

Using dispersion models as a regulatory tool in South Africa

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DEDICATION

This is for you mom.

I dedicate this study to my mother who lost her battle with cancer during the course of this study on the 2nd of June 2020.

“God is not human, that he should lie, nor a son of man, that he should change his mind. Does he speak and then not act? Does he promise and not fulfil?”

Numbers 23:19

PREFACE

The study presented in this document represents the original work conducted by the author between March 2019 and March 2021 and contains information not previously submitted for degree purposes at any university.

The author, with the help of the supervisors, was involved in the conceptualisation, data collection and analysis for the study. All the data and information that was extracted from any other sources were aptly acknowledged. The referencing style of this study follows the guidelines of the “*NWU Referencing Guide*”.

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ABSTRACT

Assessment of exposure to air pollution within densely populated low-income townships (intra-urban) is a priority area for research, due to urban air-sheds being heterogeneous with concentrations varying significantly temporally and spatially. Often assessment of exposure in developing countries is limited to sparse measurements due to costs associated with setting up monitoring networks. Therefore, the application of dispersion models can improve the accuracy of assessments which can advance understanding of the exposure at the receptors and enhances the evaluation. Until 2014, dispersion modelling practices in South Africa have been operating in the absence of a standardised regulatory approach leaving room for inconsistencies in assessment studies. Until now, the Code of Practice has uncertainties regarding modelling intra-urban environments. Therefore, a need exists to provide an assessment of the application of dispersion models in South African intra-urban environments. This research aims to evaluate the use of dispersion models as regulatory tools in modelling emissions within low-income settlements to better understand the contributions of local sources in intra-urban environments. To realise the aim, the study will identify, characterise and estimate emissions from local sources within Zamdela and Wedela (South Africa), evaluate the impact of the identified sources on ambient air of the townships using dispersion modelling, and evaluate the performance of the model against monitored particulates. Solid fuel burning, waste burning, fugitive dust, vehicle emissions and biomass burning were identified as potential sources within the study areas. These sources are usually overlooked yet their impact is immediate as they are at the breathing level. Emissions from sources were estimated using the product of emission factors and activity rates and it was observed that fugitive dust contribution is significant, yet usually ignored in research due to their complexities. The type of fuel used is mainly dependant on the quality of life and accessibility as shown by the difference in energy use between Zamdela and Wedela, hence the impact of solid fuel-burning is mostly endured by the poor. Vehicles also showed significant contribution followed by biomass burning then waste combustion. AERMOD was run for 2018 to 2020 as this period aligned with the measured data. $PM_{2.5}$ and PM_{10} were modelled, and the results were compared against measured data from sites within the townships for validation purposes. Maximum concentrations over Zamdela were predicted in the core of the township while Wedela projected highs in areas downwind of the mine tailings. Monitored and modelled data compared relatively well temporally and spatially, suggesting that the modelled sources are possibly the major contributors to the measured data. Therefore, it can be concluded that with proper inputs and procedures dispersion modelling can aid as a regulatory tool in managing air quality.

Key terms: Air pollution, intra-urban, particulate matter, dispersion modelling, low-income settlement.

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GLOSSARY

AERMOD	Gaussian steady-state air pollution dispersion model that has single point meteorology.
Air pollution	The release of substances into the atmosphere in large quantities which can be harmful to human health and the environment.
Air Quality	A measure of exposure to air pollution.
Ambient Air	The air of the surrounding environment; outdoor air.
AP-42	A compilation of emission factors and process descriptions for a broad range of emission sources. EPA Document Number AP-42, Compilation of Air Pollutant Emission Factors, Environmental protection Agency, Research Triangle Park, North Carolina. Current information on AP-42 is available on the US EPA website.
Dispersion modelling	A standard technique for evaluating the impact of emissions from different sources on the receptor.
Emission Factor	Representative value, relating the quantity of a pollutant to a specific activity resulting in the release of the pollutant to the atmosphere.
Intra-Urban	Defines the area within an urban area.
Low-income settlements	Area housing residents that receive no or low revenue (urban poor).
Particulate Matter	A combination of chemical and physical elements that vary by location and time.

ACRONYMS/ ABBREVIATIONS

AERMAP	AERMOD Mapping Program
AERMET	AERMOD Meteorological Pre-processor
AERMOD	American Meteorological Society/Environmental Protection Agency Regulatory Model Improvement Committee Dispersion Model
CBL	Convective Boundary Layer
°C	Degrees Celsius
DEA	Department of Environmental Affairs (now DFFE)
DEFF	Department of Environment Forestry and Fisheries (now DFFE)
DEM	Digital Elevation Model
DFFE	Department of Forestry, Fisheries and the Environment
GBD	Global Disease Burden
IPCC	Intergovernmental Panel on Climate Change
kg	Kilogram(s)
km	Kilometre(s)
LPG	Liquid Petroleum Gas
m	Metre(s)
m³	Cubic metre(s)
m.s⁻¹	Metres per second
µg	Microgram(s)
µg.m⁻³	Micrograms per cubic metre
MODIS	Moderate Resolution Imaging Spectroradiometer
NAAQS	National Ambient Air Quality Standards
PM_x	Particulate Matter with diameters less than x µm
PBL	Planetary Boundary Layer
PAH	Polycyclic Aromatic Hydrocarbons
SAWS	South Africa Weather Service

SAAQIS	South African Air Quality Information System
SAST	South Africa Standard Time
t.a⁻¹	Tons per annum
TSP	Total Suspended Particles
UFP	Ultrafine Particles
US EPA	United States Environmental Protection Agency
UTM	Universal Transverse Mercator
VTAPA	Vaal Triangle Airshed Priority Area
WHO	World Health Organization

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CHAPTER 1

This chapter provides the background and scope of this study. It gives an overview of the complexities governing air pollution in intra-urban environments and how this should be realised before implementing regulatory dispersion modelling.

1. INTRODUCTION

1.1 Background

Assessment of exposure to air pollution within densely populated low-income townships (intra-urban) is a research priority. This is due to several reasons including (i) previous studies which have established that air quality is the poorest in residential densely populated townships in comparison to affluent areas (Scorgie *et al.*, 2003; Hersey *et al.*, 2014; Language *et al.*, 2016; Langerman, *et al.*, 2018). Ambient PM_{2.5} concentrations were found to be about 78% higher in township areas than in industrial areas (Hersey *et al.*, 2014). (ii) Factors such as income, preferences, availability of fuels, and diversity of energy sources that continue to be used for space heating and cooking, with “dirty” fuels being prominent (Hersey *et al.*, 2014; Language *et al.*, 2016). (iii) There is a growing contribution of traffic pollution, with Hoek *et al.*, (2002) predicting that the demand for transportation will exceed improvements to emission reduction technologies. (iv) The usual location of low-income settlements is in areas directly impacted by other major sources of emissions in the form of industrial sites, mining activities and power plants (Hersey *et al.*, 2014). (v) Finally, poor service delivery in low-income settlements results in residents partaking in waste burning as garbage is often not collected, as well as the presence of unpaved roads causing dust exposure (Ferronato, & Torretta, 2019). The above reasons indicate that residents of these settlements are continuously exposed to high concentrations of pollutants, making them vulnerable to adverse health outcomes (DEA, 2019).

Section 24 of the Constitution of the Republic of South Africa (1996) advocates that every person should have access to an environment that is not harmful to their health and well-being. However, as per the prior paragraph, in many densely populated low-income communities in South Africa, this constitutional right is not realised. As a result, several strategies including offsetting programs, interventions, and regulatory standards enforced on industries have been introduced to reduce air pollution in the townships, but little to no change can be observed, especially in the Vaal Triangle Priority Area (VTAPA) (DEA, 2019).

The complexities governing air pollution are often ignored when implementing strategies. For example, the diversity of sources is sometimes overlooked and attention is often given to specific

sources (ITA, 2018), consequently, sources of large percentages of pollutants remain undefined (Petkova, 2013). The chemical composition of pollutants which vary temporally and spatially is not considered, thus the strategic interventions overlook the different activities that emitted pollutants are exposed to in the atmosphere, which controls their dispersion, deposition, lifespan and transformation (Hunter *et al.*, 2002; Xu *et al.*, 2017). Furthermore, the complexities in defining the toxic mechanism whereby multiple compounds and particles interact are overlooked (Xu *et al.*, 2017). The other variables that are often ignored include the impact of regional transportation on areas downwind of sources (Teri *et al.*, 2015), and the effect of meteorological conditions (Radaideh, 2017).

In addition to the complexities, assessment of exposure in developing countries is limited to sparse measurements due to the costs associated with setting up monitoring stations and networks (Szpiro, 2008; Carbonell *et al.*, 2010). According to Gwaze and Mashele (2018), there are about 130 monitoring stations across South Africa which is diminutive given the country covers approximately 1.22 million km² (Stats SA, 2018). Studies show that point monitoring stations do not necessarily give a reliable perspective of an area's ambient air quality, as this can vary significantly temporally and spatially in townships (Petit *et al.*, 2015; Wernecke, 2018).

An alternative approach is to utilise an integrated measurement and modelling technique which can be a solution towards effective air quality assessment (Carbonell *et al.*, 2010). The application of models such as SCIPUFF, CALPUFF and AERMOD can improve the accuracy of assessments which help in better understanding the exposure at the receptors and enhance the evaluation of compliance with standards (Carbonell *et al.*, 2010; DEA, 2014; Kumar *et al.*, 2017). These tools allow for a mathematical prediction of pollutants' concentrations using a simulation of chemical and physical processes of the atmosphere, hence, determining the behaviour, reaction and dispersion of pollutants (Hunter *et al.*, 2002; Kumar *et al.*, 2017). More so, unlike monitored data that can only give information for as long as there has been monitoring and for a specific point, modelling can create simulations of air pollutant concentration scenarios over a region for the recent past, present and future (Kumar *et al.*, 2017).

If these models are applied in the absence of proper principles, evaluation and regulations; erroneous and misleading conclusions may be drawn (Manju *et al.*, 2011; DEA, 2014). This is because models can be manipulated, therefore influencing the output to provide the desired conclusion based on motive (DEA, 2014). More so, models are often developed and evaluated based on specific meteorological conditions and source specifications and are then applied in a different setting without this consideration (Manju *et al.*, 2011).

Given the above, the regulator should come up with robust guidelines to avoid misleading findings and to promote consistent application of the models. It is vital to develop a comprehensive

approach for dealing with the implementation of models for which guidance is needed (US EPA, 2017). This includes advanced coordination with the authorities on the application of the models, and the development of modelling protocols that should be agreed upon by all parties in advance of any modelling activity to be able to evaluate the process (US EPA, 2017).

Modelling techniques are not effective management tools in many countries due to the lack of regulations (Carbonell *et al.*, 2010). Until recently, regulatory air dispersion modelling practices in South Africa have been operating in the absence of a standardised regulatory approach (DEA, 2014). In 2014 the Department of Environmental Affairs (DEA) developed the “Air dispersion modelling regulations and Code of Practice for Air Dispersion Modelling in Air Quality Management in South Africa” to provide procedures and protocols which ensure standardisation and consistency in assessing emissions when using air dispersion models (DEA, 2014).

According to Szpiro (2008), air quality exposure studies can be improved by accurate estimation of intra-urban variation in individual concentrations. However, the Code of Practice (DEA, 2014) has uncertainties regarding the modelling of intra-urban environments. Therefore, a need exists to provide an assessment of the application of dispersion models in South African intra-urban environments. This includes understating the complex nature of townships and incorporating this into the modelling process. Most studies treat townships as area sources with an individual emission rate for the entire area, this study intends to evaluate what the distinct sources within the township are contributing to the ambient concentrations (Thomas, 2009; Mkhonto, 2014). This is a challenge as it means modelling a space where the location of the sources is the same as the receptors.

1.2 Scope for this research

Urban air-sheds are heterogeneous and pollution concentrations in intra-urban environments vary significantly spatially and temporally (Kim *et al.*, 2015, O'Leary & Lemke, 2014). Air sampling networks installed to monitor adherence with national ambient air quality standards (NAAQS) are vital sources of ambient air quality information (O'Leary & Lemke, 2014; Gwaze & Mashele, 2018). While these regulatory monitoring networks provide valuable time-series measurements, they commonly lack the spatial resolution needed to provide neighbourhood-scale exposure estimates. Therefore, a need exists to develop concrete methods to incorporate detailed spatial and temporal air quality data from multiple sources that exist within low-income settlements in order to come up with feasible air quality strategies (Wilson & Zawar-Reza, 2006).

This study makes use of dispersion modelling to depict the spatial distribution in particulate matter (PM) concentrations and source contributions in order to better understand the potential for exposure characterisation error in air quality management studies. AERMOD can be used to assess how the variation of concentrations can be better shown to address the impacts of emissions sources. Two townships were strategically selected based on their location and vulnerability to poor ambient air quality. Wedela, a mining town that is exposed to dust emissions, and Zamdela, a township within the VTAPA.

1.3 Aim and objectives

The aim of this research is to evaluate the use of dispersion models as regulatory tools in modelling emissions within low-income settlements to better understand the contributions of local sources in intra-urban environments.

To realise the aim, the following objectives were set.

1. Characterise the ambient particulate loading in Wedela and Zamdela.
2. Identify, characterise and estimate emissions from local sources within Wedela and Zamdela.
3. Evaluate the impact of the identified sources on the ambient air of the townships using dispersion modelling.
4. Evaluate the performance of the model against monitored particulates.

CHAPTER 2

This chapter gives a narrative of particulate matter emissions, and their environmental and health impacts on intra-urban environments, as outlined by literature. An insight on air pollution and the possible sources within densely populated low-income settlements will be afforded. A comprehensive discussion will be given on how emissions from the possible sources can be estimated. This chapter also conceptualises the use of dispersion modelling and gives a regulatory context for modelling in South Africa.

2. LITERATURE REVIEW

2.1 Particulate Matter

Particulate matter (PM) is a combination of chemical and physical elements that vary by location (space) and time (WHO, 2013; Darçın, 2017). The most common chemical components of PM are nitrates, sulphates, and ammonium; inorganic ions such as potassium, sodium, calcium, chloride and magnesium, elemental and organic carbon; particle-bound water, crustal material; metals including cadmium, copper, nickel, vanadium and zinc and polycyclic aromatic hydrocarbons (PAH) (WHO, 2013). In addition, biological components such as allergens and microbial compounds also make up PM (WHO, 2013; Tan, 2014).

As shown in Figure 2.1-1, PM can either exist as primary – directly emitted into the air, or secondary pollutants – formed due to reactions in the atmosphere from gaseous precursors such as sulphur dioxide, oxides of nitrogen, ammonia and non-methane volatile organic compounds (WHO, 2013; Darçın, 2017).

Primary PM and precursor gases originate from both anthropogenic and natural sources (Darçın, 2017). Anthropogenic sources include combustion of diesel and petrol engines, solid fuel (coal, lignite, heavy oil and biomass) combustion for energy production in households and industries, other industrial activities (building, mining, manufacture of cement, ceramic and bricks, and smelting), and erosion of the pavement by road traffic and abrasion of brakes and tyres (Lindeque, 2018). Agriculture also contributes considerably to ambient PM, as well as being the main source of ammonium (WHO, 2013; Darçın, 2017). Natural sources of PM include sea salt, dust (crustal materials or airborne soil), biological aerosols (pollen and spores), as well as material from volcanic eruptions and wildfires (WHO, 2013). These sources of PM are complex because they contribute to ambient concentrations, which are subject to NAAQS, however little can be done to manage them (EEA, 2019).

Secondary pollutants are formed in the atmosphere through complex physical and/or chemical reactions of gaseous pollutants, or by reaction with normal atmospheric constituents, with or without photoactivation (e.g., coagulation and condensation or photochemical reactions) (Tan, 2014; Lindeque, 2018). They are mostly products of atmospheric transformation of nitrogen oxides (mainly emitted by traffic and some industrial processes) and sulphur dioxide resulting from the combustion of sulphur-containing fuels. Secondary pollutants are mostly found in fine PM. Soil and dust re-suspension is also a contributing source of PM, particularly in arid areas or during episodes of long-range transport of dust (WHO, 2013; Darçın, 2017).

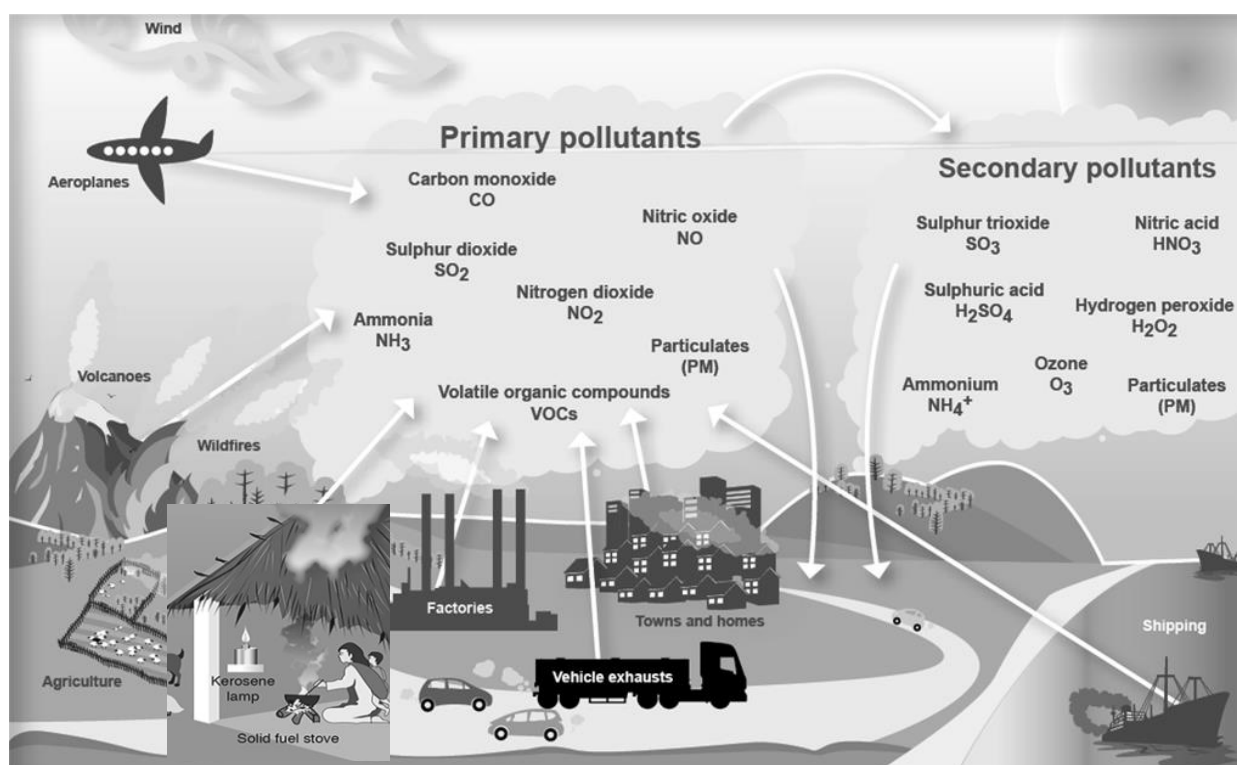


Figure 2.1-1. Typical primary and secondary pollutants and the most common sources (Source; Darçın, 2017)

The size of suspended particulates in the atmosphere varies from as little as a few nanometres (nm) to tens of micrometres (μm) (WHO, 2006; Lindeque, 2018). PM is classified into four size fractions, one of which is total suspended particles (TSP) which denote the total composition of suspended particles in the atmosphere including particles as large as 50 - 100 μm in aerodynamic diameter (WHO, 2006). Coarse fraction refers to particles less than 10 μm in aerodynamic diameter, commonly referred to as PM₁₀, which are inhalable or thoracic particles (WHO, 2006). The coarse PM fraction mostly contains particles produced from the mechanical break up of larger particles such as suspended and resuspended dust from roads and industrial activities, wind-blown dust from agricultural activities, and biological materials like sea salt, pollen and bacterial fragments (EEA, 2019; Tan, 2014).

Fine particulates (PM_{2.5}) are less than 2.5 µm in aerodynamic diameter. The fine particle mix contains directly emitted combustion particles, re-condensed organic and metal vapours, and secondary particles formed from other gaseous pollutants present in the atmosphere (WHO, 2006). Lastly, ultrafine particles (UFP) are <0.1 µm in aerodynamic diameter and are known to have the most significant impact on human health (Pascal *et al.*, 2011; Lindeque, 2018). Ultrafine particles are also emitted from combustion activities and formed through secondary atmospheric processes but have a very short atmospheric lifetime (minutes to hours) as they rapidly coagulate and condensate to form larger (PM_{2.5}) particles (Pope & Dockery, 2006). Particulate matter with particle diameters smaller than x µm is defined as PM_x (Tan, 2014).

Particulate matter has proved to have a substantial effect on human health and the environment (Yadav *et al.*, 2014, Patrick *et al.*, 2015). The fact that PM can be formed primarily or secondarily and continues to undergo chemical and physical transformation whilst in the atmosphere, further adds to its complexity (Lindeque, 2018). This variability complicates the study of exposure and risk as different characteristics of PM may cause different health effects (WHO, 2006; Tan, 2014).

2.2 Impacts of air pollution

Air pollutants are hazardous to the environment and human health if left to accumulate in the atmosphere (Patrick *et al.*, 2015; Tan, 2014).

2.2.1 Health impacts of air pollution

Comprehensive information regarding the impact that emissions have on human health and well-being is a prerequisite towards the advancement of effective strategies and regulations to mitigate the adverse impact of ambient air pollution (Bonn, 2003). World Health Organization Constitution (1948) defines health as “a state of complete physical, mental, and social well-being and not merely the absence of disease or infirmity”.

Air pollution is one of the most significant human health threats faced by countries throughout the world (Yadav *et al.*, 2014, Patrick *et al.*, 2015, Darçın, 2017). Evidence from studies on health and air pollution shows that exposure is associated with a wide range of adverse health-related outcomes (Darçın, 2017). Air pollution is a threat to people exposed to it, however, its impacts are mostly felt by certain groups, for example, aged groups, children, and people with cardiovascular and lung diseases such as asthma. Vulnerability varies between and within communities and is influenced by various factors including age, gender, living conditions, socioeconomic status and access to social support and services (Lindeque 2018). Air pollution forms in two contexts: outdoor and indoor (household).

The Global Burden of Disease risk assessment study estimates that air pollution (outdoor and indoor) exposure was responsible for about 4.9 million premature deaths worldwide in 2016, making it the fourth leading risk factor of the 79 identified (GBD, 2017) (Figure 2.2-1). This is in line with the WHO (2021), which attributed ambient air pollution to be responsible for between 4 and 9 million premature deaths globally. In South Africa, 23 547 deaths were ascribed to have been a result of exposure to air pollution (GBD, 2017). Ambient air pollution in urban areas of South Africa contributes to approximately 3.7% of mortality from cardiopulmonary disease, 5.1% from cancers of the trachea, bronchus and lung and 1.1% from acute respiratory infections in children under five (Norman *et al.*, 2007; Lindeque, 2018). In as much as air pollution is a leading risk factor for death, its effects go even further, it is one of the main contributors to global disease burden (GBD, 2017).

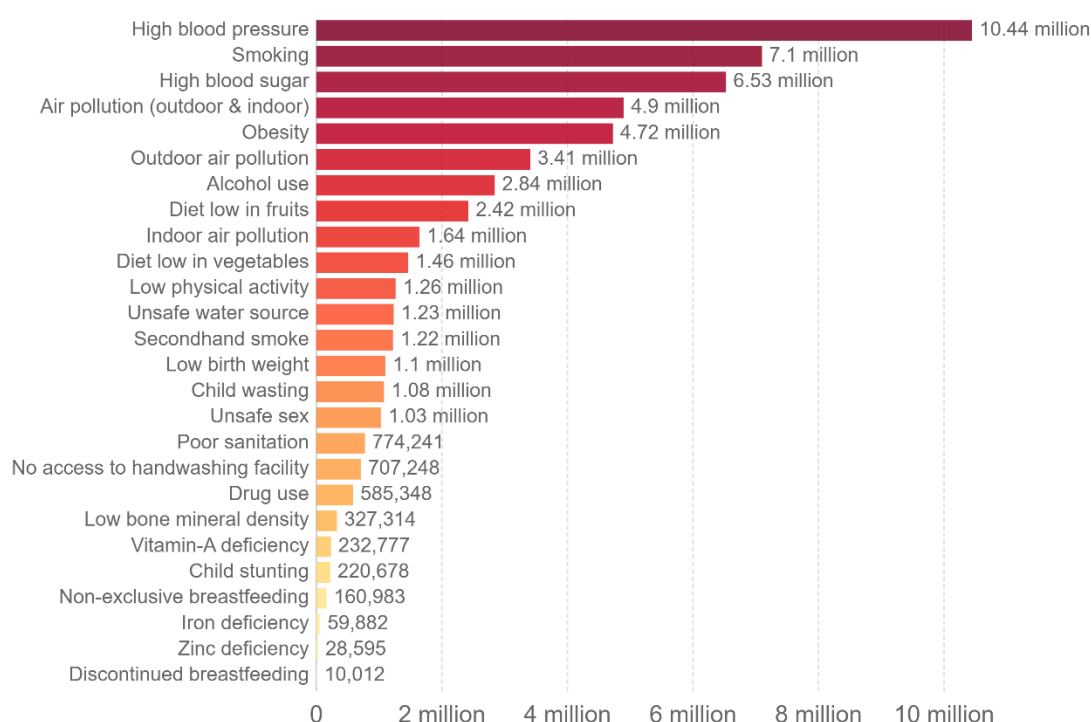


Figure 2.2-1. Total annual estimated deaths by risk factor across the world for 2017 (Source; GBD, 2017).

The quality of air plays a critical role in the state of people's physical, mental, and overall well-being (Darçın, 2017; GBD, 2017). Health hazards are due to exposure over both the short term (hours, days) and long term (months, years). Short term impacts include irritation to the nose, eyes and throat, breathing difficulties, chest tightness, coughing and wheezing (Darçın, 2017). Long term impacts include cancer, neurological, respiratory, reproductive and immune diseases (Darçın, 2017). Thus, exposure to air pollution is directly linked with respiratory diseases, and recent studies have also noted the relationship with pneumonia cases. Pneumonia is responsible for about 16% of all under-five deaths with more than half of childhood pneumonia deaths being directly associated with air pollution (Darrow *et al.*, 2014; GBD, 2017). More so, pollutants affect

cardiovascular health by hardening the arteries thus increasing the risk of strokes and heart attack. Emerging evidence has shown that air pollution could be linked to mental health conditions and degenerative brain diseases such as Alzheimer's disease, schizophrenia and Parkinson's disease (Darçın, 2017). Exposure to high levels of air pollution during pregnancy has been linked to miscarriages as well as premature birth, autism spectrum disorder and asthma in children (Darrow *et al*, 2014). Other conditions associated with high levels of air pollution are summarised in Table 2.2-1 below.

