) were found highly concentrated either during the day in summer or night in winter, or vice-versa. In an SFB household, combustion elements were mostly found during the night in winter, which is a sign of domestic burning in the evening which thus elevated PM_4 concentrations during the evening peak in the SFB house. However, according to the Principal Components Analysis (PCA) multivariate factor analysis, in both household types, most of the elemental composition was dust-related. This signifies the impact of ambient dust infiltration to the indoor environment. Winter PCA results

confirmed domestic burning because sulphur (S), potassium (K) and chlorine (Cl) were significant elements in the SFB household. The calculation and determining of non-crustal source enrichment show high impacts of ambient air on the indoor air of both household types, although most especially in NSFB households. The highly enriched S content in the PM samples in a house completely reliant on electricity was attributed to ambient source contribution such as from the industrial and energy sector. A further potential source was domestic SFB practices in the community. The study contributes to the overall aim of understanding pollution related health risks in low-income communities in South Africa. It also adds to the body of science in terms of showing the variability of low-income households and drivers to the individual use of solid fuels as household energy carriers. The study fosters an understanding of other air pollution sources to the indoor environment and the health of people thereof.

Keywords: Indoor Respirable PM (PM_4), domestic fuel combustion, elemental composition, Principal Component Analysis (PCA), Enrichment Factors (EF), Source contribution.

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DECLARATION

I, Mr Thapelo Andrew Ferdinant Letsholo (Student number: 24516562), the undersigned, declare that the dissertation:

"Characterising indoor airborne particulate matter in Sharpeville, Gauteng"

is my own work with information acquired from other sources correctly reference in accordance to the NWU Havard style of reference. It is being submitted in partial fulfilment of the requirements for the Degree of *Magister Scientiae* in Environmental Sciences in the School of Geography and Spatial Sciences at the Potchefstroom Campus of the North-West University. This dissertation has not been submitted for any degree in the North-West University or in any other institution.

X

Thapelo Letsholo
Masters Student

PREFACE: DISSERTATION FORMAT

Domestic solid fuel use in South Africa has become a significant source of particulate matter within low-income communities. South African low-income communities have access to electricity, however, households continue to rely on alternative energy carriers other than electricity. Majority of the low-income settlements use wood, coal and paraffin to meet their daily household energy needs. There are a number of factors that contribute to this including household income, fragile electrical infrastructure, household traditional preferences, and culture, to name but a few. A human health hazard arises from exposure to indoor emissions as a function of domestic solid fuel combustion.

Personal exposure to respirable airborne particles is a precursor to many pulmonary, cardiovascular and respiratory illnesses prone to cause premature mortality. These particles penetrate deeply into the lungs and disturbs the natural respiratory functions. The size and chemistry of the aerosol particles play an important role as they come into contract with the human respiratory system. The size determines how deeply the particle can advance for instance, particles above 10 µm (coarse fraction) can easily be flushed out by the body in the form of mucus however, particles <2.5 µm goes in deeper past the alveoli and cause damage. Particle chemistry, depending on the toxicity of the composition also disrupts chemical exchange processes in the respiratory system causing airway injuries and inflammation.

This study aims to characterize the typical concentration, variability and chemical characteristics of indoor airborne particulate matter in typical households of a low-income settlement in the Vaal Triangle in order to inform future health assessments in terms of sources of household energy. The following objectives have been derived to achieve the aim of the study:

- Determine the dominant household energy carriers used by the low-income community;
- Evaluate the concentration and variability of indoor PM_4 within typical low-income households; and
- Evaluate the elemental composition characteristic of indoor airborne PM in typical low-income households in Sharpeville.

Part of this work has been presented at:

- National Association for Clean Air (NACA) Conference in Vanderbijlpark, Vereeniging November 2018.

This dissertation consists of 6 chapters which are structured as follows:

DOCUMENT STRUCTURE

CHAPTER 1: Introduction

This chapter provides a brief background on indoor air pollution in low-income settlements and the health implications thereof. The aim, objectives and significance of the study are also outlined.

CHAPTER 2: Literature

This chapter provides the literature background on the topic of the study. It covers aspects that frame this study and captures the essential theoretical background for the relevant study.

CHAPTER 3: Experimental methodology

The purpose of this chapter is to explain the methods and instruments used to collect data. The analytical procedures are also comprehensively defined as well as any calculations and formulae necessary. This chapter also covers the reliability and validity of the methods and processes used to collect data as well as the analytical techniques.

CHAPTER 4: Results and discussion (1st and 2nd objectives)

This chapter provides the reader with the results from the questionnaires and continuous monitoring results of Detailed Energy Survey (DES), indoor temperature profile and indoor airborne respirable PM (PM_4) within low-income households in Sharpeville. A comparison with the World Health Organisation's (WHO) indoor temperature standards and National Ambient Air Quality Standards (NAAQS) is made.

CHAPTER 5: Results and discussion (3rd objective)

This chapter represents the results of the collected gravimetric samples from two contrasting low-income houses in Sharpeville for both winter and summer 2017. The differences in gravimetric mass concentrations are evaluated, and the elemental characteristics of indoor airborne PM_4 are represented and analysed.

CHAPTER 6: Summary and conclusion

This chapter provides a summary and a brief discussion of the key findings of this study. It also provides and draws overall conclusions based on the research outcomes.

Conference presentations and articles

Conference presentations and articles derived from the study:

Thapelo A. F. Letsholo. Marvin M. Qhekwana, Roelof P. Burger. & Stuart J. Piketh. Characterising indoor airborne particulate matter in Sharpeville, Gauteng. National Association for Clean Air Conference 28 October –01 November 2018.

ETHICAL CONSIDERATIONS

The study received ethical clearance from the North-West University-Health Research Ethics Committee (NWU-HREC) on 7 September 2018, ethical clearance number NWU-00041-17-S1. The researcher also attended a research ethics course offered by the NWU-HREC for Research on Human Subjects on 03 May 2016 and completed an online ethics training course offered by the Committee.

GLOSSARY OF ABBREVIATIONS AND ACRONYMS

List of abbreviation	
µg/m^3	Micrograms per cubic meter
Al	Aluminium
BC	Black Carbon
BrC	Brown Carbon
C1	Component 1 from the PCA factor analysis
Ca	Calcium
Cl	Chlorine
CMB	Chemical Mass Balance
D/N	Day/Night
DEA	Department of Environmental Affairs
DEAT	Department of Environmental Affairs and Tourism
DES	Detailed Energy Survey
EF	Enrichment Factors
EPA	Environmental Protection Agency
Fe	Iron
H01-17	The designated household number for research purposes
HPA	Highveld Priority Area
ISO	International Organization for Standardization
K	Potassium
Mg	Magnesium
NAAQS	National Ambient Air Quality Standards
NEM: AQA	National Environmental Management Air Quality Act 39 of 2004
NIST	National Institute of Standards and Technology
NSFB	Non-Solid fuel burning
P	Phosphorus
p.h/p.m	*per household per month
p.p/p.d	per person per day
PAH	Polycyclic Aromatic Hydrocarbons
Pb	Lead

List of abbreviation	
PCA	Principal Component Analysis
PM	Particulate matter
PM_{10}	A coarse aerosol particle with an aerodynamic diameter of 10 microns
$PM_{2.5}$	A fine aerosol particle with an aerodynamic diameter of 2.5 microns
PM_4	Respirable aerosol particle with an aerodynamic diameter of 4 microns
PMF	Positive Matrix Factorisation
RDP	Reconstruction and Development Program
ROS	reactive oxygen species
S	Sulphur
SFB	Solid Fuel Burning
Si	Silicon
UNDP	United Nations Development Programme
US	United States
VTAPA	Vaal Triangle Airshed Priority Area
WBPA	Waterberg-Bojanala Priority Area
WD-XRF	Wavelength Dispersive X-Ray Fluorescence
WHO	World Health Organization
XRF	X-ray fluorescence
Zn	Zinc

CHAPTER 1: INTRODUCTION

CHAPTER OVERVIEW: This chapter provides a brief background on indoor air pollution in low-income settlements and the health implications thereof. The aim, objectives and significance of the study are also outlined.

1.1 Introduction

Air pollution is one of the leading causes of premature deaths worldwide (Ezzati & Kammen, 2002; Sidhu *et al*., 2017). This is due to significant health implications associated with exposure to harmful pollutants emitted in the air World Health Organization (WHO), 2015). The impacts of air pollution are dependent on the number of sources, quantity of emissions and the dispersion potential of the atmosphere. Both health and environmental impacts are possible from high levels of air pollution. Health impacts are directly linked to the ambient or indoor concentrations and thus individual exposure. Historically, air pollution was considered to be linked mostly to heavily industrialised areas (Scorgie, 2012). These areas are infamous for their air-quality-degrading activities that result from manufacturing processes. The areas are typically surrounded by low-income settlements enabling residents to easily commute to work by means of public transport or on foot (Van den Berg, 2015), but in the process exposing them to the air pollution. In South Africa the largest sources of air pollution include electricity generation, domestic solid fuel combustion, agricultural and waste burning, manufacturing industries and exhaust emissions from vehicles and fugitive dust (Piketh *et al*., 1999; Klausbruckner *et al*., 2016). Besides industrial and related activities, in certain areas, high particulate pollution can be attributed directly to household solid fuel combustion (Scorgie, 2012).

Household solid fuel burning (SFB) is an activity that has been practised for centuries and for various reasons. The on-going use of solid fuels as primary energy carriers is common to over 2.8 billion households globally, despite the widespread availability of electricity, natural gas and other cleaner alternatives (Gordon *et al.,* 2014; Bruce *et al.*, 2015; Poddar & Chakrabarti, 2015; WHO, 2015; Language *et al.,* 2016; Li *et al*., 2017). Households within low-income communities commonly use wood, coal, animal dung and crop waste as energy sources to complete everyday household activities (Chafe *et al*., 2014; Wernecke *et al.*, 2015). Domestic solid fuel combustion often takes place in inefficient, simple stoves characterised by poor combustion design (Masekameni, 2015). This results in emissions of pollutants associated with incomplete combustion of which particulate matter forms an important part (Wernecke *et al.*, 2015; Sulaiman *et al*., 2017; Sidhu *et al*., 2017; Li *et al*., 2017).

Domestic SFB is a key contributor to particulate emissions in most regions globally. For instance, 50–70% of fine particulate emissions in China can be directly linked to household burning of coal and biomass fuels (He *et al*., 2004). In India, 60–90% of particulate emissions and organic carbon

is ascribed to domestic coal combustion (Bond *et al.*, 2004). In addition, Smith *et al*. (2014) articulated that cooking-stoves used within households contribute a significant amount of particulate emissions to the ambient environment.

The connection between energy use and socio-economic factors cannot be severed because of both factors having an impact on quality of life (Masekameni, 2015). Developing countries such as South Africa are often characterised by low-income settlements where solid fuels are perceived as a primary commodity by the local communities (Bai *et al.,* 2003). Socio-economic factors, geographical location and population density affect the use, access and reliance on energy resources (Kimemia & Annegarn, 2011); thus solid fuel use remains a widespread practice within developing economies.

The majority (>75%) of African households across the continent are largely dependent on solid fuels as their primary energy carrier (Barnes, 2014). Despite domestic coal burning, biomass fuel (wood-based biomass) is a dominant energy source in nearly all parts of the world where wood is in abundance. An estimated 81% of African households rely on a mixture of solid fuels, with about 70% primarily dependent on wood as a primary fuel for cooking and space heating (Africa Renewable Energy Access Program (AFREA), 2011). Sub-Saharan African households consume large amounts of wood as it provides at least 80–90% of the households' energy demand in the region, and of which South Africa is an integral part (Ezzati & Kammen, 2002; Wessels *et al.,* 2013; Sulaiman *et al.,* 2017). South Africa is just as reliant on solid fuels as an energy carrier, regardless of being a relatively well-developed country with improved electricity infrastructure and other cleaner sources of energy (Shackleton and Shackleton, 2004; Wessels *et al.,* 2013). A study by Pereira *et al.* (2011) found that 55% of the total low-income houses in most parts of South Africa have access to electricity; nonetheless, persistent wood use is still prevalent in 54% of these households. Approximately 60% of the total population rely on wood and coal as their main energy sources for household activities (Doppegieter *et al.,* 1998). Domestic coal use accounts for 3% (3.3 million tons) of the total annual use of the country's coal used by approximately 950 000 South African households (Mdluli, 2007; Van den Berg, 2015). The above-mentioned amount is attributed to a 36% contribution to national particulate emissions and over 20% of the total air pollution (Mdluli, 2007).

Approximately 7 million premature deaths per annum are attributed to air pollution.with4.3 million of these deaths being ascribed to indoor air pollution (Lim *et al.,* 2012; Bruce *et al.,* 2015). According to Lim *et al.* (2012), indoor air pollution constitutes 2.7% of the Global Burden of Disease. Domestic coal combustion was ascribed as the cause of 2.5 million premature deaths in the year 2000, 3.5 million in 2010, and 3.9 million in 2013 (Lim *et al*., 2012; Smith *et al*., 2014).

Wood and coal are often a staple option for most households, as they are frequently found to be cheaper. Ongoing dependence on these fuels for household activities is a major health risk. Indoor air pollution most affects the health of children, the elderly and women (Ezzati & Kammen, 2002; Poddar & Chakrabarti, 2015; Li *et al*., 2017). This is mainly due to the amount of time spent indoors being significantly more than that spent outdoors by these parties (Ashmore & Dimitroulopoulou, 2009). On average, the general population spends 86–87% of their time indoors and for children, it is 89–90% (Ashmore & Dimitroulopoulou, 2009). This makes the residential environment an important setting for exposure to vulnerable parties (Batterman *et al*., 2012). Consequently, exposure to indoor air pollutants poses a major health risk which can lead to cardiovascular and respiratory diseases (WHO, 2012). It can lead to occupants contracting pneumonia, ischaemic heart disease, lung cancer, acute lower respiratory infections and other respiratory dysfunctions (Fullerton *et al*., 2008; WHO, 2012).

Although there are several sources of air pollution contributing to health implications, domestic solid fuel combustion has by far the largest impact in South Africa, most especially within low-income settlements (Friedl *et al.,* 2008). For several decades, continued use of solid fuels within households in South Africa has been linked with childhood illness and increased mortality rates (Norman *et al.,* 2007; Barnes *et al.,* 2009; Barnes *et al.,* 2011). An estimated 24 893 deaths across South Africa have been attributed to domestic solid fuel combustion (Norman *et al.,* 2007; Friedl *et al.,* 2008), and an annual premature death rate of 9 000 South African residents (Norman *et al.,* 2007). Exposure to indoor air pollutants as a function of domestic solid fuel use was responsible for the premature deaths of approximately 1 400 children per year (Barnes *et al.,* 2009).

Unprocessed/poor-quality fuel and inadequate appliance designs, ignition methods and ventilation are major contributors of indoor PM loading in low-income households (Masekameni, 2014). Other factors contributing to continued use of solid fuels in South African low-income households include the backlog in provision of basic human services (Naidoo *et al.,* 2014) and the relative availability and affordability of solid fuels within low-income areas. Therefore, people find it easy to resort to solid fuels to meet their daily energy requirements and compensate for inadequate electricity provision as well as expensive tariffs.

South Africa, classified as a developing country, is home to a large number of low-income settlements. The South African Government, through a government initiative known as the Reconstruction and Development Program (RDP), built these settlements as a low-cost housing development strategy (Language *et al.,* 2016). The development of these settlements took place in existing "*townships"* built on rural areas and urban peripheries. These low-cost households were built using cheap building materials which created socio-economic issues for the residents. Problems associated with the RDP included the inability to effectively address the issues of

poverty within low-income areas, and due to the poor-quality structural material used, houses rapidly deteriorated and needed maintenance which was too expensive for these communities (Goebel, 2007). The poor-quality structural design of the houses can lead to a less desirable thermal environment. Thermal comfort can be one of the factors driving domestic solid fuel use. It is described as a human perspective where satisfaction with the surrounding environment is reached; and/or a state where perceived driving forces lead inhabitants to attempt to create environmental conditions that are thermally comfortable (Centnerova & Hensen, 2001; Djongyang *et al.*, 2010). In achieving a satisfactory state of thermal comfort, residents of low-income houses concentrated in RDP residential areas in poverty-struck communities tend to resort to indoor heating using solid fuels.

The Bill of Rights contained in the South African Constitution Section 24 (1996) ensures a safe environment that is not harmful to the health and wellbeing of the people. Aligned to this Bill of Rights, a series of environmental regulations was developed by the South African Government to deal with a range of environmental issues. The first air pollution law was the Atmospheric Pollution Prevention Act (APPA) Act 45 of 1965 which was mainly focused on point source emissions and disregarded receptor sites. Since the development of the National Environmental Management: Air Quality Act (Act No. 39 of 2004) (NEM: AQA), receptor site impact consideration became regarded as a highly important part of air quality management in South Africa. The NEM: AQA, air pollution hotspots were classified as air pollution priority areas, namely the Vaal Triangle Airshed Priority Area (VTAPA), Highveld Priority Area (HPA) and the Waterberg-Bojanala Priority Area (WBTA).

The VTAPA was the first to be declared an air pollution priority area in terms of section 18 of the NEM: AQA due to the observed increase of pollutant concentrations in the vicinity, specifically particulates. With a high concentration of industrial, mining, commercial, agricultural and residential land use close to one another, the Vaal Triangle was faced with concerning air pollution issues (Scorgie *et al.,* 2003). Solid fuels, mainly wood and coal, continue to be used by surrounding low-income areas including Evaton, Sebokeng, Sharpeville, Zamdela, Bophelong, and Boipatong. Domestic solid fuel use continues within the area mainly due to: (i) accelerated urbanisation; (ii) increased backlog to basic service delivery such as electricity provision by the rapidly growing informal settlements; and, (iii) the use of solid fuels due to their cost-effectiveness and convenience, providing a dual function of simultaneously cooking and space heating (Scorgie *et al.,* 2003).

The purpose of this study is to measure, evaluate and understand the elemental characteristics of indoor airborne respirable particles (PM_4) within the Sharpeville low-income community. Particulate matter is an imperative pollutant attributable to its significant impacts on human health. Understanding the complex nature of PM with regard to its origins and chemical composition is

important for this research experiment. Indoor measurements of PM are compared to the National Ambient Air Quality Standards (NAAQS). Although the methods of measurement for indoor PM may be different to when measuring ambient PM, because the health impacts are similar they are applicable for this study. There are currently no clear benchmark concentration levels associated with health impacts, although the NAAQS serve as a guide to acceptable levels of air pollution (Vanker *et al.,* 2015). Knowing and understanding the chemical composition of PM aids in the identification of the source (Liu *et al.,* 2005). Therefore, it is important for this research experiment to evaluate the typical chemical species found in PM concentrated within low-income households as a function of domestic use of solid fuels.

1.2. Problem statement

Domestic solid fuel use is a common predominantly within low-income communities (Smith *et al.,* 2014). The emphasis on indoor air pollution within low-income households is elevated by the increased degree of human exposure to health-damaging pollutants such as PM (coarse/fine) produced by combustion of solid fuel material (wood, coal, animal dung and agricultural waste). Certain factors such as combustion appliance conditions, type and quality of the fuel and ventilation contribute greatly to the level of pollution within low-income households (DEA, 2012; Masekameni, 2014). Domestic solid fuel combustion accounts for 75% of premature deaths in South Africa, with particulate matter being one of the most notorious pollutants to cause respiratory infections/illness (Barnes, 2014; Masekameni, 2015). Fine particulate matter has the ability to penetrate deep within the human respiratory system into the lower respiratory tract and cause harm.

Indoor air pollution from solid fuel use has been ranked the 7th leading risk factor in the Global Burden of Disease, Injury, and Risk Factor Study 2013 (Li *et al.,* 2016). It should further be noted that developing nations suffer more from exposure to indoor airborne PM. This is true in terms of the number of people exposed, intensity of exposure and time spent indoors compared to those residing in the developed world (Fullerton *et al.,* 2008). Thus, exposure to indoor air pollution prematurely claimed the lives of 425 000 people in China during the year 2000, and for the same period up to 875 000 premature deaths in India (Rohra & Taneja, 2016).

Previous research (Hoets, 1998; Kimemia *et al.,* 2014; Naidoo, 2014; Jafta *et al.,* 2017) has shown that despite the electrification program designed to reduce domestic solid fuel combustion within South African low-income settlements, the practice is still prevalent and has likely been exacerbated over the past decade by increasing electricity process and poor reliability of electricity supply. As a result, indoor air pollution and the health implications associated remain a major air quality issue in South Africa costing the Government approximately R1.2 billion per year in illness-related costs (Masekameni, 2015).

The VTAPA experiences elevated levels of air pollution due to the conglomeration of industrial, mining, power generation, commercial and agricultural activities. It is also home to a significant number of low-income settlements, which contribute to the pollution levels in the region as a function of their burning practices of solid fuels and waste. Sharpeville is one of the low-income communities located within the Vaal Triangle and surrounded by major industrial and mining activities. Adding to the pollution coming from industrial activities, the Sharpeville community uses typical solid fuels (wood, coal and paraffin) for domestic purposes.

Although the Vaal Triangle has numerous sources of PM, particulate emissions that have a significant impact on human health are those that occur within residential dwellings. Airborne particles have distinct characteristics determined by their origin, solubility, size and chemical composition (Pope & Dockery, 2006). Throughout the history of indoor air pollution research, particulate matter of the aerodynamic size 10 and 2.5 microns has received significant attention. Particulate air pollution monitoring measures various particle sizes of special relevance to inhalation and deposition, sources, or toxicity (Pope & Dockery, 2006). This research focuses on the respirable size fraction of PM (PM_4) in Sharpeville because no indoor physical measurements for PM have been conducted in the community before. PM of a size fraction 4–10 µm has a relative 50% penetration into the respiration tract (Brown *et al.,* 2013). This means PM_4 is expected to be deposited in the gas-exchange region of the lungs and cause harm (Brown *et al.,* 2013). Moreover, the study aims to determine source contribution to indoor airborne PM using receptor models. This aims to, at least partially fill data and knowledge gaps regarding source contribution to indoor airborne PM from outdoor sources in South African low-income households.

1.3. Aim

This study aims to characterise typical low-income household concentrations of airborne respirable PM (PM_4). Furthermore, the study focuses on the variability of indoor concentrations and chemical characteristics of indoor airborne particles in Sharpeville, Gauteng. The study is designed to inform future health assessments in terms of exposure to indoor air pollution and relative health implications.

1.4. Objectives

- Determine the dominant household energy carriers used by the low-income community;
- Evaluate the concentration and variability of indoor PM_4 within typical low-income households; and
- Evaluate the elemental composition characteristic of indoor airborne PM in typical low-income households in Sharpeville.

1.5. Significance of the study

Indoor air quality assessments are important to understand due to the interconnection between exposure to pollution and human health deterioration. Typically, particulate pollution within low-income settlements reach concentrations 50% higher than neighbouring urban areas and also >75% in contrast to industrial PM emissions (Hersey *et al.,* 2015; Pauw, 2017). This accentuates the significance of emissions from residential burning of solid fuel material as basic energy carriers. Therefore, indoor air pollution monitoring within low-income communities in South Africa is important. The health implications resulting from exposure to indoor airborne particles have been well documented under the World Health Organization (WHO) and a significant amount of indoor air pollution research (WHO, 2014; WHO, 2015). These health impacts range from acute lower respiratory infections to cardiovascular dysfunctions. Therefore, it is important to understand the characteristics of respirable indoor airborne PM in Sharpeville because they can reside in the air for longer and have a high potential of causing respiratory illness and cardiovascular diseases (Arora *et al.,* 2013).

The study site selected for this investigation is Sharpeville, a low-income settlement located within the VTAPA. The Vaal Triangle was declared an Air Pollution Priority Area under section 18(1) of the NEM: AQA (Act No. 39 of 2004) by the Minister of Environmental Affairs and Tourism. The VTAPA was the first South African priority area to be declared due to observed elevated air pollution within the vicinity, most especially particulates. Engelbrecht *et al.,* (1998) conducted a study in the VTAPA region and found that the combustion of solid fuel material for residential basic energy needs contributed 36.5% to ambient particulate loading and is elevated to 65% during winter.

As a product of incomplete combustion, indoor airborne PM has various characteristics including size, density, and chemical composition (Pipal *et al.,* 2014). The size fraction and chemistry of particulate matter are the two important factors that govern the health impacts on human wellbeing (Brook *et al.,* 2010). Particulate emissions from combustion processes are often composed of chemical compounds such as elemental and organic carbon, sulphates and nitrates (Pope & Dockery, 2006). Due to inefficient combustion characteristic of traditional stoves found within low-income settlements, coarse (> 10 µm), respirable (4 µm) and fine (< 2.5 µm) are typical products (Akhtar & Palagiano, 2017). This study focuses on the respirable size fraction (4 µm) of indoor airborne particulate matter and the chemical characteristics of these particles. This adds to the body of knowledge with respect to typical concentration levels of respirable PM and chemical composition characteristic of indoor air quality in low-income households for future health assessments. The importance of PM_4 in this study is derived from the underestimation of its ability to deposit beyond the terminal bronchioles and settle in the gas-exchange area of the lungs (Brown *et al.,* 2013; Kawata *et al.,* 2018). The reliance on solid fuel use by low-income

communities makes them more vulnerable to respiratory illness. Thus the health implications associated with exposure to indoor PM are important and need to be addressed. Identifying and quantifying indoor concentrations of PM aids in determining related health impacts and premature death tolls in low-income areas. The information provided on this piece of research may inform decision- and policy-makers regarding focus areas for strategies and interventions intended to reduce air pollution within low-income settlements. It also builds awareness with respect to the disadvantages of residential solid fuel use. Furthermore, this study provides an indication of how much PM_4 each household is likely to contribute to the PM_4 loading at a local scale within low-income settlements. Source contribution to indoor airborne PM has not yet been extensively researched in South African low-income communities, which thus makes this research an important contribution to the scientific community dealing with indoor air quality in South Africa.

CHAPTER 2: LITERATURE STUDY

CHAPTER OVERVIEW: This chapter provides the literature background on the topic of the study. It covers aspects that frame this research and captures the essential theoretical background relevant to the study.

2.1. Study background

2.1.1. Importance of atmospheric aerosols

Atmospheric aerosols are pollutants which have a major influence on weather and climate. Their influence on climate relies on three factors, namely: spatial and temporal distribution; optical properties; and hygroscopic ability (Charlson *et al.,* 1992). Particles in the atmosphere have a direct and indirect effect on the climate. The direct scattering of incoming short wavelength solar radiation and alterations on the reflective properties of clouds which result in an increased planetary albedo have a cooling effect on the planet (Charlson *et al.,* 1992). The reflection or backscattering occurs often when there is an elevated concentration of sulfate aerosols in the atmosphere (Charlson *et al.,* 1991). Absorption of shortwave solar radiation by aerosol particles occurs when carbonaceous aerosols (organic and/or black carbon) are highly concentrated in the air (Lammel *et al.,* 1995; Ramanathan *et al.,* 2001) which results in the warming of the atmosphere. In most cases, clouds with concentrations of black carbon-composed aerosol particles can inhibit the formation of clouds or reduce local cloud cover (Ramanathan *et al.,* 2001). This means the stability of the atmosphere will change as well as the radiation budget.

The indirect effect of aerosols occurs through acting as Could Condensation Nuclei (CCN) altering cloud properties (Lohmann & Feichter, 2005). This means rain droplets form due to the hygroscopic properties of the particles because the solutes found in aerosols can retain water in the liquid form and therefore condensation occurs. The collective impact aerosols have on climate–forcing is a complex phenomenon to quantify because of their short residence time and spatial distribution (Jimoda, 2012). However, the smaller (finer) the particle, the larger the impact (local to global scale) and the larger (coarser) they are the more localised the impact is.

2.2. Particulate matter: definition, origin and characteristics

The Earth's atmosphere is made up of gases, water vapour and aerosol particles which play an important role with respect to the physics and chemistry of the atmosphere (Barry & Chorley, 2010). Aerosols are commonly referred to as the suspension of liquid and solid particles in the air which occurs either directly from a source or indirectly in terms of chemical change in gases (Pöschl, 2005). According to Seinfeld & Pandis (1998), an aerosol consists of a single continuous unit of solid or liquid containing several molecules combined by intermolecular forces and primarily larger than molecular dimension (0.001 µm). It may also consist of two or more unit

structures combined by inter-particle adhesive factors such that it behaves as a single unit in suspension or deposition (Seinfeld & Pandis, 1998).

Due to the stable and negligible fall velocity of aerosol particles, they are the most common and persistent pollutants found in the atmosphere (McCormick and Baulch, 1962). Consequently, the close correlation between adverse health impacts and aerosol loading makes particulate pollution a major air quality problem (Seaton *et al.,* 1995; Smith *et al.,* 2000; Kim *at al.,* 2015). The chemistry and size of aerosols controls particulate loading at a given space and time (Friedlander, 1970). The toxicity of particles is determined by the availability of toxic compounds which will be found inside the particle or on the surface of the particle (Kelly & Fussell, 2012; Van den Berg, 2015). This exemplifies the complex nature of the aerosol chemical composition. Moreover, the chemistry of particulates differs per size fraction. Fine particles (< 2.5 µm) often consists of soluble and insoluble compounds including ammonium (NH_4^+), sulfate (SO_4^2), nitrates (NO_3^-), organic and elemental carbon (OC/EC), whereas coarse particles (> 10 µm) comprise mostly crustal elements such as aluminium (Al), iron (Fe), calcium (Ca), chlorine (Cl) and silicon (Si) (Watson *et al.,* 1997; Aneja *et al.,* 2006; Liu *et al.,* 2016).

To identify the source of PM, elemental data plays an important role as it provides the chemical characteristics of the particulates and the information can be also be used to evaluate the impact on human health, ecology and the environment (Pipal *et al.,* 2014). Sources of ambient PM can be determined using their chemistry because every source has unique chemical characteristics and thus can be distinguished (Table 1) (Watson *et al.,* 1997). Thus the chemistry of particulate samples will often correlate with the composition found at the source. However, indoor concentrations of PM are more complex in terms of chemical composition due to the influence of ambient sources as well as indoor activities (Tunno *et al.,* 2016). The likelihood of outdoor sources having an impact within residential areas is very high and the pollution from outside will mix with the indoor pollution.

Table 1: Chemical species classified per emission source per particle size fraction (Watson *et al.,* 1997).

Source	**Particle size**	**Chemical Abundances in % Mass**			
		< 0.1%	**0.1 to 1%**	**1 to 10%**	**> 10%**
Paved Road Dust	Coarse	CR, SR, Pb, Zr	SO_4^{2-}, Na^+, K^+, P, S, Cl, Mn, Zn, Ba, Ti	EC, Al, K, Ca, Fe	OC, Si
Unpaved Road Dust	Coarse	NO_3^-, NH_4^+, K^+, P, S, Cl, Ti	SO_4^{2-}, Na^+, K^+, P, S, Cl, Mn, Ba, Ti	OC, Al, K, Ca, Fe	Si
Construction	Coarse	Cr, Mn, Zn, Sr, Ba	SO_4^{2-}, K^+, S, Ti	OC, Al, K, Ca, Fe	Si
Agricultural Soil	Coarse	NO_3^-, NH_4^+, Cr, Zn, Sr	SO_4^{2-}, Na^+, K^+, S, Cl, Mn, Ba, Ti	Oc, Al, K, Ca, Fe	Si

Source	Particle size	Chemical Abundances in % Mass			
		< 0.1%	0.1 to 1%	1 to 10%	> 10%
Natural Soil	Coarse	Cr, Mn, Sr, Zn, Ba	Cl^-, Na^+, EC, P,S, Cl, Ti	Oc, Al, Mg, K, Ca, Fe	Si
Motor Vehicle	Fine	Cr, Ni, Y	NH_4^+, Si, Cl, Al, Si, P, Ca, Mn, Fe, Zn, Br, Pb	Cl^-, NO_3^-, SO_4^{2}, NH_4^+, S	OC, EC
Vegetative Burning	Fine	Ca, Mn, Fe, Zn, Br, Rb, Pb	NO^-, SO_4^{2}, NH_4^+, S, N^+	Cl^-, K^+, Cl, K	OC, EC
Residential Oil Combustion	Fine	K^+, OC, Cl, Ti, Cr, Co, Ga, Se	NH_4^+, Na^+, Zn, Fe, Si	V, OC, EC, Ni	S, SO_4^{2}
Incinerator	Fine	V, Mn, Cu, Ag, Sn	K^+, Al, Ti, Zn, Hg	NO_3^-, Na^+, EC, Si, S, Ca, Fe, Br, La, Pb	SO_4^{2}, NH_4^+, CO, Cl
Coal-fired boiler	Fine	Cl, Cr, Mn, Ga, As, Se, Br, Rb, Zr	NH_4^+, P, K, Ti, V, Ni, Zn, Sr, Ba, Pb	SO_4^{2}, OC, EC, Al, S, Ca, Fe	Si
Marine	Fine & Coarse	Ti, V, Ni, Sr, Zr, Pd, Ag, Sn, Sb, Pb	Al, Si, K, Ca, Fe, Cu, Zn, Ba, La	SO_4^{2}, NO_3^-, S, OC, EC	Cl^-, Na^+, Na, Cl

Aerosol particles are distinguished according to their size distribution which occurs in three phases: nucleation; accumulation; and coarse (Chow *et al.,* 1995). The nucleation and accumulation modes are both fine particles with the nuclei mode being 0.005–0.1 µm in diameter accounting for the predominance of particles in the atmosphere, as they will have higher residence time due to their ability to be suspended longer (Seinfeld & Pandis, 1998). Accumulation mode consists of particles with a sizes of 0.1–2.5 µm in diameter. The secondary source of these particles is the coagulation of nuclei mode particulates from the condensation of vapour onto existing aerosol particles which causes them to grow in size (Seinfeld & Pandis, 1998). Particles in the accumulation mode reside for an extended period in the atmosphere as well, due to particle removal mechanisms being least effective (Hicks *et al.,* 2016). The coarse mode is > 2.5 µm in diameter and is mostly from natural (volcanic eruptions, windblown dust) as well as anthropogenic sources (industry, vehicle emission, power generations, and domestic fuel burning) (Pandis *et al.,* 1992; Seinfeld & Pandis, 1998; Cyrys *et al.,* 2003). The smaller the particle the longer it stays suspended, whereas large particles have a short lifespan. Coarse aerosol particles have a few days' lifespan in the air where fine particles can be suspended for weeks (Tyson & Preston-White, 2000).

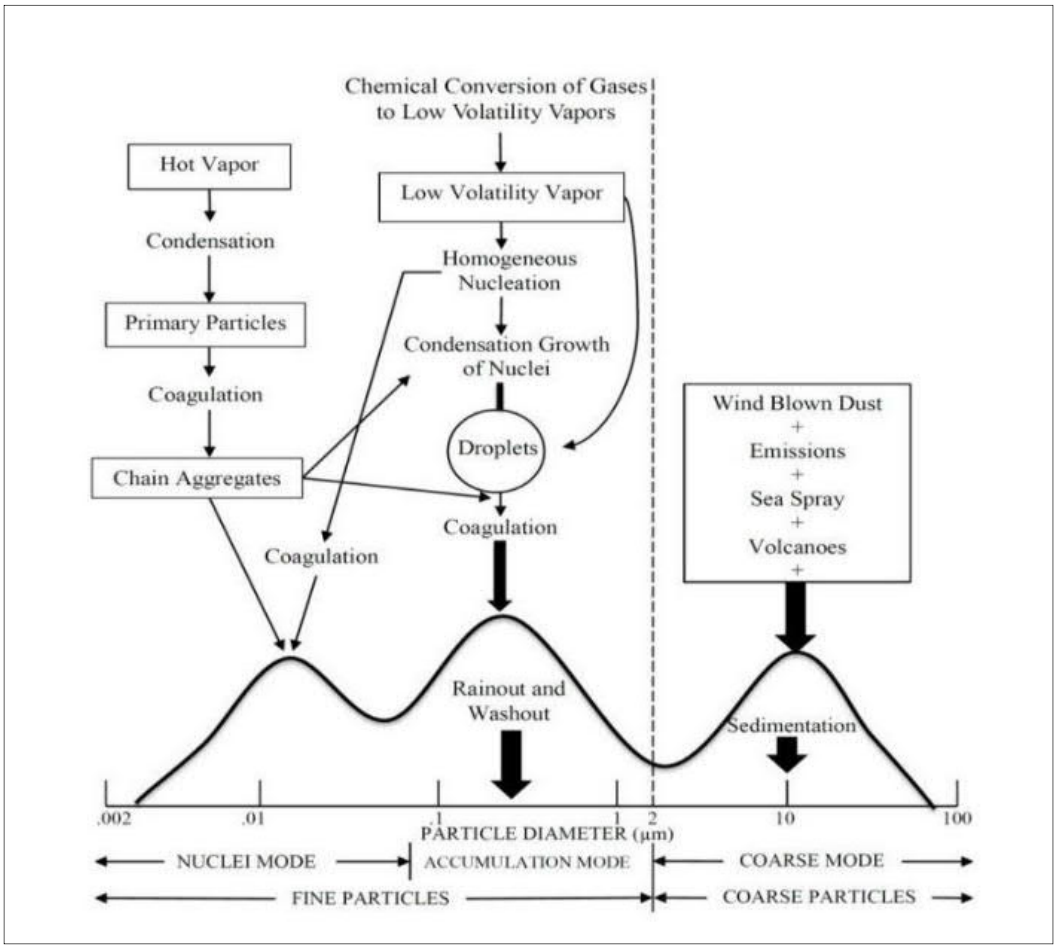

Figure 1: Schematic ideal for showing the distribution of particle surface area, size, source, formation and removal mechanisms (Seinfeld & Pandis, 1998).

2.3. Household solid fuel use in developing countries

In most high-income countries, only about 5% of the population uses solid fuels (Bonjour *et al.,* 2013). In contrast, approximately 60–95% accounts for the total energy consumption in developing countries (Leach, 1992). Furthermore, the world's population using solid fuels is highly concentrated in developing nations constituting of over 80% of households within these nations primarily reliant on solid fuels material {International Energy Agency (IEA), 2010}. Modern energy developments are increasing; however, households in developing countries continue to have limited access to these energy services (IEA, 2010; Bonjour *et al.,* 2013). The elevated use of fuels such as coal and biomass is observed in China, India and most African countries (Holdren *et al.,* 2000; Muller & Yan, 2016). Approximately 87% of the projected 1.2 billion people who will still be using solid fuels by 2030 due to insufficient access to electricity will reside in low-income areas in third world countries (IEA, 2010). An estimated 77% and 61% of the population in Africa and Southeast Asia, respectively, are reliant on solid fuels for daily domestic activities (Bonjour *et al.,* 2013). This rendered the two regions the highest-ranking in terms of domestic solid fuel use per household compared to other regions in 2010. The use of solids fuels is high during the cold winter months compared to elevated summer temperatures (Language *et al.,* 2016; Junaid *et al.,* 2018). Developing countries have more than 75% of their population reliant on solid fuels

which is attributable to the lack of electrical infrastructure, access to electricity and low per capita income {United Nations Development Programme (UNDP) & WHO, 2009}.

In Africa, over 80% of the low-income areas are reliant on biofuels for domestic cooking and space heating (Bonjour *et al.,* 2013). The abundance and accessibility of wood in this region allows residents to alternate between clean and dirty fuels. Malawi and Mozambique have an estimated 95% and 85% of the rural households burning solid fuels for daily domestic activities whereas Nigeria has about 65% (Bonjour *et al.,* 2013; Isara & Aigbokhaode, 2014; Chinyandura, 2016). The national income and household fuel use are closely associated with each other but the use of solid fuels for residential purposes can vary substantially despite this relationship (Bonjour *et al.,* 2013). The variation is governed by a variety of factors such as availability of coal, biomass and other cleaner alternatives, income distribution across the country and level of development (Bonjour *et al.,* 2013; Haltberg, 2004). In Ghana, the rural areas are more prone to using primarily wood-based fuel (charcoal, firewood) which constitutes > 60% of the population (Van Vliet *et al.,* 2013). Sub-Saharan Africa has by far the highest reliance on biofuels, where approximately 75% of the population use these as a primary source of energy, predominantly wood-based biomass fuel (fuelwood and charcoal) (Sulaiman *et al.,* 2017).

Wood provides at least 80–90% of the households' energy demand in Sub-Saharan Africa, of which South Africa is an integral part (Ezzati & Kammen, 2001; Wessels *et al.,* 2013; Sulaiman *et al.,* 2017). South Africa is similarly dependent on solid fuels as an energy source, despite being a comparatively a well-developed country with improving access to electricity and other cleaner sources of energy (Shackleton and Shackleton, 2004; Wessels *et al.,* 2013). According to Bonjour *et al.* (2013), 70% of households in South Africa were electrified in 2001; nonetheless, one third were reliant on solid fuels for domestic purposes and one fifth of households used paraffin. A study by Periera *et al.* (2011) found that 55% of the total low-income households across South Africa have access to electricity; nonetheless, persistent wood use is still prevalent in 54% of these households. Approximately 60% of the total population relies on wood and coal as their main energy sources for household activities (Doppegieter *et al.,* 1998). Domestic coal use accounts for 3% (3.3 million tons) of the total annual use of the country's coal used by approximately 950 000 South African households (Mdluli, 2007, Van den Berg, 2015). The above-mentioned is attributed to a 36% contribution to the national particulate emissions and over 20% of the total air pollution (Mdluli, 2007).

Regardless of the electrification program achieving over 80% access to electricity in South Africa, most low-income households remain reliant on solid fuels (Mdluli, 2007). It was also found that about 80% of low-income households continue to use solid fuels while having access to cleaner alternatives (Ismail & Khembo, 2015). For instance, according to Norman *et al.* (2007) "in 2001 almost 60% of households in Limpopo, a predominantly rural province, used wood as the main

source of energy for cooking (almost 3 times the national average), while in the more developed province of Gauteng less than 1% of households used wood for cooking". Low-income areas located in the Highveld region have a higher propensity for using coal than most regions due to the abundance of coal mines (Figure 2 & 3). Residential coal combustion in the Vaal Triangle thus contributes 65% of the air pollution (Wagner *et al.,* 2005).

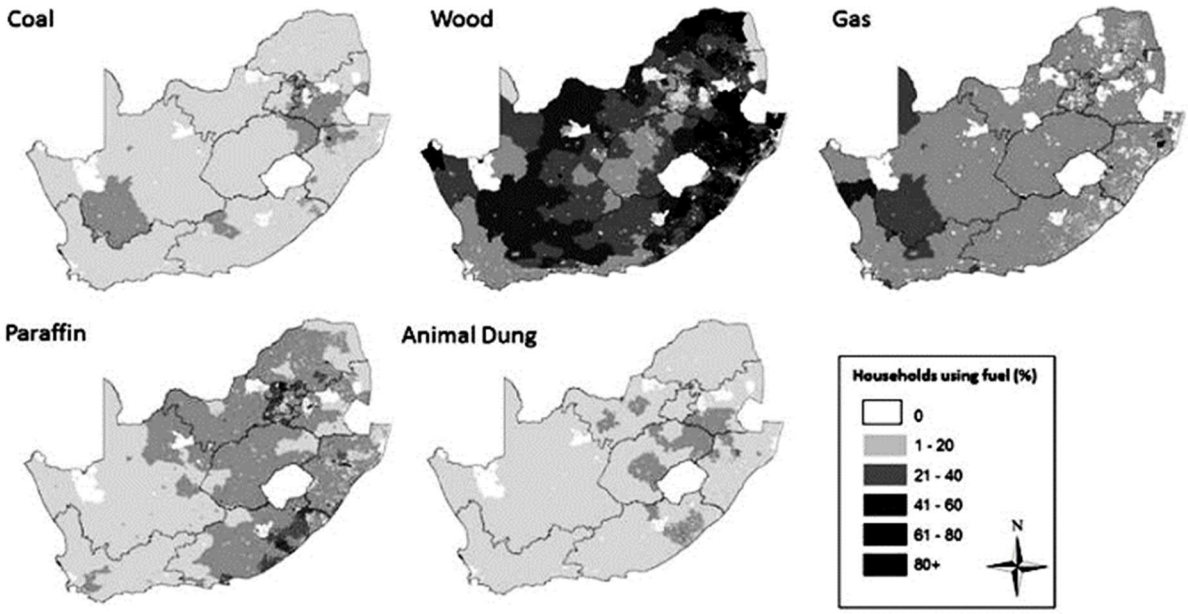

Figure 2: The modelled availability of solid fuels across South Africa (Van Den Berg, 2015).

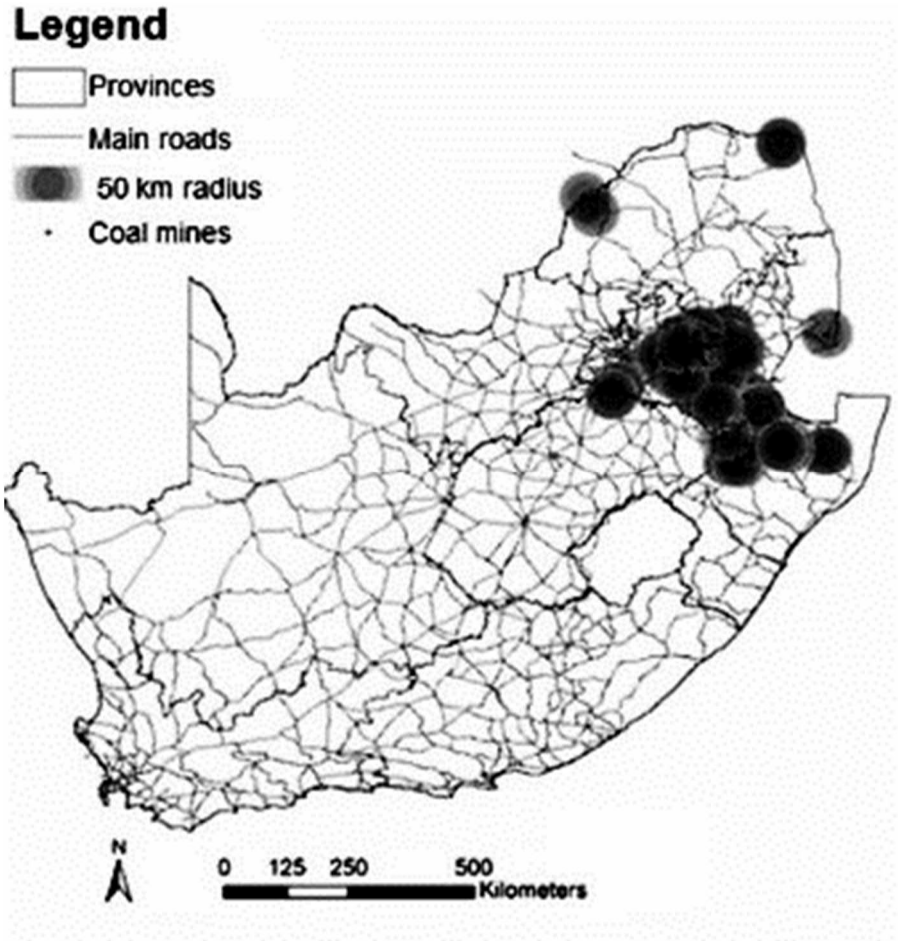

Figure 3: The spatial location of coal mines in South Africa (Van den Berg, 2015).

The industrial and energy sector is responsible for a fraction of the air pollution, whereas the majority (65%) of local air pollution is a function of domestic coal combustion in Gauteng (Scorgie *et al.,* 2003; Balmer, 2007). In comparison, 5% and 30% are contributed by the power generation and industrial sectors respectively (Balmer, 2007).

2.4. Factors determining household fuel choice and the energy ladder theorem

It is prevalent that there are factors that drive households to use certain types of energy sources to meet their daily energy needs. Theories such as the energy ladder model cover most of these fuel selection determinants. The energy ladder model is mostly based on affordability of fuels used within households. Poorer households use less clean fuels (Figure 4; Van der Kroon *et al.,* 2013; Chinyandura, 2016). As household income rises, the more advanced and cleaner the residential fuel gets (Muller & Yan, 2016). Therefore, the energy ladder model is a hierarchical representation of domestic energy use whereby traditional fuels (wood, crop waste and animal dung) are ranked at the bottom, coal, kerosene and charcoal (transition fuels) come second and advanced or modern fuels (LPG, electricity and biofuels) are ranked at the top at the top of the hierarchy (Figure 5) (Schlag & Zuzarte, 2008; Van der Kroon *et al.,* 2013). This explains the impact of income on households' choices in terms of fuel use. It is similar to the consumer economic theory where "*consumers substitute necessary goods and luxury goods for inferior goods"* (Muller

& Yan, 2016). Moreover, households have a wide range of fuels to choose from that are classified according to how advanced they are technologically which is also driven by individual household preferences (Muller & Yan, 2016). It is possible the households switch to more advanced energy sources of fuel but also continue to use traditional fuels. This is a trend that seems to be more prevalent currently that was not observed in the past.

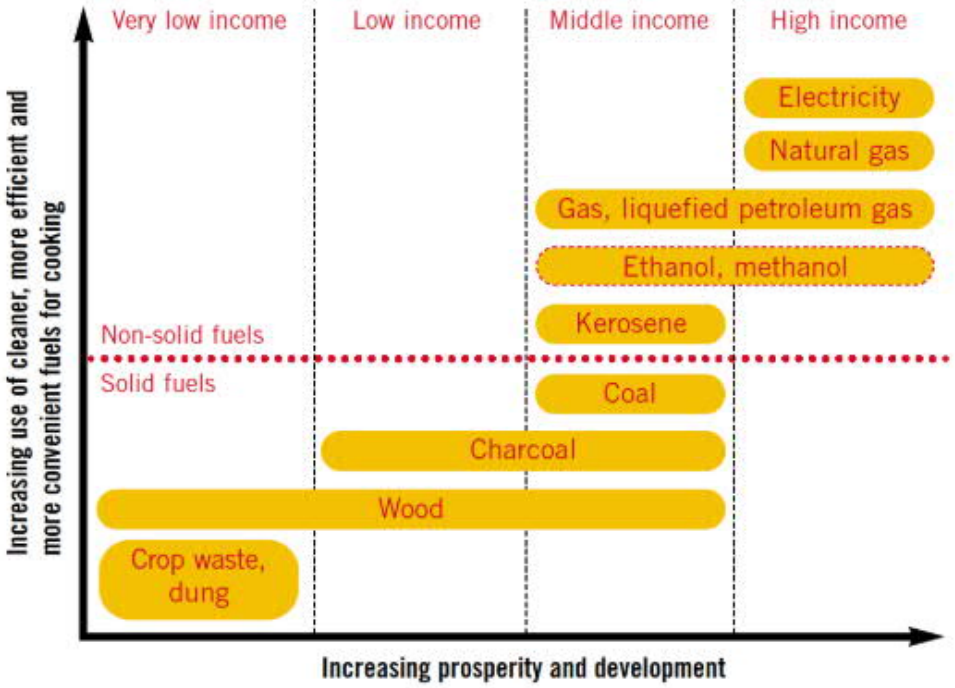

Figure 4: The energy ladder model showing fuel use based on household income (Chinyandura, 2016).

The energy ladder has been mostly used as a concept to account for patterns of domestic solid fuel combustion in third world countries. During the year 2000, a critique of the energy ladder was formalised and an energy stack model was introduced. This is somewhat similar; however, it represents a mixture of household energy sources (Figure 5) (Schlag & Zuzarte, 2008; Van der Kroon *et al.,* 2013). The model emphasizes the importance of fuel switching and different residential energy profiles (Van der Kroon *et al.,* 2013). A critique was developed by Masera *et al.* (2000) on the basis that the energy ladder is inadequate to account for the dynamics of households' fuel use. Rather, in developing countries, fuel stacking is a common practice in both rural and urban and low-income settlements. This model conforms to the multiple fuel use patterns which entail households selecting a mixture of fuels on the lower and upper level of the energy ladder satisfactory to their needs and socioeconomic standards (Masera *et al.,* 2000). Modern fuels are not very reliable; thus households prefer to use two or more fuels that will serve the purpose of substituting modern fuels in the event of temporary unavailability (Van der Kroon *et al.,* 2013). This means modern fuels serve a rather partial role as a substitute than a perfect one for traditional fuels. The multiple fuel use patterns are due to factors such as high and unstable

cost of purchase and maintenance of modern fuel appliances, periodic lack of access to modern fuels, and other circumstantial preferences leading households to not choose modern fuels as their primary energy carriers (Masera *et al.,* 2000; Van der Kroon *et al.,* 2013; Muller & Yan., 2018). Ultimately, fuel stacking reveals the complex nature of household fuel switching and multiple reasons behind residential fuel mixes which are not only limited to income.

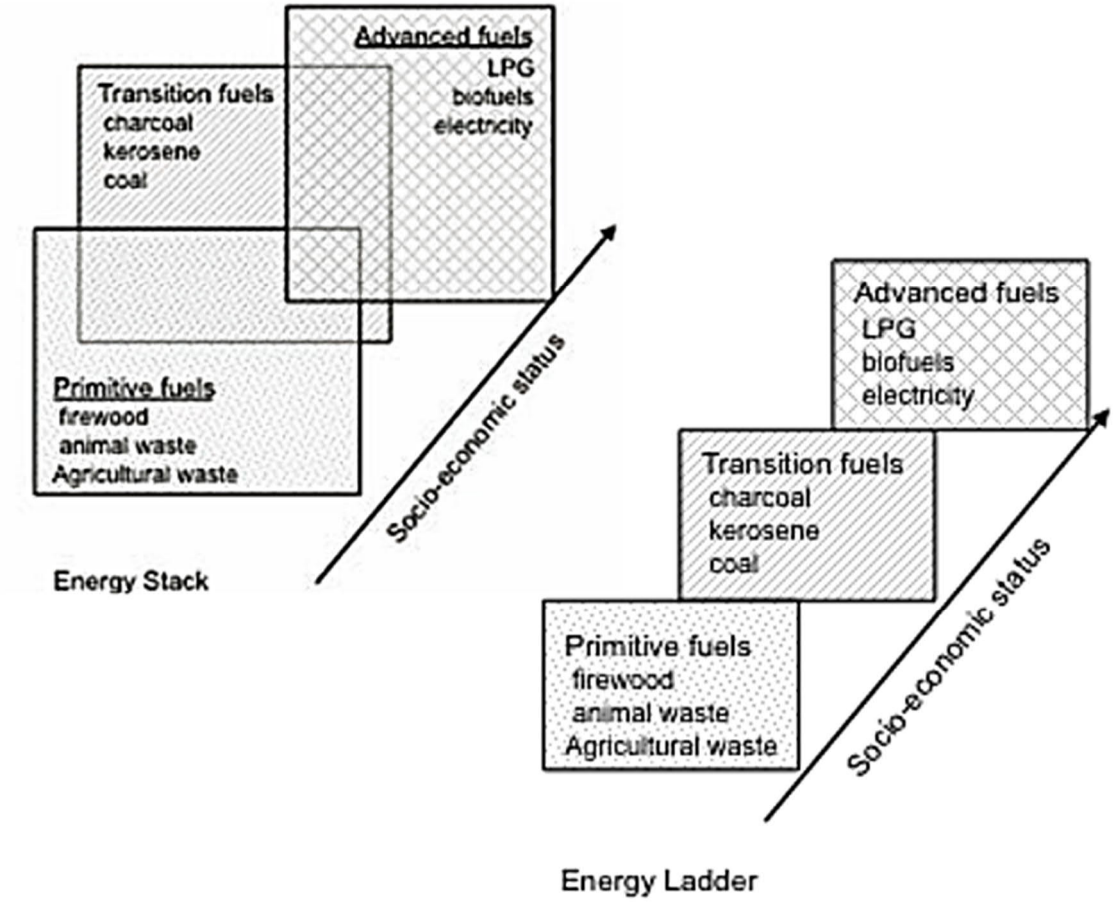

Figure 5: Energy ladder and energy stack models (Schlag & Zuzarte, 2008; Van der Kroon *et al.*, 2013).

2.5. Multiple Inter-Related Factors that Influence Residential Fuel Choice: South African Context

Residential energy use is different across regions due to the variation in availability, affordability, natural climate, individual and national income as well as the level of development (Jiang & O'Neill, 2004). The choice of household fuels is governed by a number of various and interconnected factors influencing the decision making of residents. Each of these factors can differ in terms of time and space (Davis, 1998). These factors are: 1) availability, access and affordability; 2) household demographics; 3) climate impacts; 4) people's culture and traditional beliefs; 5) level of education; and, 6) time spent in relation to fuel type (Figure 6). The importance and presence of these factors vary and are sometimes specific to certain areas. The availability

and access to the various fuels are two important factors because these allow people to actually use a certain energy source. Commercial fuel prices and the level of income per household or per capita governs the affordability aspect of fuel choice. Household demographics comprise household characteristics in terms of size, type and education to name but a few. Climate plays an important role whereby most biofuels rely on certain climate conditions for them to be available. Moreover, the difference between peoples' culture and traditional beliefs plays an important role as a fuel choice determinant. The more educated the community, the more reluctant they become to use "dirty" fuel types. In addition, people are selecting fuel based on how labour-intensive it is and how much time can be saved by using that fuel (Van der Kroon *et al.,* 2013).

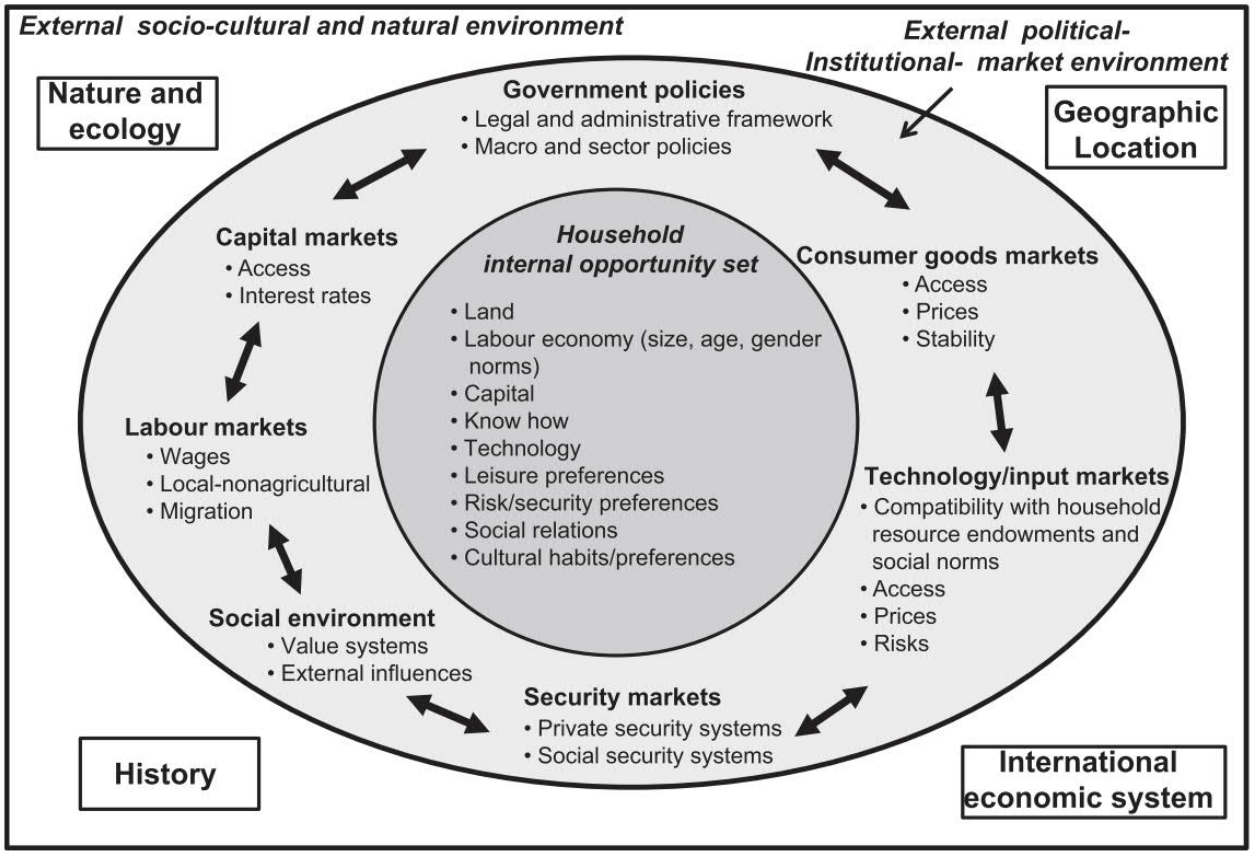

Figure 6: Typical framework that conceptualises residential energy choices and its determents (Van der Kroon *et al.,* 2013).

The following examples of factors that determine residential fuel choice are based on the above conceptual framework.

- **Climate impacts on fuel availability and consumption (geographic location)**

Location prior to other factors plays a pivotal role on fuel choice in households and can, in certain areas, offer residents with natural resources. A natural event such as weather and climate determines which natural fuel types can be available in certain regions. In the instance of communities using biomass (e.g. wood), weather and climate will determine the type of vegetation, moisture and temperature and therefore abundant biomass fuel. Climatic conditions

can be very harsh at times and may not offer the opportunity for ample vegetation growth. This can sway households into use alternative natural fuel types such as coal, animal dung and other alternatives like paraffin (kerosene). Urban and rural areas have local merchants selling traditional household (including transitional) fuels (Nkosi, 2018). Seasonal changes in temperature and local climate affects the availability of these fuels seen as a local commodity. This means the availability of these fuels is controlled by the climate and this will, in turn, affect the pricing of these fuels as well as demand and usage (Mansur *et al.,* 2008; Clarke *et al.,* 2018). Residential space heating during winter increases the demand for solid fuels as temperatures can drop to extreme lows and in summer most households use them solely for cooking. This means the number of households, burning events and time spent burning decrease by almost half in summer. Therefore, particulate pollution in the ambient air is less severe for exposed parties during the season.

- **Availability, access and affordability**

Energy use patterns vary significantly between high-, middle-, and low-income settings as well as between rural and urban areas. These different settings often contribute to the availability, access to the fuel and affordability. Household income and total expenditure allocated to energy are significant factors determining the affordability of fuel types which in certain instances governs access to this energy source (Aitken, 2007). The high- and middle-income areas will often resort to the use of more advanced or sophisticated fuels (electricity and gas) whereas low-income and rural areas tend to use a mixture of electricity and traditional fuels (Heltberg, 2004; Mensah & Adu, 2015). However, it does not mean that high- and middle-income households do not use traditional fuels at all. Traditional fuels become scarcer and costlier to acquire due to logistical reasons in high to middle-income areas thus rarely used (Leach, 1992). Moreover, the gathering of traditional fuels for use in urban areas, besides the transport cost to get the fuel to the town/city, is labour intensive more so than in rural areas where they are typically abudant (Jiang & O'Neill, 2004). Thus urban areas tend to rely on electricity as it is readily available from the grid. Most high to middle-income areas have more access to advanced residential energy carriers because they can afford most of them, unlike low-income and rural areas. In low-income and rural settlements, traditional fuels are the norm and are very dependable for these communities. On the other hand, the general supply of electricity in these regions is not reliable (Kemmler, 2007). Another hindrance to sole reliance on electricity in these communities are the costs and maintenance of electrical appliances as well as prepaid electricity levies (Kemmler, 2007; Ahmad *et al.,* 2014). Thus traditional fuels are a preferable standard to meet the day-to-day residential energy needs.

The use of traditional fuels is inversely related to income because when monthly income decreases, there is an increase in the use of these fuels (Nkosi, 2018). This applies for all income groups, with the difference being total expenditure on residential energy. According to Davis

(1998), high-income groups spend proportionately less on household energy (5.5%), unlike the lower income groups which spend substantial amounts of their total income on residential energy needs. In areas such as the Himalayas, household income has been found to be a major factor that governs fuel choice (Mustaq *et al*, 2014) whereas in areas like Nigeria it does not impact on the choice of fuel (Ogwuche & Asobo, 2013). This can be as a result of the differences in terms of the culture of the people per region. It is also most likely due to the appropriation of a certain household economic status relative to specific domestic energy sources (Xiaohua & Zhenmin, 2003). For instance, in Malawi, with increasing household income, the majority of residents found it preferable to switch to fuel types that were cleaner, more economical and efficient (Nkosi, 2018). This shows that income can be a determinant of household affordability of certain energy carriers for residential purposes and preference.

- **Household Demographics**

Household demographic qualities also play an important role in determining the type of fuel resident prefer to choose and use. These include numbers in the household, family size, age, building material used for construction of the home, level of education, gender and ethnicity. The latter demographic elements are considered strongly influential in domestic fuel choice. With regard to household size in developing countries, the smaller the household, the higher the per capita fuel consumption, but larger families tend to use advanced energy devices like heaters for space heating (Manyo-Plange, 2011). However, often larger families are disadvantaged and potentially unable to afford more sophisticated energy carriers because they are likely to have "more mouths to feed" than be able to sustain their energy needs.

Building material used for construction of homes has an impact on the type and amount of fuel used for domestic purposes since this influences indoor temperatures. For instance, rural houses built from corrugated iron and wood are prone to experiencing harsh temperatures (very hot and very cold) depending on the season. Thus, during cold winter months, there is a spike in fuel combustion within these houses built of materials with poor thermal properties. These households are therefore more likely to use traditional dirty fuels. Houses built using concrete bricks mostly have good insulation properties and are thus expected to make more use of modern fuels (electricity) than solid traditional fuels.

Education is also key to residential fuel choice; with the likelihood of a household led by an uneducated person to use modern fuels being lower than if educated (Panchauri & Jainm, 2008). This is due to the positive relationship found between education and income whereby the more educated the person is, the higher the likelihood of employment (source of income) and opportunity to choose cleaner energy alternatives. Level of education increases knowledge on the effects of household fuels; thus, in most cases households led by educated persons use

cleaner fuels, unlike uneducated people which mostly use solid traditional fuels (Filiooini & Panchauri, 2004). A study was done by Boadi & Kuitenen (2006) in Ghana found that residents with primary to no education continued using charcoal (traditional fuel) unlike those who had attained secondary and higher education qualifications. Contrary to what was found by Boadi & Kuitenen (2006), education has little to no influence on residential fuel choice in other regions; here it was rather governed by certain elements such as gender of household head and age (Wu *et al.,* 2012). Female household heads are more likely to use solely traditional fuels or a mixture to meet their energy needs (Mekonnen, 2009). They also are linked with using fuels that consume time (Haltberg, 2005) and the use of a greater proportion of solid fuels (Mustaq, 2014). Households led by an older person also typically consume a greater proportion of solid traditional fuels, unlike the younger generation which is more open to cleaner and more convenient fuel alternatives (Mekonnen & Kohlin 2008; Mekonnen, 2009). This can be ascribed to older people being used to customs and beliefs that they grew up with.

- **Culture and traditional beliefs**

Cooking with fire in some areas has a deep cultural and spiritual meaning (Smith, 2006) with the traditional methods for cooking also affecting the taste of food. For instance, areas, where coal and wood are used for cooking it, is believed that they preserve the culture and traditional taste of their cuisine (Nansaior *et al.,* 2011). Therefore, the fuel choice is also a function of traditional and cultural perceptions of the community. Even well-off households that use advanced energy carriers continue to use traditional cooking fuels because of their cultural significance (Manyo-Plange 2011).

2.6. Characteristics of particulate matter emissions from burning of residential fuels

The combustion of traditional solid fuels for industrial, commercial and domestic purposes is a significant source of gaseous (SO_2, NO_2, NO_x) and particulate emissions (Williams *et al.,* 2012; Olave *et al.,* 2017). Household burning of solid fuels is a common practice in low-income South African communities and accounts high particulate loading in the atmosphere on a local scale (Makonese *et al.,* 2016; Nkosi *et al*, 2017). Emissions of particulate matter are regarded as by far the most harmful of all household burning products, especially fine particles which can be inhaled easily deeply in the respiratory system (Graham & Dutkiewicz, 1999; Olave *et al.,* 2017). The cook-stove test study conducted by Graham and Dutkiewicz (1999) found high emissions of total suspended (TSP) and fine particle emissions from all domestic fuel-burning appliances (Figure 7). A study done in Qalabotjha in Freestate Province found that the largest contributor to ambient PM concentrations was household coal burning to account for 62.1% of $PM_{2.5}$ and 42.6% of PM_{10} compared to other sources (Engelbrecht *et al.,* 2002). Similarly, domestic burning emissions in Soweto contributes up to 75% of ambient fine particulate matter (Nuwarinda *et al.,* 2007).

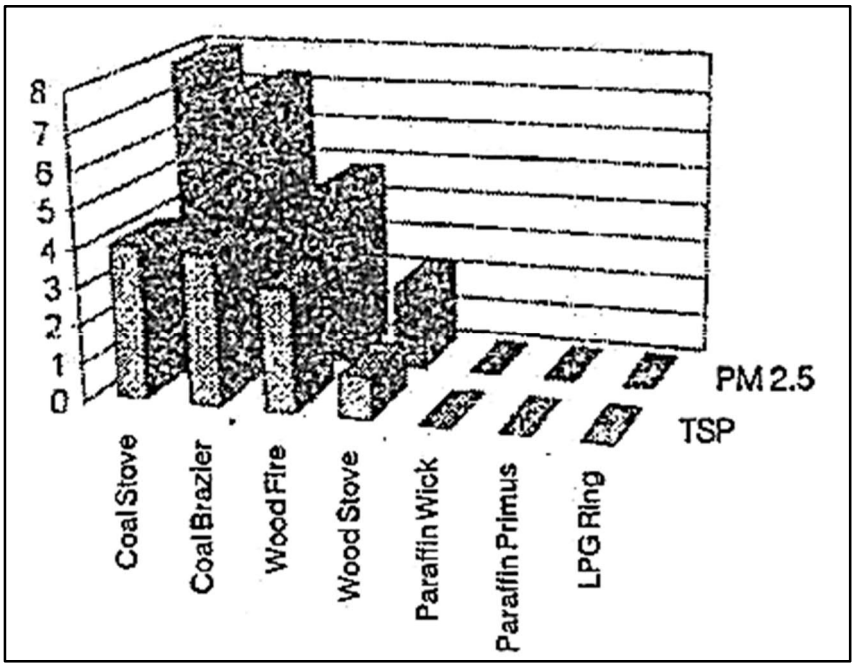

Figure 7: Various emission intensities of particulates from a number of domestic cooking appliances (Graham & Dutkiewicz, 1999).

Emissions from domestic fuel combustion are high due to insufficient emissions control and poor combustion design (Streets, 2002). This can also be coupled with inadequate stove operation techniques and the quality and characteristics of fuel used (Bi *et al.,* 2008). Studies have found that D-grade coal, commonly used for domestic burning, affects the $PM_{2.5}$ emission factors to range between 0.2 and 3.3 g/MJ due to the different ventilation and ignition methods (Wernecke, 2018). As a result, certain low-income communities experience over 100 µg/m3 ambient particulate concentrations during burning periods (Nkosi, 2017). However, particulates have different sources and the chemical constituents provide source information such as biomass burning (veld fires), industrial, power generation or domestic burning (Lina *et al.,* 2014). Due to the poor quality of coal used in low-income areas, a large amount of polycyclic aromatic hydrocarbons (PAH) are found highly concentrated in the air (Zhang *et al.,* 2011). Incomplete combustion is a typical process that occurs throughout the domestic burning of traditional fuels. This process releases high amounts of carbonaceous particles which will result in the formation of soot (Garcia-Maraver, 2014). The burning temperature and duration of the burning period have an effect on the chemistry of particulates (Zhao *et al.,* 2007). Aerosol particles released from SFB include PM_{10} (coarse) and $PM_{2.5}$ (fine). Domestic fuel combustion is a major source of fine particles (< 2.5 µm) composed of sulfate, nitrate, ammonium, lead, organic and elemental carbon (Figure 8) (Aneja *et al.,* 2006).

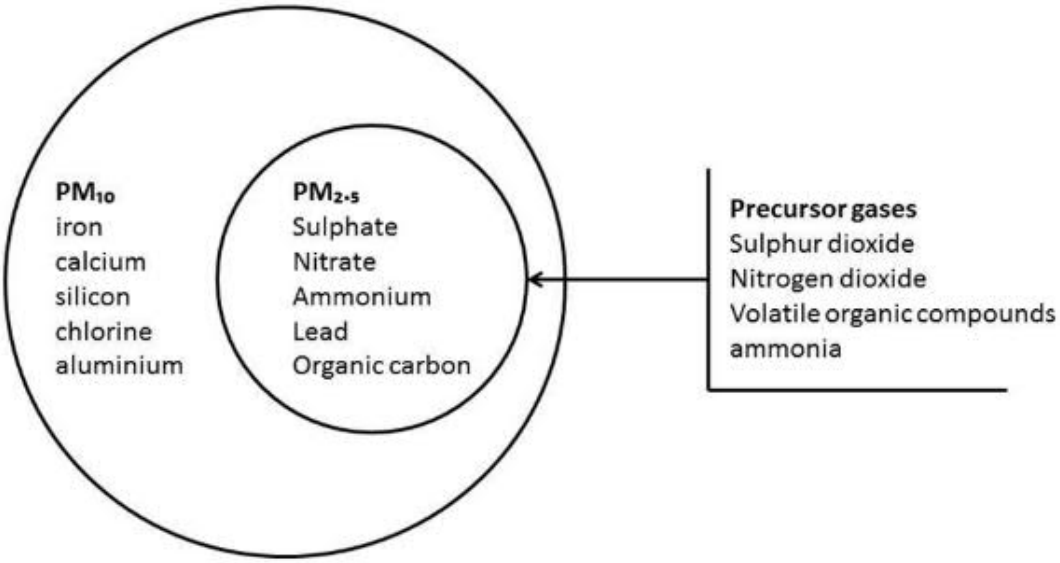

Figure 8: Typical chemical composition found in aerosol particles (Van den Berg, 2015).

2.7. Health Impacts Associated with Poor Indoor Air Quality

The correlation between air pollution and increased mortality and morbidity due to negative health implications has been well documented. It is described as a "multifaceted mix" combining airborne aerosol particles and gases (Brunekreef & Holgate, 2002; Wernecke, 2018). Approximately 2 billion people are exposed to a high concentration of PM and gases 10–20 times higher than ambient levels due to residential cooking and heating using open fires (Balakrishnan *et al.,* 2004). The importance of PM in residential environments is the health implication due to long-term exposure, significantly associated with premature causes of death and hence the reduction of life expectancy (Lwai *et al.,* 2005; Krall *et al.,* 2013). Short-term exposure to indoor suspended PM also impacts negatively on health causing pulmonary and acute cardiovascular diseases (Brook *et al.,* 2010). An average of 4–8% of premature deaths can be ascribed to being a function of exposure to TSP, mostly fine size fraction ($PM_{2.5}$) in both ambient and indoor settings (Wernecke, 2018).

Size and chemistry of airborne particles within households are important factors as they are both harmful when they come in contact with respiratory organs. The size fraction of PM determines the amount of damage it may cause. The size is directly associated with the possible health consequences (Kim *et al.,* 2015). These small particles include respirable coarse particles with an aerodynamic diameter between 2.5 and 10 µm and fine particles less than 2.5 µm (Kim *et al.,* 2015). In general, the smaller the particle, the deeper and more rapidly it is deposited within the respiratory system. It is unlikely that PM >10 µm will pass the trachea, and it is thus eliminated through coughing or sneezing and causing less harm to the body (Cadelis *et al.,* 2014; Kim *et al.,* 2015). Due to their size fraction, particles 5 to 10 µm can enter the tracheobronchial tree (upper

respiratory system), whereas particles 1 to 5 µm potentially deposit in the alveoli and bronchioles where the exchange of gas takes place (Figure 9) (Kim *et al.,* 2015).

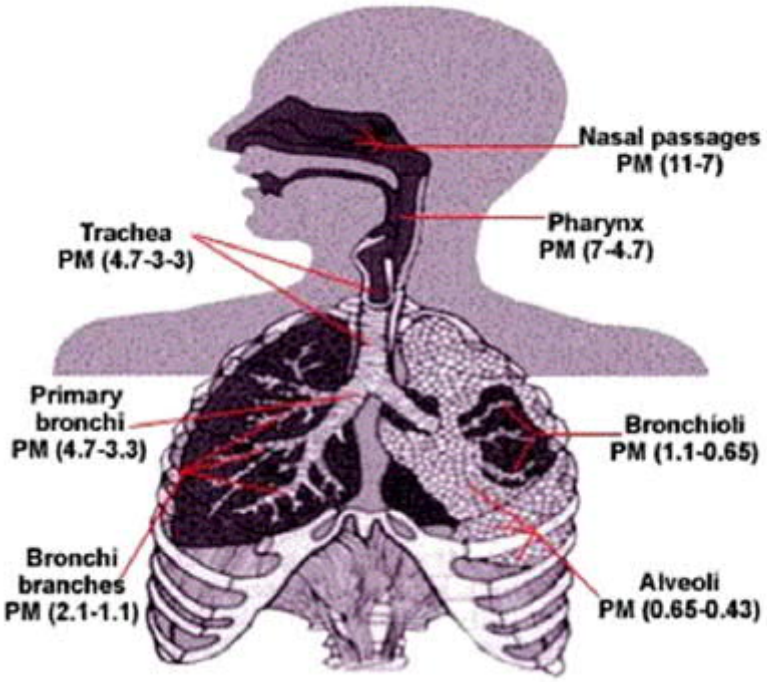

Figure 9: The potential of particles of various sizes to deposit in the various locations in the respiratory system (Source: Kim *et al.,* 2015).

Other than size, the toxicity levels of the chemical elements within PM are also an important contributing factor to the cause of health problems. One of the most impactful elements found in PM is transition metals, especially iron (Fe) (Kim *et al.,* 2015). It has been found that following exposure to PM, Fe contents increase the production rate of reactive oxygen species (ROS) within the respiratory tract (Kadiiska *et al.,* 1997; Knaapen *et al.,* 2002; Kim *et al.,* 2015). Thus transition metals within PM play a potential mediator role for causing airway injuries and inflammation when they undergo the Fenton reaction (Knaapen *et al.,* 2002; Gonzales-Flecha, 2004). During the Fenton reaction, transition metals reduce hydrogen peroxide and thus generate hydroxyl free radicals and this is related to inducing oxidative stress and inflammation (Sørensen *et al.,* 2005). Further, PM formed as a function of anthropogenic processes (domestic burning) have greater effects on lung and systemic cardiovascular disease due to possible induced production of pro-inflammatory cytokines (Araujo, 2011).