Table 2.2-1. Main criteria air pollutants, origin, and their health impacts (Source; compiled from WHO, 2006; Darçın, 2017; Lindeque, 2018; GBD, 2017).

Air pollutant	Origin	Health impact
Nitrogen dioxide (NO₂)		
• A highly reactive brownish gas that forms rapidly when fuels are burned at high temperatures. • It contributes to the formation of PM_{2.5} and transforms to form nitric acid (HNO₃) in the atmosphere.	• Power plants • Motor vehicles • Industrial plants	• The overall function of the lungs is reduced. • Aggravate susceptibility to respiratory bacterial and viral infections. • Asthma and bronchitis symptoms increase in children with prolonged exposure. • Irritation and inflammation of the airways and lungs. • Intensify existing heart, lung and bronchial diseases.
Sulphur dioxide (SO₂)		
• Highly reactive, irritating and colourless gas which is pungent at high concentrations. • Linked with the formation of aerosols and a driver of acid rain.	• Power plants • Volcanic eruptions • Sulphur containing fossil fuels (coal, petroleum) combustion	• Wheezing and worsening of asthma and respiratory diseases. • Cardiovascular illnesses. • Irritation of the throat, nose and eyes. • Cause coughing and mucus secretion. • Inflammation of the respiratory tract and symptoms of lung disease is increased.
Ozone (O₃)		
• It is a highly active oxidation gas, which is colourless and has a strong odour. • Formed because of chemical reactions between oxygen, NO_x and VOCs.	• Secondary pollutants formed in sunlight as a result of chemical reactions of products emitted by petrochemicals, vehicles, landfills, electric utilities.	• Chest tightening and pain, coughing and wheezing • Exacerbate asthma, bronchitis and emphysema. • Permanent damage to lung tissue, decreasing lung function. • Weariness and headache • Increased vulnerability to bacterial and viral infections. • Irritates the

Lead

- | | | |
|---|---|--|
| <ul style="list-style-type: none">• A heavy metal that is denser than other materials.• Lead is silvery with a hint of blue, it turns to a dull grey colour when exposed to air. | <ul style="list-style-type: none">• Fuel combustion• domestic wood fires• motor engines | <ul style="list-style-type: none">• Causes damage to the central nervous system• hematologic dysfunctions• affect the reproductive system• eye irritation |
|---|---|--|
-

This study focuses on the impacts of PM; therefore, an in-depth discussion of its effect is provided. PM is associated with many adverse human health impacts (Pascal *et al.*, 2013; Yadava *et al.*, 2014; Kim *et al.*, 2015), hence the pollutant of focus for this study is PM_{2.5} and PM₁₀. This is because PM₁₀ and PM_{2.5} particles are very small and can be inhaled, entering the respiratory and cardiovascular systems (Pope III & Dockery, 2006; Lindeque, 2018).

Studies have revealed that exposure to PM can result in respiratory disorders like sneezing, nose blockage, hyperacidity and coughing, eye irritation and premature mortality (Pascal *et al.*, 2013; Yadava *et al.*, 2014; Kim *et al.*, 2015; Yang *et al.*, 2015; Rai, 2015). According to Pascal *et al.* (2011), it is also attributed to affect infant birth weight and cause infant death syndrome. Some specific physiologic endpoints are also associated with PM exposure, including reduced lung function which causes lung inflammation, and enlarged blood plasma viscosity which affects vascular tone and endothelial function, leading to reduced heart rate variability and inflammation mild hypoxemia resulting in a decline in blood oxygen saturation (Darçın, 2017; GBD, 2017).

Exposure to PM_{2.5} is related to “myocardial infarction (MI), ischemic heart disease, dysrhythmias, heart failure, cardiac arrest, and increased carotid intima-media thickness (CIMT)-a measure of subclinical atherosclerosis” (Rai, 2015). PM_{2.5} also has an acute inflammatory consequence causing “neutrophilia, activation of mast cells and neutrophils and the production of cytokines and chemokine associated with the neutrophil accumulation and activation” (Kim *et al.*, 2015; Rai, 2015).

PM₁₀ exposure is mainly associated with increased ischemic heart diseases common among the aged group and poses a greater risk of myocardial infarction (GBD, 2017). A high enough concentration of dust particles in the atmosphere results in acute respiratory illnesses such as “sinusitis, bronchitis, asthma and allergy and damage to the defensive functions of alveolar macrophages leading to increase respiratory infections”, causing one to be vulnerable to other respiratory diseases (Rai, 2015; Yang *et al.*, 2015).

Figure 2.2-2 summarises some of the cardiovascular health impacts associated with exposure to high levels of PM.

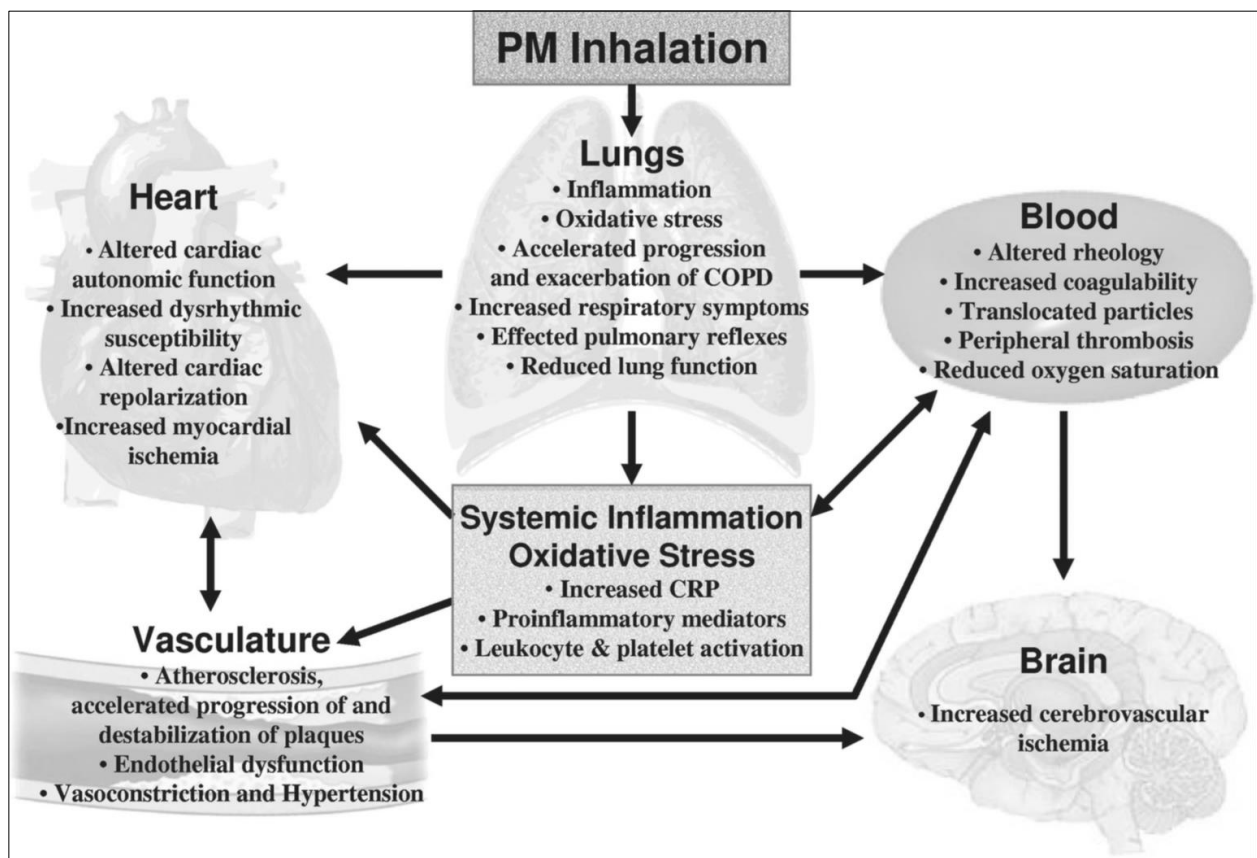


Figure 2.2-2. The pathophysiological pathways connecting PM exposure with cardiopulmonary diseases (Pope & Dockery, 2006).

2.2.2 Environmental effects of air pollution

In addition to the impact on human health, air pollution has effects on the environment resulting in acid rain, eutrophication, haze, ozone depletion, crop damage, effects on wildlife, and is the main driver of global climate change (Patrick, *et al*, 2015).

Some compounds emitted into the atmosphere are converted to acidic species. For instance, through oxidation and dissolution, emitted NO_x and SO_2 are converted to nitric acid (HNO_3) and sulfuric acid (H_2SO_4) respectively (Figure 2.2-3), and similarly, ammonia gas (NH_3) is transformed into ammonium (NH_4^+) (Tan, 2014). The deposition of such species in excessive amounts can cause ecosystem damage (Stanitski *et al*, 2000). The deposition of such species in excessive amounts causes ecosystem damage, as well as damage to buildings and other objects (USEPA, 2021).

According to H. John Heinz III Centre for Science (2008), acid rain can be very damaging on trees. It weakens them by washing away the protective film on leaves and stunts growth. Acid rain also alters the chemistry of soil and water bodies, making them uninhabitable for native animals and plants. For example, healthy lakes have a pH of 6.5 or higher. As acid rain increases the level of acidity, the death of fish in the waterbody may occur (USEPA, 2021). Most fish species

cannot endure a water pH under 5, therefore when pH is less than 4, the lake is considered dead (Mattson, 1999).

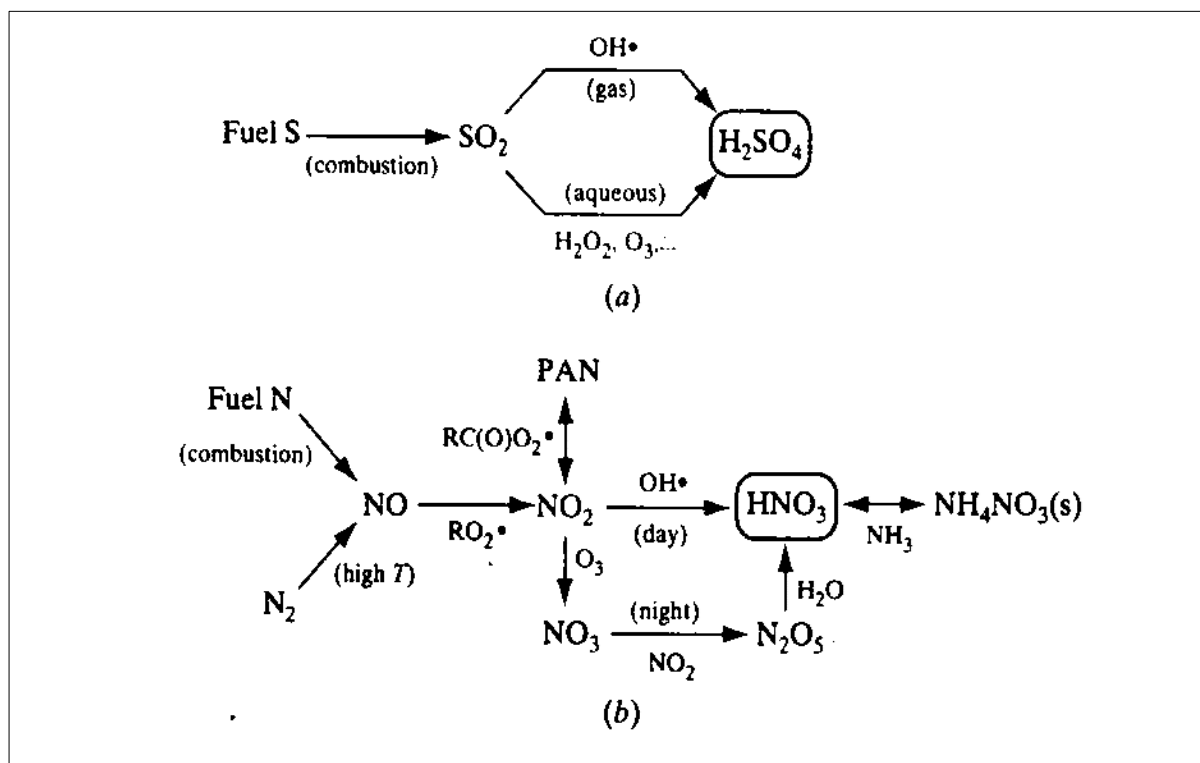


Figure 2.2-3. Transformation dynamics that produce atmospheric acids from (top) sulphur oxides and (bottom) nitrogen oxides. Where R is a hydrocarbon fragment such as CH₃, PAN is Peroxyacetyl nitrate (Tan, 2014).

Compounds tend to display different chemical behaviour when excited. Some pollutant species react under the influence of sunlight known as photochemical reactions (Mohammadi, 2012). If a mixture of organic gases, nitrogen (N), and oxides (O_x) are exposed to sunlight, nitrogen dioxide (NO_x) will dissociate, liberating oxygen radical that combines with an oxygen molecule to produce ozone (Lobo *et al.*, 2010) (Figure 2.2-4). According to Patrick *et al.* (2015), central to the reactions are transformation processes, chemical kinetic reactions, and phase changes that produce secondary pollutants whose effects are worse than the primary emissions.

Photochemical smog has adverse effects on the environment. This is because the chemicals found in photochemical smog can cause harm to plants and animal life (Mohammadi, 2012). Some plants such as spinach, tobacco and tomato respond highly to ozone, hence photochemical smog can affect the growth and productivity of these sensitive crops (Mohammadi, 2012; Charmaine, 2019). Ozone also causes uncharacteristic effects on the upper surfaces of tree leaves. The effects of smog on animals are similar to its effect on humans; it decreases lung capacity and lung elasticity (Charmaine, 2019).

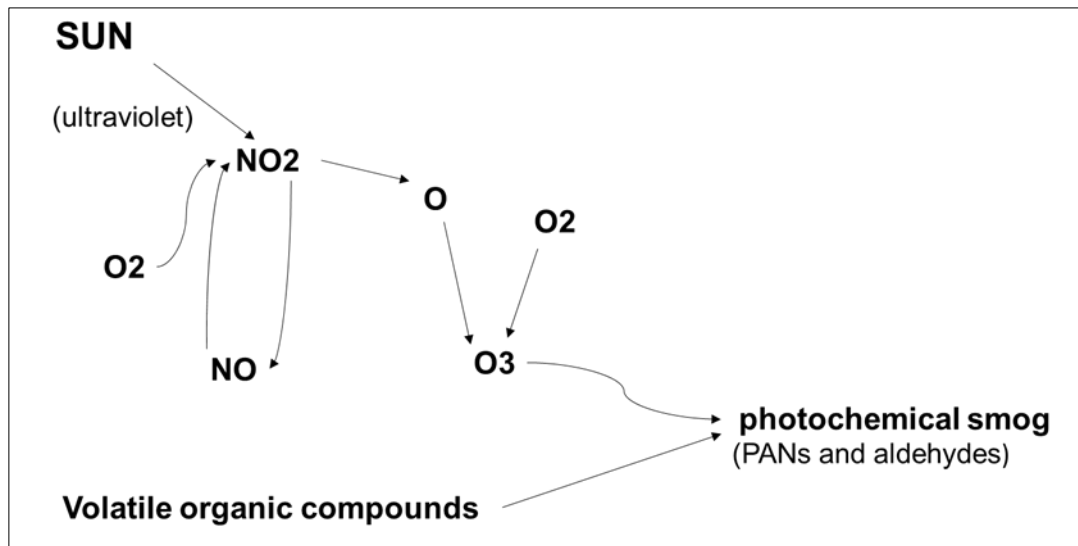


Figure 2.2-4. Transformation dynamics leading to the formation of Photochemical Smog in the atmosphere (Lobo et al., 2010).

Air pollution is closely related to climate change as the main driver of climate change is fossil fuel combustion (Kumar *et al.*, 2017). Emissions of pollutants into the air can result in changes to the climate. Ozone in the atmosphere warms the environment, while different components of PM can have either warming or cooling effects on the climate (IPCC, 2013). For example, black carbon, a particulate pollutant from combustion, contributes to the warming of the Earth, while particulate sulphates cool the earth's atmosphere. Increased concentrations of airborne particulates in the atmosphere have already caused the warming of the planet significantly, resulting in more severe and lengthy heat waves, temperature variability, veld fires, droughts and floods (IPCC, 2013; Solomon *et al.*, 2007).

2.3 Air pollution in townships of South Africa

Several studies have settled that the poorest air quality is found within the densely populated low-income areas (Scorgie *et al.*, 2003; Hersey *et al.*, 2014; Language *et al.*, 2016; Langerman, *et al.*, 2018). Langerman *et al.* (2018) showed a contrast in PM between two neighbouring towns, KwaZamokuhle (a low-income settlement) and Hendrina (an affluent settlement) (Figure 2.3-1). Despite them sharing the same regional pollution sources, the meteorological and topographic settings recorded contrasting particulates measurements, verifying the significance of local sources within townships (Langerman, *et al.*, 2018). The area between the red and blue lines is argued to indicate the contribution of domestic burning and traffic as morning and evening peaks are clear (Langerman, *et al.*, 2018; Chidhindi *et al.*, 2019).

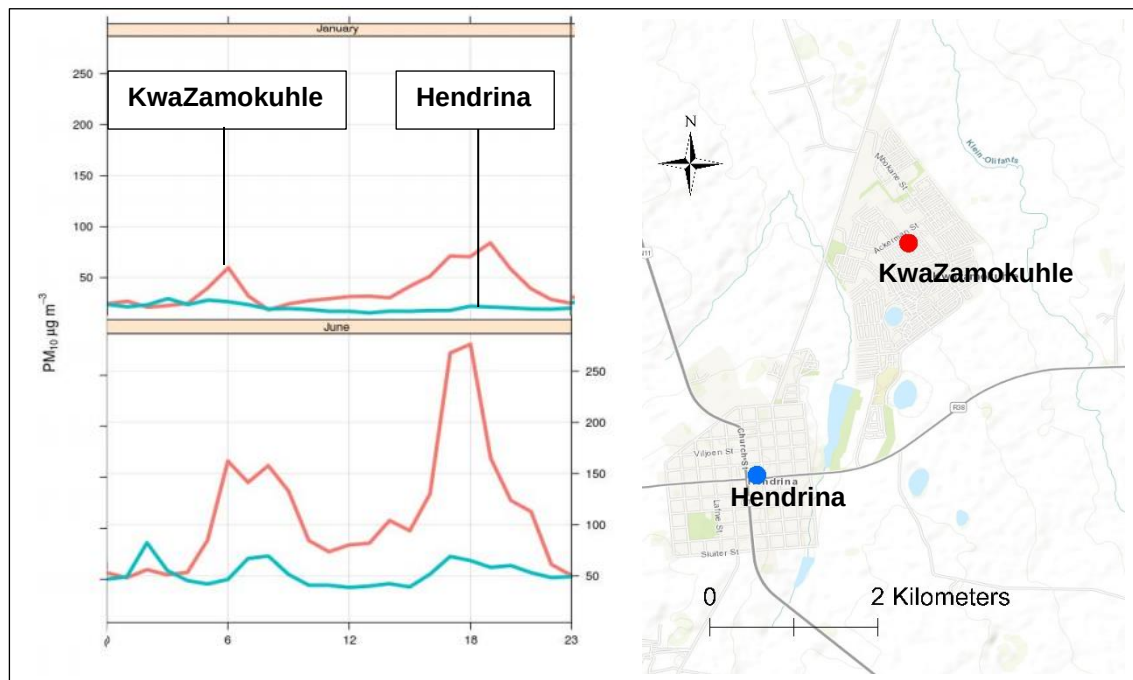


Figure 2.3-1. Average diurnal profiles of PM₁₀ in summer and winter respectively as measured in KwaZamokuhle (indicated in red) and an adjacent settlement Hendrina (indicated in blue) (Langerman et al., 2018).

South Africa has a complex history, which created a huge class gap, hence it has air quality problems related to both developed and developing countries (Lindeque, 2018). While industrial activities and high numbers of vehicle ownership are major sources of emissions, at the same time, solid fuel burning and waste burning also contribute extensively to air quality problems in the country (DEA, 2019).

Most people in low-income settlements earn just enough to afford basic needs and cannot make use of clean energy options for heating and cooking, so they settle for the more affordable and easily accessible options such as wood, paraffin fuel, and coal (John & Sonali 2012; Language *et al.*, 2016). Additionally, dense low-income settlements typically have unpaved roads, few trees and bare soil that exacerbates air pollution (DEA, 2019). The public health risk due to pollution is amplified further by socio-economic factors such as malnutrition, poverty and underdevelopment (Thabethe *et al.*, 2014). Therefore, since low-income settlements are typically related to high levels of poverty, lack of resources and poor service delivery (Language *et al.*, 2016), their residents are more vulnerable to the impacts of poor ambient air quality (Hersey *et al.*, 2014).

2.4 Sources of air pollution within low-income settlements

Emissions in low-income settlements vary both spatially and temporally (DEA, 2019). This variability is highest at the local scale, where there are steep population gradients and high energy use diversity (Lindeque, 2018).

2.4.1 Domestic fuels burning

The highest concentrations of both PM₁₀ and PM_{2.5} in low-income settlements are measured during winter when residents burn more solid fuels (Hersey *et al.*, 2015; Lindeque, 2018). About 95% of low-income residents in third-world countries burn solid fuel for cooking and space heating, predominantly in winter (Thabethe *et al.*, 2014; Nkosi, 2018). More than 81% of the population in Africa depend on biofuels as their primary source of energy (Nyika *et al.*, 2020). Approximately 85% of South Africans have access to electricity but many low-income households cannot afford to use it as their only energy source (Language *et al.*, 2016; Stats SA, 2018; DEA, 2019). These households supplement their energy needs by burning solid fuels for cooking and/or heating purposes (Nkosi, 2018). Solid fuels are cost-effective alternatives and are often more convenient as the same appliance is used for cooking and space heating simultaneously (Balmer, 2007; Naidoo *et al.*, 2014).

Studies show that energy use between households is highly variable and in houses where solid fuels are burnt, the amount of fuel being used varies significantly temporally and spatially (Scorgie *et al.*, 2003, Nkosi, 2018). The variability in the use of energy is due to several factors including fuel accessibility and availability, season, location, social preferences, type of house (formal house or informal house), and income (Scorgie *et al.*, 2003; Langerman *et al.*, 2015; Nkosi, 2018). This explains the complexities in estimating domestic burning emissions. A consequence of this is the significant variation in the emission factors calculated in different studies (DEFF, 2020; Mkhonto, 2014; Makonese, *et al.*, 2015; Nkosi, 2018).

A range of fuel sources is utilised during domestic burning events in South African townships (Nkosi, 2018). The two common energy sources used in these areas during winter are coal and wood (Naidoo *et al.*, 2014), while other fuels, for example, electricity, paraffin and liquid petroleum gas (LPG) are also used alternatively or in combination with the former in informal settlements (Naidoo *et al.*, 2014, Nkosi, 2018). Gas is not a common choice for cooking purposes because it is costly when compared to other fuels, while paraffin is the most preferred source among completely unelectrified settlements, as it can be used for both cooking and lighting through paraffin lanterns (Naidoo *et al.*, 2014). Low-income residents tend to use coal and wood due to their multi-functional nature as these fuels can be used for cooking, lighting functions and space heating purposes simultaneously (Scorgie, 2003; Naidoo *et al.*, 2014; Nkosi, 2018).

2.4.2 Waste burning

Waste burning contributes considerably to PM and the release of a variety of chemical compounds, ash and toxic gases. Waste is defined as “an unavoidable by-product of most human activities” for example, paper, plastic, food leftovers and yard trash among other refuse

(Nkosi *et al.*, 2013, Das *et al.*, 2018). The estimated amount of domestic waste generated per year in South Africa is 42 million m³ (Nkosi *et al.*, 2013). South Africa generated ~108 million tonnes of waste in 2011 (Stat SA, 2018). It is approximated that 0.7 kg of waste is generated per capita per day, with the generation rate further broken down according to income classes. Low-income residents generate 0.41 kg/person/day and middle income generate 0.74kg/person/day, while high income individuals produce 1.29 kg/person/day (DEA, 2009).

Solid waste management (SWM) has become a major concern, especially in densely populated low-income areas of developing countries (McAllister, 2015). Most municipalities are facing extreme environmental degradation, along with public health risks due to poor waste management and unhygienic disposal practices (Wiedinmyer *et al.*, 2014). Domestic waste burning is a method of disposing of uncollected garbage (Keita *et al.*, 2017). The public and elected officials, on the other hand, are relatively unaware and are ignorant of the significance and impacts of open waste burning (Das *et al.*, 2018; RSA, 2019).

Few emission inventories studies have included open burning of municipal solid waste (MSW), with many including a very crude estimate(Keita *et al.*, 2017; Bhaunju *et al.*, 2012; Wiedinmyer *et al.*, 2014). This is mainly because of the difficulties and large uncertainty in estimating how much waste is really burnt (Nagpure *et al.*, 2015). Sources of uncertainties include (i) emissions factors; (ii) activity parameters such as the fraction of the population that burns their waste and the fraction of waste that is combustible; and (iii) spatial allocation of waste-burning activities (Das *et al.*, 2018).

2.4.3 Fugitive dust emissions

Windblown dust emissions from natural and anthropogenic sources contribute significantly to ambient particulate concentrations (Friedrich, 2009; Liebenberg-Enslin, 2014). Fugitive dust sources include agricultural operations, construction and mining operations, road dust, and windblown dust from vacant land (Countess *et al.*, 2001).

Wind erosion is sensitive to a range of environmental factors; hence, dust emission events are often spatially and temporally variable (Lu & Shao 2001). This is due to many variables that influence the source strength and therefore the dispersion ability of the dust concentrations (Liebenberg-Enslin, 2014).

Particle transport, entrainment and deposition have been identified as the main drivers for dust mobilisation by wind (Countess, *et al.*, 2001). The drivers of dust mobilisation are influenced by soil properties (soil texture, composition and aggregation); meteorological conditions (wind, precipitation and temperature); land surface characteristics (topography, moisture, vegetation and aerodynamic roughness length), land use (farming, mining, residential settlements) and particle

characteristics (particle size, shape and density) (Figure 2.4-1) (Lu & Shao, 2001; Liebenberg-Enslin, 2014).

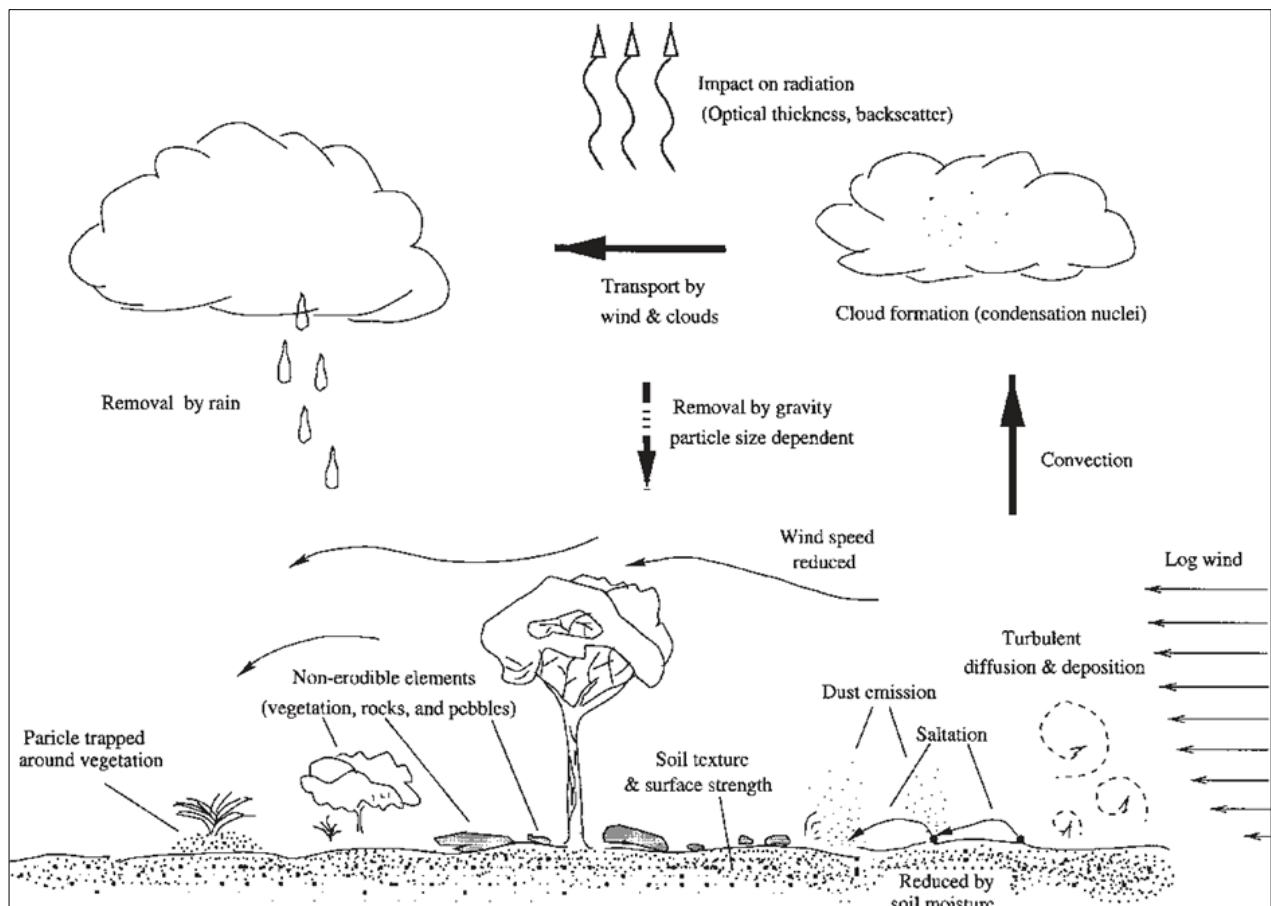


Figure 2.4-1. An illustration of the physical processes which influence dust emission, transport and deposition (Lu & Shao, 2001).

Owing to gravitational settling velocity, (PM_{10}) and ($PM_{2.5}$) deposit at different rates. PM_{10} deposits relatively quickly ($0.5 - 5 \text{ cm.s}^{-1}$), while $PM_{2.5}$ deposits more slowly ($0.05 - 0.2 \text{ cm.s}^{-1}$) (Countess *et al.*, 2001). Source apportionment studies over KwaDela and VTAPA show that on average, dust emissions contribute between ~19% and 24% respectively towards total PM in urban areas (Van der Berg, 2015; DEFF, 2020).

2.4.4 Vehicle emissions

Pollution from road transport is one of the major threats to urban air quality and global warming (Colville *et al.*, 2001). Fossil fuel-powered engines are important workhorses, running bus fleets, large trucks, utility pickups, heavy haulers, graders, passenger vehicles and bulldozers necessary for meeting day to day human needs. However, vehicle exhausts emit a complex mixture of airborne particles that have numerous effects on human health, the environment and materials (Bolaji & Adejuyigbe, 2006). PM emissions from vehicles, particularly in the ultrafine fraction, have

been explicitly associated with endpoints such as oxidative stress and mitochondrial damage (IPCC, 2013).

Considering all major anthropogenic source categories, apart from agriculture, vehicular operations release about 20 – 30% of the total emissions of nitrogen oxides (NO_x), volatile organic compounds (VOCs) and lead (Pb) and more than 60% of the carbon monoxide (CO) (IPCC, 2013). A typical passenger vehicle, for instance, emits about 4.6 tons of CO₂ per year, which varies significantly based on the fuel a vehicle uses, fuel economy, the speed at which one is driving and the number of kilometres driven per year, amongst other factors (US EPA, 2005a).

PM emissions result mainly from mechanical abrasion of the vehicle's brakes and clutch, tyres, corrosion of the chassis, bodywork and other vehicle components and the road surface wear (EEA, 2019). The origins of emissions from vehicles are further shown in Figure 2.4-2 below.

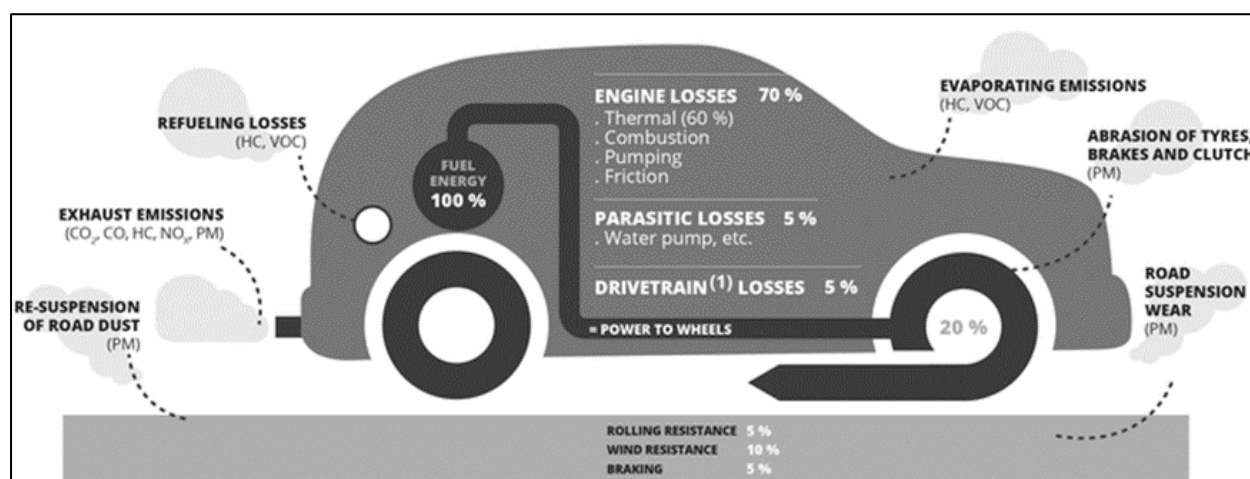


Figure 2.4-2. Sources from which vehicles emit pollution (EEA, 2019).

2.4.5 Biomass burning

Biomass burning (BB) is a major source of gaseous and PM emissions to the atmosphere (Wiedinmyer *et al.*, 2006; Yin *et al.* 2019). Fires tend to have great inter-annual variability; therefore, the resulting emissions are typically variable in time and space.

BB is estimated to contribute approximately 20 to 30% of CO₂ emissions and gases such as hydrocarbons, CO, and NO_x globally (Yin *et al.*, 2019). BB also contributes around 42% of black carbon and 74% of organic carbon compounds that have a significant impact on ambient air quality, human health and the environment (Bond *et al.*, 2004; Michel *et al.*, 2005).

2.5 Emission calculations

Emission estimates are a vital component for efficient and effective air quality management (AQM) (Bachmann, 2007; Chow *et al.*, 2007). Sources of emissions can be divided into mobile,

point and area types, of which different monitoring and measuring methods apply to each one. Directly measuring emissions from all possible sources for all species is practically impossible, therefore, to estimate emissions, models are often used to simulate approximations based on measurements from comparable source types (Watson *et al*, 2012). The most common method is the equation below.

$$ER_{ijk} = EF_{jk} \times A_{jk} \times F_{ij} \quad (1)$$

Where, ER_{ijk} , is the mass emission rate measured as either $\mu\text{g.s}^{-1}$, t.a^{-1} , or other mass per unit time of pollutant i from source type j corresponding to period k . EF_{jk} , represents the emission factor which is the mass rate of emissions (g.unit^{-1} of activity) for source type j corresponding to time k . A_{jk} , is the activity rate associated with the emissions (unit of activity/s) for source type j over corresponding time k (Watson *et al*, 2012; Wiedinmyer *et al.*, 2014).

For PM measurements, F_{ij} represents the fractional quantity of pollutant i in source type j (Watson *et al*, 2012). This means that source profile abundances are normalized to mass emissions from a source in the desired size range and averaged over several tests on similar sources (Watson *et al.*, 2008).

Each of the components of the emission rate (ER) is empirically derived from several tests. The ER is summed over j for period k to obtain total emissions for species i . Air pollution emission characterisation often referred to as “source testing” is performed periodically to determine EF_{jk} (Watson *et al.*, 2012).

According to Chow *et al.* (2007), emission factors and activity rates may change with time (seasonally or time of day), and temperature is an important consideration (cold engine starts, roadway de-icing, evaporating fuels, and solvents). Activity (A) is typically derived from records on amounts of fuel used, vehicle kilometres travelled, population densities and amounts of product produced (Wiedinmyer *et al.*, 2014).

2.6 Pollutants transport phenomena

Pollutants' transport describes the movement of aerosols and species from the site at which they are emitted, resulting in impacts that can be adverse at a distance from the source (Vallero, 2019). Pollutant movement in the atmosphere is initiated by dispersion and deposition (Watson & Kennedy, 1988). Dispersion results from local turbulence, that is, motions that result from the time-averaged wind movement (Watson & Kennedy, 1988; Vallero, 2019). Deposition, on the other hand, is governed by natural processes including scavenging, sedimentation and precipitation which cause downward movement of aerosols, which eventually move the pollutants to the ground surface.

The study aims to assess the applicability of dispersion models; therefore, the focus was on the dispersion transport process.

2.6.1 Dispersion

Dispersion describes the spreading of aerosols and pollutants from a source of discharge in the atmosphere (Vallero, 2019). Several factors influence the dispersion of pollutants including, the height of discharges, species' chemistry, meteorology and the nature of the local terrain (NRC, 2000).

Given the complexity of the movement field, it is impractical to describe velocity as a function of time and space only. Dispersion describes pollutants transportation more realistically as it incorporates the fundamental mechanisms of advection and molecular diffusion. As illustrated in Equation 2, dispersion is treated as a random process where the molecular diffusion coefficient is replaced by a dispersion coefficient in the Fick's first law of diffusion (Kaufmann, 1998).

$$J_{\text{dispersion}} = -\epsilon \frac{\Delta C}{\Delta x} \quad (2)$$

Where; ***J_{dispersion}*** is the dispersion flux density (chemicals per cross-section area per time), ΔC is the change in concentration of molecules across the area, and Δx is the length covered by the pollutants. We subtract (-) because the flux proceeds in the direction of the decreasing concentrations. ϵ is the dispersion coefficient obtained through a combination of empirical data and theoretical equations (Kaufmann, 1998).

2.7 Atmospheric dispersion modelling

Dispersion modelling is a standard technique for evaluating the impact of emissions from different sources on a receptor (Cora & Hung, 2003; US EPA, 2005b). The modelling system uses mathematical equations that describe the atmosphere, dispersion, and chemical and physical processes within the plume to calculate concentrations at various locations (Cora & Hung, 2003; US EPA, 2017; Holmes & Morawska, 2006). Based on the emissions release rate, and the distance of receptor from source and meteorological inputs, the model can simulate concentrations at selected downwind receptor locations (Mtiya, 2013). Such models estimate the reduction in concentration that occurs throughout the dispersion of a pollutant over the modelling area (Mtiya 2013; Cora & Hung, 2003).

United States Environmental Protection Agency (US EPA) air dispersion models have been developed over the years; however, they differ in range, complexity and applicability, hence more complex models are more reliable (Tan, 2014). The Department of Forestry, Fisheries and the Environment (DFFE) recommends SCIPUFF, AERSCREEN, CALPUFF, AERMOD and SCREEN3 as South African standard atmospheric dispersion models, with all being recognised

as US EPA regulatory models (DEA, 2014). AERMOD is a steady-state Gaussian model and is the preferred model for this study as it can be used to demonstrate regulatory compliance in near field (<50 km) studies, and therefore is relevant for intra-urban assessment (Rood, 2014).

2.7.1 Gaussian dispersion model

Gaussian dispersion (GD) models are preferred for estimating the impact of nonreactive air pollutants. They are material balance models for source emissions over a modelling domain (Tan, 2014). The model is considered a statistical model as it shows the “normal distribution” of concentration over 15 minutes or longer periods. Gaussian or normal distribution means that the plume is dispersed in vertical and horizontal directions under specific atmospheric conditions (Holmes & Morawska, 2006; Tan, 2014).

According to Tan (2014), the model does not predict the concentration in a plume at any instant, but rather the statistical distribution of the pollutant concentration about the plume centre line, which is a normal distribution. The normal distribution of the plume is modified at greater distances because of turbulent reflection from the surface of the earth and at the boundary layer depending on the mixing height. The width of the plume is determined by σ_y and σ_z , which are defined either by stability classes or travel time from the source. As illustrated in Figure 2.7-1, the instant plume appears like the shaded area, but the time-averaged concentration may be different from what it appears to be at that instant.

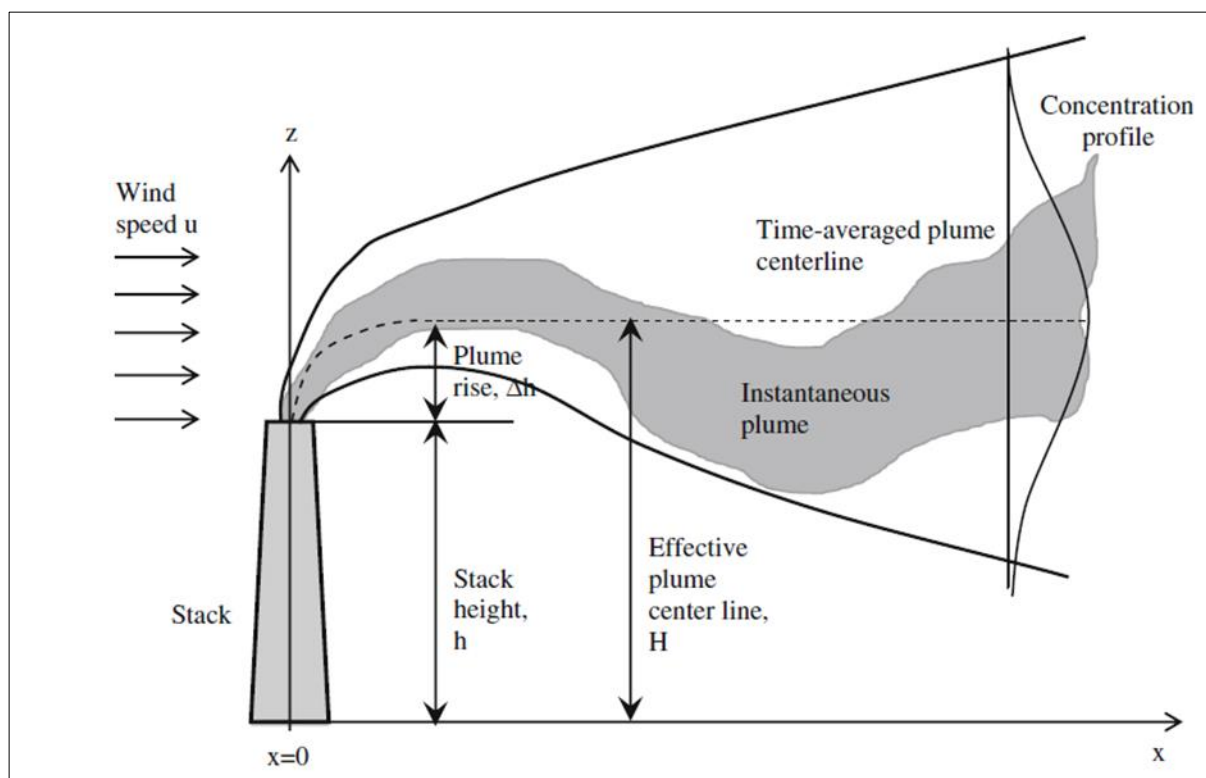


Figure 2.7-1. Schematic illustration of Gaussian dispersion of a plume (Tan, 2014).

GD models assume that a stack with an effective height of H will have a plume rise of $(Dh \frac{1}{4} Hh)$. The emissions are subjected to a crosswind with a speed of u . The emissions move in a downwind direction; therefore, the plume rises from the stack and then travels in the x -direction (Tan, 2014). In summary, GD models describe a three-dimensional emissions concentration field produced by a point source, assuming steady meteorological and pollutants release conditions. The same concept is applied for line and area sources (Daly & Zannetti, 2007).

Several limitations associated with GD models have been identified. One plain limitation of GD models is that they use steady-state estimates which do not take into consideration the time required for the emissions to travel to the receptor. Consequently, aerosol dynamics must be calculated by post-processing management of the results (Holmes & Morawska, 2006). Furthermore, GD modelling systems require one to incorporate chemical modelling to accurately simulate the formation of particles through secondary organic aerosol (SOA) formation (Caputo *et al.*, 2003). This is because the Gaussian equation assumes no interaction between plumes, which is significant when modelling intra-urban environments. The Gaussian equation does not consider the recirculation effects instigated by multiple buildings or at intersections (Holmes & Morawska, 2006).

GD models are not designed for dispersion under low wind conditions or at sites close to the source, for example distances less than 100 m. The models tend to over-predict concentrations in low wind conditions. Furthermore, the models are best suited for short ranges (sites less than 50 km from the sources) due to the Gaussian plume equations adopting a homogeneous wind field, and therefore it is not recommended for far-field modelling as the meteorology is expected to change (Caputo *et al.*, 2003; Holmes & Morawska, 2006).

To overcome the limitations, algorithms have been developed that enable modelling of the chemistry and physical processes within the plume and dispersion around buildings. More so, the effect of wakes from buildings can be achieved by modifying the dispersion coefficients, σ_y and σ_z . To calculate the concentration of pollutants over an urban area multiple source plumes are often used (Tan, 2014). Thus, the different equations used should be based on the nature and heights of the sources and receptors.

Given the limitations, GD models remain widely used due to their characteristics that make them convenient tools. This includes their ability to simulate conservative results that are comparable to monitored data (Holmes & Morawska, 2006). Thus, GD models typically overpredict more than they underpredict concentrations, hence offering a degree of assurance in the regulatory environment when estimating contributions of low-level sources. The models are relatively easy to use as they require a small number of variables and do not need powerful computer resources (Brusca *et al.*, 2016). GD models also offer simple meteorological data requirements, thus

standard measurements can be used as input datasets. In addition, they are quicker compared to numerical models, with the ability to simulate a years' worth of data in minutes (US EPA, 2002).

AERMOD is the Gaussian model of focus for this study, therefore it is discussed in more detail below.

2.7.1.1 AERMOD modelling system

AERMOD software is the result of a collaboration between the American Meteorological Society and the US EPA (AMS/US EPA) (US EPA, 2005b). It is a near-source steady-state Gaussian plume model which is based on planetary boundary layer (PBL) turbulence structure and scaling concepts, applicable to multiple areas on an up-to-date characterisation of the atmospheric boundary layer sources over simple and complex terrain (US EPA, 2002; Masuraha, 2006). AERMOD can be applied to multiple pollutant sources of different natures including point, volume, and area sources (Masuraha, 2006). AERMOD can be applied in modelling buoyant plumes and incorporates a treatment of lofting, where the emissions remain near the top of the PBL before mixing with the convective boundary layer (CBL). Although GD models are limited to simulating concentrations over simple terrain, AERMOD can approximate flows over complex terrain (Holmes & Morawska, 2006). AERMOD is a local scale model which is applied to urban sources at a small regional sphere because it assumes that meteorological conditions are uniform across the modelling domain (Kumar *et al.*, 2017). The model does not include dry or wet deposition of gases and only includes a simple treatment of dry deposition using a reflection algorithm that assumes the change in emission based on measured datasets (Holmes & Morawska, 2006). Treatment of both surface and elevated point sources, area sources, and volume sources in a simple or complex terrain model domain are addressed in the model (Rood, 2014). More so, the model possesses the ability to generate daily, monthly and annual averages of pollutants concentrations (US EPA, 2002).

AERMOD uses inputs from two pre-processors, i.e., AERMET and AERMAP. AERMET provides a meteorological pre-processor for consolidating available climatological data into a format appropriate for use by the AERMOD (US EPA, 2005b). AERMET software uses typical surface parameters and meteorological measurements as its inputs to compute the boundary layer. Parameters and meteorological measurements used in AERMET include roughness, albedo and Bowen ratios, temperature, wind and turbulence (US EPA, 2016a). AERMAP software on the other hand processes terrain data for AERMOD using a layout of receptors and sources. Terrain data can be accessed on commercial sources online and it comes in the form of computer terrain elevation data files. This data may be obtained in numerous map scales and data formats. AERMAP can process several of these standardised data formats and thereafter produce terrain base elevations for each receptor and source, and a hill height scale value for each receptor

(US EPA, 2016b). Figure 2.7-2 summarises the inputs necessary for the running of AERMOD and the outputs thereof.

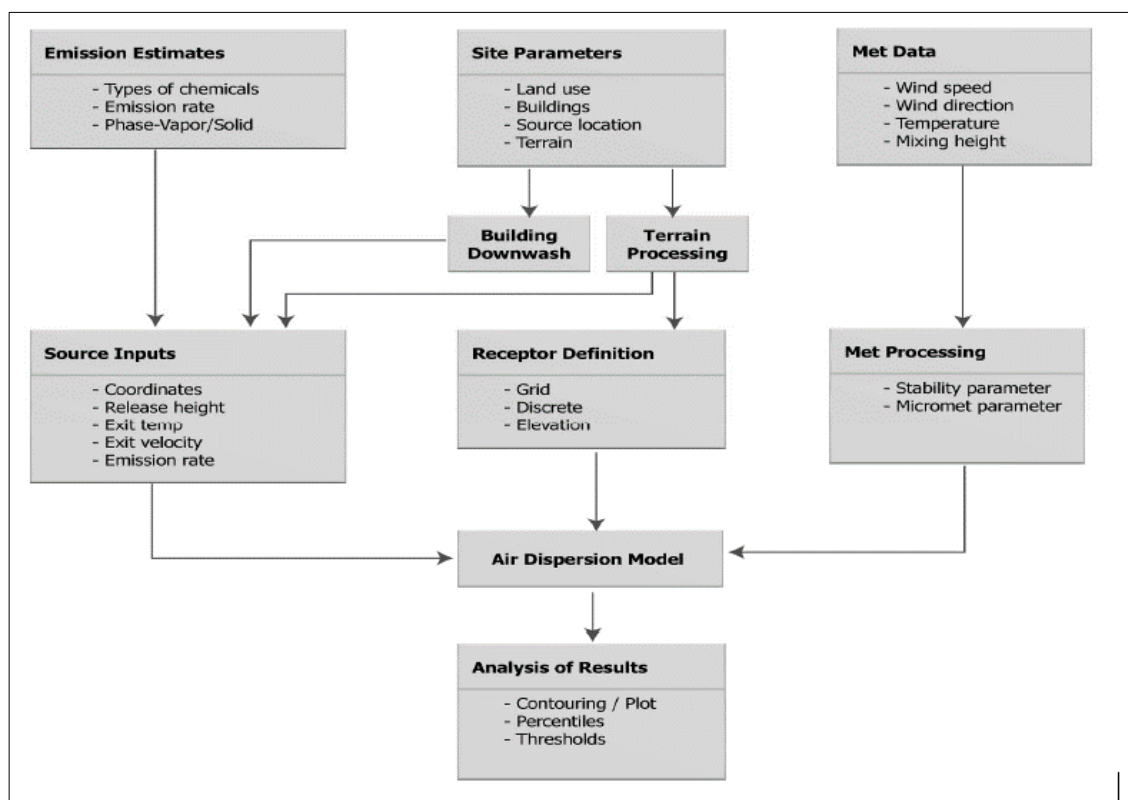


Figure 2.7-2. Inputs and outputs of the AERMOD modelling system (US EPA, 2016b).

The performance and applicability of regulatory models have been investigated in several studies (Perry et al., 2005; Krishna, et al., 2005; Boylan and Russell, 2006; Holmes & Morawska, 2006; Masuraha, 2006; Buthelezi, 2009; Mtiya, 2013; Rood, 2014; Kumar et al., 2017). Perry et al. (2005) did a study using AERMOD and other models on 17 field study databases to evaluate emissions from several individual stacks. The study was done on sites with complex and flat terrain, rural and urban conditions and surfaces with and without structure wake effects. AERMOD estimates were compared with those of other applied models. In model-to-model comparison, AERMOD's performance was superior to that of applied models (Perry et al., 2005).

Buthelezi (2009) used AERMOD to examine the dispersion likelihood of fugitive and stack emissions from Anglo Gold Ashanti's sulphuric acid (H_2SO_4) plant located in Klerksdorp, North-West. The study considered SO_2 stack emissions in the modelling process. From this study, the model proved to be an effective tool in modelling dispersion from stack emissions (Buthelezi, 2009). AERMOD demonstrated satisfactory 1-hour and 24-hour concentrations simulations of the stack emissions from the plant. However, when compared to the monitored ambient air quality, the modelled data was underestimated, which was attributed to the lack of inclusive emission factors for fugitive sources.