2.8. Indoor airborne PM characterisation: comparison of techniques

Particulate loading differs in space and time because different regions have different PM sources contributing to the regional atmospheric aerosol loading. In order to understand the impacts of domestic fuel combustion, both qualitative and quantitative data is important. It is also important to find methods and techniques suitable for the scope of the study. In this regard, receptor models

are suitable for indoor air pollutants' source attribution. Prior to the use of receptor models, determination on of elemental composition within PM is required. Therefore, PM samples undergo chemical analysis techniques to acquire elemental data and thus follow the receptor models to attribute the PM to internal and external sources.

2.8.1. Elemental composition analysis

There are different techniques that may be used for the determination of chemical elements in PM samples. These include the widely used spectroscopic methods which evaluate the "interaction of electromagnetic radiation with samples" (Radojevic & Bashkin, 1999), such as gravimetric techniques, calorimetry and x-rays. For the purpose of this study, X-ray fluorescence (XRF) was used to determine the elemental composition of the indoor PM samples.

About a century ago, Henry Moseley introduced the XRF method which contributed to the elements presented in the periodic table (Shackley, 2011). It is a widely used method for a variety of commercial and environmental applications and in the field of species detection. It is a non-destructive method of analysis which means samples can go through XRF process and be used for other analytical purposes. Minimal effort for sample preparation is required for this method, whereas other techniques are time-consuming (especially when samples need to be diluted). It is also cost effective and easy to learn and use; furthermore, with its high precision, the results produced are highly accurate (AMC, 2017). Thus, XRF was selected for this study because the scope is low complexity; this method is thus more suitable than other techniques.

2.8.2. Receptor models

A number of sources are able to contribute significantly to indoor PM and this can be estimated using a variety of source-apportionment techniques. Complex dispersion models can be used to determine the source contribution of pollutants to a receptor site such as low-income settlements often located downwind. However, receptor models that work with multivariate measurements from one or two sites are more easy to use. These models can be fixed to work for indoor, outdoor and mobile monitors (Watson *et al.,* 2002). The models are not just based on statistics but rather use the same scientific principles used in source contribution methods, they are on the contrary explanatory of source contribution rather than being predictive (Watson *et al.,* 2002). There are various receptor models used for source apportionment application; these include chemical mass balance (CMB), principal component analysis (PCA), enrichment factors (EF) and positive matrix factorisation (PMF) (Viana *et al.,* 2008; Callén *et al.,* 2009). The CMB and PMF methods will be briefly described and the emphasis is placed on PCA and EF which were applied for this study.

The CMB models consider the chemical composition of the emissions for all relevant sources (Viana *et al.,* 2008); this entails that the chemical and physical characteristics are combined to

quantify and determine the presence of particle sources in a receptor site (Van den Berg, 2015). The model "*consists of a least squares solutions to a set of mass-balance equations that express each receptor chemical concentration as a linear sum of products of source profile abundances and source contribution*" (Engelbrecht *et al.,* 2002). This occurs through quantifying the contribution using the chemistry of source emissions instead of measuring every polluter (Begum *et al.,* 2007).

The PMF model has been used successfully for the determination of particle source contribution in the areas such as Hong Kong (Lee *et al.,* 1999), Atlanta (Kim *et al.,* 2003) and the Indian Ocean (Bhanuprasad *et al.,* 2008) and many other countries worldwide. The model is a multivariate factor analysis tool that uses a matrix of sample data separated into two matrices – source profile and source contribution (US: EPA, 2008). The model uses a "point-by-point least-squares minimisation" system and not correlation matrix information; this allows for the profiles to be compared directly to the input matrix without transformation (Lee *et al.,* 1999). The Positive Matrix Factorisation model is advantageous for representing data below the detection limit and the non-negativity built in the model also offers the minimisation of error in estimating source contribution; PMF results are also comparable to CMB output results (Paatero & Tapper, 1994; Begum *et al.,* 2007; Lee *et al.,* 2008).

Principal component analysis is a statistical tool for analysing structure in multivariate datasets using the conversion of mass law to calculate the constituents of particles (Callén *et al.,* 2009). In a sense, PCA is a method for data reduction through determining linear combinations of original variables and in so doing, accounting for the total variance (Shen *et al.,* 2006). This is a correlation matrix that determines the Eigenvectors and where a subset is rotated to increase or decrease the correlation of each factor with each species in the measurements (Watson *et al.,* 2002). In so doing, PCA transforms complex datasets to less significant dimensions to show similarities in the dataset. The method is good for determining the origin of chemical species through comparing factor loadings with source measurements and thus establishes a source profile (Watson *et al.,* 2002; Shen *et al.,* 2006). For this reason, PCA was chosen as the most suitable receptor model to understand the origin of airborne respirable particles found within low-income settlements in the present study.

Another model used in this research to determine source contribution to indoor airborne PM is the EF. The EF model determines the atmospheric concentration of elements by comparing them to the ratio of geological or marine material (Watson *et al.,* 2002). In this case, aluminium (Al) is used as a reference geological element to determine the anthropogenic contribution to indoor airborne PM. The differences in enrichment entail anthropogenic contribution. The significance of enriched elements can ascertain and attribute the highly enriched element to a source of origin. For instance, sulphur (S) shows signs of secondary sulphate contribution whereas heavy metals

are attributed to industrial emissions (Reinmann & de Caritat, 2005). There are various reasons why EF can be high or low, such as contamination which can be from indoor or outdoor sources of PM.

The PCA and EF models were used for this research to understand anthropogenic source contribution to indoor airborne respirable PM (PM_4). Using the two models combined has the ability to estimate the origin and impact of outdoor sources to the indoor environment, as well as identify major polluters in Sharpeville that contribute to indoor PM.

CHAPTER 3: METHODOLOGY

CHAPTER OVERVIEW: The purpose of this chapter is to explain the data utilized in the analysis as well as the methods of collecting and obtaining the elemental composition of collected indoor samples. The analytical procedures are comprehensively outlined, as well as any calculations and formulae necessary to obtain usable verified data.

3.1. Experimental methods

The extent of solid fuel use within low-income households in South Africa for daily household activities was explained using two approaches: (1) questionnaire-based interviews with the low-income community members; and, (2) physical measurements were taken to investigate the state of the air people are exposed to within their homes. The scope of the study was on a local scale where a single low-income settlement was selected. These two approaches were then combined to represent the importance of domestic solid fuel combustion in South African low-income communities.

3.2. Study site description

Sharpeville is a low-income settlement located in the VTAPA that was selected for completing the objectives of this research study. The Vaal Triangle is well documented to have poor air quality due to numerous sources of air pollution (DEAT, 2008; Scorgie *et al.,* 2003). These sources include mining, industry, power generation, domestic burning, transport and agriculture which contribute significantly to the particulate loading in the air. However, in this study, the focus is only on the domestic sources of PM.

Sharpeville is located between Vanderbijlpark and Vereeninging in the Emfuleni Local Municipality. These two towns are notorious for the concentration of industrial and mining activities. The site layout and sampling locations are presented in Figure 10. Sharpeville is one of the most densely populated low-income settlements in the VTAPA region with a population density of 7536 persons/km^2 and a total population of 37,599 (Census 2011; StatsSA, 2018).

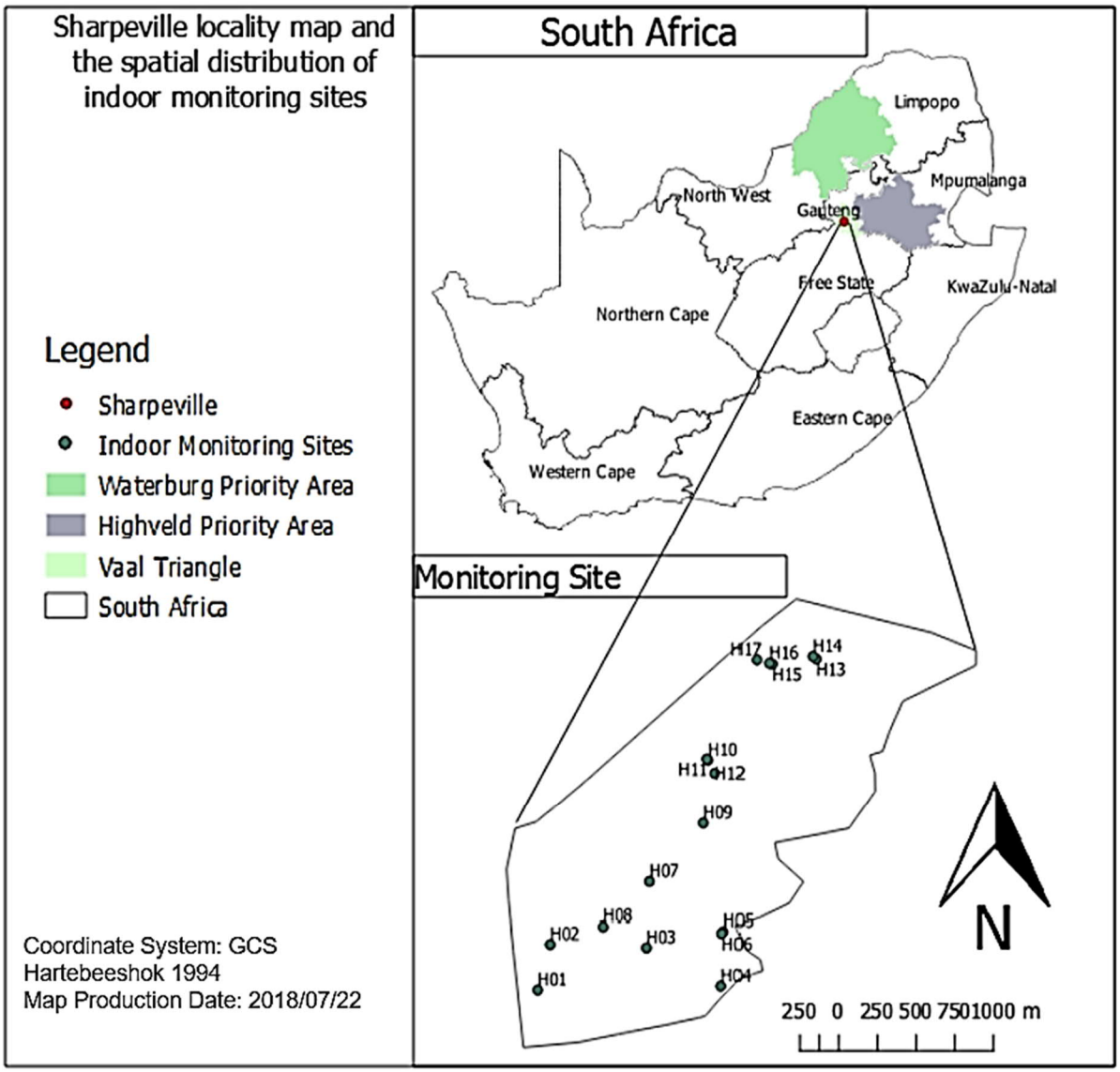

Figure 10: Locality map of the Sharpeville site with all sampling points.

Sharpeville has a flat and rolling topography between 13.000 to 19.000 masl (Figure 11). The gentle topography allows for a homogeneous local climate and occasional valley winds. Sharpeville resides in a grassland dominant region mainly suitable for commercial farming. The Emfuleni Local Municipality is within the Sedibeng District Municipality which is a highly industrialised area characterised by a variety of mining (Seriti – New Vaal Coal Colliery), industrial (ArseloMital), power generation (Eskom Lethabo Power Station) and petrochemical production (Sasol plant).

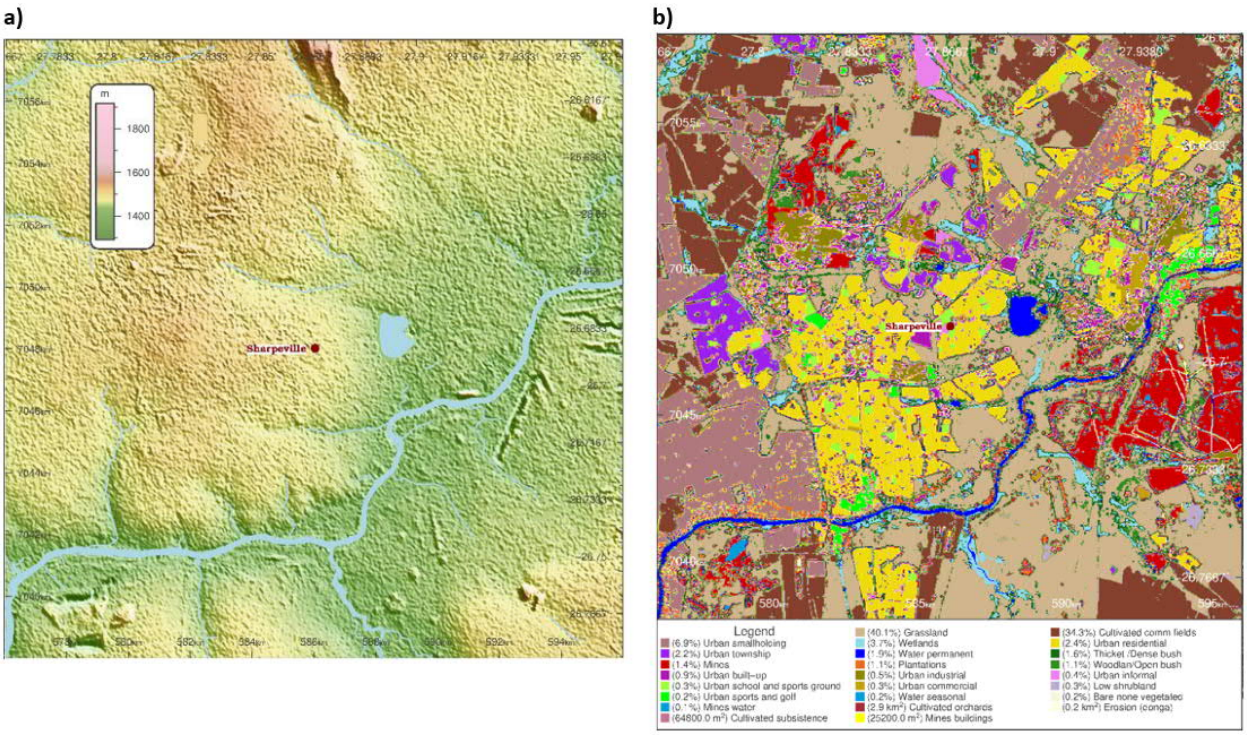

Figure 11: The topography of Sharpeville (a) and land cover (b) are shown.

3.3. Indoor measurements

3.3.1. Study design

Baseline indoor air pollution monitoring was conducted in Sharpeville over both winter and summer 2017. The experiment within low-income household's characteristic of the Sharpeville community included conducting a Detailed Energy Survey (DES) by the Nova Research Institute; deployment of continuous respirable PM monitoring equipment; collection of PM samples for chemical characterisation using Quartz and Mixed Cellulose Ester (MCE) filters; a continuous black carbon sampler; and, temperature iButtons. Following the DES, a total of 16 different households were randomly selected during the winter and summer sampling campaigns (Table 2). The household House-15 (H15) used during the winter campaign withdrew from the study and was replaced by House-17 (H17) for the summer campaign. Therefore, both campaigns only consisted of a subset 16 sample houses numbered H01 to H17. A brief summary of the key features of each of the houses in which sampling was conducted is provided in Table 3.

Table 2: Indoor particulate monitoring households, including the installation and removal dates for each household, according to sampling campaign.

Household No.	Winter 2017 (10 July–4 September)			Summer 2017 (23 October–4 December)		
	Installation Date	Removal Date	Days Active	Installation Date	Removal Date	Days Active
H01	2017/07/10	2017/07/24	15	2017/10/23	2017/11/06	15
H02	2017/07/10	2017/07/24	15	2017/10/23	2017/11/20	20
H03	2017/07/10	2017/07/24	15	2017/10/23	2017/11/06	15
H04	2017/07/10	2017/07/24	15	2017/10/23	2017/11/06	15
H05	2017/07/24	2017/08/10	18	2017/10/23	2017/11/20	29
H06	2017/07/24	2017/08/05	15	2017/11/06	2017/11/20	15
H07	2017/07/24	2017/08/10	18	2017/10/23	2017/11/20	29
H08	2017/07/24	2017/08/05	15	2017/11/06	2017/11/20	15
H09	2017/08/21	2017/09/04	15	2017/11/06	2017/11/20	15
H10	2017/08/07	2017/08/21	15	2017/11/20	2017/12/04	15
H11	2017/08/07	2017/08/21	15	2017/11/20	2017/12/04	15
H12	2017/08/07	2017/08/21	15	2017/11/20	2017/12/04	15
H13	2017/08/21	2017/09/04	15	2017/11/20	2017/12/04	15
H14	2017/08/07	2017/08/21	15	2017/11/20	2017/12/04	15
H15	2017/08/21	2017/09/04	15	-	-	-
H16	2017/08/21	2017/09/04	15	2017/11/20	2017/12/04	15
H17	-	-	-	2017/11/06	2017/11/20	15

Table 3: Important characteristics of each of the selected houses in which indoor measurements of respirable particulate matter and temperature where measured during winter and summer 2017.

	Characteristics of the Household					
	House type	**Main fuel use**			**Other sources of PM**	**Ventilation**
		Cooking	***Heating***	***Lighting***		
H01	Formal Extended RDP	Elec	Elec	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H02	Formal RDP	Elec & Wood	Elec & Wood	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H03	Formal Extended RDP	Elec	Elec	Elec	Next to an unpaved road	Windows and doors; open during the day; closed at night
H04	Formal Extended RDP	Elec & Wood	Elec & Wood	Elec	No ceiling; Waste burning; next to an informal dumping site	Windows and doors; open during the day; closed at night
H05	Formal RDP	Elec	Elec	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H06	Formal RDP	Elec	Paraffin	Elec	No ceiling	Windows and doors; open during the day; closed at night
H07	Formal RDP	Elec & Wood	Elec & Wood	Elec	Next to an unpaved road; No ceiling; waste burning	Windows and doors; open during the day; closed at night
H08	Formal RDP	Elec & Wood	Elec, Wood & Paraffin	Elec	Waste burning; smoking inside the house	Windows and doors; open during the day; closed at night
H09	Formal RDP	Elec & Wood	Elec & Wood	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H10	Formal RDP	Elec	Elec	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H11	Formal RDP	Elec	Elec	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H12	Formal Extended RDP	Elec	Elec	Elec	Next to an unpaved road	Windows and doors; open during the day; closed at night
H13	Formal Extended RDP	Elec	Elec	Elec	Next to an unpaved road; No ceiling	Windows and doors; open during the day; closed at night
H14	Formal Extended RDP	Elec & Wood	Wood	Elec	Next to an unpaved road	Windows and doors; open during the day; closed at night
H15	Formal Extended RDP	Elec	Elec	Elec	Next to an unpaved road	Windows and doors; open during the day; closed at night
H16	Formal RDP	Elec	Elec	Elec	Next to an unpaved road; smoking inside the house; construction of the road during sampling	Windows and doors; open during the day; closed at night
H17	Formal Extended RDP	Elec & Wood	Elec & Wood	Elec	Next to an unpaved road; No ceiling; smoking inside the house; Waste burning; use of braziers outside for cooking	Windows and doors; open during the day; closed at night

Thermochron iButton temperature sensors were installed in each of the 16 households in Sharpeville during the winter (10 July–4 September) and summer (23 October–4 December) sampling periods. The iButton sensor location for each household is represented in Table 4. The sensors were located in the same position during both campaigns unless otherwise specified. The sensors were set to a logging interval of 10 minutes. The purpose of the temperature sensors was to establish the profile of thermal variability in each household to understand thermal extremes experienced by inhabitants compared to ambient conditions.

Table 4: Temperature monitoring (location and direction) conducted at individual households using Thermochron iButtons during the winter (10 July–4 September) and summer (23 October–4 December) campaigns in 2017.

	Winter 2017					**Summer 2017**				
House No.	**External**	**Living room**	**Kitchen**	**Main Bedroom**	**Secondary Bedroom**	**External**	**Living room**	**Kitchen**	**Main Bedroom**	**Secondary Bedroom**
H01	S		S	N		S	W	S	N	
H02			S	N			E	S	N	
H03	S		E	S		S	W	E	N	
H04	S		W	N		S	S	E	N	
H05		W	W	S			W	W	S	
H06		S	E	N			S	E	N	
H07		W	S	N			W	S	N	
H08	S		S	W			W	S	W	
H09		S	E	W	N		S	E	W	N
H10		N	W	E	S		N	W	E	S
H11		N	W	E	S		N	W	E	S
H12		E	N	E	W		E	N	E	W
H13		E	W	E	S		E	W	E	S
H14		S	E	N	E		S	E	N	E
H15		E	N	S	N		E	N	S	N
H16		N	W	N			N	W	N	
H17						S	E	N	E	
Key	**N-North**		**E-East**			**S-South**			**W-West**	

3.3.2. Instrumentation used for indoor physical measurements

3.3.2.1. Continuous PM monitoring equipment

Continuous monitoring of respirable particles within houses requires a small, portable and easy to set up equipment. The TSI DustTrak 8530 and SidePak AM510 (TSI, Inc) continuous were used for the purpose of this study as they are small enough not to inconvenience participants in

their houses (Figure 12). Size, noise and energy consumption were critical components in the selection of the instrument in consideration of the sensitivity of the study. These instruments are both light-scattering nephelometers that use a long-wavelength laser (λ = 680 - 780 nm) and undergo factory calibration using the Arizona Road Dust Test standard (ISO 12103-1). The laser diode on the monitors enables measurement of particulates ranging between 0.0001 – 100 mg/m3 (Wallace *et al.,* 2011; TSI Inc., 2002).

The instruments were fitted with a 10-mm Nylon Dorr-Oliver Cyclone which allowed the separation of particles at various size fractions (<10 µg/m^3). The monitors were set to sample through a PM_{10} inlet at 1.7L/min^{-1} to archive a 50% cut-off at 4 µm (Watson *et al.,* 2011; TSI, 2010). Data was logged at a 5 minute intervals to acquire better temporal resolution during analysis. A total of 16 different houses had DustTrak samplers installed during the winter and summer sampling campaigns. The alternative H17 had the TSI AM510 Sidepak photometric monitor used to continuously measure indoor PM during summer 2017 (Figure 12b). The photometric instruments are prone to under- and overestimation of particulate concentrations (Wallace *et al.,* 2011). This is due the surface albedo of the particles passing through the laser beam have an impact on the absorption and scattering of light (Kim *et al.,* 2004; Watson *et al.,* 2011). This means darker surface particles absorbs the light and result in underestimation whereas bright-surface particles reflect light at a higher intensity and result in overestimation of particulate concentrations. Therefore, a reference method is important to calculate the correction factor for these uncertainties of measurements (Watson *et al.,* 2011).

Figure 12: The location of the DustTrak (a) and SidePak (b) particulate matter sampling equipment and inlets within households at Sharpeville during winter and summer 2017.

3.3.2.2. Gravimetric sampling

i. Preparation prior to field sampling

Preparations prior to field deployment are important. Filters need to be equilibrated in the weighing lab ±2 hours before they can be weighed (Figure 13a). The calibration of the microbalance scale is checked daily by using calibration weights (1g and 2g) before weighing any filters. This process is recorded to provide traceability and accountability. Incorrect mass reading is limited because prior to each weighing session the microbalance undergoes a zero calibration ('zeroed'). Weighing of each filter is repeated three times. The XP26 DeltaRange Microbalance (Mettler-Toledo AG, Greifensee, CH) was used during this study; it was thus not necessary to use an external anti-static radiation source. The antistatic kits integrated with the microbalance releases static samples during the weighing the process reducing any risk of contamination of samples (Mettler-Toledo GmbH, 2019). To prevent infiltration of polluted air into the 37mm cassettes, shrink seals were used during the preparation process. The cassettes were further loaded and sealed in airtight containers to avoid contamination (Figure 13b). Gravimetric sampling needs use of the Dorr-Oliver cyclone accompanied by checking and cleaning procedures as mentioned previously.

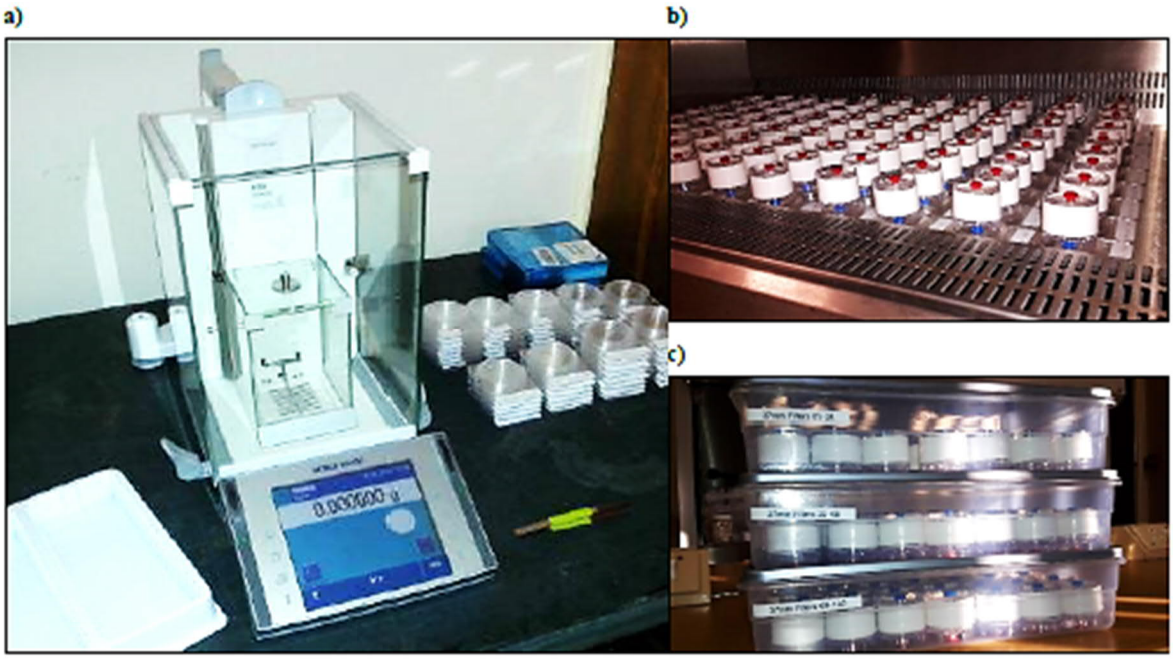

Figure 13: Representation of the XP26 DeltaRange Microbalance scale used: (a) for filter weighing prior to loading; (b) 37mm cassettes (c) airtight containiners for used during collection and transport of filter samples (Images taken by B. Language, 2015).

ii. Field sampling

Gilian GilAir sampling pumps (Senidyne, Clearwater, FL, USA) (Figure 14) were used for the gravimetric sampling. These instruments were with a cyclone inlet offering the same PM cut-size (50% at 4 microns) and connected to a 37mm cassette containing MCE filters. The sampling was set to run at the same volume of air at 1.7 $L.m^{-1}$ (±10%). The sampling heads were assembled to make sure that the alignment of the filter cassettes and cyclones were unobstructed to allow free airflow as well as prevent air leaks.

From a subset of 17 households, 2 sample houses were selected for the gravimetric sampling during winter (26 July–09 August 2017) and summer (1–19 November 2017). One house used a mixture of sild fuels and electricity in its energy mix (SFB), whereas the other was only using electricity as a primary household energy source (NSFB).

Figure 14: The set up used for both gravimetric and photometric sampling during winter (26 July–09 August 2017) and summer (1–19 November 2017).

The cyclone inlets were placed at representative breathing height of 1.6m above the ground. Filter samples were collected at intervals of ~11- hours, leaving one hour for filter changes and to ensure the sampling connection is in good working condition before continuing with the campaign. To prevent filters from overloading and also not misrepresent peak hours, the sampling interval was for filter changes was between 10h00 to 22h00. This means the morning and evening peak concentrations ae captured. Inverting the cyclone was carefully avoided as this could result in a large deposit of non respirable particles on the filers. The accumulation of larger particles on the

grit pot onto the filters would mean incorrect sampling and therefore no results. Following every sampling event, 37mm cassettes were removed from the inlet and placed in the airtight containers for storage before sent to the lab.

iii. Post-sampling

At the end of each sampling campaign, the filters were offloaded and taken to the weighing lab and left overnight to equilibrate before they can weighed on the microbalance to acquire post-mass (Figure 15). Damaged filters were excluded from the sample batch. Following the acquisition of the post-mass, the samples were stored inside petri-dishes for further chemical analysis.

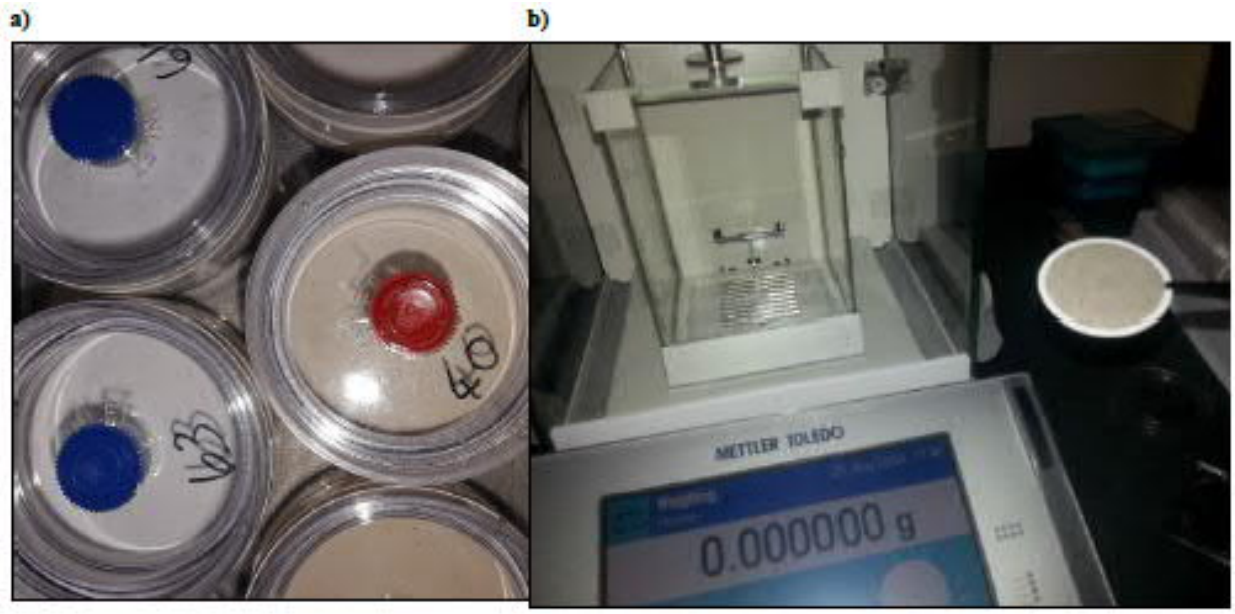

Figure 15. Representation of non-exposed filter (blue caps) and exposed (red caps) filters (left); and post-exposure filter weighting (right). (Images taken by B. Language, 2015).

A Series 4000 flow meters need a reverse flow correction while working with flows above 10 $L.min^{-1}$; however, the indoor sampling is needed for 1.7 $L.min^{-1}$ flow making the reverse flow correction redundant. The main reason for the redundancy is the uncertainty surrounding the correction that would be too large.

iv. Calculation of gravimetric mass

Gravimetric mass is the total collected particulate mass post-exposure of the filters below was used to calculate the actual particulate mass on the filters following the gravimetric sampling of indoor airborne PM_4:

$$PM_{4(g)} = (M_f - M_i) - (B_f - B_i)10^6$$

Equation 1. Actual PM mass collected in grams (g).

where $PM_{4(g)}$ is the actual mass collected on the filter in g, M_f is the post-exposure filter weight in grams, M_i is the pre-exposure filter weight in grams, B_f is the mean initial blank filter weight in grams, B_i is the mean final blank filter weight in grams, and 10^6 is the unit conversion from grams to micrograms. The total volume of air sampled was calculated with:

$$V_T = \frac{(F_{ave})(t)}{10^{-3}}$$

Equation 2. Total volume sampled in cubic meter (m^{-3}).

Where V_T is the total volume of air sampled in m^{-3}, F_{ave} is the average flow rate in $L.min^{-1}$, and t represents the elapsed time in minutes. The gravimetric PM_4 concentration was calculated from the laboratory data and the sampler volume using:

$$PM_{4(\mu g.m^{-3})} = \frac{[(W_f - W_i) - blank\ filter\ mass]x10^6}{V}$$

Equation 3. Gravimetric mass concentration in microgram per cubic meter ($\mu g.m^{-3}$) (Shezi *et al.,* 2017).

Where $PM_{4(\mu g.m\text{-}3)}$ is the gravimetric mass in $\mu g.m^{-3}$ W_f is the post-exposure filter weight in g, W_i is the pre-exposure filter weight in g (Shezi *et al.,* 2017). The blank filter mass is the mean pre- and post-exposure blank filter weight in g, and V is the total volume of air sampled in m^3. The results are then converted from g to µg by multiplying with 10^6 {National Institute for Occupational Safety and Health (NIOSH), 1998}.

3.2.2.3. Indoor temperature profile

Thermochron iBotton (DS1921G) temperature sensors were used for the purpose of measuring indoor and outdoor temperatures in Sharpeville. These small, mobile and durable independent sensors are able to measure and record temperature data in a secure internal memory unit. They are user-friendly and allow for parameters to be set according to the users' preferences. The device can store up to 2048 temperature values for 1–255 minutes recorded at equal intervals. This makes the device suitable for this study because at 10-minute intervals 2 weeks' worth of temperature data can be acquired before downloading. The 10-minute intervals are suitable for analysis of temperature trends and therefore provides clear temperature profiles of the households. These devices also have a resolution of 2.0°C and an additional 5012 bytes battery-backed SRAM which provides enough storage for the required dataset. The iButtons were factory-lasered with a guaranteed distinctive, electrically reliable, 64-bit registration that allows for

absolute reliability. The extraction of the data from the device occurs serially via the 1-Wire protocol which needs a single data lead and a ground return. These buttons were installed at a standardised height of 1.6 m on the wall (Figure 16). Ambient temperature measurements were also taken using the iButton temperature sensor in a plastic container filled with weights to avoid loss via wind and faulty temperature readings.

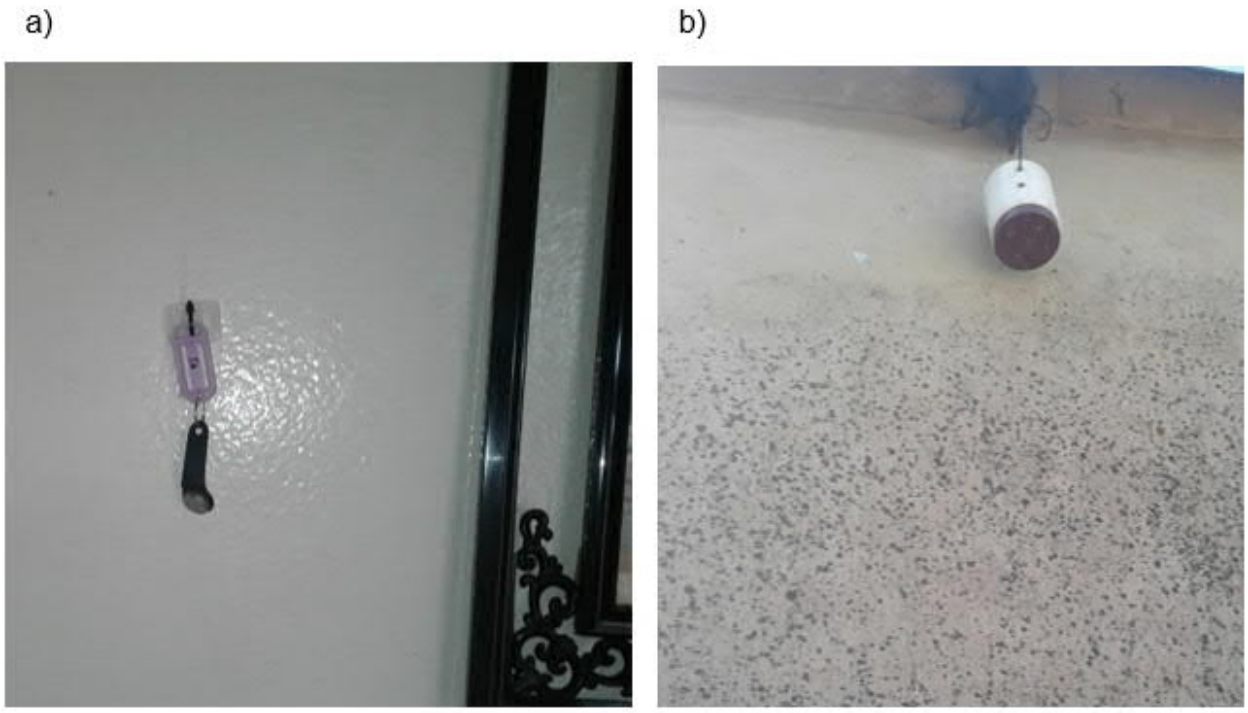

Figure 16: Indoor (a) and outdoor (b) temperature measurements using Thermochron iButton temperature sensors during winter (10 July–4 September) and summer (23 October–4 December) campaigns in 2017.

3.3.2.4. Indoor BC concentration

Indoor BC measurements were taken as an additional measurement in the selected SFB household during the summer campaign of 2017. These measurements were continuous and taken simultaneously with both photometric and gravimetric measurements. The data period of BC measurements took place for 19 days (1–19 November 2017). The micorAeth AE51 (AethLabs, San Francisco, California) was used to continuously sample BC concentrations. This is a portable aethalometer with built-in optical absorption technology that allows for rea- time measurements of BC (Figure 17). The aethalometer measures the rate of change in absorption of light as aerosol particles accumulate on the T60 Teflon-coated borosilicate glass fiber filter. The aethalometer has light-emitting diodes with unique wavelengths. The AE51 uses the 880nm wavelength to measure BC concentrations. The sampler has a measurement resolution and precision of 0.001 µg/m^3 and ±0.1 µg/m^3 (1 minute average at 150 ml/min flow rate), respectively.

These specifications allow for quality data acquisition within indoor environments. The instrument's inlet was set to a breathing height, as were the continuous samplers.