A study done by Mtiya (2013) showed that AERMOD performed well on several databases. The aim was to use AERMOD and CALPUFF to simulate Chevron Refinery's (Cape Town, Western Cape) contributions to ground level SO₂ concentrations. The models' predictions were validated against the monitored ambient data over a year. AERMOD predicted 24-hour averages that correlated with the measured values, overpredicting by 9%, while the annual simulations were underpredicted by 11% in Table View and overpredicted by 17% in Bothasig sited. This accentuates that AERMOD is effective over complex terrain and even outperforms the more complex models on several databases. Furthermore, in stable conditions, AERMOD can factor in temperature and wind variations above stack top, while in unstable conditions it accounts for convective updrafts and downdrafts (US EPA, 2005b).

2.8 The regulatory context for air quality and dispersion modelling in South Africa

Air dispersion models can be a useful regulatory tool for air quality management. However, the application of such models in the absence of proper evaluation and regulations may lead to inaccurate and biased conclusions (Manju *et al.*, 2011). Often, models are developed and evaluated based on specific source characteristics and meteorological conditions and then are applied in a different setting. This is the case in South Africa where the recommended dispersion models used were developed by the US EPA (DEA, 2014). With proper application and regulations, these air quality models can be useful in determining facilities' compliance with NAAQS (Table 2.8-1) and other monitoring requirements including New Source Review (NSR) and Prevention of Significant Deterioration (PSD) regulations. Given the dynamics around pollution and air quality management, such guidelines should be revised periodically to ensure that new model developments or expanded regulatory amendments are incorporated (US EPA, 2017).

Table 2.8-1. National ambient air quality standards for PM_{2.5} and PM₁₀.

Pollutants	Averaging Period	Concentrations	Allowed exceedance per year
PM _{2.5}	24-hour	75	4
	Annual	40	0
PM ₁₀	24-hour	40	4
	Annual	20	0

The use of dispersion models has been on the rise in South Africa. However, according to DEA (2014), if such practices operate in the absence of a standardised regulatory approach, it poses several challenges including; (i) air dispersion models being used by industry, consultancies and all levels of government at the discretion of modellers; (ii) assessments to estimate the impacts

of proposed or existing regulated industries on ambient air quality vary in terms of quality; (iii) different regulatory authorities have different requirements in respect to estimating the impacts of proposed or existing regulated industries on ambient air quality and (iv) the choice of the methodology used to estimate the impacts of proposed or existing regulated industries on ambient air quality may be improperly influenced by the desired outcome of estimates. Hence the development of the “Regulations Regarding Air Dispersion Modelling (No. R. 533 of 2014)”.

The Regulations aim to create a “code of practice for dispersion modelling in Air Quality Management (AQM) in South Africa”. Its objectives are to standardise model applications for regulatory purposes and to ensure that dispersion modelling practices in South Africa are undertaken in a compatible form, therefore guaranteeing that results from one dispersion modelling study can be compared directly to another (DEA, 2014). The DEA (2014) argue that achieving the objective will; (i) guarantee that there be consistency and equity in the use of models from procedures, possibilities and limitations of their application, (ii) create confidence and transparency of appropriate model applications, (iii) encourage the application of the best available science in regulatory practices so that models are used as “fit for purpose” tools, and (iv) allow regulators to interrogate the results of dispersion modelling studies in an informed and standard method.

The regulations intend to provide model users with a step by step approach for correct modelling practices to apply when; (i) defining assessment objectives and scope of a modelling study, (ii) choosing the most appropriate model for a task based on the required air quality assessment levels and criteria, (iii) selecting the most appropriate data (emissions, meteorological and terrain data) to put into the model, and (iv) presenting modelling study reports with sufficient information for the regulatory authority to make an informed decision (DEA, 2014).

The Regulations recommend SCREEN3, AERSCREEN, AERMOD, CALPUFF and SCIPUFF as South African standard atmospheric dispersion models (DEA, 2014). AERMOD is discussed further as it was applied for this study.

2.8.1 AERMOD regulations

The DEA, (2014) regulations recommend the AERMOD model for near-source (< 50 km) applications in any terrain type. DEA (2014) states that the model is appropriate for Level 2 assessments, which refers to air quality impact assessment in generic licencing or amendment processes where:

- *The distribution of pollutant concentrations and depositions are required in time and space.*

- *Pollutant dispersion can be reasonably treated by a straight-line, steady-state, Gaussian plume model with first-order chemical transformation. Although more complicated processes may be occurring (i.e., curved plume trajectory), a more complicated model that explicitly treats these processes may not be necessary depending on the purposes of the modelling and the zone of interest. For example, if the area of interest is within 100 m, then curvature effects and chemical transformations may not be critical.*
- *Emissions are from sources where the greatest impacts are in the order of a few kilometres (less than 50 km) downwind.*

2.9 Chapter Conclusion

PM, its two forms (primary and secondary) and size fractions have been discussed in this chapter. Subsequently, the health and environmental impacts of PM were highlighted as previous studies have indicated the significant impacts PM has, especially in low-income settlements. A discussion on air pollution in low-income settlements in South Africa was presented, including an identification of the common sources found in these settlements. A brief outline of how emissions are estimated was provided, leading to the discussion on pollutants transport phenomena. Atmospheric dispersion and GD modelling was defined, which was principal to the discussion on AERMOD. Lastly, the regulatory context for air quality dispersion modelling was summarised, which was then narrowed down to specific guidelines on the application of AERMOD.

CHAPTER 3

This chapter aims to give a comprehensive narrative of the data and methods used in this research. Firstly, a description of the sites, Zamdela and Wedela will be undertaken. A summary of the acquisition of monitored data is provided. The procedure for estimating emissions and how the AERMOD dispersion model was set up is then outlined.

3. RESEARCH METHODOLOGY

The research aimed at evaluating the use of dispersion models as regulatory tools in modelling emissions within low-income settlements to better understand the contributions of local sources in intra-urban environments. This would be achieved through the identification of potential sources and their activities within two townships, estimating the contributions of the sources using emission factors and activity rates and through AERMOD, the dispersion of the emissions was simulated. The study also considered and compared monitored time series against the modelled time series.

3.1 Study areas

Two sites in Wedela and Zamdela were considered. These two sites were strategically selected as they are low-income settlements that are densely populated and have a range of sources impacting ambient air quality.

3.1.1 Zamdela

Zamdela (-26.865989 S, 27.869287 E) (Figure 3.1-1) is in the Free State province, South Africa, close to the industrial area of Sasolburg. Zamdela was established in 1954 by the South African Synthetic Oil Limited (SASOL) company as a hostel township for its African employees who came from various parts of South Africa and neighbouring countries (Moloi, 2018). Zamdela is a low-income residential area located within the administrative boundaries of the Metsimaholo local municipality. Approximately 84% of the municipality is rural and 16% is urbanised, with a poverty rate of 42% (Stat SA, 2011b). The unemployment rate is approximately 43% with the majority of the population (approximately 55%) earning less than R1000 per month including social grants (Stat SA, 2018). Zamdela is a community with an estimated 25067 households residing on approximately 1 172.47 per km² (Stat SA, 2011b). The estimated population of 82339 is located on approximately 17000 residential stands with approximately 5979 formal houses, 5334 residential stands with informal houses, 1282 flats, 560 hostels, 1230 informal backyard houses and 5200 informal houses in open spaces (Wagner *et al.*, 2005; Stat SA, 2011b). It is a significant

site as it is part of VTAPA (NEM: AQA, Act no. 39 of 2004). According to DEFF (2020), Zamdela is one of the hotspots in the VTAPA with the highest PM concentrations (PM_{10} , $76.1 \mu g.m^{-3}$ and $PM_{2.5}$, $31 \mu g.m^{-3}$ - annual averages). This is attributed to the growing densely populated informal settlements, the proximity of the township to the Sasol One site and Lethabo power plants, and relatively cheap and easily accessible solid fuels and diversity of sources (Scorgie *et al.*, 2003). More so, several households in Zamdela use either wood, coal, or a combination of both as a substitute for electricity (DEFF, 2020).

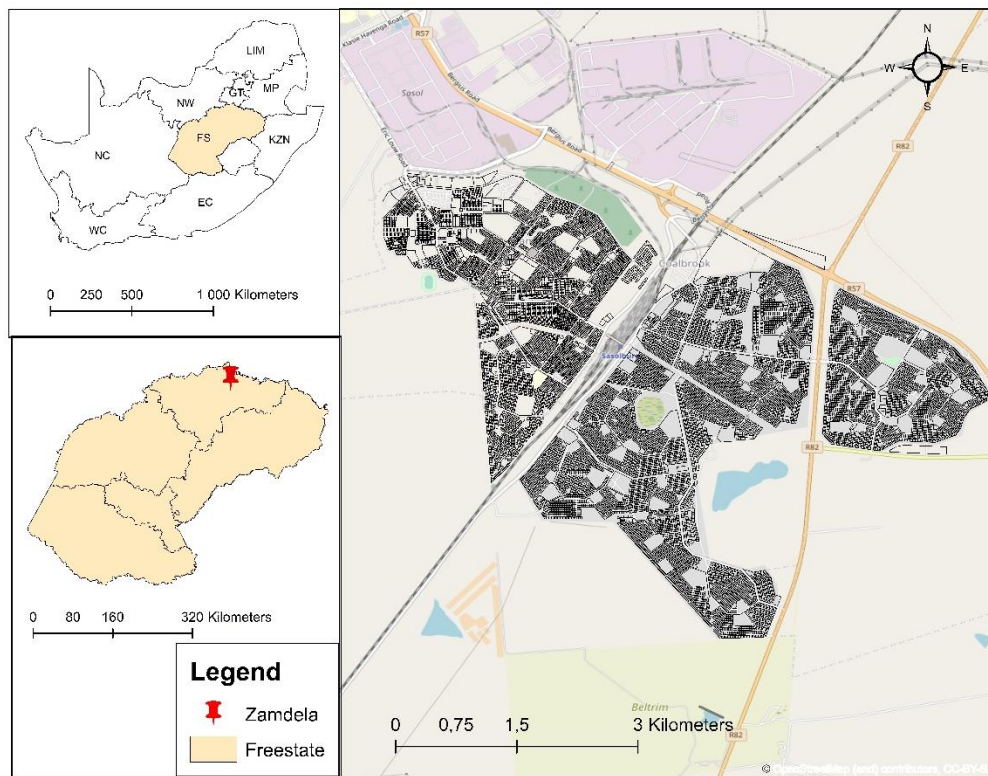


Figure 3.1-1. Map showing Zamdela, Free State province and South Africa.

3.1.2 Wedela

Wedela (-26.470308S, 27.376999E) is situated south of the Gatsrand mountain range within the Merafong City Local Municipality, close to Carletonville, Gauteng (Figure 3.1-2). Owing to the discovery of the gold deposits in the 1930s, the municipality is situated on the South African gold mining belt, hence, Wedela was established as a mining village around 1978 to accommodate mine workers. The settlement is located between the Western Deep Levels and Elandsrand Gold Mine and derives its name from the mines' prefixes; Wed-' and '-ela' respectively (Anon, 2008).

Wedela has a population of around 17928 residing on approximately 2.81 km^2 (Stat SA, 2018). The township is predominantly used for residential purposes, comprising approximately 2300 formal and 1095 informal housing units. Most of the houses are small with approximately 40% accommodating a single member (Stat SA, 2011a, Stat SA, 2018). The socio-economic status of

Wedela is representative of a common mining town. A significant portion of the population is of working age, with 50% of the population earning an annual income of less than R 40000.00, while a small percentage earn a greater annual income (Anon, 2008; Stat SA, 2011a, Otto, 2018).

Approximately 77% of the households reside in formal housing, with 96% making use of electricity for lighting and 98% receiving refuse removal for once a week. Most of the roads within Wedela are formal, owing to the extension of the settlement, where nearly 4.08 km of the roads are unpaved which results in vehicle entrained dust and an increase in PM (Stat SA, 2011a).

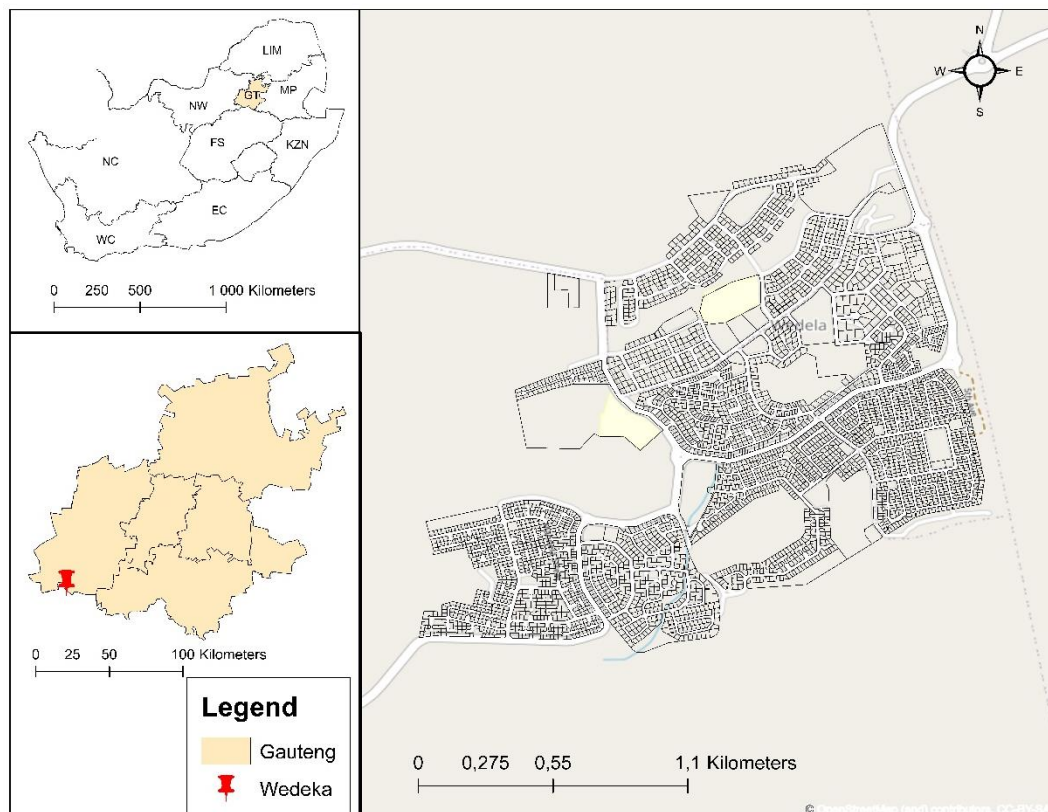


Figure 3.1-2. Map showing the location of Wedela, Gauteng province and South Africa.

3.2 Monitored Data

By tradition, air pollution concentrations are measured at fixed monitoring stations that are mainly built and managed by environmental or governmental authorities (Gwaze & Mashele, 2018). The main advantages of these stations lie in the measurement availability for a variety of pollutants, and the measurement reliability, which benefits long-term pollution estimation (Holmes & Morawska, 2006).

Ground ambient air concentrations of $PM_{2.5}$ and PM_{10} were obtained from the NWU-Anglo monitoring station in Wedela (-26.460073S, 27.384414E), and from the NWU-SASOL monitoring station in Zamdela (-26.852576S, 27.856447E). This data was used for validation with the simulated concentrations.

Ambient PM in Wedela was measured through continuous sampling by two Met One Environmental Beta Attenuation Monitor (EBAM) plus instruments (Figure 3.2-1). The instruments automatically measure and record ambient PM₁₀ or PM_{2.5} using the beta ray attenuation principle. Air is drawn through a PM₁₀ and PM_{2.5} inlet at a constant flow rate of 16 l.m⁻¹. Beta particles are then detected and counted by a sensitive scintillation detector, and a vacuum pump then pulls the measured particulates through a filter taper positioned between the source and detector, allowing for the attenuation of the beta particle signal. Thereby, the degree of attenuation determines the mass concentration of the PM on the filter and the volumetric concentration of PM in ambient air (Met One Instruments Inc, 2021). Met One MSO 232 weather sensors were attached to the two EBAM instruments. This allowed for real-time monitoring of wind speed and direction, temperature, relative humidity and pressure.

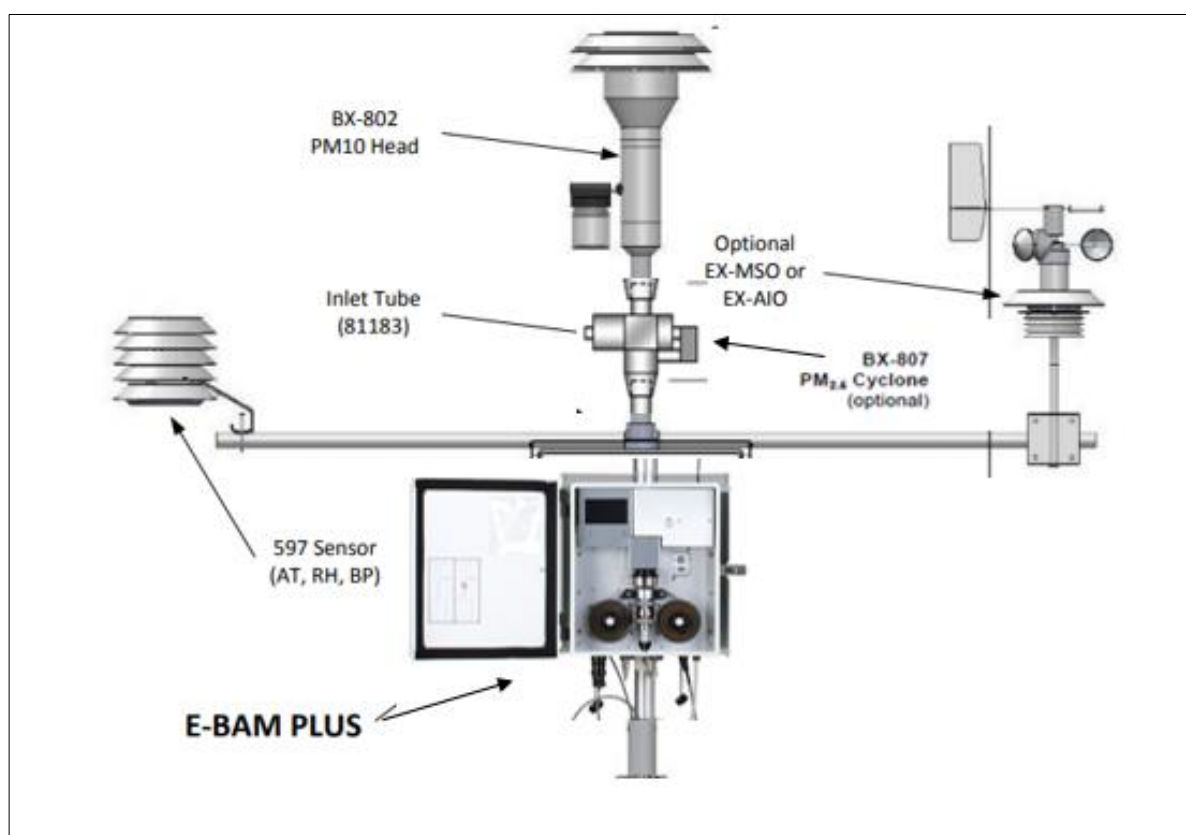


Figure 3.2-1. The standard E-BAM Plus configuration (Met One Instruments Inc, 2021).

The NW-SASOL monitoring station in Zamdela on the other hand is equipped with two Beta Attenuation Monitors (BAM) 1020 to measure ambient PM. In the same way as the EBAM, BAM uses the beta ray attenuation principle. Data at both sites was measured continuously at 1-minute average intervals, recorded on a CR1000 Campbell Scientific data logger, and uploaded for remote monitoring to ECOSTAT (www.ecostat.co.za). The monitoring station is equipped with a fully automated weather station, including the following instruments: RM Young 05103 to

continuously record, rainfall, pressure, wind speed and direction; LICOR LI-200 pyranometer to monitor solar radiation, and Vaisala HMP60 to measure humidity and temperature (Figure 3.2-2).

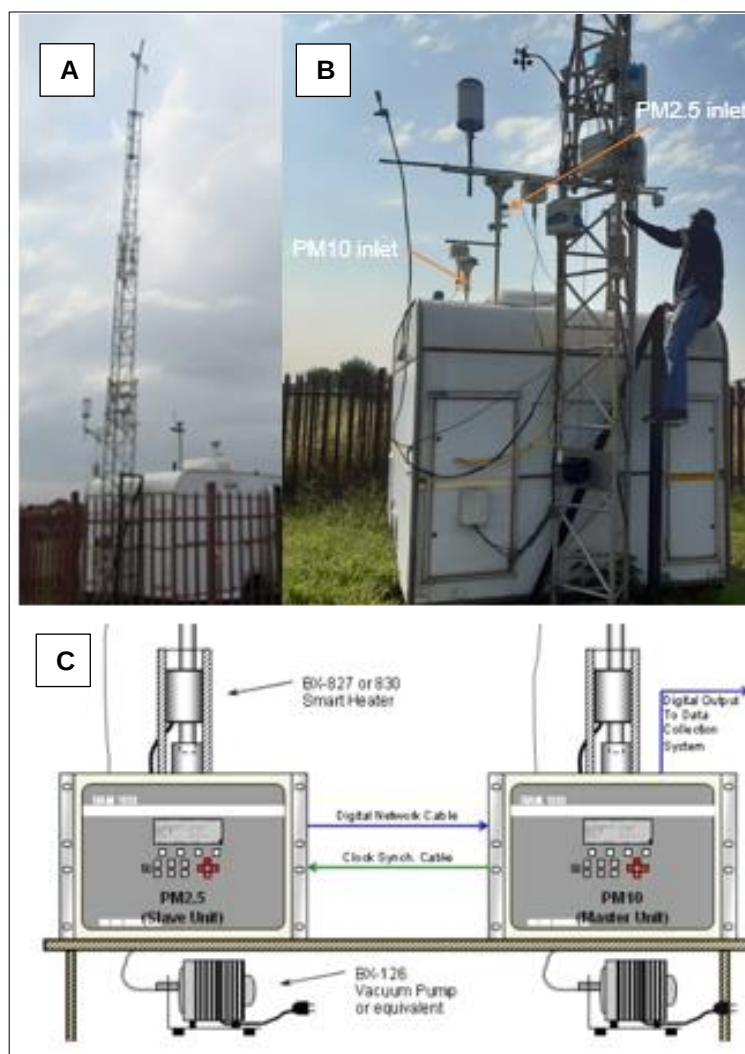


Figure 3.2-2. A: Meteorological tower and cabin housing the ambient sensors at the NWU-SASOL station in Zamdela. B: PM inlets and meteorological sensors. C: Pictorial representation of BAM 1020 PM_{2.5} and PM₁₀ instruments (Met One Instruments Inc, 2009).

3.2.1 Quality control

It is recommended that at least 70% of data be available for quality assurance and data analysis (Belelie, 2018, SANAS 2019). Table 3.2-1 shows that data availability was high except for PM₁₀ in Wedela, however, it met the national annual threshold of 70%, except in 2018. For this study, values that were above and below the instrument's detection limit were flagged and either removed or adjusted to the minimum limit, as suggested by Croghan & Egeghy (2003) or set to be missing depending on the deviation. The same was applied to outliers and random spikes and dips. All values above 1000 $\mu\text{g}\cdot\text{m}^{-3}$ were disregarded, as it is very unlikely that PM levels will exceed 1000 $\mu\text{g}\cdot\text{m}^{-3}$ without first clogging the BAM filter tape, hence the limit is usually set below this value.

Table 3.2-1. Data availability (%) of PM_{2.5} and PM₁₀ concentrations monitored at the respective sites.

Monitoring site	Data availability by Year (%)					
	2018		2019		2020	
	PM _{2.5}	PM ₁₀	PM _{2.5}	PM ₁₀	PM _{2.5}	PM ₁₀
NWU-SASOL (Zamdela)	89.5	78.9	77.3	80.3	75.9	76
NWU-Anglo (Wedela)	89.7	64.1	86.3	70	97.1	71.4

The ambient monitored data was analysed using Python and R through the creation of wind rose diagrams, time series plots, and correlations. The meteorological data were used as input in the AERMET and AERMOD models to determine the atmospheric dispersion potential in the study areas. Ambient monitored particulates were used to validate the modelled data, as the location of the stations were inputted as discrete receptors.

3.3 Identification of sources

Existing data from literature and previous source apportionment projects were used as a guide in identifying potential sources within the study areas. The study also made use of satellite imagery, land use maps, and surveys were conducted in the townships. All the methodologies for source identification have strengths and limitations, hence an integrative approach was used (Belis *et al.*, 2013).

The most direct way of identifying sources is to visually trail potential sources, by tracking the plumes and observing activities. Time was dedicated to visiting the two townships where the researcher observed the possible source locations and frequency of activities that potentially emit PM into the atmosphere. A Global Positioning System (GPS) was used to measure and log the coordinates of sources, and smartphones were used to capture images of the activities. A month-long car count analysis was carried out in both townships. The data was recorded on log sheets which accounted for the vehicle type (sedan, truck, LDVs, mini-bus, motorbikes and others), date, time of day (divided into hours), location of observation (street and coordinates), and the weather (see annexure). The data was statistically analysed to show the temporal variation in the number of vehicles and further used as variability input in AERMOD. The results were compared to other similar studies for validation and to fill in hours not surveyed.

3.4 Estimation of PM emissions

Emissions were quantified for the identified potential sources within the two townships and were further used as input for modelling atmospheric dispersion. Directly measuring emissions from all possible sources for all species is practically impossible, therefore, to estimate emissions, models

are often used to simulate approximations based on measurements from comparable source types (Watson, 2012). Emission rates are a product of activity rates and emission factors.

3.4.1 Emission factors

An emission factor is a variable that specifies the mass of pollutants emitted from an activity normalized by the intensity of the activity (Ho *et al.*, 2019). For example, emission factors for fine particulates from a vehicle source can be expressed in terms of grams (g) of PM_{2.5} emitted per kilometer (km) travelled, or as a ratio of the fuel consumed (Ho *et al.*, 2019). In this study, the emission factors used are based on existing literature that focused mainly on low-income settlements. This is in line with the recommendations of the regulations guiding modelling (DEA, 2014). Different and varied sources were used since there is no individual study that comprehensively covered all the sources deemed potential by this study. According to Mkhonto (2014), the variability of data sources can negatively influence analyses, therefore quality control was necessary. Provided below is how each source contribution was estimated using emission factors.