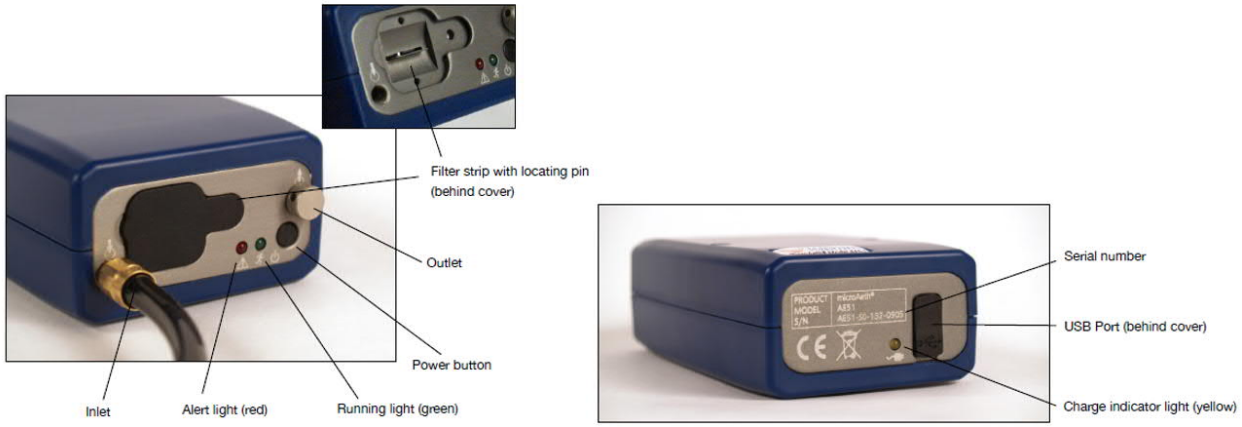

Figure 17: MicroAeth portable BC aethalometer (microAeth, 2016).

3.4. Elemental composition analysis

The XRF method was selected for analysis of elemental composition in this study. This technique does not require any treatment process of the filter samples before analysis and does not impose permanent damage to the filters. The XRF analysis method is susceptible to aspects such as matrix-effect corrections {Engelbrecht, 2011; Analytical Methods Committee Technical Brief (AMCTB), 2011}; however, these effects are minimal for thin film samples such as aerosol filters (Vanhoof *et al.*, 2003).

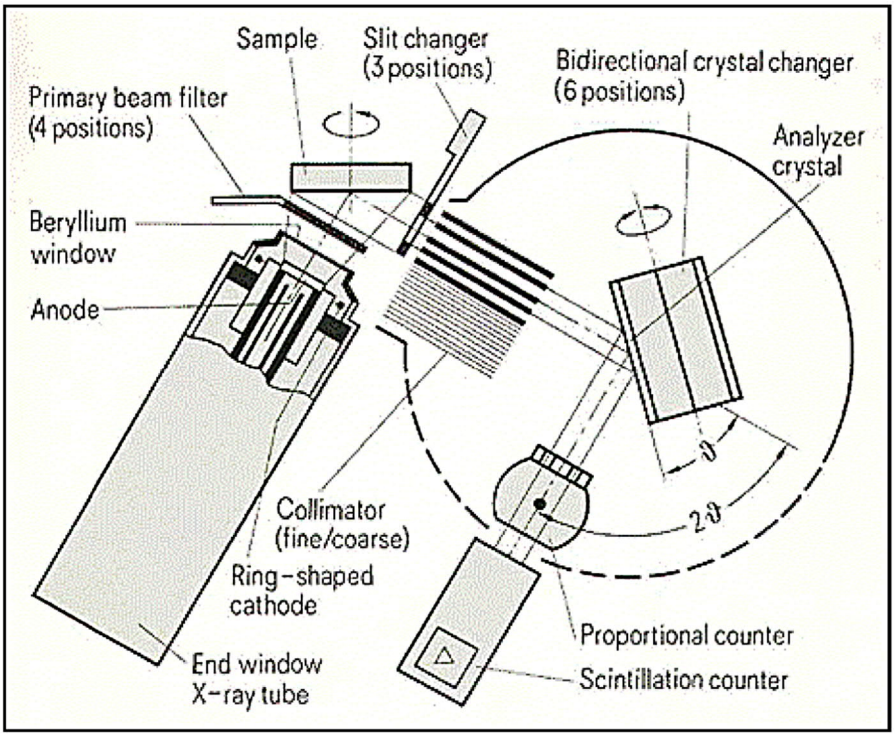

Figure 18: A schematic representation of the wavelength dispersive x-ray fluorescence (WD-XRF) spectrometer components (Engelbrecht, 2011).

The PANalytical Axios^{maX} wavelength dispersive x-ray fluorescence spectrometer (WD-XRF) was used for the elemental characterisation. The spectrometer uses a sequence to analyse the elements one at a time. The components illustrated in Figure 18 are important to be adjusted accordingly during the analysis to ensure relevant application. Seven trays were fitted in the sample changer with each of the seven trays holding the 37mm samples. Therefore only 56-positions could be used during analysis. An area of 20mm on the filter samples was analysed on every sample, this was due to the 20mm collimator mask selected on the XRF unit.

A selected number of elements (~47) were analysed as per the limitations of the XRF unit. These include Na; Li; Al; Mg; P; Si; Cl; S; Ca; K; V; Ti; Mn; Cr; Co; Fe; Cu; Ni; Zn; Cu; Ge; Ga; Se; As; Rb; Br; Y; Sr; Nb; Zr; Pd; Mo; Cd; Ag; Sn; In; I; Sb; Ba; Cs; W; Ce; Pt; Au; Hg; Tl; Bi and Pb (Willis et al., 2014; Willis *et al.,* 2014). The elements included in the analysis were a wide range of metals, metalloids and non-metallic, with an exclusion of actinoids, lanthanoids and noble gases as per XRF unit limitations. For calibration purposes, MICROPATTERN calibration standards were used. Table 5 (a and b) below presents a summary of elements analysed and calibration standards used per filter application.

Table 5(a). Elements included in the WD-XRF analysis and the associated MICROMATTER$^{(TM)}$ .XRF Standards applied during the calibration of the PANalytical AxiosmaX spectrometry instrument as used for the filter application.

Element	Metals					Metalloids	Non-metals	Description	(µg.cm^{-2}) +/- 5%	
	A	AE	PT	T	L				H	VL
Li	√							Lithium as LiF	53.6	4.1
Na	√							Sodium or Chlorine as NaCl	519	7.5
Mg		√						Magnesium as MgF2	51.8	6.6
Al			√					Aluminium as Al metal	53.1	7.1
Si						√		Silicon as SiO	58.5	5.3
P							√	*Gallium or Phosphorus as GaP ; Ga=31.5 & 3.1 P=11.8 & 3.8	43.3	6.9
S							√	Sulphur as CuSx Cu=31.9 S=10.5 and as InSx In=5.6 S=1.4	42.4	7.0
Cl							√	Chlorine as KCl	47.9	7.3
K	√							Potassium or Chlorine as KCl	46.4	7.2
Ca		√						Calcium as CaF2	45.4	6.8
Ti				√				Titanium as Ti metal	51.8	6.6
V				√				**Vanadium as V metal	38.3	7.8
Cr				√				Chromium as Cr metal	39.0	6.0
Mn				√				**Manganese as Mn metal	54.8	6.3
Fe				√				**Iron as Fe metal	46.1	6.7
Co				√				Cobalt as Co metal	46.5	4.8
Ni				√				Nickel as Ni metal	50.3	7.1
Cu				√				Copper as Cu metal	48.9	6.0
Zn				√				*Zinc as ZnTe	53.2	7.1

Element	Metals					Metalloids	Non-metals	Description	(µg.cm^{-2}) +/- 5%	
	A	AE	PT	T	L				H	VL
Ga			√					*Gallium or Phosphorous as GaP; Ga=41.0 & 4.2 P=9.4 & 4.4	50.4	78.6
Ge						√		Germanium as Ge metal	49.4	6.3
As						√		*Arsenic or Gallium as GaAs; Ga=19.2 & 3.0 As=30.8 & 4.0	50.0	7.0
Se							√	Selenium as Se metal	46.8	4.5
Br							√	Bromine or Caesium as CsBr	46.6	6.5

Table 5(a). Elements included in the WD-XRF analysis and the associated MICROMATTER(TM) .XRF Standards applied during the calibration of the PANalytical AxiosmaX spectrometry instrument as used for the filter application.

Element	Metals					Metalloids	Non-metals	Description	(µg.cm^{-2}) +/- 5%	
	A	AE	PT	T	L				H	VL
Ba		√						Barium as BaF2	48.9	6.5
Ce					√			Cerium as CeF3	51.0	7.23
W				√				Tungsten as WO_3	47.9	6.4
Pt				√				Platinum as Pt Metal	51.3	6.0
Au				√				Gold as Au metal	48.6	4.7
Hg				√				Mercury as AgHg; Ag=28.5 Hg=27.5 and *HgTe; Hg=3.7 Te=2.5	56.0	6.2
Tl			√					Thallium as TlCl	45.1	5.9
Pb			√					**Lead as Pb metal	52.3	7.1
Bi			√					Bismuth as Bimetal	51.6	7.7
Sr		√						Strontium as SrF_2	50.9	5.9
Y				√				Yttrium as YF_3	51.8	5.27
Zr				√				Zirconium as ZrF_4	59.2	4.17
Nb				√				Niobium as Nb_2O_3	50.1	6.6
Mo				√				Molybdenum as MoO_3	59.8	6.8
Pd				√				Palladium as Pd metal	52.1	6.1
Ag				√				**Silver as Ag metal	52.3	7.0
Cd				√				*Cadmium as CdSe	53.0	5.72
In			√					Indium as In metal	52.6	6.0
Sn			√					Tin as Sn metal	55.5	5.3
Sb						√		Antimony as Sb metal	53.5	6.7
I							√	Iodine or Rubium as RbI	54.0	
Cs	√							Bromine or Caesium as CsBr	46.6	6.5
Rb	√							Rubidium or Iodine as RbI	42.8	7.0

A- Alkali; AE – Alkali Earth; PT –Post Transition; T – Transition; L – Lanthanoids
*** may not be stoichiometric; ** will oxidize; H – Heavy Standard; VL – Very Light Standard**

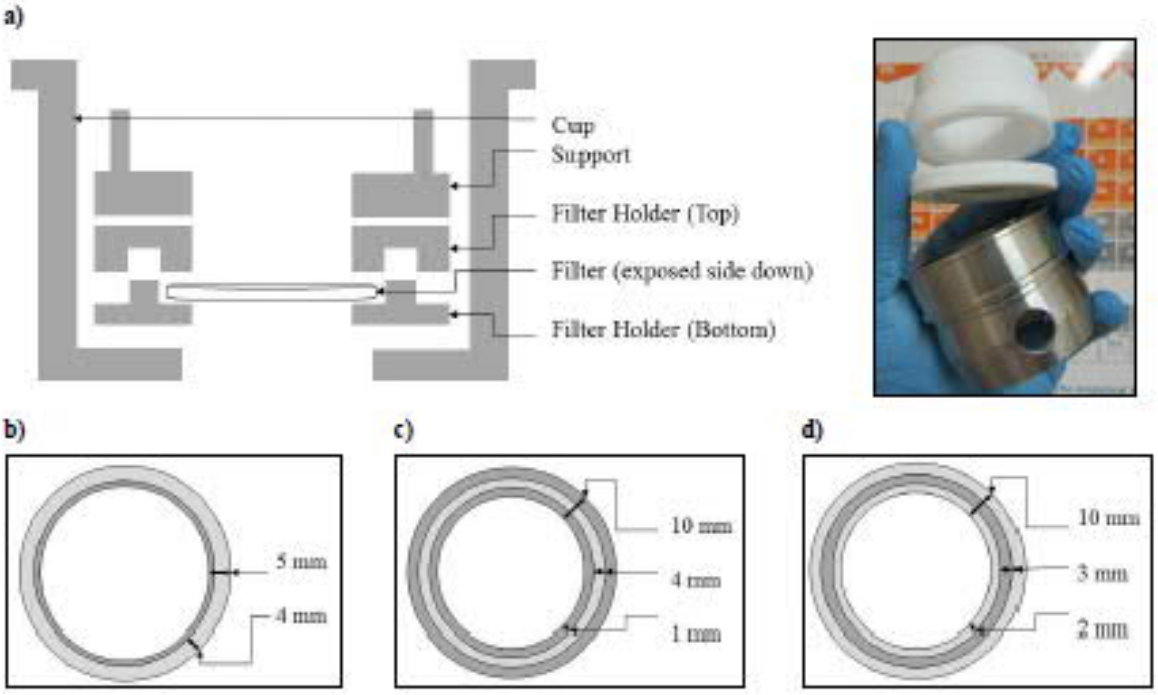

Figure 19: Schematic depicting filter holder designed for use in a 37 mm cup (a); filter holder support (b); the top half of filter holder (c); and bottom half of filter holder (d); (Images by B. Language, 2016)

3.5. Indoor airborne PM source characterisation

3.5.1. Receptor modelling: Enrichment factors (EF) and principal components analysis (PCA)

Two receptor models were employed for the purpose of this study namely PCA and EF. The multivariate model PCA is operates without source composition as input data. The important input dataset is measured elemental concentration data for the model to identify sources and estimate source contribution without source characteristic data. The model is suitable for this study following the collection of PM samples using filters that undergo XRF to acquire elemental composition data. Modelling using PCA works on the principle of correlation between measured concentrations of elemental species and assumes that highly correlated species are from the same source. Prior understanding of possible source characteristics is important when interpreting the output from the model to limit hypothetical and tentative interpretation, or even misinterpretation of the results. The estimation of anthropogenic contribution is estimated using the EF model. The anthropogenic contribution is determined using a reference crustal element such as Al and anything higher than the allocated number (1) represents anthropogenic impact.

An 80% minimum of both valid cases per variable, as well as variables used prior to PCA processing, was set. This was important for removing unqualified variables and cases meaning variables and cases that were missing or have no variance. *STATISTICA* (Dell Software) uses the *NIPALS* algorithm, which extracts the principal components sequentially in a finite number of

steps. This option specifies the maximum number of iterations the *NIPALS* algorithm is allowed to take for extracting a single component. The actual number of iterations the algorithm may take to find a component might actually be less than the maximum number specified depending on the setting of the Convergence criterion. The convergence criterion specifies a minimum improvement (drop) in error that must be made; if the rate of improvement drops below this level, iterations are terminated. The default value is 0.0001, but the setting can be changed to suit the analysis. Note that if you set the convergence to zero, the algorithm will continue until the maximum number of iterations is reached.

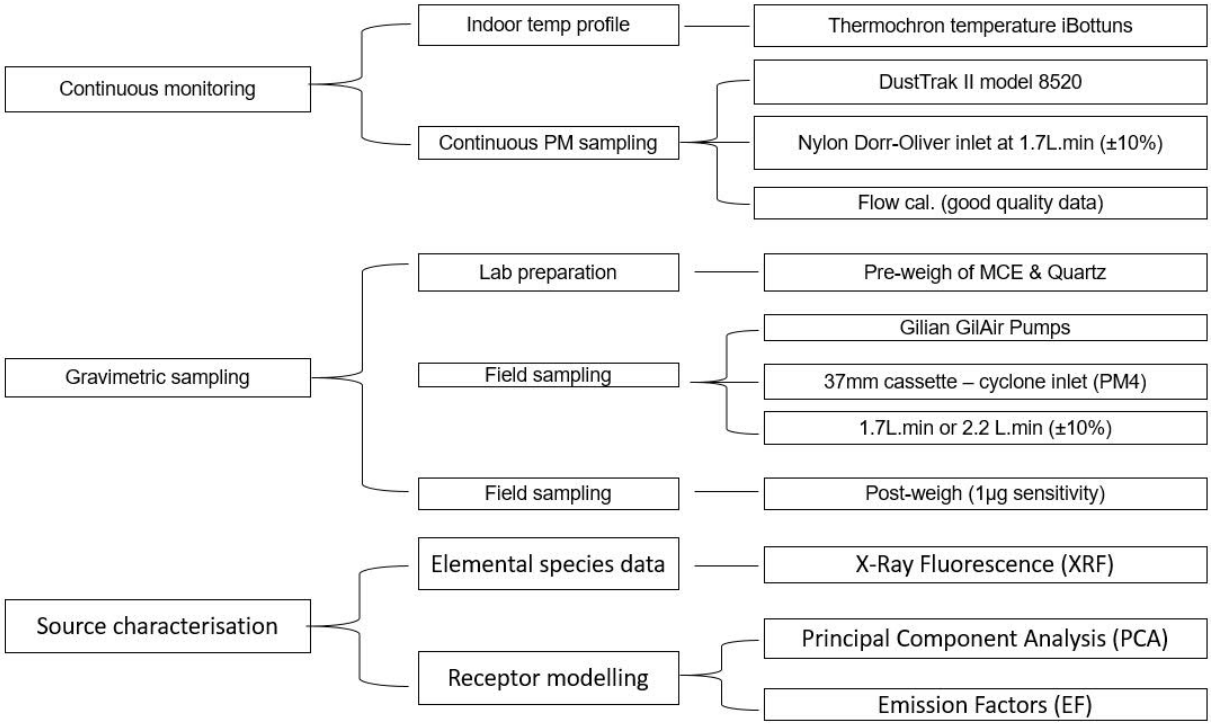

Figure 20: Summary schematic of the tools and methods employed in this research

3.6. Quality assurance and quality control

Preceding the acquisition of households' energy mix of the Sharpeville community, a regular grid was randomly positioned over the area (Figure 21). At each grid intersection, we selected clusters made up of all households on the block to ensure that the selection of the sample houses is not bias. Prior to the start of the sampling campaign, the photometric monitors undergo a yearly dynamic factory calibration using the respirable fraction ISO standard 12103-1, A1 Arizona test dust. Before and after every week of sampling, both photometric instruments and Gilian GilAir Air Pumps were flow calibrated using the cyclone and a Gilian Gilibrator Bubble Flow Meter according to the manufacturer's instructions to correct the air flow of the instruments. Concurrent with the sampling, zero calibrations were done after every data download on the instruments to ensure and maintain functional quality data samples. Filter samples, in general, are prone to contamination of the filters. Therefore, filters were contained in air-tight containers to reduce the

possibility of damage and contamination which could affect the data (Figure 13c). Careful laboratory conditions were maintained during post-weighing of the filter mass.

Figure 21: Grid used for random selection of the sample sites in Sharpeville

The continuous PM_4 data is susceptible to technically induced anomalies. These anomalies are inclusive of the noise observed in the data and random errors which show unexpected and abnormal behaviour in the concertation patterns. To ensure clean, representable and sensible data, a time series was created to detect these anomalies which were mostly outliers and nonsensical spikes on the data. These values were then removed.

It is noted that photometric measurements are prone overestimations and other relative erroneous data. Although gravimetric sampling was conducted simultaneous with photometric, this was done so for the purpose of elemental characterisation. A reference correction factor of 0.715 was adopted from Language *et al.* (2016) conducted in South Africa under relatively similar low-income household environments. The correction factor was then applied to the continuous PM_4 dataset.

CHAPTER 4: ENERGY USE, CONTINUOUS INDOOR PM, BC AND TEMPERATURE MONITORING

CHAPTER OVERVIEW: The results from the questionnaires and continuous monitoring of indoor respirable PM, BC concentrations and temperature within low-income households in Sharpeville are discussed in this chapter.

4.1. Household energy source profile per low-income household in Sharpeville

According to the South African National Census (2011) data, Sharpeville's total population stands at 37,599 people occupying 12,170 households. The average household size is 3.1 inhabitants with 41.1% of the households headed by females. The majority (96%) of households in Sharpeville have access to electricity. In comparison to the 2017 survey conducted by Nova (Table 6), the demographic profile of Sharpeville is slightly different indicating growth (calculated using growth rates from Census 2001–2011 used with 2011 numbers for both total population and number of households). Household income is considered statistically representative of the area rather than accurate because not all respondents were willing to disclose their income. This is a common problem observed in conducting household surveys where respondents are not willing to disclose and thus lead to under- or over-estimation of household income especially low-income communities (Meyer *et al.,* 2015). Therefore ±R3,284 is the estimated total household income p.h/p.m with an average per capita income p.p/p.d of ± R35.

Table 6: A Demographic profile of Sharpeville in 2017 (Source: Nova Detailed Energy Survey, 2017).

Domestic energy usage in Sharpeville 2017 (summary matrix)	
Indicators	
Demographics	
Total population	38,990
No. of households	12,620
Total income p/household p/month	3,284
Mean p/capita income p/person p/day (ZAR)	35
Approximate size (km^2)	5
Population density (persons p/km^2)	1,814

Although 96% of the households are electrified, solid fuel use is still a common practice within the community. A total of 225 DES were obtained by the Nova Research Institute. The results from the DES show a distinct but common energy usage per season where winter fuel consumption is higher than summer. The DES was an exceptional tool for the first research objective that is to determine dominant energy carriers within the community. Firstly, Table 7 show the percentages (%) and the total number of household using solid fuels in their energy mix (only coal, wood and paraffin were considered as they were more common in the area). During winter 24% (2,973) of

the households use wood whereas coal and paraffin are 15% (1,901) and 3% (337), respectively. This indicates that wood is the dominant fuel used for domestic activities even during summer; where more households (1,122 or 9%) use wood compared to the 280 (2%) for both coal and paraffin users. These results are based on the calculated percentage and number of households using wood, coal and paraffin.

Table 7: Summary matrix indicating the Percentage of household energy source in Sharpeville 2017 (Source: Nova Detailed Energy Survey, 2017).

Domestic energy usage in Sharpeville 2017 (summary matrix)		
Indicators		
Dirty energy carrier profile	Winter	Summer
Households using coal	15% (1,907)	2% (280)
Households using wood	24% (2,973)	9% (1,122)
Households using paraffin	3% (337)	2% (280)

Fuel consumption varied per season as shown by the DES. The DES also accounted for the quantities (mass/time) for the monthly and annual consumption of these household energy carriers. It was also found that the consumption of all fuels increased considerably during winter (Table 8), with coal usage being higher than wood. About 85 kg/month of coal was found to be consumed whereas wood and paraffin usage was 47 kg/month and 14 litres/month respectively. The fuel usage was quantified per coal-, wood- and paraffin-using household and not in terms of the total number of households using a fuel type. It is found that some households prefer to use coal more often than individual wood-using households. Even when the usage was averaged to be representative of the whole community, coal usage was higher than wood and paraffin in winter by a factor of 1.2 and 32, respectively. Considering the location of this study area, where minimum ambient winter temperature conditions range between 1 and 17°C compared to summer 15–32°C ranges. Therefore, the consumption of fuel is higher in these cold months. This implies coal is dominant in winter per coal-using household than wood whereas in summer wood use per wood-using household is higher.

Table 8: Summary matrix showing quantities of domestic energy carriers differentiated per season (mass/month) in Sharpeville 2017 (Source: Nova Detailed Energy Survey, 2017).

Domestic energy usage in Sharpeville 2017 (summary matrix)		
Indicators		
Dirty energy carrier profile (in quantities)	Winter	Summer
kg/month (litre/month for paraffin)		
Average coal use per coal-using household	85	8
Average wood use per wood-using household	47	13
Average paraffin use per paraffin-using household	14	9
tonne/month (kilolitre/annum for paraffin)		
Average coal use per community	162	2
Average wood use per community	138	15
Average paraffin use per community	5	3

Wood, coal and paraffin annual quantities consumed are presented in Table 10 below. Coal use per coal-using household remained higher than wood use per wood-using household at 400 kg/annum and 292 kg/annum, respectively. However, when the quantities are averaged to represent annual coal and wood used by the community, wood exceeded the total coal use by a factor of 1 which is not significantly high but shows variation. Paraffin remained the least-used fuel for domestic purposes with an annual 129 kilolitres/annum per paraffin-using household and 40 kilolitres/annum for the entire community. Seasonal differences in fuel used clearly show that coal per coal-using household was the preferred energy carrier. Due to the decreased coal use but elevated wood use in the summer, annual consumption of wood exceeded that of coal. The durability and denseness of coal explain the elevated use in winter whereby households could burn this fuel for extended periods of time.

Table 9: Summary matrix showing quantities of domestic energy carriers per year (in mass/annum) in Sharpeville 2017 (Source: Nova Detailed Energy Survey, 2017).

Domestic energy usage in Sharpeville 2017 (summary matrix)	
Indicators	
2) Dirty energy carrier profile (in quantities)	
kg/annum (litre/annum for paraffin)	
Annualised coal use per coal-using household	400
Annualised wood use per wood-using household	292
Annualised paraffin use per paraffin-using household	129
tonne/annum (kilolitre/month for paraffin)	
Annualised coal use per community	664
Annualised wood use per community	672
Annualised paraffin use per community	40

4.2. Continuous indoor aerosol monitoring

4.2.1. Summary descriptive statistics

Summary statistics of the indoor continuous PM_4 concentrations for the two seasons are provided in Table 10 below. Most of the PM_4 sampling was done in the kitchens and living rooms of the selected households. The houses varied in terms of physical structure, insulation and ventilation methods as well as household energy profile. All houses had access to electricity and were formal in structure. The difference was that other standard issued RDP houses had been extended and refurbished with different building material. Insulation measures mainly included ceilings, and all houses used windows and doors for ventilation purposes. Wood and paraffin were the most widely used energy carriers for domestic activities within the sample houses. Sources of PM within and around the homes included domestic burning and close proximity to intermittent waste burning and unpaved roads. Indoor concentrations are compared to the 24hr National Ambient Air Quality Standards (NAAQS). It is understood that there are not indoor standards for indoor particulate concentrations. However, the draft guideline for monitoring domestic indoor air quality mentions stated the use of the NAAQS as a proxy for indoor PM measurements because the health implications are the same (DoH, 2009).

During winter, continuous measurements of respirable airborne PM (PM_4) showed that on average NSFB and SFB households experienced concentrations of 150 (± 85) µg/m^3 and 220 (± 131) µg/m^3 respectively. Summer showed a less extreme scenario than winter where NSFB and SFB households' mean PM_4 concentrations were 70 (±50) µg/m^3 and 99 (±51) µg/m^3, respectively. For both household types, winter concentrations on average twice as high as those experienced in summer. Four houses in winter showed very high mean concentrations of 107

(±33) µg/m^3, 205 (±71) µg/m^3, 163±56.06 µg/m^3, and 107 (±55) µg/m^3, in H06, H07, H08 and H14 respectively. Mean concentrations in summer remained high in two of these houses, namely H07 and H17, which were reliant on solid fuels in their household energy mix.

Table 10: Summary of descriptive statistics of daily average indoor PM_4 concentrations (µg/m^3) in Sharpeville for winter and summer 2017 measured with a DustTrak II.

Winter 2017							**Summer 2017**					
	N	***Mean***	***Median***	***Min***	***Max***	***SD***	***N***	***Mean***	***Median***	***Min***	***Max***	***SD***
H01	15	92	74	46	201	43	15	38	35	19	60	11
H02	15	96	87	25	181	46	19	36	32	22	80	15
H03	15	108	99	38	305	67	13	41	44	19	55	13
H04	13	84	81	39	141	31	15	39	37	20	62	12
H05	17	71	58	30	154	37	29	32	31	14	68	12
H06	13	107	94	67	175	33	15	35	37	11	66	13
H07	16	205	180	125	349	71	29	148	141	74	296	51
H08	12	163	153	65	282	56	15	52	45	13	149	35
H09	15	54	57	27	73	15	15	30	30	7	57	13
H10	15	89	76	23	303	67	15	26	26	16	41	8
H11	14	72	70	27	104	23	15	32	33	19	56	9
H12	15	51	53	8	86	24	15	26	25	17	39	7
H13	14	69	71	30	97	19	15	37	33	22	82	15
H14	12	107	93	36	193	55	15	33	30	22	71	12
H15	15	64	70	28	101	20	15	46	41	20	101	20
H16	15	87	94	43	136	26	-	-	-	-	-	-
H17	-	-	-	-	-	-	15	81	76	33	168	33

4.2.2. Daily indoor aerosol concentrations

During winter all indoor PM_4 concentrations exceeded the $PM_{2.5}$ 24hr National Ambient Air Quality Standard (NAAQS) in all households. Seven houses (H01, H02, H03, H06, H07, H08 and H14) stand out in that their median 24hr concentrations exceeded both the 24hr average PM_{10} and $PM_{2.5}$ NAAQS (> 75 µg.m^3) (Figure 22 a and b). The remaining nine households were well below the 24hr PM_{10} NAAQS (< 75 µg.m^3); however, they exceeded the 24hr $PM_{2.5}$ NAAQS (≥ 40 µg.m^3). Winter 24hr median indoor PM_4 concentrations in households H06, H07 and H08 were higher than the other houses (106 µg.m^3, 165 µg.m^3 and 80 µg.m^3 respectively). Household H06 is reliant on electricity but nevertheless experienced elevated indoor particulate concentrations during winter.

Summer concentrations were frequently below both the $PM_{2.5}$ and PM_{10} NAAQS (Figure 23 a. and b.). There was a significant difference between indoor PM_4 concentrations from winter to summer. Respirable particulate loading within households goes from > 75 µg/m^3 (PM_{10} NAAQS) in winter to < 40 µg/m^3 ($PM_{2.5}$ NAAQS) in summer which represents ~47% decrease. Two houses,

namely H08 and H17, showed indoor concentrations exceeding only the $PM_{2.5}$ 24hr standard in summer with average concentrations of 50 µg/m^3 and 60 µg/m^3 respectively. One household, H07, showed extreme indoor concentrations up to a maximum of 361.8 µg/m^3 and a median of 114 µg/m^3 which is above both $PM_{2.5}$ and PM_{10} standards. This may be considered representative of households reliant almost exclusively on solid fuels as their primary energy carriers in both seasonal campaigns.

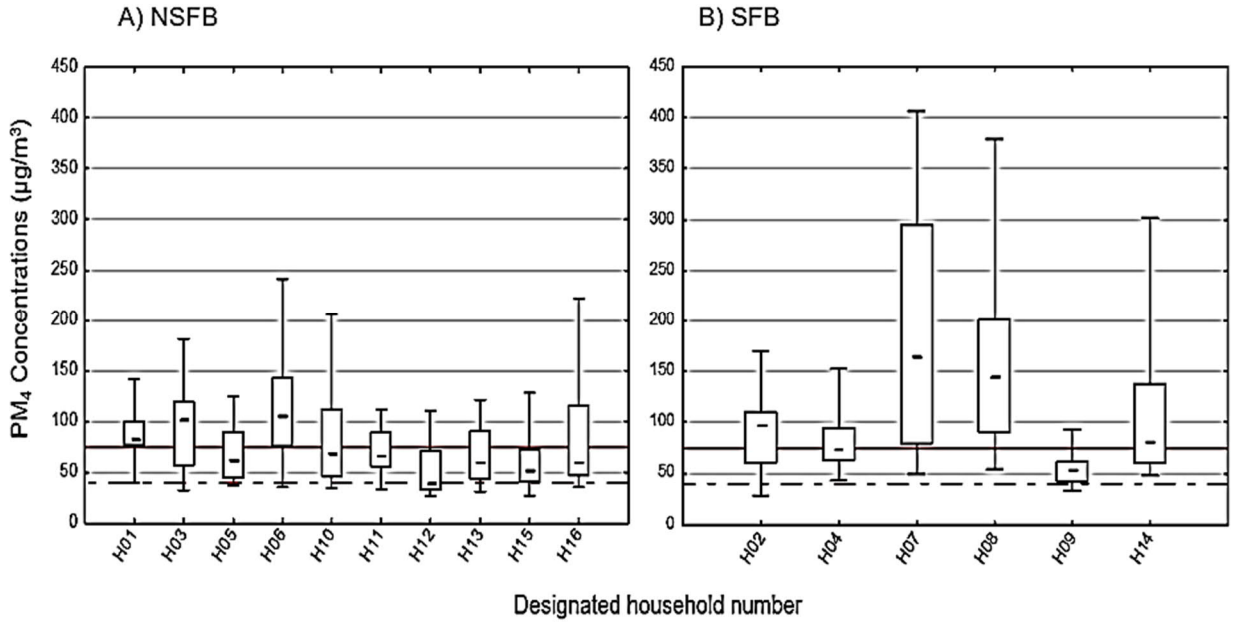

Figure 22: Box & Whisker Plot representing 24hr indoor average concentrations for winter 2017 at Sharpeville compared to both 24hr $PM_{2.5}$ (dotted line) and PM_{10} (solid line) NAAQS (Box = 25th and 75th percentiles; Dash = median; Whiskers = Min & Max).

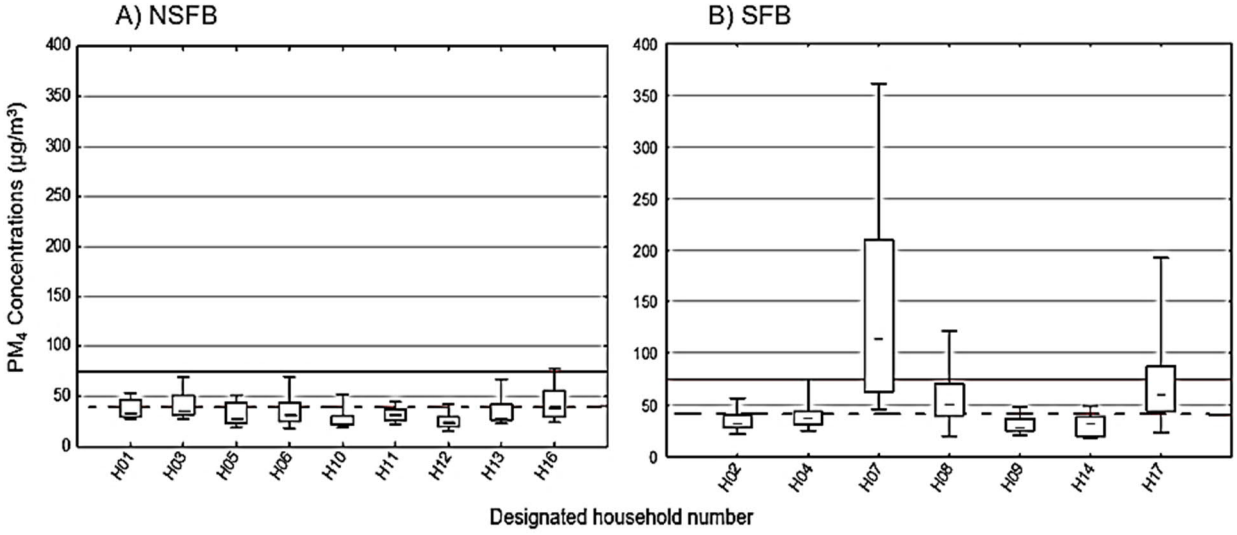

Figure 23: Box & Whisker Plot representing 24hr indoor average concentrations for summer 2017 at Sharpeville compared to both 24hr $PM_{2.5}$ (dotted line) and PM_{10} (solid line) NAAQS (Box = 25th and 75th percentiles; Dash = median; Whiskers = Min & Max).

4.2.3. Comparison between SFB and NSFB houses

In order to evaluate the benefits of implementing air pollution reduction strategies in low-income houses, a comparison between the houses that use solid fuel combustion and those that do not is important. It is clear from the statistics as presented for each house (Table 10, Figure 22 and Figure 23) that the data is highly variable. This makes the comparison between houses more complicated as differences may not be greater than the standard deviation suggesting no variation between houses. A key characteristic in the data for comparison is the level of variability between SFB and NSFB houses. Daily averages were used to determine the significance of variability between these household categories.

4.2.3.1. Variation per household category

An unpaired T-test, by conventional criteria, was undertaken to evaluate the SFB and NSFB houses. The test indicates that the two household categories indeed are statistically significantly (p <0.05) different. This clearly indicates that by illuminating the solid fuel combustion in houses (even if not all simultaneously) would significantly reduce (by a factor of 2) the indoor PM_4 concentrations and therefore resident exposure to inhalable pollution. The benefits of such interventions are likely to be even more significant if implemented on a larger scale as the ambient concentrations would start to drop which would in the longer term also reduce the exposure in the NSFB houses.

Table 11: T-test for independent samples variation between both household categories for winter and summer

	SFB vs. NSFB	
	Winter	**Summer**
Mean (SFB)	114 (µg/m^3)	62 (µg/m^3)
Mean (NSFB)	80 (µg/m^3)	35 (µg/m^3)
t-value	4	7
p-Value	0.000396	0.000000
Valid N (Winter)	15	15
Valid N (Summer)	15	15
Std.Dev. (Winter)	21	13
Std.Dev. (Summer)	25	7
F-ratio	1.394348	3.220870
p-Variance	0.542164	0.036295

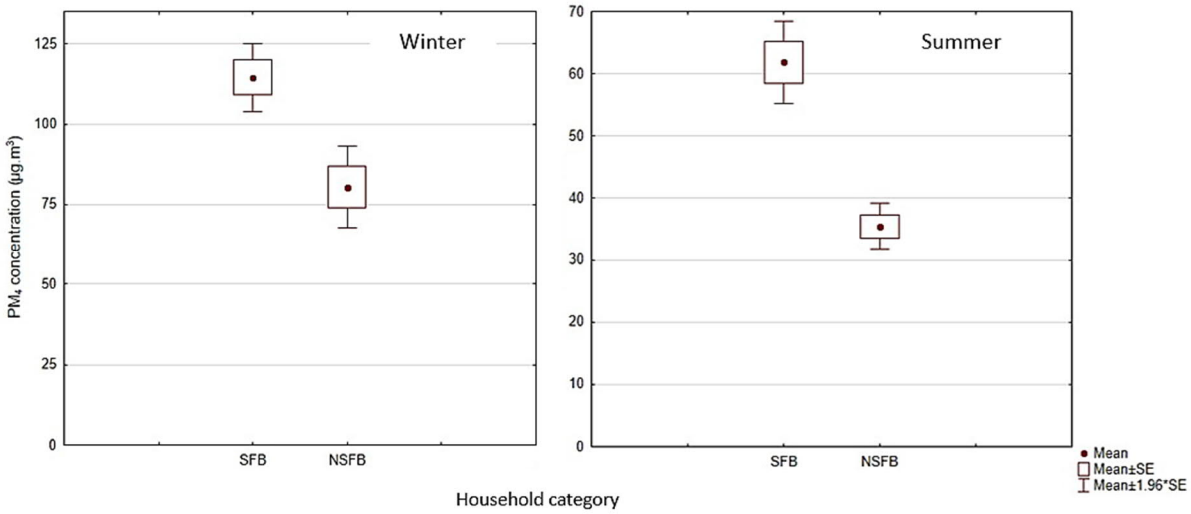

Figure 24: Representation of the statistical variability between household categories during winter and summer (Note: y-axis scales differ).

4.2.3.2. Variation in households within the same category by season

In addition to the household categories being different, a test has also been done to see if the houses were statistically; different between seasons. The test shows clearly that concentrations of PM_4 in NSFB houses is significantly higher during winter (*Mean* = 80 µg/m^3) and summer (*Mean* = 35 µg/m^3) a *p*-Value of <0.05 was calculated (Table 12). A similar result was obtained for the SFB houses where concentrations in winter (*Mean* = 114 µg/m^3) are significantly higher than during summer (*Mean* = 62 µg/m^3) *p*-Value <0.05. The seasonal differences are driven by the changes in the use of the amount of solid fuel combustion in each SFB house. According to Language *et al.* (2016), the increased use of solid fuels in winter coupled with the less ventilation activity (closed doors and windows to contain the heat) exacerbates indoor concentrations of PM_4. In addition, emissions resulting from the higher burning rates increase emissions on the local atmosphere. This leads to the increase in PM concentrations also in the NSFB houses.

Table 12: T-test for Independent Samples variation by season for each household type.

	NSFB	SFB
	Winter vs. Summer	Winter vs. Summer
Mean (Winter)	80 (µg/m^3)	114 (µg/m^3)
Mean (Summer)	35 (µg/m^3)	62 (µg/m^3)
t-value	7	8
p-Value	<0.0001	0.0001
Valid N (Winter)	15	15
Valid N (Summer)	15	15
Std.Dev. (Winter)	25	21
Std.Dev. (Summer)	7	13
F-ratio	12	3
p-Variance	0.000040	0.081322

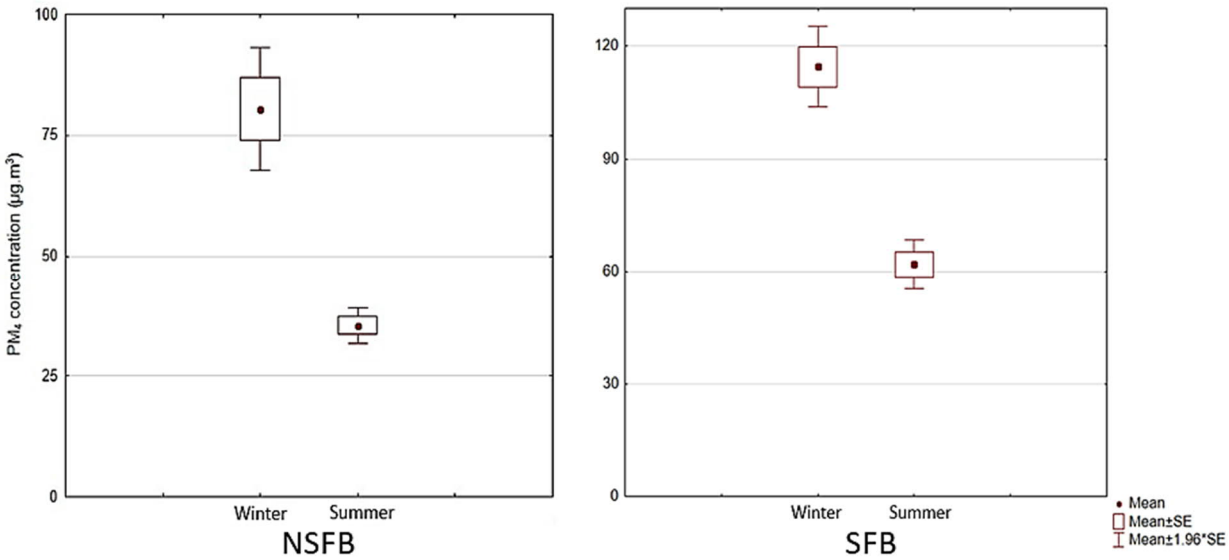

Figure 25: Seasonal variability between winter and summer for both household types (Note y-axis scales differ).

4.2.3.3. Variability of each household to one another

The difference amongst households was determined using matrix created where the *p*-Values were used to analyse the statistical significance of the variability in each sample (Table 13 and Table 14). Red-shaded cells represent cells with *p*-values < 0.05, which means the difference is statistically significant between the household samples. The non-shaded cells represent *p*-values > 0.05 which implies the difference is not statistically significant Different colours were also used for the top (x-axis) and the left (y-axis) represent the different household categories within the sample, where green depicts NSFB and grey is used for SFB.

Winter comparisons showed a number of NSFB houses' PM_4 concentrations with no significant statistical variation for some of the SFB houses (Table 13). Households H01, H02, H03, H04, H05, H10 and H14, all had $p > 0.05$ ranging to as high as $p = 0.95$. This implies, although H02, H04 and H14 are SFB households as opposed to H01, H03, H05 and H10, the variability between them was not statistically significant. Households H05, H09, H10 and H15 also showed comparable indoor concentrations ($p > 0.05$). Households H07 and H08 had a *p*-value of 0.1, making them not that different to one another. However, they showed a difference that is statistically significant ($p < 0.05$) when compared to all of the sample houses. Overall, within the SFB household category, H02, H04 and H14 appeared to have less statistical difference to most of the NSFB samples.

Table 13: Test of means to assess the variability for all houses in relation to one another during winter (green shaded cells: NSFB, grey shaded cells: SFB, red shaded cells p <0.05).

	H01	H02	H03	H04	H05	H06	H07	H08	H09	H10	H11	H12	H13	H14	H15	H16
H01		0.78	0.43	0.58	0.15	0.30	0.00	0.00	0.00	0.90	0.13	0.00	0.08	0.44	0.03	0.72
H02	0.78		0.58	0.41	0.09	0.48	0.00	0.00	0.00	0.73	0.08	0.00	0.05	0.60	0.02	0.50
H03	0.43	0.58		0.23	0.06	0.97	0.00	0.03	0.00	0.44	0.06	0.00	0.04	0.95	0.02	0.26
H04	0.58	0.41	0.23		0.33	0.07	0.00	0.00	0.00	0.79	0.26	0.00	0.15	0.21	0.05	0.75
H05	0.15	0.09	0.06	0.33		0.01	0.00	0.00	0.11	0.34	0.96	0.09	0.86	0.05	0.50	0.17
H06	0.30	0.48	0.97	0.07	0.01		0.00	0.01	0.00	0.38	0.00	0.00	0.00	0.96	0.00	0.08
H07	0.00	0.00	0.00	0.00	0.00	0.00		0.10	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H08	0.00	0.00	0.03	0.00	0.00	0.01	0.10		0.00	0.01	0.00	0.00	0.00	0.02	0.00	0.00
H09	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00		0.06	0.02	0.65	0.02	0.00	0.16	0.00
H10	0.90	0.73	0.44	0.79	0.34	0.38	0.00	0.01	0.06		0.36	0.05	0.29	0.47	0.17	0.92
H11	0.13	0.08	0.06	0.26	0.96	0.00	0.00	0.00	0.02	0.36		0.03	0.75	0.04	0.33	0.10
H12	0.00	0.00	0.00	0.00	0.09	0.00	0.00	0.00	0.65	0.05	0.03		0.03	0.00	0.13	0.00
H13	0.08	0.05	0.04	0.15	0.86	0.00	0.00	0.00	0.02	0.29	0.75	0.03		0.02	0.47	0.04
H14	0.44	0.60	0.95	0.21	0.05	0.96	0.00	0.02	0.00	0.47	0.04	0.00	0.02		0.01	0.24
H15	0.03	0.02	0.02	0.05	0.50	0.00	0.00	0.00	0.16	0.17	0.33	0.13	0.47	0.01		0.01
H16	0.72	0.50	0.26	0.75	0.17	0.08	0.00	0.00	0.00	0.92	0.10	0.00	0.04	0.24	0.01	

Table 14: Test of means to assess the variability across all houses in relation to one another during summer (green shaded cells: NSFB, grey shaded cells: SFB, red shaded cells p <0.05).

	H01	H02	H03	H04	H05	H06	H07	H08	H09	H10	H11	H12	H13	H14	H16	H17
H01		0.85	0.52	0.75	0.84	0.54	0.00	0.14	0.06	0.00	0.10	0.00	0.82	0.22	0.20	0.00
H02	0.85		0.71	0.94	0.99	0.50	0.00	0.18	0.08	0.01	0.14	0.00	0.72	0.24	0.31	0.00
H03	0.52	0.71		0.74	0.67	0.27	0.00	0.28	0.03	0.00	0.04	0.00	0.46	0.10	0.48	0.00
H04	0.75	0.94	0.74		0.92	0.40	0.00	0.19	0.04	0.00	0.07	0.00	0.63	0.16	0.31	0.00
H05	0.84	0.99	0.67	0.92		0.46	0.00	0.17	0.06	0.00	0.09	0.00	0.71	0.19	0.27	0.00
H06	0.54	0.50	0.27	0.40	0.46		0.00	0.09	0.23	0.03	0.42	0.02	0.76	0.59	0.10	0.00
H07	0.00	0.00	0.00	0.00	0.00	0.00		0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
H08	0.14	0.18	0.28	0.19	0.17	0.09	0.00		0.02	0.01	0.04	0.01	0.13	0.05	0.53	0.03
H09	0.06	0.08	0.03	0.04	0.06	0.23	0.00	0.02		0.44	0.55	0.31	0.16	0.48	0.01	0.00
H10	0.00	0.01	0.00	0.00	0.00	0.03	0.00	0.01	0.44		0.08	0.75	0.02	0.11	0.00	0.00
H11	0.10	0.14	0.04	0.07	0.09	0.42	0.00	0.04	0.55	0.08		0.03	0.28	0.84	0.02	0.00
H12	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.31	0.75	0.03		0.01	0.06	0.00	0.00
H13	0.82	0.72	0.46	0.63	0.71	0.76	0.00	0.13	0.16	0.02	0.28	0.01		0.41	0.19	0.00
H14	0.22	0.24	0.10	0.16	0.19	0.59	0.00	0.05	0.48	0.11	0.84	0.06	0.41		0.04	0.00
H16	0.20	0.31	0.48	0.31	0.27	0.10	0.00	0.53	0.01	0.00	0.02	0.00	0.19	0.04		0.00
H17	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	

Summer concentrations showed a significant statistical difference (p < 0.05) between H07 and H17 to all the other household samples (Table 14). This could have been caused by these houses remaining reliant on the use of solid fuels throughout the summer campaign as opposed to other households that reduced solid fuel use. Households H02, H04, H08, H09 and H14 appeared to

have concentrations with less significant differences, than most of the NSFB households. H10 and H12 are extremely statistically different with 69% and 75% of the sample group, respectively.

Looking at the variability between all the household samples means that indoor PM_4 concentrations within these categories could not be generalised. Normally, assumptions and conclusions are drawn by looking at the mean. Consequently, the mean for all houses did not provide a good representative assessment of individual exposure risk for residents. Average concentrations within SFB households were higher than NSFB samples. However, depending on the distribution and the variance within the distribution, the statistical significance can be there or not. Determining the statistical difference allows for less generalisation of the results. It is clear that each home is different and that indoor sources, household characteristics and surrounding sources affected the behaviour of indoor particulate matter concentrations.

4.2.4. Diurnal concentrations

4.2.4.1. Indoor diurnal concentrations within NSFB households

Changes in concentrations throughout the day represent the distribution and behaviour of airborne respirable particulates and their sources. The average diurnal pattern of PM_4 concentrations showed a bi-modal distribution. Morning and evening peaks are observed in both household types for both winter and summer 2017. During winter, NSFB households' morning peaks occurred from 06h00 until 09h00 with concentrations ranging between 80 µg/m^3 and 150 µg/m^3 (Figure 26 a and b). In the evening, from 19h00 to 22h00, PM_4 peak concentration ranged from 70 µg/m^3 to 175 µg/m^3. Elevated concentrations were often experienced in the evening between 19h00 and 22h00, with maximum values > 500 µg/m^3 (households H03 and H16). Households H05, H11 and H12 on average experienced relatively lower concentrations not exceeding 100 µg/m^3 throughout the day. Although these households rely solely on electricity as a source of energy for domestic activities, indoor concentrations appeared to be elevated above the 24hr NAAQS for PM_{10} and $PM_{2.5}$ (> 75 µg/m^3) during peak hours in winter.

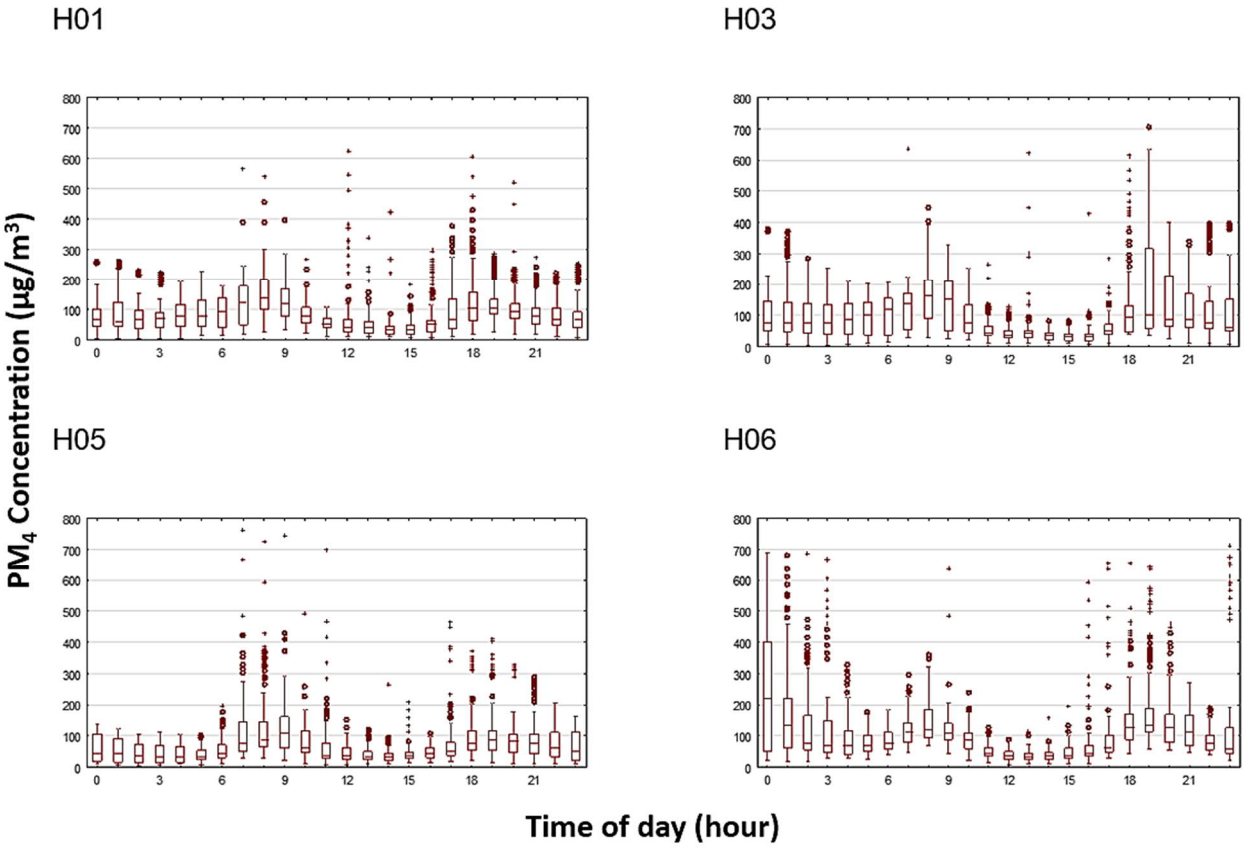

Figure 26(a): Diurnal patterns of hourly-averaged PM_4 concentrations measured during winter 2017 at Sharpeville in non-solid fuel burning (NSFB) households (Box = 25th & 75th percentile; Whiskers = Min & Max; Dash = Mean; Star shapes = Extremes; Circles = outliers). (Note: y-axis scales differ).

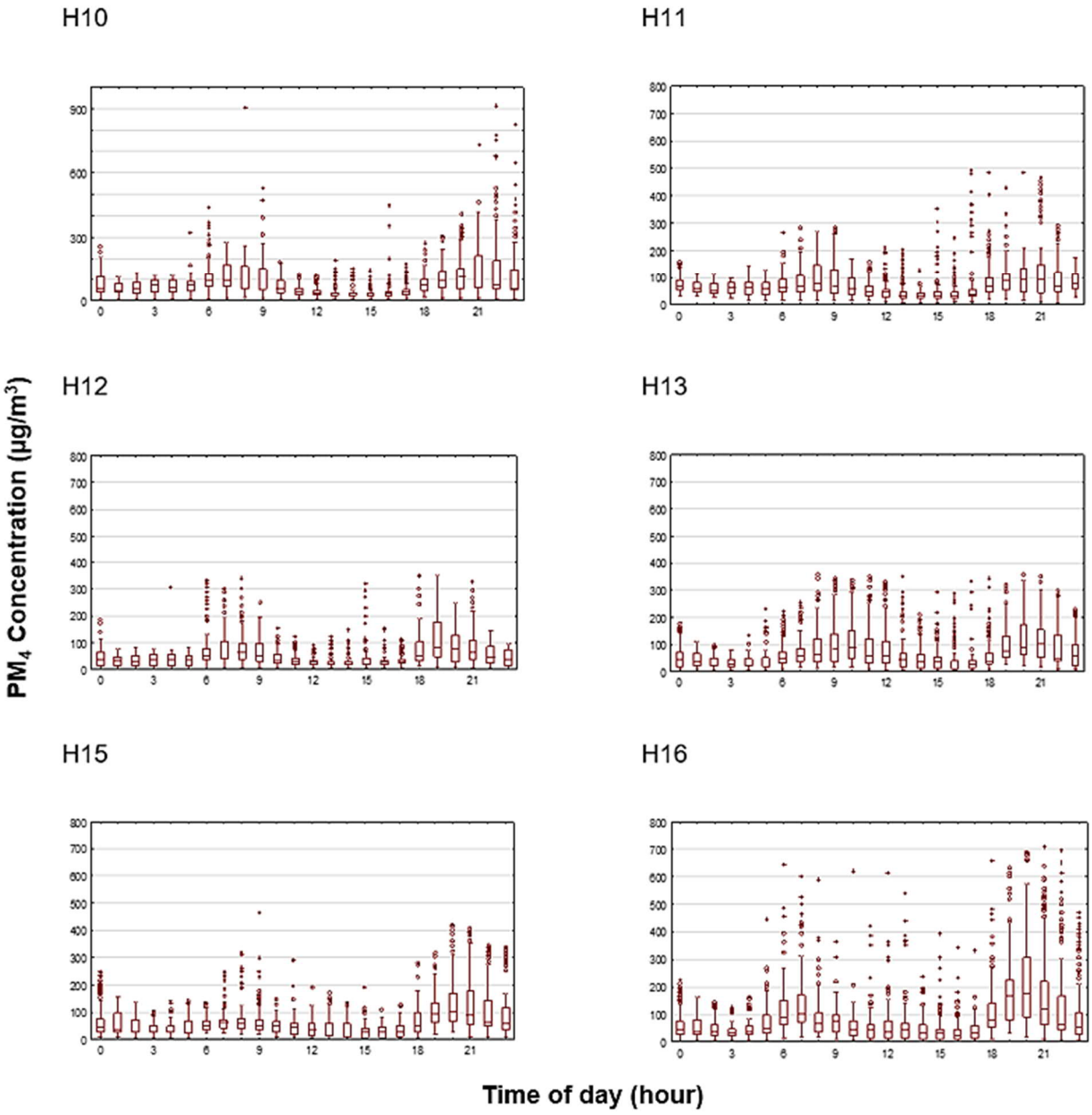

Figure 26(a): Diurnal patterns of hourly-averaged PM4 concentrations measured during winter 2017 at Sharpeville in non-solid fuel burning (NSFB) households (Box = 25th & 75th percentile; Whiskers = Min & Max; Dash = Mean; Star shapes = Extremes; Circles = outliers). (Note: y-axis scales differ).)

Summer average indoor PM_4 mass concentrations show a less extreme scenario compared to winter. Average indoor concentrations within NSFB households (Figure 27 a and b) were 50 µg/m^3 and lower for both morning and evening peak hours (06h00–09h00 and 17h00–-21h00). Highest measured maximum concentrations were observed in H16 (> 200 µg/m^3) in the evening. Although these concentrations levels showed a less extreme scenario, compared to the NAAQS they exceeded the average 24hr $PM_{2.5}$ standard of 40 µg/m^3.

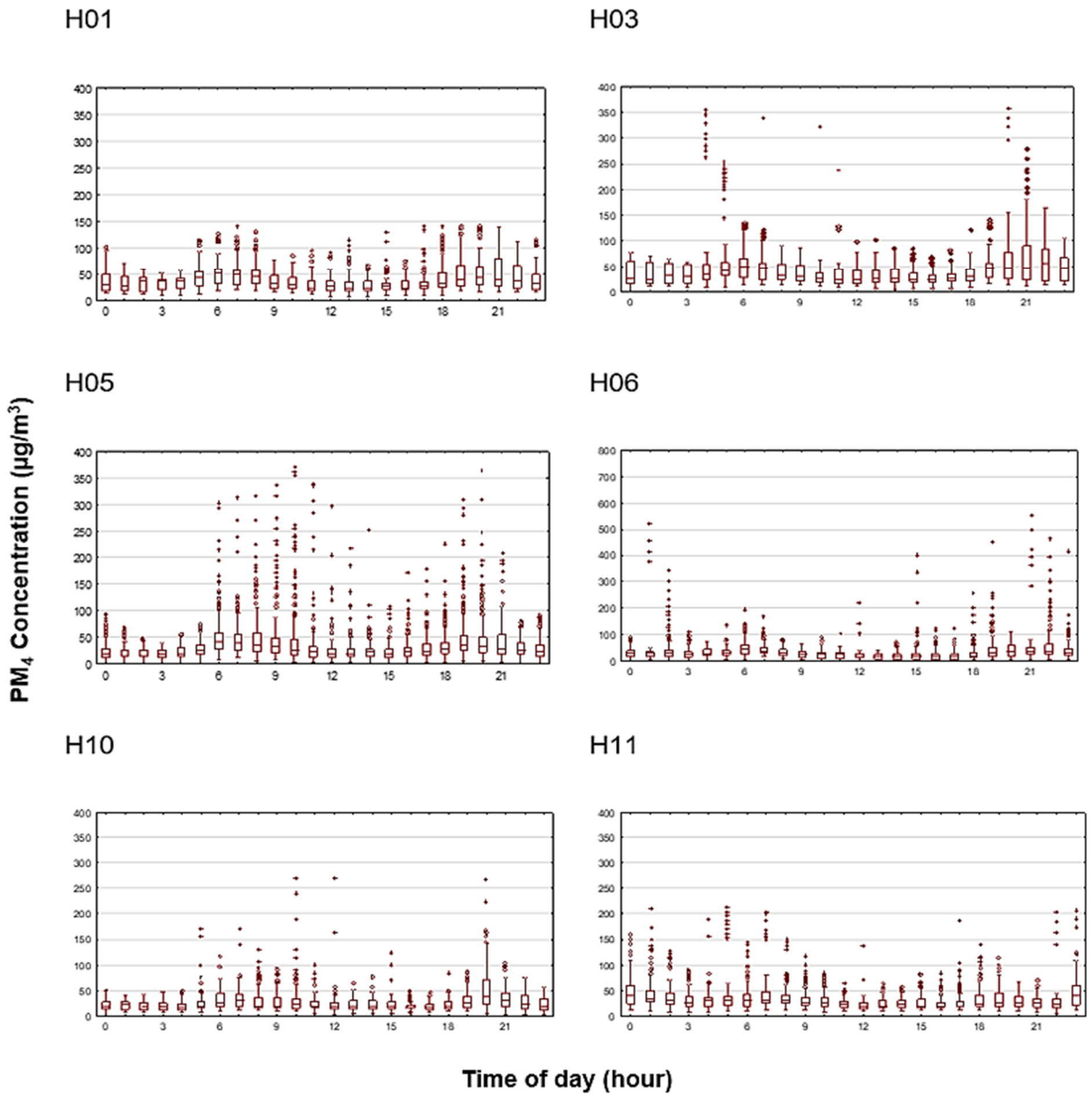

Figure 27(a): Diurnal pattern of hourly-averaged PM_4 concentrations measured during summer 2017 at Sharpeville in non-solid fuel burning (NSFB) households (Box = 25th & 75th percentile; Whiskers = Min & Max; Dash = Mean; Star shapes = Extremes; Circles = outliers). (Note: x- and y-axis scales differ).

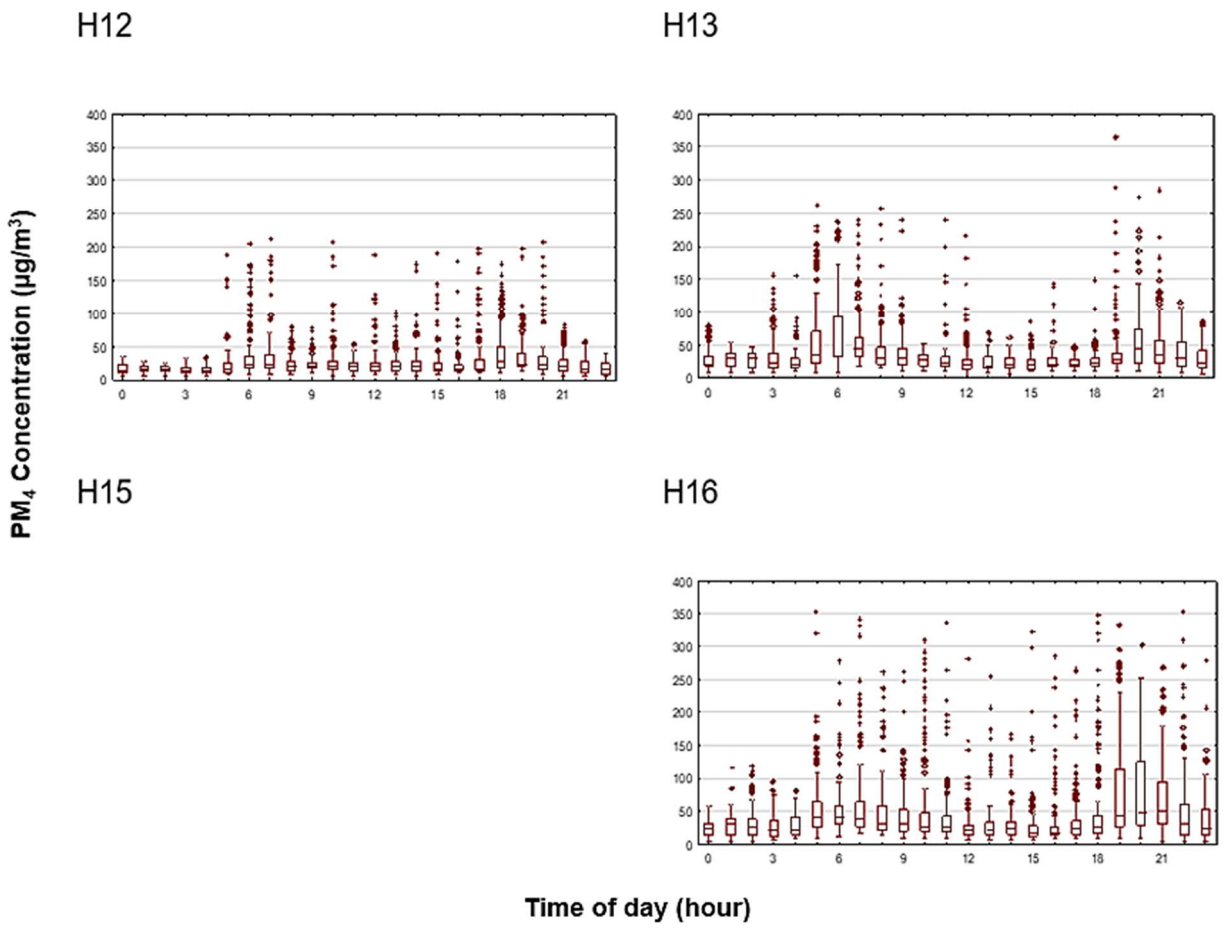

Figure 27(a): Diurnal pattern of hourly-averaged PM4 concentrations measured during summer 2017 at Sharpeville in non-solid fuel burning (NSFB) households (Box = 25th & 75th percentile; Whiskers = Min & Max; Dash = Mean; Star shapes = Extremes; Circles = outliers). (Note: x- and y-axis scales differ).

4.2.4.2. Indoor diurnal concentrations within SFB households

In comparison to NSFB households, concentrations of indoor airborne PM_4 in SFB houses were problematic in winter (Figure 28). Morning peaks within the households were on average > 150 µg/m^3 (06h00 and 09h00) and evening peaks were > 200 µg/m^3. During the winter sampling period, not all SFB houses used alternative fuels throughout the campaign; thus, certain sample houses experienced lower concentrations than others. Three households (H07, H08 and H14) experienced elevated concentrations > 300 µg/m^3 during peak hours compared to H02, H04 and H09. Of all the SFB households, H09 had the lowest average indoor concentrations (< 100 µg/m^3).

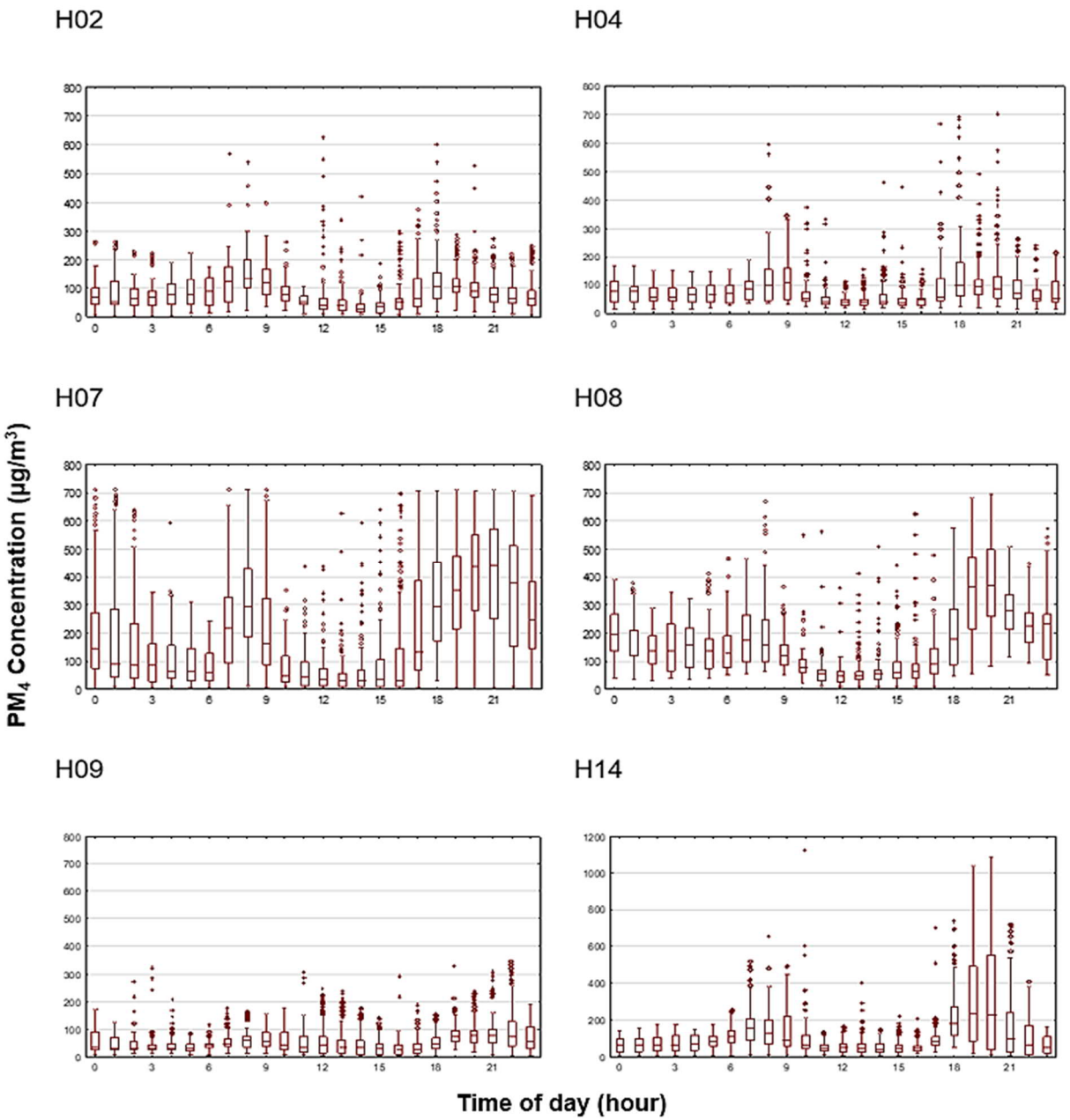

Figure 28: Diurnal pattern of hourly-averaged PM_4 concentrations measured during winter 2017 at Sharpeville in solid fuel burning (SFB) households (Box= 25th & 75th percentile; Whiskers = Min & Max; Dash = Mean; Star shapes = Extremes; Circles = outliers). (Note: x- and y-axis scales differ).

SFB households show decreased indoor PM_4 mass concentrations in summer, with measured particulate levels < 50 µg/m^3 in households H02, H04, H09 and H14 for both peak events of the day (Figure 29 a and b). Elevated ambient temperatures reduce the need to burn fuels for space heating; therefore less extreme indoor PM_4 concentrations were observed for this period. However, H08 experienced elevated concentrations between 50 and 80 µg/m^3 for both peak events. Respirable particulate pollution was a concern for households H07 and H17. Average diurnal concentrations in H07 rose above 100 µg/m^3 during the morning peak (05h00–07h00) while during the evening peak (19h00-22h00) PM_4 levels were even more elevated, at > 200

µg/m^3 (between 211 µg/m^3 and 390 µg/m^3). Households H07 and H17 were the two houses with maximum concentrations measured > 500 µg/m^3; where the latter households only used solid fuels for cooking while electricity was reserved for lighting only.

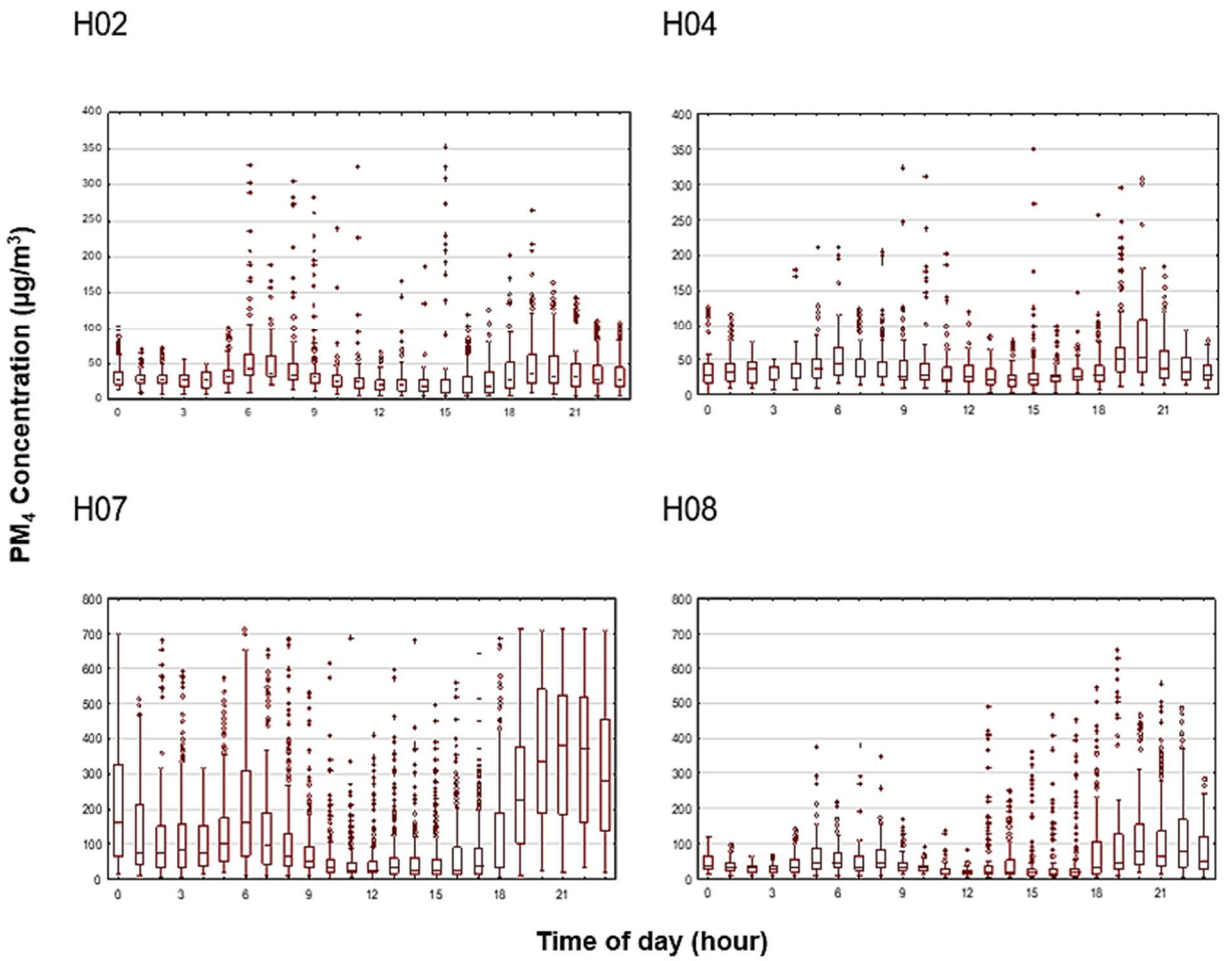

Figure 29(a): Diurnal pattern of hourly-averaged PM_4 concentrations measured during summer 2017 at Sharpeville in solid fuel burning (SFB) households (Box= 25th & 75th percentile; Dash= Mean; Whiskers= Min & Max; Star shapes= Extremes; Circles= outliers). (Note: x- and y-axis scales differ)

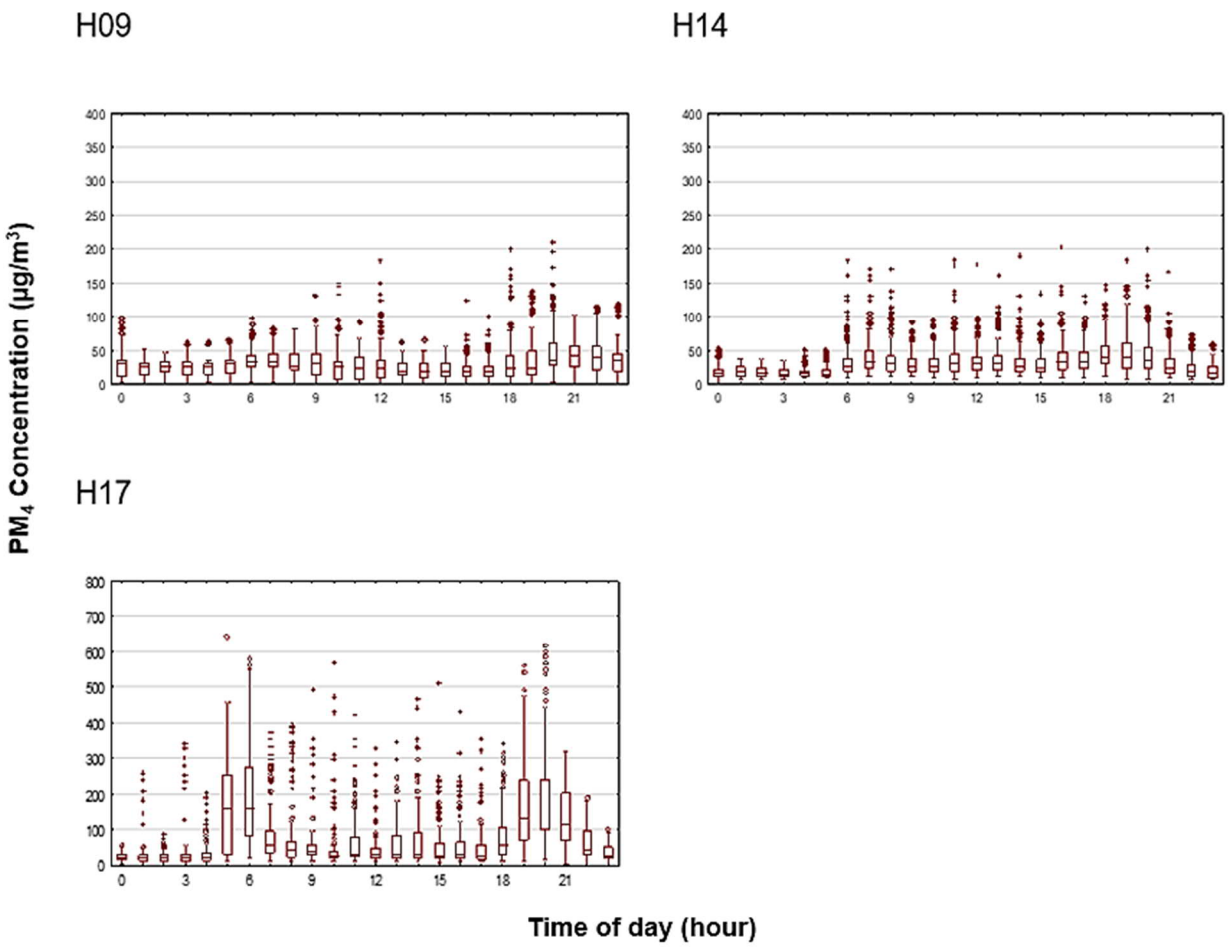

Figure 29(a): Diurnal pattern of hourly-averaged PM_4 concentrations measured during summer 2017 at Sharpeville in solid fuel burning (SFB) households (Box= 25th & 75th percentile; Dash= Mean; Whiskers= Min & Max; Star shapes= Extremes; Circles= outliers). (Note: x- and y-axis scales differ).