3.4.1.1 Domestic fuel combustion estimates

The commonly used energy sources in townships are electricity, coal, wood and paraffin (Ludwig *et al.*, 2003; Roberts & Wentzel, 2006; Naidoo, *et al.*, 2014; DEFF, 2020). Emission rates were estimated by multiplying fuel consumption by the emission factor, similar to studies by Mkhonto (2014) and Nkosi (2018) who focused on settlements comparable to this study (Equation 3).

$$E(t.a^{-1}) = EF(g.kg^{-1}) \times BC(kg.HH.yr^{-1}) \quad (3)$$

Where, (*E*) is the emission estimate in tons per year,

(*EF*) represents the emission factor estimated in grams per kilogram of the fuel burned,

(*BC*) is the fuel consumption rate measured as kg of fuel per household per burning period.

The emission factor was adopted from studies by Nkosi (2018), DEFF (2020), Mkhonto (2014), Makonese *et al.* (2015), and Scorgie, *et al.* (2003) and are summarised in

Table 3.4-1 below.

Table 3.4-1. Emission factors for domestic fuel burning by particle size.

Fuel	Emission Factor by particle size (g.kg ⁻¹)	
	PM _{2.5}	PM ₁₀
Wood	13.5 ± 7.2 (a, b, c)	7.5 ± 5.3 (a, b, d)
Coal	10.15 ± 6.7 (b, c, e)	6.5 ± 4.4 (b, c, f)
Paraffin	0.63 ± 0.02 (b)	0.2 ± 0.07 (d)
Gas	0.068 ± 0.01 (a, b)	0.188 ± 0.03 (a)

(a) AP- 42; (b) DEFF, 2020; (c) Makonese *et al.*, 2015; (d) Mkhonto, 2014; (e) Nkosi, 2018; (f) Naidoo *et al*, 2014

The fuel consumption rate was acquired from literature including Roberts and Wentzel (2006); Paulsen *et al.* (2010); Department of Energy (DOE) (2016); Nova Institute (2017); Nkosi (2018); DEFF (2020); Pauw (2020); Census 2011 and Community Survey 2016 acquired from Stats SA. Nkosi (2018) approximated that on average, the household coal consumption is between 4.9 kg to 9.1 kg on a winter day and the daily summer usage is in the range of 3.3 kg and 6.9 kg. This is within the range of household coal use of about 50 kg of coal per week during winter and approximately 20 kg in summer (Scorgie *et al.*, 2003). DEFF (2020) provided the wood consumption estimates which were validated by Kaoma & Shackleton's (2015) study. Estimates of paraffin consumption rates by Roberts and Wentzel (2006) and Paulsen *et al.* (2010) were adopted, with the value converted from litres to kg which is appropriate for emission calculations (a density of 800 kg.m⁻³ was used). Gas consumption data was derived from the VTAPA second edition study (DEFF, 2020) and was narrowed down to the respective townships using Stats SA information. Gas was converted from litres to tons required for emissions estimates using a density of 0.54 kg.l⁻¹ as specified by Afrox. The Community Survey 2016 provided the definite number of households fuel usage for all the sources and it had information narrowed down to small area level (SAL), while the Census 2011 questions offered answers to the multiple fuel types used and different applications of cooking, lighting and space heating.

3.4.1.2 Waste burning emissions estimation

Due to poor service delivery in the township, open waste burning is common in such places. To estimate the emissions from this source, a formula previously used by Wiedinmyer *et al.* (2014) was assumed (Equation 4).

$$E(t.yr^{-1}) = EF(g.kg^{-1}) \times WB(kg.yr^{-1}) \quad (4)$$

Where, (*EF*) represents the emission factor estimated in grams per kilogram of the fuel burned, (*WB*) is the total amount of solid waste burnt in an open space, it is measured as kg per burning period.

Emission factors used to estimate pollutants released from waste burning and the sources used as reference are given in Table 3.4-2.

Table 3.4-2. Emission factors for open waste burning by particle size.

	PM_{2.5}	PM₁₀
Emission factors (g.kg⁻¹)	9.8 (a, b, c)	11.9 (b, c, d)

(a) Akagi *et al.* (2013); (b) Wiedinmyer *et al.* (2014); (c) DEFF, (2020);

The amount of waste open-burned is the most important activity data required for estimating emissions from waste burning. Equation 5 below was used to estimate the total amount of waste open-burned (Wiedinmyer *et al.* 2014).

$$WB(kg.yr^{-1}) = P \times P_{frac} \times MSW_g(kg.person.day^{-1}) \times B_{frac} \times 365(days) \quad (5)$$

Where population (*P*) is the total number of people in the settlement. The population estimates were acquired from Stats SA's Community Survey of 2016 as statistics for subsequent years were not yet available.

(*P_{frac}*) is the fraction of the total population that burn waste. It was estimated as being the fraction of the population whose waste is not collected regularly by collection structures. The refuse removal data used was acquired from the General Household Survey of 2018 (Stats SA, 2018). Approximately 10% of waste generated is not collected in both low-income settlements.

(*MSW_g*) is the annual waste generation per capita. This was attained from findings by DEA (2009) who claim that around 0.41 kg/person/day waste is generated in low-income settlements.

(B_{frac}) is the fraction of the waste that is burnt relative to the total amount of waste exposed to the fire and is related to waste composition. For this study 0.6, as recommended by the IPCC, was adopted based on the findings of Wiedinmyer *et al.* (2014).

(365) is the number of days in a year since the amount of waste burnt is assumed to be yearly.

3.4.1.3 Dust Emission

Suspended dust contributes to particulates that usually exceed NAAQS (Watson & Chow 2000; Hersey *et al.*, 2015). Activities that influence emissions include estimates of bare open spaces, kilometres of unpaved roads, mine tailings, vegetation coverage, and areas under cultivation (Countess *et al.*, 2001). There are a lot of assumptions when calculating emissions from this source. This includes the fact that approximations assume that emissions follow a continuous process of which they are intermittent based on the activity variables and metrological factors (NRC, 2000; Liebenberg-Enslin, 2014). More so, dust emissions depend on particle size, surface conditions, surface loading and the activity causing the suspension of dust (Countess *et al.*, 2001).

Emissions were quantified based on the Airborne Dust Dispersion model from Area Sources (ADDAS) model (Equations 6 to 12) (Marticorena & Bergametti, 1995; Burger *et al.*, 1997). Inputs were set according to Marticorena and Bergametti (1995), Liebenberg-Enslin (2014), and (DEFF, 2020).

$$E (kg.m^{-2}.s^{-1}) = 10^{(0.134c-6)} \times Q \times EF \quad (6)$$

$$Q = C_o \frac{\rho_a}{g} u^{*3} \times (1 + Y) \times (1 - Y) \quad (7)$$

$$Y = \frac{u_t^*}{u^*} \quad (8)$$

Where E is the emissions rate (size category); c is clay content (%); ρ_a is air density; g is gravitational acceleration; u^* is frictional velocity; u_t^* is threshold friction velocity, and C_o is a constant of proportional valued at 2.61. Air density was assumed to be 1.22 kg.cm⁻³ as typical of the atmosphere, and gravitational acceleration of 9.80665 m.s⁻² was used. Friction velocity in the boundary layer is usually expressed in terms of the friction velocity and the height (z) as:

$$u(z) = \frac{u^*}{K} \ln\left(\frac{z}{z_o}\right) \quad (9)$$

Where z_o is the aerodynamic roughness length and K is the von Karman constant (~0.4). While threshold friction velocity was assumed as:

$$F \quad u_t^*(d) = \frac{0.129 \times K}{(1.928 \times RE^{0.092} - 0)^{0.5}} \quad 0.03 < RE < 10 \quad (10)$$

$$K = \left(\frac{P_p g d}{\rho_a} \right)^{0.5} \times \left(1 + \frac{0.006}{P_p g d^{2.5}} \right)^{0.5} \quad (11)$$

$$Re = 1331d^{1.56} + 0.38 \quad (12)$$

Where:

P_p particle density = 2.65 g.cm⁻³ (constant used)

g gravitational acceleration = 9.80 m.s⁻²

d particle diameter = 2.5 and 10 µm

ρ_a is air density = 1.22 kg.cm⁻³

3.4.1.4 Vehicle emissions estimations

Several factors regulate how much a vehicle emits, hence making it difficult to comprehensively estimate emissions from vehicles. Previous studies however agree that vehicle exhausts represent an important source of fine PM emissions (Mazzoleni *et al.*, 2004). Vehicle emissions were calculated as outlined by Colville *et al.* (2001), Thomas (2009), and EEA (2019).

$$E \text{ (t.yr}^{-1}\text{)} = EF \text{ (g.veh}^{-1}\text{.km}^{-1}\text{)} \times VF \times L \text{ (km)} \times 365 \quad (13)$$

Where, (EF) represents the emission factor estimated in grams per kilometre travelled, (VF) is the vehicle flow which gives the information on volume and type of transportation, and (L) is the estimated road length a vehicle travelled.

Emission factors were validated between several studies as outlined in Table 3.4-3. The vehicle categories to be considered are passenger cars, light-duty vehicles, heavy-duty vehicles, and mini-buses.

Table 3.4-3. Vehicle emission factors in (g.veh⁻¹.km⁻¹) by particle size.

Vehicle type	PM _{2.5}	PM ₁₀
Passenger cars	0.0095 ± 0.007	0.013 ± 0.005
Light-duty vehicles (LDVs)	0.015 ± 0.0015	0.019 ± 0.002
Heavy-duty vehicles (HDVs)	0.130 ± 0.019	0.170 ± 0.024
Minibus	0.038 ± 0.004	0.074 ± 0.002

Source: Mazzoleni, *et al.*, (2004); Smit, 2017; Zhao, *et al.*, (2018) and EEA, (2019)

As there are no SANRAL or any available traffic counts in the townships, this study made use of physical traffic counts. Traffic flow was recorded from 06:00 to 18:00 South Africa Standard Time (SAST), which is the standard 12-hour count (Colville *et al.*, 2001). This allowed the study to

capture the early morning rush, off-peak and early evening rush hours. The car counts accounted for the vehicle type (sedan, truck, LDVs, min-bus, motorbikes and others), date, time of day (divided into hours), location of observation (street and coordinates and the weather).

GIS roads datasets and google maps data were used to estimate the lengths of the roads within the two townships. The attribute tables of the GIS data also provided information on the type of road (residential, tertiary and highway) which is important when assigning emissions factors. Since it is an intra-urban study, it assumed the general speed limit in terms of the “South African National Road Traffic Act, 1989 and its regulations” of 60 km.h⁻¹ on a public road within an urban area.

3.4.1.5 Biomass burning

Biomass burning is a major source of gaseous and particulate matter emissions to the troposphere (Wiedinmyer *et al.*, 2006; Yin *et al.* 2019). Pollutant emissions (*E*) were calculated as the product of dry mass burned (*M*) and a corresponding emission factor (*EF*) as expressed by equation 14.

$$E(t.yr^{-1}) = EF(g.kg^{-1}) \times M(kg.yr^{-1}) \quad (14)$$

Emission Factors were obtained from previous studies, as summarised in Table 3.4-4. Land use maps were used to identify the type of vegetation cover in the areas.

The mass of dry matter burnt is estimated as outlined in Equation 17 below.

$$M = A \times B \times CE \quad (15)$$

Where, (*A*) is the area exposed to fire,

(*B*) is the fuel loading or the dry matter density (mass of biomass per area), and

(*CE*) is the combustion efficiency (fraction of biomass burnt).

Table 3.4-4. PM Emission factors for each vegetation cover type (unit: g.kg⁻¹).

Category	Land-use type	PM _{2.5}	PM ₁₀
Forest	Dominated by evergreen conifer trees (canopy >2m). Tree cover >60%	12.7	13.1
Forest	Dominated by evergreen broadleaf and palmate trees (canopy >2m). Tree cover >60%.	10.2	12.8
Forest	Dominated by deciduous needle leaf (larch) trees (canopy >2m). Tree cover >60%.	12.7	13.1

Forest	Dominated by deciduous broadleaf trees (canopy >2m). Tree cover >60%.	12.3	12.8
Forest	Dominated by neither deciduous nor evergreen (40-60% of each) tree type (canopy >2m). Tree cover >60%.	11.7	12.9
Shrubland	Dominated by woody perennials (1-2m height) >60% cover	7.9	8.5
Shrubland	Dominated by woody perennials (1-2m height) 10-60% cover.	7.9	8.5
Grassland	Trees cover 30-60% (canopy >2m).	7.9	8.5
Grassland	Trees cover 10-30% (canopy >2m).	7.1	9.2
Grassland	Dominated by herbaceous annuals (<2m).	7.1	9.2
Cropland	At least 60% of the area is cultivated cropland.	5.0	6.3
Cropland	Mosaics of small-scale cultivation 40-60% with the natural tree, shrub, or herbaceous vegetation.	7.9	8.5

Sources: Andreae & Merlet, (2001); Streets *et al.*, (2003); Michel *et al.*, (2005); Wiedinmyer *et al.*, (2006), Akagi *et al.* (2013).

Identification of fire activity for this study was obtained from satellite drivers, including the Moderate Resolution Imaging Spectroradiometer (MODIS) Burned Area Product and the Land Cover Characteristics datasets. Data for Southern Africa was obtained in Shapefile format from (ba1.geog.umd.edu) which enabled the researcher to narrow it to the desired townships in ArcGIS. The fuel loading and combustion efficiency were extracted from a study by Van Leeuwen *et al.* (2014) and are summarised in Table 3.4-5.

Table 3.4-5. Fuel load (B) and combustion efficiency (CE) for the different vegetation categories

Vegetation category	Fuel load (t.ha⁻¹)	combustion efficiency %
Dicots	0.4 (0.5)	91 (12)
Grass-dormant	1.9 (1.4)	93 (14)
Grass-green	0.4 (0.2)	88 (23)
Litter	2.1 (0.5)	88 (13)
Tree/shrub leaves	0.4 (0.8)	64 (12)
Woody debris (0–0.64 cm)	0.6 (0.7)	65 (16)
Woody debris (0.64–2.54 cm)	0.9 (1.0)	39 (25)
Woody debris (> 2.54 cm)	1.0 (1.1)	21 (12)
Crop residue	8.3 (9.9)	75 (21)
Source: Van Leeuwen <i>et al.</i> , (2014)		

3.5 Atmospheric dispersion modelling

Dispersion models are tools that can be used to simulate ambient concentrations as a function of meteorological characteristics, emission estimates, and source configurations (US EPA, 2005b). Using this tool allows for the replication of spatial and temporal variations in the ground level concentrations coming from the emissions of several possible sources (Mtiya 2013; Cora & Hung, 2003). Increasing confidence is being put on low-level air pollution concentration estimates from modelling as the principal basis for environmental and health impact assessments. Therefore, attention was put into selecting a suitable model for this study. DFFE and US EPA approved AERMOD modelling protocol was selected for use in this assessment study. Ministry for the Environment Manatū Mō Te Taiao (2004) gives an outline of the modelling process which was followed in this study (Figure 3.5-1).

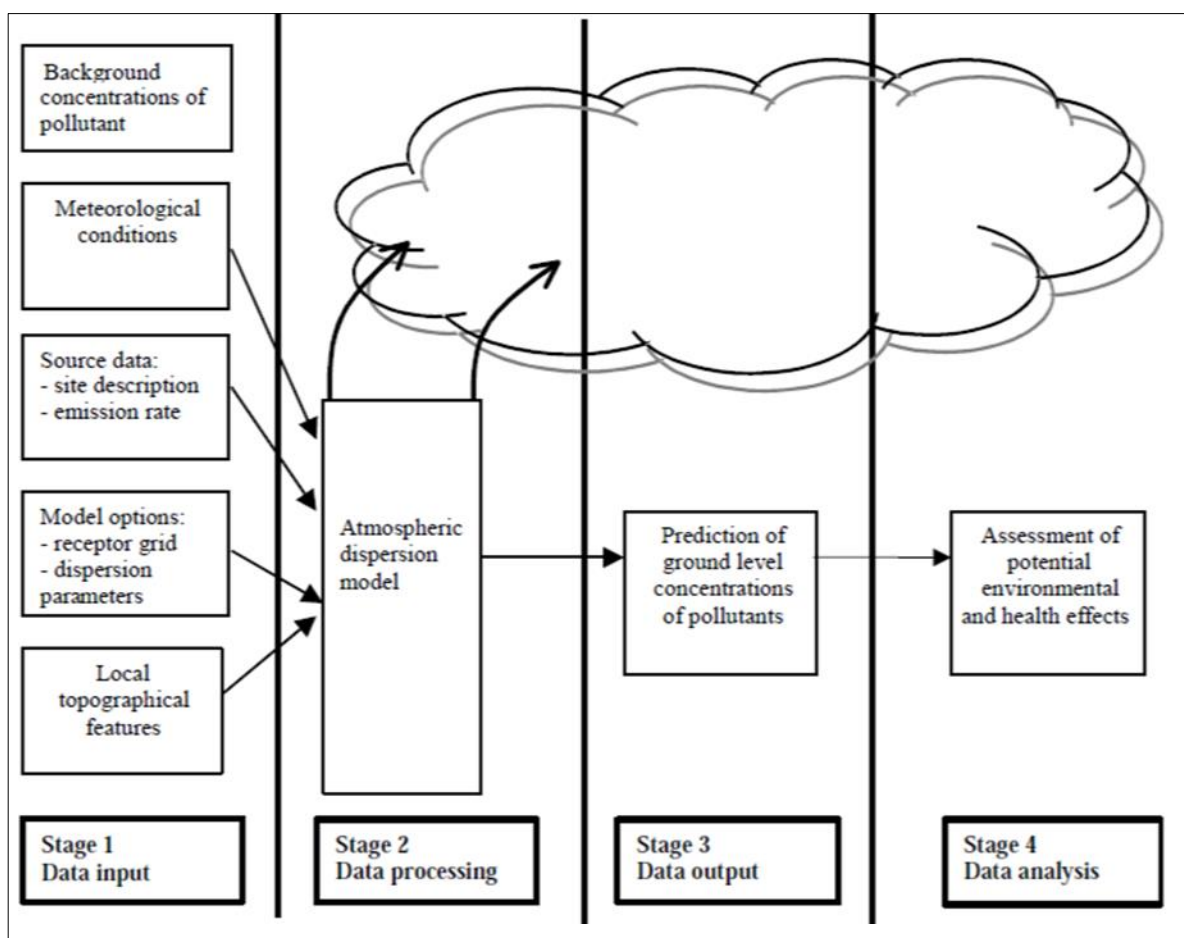


Figure 3.5-1. Overview of air dispersion modelling (Ministry for the Environment Manatū Mō Te Taiao, 2004).

3.5.1 AERMOD system

The modelling system comprises of the AERMOD Dispersion model (US EPA, 2005b) and requires two input data processors that are regulatory components: AERMET, a meteorological data processor (US EPA, 2016a), and AERMAP, a terrain data processor (US EPA, 2016b).

Additional non-statutory components of this system include AERSURFACE, a surface appearances processor, and BPIP-PRIME for handling data from structures and obstacles near emission sources to define their interference with plume rise and to approximate the variables required by AERMOD to evaluate building downwash effect (US EPA, 2005b).

3.5.1.1 AERMET

AERMET defines the ONSITE trail for recording hourly surface observations acquired from a site-specific meteorological station. AERMET can process three forms of data inputs; (i) hourly surface data, (ii) upper air meteorological normally collected by the National Weather Service (NWS), and (iii) data can also be collected from a site-specific prognostic meteorological data processed through a processor such as the Mesoscale Model Interface (MMIF) (US EPA, 2017). It also estimates the mixing height in the CBL, considering its reliance on both mechanical and convective processes. AERMET estimates the mixing height based on the following standards:

- i. Throughout the day, when the Monin-Obukhov Length is negative, it is projected as higher than the convective or the mechanical mixing height.
- ii. During the night, when the Monin-Obukhov Length is positive, it is estimated as equal to the mechanical mixing height.

AERMET data processing occurs in three discrete stages, each required to be run separately as shown in Figure 3.5-2 (US EPA, 2017). The first stage extracts the surface and upper-air data from files into specific archive formats that are appropriate for AERMOD. It is also at this stage that the quality assurance (QA) of the surface, upper air, and site-specific data is considered. The second stage combines the processed surface and upper-air data with the site-specific data into discrete 24-hour periods and writes the merged data to an intermediate file. The final stage reads the merged data file, calculates the boundary layer parameters (e.g., surface friction velocity, mixing height, and Monin-Obukhov length) required by AERMOD, and generates two AERMOD ready meteorological data files. The first file contains the observed surface parameters (e.g., temperature, wind speed, and wind direction) while the second file contains a profile of winds, temperature and the standard deviation of the fluctuating components of the wind if provided (US EPA, 2017).

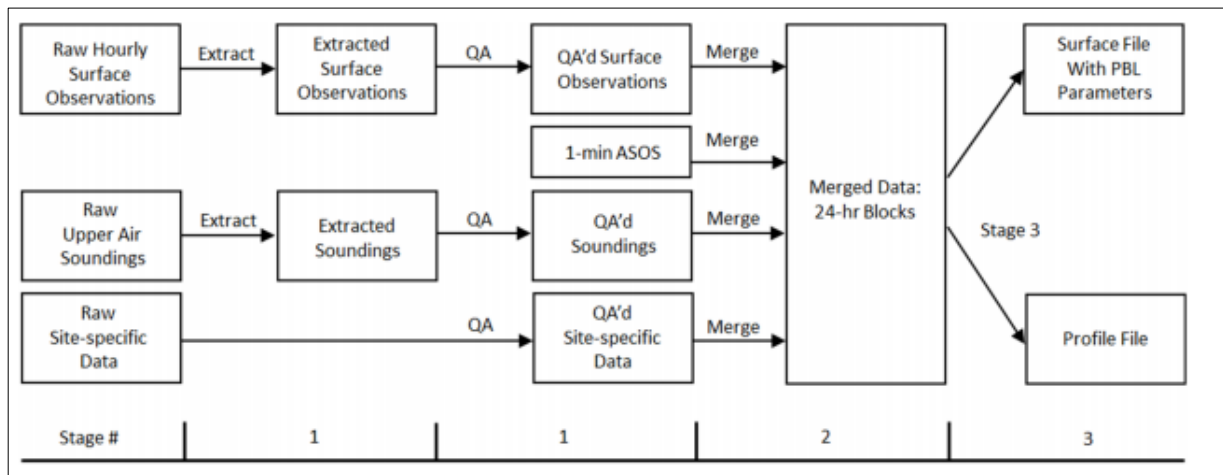


Figure 3.5-2. AERMET processing framework (US EPA, 20016a).

3.5.1.2 AERMAP

AERMAP was established to process terrain data in combination with a layer depicting receptors and sources. It can process several standardised data formats, which makes it possible to produce terrain base elevations for specific receptors and sources as well as a hill height scale value for each respective receptor (US EPA, 2005). With the notion that terrain influences air quality concentrations at distinct receptors, AERMAP helps to define the base elevation at each receptor and source. Thus, AERMAP assumes the terrain height and location that has the greatest influence on dispersion for each receptor. This height is referred to as the hill height scale (US EPA, 2005).

One of the main limitations for the use of AERMAP in developing countries is the availability of a digital elevation model (DEM) containing topographical data of the modelling domain with an adequate resolution (US EPA, 2005). Therefore, online free DEM sources are mostly preferred when running AERMAP in cases where a local one is not readily available. For this study, the topographic data required by AERMAP was obtained from Shuttle Radar Topography Mission (SRTM1), whereas the land cover data was attained from WebGIS, with a resolution of ~30 m. The World Geodetic System 1984 (WGS84) projection datum was used, with the terrain data calculated based on the Universal Transverse Mercator (UTM) coordinate system (35 South). The receptors' grid was set at 50m by 50m over the two townships.

3.5.1.3 AERMOD

The execution of the AERMOD dispersion model, in addition to the previous steps, included:

- i. Application of the control options:
 - Regulatory options were set on default with concentration selected as output type.

- Selection of the averaging periods, thus hourly, daily and annual were opted for given the pollutants established NAAQS and analysing needs.
 - Some default output options used include the concentrations and depositions at each receptor due to emissions from each group of sources, the average during long periods (annual or whole period), and the maximum in each short averaging period (PLOT-FILE option). Secondly, concentrations higher than the threshold values due to emissions from each group of sources in each short averaging period (MAXIFILE option). Lastly, all concentrations at each receptor in each short averaging period (POST-FILE option); for substantiation against monitored data.
- ii. Application of methodologies to produce the input data required by the model for the description of emission sources, which involved:
- Defining the pollutants emitted and the size of the particulate, and in this study PM₁₀ and PM_{2.5} were considered.
 - Determining how to best input the sources based on the data available, as well as the corresponding release height, velocity, and diameter where applicable.
- iii. Deciding on locale, which included urban and rural conditions, as given by the Stats SA definition of townships.

Figure 3.5-3 summarises the processes and data flow stages required to successfully perform air dispersion modelling analysis using AERMOD.