During winter, average diurnal indoor concentrations of PM_4 were higher in SFB than NSFB households by a factor of three and higher. There was less diurnal variability between the two household types during the day from 11h00–14h00 where indoor concentrations were 60 and 50 µg/m^3 respectively for SFB and NSFB. Solid fuel burning households typically resulted in higher concentrations during peak hours compared to NSFB. This shows that complete reliance on solid fuels is problematic and result in a health hazard for residents exposed to respirable particles.

Summer average diurnal indoor concentrations of PM_4 remained higher in SFB compared to NSFB households by a factor of three during both peak hours of the day. There is less diurnal variability between both household types during the day between 11h00 and 14h00 where indoor concentrations were 40–50 µg/m^3. Both household types experienced decreased indoor particulate concentrations compared to the levels during winter. This may be a function of warmer ambient temperatures when the need to burn solid fuels for space heating is lower.

The households using only electricity as their main energy carrier had similar diurnal indoor PM concentration patterns for both winter and summer. On the other hand, the SFB houses showed significant seasonal variations in indoor concentrations. Highest concentrations are recorded in winter for both NSFB and SFB houses. For NSFB households, the indoor concentrations in summer decreased by ~58.3% in the summer morning peak (06h00–09h00) and 60.9% in the evening peak (18h00–21h00). In SFB houses, the trend was similar in summer with decreases of between 47.5% and 50.9% recorded in both peaks. This implies that on average, both household samples are exposed to almost half of the particulate loading in summer compared to winter. Since the bulk of the SFB occurs indoors and winter temperatures force residents to keep windows and doors closed, this decreases ventilation and the potential dispersion of PM_4. In turn this results in elevated concentrations (Language *et al.,* 2016).

This study can be considered to confirm that the households are exposed to elevated PM_4 concentrations for both diurnal and daily temporal scales. There is an increased likelihood of residents in this community suffering from negative health impacts since the respirable fraction of PM measured in this study is a health hazard associated with respiratory illnesses. This is due to the consequences of inhalation of PM_4 which can cause harm by being deposited in the nasal pharyngeal and laryngeal areas of the respiratory system (Kapwata *et al.,* 2018).

There are multiple household activities, characteristics and ambient sources of PM that contribute to the variability associated with indoor diurnal trends of PM_4 concentration (Shezi *et al.,* 2018). Airborne particulates are generated by a number of activities within the household. These include activities such as cooking, cleaning, sweeping, ventilation properties and household physical structures. All the sample houses had access to electricity and alternatively used other energy carriers depending on but not limited to preference and affordability. It has been well established that winter solid fuel use is high because of the duality of cooking and space heating, whereas such use is reduced in summer due to residents not needing to heat their homes. The number of people in the house (household size) also often affects the amount of cooking that takes place, as well as the duration of meal preparation. The contribution of SFB to household indoor airborne concentrations tends to vary when factoring in the amount of time the fuel is burnt to cook and/or heat the living space. Smaller families can end up consuming less fuel and thus suffer less exposure to particulate emissions compared to larger families dependent on solid fuels as the primary energy carrier. This explains statistically significant variations observed in SFB households despite the households falling under one category.

Housekeeping activities such as cleaning and sweeping have an impact on airborne particulate concentrations. However, housekeeping is typically done during the day and simultaneously with cooking which occurs in the early morning and afternoons; thus concentrations during the day (especially 11h00–16h00) are lower. Common to all the sampled houses, ventilation practices

include opening doors and windows during the day and keeping them closed at night. None of the sampled houses had installed air conditioning systems which would have impacted the traditional ventilation process. For instance, with air conditioning systems residents would have kept doors and windows closed for most parts of the day which in turn reduces the likelihood of outdoor particulates depositing inside the houses.

Housing structure and physical properties such as roof type, floor type, walls and insulation play an important role in determining indoor PM levels. About 65% of the sampled houses were not retrofitted with a ceiling thus increasing the infiltration of outdoor dust through cracks in the roof. Although most of the sampled houses were formal (either semi-detached or free-standing, the majority had a galvanised iron or fibre-cement roof type. These roof types are prone to increase more suspended airborne particles into the house than tiled roofs with ceiling.

The most common outdoor source most of the sample houses were situated close to was unpaved roads. Thus vehicle entrainment of dust and wind-blown dust are likely to have had an impact of the infiltration of dust from the ambient environment within the sampled houses. These would also have had an impact on the variability of concentration of PM depending on the interaction of all the factors at the time. For instance, the amount of traffic on the unpaved roads, meteorology and ventilation activities can work together to increase the rate of penetration of ambient dust particles to the indoor environment. In addition to unpaved roads, informal dumping sites and unauthorised waste burning were significant sources of PM within the area. The level of interaction between all the sources and household activities was not measured in this study; they are, however, mentioned as potential factors affecting the variability in indoor concentration of PM within the sample group observed in Table 13 and Table 14 .

4.3. Indoor Black carbon (BC) measurements

The use of solid fuels leads to elevated emissions of gaseous and particulate pollutants. As a by-product of incomplete combustion, Black Carbon (BC), among other pollutants, was measured at H07. Solid fuel combustion can generate significant volumes of PM emissions, including BC and brown carbon (BrC) (Xulu, 2017). BC containing aerosols can cause health impacts that are 5 times more dangerous to humans than inorganic particles (Smith *et al.,* 2009; Lilieveld *et al.,* 2015). Approximately, the health effects of a 1 µg/m^3 increase of BC were found to be higher than those of PM2.5 and PM10 which are likely exacerbated by the toxicity levels of BC and other hazardous compounds absorbed by BC particulates (Janssen *et al.,* 2011; Xulu, 2017). Therefore, human exposure to BC has grievous health implications and is important to measure for future health studies.

This is the first time that BC has been measured indoors in South Africa. The indoor measurements were taken in H07 for a period of 19 days (01–19 November 2017) during the

summer sampling campaign. Summary descriptive statistics of the daily BC concentrations for summer are provided in Table 15 below. The average concentrations throughout the summer were 3.60 ± 4.34 µg/m^3. Figure 30 provides the temporal variation of BC mass concentration. There are certain days when BC concentrations in household H07 were extremely elevated, reaching concentrations with a maximum of 107.11 µg/m^3, 59.52 µg/m^3 and 58.46 µg/m^3 on the 03, 11 and 15 November 2017 respectively.

Table 15: Descriptive statistics of daily BC concentrations in H07 for the summer sampling campaign.

	N	Mean	Median	Min	Max	10%	90%	Std.Dev.
BC	19	3.60	2.21	0.001	107.11	1.05	4.36	4.34

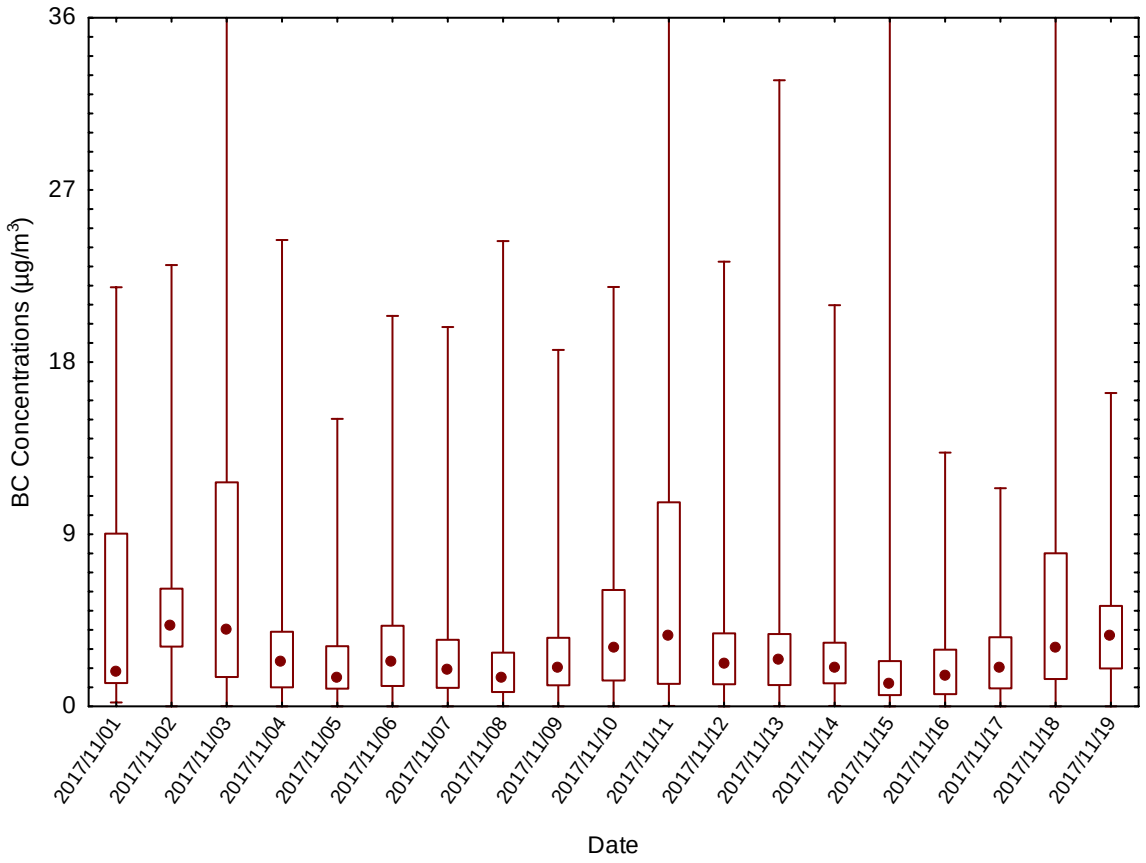

Figure 30: Daily average concentrations of indoor BC in H07 for the summer sampling period (Whiskers = Min & Max; Box = percentiles; Middle dot = Median).

The diurnal trend of indoor BC mass concentration showed a bimodal distribution with morning and evening peaks (Figure 31). Elevated concentrations occurred in the morning hours from 05h00– 08h00 in the morning and from 18h00 - 22h00 in the evening. The bimodal distribution suggests that BC concentrations were influenced by domestic burning of fossil fuel. Although a predominance in use of solid fuel use was observed in winter, for household H07, concentrations

of BC and PM_4 remained problematic for summer as well. In general, low-income households prefer the use of non-solid fuels or cleaner energy sources during summer (Naidoo *et al.,* 2014); based on personal observations during winter and summer 2017, this is also applicable for Sharpeville households. However, Figure 32 below shows a bimodal distribution of indoor BC and PM_4 mass concentrations and thus affirms the continued use of fossil fuels during summer in household H07. The ratio of BC is to PM_4 in H07 for the summer sampling campaign was <10% (7.5%). Nevertheless, it was found most households strongly preferred reduction in the use of solid fuels in summer.

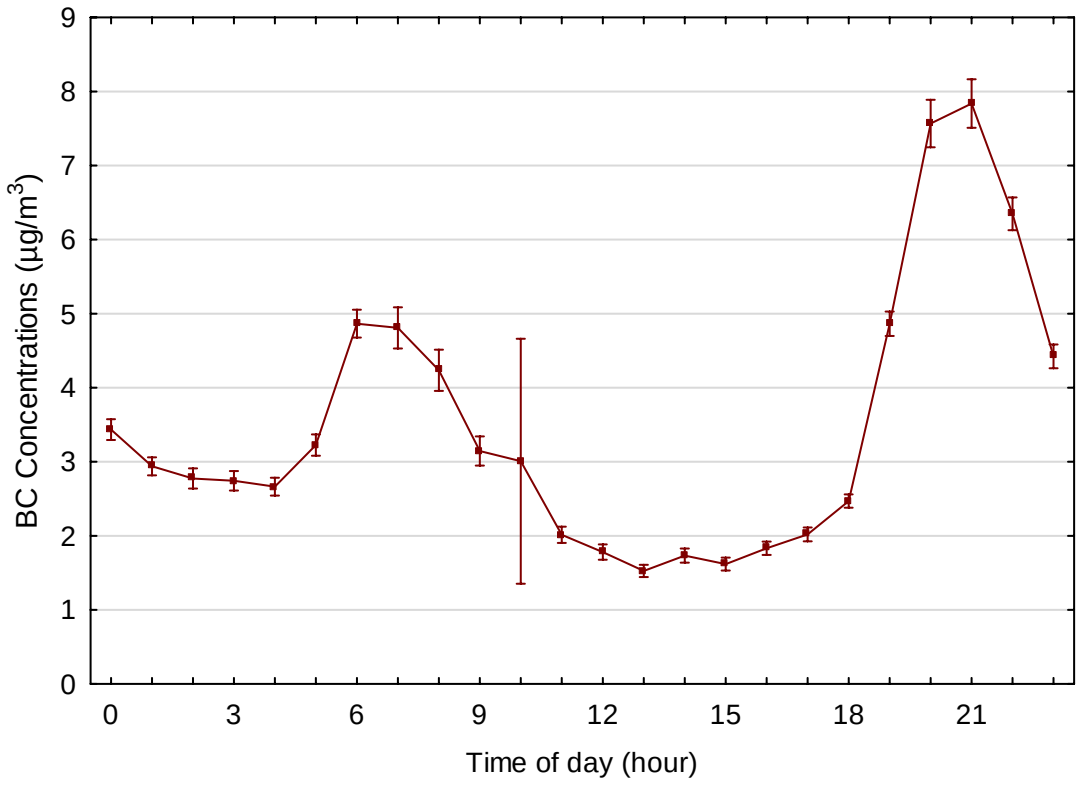

Figure 31: Diurnal concentrations of BC in H07 during the summer monitoring period.

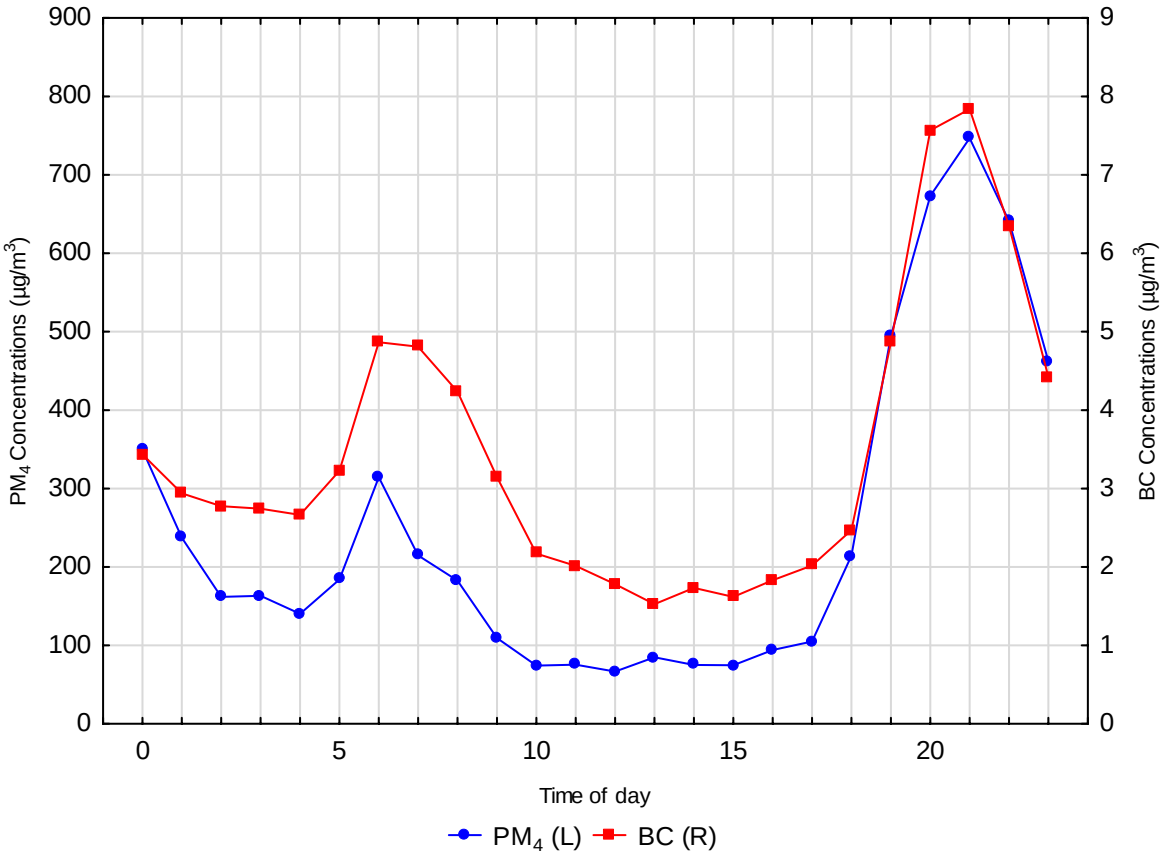

Figure 32: Diurnal plot for both BC and PM_4 in H07 for the summer sampling campaign.

4.4. Continuous indoor temperature

Indoor temperatures are one of the important drivers of domestic solid fuel use. Modern-day low-income houses in South Africa are often structurally inadequate to provide a thermally satisfactory environment for their inhabitants (Klunne, 2002). Most of the temperature sampling was done in 3 to 4 rooms in the house (kitchen, living room, main bedroom, secondary bedroom). Indoor temperatures were used in this study to evaluate the impact they have on solid fuel use and consumption, including the thermal efficiency of the house and the duration of burning which ultimately determines airborne PM_4 concentrations within low-income households in Sharpeville.

4.4.1. Summary descriptive statistics

During winter the outdoor temperatures ranged between 5°C and 23°C. Indoor temperatures were overall 5°C warmer than the ambient temperatures, while the maximum values exceeded 25°C in some houses (Table 16). The variation may be explained in terms of the structural difference of the houses where certain houses have efficient thermal properties (retrofitted with a ceiling and other material, see Table 3) and other houses were still as basic as when they were issued by the government under the RDP programme. Summer indoor temperatures varied between 21°C and 28°C and often reached a maximum of 31°C for certain houses (Table 17). Due to the warmer

ambient temperatures in summer, household combustion of solid and other energy sources had less impact on the temperature indoors.

Table 16: Descriptive statistics for hourly averaged temperature measurements for each household during winter 2017.

No.	Mean	Median	Min	Max	99th	SD	No.	Mean	Median	Min	Max	99th	SD
H01-B	15	15	11	20	20	3	**H10-B**	18	19	15	21	21	2
H01-K	15	15	11	20	20	3	**H10-L**	20	20	16	23	23	2
H01-E	13	12	5	21	21	6	**H10-b**	18	18	15	20	20	2
H02-B	19	19	15	23	23	3	**H10-K**	19	19	15	22	22	2
H02-K	19	19	15	22	22	2	**H11-K**	17	18	13	22	22	3
H03-B	21	22	20	23	23	1	**H11-L**	20	20	15	24	24	3
H03-K	16	16	15	18	18	1	**H11-b**	15	15	11	20	20	3
H03-E	13	12	6	22	22	6	**H11-B**	17	18	12	22	22	3
H04-B	18	18	15	21	21	2	**H12-K**	20	20	16	23	23	2
H04-K	20	20	16	24	24	3	**H12-L**	21	22	17	25	25	3
H04-E	14	13	6	23	23	6	**H12-b**	19	20	16	22	22	2
H05-B	16	16	12	21	21	3	**H12-B**	20	20	16	23	23	2
H05-L	17	17	12	22	22	4	**H13-B**	16	16	12	20	20	3
H05-K	18	18	12	25	25	4	**H13-b**	17	17	13	21	21	3
H06-B	20	19	14	26	26	4	**H13-L**	19	19	14	22	22	3
H06-L	21	22	16	25	25	3	**H13-K**	17	17	12	22	22	3
H06-K	21	21	15	25	25	3	**H14-B**	18	18	15	20	20	2
H07-B	18	18	12	24	24	4	**H14-b**	19	19	16	20	20	1
H07-L	17	17	12	23	23	4	**H14-L**	17	17	14	20	20	2
H07-K	17	17	12	21	21	3	**H14-K**	17	17	14	19	19	2
H08-B	18	18	15	22	22	2	**H15-L**	21	22	16	25	25	3
H08-K	17	17	14	19	19	2	**H15-b**	17	17	14	21	21	2
H08-E	13	12	5	21	21	5	**H15-B**	16	16	14	17	17	1
H09-B	18	18	15	22	22	2	**H15-K**	17	17	14	20	20	2
H09-b	19	19	15	22	22	2	**H16-B**	17	17	12	23	23	4
H09-L	17	17	14	20	20	2	**H16-L**	18	19	13	23	23	3
H09-K	19	19	16	23	23	2	**H16-K**	17	17	13	21	21	3
Average	18	18	14	22	22	3							

Table 17: Descriptive statistics for hourly averaged temperature measurements for each household during summer 2017.

No.	Mean	Median	Min	Max	99th %	SD	No.	Mean	Median	Min	Max	99th %	SD
H01-B	25	25	21	28	28	2	**H10-B**	26	26	23	28	28	2
H01-E	21	21	14	28	28	5	**H10-b**	26	26	24	28	28	1
H01-K	25	25	21	29	29	3	**H10-K**	26	26	23	28	28	2
H01-L	25	24	20	29	29	3	**H10-L**	26	26	23	28	28	2
H02-B	25	25	22	28	28	2	**H11-B**	25	25	20	29	29	3
H02-K	25	25	22	27	27	2	**H11-b**	24	24	20	28	28	3
H02-L	25	25	22	29	29	2	**H11-K**	25	25	21	28	28	3
H03-B	24	24	22	26	26	1	**H11-L**	25	26	22	28	28	2
H03-E	21	21	15	28	28	5	**H12-B**	25	26	22	28	28	2
H03-K	24	24	23	25	25	1	**H12-b**	25	26	22	28	28	2
H03-L	24	24	22	25	25	1	**H12-K**	25	25	22	28	28	2
H04-B	24	24	20	27	27	2	**H12-L**	26	26	22	29	29	2
H04-E	24	23	17	30	30	5	**H13-b**	25	25	21	29	29	3
H04-K	25	25	22	28	28	2	**H13-B**	26	26	22	29	29	2
H04-L	24	25	21	27	27	2	**H13-K**	25	25	20	28	28	3
H05-B	24	24	20	29	29	3	**H13-L**	26	26	21	30	30	3
H05-K	24	24	20	29	29	3	**H14-B**	24	24	22	26	26	1
H05-L	25	25	20	29	29	3	**H14-b**	25	25	23	26	26	1
H06-B	25	25	21	30	30	3	**H14-K**	24	24	22	26	26	1
H06-K	26	26	22	29	29	2	**H14-L**	23	24	21	25	25	2
H06-L	26	26	22	29	29	2	**H16-B**	25	25	20	30	30	3
H07-B	23	23	19	29	29	3	**H16-K**	25	25	21	29	29	3
H07-K	24	24	20	27	27	2	**H16-L**	25	25	20	30	30	3
H07-L	24	24	20	29	29	3	**H17-E**	20	19	14	29	29	5
H08-B	25	25	22	28	28	2	**H17-K**	25	25	19	31	31	4
H08-K	24	24	21	27	27	2	**H17-L**	25	25	21	28	28	2
H08-L	24	24	21	27	27	2							
H09-b	25	25	22	28	28	2	**L – Living Room**						
H09-B	25	25	22	29	29	2	**B – Main Bedroom**						
H09-K	26	26	23	28	28	2	**b – Secondary Bedroom**						
H09-L	25	25	22	29	29	2	**K – Kitchen**						
Average	25	25	21	28	28	2	**E - External**						

4.4.2. Diurnal indoor temperature profile

In winter, the temperature measurements in Sharpeville showed outdoor temperatures were below the WHO minimum (15 °C) value for 62% of the day (Figure 33 a). Temperatures started to increase from 10h00 in the morning and remained well below the WHO maximum (25 °C) value for approximately 8 hours (until 18h00). In some houses, temperatures were below the WHO minimum value (15 °C) for most of the morning (05h00–10h00) (Figure 33 b). Only a fraction of

the sampled houses exceeded the WHO maximum temperature value during winter, with the bulk of the houses remaining within the guidelines for daytime. The may be indicative of measures taken by homeowners to keep the houses warm, such as space heating using either wood and coal or cleaner alternatives like electricity.

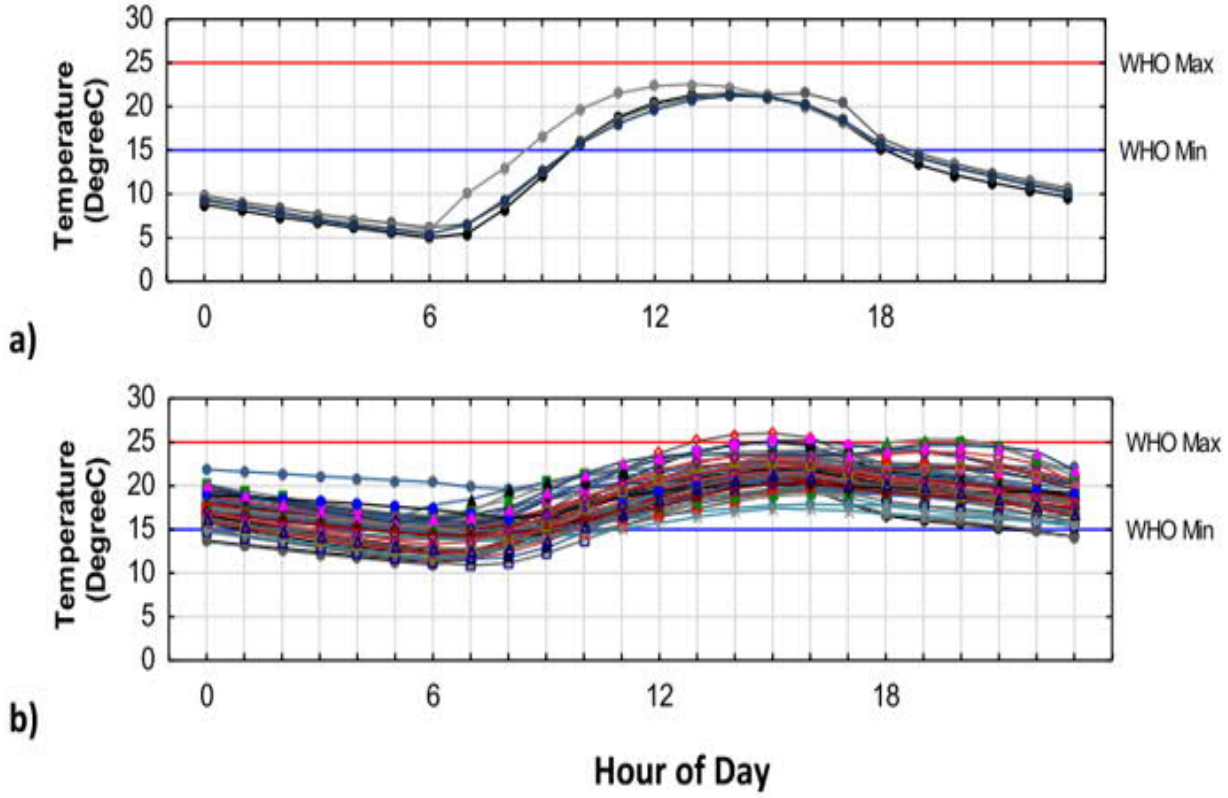

Figure 33: Diurnal pattern of hourly averaged a) ambient and b) indoor temperature measurements in Sharpeville for winter 2017.

During summer the temperature measurements show outdoor temperature exceeding the WHO maximum value from as early as around 10h00 in the morning and remaining high for approximately 7 hours (until 17h00) (Figure 34 a). The temperature in the ambient environment dropped below 15 C at three of the four sites at approximately 02h00 and remained just below this temperature until around 05h00. Indoor temperatures (Figure 34 b) during summer in all houses were in a very narrow band of values and ranged between 21 C and 28 C indoors. The minimum temperatures in summer were recorded between 06h00 and 07h00 and never dropped below 15°C. Indoor temperatures did not drop below the WHO minimum in summer; however some houses did exceed the recommended WHO maximum value. All sampled households' indoor temperatures routinely exceeded the WHO recommended maximum value of 25°C in summer between 13h00 and 17h00.

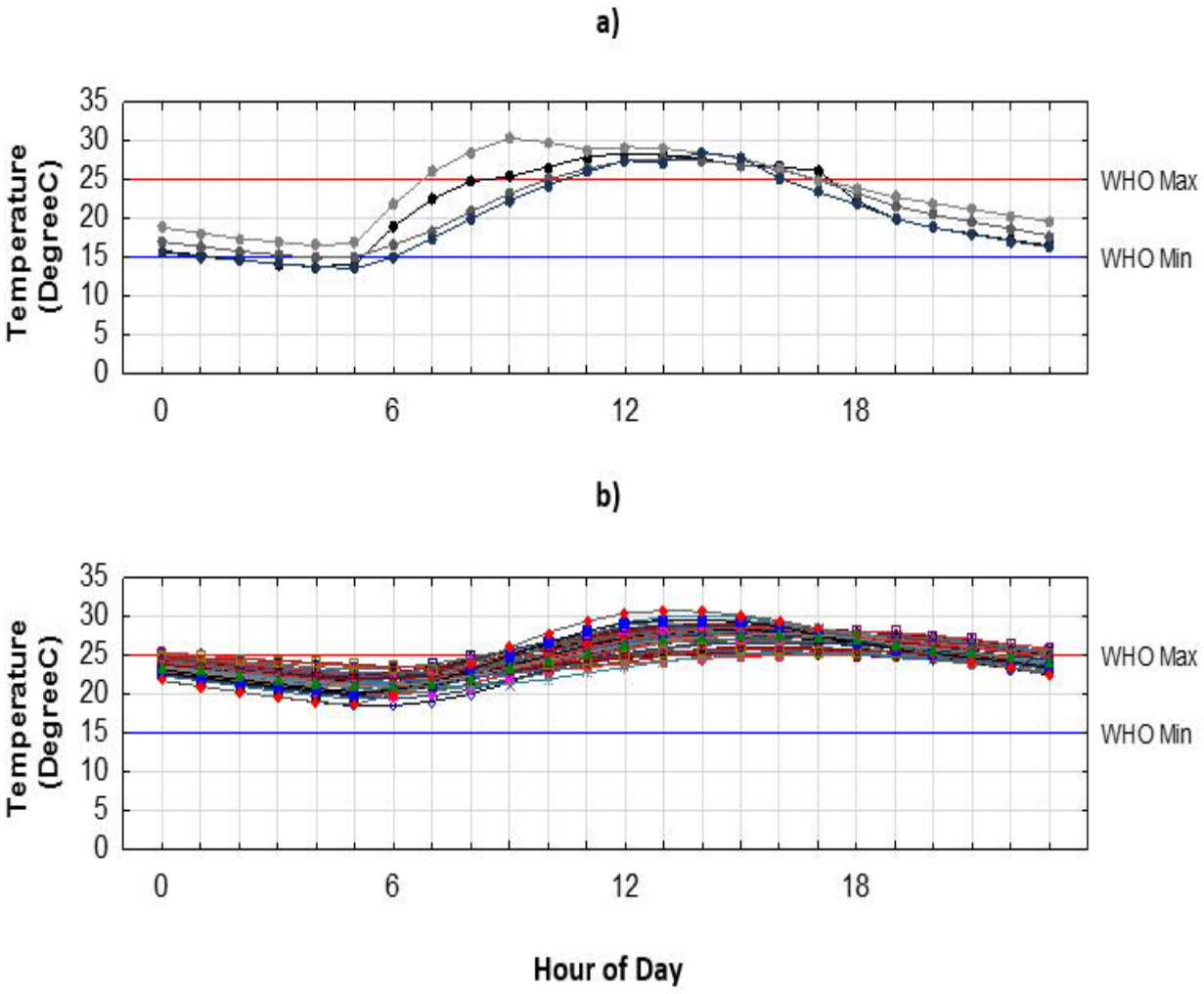

Figure 34: Diurnal pattern of hourly averaged a) ambient and b) indoor temperatures measured in Sharpeville for summer 2017.

CHAPTER 5: GRAVIMETRIC MASS CONCENTRATION AND ELEMENTAL COMPOSITION

CHAPTER OVERVIEW: This chapter represents the results of the collected gravimetric samples from two contrasting low-income houses in Sharpeville for both winter and summer 2017. The differences in gravimetric mass concentrations are evaluated and the elemental characteristics of indoor airborne PM_4 are presented.

5.1. Gravimetric mass concentrations

Gravimetric mass concentration measurements were conducted within two selected low-income households for both the winter and summer 2017 sampling campaigns. One of the criteria for selection was energy use, as the sampling required two households different in terms of primary and secondary energy sources for domestic activities. House No.05 (H05) used electricity for all their energy requirements (designated the 'control house'). House No. 07 (H07) used solid fuels (coal and wood) for cooking and space heating, while electricity was used for lighting (sampling house). A total of 25 and 24 valid samples were collected for H05 and H07 in winter, respectively. During summer, 34 valid samples were collected in H05 and 37 in H07.

Gravimetric mass concentration recorded in H07 was greater than that of H05 during winter, whereas in summer these houses showed a comparable concentration trend. Winter mass concentration in the NSFB household H05 (Table 18) was well below the 24hr $P_{M2.5}$ NAAQS with an average mass concentration of 32 (± 23.3) µg/m^3. Due to a cleaner residential energy use pattern (mainly based on electricity), it is safe to assume that in this house PM was characteristic of indoor airborne particulate pollution. Indoor activities such as household cleaning (inclusive of sweeping and dusting of furniture), movement inside the house and ventilation mechanisms would have mobilised PM deposits (Bramwell, 2013; Marawska *et al.,* 2013). Elevated concentrations can also be attributed to outdoor sources of particulates such as vehicle road dust entrainment. Within the SFB household, the winter sampling represented a case where indoor PM mass were higher than both $PM_{2.5}$ and PM_{10} 24hr NAAQS (85±100.3 µg/m^3). The household relied on sold fuels more often during this sampling period. Although domestic SFB cannot be the only source (this is explained in detail in the elemental section below), it is, however, the most significant contributor within H07.

Table 18: Gravimetric respirable PM mass concentration descriptive statistics (µg/m^3) showing the mean, median, standard deviation, minimum, maximum and percentiles (25th and 75th) in both non-fuel burning (H05) and fuel-burning (H07) households at Sharpeville for winter and summer 2017.

		N	***Mean***	***Median***	***Min***	***Max***	***25th%***	***75th%***	***Std.Dev.***
Winter	*H05*	25	32	21	3	77	12	55	23.3
	H07	24	85	53	1	479	45	85	100.3
Summer	*H05*	34	57	50	5	147	36	80	29.4
	H07	37	74	67	12	142	52	95	32.7

The only relevant difference between the H05 and H07 households was the observed increase in average mass concentration for the summer for the former (NFSB) house. This was higher than in winter by a factor of 1.8 representing constituting a 43.9% increase. The PM loading was recorded at 57 (± 29.4) µg/m^3 in H05 which is 42.5% greater than the 24hr average $PM_{2.5}$ NAAQS and a slight decrease in the SFB house. On the other hand, the summer results from the SFB house were lower than the winter levels by a factor of only 1.2 representing a much smaller 12.94% decrease between the campaigns compared to the NSFB house. The indoor mass concentrations were elevated and hazardous to the exposed parties in H07, since they fell only a single µg/m^3 lower [74(±32.7] than the 75 µg/m^3 PM_{10} 24rh NAAQS.

Time series representing gravimetric respirable PM loading in the two households for winter and summer are shown in Figures 35 and 36. For the winter campaign (26 July– 9 August 2017), 20 valid samples were gathered per household and are represented on the time series (Figure 35). Similarly, for the summer campaign (1–19 November 2017), 34 valid samples per household were collected and are represented on the time series (Figure 36). This was done in such a way that samples that match in time and date from the two houses were deemed representative of the temporal variation between the two sampling houses. The respirable size fraction mass concentrations from the SFB household was higher than that recorded in the NSFB household during winter. On average, the gravimetric mass concentrations of PM in the SFB house were higher than that in the NSFB house. However, the time series shows a trend whereby on certain sampling days, the NSFB household experienced higher PM loading than that for the SFB house.