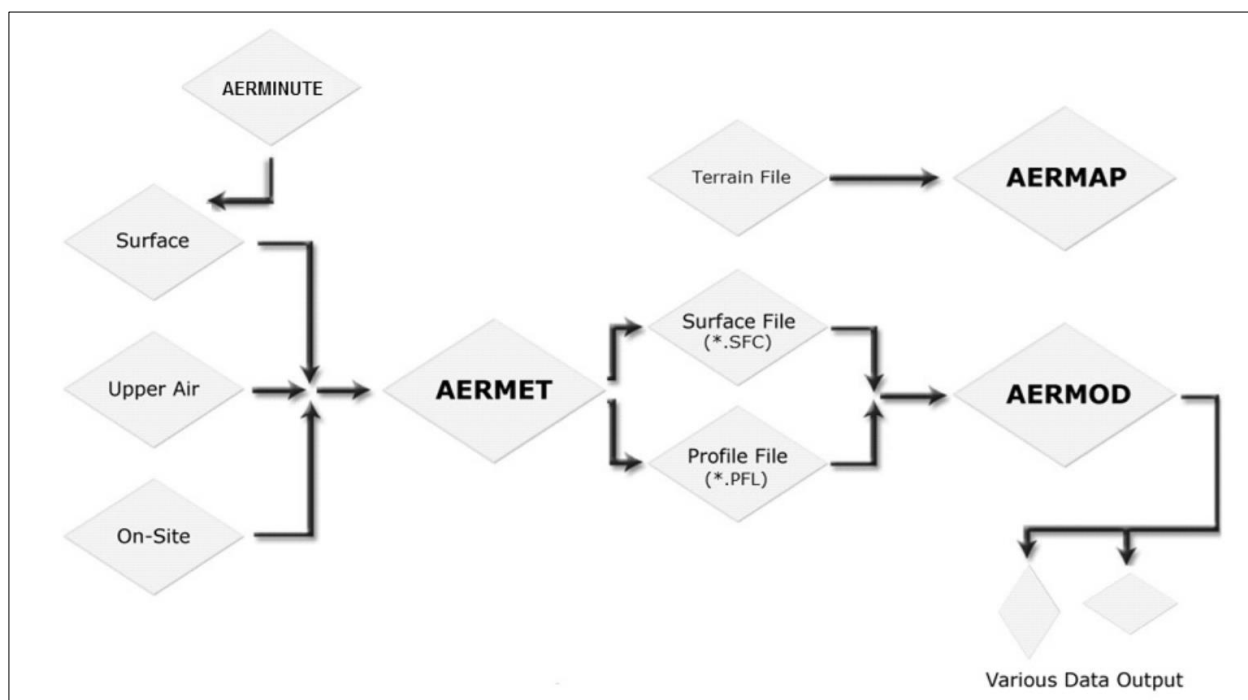


Figure 3.5-3. AERMOD modelling system's data flow (Villalvazo, Davila & Reed G. 2007)

3.5.2 Input data

3.5.2.1 Modelling domain and receptors

In this study, AERMOD was set up and run for Zamdela and Wedela as illustrated in Figure 3.5-4 and Figure 3.5-5. The size of each domain was determined by the aim of the study, which is to understand intra-urban particulate concentrations, thus they covered the area of each township in question. The grid cell size selection was determined by the desire to define terrain and meteorological variations on a high resolution and considering the computation time, it would take to model. To provide a detailed simulation of the localised impact of pollutants the receptors' grid was set at 50 m resolution as recommended by DEA (2014) which was overlaid by plant blanking boundaries outlining the townships. Further separate sensitive discrete receptors were placed in areas of concern including schools, bus ranks and hospitals (DEA, 2014).

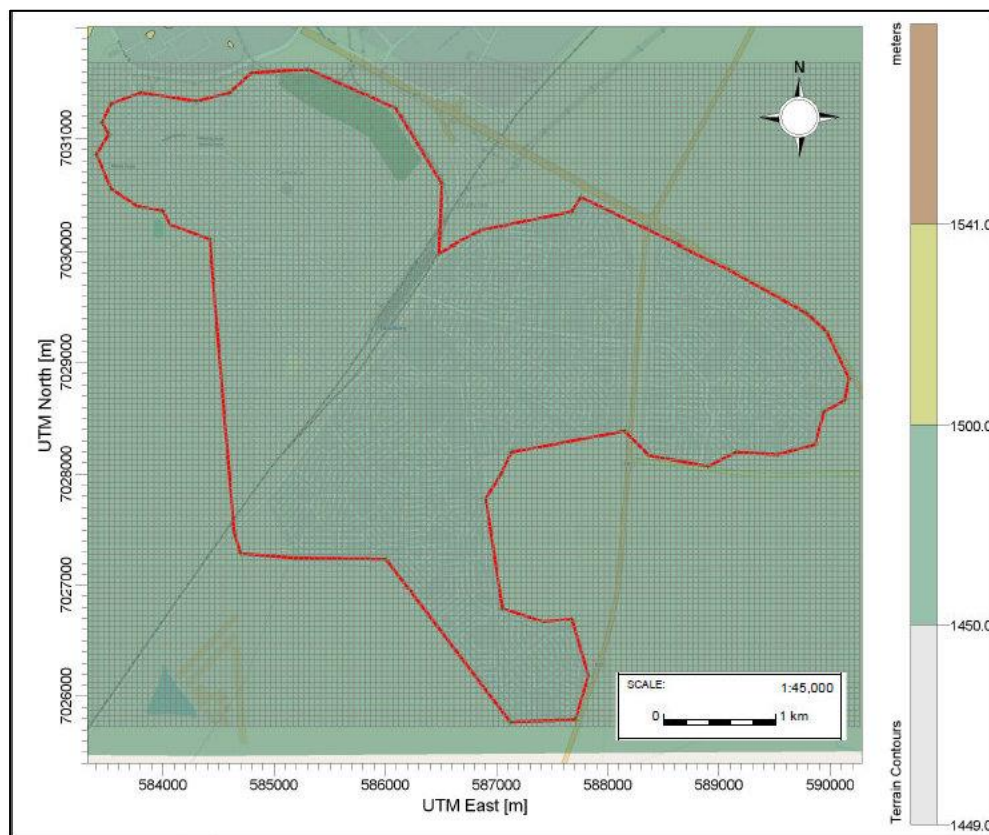


Figure 3.5-4. Modelling domain assumed for Zamdela overlaid by the uniform receptor.

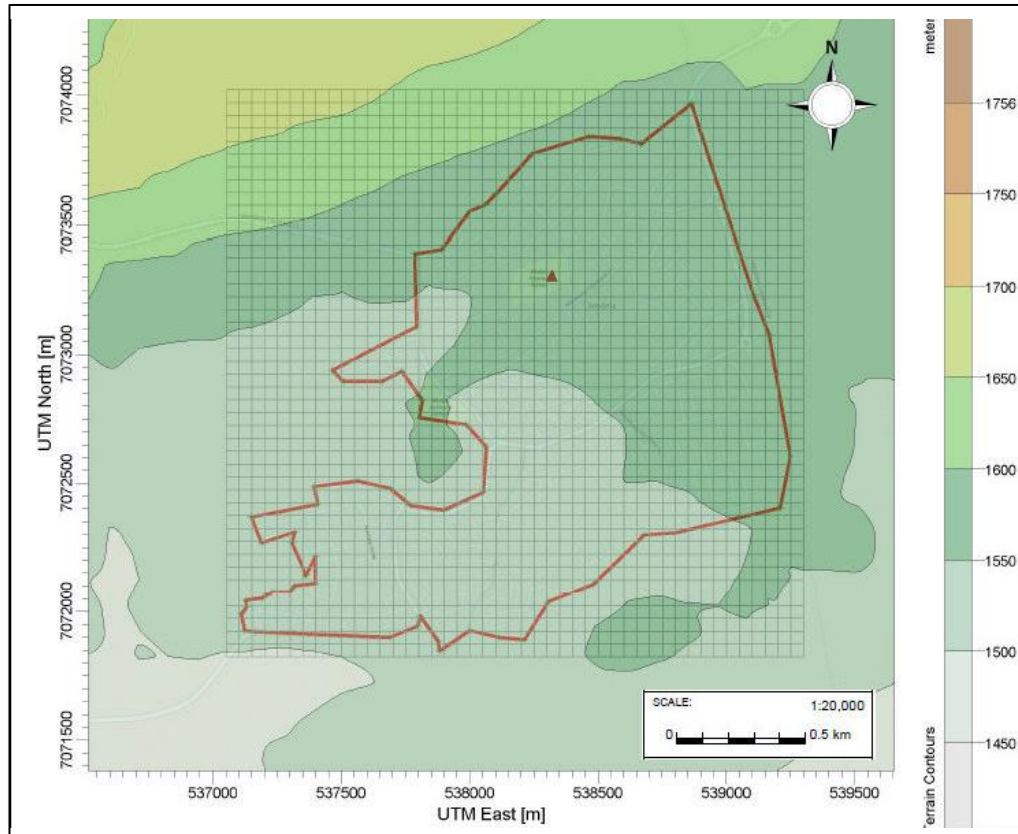


Figure 3.5-5. Modelling domain assumed for Wedela overlaid by the uniform receptor.

3.5.2.2 Sources

AERMOD modelling system supports a range of source types which allows for the characterization of most emissions within an area. The following sections summarise the source types and their input requirements that are used in this study.

Area sources are used to model low level or ground level releases where releases occur over an extent (e.g., landfills, storage piles, waste burning, and biomass burning). AERMOD models allow the definition of rectangular areas which can also be rotated at an angle specified relative to a north-south orientation, along with a range of other shapes (US EPA, 2005b). For this study domestic burning was assumed to be from multiple area sources, thus each house that was identified as a potential source was inputted likewise. Biomass burning was also assumed to be an area source with the patches identified by Modis imported to AERMOD. The same was applied to waste burning sites and dust sources. The figures depicting the sources are shown in the result section. For area sources; central locations, dimensions, emission rate, release height, and coordinates were the main variables used in simulating the sources.

Examples of line sources are roads and rail lines. AERMOD does not have a default line source type (US EPA, 2005b). However, it can simulate line sources through a series of volume sources as described in *Volume II of the U.S. EPA User's Guide for the Industrial Source Complex (ISC3)*

*Dispersion Models*²⁵”, hence roads were assumed to be line volume sources. This source type requires emission rates, configuration (set as adjacent), plume height and width, start and end coordinates, release and base height.

All the source input files used for this study can be accessed on <https://nextcloud.nwu.ac.za/index.php/s/R6xya86CXyjYwnC> as the dataset was too large to add on this draft.

3.6 Chapter conclusion

The chapter provided a detailed description of the study areas, data and methods used in this research. Different data gathering instruments, techniques and procedures followed to analyse the data are concisely defined. Accompanying formulas and calculations are also provided and explained. The atmospheric model used to study the dispersion of PM over the townships is described as comprehensive as possible. Thus, the chapter presented data analysis plans for different data frames to meet the four study objectives.

CHAPTER 4

This chapter presents the results of the characterisation of ambient monitored data within Zamdela and Wedela. In analysing ambient data, only readings that were within the reasonable level of accuracy between 2018 and 2020 were utilised. This is important in understanding the meteorological setting and background concentrations within the towns, hence showing why the study areas are important to research. A meteoroidal overview of the two towns is summarised first, with the following section describing the temporal variations in PM at the monitoring stations.

4. MEASURED AMBIENT DATA WITHIN THE STUDY AREAS

4.1 Synoptic circulation patterns over South Africa

In order to understand the microscale circulation patterns over an area it is important to reflect on the meso-scale patterns governing the area.

Southern Africa is located in the region of large-scale subsidence occurring between the Hadley and Ferrell cells of the southern hemisphere’s general circulation (Liebenberg-Enslin, 2014).

Circulation over the region is governed by the systems that originate from the tropics in the north, and temperate latitude to the south (Walton, 2021). Five major synoptic-scale circulation patterns dominate over southern Africa (continental anticyclone, ridging anticyclone, westerly trough, cut-off low and tropical easterly disturbances), of which the most predominant is the semi-permanent subtropical continental anticyclone (Tyson and Preston-Whyte, 2000, Walton, 2021).

Anticyclonic conditions are most prevalent during the winter months, occurring with a frequency of 70 – 80% during June and July. Easterly waves occur during summer while ridging anticyclones occur frequently throughout the year. Westerly disturbances have a semi-annual cycle, peaking in the spring and autumn months while continental anticyclones are the single most frequently occurring systems from March to September (Tyson et al., 1996, Liebenberg-Enslin, 2014).

Based on the major circulation patterns dominating over southern Africa, five aerosol and trace gas transport modes have been identified, thus, direct easterly or westerly transport, easterly or westerly advection out of an initially anticyclonic circulation and anticyclonic recirculation (Figure 4.1-1). Of these, anticyclonic transport to the east coast of South Africa and the Indian Ocean is the dominant transport mode ((Tyson et al., 1996, Walton, 2021).

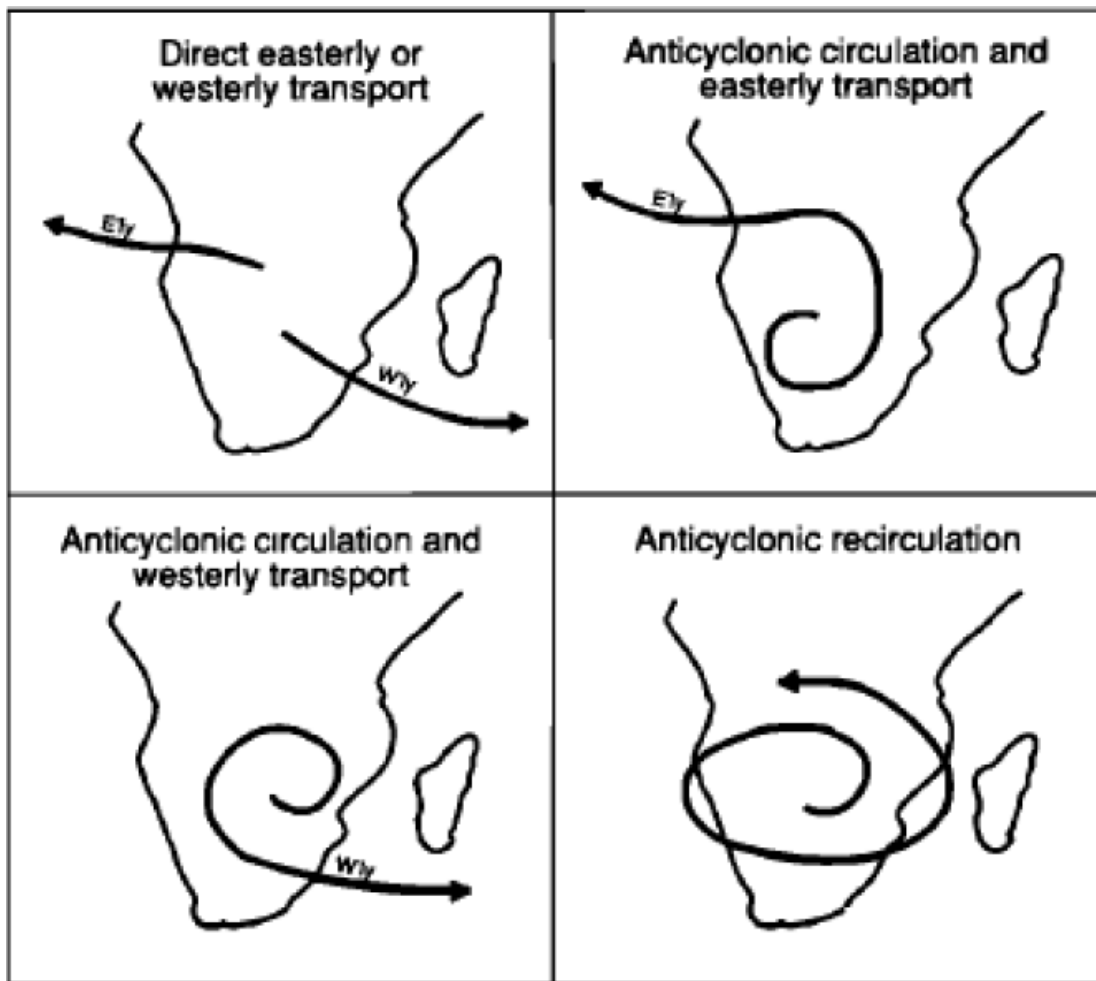


Figure 4.1-1. Schematic representation of the major aerosol and trace gas transportation over south-Africa (Walton, 2021).

4.2 Meteorological overview of the study area

Meteorological parameters that typify the microscale transport potentials of an area include wind speed and direction, ambient atmospheric temperature, precipitation, and relative humidity among others.

4.2.1 Wind Field

Wind is a significant variable in the dispersion, transportation and eventual removal of pollutants. Thus, the speed of wind governs both the distance of downwind transport and the rate of pollutants' dilution. The stronger the wind speed, the faster the dilution of pollutants, therefore lowering the concentrations at ground level (Vallero, 2019). Weaker wind speed means less dilution of pollutants occurs resulting in higher concentrations at ground level (Thomas, 2009).

The wind roses provided for the respective monitoring stations consist of twelve cardinal wind directions drawn in the blowing from vector, with the frequency of wind indicated with the dotted circles. Each circle indicates a 5% frequency of occurrence.

Figure 4.2-1 displays the annual average wind roses for Zamdela and Wedela respectively. Zamdela has prevailing easterly winds with speeds between 4 and 8 m.s⁻¹ occurring for ~15% of the time. Wedela experiences light and variable winds with a mean of ~2 and 5% of the time recording calm winds. The wind in the area frequently blows from the south westerly sector with prevailing speed between 6 and 6 m.s⁻¹.

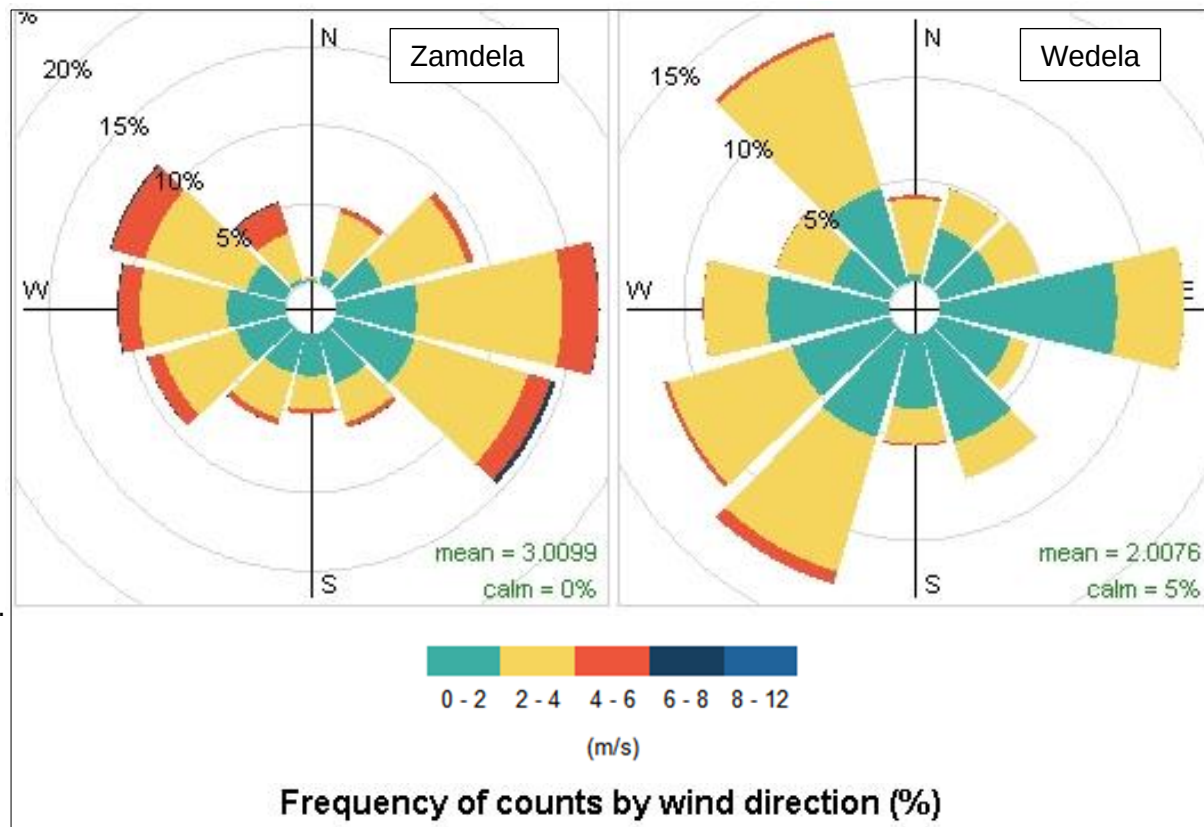


Figure 4.2-1. Annual average wind roses for the monitoring stations within Zamdela and Wedela respectively, for the period 2018-2020.

Seasonal average wind roses are reflected in Figure 4.2-2 and Figure 4.2-3 for the respective study areas. The temporal variability in the wind field is evident in the wind roses. Autumn shows relatively weak winds in both areas with the highest averages barely exceeding 4 m.s⁻¹. Easterly winds dominate over Zamdela and south-easterly are prominent in Wedela.

The surface wind field at Zamdela is dominated by winds from the easterly and westerly sectors. Mean speed between 2.4 m.s⁻¹ and 3.4 m.s⁻¹ were recorded with wind predominantly blowing from the east. Summer recorded the highest wind speed of ~8 m.s⁻¹ which blew from south-east. The winds are relatively weak in Wedela with calms of 1.1% in autumn, 7.3% in winter, 6.2% in spring and 5.4% in summer. During all seasons the wind speed hardly exceeded 6 m.s⁻¹ which is frequently observed in spring. The wind roses show that the most frequent winds blew from the

south-east during autumn ~20% of the time, from the east in winter ~20% of the time, and from the south-west in summer above 30% of the time.

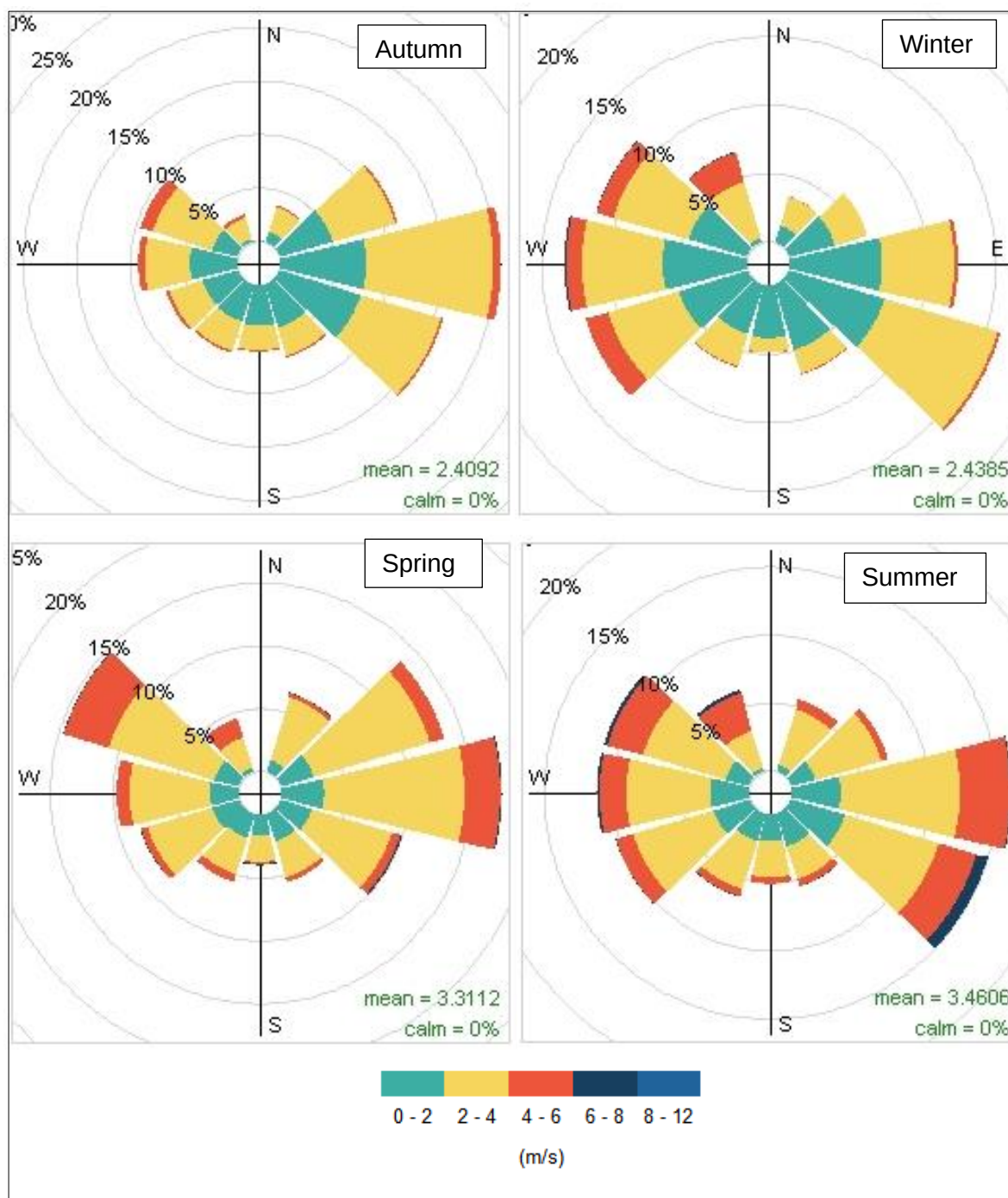


Figure 4.2-2. Seasonal average wind roses for the monitoring stations within Zamdela, for the period 2018-2020.

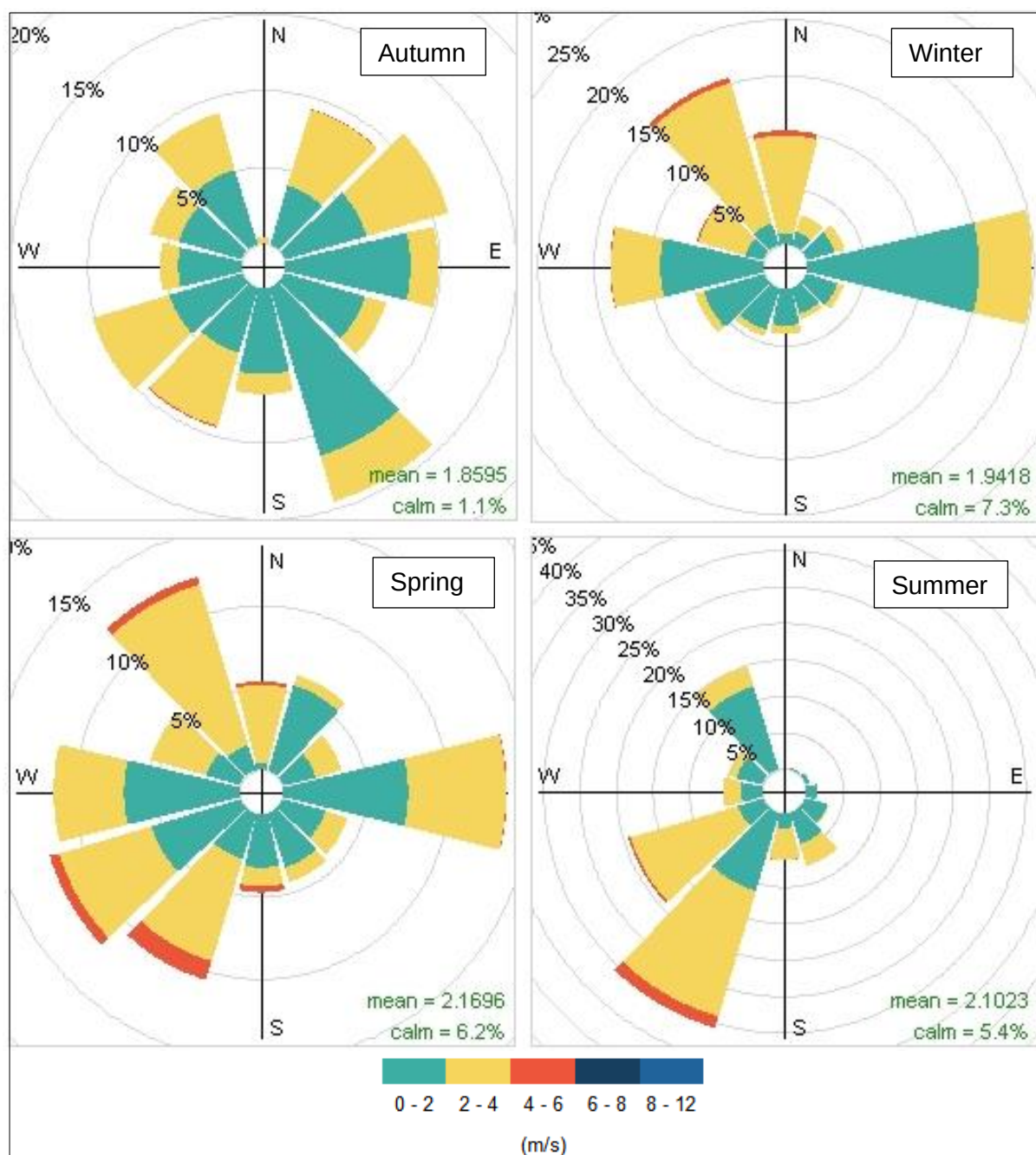


Figure 4.2-3. Seasonal average wind roses for the monitoring stations within Wedela, for the period 2018-2020.

The daytime and night-time average wind roses are provided in Figure 4.2-4. Night-time conditions are typically characterised by calm conditions and low wind speeds while stronger winds occur during the day. Apart from this, frequent winds are observed from the westerly sector in Zamdela and north-westerly and south-westerly in Wedela during the day. Contrary, during the night easterly winds are more frequent for both settlements.

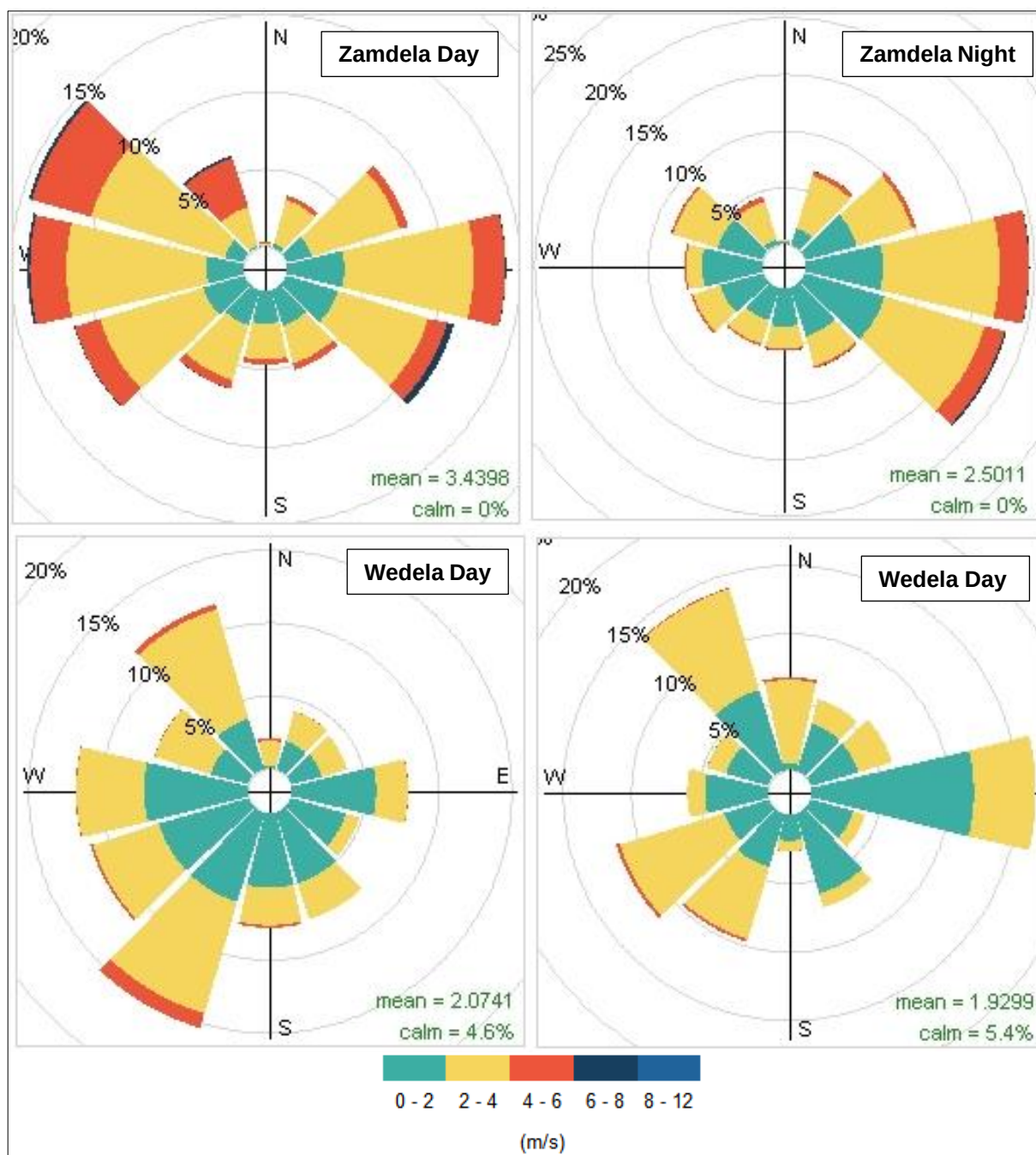


Figure 4.2-4. Day and night average wind roses for the monitoring stations within Zamdela (top) and Wedela (bottom) respectively for the period 2018-2020.

4.2.2 Temperature variations

Atmospheric temperature is an important factor in the development of the inversion and mixing layers. It also regulates the effect of plume buoyancy since the greater the temperature difference between ambient air and the plume, the higher the plume will rise. Consequently, this will affect the rate of dispersion of pollutants before it reaches ground level (Thomas, 2009).

Monthly temperatures as recorded by the stations in the study areas are given in Table 4.2-1. The maximum temperatures occur during November, December and January, with the minimum

temperatures occurring during June, July and September in Zamdela, while in Wedela the lows stretch from June until November. The mean monthly temperatures range between 10.6°C – 22.0°C for Zamdela and 7.7°C – 17.6°C for Wedela.

Table 4.2-1. Maximum, minimum, and mean temperatures measured at stations over the study areas.

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Maximum												
Zamdela	35.3	31.97	32.2	28.2	28.5	30.6	25.9	27.8	32.9	34.0	35.5	36.3
Wedela	27.6	26.6	26.1	21.5	21.8	17.5	19.9	20.6	26.1	26.2	27.9	29.9
Minimum												
Zamdela	11.0	12.5	11.1	4.8	1.8	-2.1	-3.1	3.5	-1.0	3.8	5.18	9.4
Wedela	7.9	8.1	7.9	4.8	0.7	0.1	0.1	0.2	0.1	0.2	0.3	6.3
Mean												
Zamdela	22.0	20.6	20.1	16.7	13.6	10.6	11.2	13.6	17.8	18.5	20.3	21.9
Wedela	16.6	15.6	16.4	11.3	10.9	7.7	7.9	11.7	13.3	14.6	15.2	17.6

A similar diurnal temperature variation is observed over both study areas (Figure 4.2-5). As the temperature lowers during night-time, the air in direct contact with the earth's surface is forced to cool accordingly (Thomas, 2009). The lowest temperatures occur in the morning between 05:00 and 06:00 SAST. As the sun rises, the incoming shortwave radiation warms the earth's surface, resulting in the heating of the layer of air directly above it causing this layer to peak around 15:00 SAST.

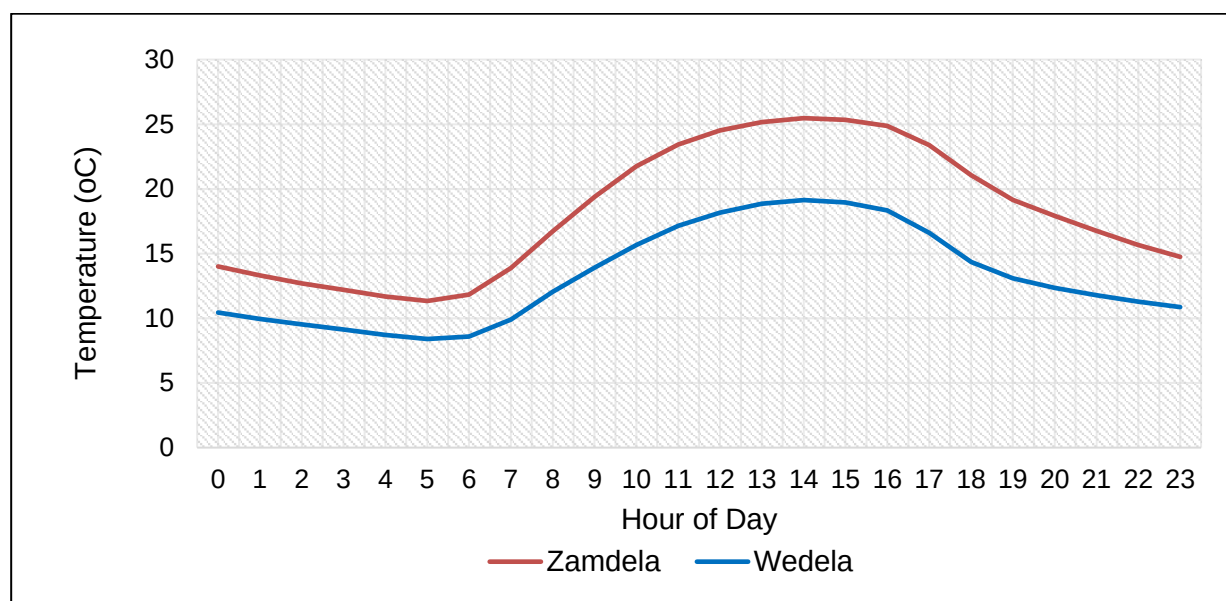


Figure 4.2-5. Mean diurnal temperature variations recorded at the respective monitoring stations within the study area for the period 2018-2020.

4.2.3 Precipitation

Precipitation aids in reducing PM formed by activities such as erosion, construction, and some industrial processes, as well as washing pollutant particles from the air. However, precipitation can also react with pollutants in the air to form more hazardous secondary pollutants, for example, the elements responsible for acid rain (Thomas, 2009).

The monthly rainfall for the respective monitoring stations is summarised in Table 4.2-2. There was no rainfall data for Wedela, therefore measurements from Potchefstroom (41 km from the township) were assumed. It can be observed from the record that the main rainy season during the study period in both settlements is between November and April. In Zamdela, the highest recorded monthly rainfall was 205.2 mm observed in March whereas, the peak in Potchefstroom was recorded in January with 383.8 mm, while winter months were relatively dry.

Table 4.2-2. Total monthly rainfall figures (mm) from the meteorological monitoring stations within the respective study areas.

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Zamdela	91.4	130	205.2	103.4	40	4.8	0	0	0	22.4	47.2	131
Wedela (Potch)	383.8	260.4	232	301.6	20.4	10.8	6.6	3.4	3.8	50.8	305.6	310.4

In Zamdela, the annual precipitation was measured at 462.2 mm and 313.2 mm for the period of 2018 and 2019 respectively (Figure 4.2-6). The maximum rainfall was observed in March and April in the respective years with winter months July and August receiving no rainfall.

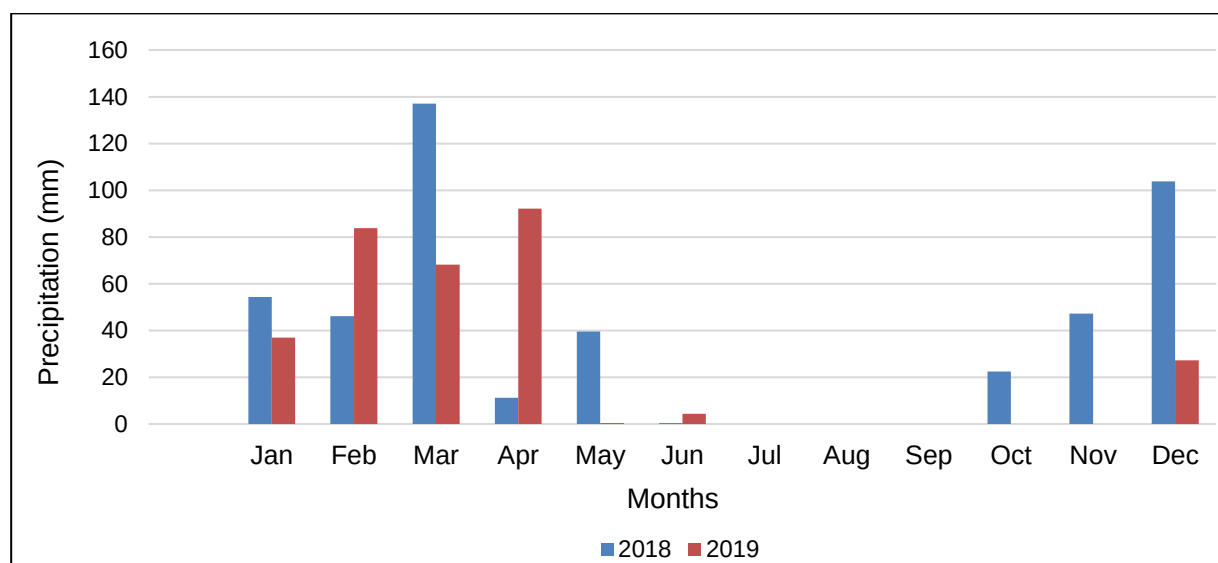


Figure 4.2-6. Monthly measured rainfall for the meteorological station of Zamdela over the period 2018-2019.

4.2.4 Relative Humidity

Relative humidity is an inverse function of ambient atmospheric temperature. As the ambient air temperature increases, the relative humidity in the atmosphere decreases given the moisture content remain constant (Thomas, 2009). This is observed in the diurnal trend in Figure 4.2-7 which shows a counter pattern of Figure 4.2-5.

Relative humidity will increase during the night to reach a maximum just before sunrise (5:00-7:00 SAST). This coincides with the lowest observed ambient air temperatures (Figure 4.2-5). As the ambient temperature begins to increase, the relative humidity decreases to reach its minimum during the warmest times of the afternoon (14:00 to 15:00). The maximum measured relative humidity measure ranged from 91.2% – 94.9% and 98 – 100% with the minimum ranging from 0% – 26% and 6 – 12% for Zamdela and Wedela respectively.

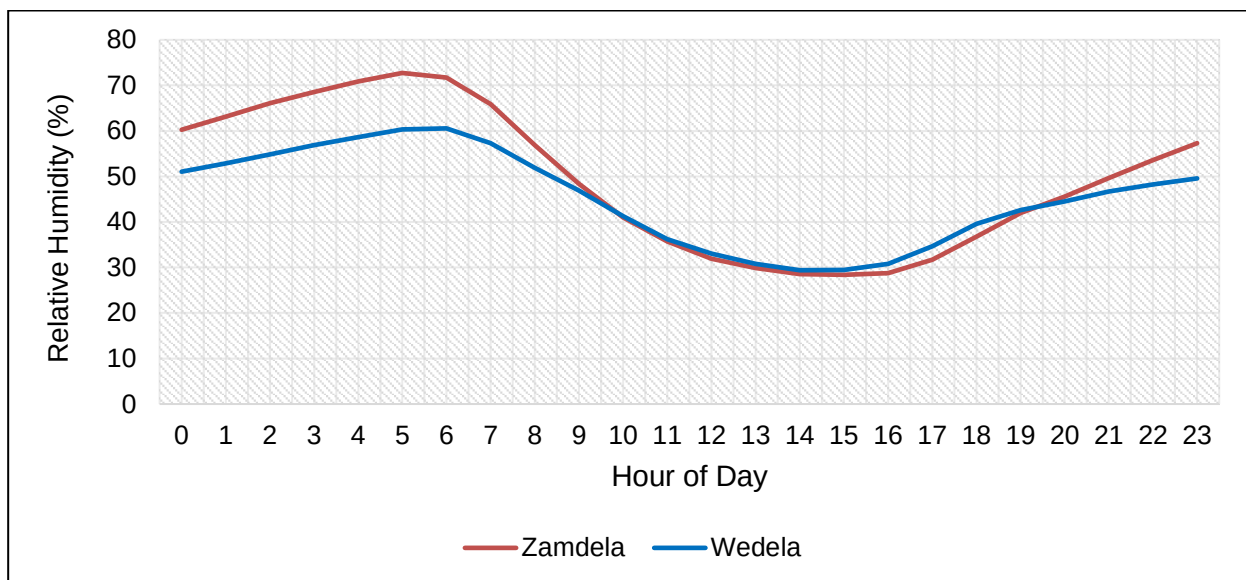


Figure 4.2-7. Mean diurnal variation of relative humidity measured at the respective monitoring stations within the study area for the period 2018-2020.

Figure 4.2-8 gives a summary of the mean monthly relative humidity values for the monitoring stations over the respective study areas. Like temperature, the average monthly relative humidity decreases during the dryer, cooler months (May to September), and increases during the wetter, warmer months (January through April and October to December). The mean monthly relative humidity range is 31.4% – 69.6% for Zamdela, and 26.7% – 70.8% for Wedela.

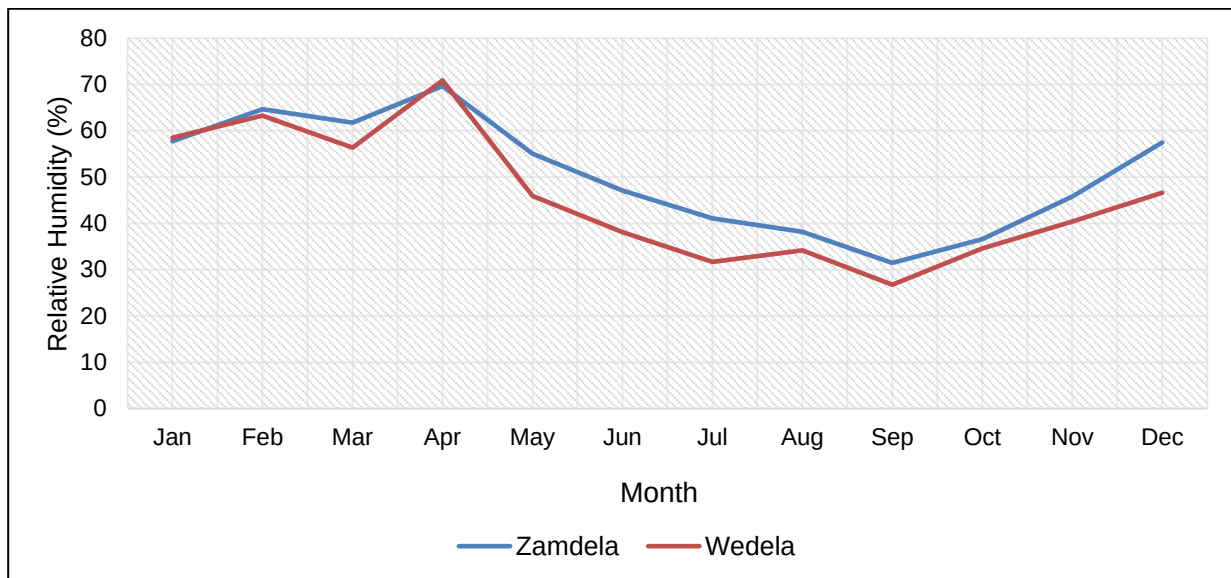


Figure 4.2-8. Mean monthly variation of relative humidity measured at the respective monitoring stations within the study area for the period 2018-2020.

4.3 Measured ambient air quality within the study area

In analysing the ambient air quality monitoring data, only data to which a sensible level of accuracy could be attached for the period 2018 – 2020 was used. Elevated concentrations of airborne PM do occur over Zamdela due to its proximity to mining activities, while Wedela conceivably suffers from the impact of dust erosion. This subsection will provide an outline on the extent of such concentrations reflecting on the temporal trends, which can help in defining the nature of sources contributing to the PM.

4.3.1 Hourly variation of PM concentrations

The mean hourly concentration of PM observed during the monitoring period was approximately $42.63 \mu\text{g.m}^{-3}$ ($\text{PM}_{2.5}$) and $103.69 \mu\text{g.m}^{-3}$ (PM_{10}) at Zamdela, while the recorded averages over Wedela were $13.46 \mu\text{g.m}^{-3}$ ($\text{PM}_{2.5}$) and $34.70 \mu\text{g.m}^{-3}$ (PM_{10}). The time series of PM concentrations at the stations are illustrated in Figure 4.3-1. The concentrations observed at Zamdela are significantly higher than those recorded in Wedela, which could be attributed to the possible sources found in the towns. At the Wedela monitoring site, it can be argued that the sources contributing to $\text{PM}_{2.5}$ and PM_{10} are not always the same as the variations do not agree with Zamdela's time trends. Regardless of this, both sites show highs during the winter and spring period which can be due to increased burning events for space heating in winter and wind effects in spring.

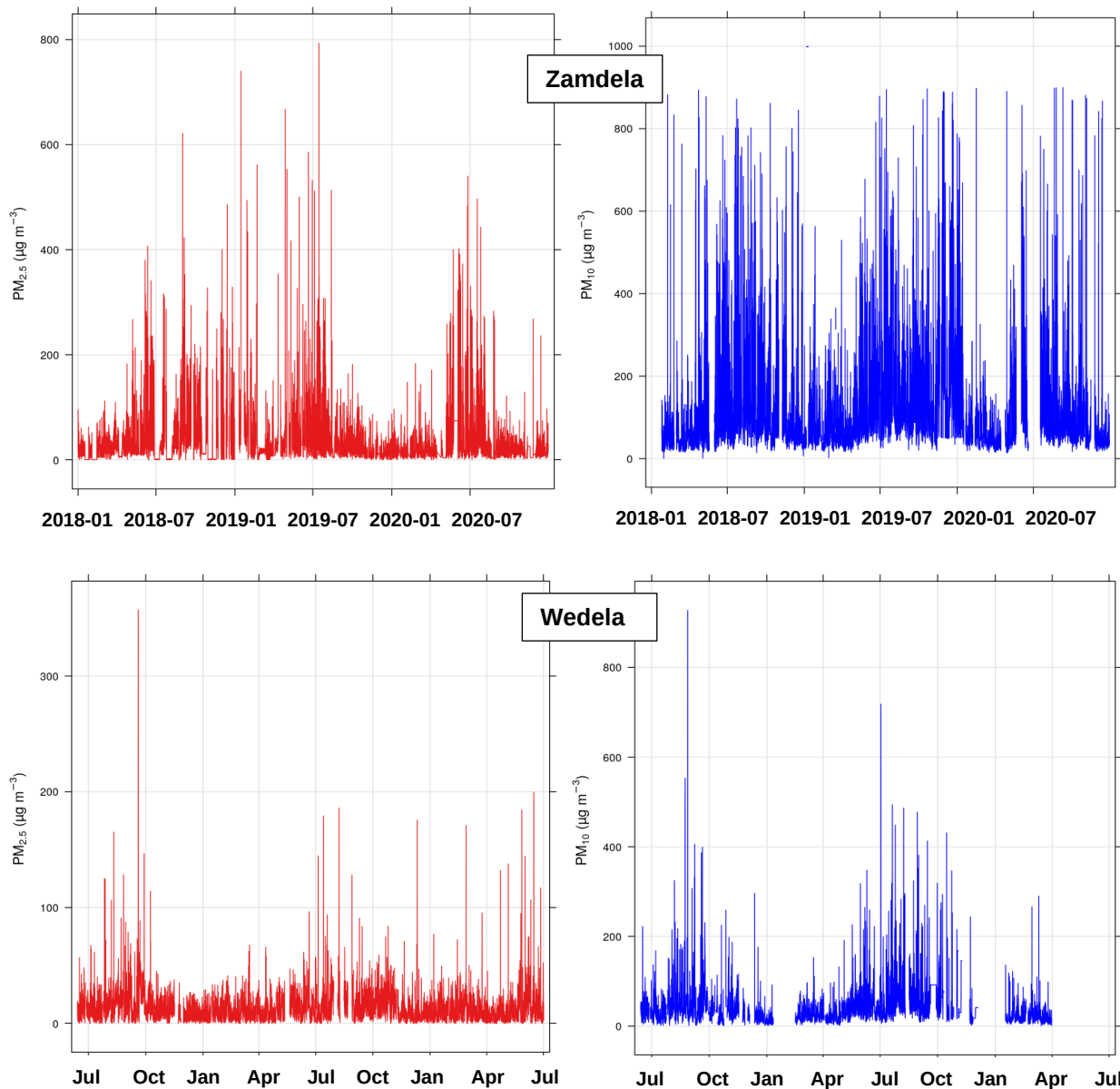


Figure 4.3-1. Time series of PM concentrations using hourly averages at Zamdela (top) and Wedela (bottom).

The frequency of exceedance of the proposed NAAQS daily and annual limits are illustrated in Table 4.3-1. Elevated inhalable particulate concentrations were recorded over Zamdela with both daily and annual average ground level concentrations exceeding the NAAQS limits at the monitoring stations. Both $PM_{2.5}$ and PM_{10} exceeded the standards more than half the time of the monitoring with annual averages of $43.36 \pm 3.01 \mu\text{g.m}^{-3}$ and $103.17 \pm 1.9 \mu\text{g.m}^{-3}$ respectively. On the contrary, Wedela is in adherence most of the time with only 4 exceedances observed with annual averages between $13.64 \pm 1.15 \mu\text{g.m}^{-3}$ ($PM_{2.5}$) and $30.67 \pm 4.89 \mu\text{g.m}^{-3}$ (PM_{10}).

Table 4.3-1. National ambient air quality standards for PM_{2.5} and PM₁₀ compared to observed inhalable PM at ambient stations.