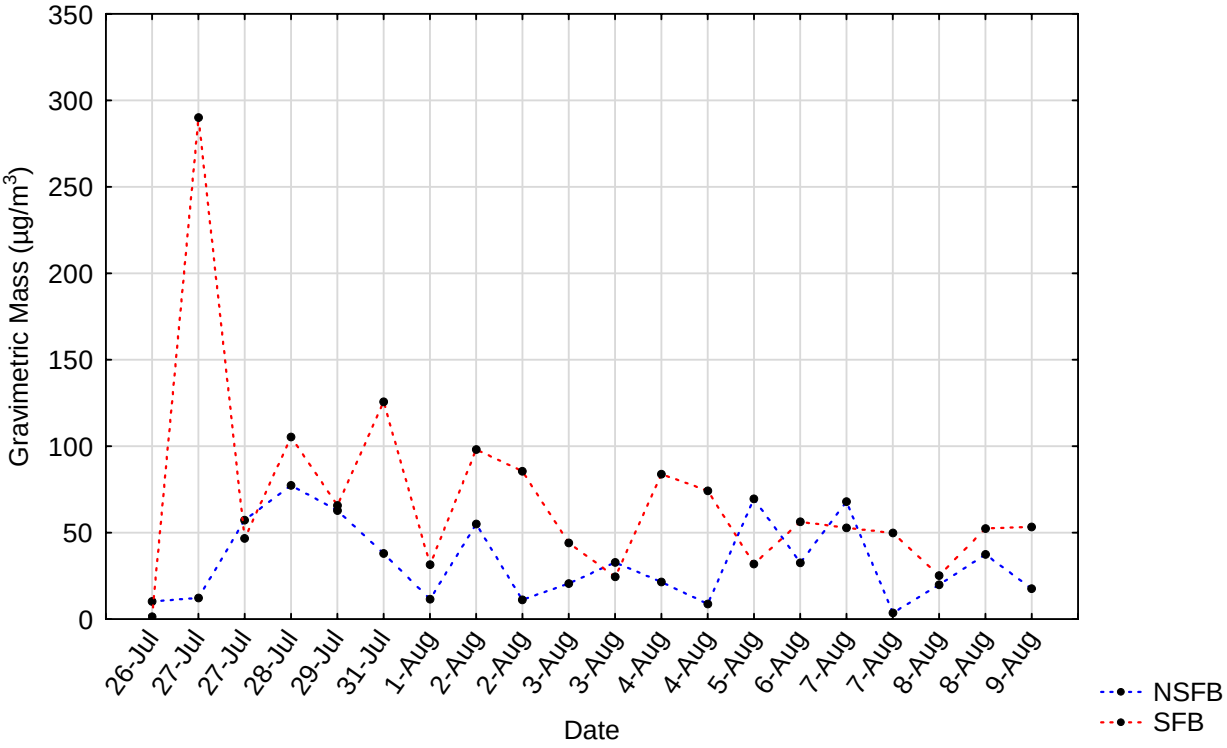

Figure 35: 12 hourly time series of indoor particulate mass concentration (µg/m^3) in both NSFB and SFB households for the winter period in Sharpeville 2017.

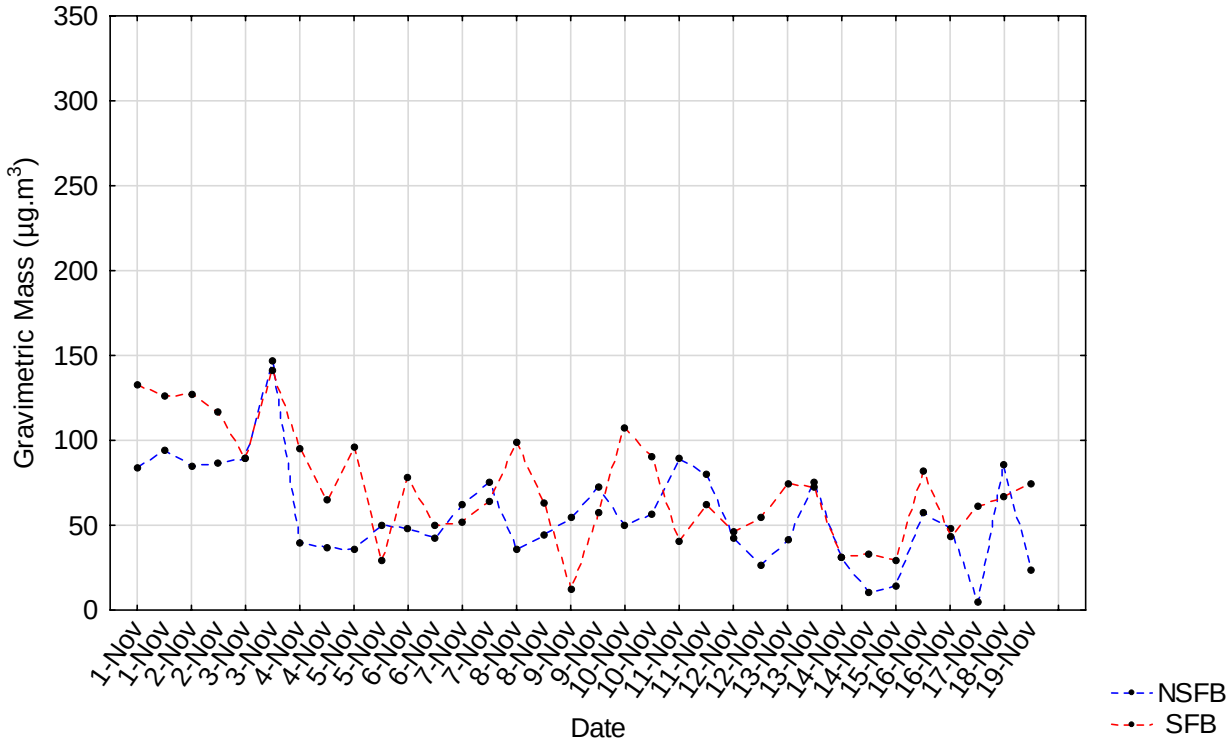

Figure 36: 12-hourly time series of indoor particulate mass concentration (µg/m^3) in both NSFB and SFB households for the summer period in Sharpeville 2017.

5.2. Elemental composition

Elemental characteristics of particulates are dependent on the contributing sources. Emissions from different sources consist of specific elemental species that are regarded as being unique (Annegarn *et al.,* 1987), and relative to the nature of the polluting source. Depending on—but not limited to—their geographic location, densely populated low-income settlements in South Africa have distinct sources of air pollution. Potential sources of PM in Sharpeville include fugitive dust, domestic solid fuel burning, waste burning (mainly informal), power generation, vehicle emissions and biomass burning (localised veldfires). Filter sampling was conducted in two sample houses in Sharpeville for the purpose of evaluating elemental composition from indoor air pollution as a function of solid fuel use. The selected houses were characterised as a solid fuel burning (sampling house) and a non-solid fuel burning (control house). The sampling was conducted for both winter and summer 2017.

Fugitive dust is mainly made up of topsoil and road dust of which the elemental species composition includes mainly Al, Si, Ti, Fe, and Ca (Watson *et al.,* 2002). Domestic solid fuel combustion of coal and wood is primarily prone to emitting particles composed with S and K (Wang *et al.,* 2006). These element are generally found within indoor PM due to the domestic combustion of coal and wood (or other biomass products). Typical emissions from the use of alternative energy carriers for domestic purposes include the release of CO, fine aerosol particles, PAH, SO_2 and NO_2 (Walton *et al.,* 2013). On the other hand, the abundance of low-grade coal which is normally used in low-income settlements because it is cheap, is infamous for the emission of heavy metals such as uranium (U) (Nuwarinda, 2007) and other chemical components such as iron sulphide (FeS) and arsenic sulphide (AsS) (Walton *et al.,* 2013). Informal waste burning is a significant contributing source of PM in dense low-income settlements.

Waste incineration is supposed to be undertaken in an environment where control measures include burning of the waste at high temperatures to minimise impacts. Informal waste burning is problematic because the waste is often burnt at lower temperatures than those required to minimise hazardous emissions. Annegarn *et al.* (1998) articulated that pollutants emitted from informal refuse combustion are highly toxic and a significant health hazard. Particles released from waste burning at a lower temperature contain a broad spectrum of chemical components including ash, toxins and VOCs (Li *et al.,* 2012). Most of the general waste material burnt is made up of a mixture of organic and inorganic substances with the by-products contain biomass constituents such as Zn, K, C and Br. Vehicle emissions contribute substantially to ambient $PM_{2.5}$ with elevated concentrations of BC, Mg, Si, Fe, S and Al (Kim *et al.,* 2003; Begum *et al.,* 2004). The R28 roadway passing close to Sharpeville is a busy route and it is important to note this as that the source of the majority of vehicular emissions. Power stations are considered to be major contributors to poor air quality in South Africa. Eskom Lethabo is a power generation facility

located ± 15 km away from Sharpeville, with both BC and S are typically the dominant emissions from coal combustion. The above-mentioned sources all consist of distinct chemical and elemental signatures which are useful for the analysis of source contribution estimates.

The elemental mean concentrations (and standard deviation) for day and night in summer and winter (2017) for the respirable size fraction at the Sharpeville are presented for both the NSFB and SFB households. Summary descriptive statistics for the NSFB and SFB households' elemental composition for both winter and summer are provided in Tables 19 and 20.

5.2.1. Non-Solid Fuel Burning Household (H05)

5.2.1.1. Dust-related elements

Fugitive dust is a significant source of PM in Sharpeville mostly due to the unpaved roads. The NSFB sample house was located next to an unpaved road. The dust related elements focused on include Al, Si, Ca, Fe and Mg of which accounted for 87.75% and 29.66% of the total gravimetric mass concentration in winter and summer respectively and earmarking a 66.21% decrease. This shows dust was a major contributing source to the indoor air at H05 during winter. ***Aluminium (Al)*** levels were higher for indoor particulates in winter compared to summer with the mean day-time (winter: 5.27 ± 2.31 µg.m^3; summer: 4.24 ± 2.54 µg.m^3) and night-time (winter: 3.56 ± 2.59 µg.m^3; summer: 3.47 ± 2.21 µg.m^3) (Table 19). Although it is true that winter values were higher than summer values, the variability between the two seasons in terms of Al concentration within indoor particles was low.

In contrast, ***silicon (Si)*** showed significantly higher winter than summer concentrations, particularly for day-time levels. The mean day-time (winter: 14.39 ± 6.36 µg.m^3; summer: 11.04 ± 6.28 µg.m^3) and night-time (winter: 9.86 ± 7.25 µg.m^3; summer: 9.25 ± 5.75 µg.m^{-3}). Day-time winter concentrations were 23.3% higher than day-time summer and 6.2% during the night. ***Calcium (Ca)*** concentration were also higher in winter than in summer by a factor of 2. Average day-time Ca concentration were (winter: 5.10 ± 2.70 µg.m^3, summer: 2.29 ± 1.52 µg.m^3) and night-time (winter: 4.52 ± 3.14 µg.m^3, summer: 1.93 ± 1.56 µg.m^3). ***Iron (Fe)*** was measured at highly concentrated winter day-time levels (13.08 ± 5.61 µg.m^3), a factor of 8 times higher than summer day-time concentrations. Night time concentrations were relatively constant for both seasons (winter: 1.79 ± 1.09 µg.m^3; summer: 1.33 ± 0.76 µg.m^3). ***Magnesium (Mg)*** was also highly concentrated during winter compared to summer, with the mean day-time (winter: 1.52 ± 0.68 µg.m^{-3}; summer: 1.07 ± 0.55 µg.m^{-3}) and night-time (winter: 1.25 ± 0.80 µg.m^{-3}; summer: 0.96 ± 0.47 µg.m^{-3}) concentrations 26.7% lower during the summer compared to winter night-time levels.

The latter elements are the most prominent topsoil elements and thus the focus when discussing dust-related elements. These trace elements were also converted to their mineral form (Al_2O_3,

SiO_2, CaO, FeO and MgO), and they accounted for >95% (97.65%) of the total mass in winter and 72.69% in summer. The element Si and mineral SiO_2 seems to be the highest of all the focal dust elements accounting for 40.21% and 41.26% of winter and summer concentrations, respectively.

5.2.1.2. Coal and biomass-related combustion elements

The following elements are associated with combustion processes in the area. For the purpose of this research, only three elemental species are central, Phosphorus (P), Sulphur (S) and Potassium (K). The combined mass of these elements account for 13.69% and 3.83% of the total mass for winter and summer respectively. The results indicates increased combustion of solid fuel material within the community. Combustion trace elements measured within H05 are associated mainly with ambient pollution from combustion sources such domestic fuel burning in the community.

On average, ***phosphorus (P)*** loadings were very low, with day- and night-time concentrations during winter (0.11 ± 0.06 µg.m^3) and summer (0.08 ± 0.5 µg.m^3) showing low significant differences across seasons. ***Sulphur (S)*** was routinely detected at levels higher than the MDL. During winter S concentrations were twice as high as in the summer samples with day-time concentrations (winter: 2.18 ± 0.98 µg.m^3; summer: 1.09 ± 0.51 µg.m^3); and night-concentrations (winter: 1.70 ± 0.99 µg.m^{-3}; summer: 1.19 ± 0.43 µg.m^{-3}). ***Potassium (K)*** is a prominent trace element for biomass burning which in Sharpeville includes domestic wood burning and communal informal waste combustion. Mean concentrations for day-time (winter: 1.08 ± 0.48 µg.m^{-3}; summer: 0.86 ± 0.43 µg.m^{-3}) and night-time (winter: 1.07 ± 0.56 µg.m^{-3}; summer: 0.88 ± 0.56 µg.m^{-3}) can be almost completely attributed to the latter sources.

5.2.1.3. Other sources

Other sources central to the discussion are limited to vehicle emissions considering the prominent impact they have on the air quality in the VTAPA region. Two most associated elements are taken in to account for the purpose of this research, namely lead (Pb) and zinc (Zn). The latter elements are eminent tracers for vehicle exhaust emissions (Huang *et al.,* 1994; Monaci & Bargagli, 1997). Both trace elements were found higher in winter than in summer. Winter Pb and Zn total mass accounted for were 10.63% and 1.08% whereas summer mass concentrations were 2.48% and 0.89%, respectively. Average ***lead (Pb)*** concentrations for day-time (winter: 3.39 ± 0.45 µg.m^3, summer: 0.31 ± 0.13 µg.m^3) and night-time (winter: 2.56 ± 0.45 µg.m^3, summer: 0.47 ± 0.41 µg.m^3) were measured. ***Zinc (Zn)*** was not detected in any of the day-time samples during winter with night-time concentrations showing 0.28 ± 0.35 µg.m^3. Summer day-time and night-time concentrations were 0.32 ± 0.41 µg.m^{-3} and 0.51 ± 0.94 µg.m^{-3} respectively. The decrease in

concentrations is attributed to the elevated dispersion of air during summer and increased household ventilation activities.

Table 19: Mean indoor concentrations and standard deviation in µg.m^{-3} of selected elements during summer and winter 2018 in the NSFB household.

Season		**Winter**		**Summer**	
Variable		**Mean**	**±SD**	**Mean**	**±SD**
Na	*Day*	6.02	2.71	2.71	1.47
	Night	3.80	2.77	3.28	1.50
Mg	*Day*	1.52	0.68	1.07	0.55
	Night	1.25	0.80	0.96	0.47
Al	*Day*	5.27	2.31	4.24	2.54
	Night	3.56	2.59	3.47	2.21
Si	*Day*	14.39	6.36	11.04	6.28
	Night	9.86	7.25	9.25	5.75
P	*Day*	0.11	0.06	0.08	0.05
	Night	0.11	0.06	0.07	0.05
S	*Day*	2.18	0.98	1.09	0.51
	Night	1.70	0.99	1.19	0.43
Cl	*Day*	1.05	0.37	0.29	0.25
	Night	1.01	0.40	0.39	0.35
K	*Day*	1.08	0.48	0.86	0.43
	Night	1.07	0.56	0.88	0.56
Ca	*Day*	5.10	2.70	2.29	1.52
	Night	4.52	3.14	1.93	1.56
Ti	*Day*	0.10	0.10	0.20	0.12
	Night	0.19	-	0.20	0.12
V	*Day*	0.05	0.03	0.06	0.03
	Night	0.06	0.02	0.07	0.04
Cr	*Day*	1.98	6.56	0.06	0.04
	Night	0.11	0.04	0.05	0.04
Mn	*Day*	0.10	0.07	0.10	0.07
	Night	0.09	0.05	0.14	0.06
Fe	*Day*	13.08	5.61	1.62	0.88
	Night	1.79	1.09	1.33	0.76
Ni	*Day*	-	2.87	0.05	0.04
	Night	-	-	0.04	0.02
Cu	*Day*	-	-	0.04	0.04
	Night	-	-	0.09	0.04
Zn	*Day*	-	-	0.32	0.41
	Night	0.28	0.35	0.51	0.94
Pb	*Day*	3.39	0.45	0.31	0.13
	Night	2.56	0.45	0.47	0.40

The median day/night (D/N) ratios were used to evaluate if sources changed or were consistent during the day-time and night-time samples collected at Sharpeville. The D/N ratio of the median

concentration for each element is given in Figure 37. Elemental species associated with dust (Al, Ca, Si, and Fe) have concentration ratios >1 in summer, indicating high episodes of dust during the day in summer. These elements were <1 in winter which showed higher dust episodes during the night. The significance is that for Al and Si the day-time ratio changes indicates that the dust source is more prevalent in the day. Wood and coal combustion elements S, Cl and K showed a different D/N behaviour where Cl and S were prevalent during the evening in both seasons, and K had higher episodes (>1) during the day in summer and high (<1) in the evening during winter.

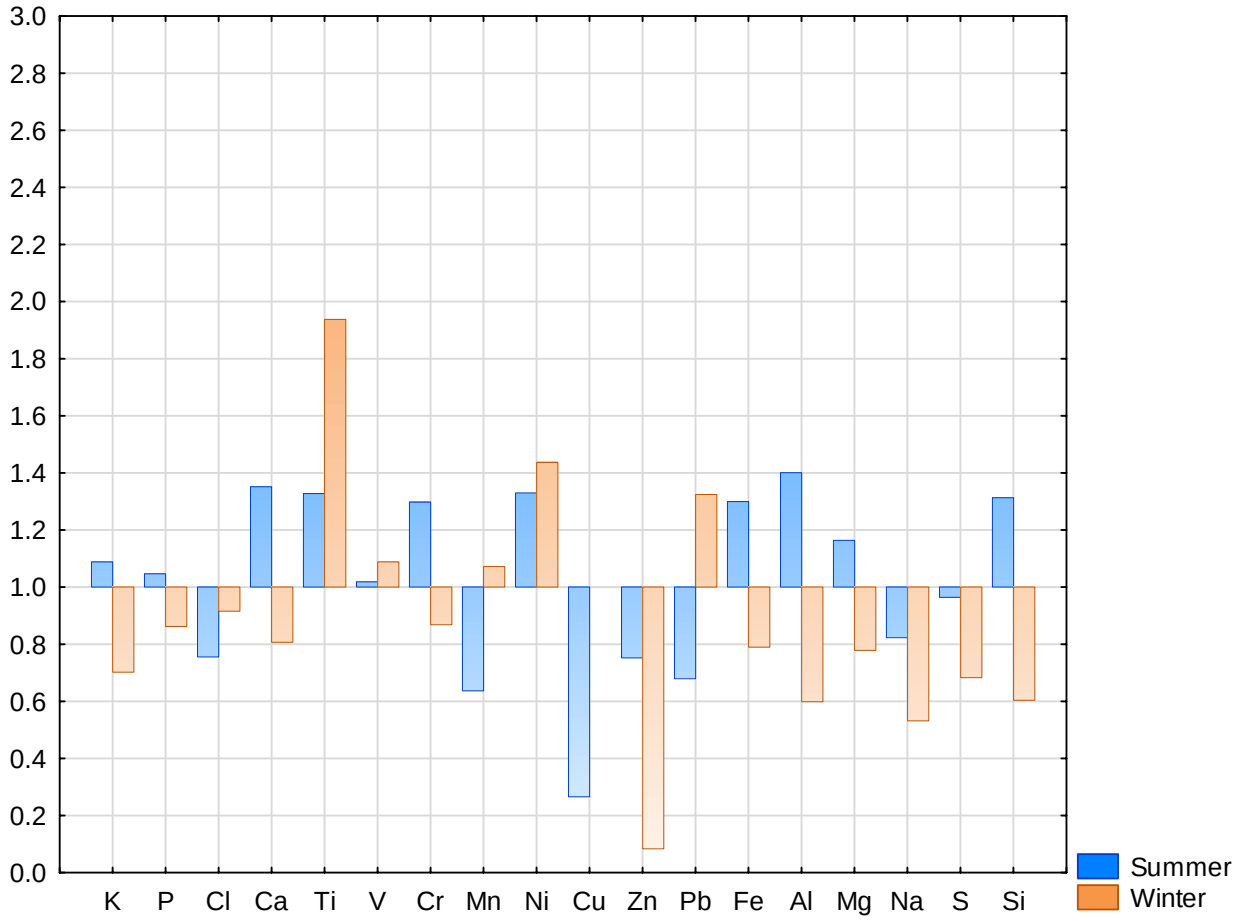

Figure 37: Ratio of median atmospheric concentration for day-time periods compared to night time concentrations for aerosol species in the NSFB household during summer and winter 2017 in Sharpeville.

5.2.2. Solid Fuel Burning Household (H07)

5.2.2.1. Dust-related elements

Dust elements combined (Al, Si, Ca, Fe and Mg) for winter in H07 accounted for 17.95% of the total mass and 42.62% in their mineral form (Al_2O_3, SiO_2, CaO, FeO and MgO). Summer total mass from the latter were as follows: 14.07% and 33.08% respectively. ***Aluminium (Al)*** winter concentrations were high compared to the levels in summer as follows: mean day-time

concentrations (winter: 3.58±1.24 µg.m^3; summer: 2.92±1.83 µg.m^3); and night-time levels (winter: 2.68±1.41 µg.m^{-3}; summer: 1.58±0.70 µg.m^{-3}). ***Silicon (Si)*** also showed higher winter than summer, although the concentrations were greater than for Al. The concentrations were as follows: mean day-time concentrations (winter: 9.45±3.28 µg.m^3; summer: 7.70±4.69 µg.m^3); and night-time levels (winter: 6.58±4.05 µg.m^3; summer: 4.08±1.73 µg.m^3). Day-time winter concentrations of Si were a factor of 1.2 times higher than day-time summer levels, and 1.6 times those measured during the night. The ***Calcium (Ca)*** concentration trend was similar to Si and Al in that it was also higher in winter than in summer. However, it is noteworthy that concentrations were five times higher at night in winter compared to summer. ***Iron (Fe)*** was determined to be highly concentrated in day-time samples during (winter: 1.55±0.65 µg.m^3; summer: 2.41±4.60 µg.m^3). Night time concentrations were relatively stable for both seasons (winter: 1.53±0.93 µg.m^3; summer: 0.76±0.45 µg.m^3). ***Magnesium (Mg)*** concentrations were high during winter compared to summer, as follows: mean day-time (winter: 0.98±0.35 µg.m^3; summer: 0.84±0.39 µg.m^3); and night-time (winter: 1.25±0.80 µg.m^3; summer: 0.96 ± 0.47 µg.m^3).

5.2.2.2. Coal and biomass combustion related elements

Combustion tracer elements central to the discussion (P, S, and K) accounted for a combined total mass of 17.75% and 14.46% for both winter and summer respectively. Average ***phosphorus (P)*** loadings were very low (<1 µg.m^3), day and night-time concentration for winter were 0.14 ± 0.06 µg.m^3 and 0.20 ± 0.10 µg.m^3, summer measured 0.11±0.06 µg.m^3 and 0.09±0.5 µg.m^3. The concentrations differed between seasons but significantly low. ***Sulphur (S)*** had double the day-time concentrations in winter compared to summer. Concentrations were as follows: day-time levels (winter: 2.32±0.99 µg.m^3; summer: 1.13±0.61 µg.m^3); and night-time levels (winter: 2.00±0.96 µg.m^3; summer: 1.17±0.54 µg.m^3). ***Potassium (K)*** is a biomass-linked element and was detected to have lower day-time mean concentrations (winter: 3.66 ± 1.51 µg.m^3; summer: 2.26 ± 1.11 µg.m^3) compared to night-time levels (winter: 4.75 ± 2.78 µg.m^3; summer: 2.38 ± 1.32 µg.m^3). Both S and K were found elevated in winter compared to summer which indicates combustion activities within H07.

5.2.2.3. Other sources

Vehicle emissions related elements Pb and Zn were also taken into consideration in H07. They accounted for a total mass of 4.1% and 2.69% for winter and summer respectively. ***Lead (Pb)*** average day- and night-time concentrations are as follows: winter 3.43 ± 1.25 µg.m^3 and 2.49 ± 1.16 µg.m^3; summer 1.83 ± 0.31 µg.m^3 and 1.60 ± 0.25 µg.m^3. ***Zinc (Zn)*** day-time (winter: 0.14 ± 0.10 µg.m^{-3}; summer: 0.32 ± 0.36 µg.m^{-3}) and night-time (winter: 0.24±0.19 µg.m^{-3}; summer: 0.39 ± 0.42 µg.m^{-3}) concentrations showed less significant concentrations compared to Pb.

Table 20: Mean indoor concentrations and standard deviation in µg.m^{-3} of selected elements during summer and winter 2018 in the SFB household.

Season		**Winter**		**Summer**	
Variable		**Mean**	**±SD**	**Mean**	**±SD**
Na	*Day*	2.69	1.06	1.72	1.16
	Night	1.86	0.66	1.63	1.10
Mg	*Day*	0.98	0.35	0.84	0.39
	Night	0.86	0.43	0.55	0.21
Al	*Day*	3.58	1.24	2.92	1.83
	Night	2.68	1.41	1.58	0.70
Si	*Day*	9.45	3.28	7.70	4.69
	Night	6.56	4.05	4.06	1.73
P	*Day*	0.14	0.06	0.11	0.06
	Night	0.20	0.10	0.09	0.05
S	*Day*	2.32	0.99	1.13	0.61
	Night	2.00	0.96	1.17	0.54
Cl	*Day*	1.75	0.47	0.86	0.88
	Night	1.94	0.93	0.81	1.17
K	*Day*	3.66	1.51	2.26	1.11
	Night	4.75	2.78	2.38	1.32
Ca	*Day*	2.50	1.52	2.00	1.31
	Night	2.38	1.07	0.51	0.20
Ti	*Day*	-	-	0.14	0.10
	Night	-	-	0.09	0.04
V	*Day*	0.06	0.03	0.08	0.04
	Night	0.04	0.02	0.05	0.03
Cr	*Day*	0.11	0.04	0.48	1.67
	Night	0.12	0.05	0.04	0.03
Mn	*Day*	0.08	0.05	0.10	0.07
	Night	0.08	0.04	0.10	0.06
Fe	*Day*	1.55	0.65	2.41	4.60
	Night	1.53	0.93	0.76	0.45
Ni	*Day*	-	-	0.38	0.80
	Night	-	-	0.04	0.03
Cu	*Day*	-	-	0.04	0.03
	Night	-	-	0.03	0.03
Zn	*Day*	0.14	0.10	0.32	0.36
	Night	0.24	0.19	0.39	0.42
Pb	*Day*	3.43	1.25	1.83	0.31
	Night	2.49	1.16	1.60	0.25

The median day/night (D/N) ratios were used to evaluate whether sources (dust, combustion and vehicle emissions) changed or were consistent for the day-time and night-time samples collected at Sharpeville. The D/N ratio of the median concentration for each element is given in Figure 40. In summer, ratios <1 were observed for the following elements: K, Mn, Ni, Cu and Zn but were not detected during winter. Significant loadings of the crustal source elements Ca, Fe, Al, Mg, Ti,

V, and Si were detected in both seasons. These are soil-related elements. Combustion-associated elemental species S, K, and Cl were detected for both winter and summer with concentrations ratios of 0.83:1.03, 0.95:1.07 and 1.17:1.12 respectively. S levels were high during the winter evenings with a D/N ratio < 1 indicating most of the indoor burning took place during the evening. Soil dust elements (Al, Na, and Si) had D/N ratios below one during winter, indicating higher dust episodes during the evening, while during the summer the D/N ratios were above one indicating higher day-time dust episodes. Ca levels were > 1 for both seasons which indicates a dominant source of elemental Ca is prevalent during the day. This also applies to the following elements: Cl; P; Fe; and Mg. The ability of PM_4 to spread past the terminal bronchioles and thus be deposited in the gas-exchange parts of the lungs (Kapwata *et al.,* 2018) while these parts contain transition metals exposes householders to increased ROS which cause airway injuries and inflammation (Knaapen *et al.,* 2002; Gonzales-Flecha, 2004).

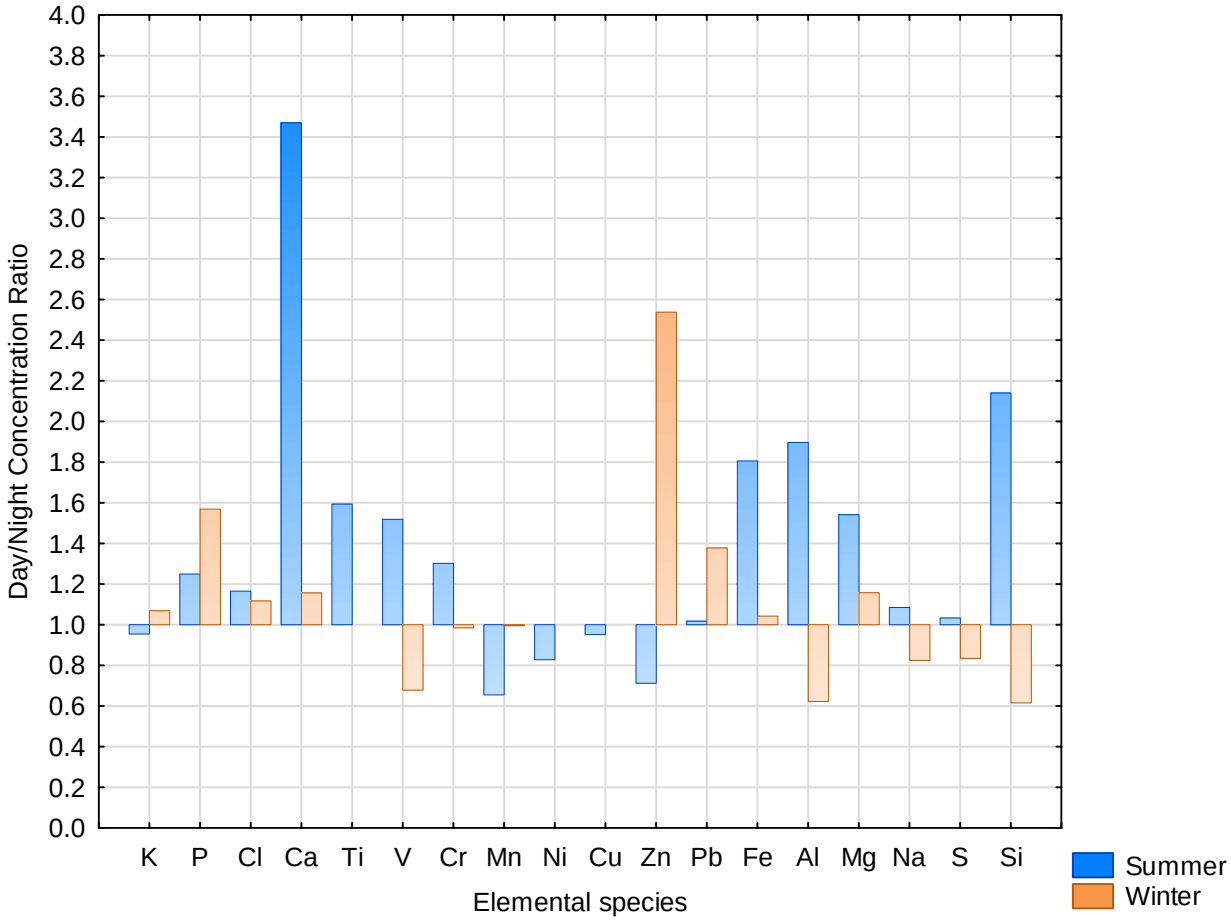

Figure 38: Ratio of median atmospheric concentration for day-time to night-time for aerosol species in the SFB household during summer and winter 2017 in Sharpeville.

The dust elements Al, Si, Ca, and Fe are much higher in SFB houses than NSFB houses during the day. Crustal sources occur mainly during the day in the SFB house whereas dust deposits are more predominant at night in the NSFB samples. In contrast to SFB houses, Ca in NSFB

samples had a day-time night-time source distribution. Sources of K and S, on the other hand, showed a reversed pattern in the SFB samples. S is much higher during the night in winter and is prevalent during the day in summer, whereas K sources are apparent in the day-time in winter and night-time in summer. The NSFB house differs in that sources of S appear dominant during the night in both seasons, while K is prevalent throughout the day in summer and during the night in winter.

5.3. Enrichment factors

The strength of the crustal and non-crustal sources was evaluated by calculating the enrichment factor (EF) for the elements quantified in this study. The EF provides an estimation of the amount an element is in excess compared to the concentration of the reference element. The EF was calculated using Al as a reference element in accordance to Mason's (1966) crustal chemical composition (Wang *et al.,* 2006). The EF is the ratio of an element of choice to that of Al in the air and in the average crustal material, respectively (Wang *et al.,* 2006). Day- and night-time enrichment factors for non-solid fuel burning (NSFB) and solid fuel burning (SFB) households are provided in Figures 39 and 40.

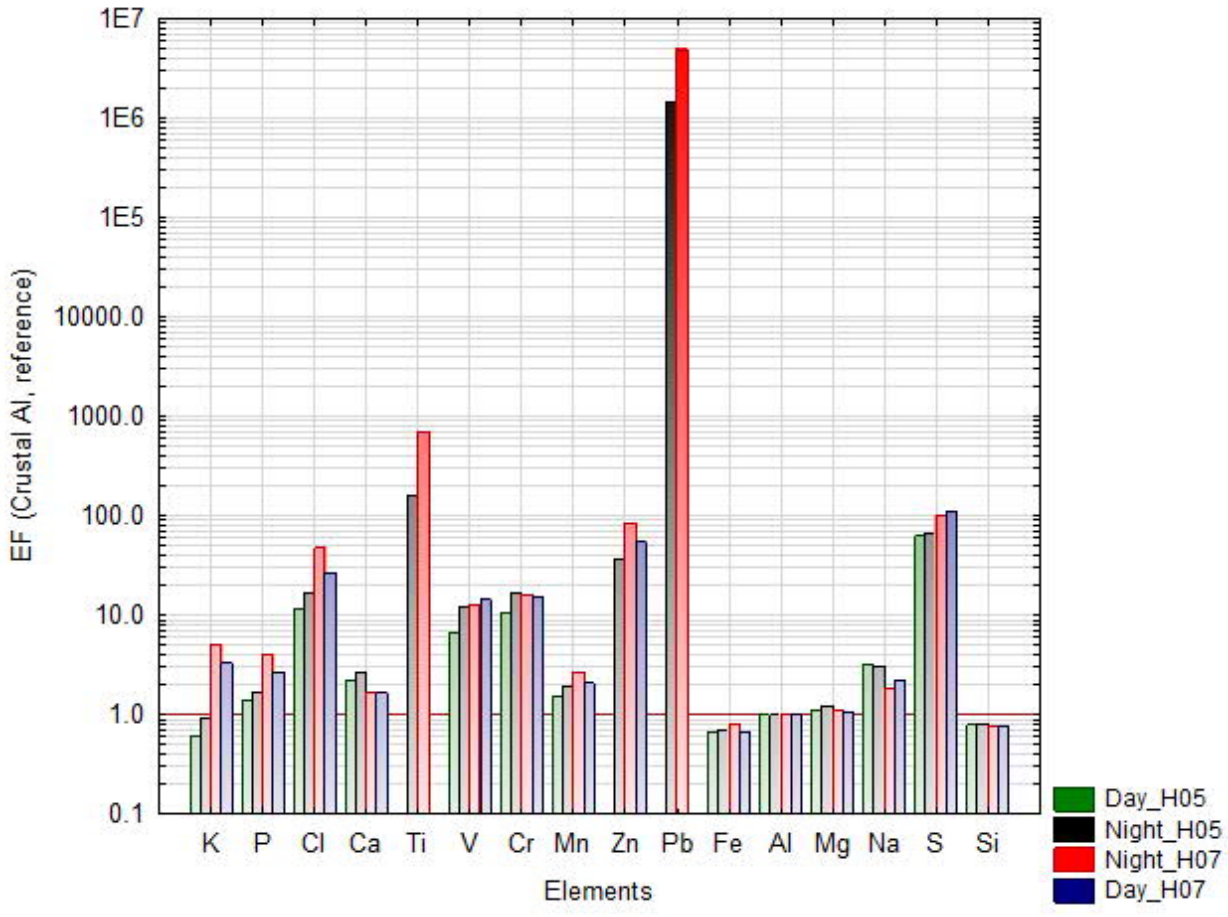

Figure 39: Day- and Night-time EF's for both household types during winter 2017 (Crustal Al used as the base element).

Enrichment factors are important as they can indicate a contribution to key signature elements other than soil dust. In the graph shown in Figure 39 and Figure 40, anything above 1 is enriched. In winter (Figure 39), key soil elements iron (Fe), Sodium (Na) and silicon (Si) showed no enrichment (<1) whereas Manganese (Mn) and Magnesium (Mg) had low enrichment. Potassium (K) indicated no enrichment in the NSFB sample house. In contrast, the SFB household showed elevated K enrichment which signifies indoor burning activity. Phosphorus (P) enrichment was higher in SFB compared to the NSFB sample house. Both NSFB and SFB houses showed high enrichment levels of Sulphur (S) in winter by a factor of a 100. For the NSFB household, this indicates ambient air pollution contribution within the household from residential burning in the community and industrial sources (mainly power generation). In comparison, the SFB house elevated S enrichment was a function of predominantly indoor solid fuel burning practices more than outdoor contributions. Lead (Pb) and Zinc (Zn) also were highly enriched in the SFB house. Pb showed enrichment by factors greater than a million and Zn almost a 100. Both Pb and Zn indicate the impact of vehicle exhaust emissions on the indoor environment.

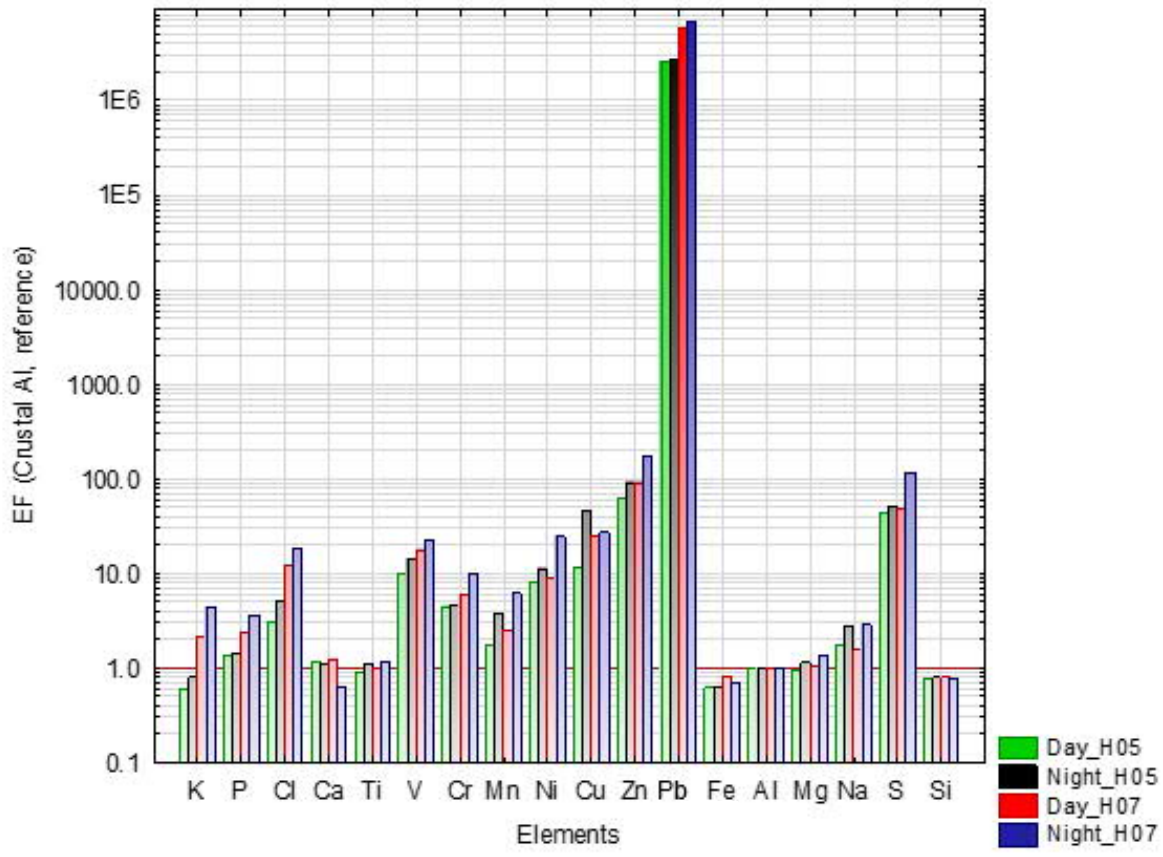

Figure 40: Day- and Night-time EF's for both household types during summer 2017 (Crustal Al used as the base element).

Summer elemental enrichment levels show a similar trend to winter patterns. Dust related elements show low to no enrichment in both household types. Combustion elements vary from

NSFB to SFB sample house, whereby K show no enrichment in the NSFB compared to the SFB house. P shows low enrichment in the NSFB compared to the moderate enrichment level in the SFB samples. S indicated elevated enrichment in both houses by a factor of almost 100 which, in the NSFB it was mainly the impact from ambient air pollution whereas solid fuel combustion was the primary contributor in the SFB sample house. Both households showed elevated signature of enriched Pb and Zn. This can be attributed to vehicular exhaust emissions during summer as Pb and Zn are known signatures of traffic emissions. Emissions from congested traffic, depending on the local weather and ventilations of houses, occur as a function of deposition and infiltration of outdoor particles within residences. Generally, summer temperatures are high and calls for household ventilation, which in most low-income houses, is done through the opening of windows and doors thus the ability of ambient polluted air to travel inside the houses. Therefore, the likelihood of traffic related emissions to occur within households in Sharpeville was plausible.

5.4. Source contribution to indoor airborne PM loading

5.4.1. Source contribution analysis using principal component analysis (PCA)

Receptor models infer contributions from different source types using multivariate measurements taken at one or more receptor locations. Once the samples had been analysed, a statistical receptor model was then applied to the aerosol data for source apportionment purposes. Receptor modelling is used to characterise particulate air pollutant sources and calculate each source contribution to a particular pollutant. The PCA receptor model technique was used for the purpose of this study to determine indoor particulate sources based on multivariate statistics. The model was applied to the source apportionment of 19 to 20 chemical species in the PM_4 (PCA). This model has been widely used for aerosol source apportionment studies mostly coupled with Chemical Mass Balance (CMB) for ambient source apportionment (Harrison and Yin, 2004).

In carrying out the PCA, three to five factors with Eigenvalues were obtained. Factor (components) loadings ≥0.6 are presented in Table 21 and 22 in bold case. These loadings correspond to statistically significant variables for both seasons for H05 and H07 presented in Tables 21 and 22 respectively.

The principal components accounted for 85.03% of the total variance during winter in H05 (Table 21). The first factor for winter in H05 (Component 1, 58.39% of the variance) was characterised by K (0.88), P (0.95), Cl (0.61), Ca (0.96), Mn (0.75), Fe (0.55), Al (0.96), Mg (0.96), Na (0.94), S (0.82) and Si (0.96). Elevated loadings of these elements indicated a strong dominance of crustal sources such as dust. Moreover, combustion trace elements K and S were found highly composite within the PM mass in H05 during the winter season. This shows contributions of a combustion source (be it coal and biomass) in close proximity to the NSFB sample house. Component 2 in

the PCA matrix show an impact from a secondary crustal source whereby Cr (-0.83) and Fe (-0.78) are the predominant trace metals.

During the summer, the principal components accounted for 75.25% of the total variance in H05 (Table 21). Component 1 of the PCA was characterised by K (0.83), P (0.96), Cl (0.65), Ca (0.87), Ti (0.95), Mn (0.60), Fe (0.95), Al (0.97), Mg (0.97), Na (0.78), and Si (0.97). Similar to winter, this shows that dust was the dominant source of PM in H05 during the summer sampling campaign. Traces of S were less significant in summer than observed in winter. General reasoning of these low concentrations of S is the reduced burning activity during summer months and elevated atmospheric dispersion. However, the K signature provides with leeway to assume biomass burning (mainly informal waste combustion) was still a prevalent source within the community. Component 2 of the summer campaign indicates vehicle emissions as a prominent source whereby Zn (0.90) was significant.

Table 21: Factor (Components) Loading Matrix (PCA) for the elemental species concentrations in PM_4 with loadings ≥ 0.6 in bold cases for a NSFB household

	NSFB winter 2017				**NSFB summer 2017**		
	C1	C2	C3	C4	C1	C2	C3
K	**0.88**	0.29	0.18	0.03	**0.83**	-0.04	-0.08
P	**0.95**	0.21	0.03	-0.17	**0.96**	-0.04	-0.03
Cl	**0.61**	0.33	0.15	**0.59**	**0.65**	**0.57**	0.21
Ca	**0.96**	0.13	0.03	-0.01	**0.87**	-0.15	-0.07
Ti	-	-	-	-	**0.95**	-0.07	0.03
Cr	0.48	**-0.83**	0.16	0.03	0.06	-0.18	**0.90**
Mn	**0.75**	-0.35	0.17	0.01	**0.60**	0.41	0.23
Zn	-	-	-	-	0.02	**0.90**	-0.18
Fe	**0.55**	**-0.78**	0.16	0.04	**0.95**	-0.14	-0.10
Al	**0.96**	0.19	-0.11	-0.07	**0.97**	-0.15	-0.09
Mg	**0.96**	0.22	-0.08	-0.08	**0.97**	-0.05	-0.08
Na	**0.94**	0.12	-0.16	-0.02	**0.78**	0.52	-0.04
S	**0.82**	0.29	-0.03	-0.08	0.40	-0.21	0.24
Si	**0.96**	0.18	-0.11	-0.08	**0.97**	-0.16	-0.10
% of variation	58.39	20.17	11.21	5.26	54.10	11.29	9.86
Total % of variation	85.03				75.25		

Figure 41 below shows a strong correlation (0.9) for soil elements (Si, Na, Mg, and Ca). Iron (Fe) seems to have another source that is unrelated to Al. A significant attributable source of Fe emissions close to Sharpeville is AcerloMittal, the steel manufacturing company. Wood and coal combustion elements (S, Cl and K) have a positive relationship, although chlorine appears to have another source unrelated to sulphur (Figure 42), which can may occur from other anthropogenic sources of Cl within the vicinity.

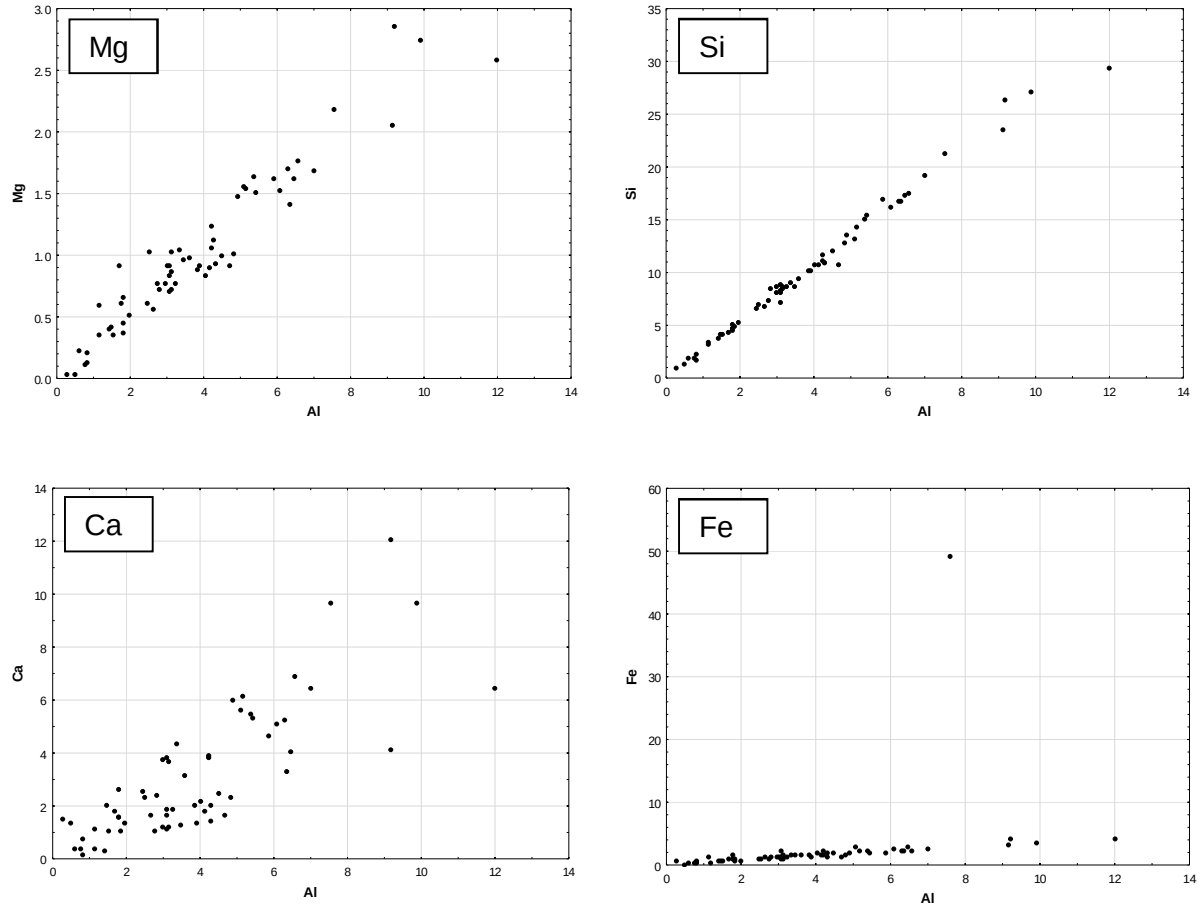

Figure 41: Correlation observed in soil elements using Al as the base element for the NSFB household.

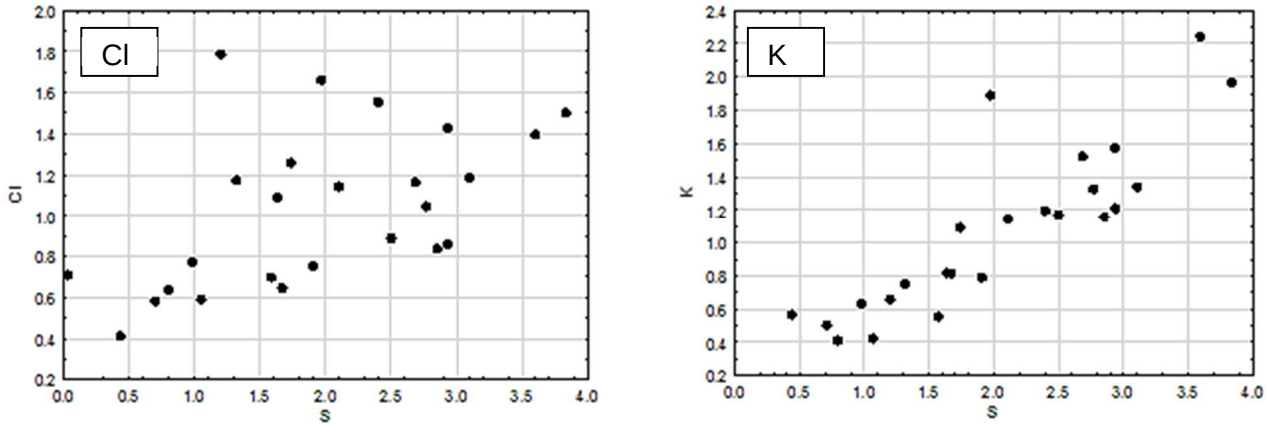

Figure 42: Correlation observed between wood and coal combustion related elements using S as the base element in the NSFB household.

Winter principal components in H07, the SFB household, accounted for 70.41% of the total variance (Table 22). There is a strong dominance of crustal elements in Component 1 {Ca (-0.92), Mn (-0.62), Fe (-0.80), Al (-0.97), Mg (-0.92), Na (-0.65) and Si (-0.97)} which indicates dust as the significant source of PM accounting for 29.91% of the variance. The dust is mainly due to the sample house being located in close proximity to an unpaved road; ventilation methods; no ceiling

available and housekeeping activities. Trace elements for combustion activities (mainly indoor SFB) S (0.55) and K (0.97) fall under Component 2 accounting for 18.14% of the variance. With Cr (0.72), Cl (0.91) and P (0.81) being significant in C2, this indicates industrial emissions impacts within the H07. Therefore, indoor SFB is not the only contributing source to airborne particles inside H07 there is a considerable outdoor impact. The third component accounting for 13.50% of the variance has a signature of S (0.61) which is mainly attributed to indoor burning of coal and ambient pollution from the power generation facility within the area.

The PCA for the summer campaign (68.52% of the total variance) show two components with the first indicating dust as the predominant source of indoor airborne PM in H07. This is mainly due to the following elemental species being significant, P (0.72), Al (0.89), Mg (0.92), Na (0.65) and Si (0.87) with an overall 29.91% of variance. Biomass burning (mainly wood) for domestic activities can be attributed to the K (-0.64) showing significant contribution to the indoor PM composition. Nevertheless, trace metals Cr (-0.64) and Fe (-0.65) in C2, indicate ambient contributions from industrial activities such as the smelter facility located close to Sharpeville. Therefore, three main activities of indoor and outdoor nature (SFB, windblown dust, industry) were the major sources of identified through the PCA matrix from the elemental composition of PM in H07.

Table 22: Factor (Components) Loading Matrix (PCA) for the elemental species concentrations in PM_4 with loadings ≥ 0.6 in bold cases for a SFB household

	SFB winter 2017			**SFB summer 2017**		
	C1	C2	C3	C1	C2	C3
K	-0.01	**0.97**	0.04	0.30	**-0.64**	-0.16
P	-0.11	**0.81**	-0.19	**0.72**	-0.30	-0.12
Cl	-0.09	**0.91**	-0.09	0.03	**-0.56**	0.04
Ca	**-0.92**	-0.09	-0.36	0.48	0.32	0.05
Ti	-	-	-	**0.89**	0.11	-0.09
Cr	0.02	**0.72**	-0.15	0.06	**-0.64**	0.13
Mn	**-0.62**	-0.07	-0.54	0.52	0.08	0.39
Zn	-	-	-	-0.02	0.06	0.19
Fe	**-0.80**	-0.07	-0.19	0.23	**-0.65**	0.11
Al	**-0.97**	-0.02	0.00	**0.89**	0.17	-0.11
Mg	**-0.92**	-0.03	-0.14	**0.92**	0.20	0.01
Na	**-0.65**	0.22	0.46	**0.65**	0.17	0.09
S	-0.38	**0.55**	**0.61**	0.44	-0.14	0.12
Si	**-0.97**	-0.01	-0.02	**0.87**	0.19	-0.11
% of variance	29.91	18.14	13.50	29.91	13.50	6.97
Total %	70.41			68.52		

Soil elements were all plotted against Al in Figure 43. Si, Ca and Mg showed a positive relationship with Al, whereas Fe, P, K, Cl, and Mn appeared to have a source unrelated to Al. This means that there are other natural or anthropogenic sources of Fe, K, Cl and Mn that is not from the top soil. Fe and Cl are likely from an industrial source whereas K is a key signature of

biomass burning and possibly local informal waste burning. Wood and coal combustion elements Cl and K were plotted against S and they seemed to have a less positive correlation, whereby Cl and K were indicated to have another source unrelated to S (Figure 44). This essentially means there are other anthropogenic sources of Cl that had an impact on the indoor environment unrelated to the already existent solid fuel combustion. This may be attributed to the industrial operation close to the Sharpeville community. The relationship between S and K show that K had a source other than coal which is possibly biomass burning from wood and informal waste burning within the community. Further investigations can be conducted to determine the exact sources of Cl and K to understand their impact on indoor air pollution in the Sharpeville community.

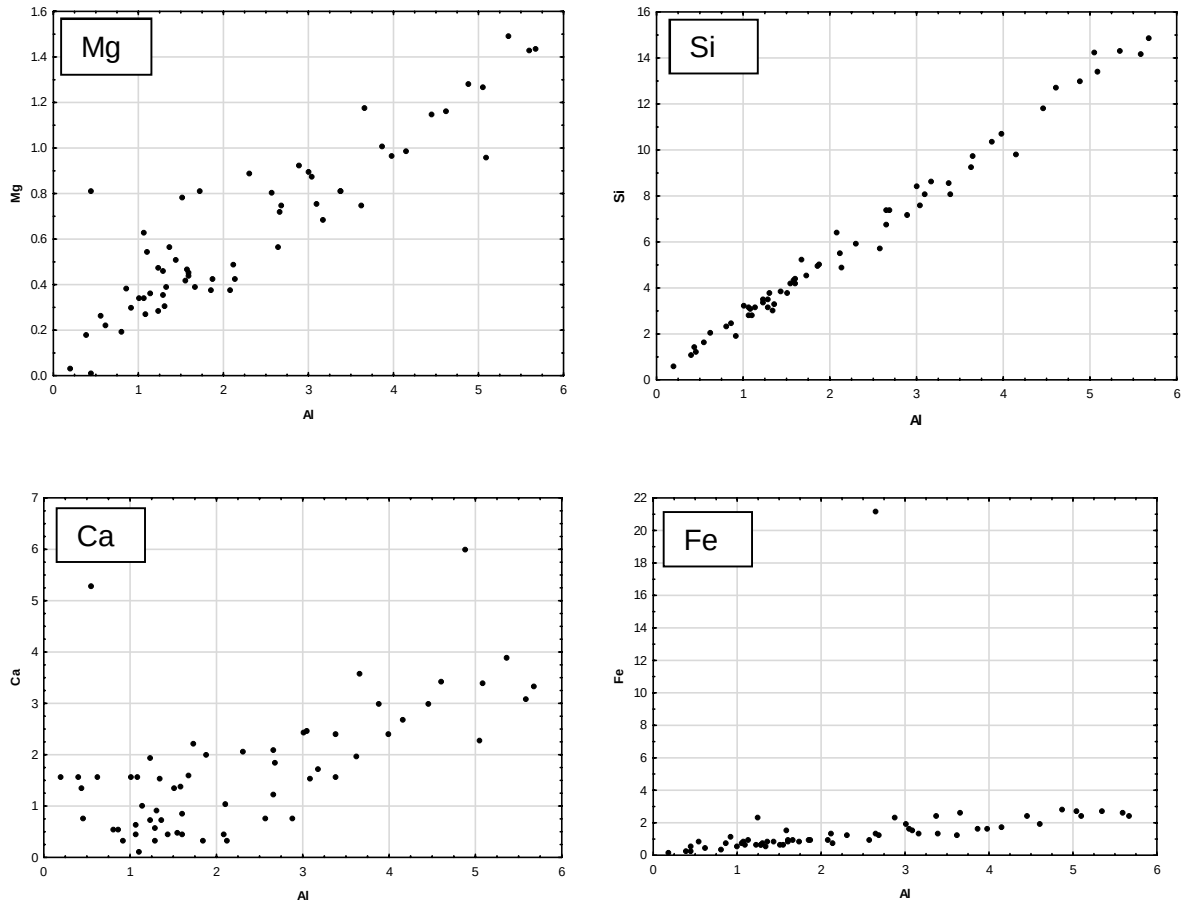

Figure 43: Correlation observed in crustal soil elements results using Al as the base element for the SFB household

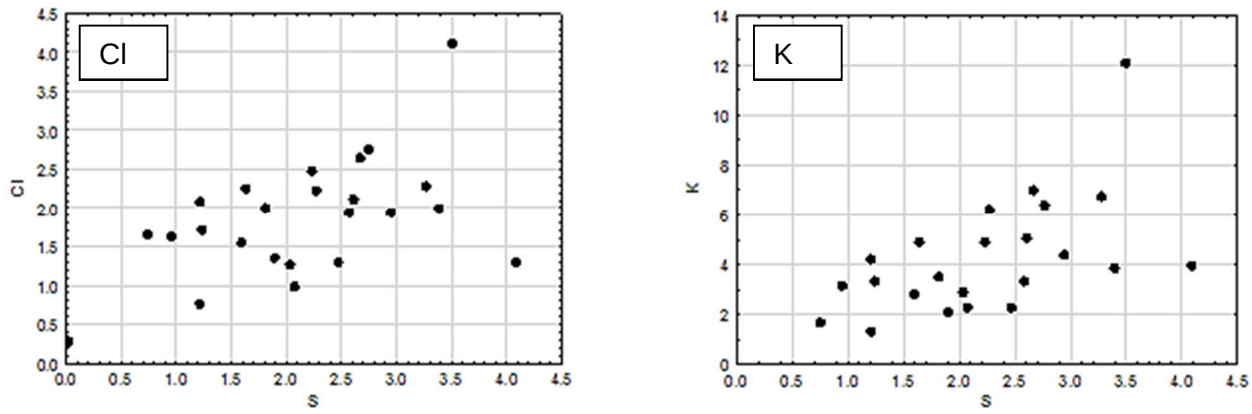

Figure 44: Correlation observed in combustion elements in the PCA results using Al as the base element for the SFB household

CHAPTER 6: CONCLUSIONS

Chapter overview: This chapter provides a summary and brief discussion on the key findings of this study. An overall conclusion is made on the outcome of the study.

6.1. Summary and conclusions

Solid fuel use for residential purposes is of great concern as it contributes significantly to indoor and ambient air pollution. It also raises concern due to the associated human health implications both for the occupants of the relevant house and people in the broader community. Addressing domestic solid fuel use in low-income areas, in South Africa, has been an issue due to the limited data available. The aim of this study was to characterise indoor airborne particulate matter within typical residential households found in a low-income settlement in South Africa. Firstly, the household energy source profile of the Sharpeville low-income settlement was explored. Secondly, the temporal variability (diurnal, daily and seasonal) of indoor respirable particulate matter was characterised from the Sharpeville low-income community. In addition, the concentrations were evaluated against the NAAQS. Although the NAAQS were set for ambient air pollution, they were applicable to this research because exposure to indoor particles causes similar health problems. Lastly, the elemental composition of the indoor respirable particulate matter was characterised and source contribution estimated through PCA.

6.2. Summary and key finding

6.2.1. Household energy source profiles in Sharpeville

Sharpeville has a rather typical household energy source profile compared to other South African low-income communities, although there are some differences. The Detailed Energy Survey (DES) identified electricity as the main energy source, with 96% of the community being electrified. However, solid fuel use for domestic purposes is still a prevalent practice. The household size in the low-income settlement is 3.8 persons per household. Income level estimations per household per month were ±R3,284 which is roughly ~R35 per person per day. This may not be accurate due to most respondents not being willing to disclose their income. It is, however, representative of the typical monthly household in low-income communities. The low household income level could explain the use of solid fuels for residential purposes since electricity tariffs and other cleaner energy alternatives are costly.

There are seasonal household differences in the consumption of wood, coal and paraffin. During winter 24% (2,973) of the households use wood, whereas 15% (1,901) and 3% (337) use coal and paraffin, respectively. This indicates that wood is the dominant fuel used for domestic activities (cooking and heating). During summer, these percentages drop to 1122 (9%) for wood use compared to the 280 (2%) using both coal and paraffin, respectively. Although the number of

wood-using households is higher than those who use coal, it is found that per coal-using household, the quantity of coal used in a given period is higher than that of wood consumed per wood-using household. Each household consumes ~85 kg of coal per month, whereas wood and paraffin consumption is ~47 kg and ~14 litres per month, respectively. On average, the whole communities' consumption of coal is a factor of 1.2 and 3.2 times higher than wood and paraffin in winter. Vice versa, in summer only 2 tonnes/month of coal is consumed and 15 tonnes/month of wood which makes wood use 7 times higher than coal. This means that winter is dominated by coal use and summer by wood.

This is also supported by the annual consumption per coal-using household at 400 kg/annum and wood per wood-using household is 292 kg/annum. However, wood is the dominant household energy carrier when the consumption level is extrapolated for the whole community. This is reflected in the 672 tonnes/annum of wood estimated to be consumed by the community. In a nutshell, wood is the most widely consumed household fuel whereas coal is used intensively per coal-using household. This can be explained in terms of fuel preferences and durability, whereby coal burns longer than wood; it is generally preferred during cold winter months. Wood is used more than coal in summer because residents might not need to burn the fuel for an extended number of hours to heat the interior of their homes. It is mostly used for cooking purposes during summer.

6.2.2. Continuous measurements: Indoor PM_4 concentration characteristics

6.2.2.1. Daily (24hr) indoor PM_4 concentrations

Household concentrations were measured against PM_{10} and $PM_{2.5}$ 24hr average National Ambient Air Quality Standards (NAAQS). The NAAQS were used as a proxy for the non-existent indoor air quality standards in South Africa. Daily average concentrations during winter for all sample houses were above the 24hr $PM_{2.5}$ NAAQS thus a 100% exceedance rate for winter. Only 46% of the sample houses exceeded the PM_{10} 24hr NAAQS in winter. Summer daily average concentrations were below the 24hr exposure threshold (24hr NAAQS) of $PM_{2.5}$ for about 80% of the households. Two exceedances of the $PM_{2.5}$ 24hr threshold were recorded accounting 13% of the exceedance rate. Only 7% exceedance rate of both PM_{10} and $PM_{2.5}$ was observed during the summer sampling period. In terms of non-solid fuel burning households (NSFB), all sample houses were above the $PM_{2.5}$ NAAQS during winter with only 30% of these households above both PM_{10} and $PM_{2.5}$ 24hr NAAQS. Similarly, all solid fuel burning households (SFB) experienced 100% exceedance rate of both PM_{10} and $PM_{2.5}$ 24hr NAAQS. However, an 83% exceedance rate of both $PM_{2.5}$ and PM_{10} 24hr NAAQS was observed within the SFB household category. This means SFB households experienced elevated concentrations of PM_4 during winter compared to

NSFB. Summer concentrations show no exceedances of the PM_{10} and $PM_{2.5}$ 24hr NAAQS within the NSFB sample houses.

During winter, continuous measurements of PM_4 show that on average NSFB and SFB households experience concentrations of 150±85 µg/m^3 and 220±131 µg/m^3 respectively. Summer shows a less extreme scenario than winter where NSFB and SFB households' mean concentrations were 70±50 µg/m^3 and 99±51 µg/m^3, respectively. For both household types, winter concentrations were on average 2 magnitudes higher than those experienced in summer, with SFB concentrations being higher than those NSFB sample houses.

A student T-test was performed to measure the variability between household categories (SFB versus NSFB), seasons (winter versus summer) and households to each other. The difference in indoor PM_4 concentrations between SFB and NSFB houses is statistically significant (p <0.05) for both (that is, summer and winter) sampling campaigns. Similarly, the seasonal variation was found to be strongly significant, which means that indoor PM_4 concentrations were higher in winter compared to summer.

Average concentrations provide limited information regarding the variability of the entire sample to one another. Therefore, to limit the generalisation of the results, a test of daily means was conducted. The test explored the significance of the statistical variance between houses sampled in the individual campaigns (winter and summer 2017). During winter the indoor PM_4 concentrations in the NSFB households showed variability with low statistical significance (p >0.05) compared to the SFB houses. Summer concentrations show a similar trend where indoor PM_4 concentrations in the SFB houses did not vary significantly from NSFB households. Moreover, two households that remained highly reliant on solid fuels during summer showed high variability to 87% of the sample houses. This means out of the whole sample pool, only two households vary significantly due to the extensive use of solid fuels. These results allow for better understanding and approach when interpreting indoor particulate pollution datasets by limiting general assumptions based solely on average concentrations.

6.2.2.2. Diurnal PM_4 concentrations

The diurnal pattern of concentrations shows a bi-modal distribution of morning and evening peaks for both household types during both seasons. On average, all NSFB households experience morning peaks (06h00-09h00) ranging between 80 µg/m^3 and 150 µg/m^3 and evening peaks (19h00-22h00) at 70 µg/m^3 and 175 µg/m^3. Two NSFB houses, H03 and H16, often experienced significantly high concentration during late evenings with maximum peaks >500 µg/m^3. Three other NSFB houses (H05, H11 and H12) showed to experience relatively low concentrations

throughout the day <100 µg/m^3. Compared to the 24hr PM_{10} and $PM_{2.5}$ NAAQS, NSFB households show exceedances of the standards during peak hours.

SFB households were experiencing high indoor concentrations during winter peak hours (same peak hours for all houses) compared to NSFB houses. Morning peaks were >150 µg/m^3 with even higher evening peaks >200 µg/m^3. Three households (H07, H08 and H14) experienced elevated concentrations >300 µg/m^3 during both peak hours (morning and evening) compared to H02, H04 and H09. Of all the SFB households, H09 has by far the lowest indoor concentrations (<100 µg/m^3). A maximum of ±700 µg/m^3 and ±1050 µg/m^3 were observed in the evening within H07 and H14 respectively. A comparison between both household types shows that indoor concentrations in SFB houses were higher by a factor of 3 than in NSFB houses in morning and evening peak hours during winter.

During summer, diurnal concentrations in NSFB households were well below 50 µg/m^3 for both morning and evening peak hours. House 16 (H16) experienced maximum PM_4 levels of about 200 µg/m^3 in the evening. NSFB houses exceeded the NAAQS $PM_{2.5}$ 24hr averaged guideline during these peak hours. This is similar to most of SFB households whereby indoor concentrations were found to be below 50 µg/m^3 throughout the day except for households H07, H08 and H17. The latter households had concentrations of 100–390 µg/m^3, 50–80 µg/m^3, 140–170 µg/m^3, respectively. These houses exceeded both the 24hr $PM_{2.5}$ and PM_{10} NAAQS throughout the day.

Diurnal concentrations per household types show a seasonal impact on the indoor PM_4 concentrations. NSFB households show a decrease of ~58.3% in summer morning peaks (06h00-09h00) and by ~60.9% evening peaks (18h00-21h00). A decrease of ~50% (~47.5% to 50.9%) in SFB households was observed. This implies that NSFB households experience ~57.2% less of the indoor PM_4 in summer than in winter whereas as SFB households is ~47.5%. Also, SFB households were exposed to half the particulates in summer compared to winter during the peak events of the day.

6.2.2.3. Black Carbon (BC) and Temperature profile description

Two additional black carbon (BC) and temperature profile measurements were taken as supplementary datasets for characterising PM_4 concentrations. BC measurements were taken in H07 to conduct a real-time assessment concurrent with PM_4 measurement only for the summer campaign. Daily average concentrations were 3.60±4.34 µg/m^3. No standards were used to compare severity of BC concentrations. This was done solely to assess continued use of solid fuels during the summer period in H07. The diurnal distribution shows a bimodal concentration pattern with morning (05h00–08h00) and evening peaks (18h00–22h00) similar to indoor PM_4

measurements. Elevated BC and particulate loading in H07 indicates continued use of solid fuels during summer because BC is a key signature of solid fuel combustion.

Household structure needs to have temperature properties suitable to keep the indoor environment thermally satisfactory to the dwellers. Most low-cost houses built through the RDP government program were not structurally equipped to offer thermal satisfaction. The houses are often built using cheap material which compromises the thermal efficiency of the house for the dwellers. Temperature profiles during winter are below the minimum WHO value for over 60% of the day. These low temperatures prompt residents to burn more and reach a satisfactory level of thermal comfort. Temperatures start to pick from 10h00 remain well within the WHO indoor temperature guidelines and remain above the minimum standard for the rest of the day. Indoor warming through heating (using clean or solid fuel) were observed and assumed to occur in the evening when ambient temperatures drop below 15°C. Indoor temperatures remain, on average, above 17 C and rise to about 19 C for 2 hours and this drops, but not below the minimum value. In some of the houses, the temperature can go as low as <5°C which can be uncomfortably cold. However, on average indoor temperatures range between 14°C and 22°C. Summer indoor temperatures range between 21°C and 28°C and are well within the WHO guideline for about 75% of the day. Extreme indoor temperatures (above WHO max) are experienced between 10h00 and 17h00. This shows that temperature profiles are potential drivers for indoor burning. The higher the indoor temperatures, the lesser the burning which is used to explain the low indoor PM_4 concentrations in SFB households. Observations in SFB households show that with lower temperatures there is an increase in PM concentrations whereas higher temperatures show the opposite. This showed a relationship between PM concentrations and indoor temperature profiles.

6.2.3. Elemental composition characteristics of indoor airborne respirable PM_4

Based on the filter sampling, mass concentrations in winter were high in the solid fuel burning households (SFB) during winter whereas, in summer both SFB and non-solid fuel burning houses (NSFB) showed similar concentrations. Winter average concentrations were 32 (± 23.3) µg/m^3 and 85 (± 100.3) µg/m^3 for NSFB and SFB houses respectively. Summer concentrations were 57 (± 29.4) µg/m^3 and 74 (± 32.7) µg/m^3 for NSFB and SFB sample houses respectively. Seasonal differences observed involved a 43.9% increase from winter to summer in the NSFB and 12.94% decrease in the SFB sample. This shows summer concentrations in the NSFB house were higher than in winter. Due to the lowered ventilation activities in winter, it was expected that the results show an increase of mass concentrations in summer where ventilation is prioritised NSFB samples. A decrease in mass concentrations in SFB samples is a key signature of reduced indoor burning of solid fuels.

Elemental concentrations were represented in a day and night (D/N) ratio for both NSFB and SFB households. This was done so as to understand which elemental species were found highly concentrated during the day and night, respectively. In the NSFB, crustal elements have high-concentration episodes during the day in summer and in the night during winter. Combustion elements vary, whereby S and Cl were dominant in the night for both seasons whereas K has shown elevated concentrations during day-time in summer and in night-time during winter. The SFB household showed a similar concentration ratio as the NSFB house in terms of crustal soil elements. However, Ca was > 1 for both seasons which indicates a dominant source of elemental Ca is prevalent during the day. Two combustion elements (S and K) were found to be high during the night in winter and during the day in summer, whereas high Cl concentrations were experienced during the day in both seasons.

Aluminium (Al) was used as the reference element to evaluate level of enrichment for the elements. This enrichment factor indicated whether the element was of crustal soil- or anthropogenic origin. Both seasons indicate that dust related elements showed low to no enrichment. However, Calcium (Ca) demonstrates seasonal signature which indicates an additional source of fine particulate Ca in summer. The enrichment of combustion elements varied between the household types. Potassium (K) remained <1 in the NSFB which indicates no enrichment in contrast to the moderate enrichment in the SFB samples. K is a key signature of biomass burning which provides reason to assume that it was enriched as a function of wood burning in the SFB house. Phosphorus (P) showed low and moderate enrichment in NSFB and SFB houses, respectively. Sulphur (S) was highly enriched by a factor of almost a 100 in both household types. Winter and summer enrichment levels were indicative of ambient pollution impact within the NSFB household. In comparison, S in SFB samples was primarily from the solid fuel combustion. Other elements that showed elevated enrichment factors of a 100 and greater than a million were Zinc (Zn) and Lead (Pb). These elements are originally associated with vehicle emissions. This means that particulates sampled in Sharpeville are mainly characterised by crustal, combustion and vehicle emissions.

PCA receptor-modelling was employed for factor analysis and source characterisation of the respirable PM based on elemental species. The high loadings for these elements show the strong dominance of crustal sources such as topsoil for both household types, >50% in NSFB and >29% in SFB. Transition metals Cr, Mo, Ag and Fe showed high levels of contribution (over 20%) and ascribed to secondary crustal sources in the NSFB household. There were less significant signatures of S in NSFB for both seasons. However, sulphur (S) found highly concentrated in the crustal element Component 1 during winter in NSFB can be naturally reserved in the organic matter concentrated within the topsoil. The SFB household show Component 1 (>29%) represent a crustal source of particulates mainly topsoil. Nevertheless, Component 2 accounting for 18% in

winter and 13.5% in summer show significant signatures of biomass burning impacts within the household. Sharpeville is notorious for elevated informal waste burning which thus has an impact of indoor air quality. Winter also Component 3 accounting for 13.5% of the contribution being from coal burning as sulphur was observed to be the significant element. This indicates preeminent domestic combustion of solid fuels material during winter. Low sulphur signatures in summer is attributed to less indoor coal burning activities.

6.3. Study conclusions

Household energy profiles within the Sharpeville community consist of electricity, wood, coal and paraffin. Electricity is the predominant clean energy carrier in the area. Wood, coal and paraffin are mainly alternative fuels used for domestic activities (cooking and space heating). Coal was found to be the most preferred during the winter campaign whereas wood was used more in summer. However, wood is the dominant household energy carrier in the Sharpeville community.

Indoor burning of solid fuel contribute significantly to indoor airborne particles exposing residents hazardous levels of particulates. Typical indoor respirable particulate matter in solid fuel burning (SFB) households indicate concentrations above the average daily threshold of the National Ambient Air Quality Standards (NAAQS). Although the NAAQS are not necessarily indoor guidelines, they however emphasise the health impacts Sharpeville residents are at risk of. Similar to most indoor respirable PM studies as a function of domestic fuel burning in South Africa winter concentrations represent a more severe scenario compared to summer due to the low ambient temperatures and extensive solid fuel use. This is also found to occur in the Sharpeville low-income community. Moreover, residents reliant on cleaner household energy carriers experience less extreme exposure to airborne particles. However, the concentrations in these households tend to exceed 24hr $PM_{2.5}$ NAAQS in the winter during peak hours of the day. Summer concentrations in NSFB households were mostly below both 24hr PM_{10} and $PM_{2.5}$ NAAQS. Although there is a decrease in concentration levels of respirable PM in SFB households, PM_4 is problematic during both seasons in these houses. In most cases, residents of SFB households were exposed to PM_4 levels above the 24hr $PM_{2.5}$ and PM_{10} NAAQS and this is a considerable health hazard due to poor dispersion within residential environments. On a daily average temporal scale, most of the households experienced lower PM_4 concentrations in the summer whereas, in a number of SFB households, PM_4 concentrations remained problematic. Such PM is thus a significant health hazard in need of attention. Measurements of BC conducted in an SFB household supports the assumption of continued fuel-burning in summer within these houses.

The elemental components show that indoor particles are from dust re-suspension within the households. Key crustal elements were detected with elevated loading and consistent in both seasons. Moreover, fundamental combustion elements were identified to be of indoor solid fuel burning, ambient emissions from power generation and industrial activities as well as informal

waste incineration. This gives an indication of the impact outdoor pollution has into the homes of the Sharpeville community. This means not only are residents exposed to elevated levels of indoor PM_4 concentrations, other sources such industrial and vehicle emissions contribute to these indoor concentrations.

The study contributes to the overall aim of understanding pollution related health risks in low-income communities in South Africa. It also adds to the body of science in terms of showing the variability of low-income households and drivers to the individual use of solid fuels as household energy carriers. The study fosters an understanding of other air pollution sources to the indoor environment and the health of people thereof.

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