Pollutant	Averaging Period	Maximum allowed concentrations	Allowed exceedances per year	Observed exceedances		99 th percentile concentration (µg.m ⁻³)	
				Zamdela	Wedela	Zamdela	Wedela
PM _{2.5}	24-hour	40 µg.m ⁻³	4	(n=314) 160	(n=348) 4	179.21	41
	Annual	20 µg.m ⁻³	0	2	0	--	--
PM ₁₀	24-hour	75 µg.m ⁻³	4	(n=353) 219	(n=349) 16	295	141.64
	Annual	40 µg.m ⁻³	0	2	0	--	--

Figure 4.3-2 illustrates the diurnal variation of the monitored PM at the respective sites. A prominent bimodal diurnal pattern is exhibited at Zamdela with increases in the concentrations in the early morning (06:00 – 10:00) and evening (17:00 -21:00), which is typical of domestic fuel burning activities and traffic trends (Langeman *et al.*, 2015; Language *et al.*, 2016; Nkosi, 2018). This diurnal trend can also be a result of plumes naturally descending to the ground during the period of atmospheric instability. Such vertical turbulence is related to the breakup of inversion layers and convective mixing of emissions (Belelie, 2018). Wedela on the other hand shows two distinct variations with PM_{2.5} slightly peaking during the early morning (01:00 – 04:00) and significantly peaking during the evening (17:00 -21:00). PM_{2.5} emissions are mainly from anthropogenic activities with the early increase in concentration possible due to heavy vehicles to and from the mines and the crusher near the township, while the evening increase coincides with traffic peaks and domestic burning for cooking and space heating. PM₁₀ concentrations follow an opposite pattern with peaks recorded during the afternoon. This is typical of dust erosion which is arguably the main source of coarse particulates over the township given its proximity to mine dumps, mines and a stone crusher. This agrees with Thomas (2009) who suggested that peak concentrations from industrial activities occur during the day between 10:00 and 16:00.

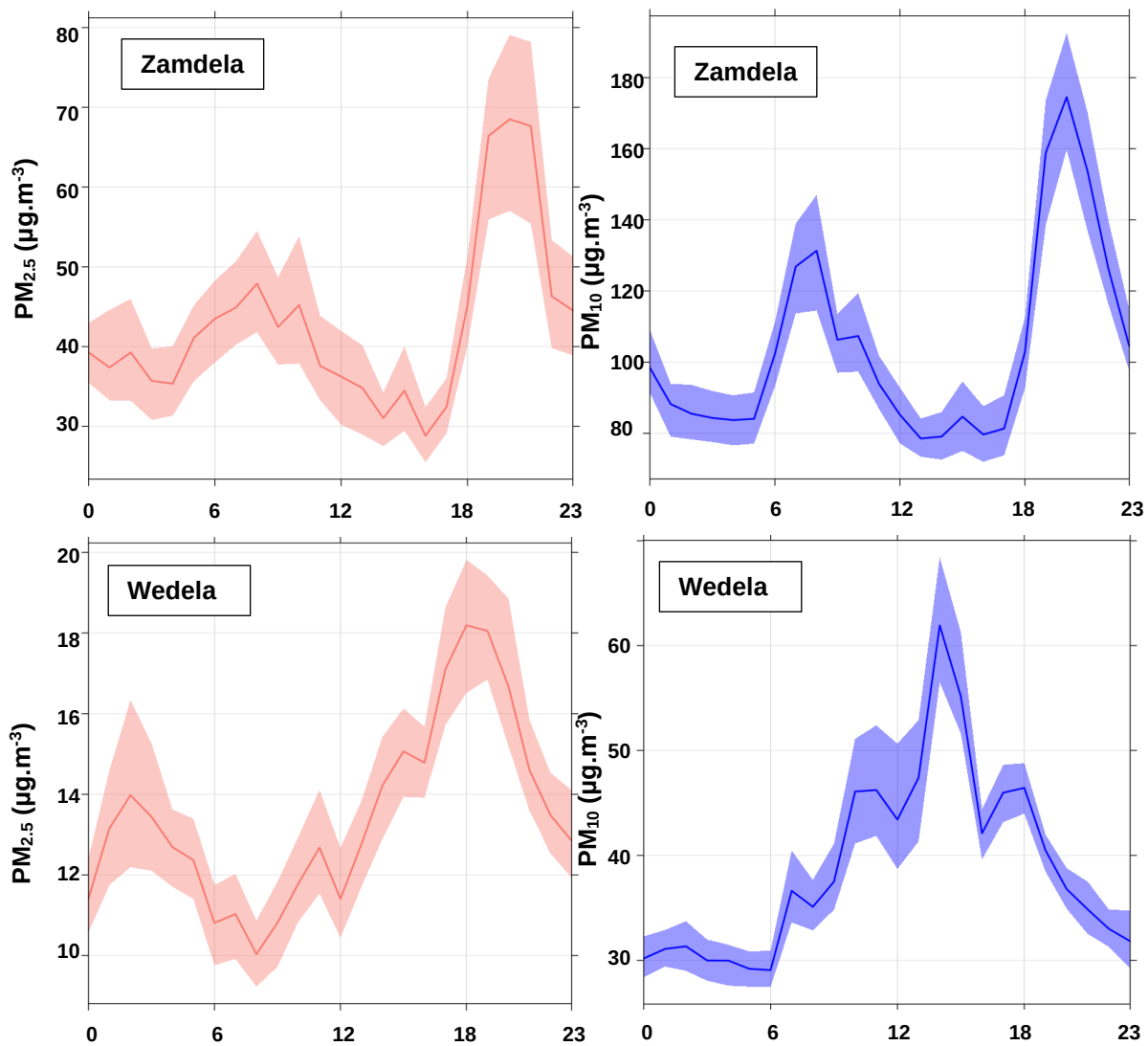


Figure 4.3-2. Average hourly variation of $PM_{2.5}$ and PM_{10} concentrations ($\mu g.m^{-3}$) at Zamdela and Wedela respectively.

4.3.2 Seasonal variation of PM concentrations

Seasonal average hourly variations of concentrations are summarised in Table 4.3-2. The seasonal variation of $PM_{2.5}$ and PM_{10} concentrations at Zamdela and Wedela are further illustrated in Figure 4.3-3 through Figure 4.3-6. The highest hourly $PM_{2.5}$ concentration at Zamdela was observed in winter ($793.62 \mu g.m^{-3}$) followed by summer ($740.05 \mu g.m^{-3}$), autumn ($667.13 \mu g.m^{-3}$), and spring ($621.05 \mu g.m^{-3}$) in descending order. PM_{10} declined from winter ($895.88 \mu g.m^{-3}$) to spring ($861.35 \mu g.m^{-3}$), summer ($854.21 \mu g.m^{-3}$) and autumn ($677.53 \mu g.m^{-3}$). In contrast to Zamdela, the highest fine particulate concentration in Wedela was observed in spring ($357.08 \mu g.m^{-3}$), winter ($179.25 \mu g.m^{-3}$), autumn ($67.58 \mu g.m^{-3}$) and summer ($47 \mu g.m^{-3}$) in that descending order, while PM_{10} fell from winter ($676.5 \mu g.m^{-3}$), to spring ($405.83 \mu g.m^{-3}$), summer ($176.7 \mu g.m^{-3}$) and autumn ($48.5 \mu g.m^{-3}$).

The minimum $PM_{2.5}$ and PM_{10} concentrations at the Zamdela site were recorded in summer (0.03 and $1.88 \mu g.m^{-3}$) with winter recording the highest minimums (1.35 and $25.58 \mu g.m^{-3}$) respectively. These values suggest that the activities of possible sources contributing to PM are subjective to seasons, for example, domestic fuel burning events increase in winter, as does the demand on electricity generation, notwithstanding the impact of meteorology on transportation. Wedela on the other hand showed a constant minimum $PM_{2.5}$ concentration throughout the seasons ($0.08 \mu g.m^{-3}$) with the same recorded in summer and autumn. This can be attributed to sources that operate constantly and are not influenced by seasons, for example, some industries like mining and crusher are influenced by the market demand more than the season.

Table 4.3-2. Statistical summary of PM concentrations ($\mu g.m^{-3}$) at Zamdela and Wedela respectively.

		N (hourly)		Mean		Max		Min		SD	
		$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}	$PM_{2.5}$	PM_{10}
Z a m d e l a	Summer	982	1601	47.9	68.4	740.1	854.2	0.1	1.9	55.5	61.9
	Autumn	1912	1764	28.1	81.5	667.1	677.5	0.1	15.3	36.8	70.3
	Winter	1947	1463	91.9	162.7	793.6	894.9	1.4	25.6	67.2	120.4
	Spring	1294	1905	46.6	108.6	621.1	861.4	0.1	14.3	51.	89.6
W e d e l a	Summer	1995	1830	9.2	20.2	47	176	0.1	0.1	6.9	14.2
	Autumn	1858	2019	12.4	35.7	67.6	48.5	0.1	0.1	8.9	40.9
	Winter	2066	2068	16.1	40.6	179.2	676.5	0.1	0.3	13.7	54.5
	Spring	1877	2068	16.2	41.1	357.1	405.8	0.1	0.3	15.6	35.3

Winter shows the highest averages of PM at both sites, which can be explained by the consistent and persistent presence of the inversion layer during the season. It has also been observed that during winter, there is an increase in domestic fuel burning events as the lower temperatures lead to more combustion for heating purposes. As illustrated by Figure 4.3-3 to Figure 4.3-6 the distribution of the concentration between the quartiles is large in Zamdela compared to Wedela which suggests the nature of sources in the different settlements.

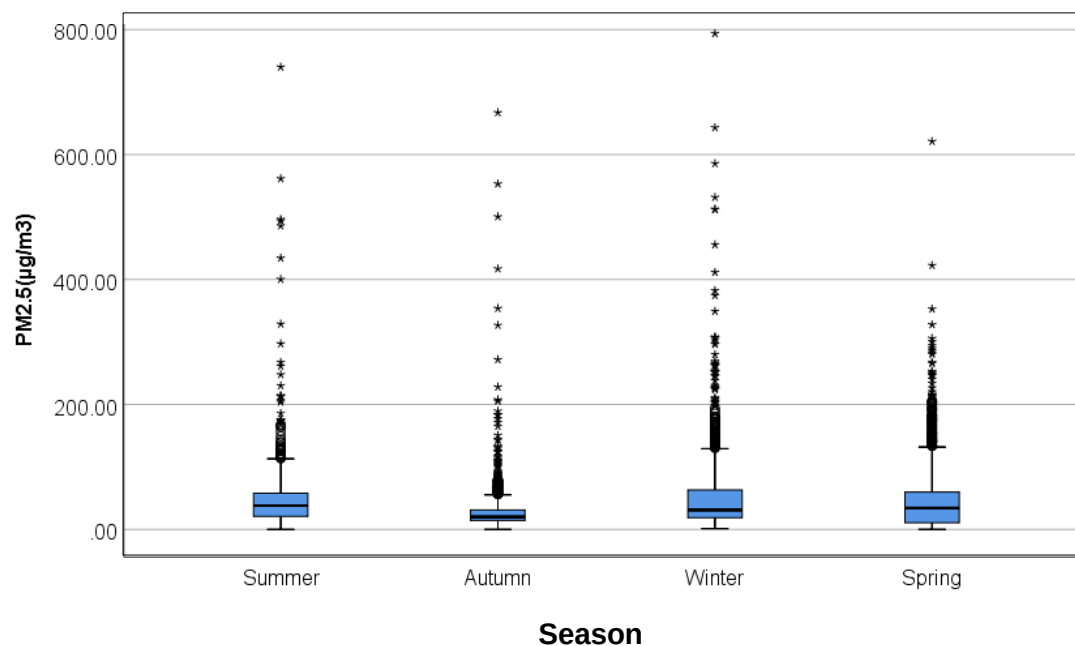


Figure 4.3-3. Seasonal variation of PM_{2.5} concentrations at Zamdela. The horizontal lines in the boxes represent the 25th, 50th, and 75th percentiles.

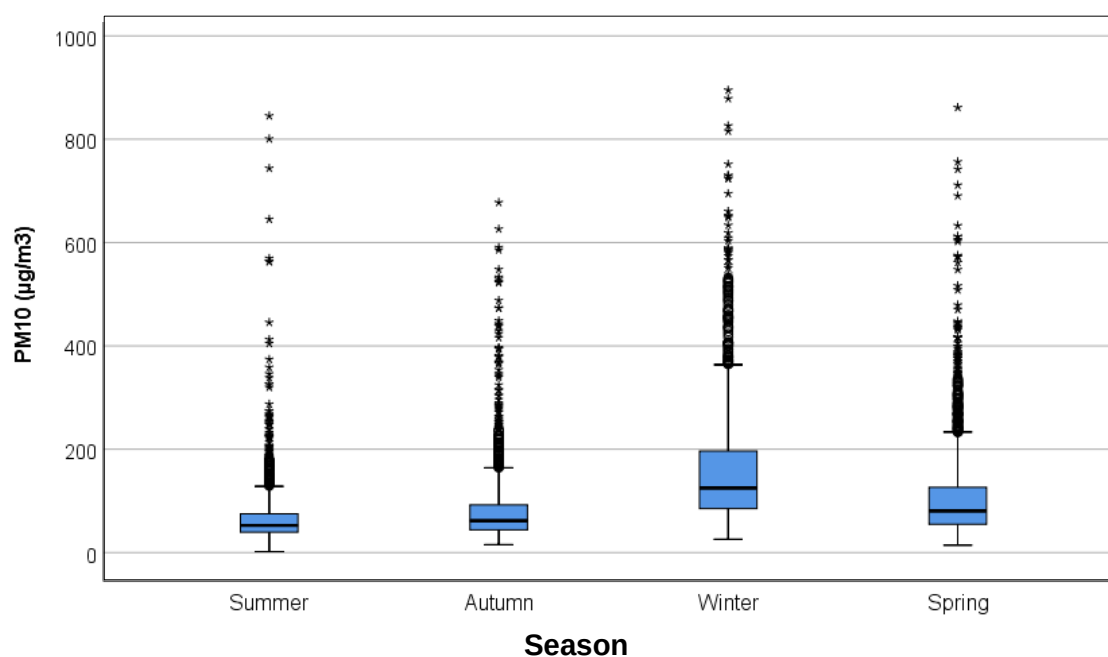


Figure 4.3-4. Seasonal variation of PM₁₀ concentrations at Zamdela. The horizontal lines in the boxes represent the 25th, 50th, and 75th percentiles.

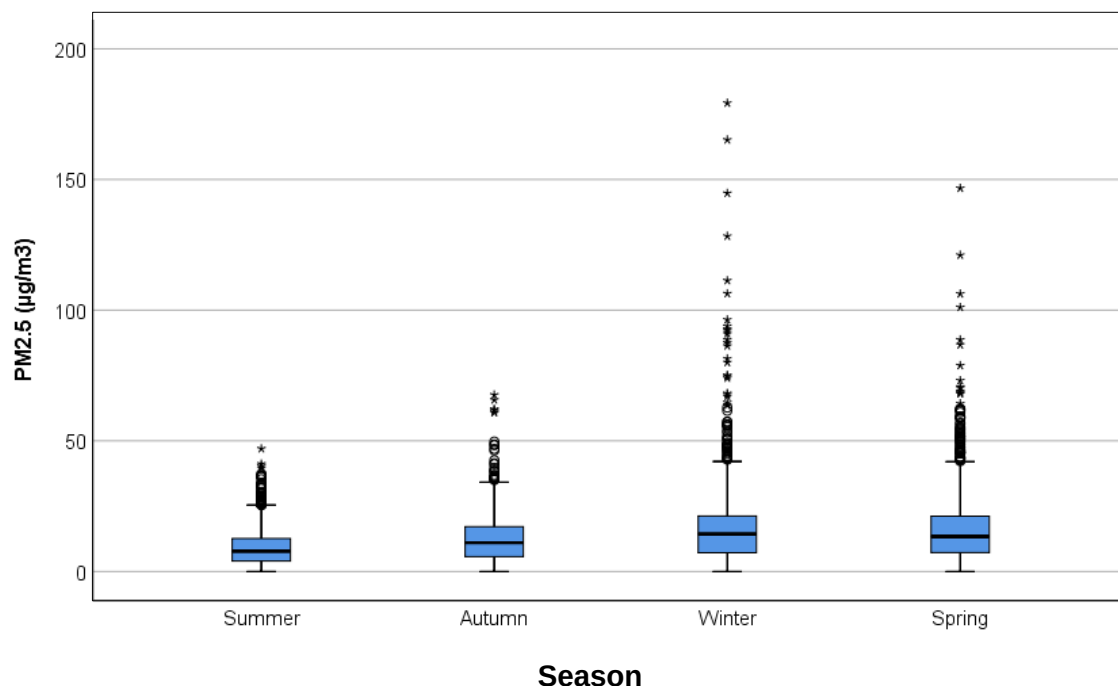


Figure 4.3-5. Seasonal variation of PM_{2.5} concentrations at Wedela. The horizontal lines in the boxes represent the 25th, 50th, and 75th percentiles.

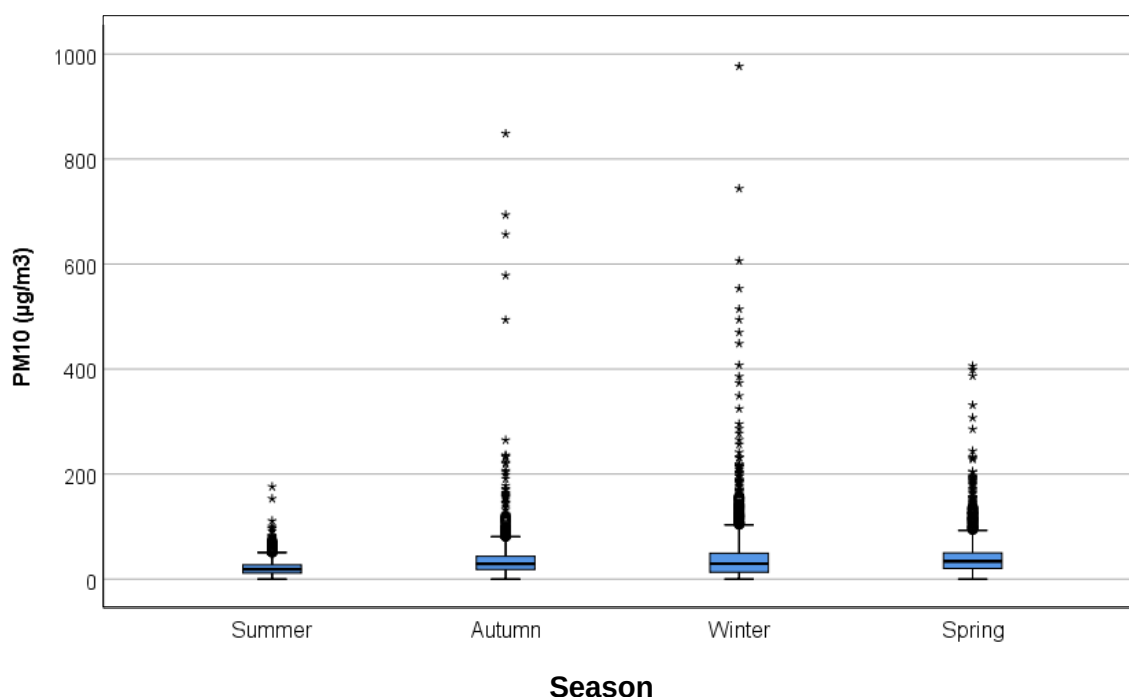


Figure 4.3-6. Seasonal variation of PM₁₀ concentrations at Wedela. The horizontal lines in the boxes represent the 25th, 50th, and 75th percentiles.

4.3.3 Correlation of ambient PM concentrations and meteorology

A correlation of PM concentrations at the sites with measured meteorological parameters is illustrated in Table 4.3-3. Very small correlations were observed at both sites for all monitored meteorological parameters. This suggests that the concentrations are mostly governed by the sources' activities with little influence from meteorological variables. This supports the narrative that source activities vary with time and seasons as demand rises and decreases.

Table 4.3-3. Correlation of ambient PM concentrations and meteorological parameters at Zamdela and Wedela respectively.

		Rainfall	RH	Temperature	WD	WS
Zamdela	PM_{2.5}	-0.01	-0.04	-0.15	-0.21	-0.16
	PM₁₀	-0.01	-0.21	-0.24	-0.18	-0.23
Wedela	PM_{2.5}	---	-0.12	0.03	-0.16	0.14
	PM₁₀	---	-0.11	-0.12	-0.21	0.19

4.4 Chapter conclusion

Ambient PM concentration temporal variations were analysed for both sites, with Zamdela showing significant standard deviations, signalling the profound influence of the sources around the monitoring site. The observed diurnal and seasonal variations in Zamdela suggest the probable impact of domestic fuel use and mobile sources' trends, while the patterns in Wedela suggest the impact of dust activities with PM₁₀ peaking in the afternoon. Significant exceedances of current airborne particulates are exceeded half the time in Zamdela while Wedela tends to be in adherence most of the time. Little correlation was observed between the measured PM and meteorological parameters at both sites. The main conclusion drawn is that ambient particulate matter concentrations are constantly elevated across Zamdela throughout the monitoring period. Given that these measurements are made at low levels, observed concentrations pose a significant health risk. Source apportionment studies can help attribute the emissions to the possible sources. This can be particularly helpful in determining the contribution of different potential sources. This is important as industrial activities are often regarded as the overriding source of PM in South Africa's townships.

CHAPTER 5

This chapter provides the PM concentrations estimates from identified potential sources within Zamdela and Wedela. Firstly, the identified sources will be given, followed by the emissions that are estimated to originate from the sources.

5. INTRA-URBAN EMISSIONS INVENTORY FOR WEDELA AND ZAMDELA

The identification and quantification of potential sources of emissions in addition to the determination of ambient concentrations are vital in the assessment of the possible increasing impacts associated with emissions. A PM emissions inventory for this study was established for sources within the study areas (intra-urban), with Szpiro (2008) and Wernecke (2018) stating that the assessment of exposure to air pollution can be enhanced by accurately measuring the **intra-urban** variations in ambient pollutant concentrations.

5.1 Identification of sources

The most important thing to do in response to the impact of air pollution is to identify and define the characteristics of the emissions sources (Szpiro, 2008). The emissions inventory discussed in this paper exclusively contains sources that were identified within Zamdela and Wedela.

5.1.1 Evidence of possible sources from ambient data

According to Carslaw (2015), variations of pollutants by time can be useful in revealing likely sources. This section provides an interpretation of diurnal, weekly and seasonal cycles of 1-hour average PM (PM_{10} and $PM_{2.5}$) concentrations measured within the two townships between 2018 and 2020.

At Wedela (Figure 5.1-1), the diurnal pattern shows a PM_{10} peak between 13:00 and 15:00 which can be associated with lunchtime vehicle movement and given the proximity of the town to mining activities, afternoon peaks can also be associated with dust erosion as the wind speed tends to peak in the afternoon. $PM_{2.5}$ peak can be observed in the early morning and evening, which can be associated with vehicle movement peaks and domestic fuel burning for cooking.

A decrease in concentrations of PM observed at the end of the weekend can be linked to a low level of human activities during the weekends, while peaks can be seen on Monday and Friday when people drive in and out of their working environments, (Riffault, 2017). The seasonal data shows high concentration readings between May and September as these are dry seasons allowing for erosion and are also cooler temperatures that lead to an increase in burning events.

This is also the period that domestic fuel burning is at its peak as households warm their surroundings.

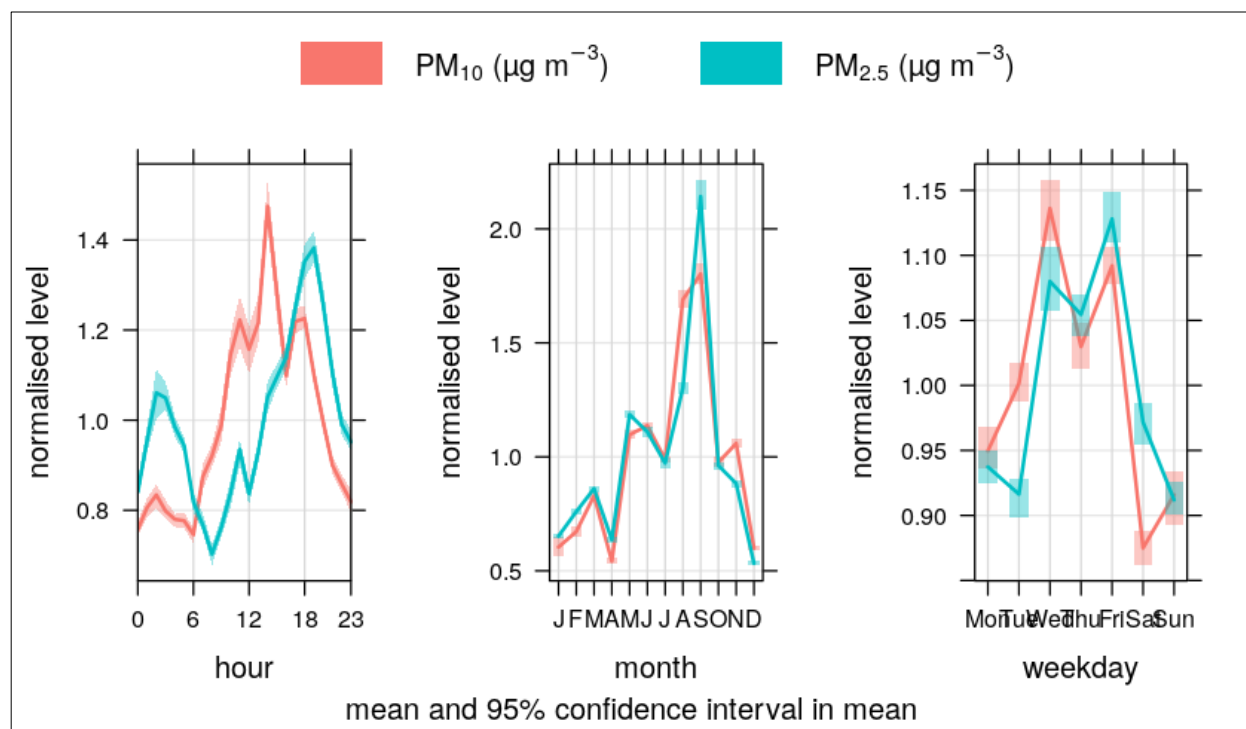


Figure 5.1-1. Normalised ambient temporal variations in concentrations as measured at Wedela monitoring site.

Zamdela shows a distinct diurnal variation with peaks in the morning (06:00 – 09:00) and evening (16:00 -22:00), which can be attributed to traffic peaks due to people going to and returning from work respectively, as well as to the typical township burning events for cooking and water heating. Average winter PM concentrations are about 2 times higher than in summer (Table 4.3-2) which shows an increase in burning events, as space heating increases (Nkosi, 2018).

September also recorded high PM concentrations which can be attributed to increased windblown dust and regional pollutants transportation, since wind speed is at its highest during this month. A drop is observed in August which can be attributed to an increase in temperature resulting in reduced domestic burning events.

Just like Wedela, the weekly PM in Zamdela drops significantly during the weekends as not much activity occurs, with few people going to work and school (Figure 5.1-2).

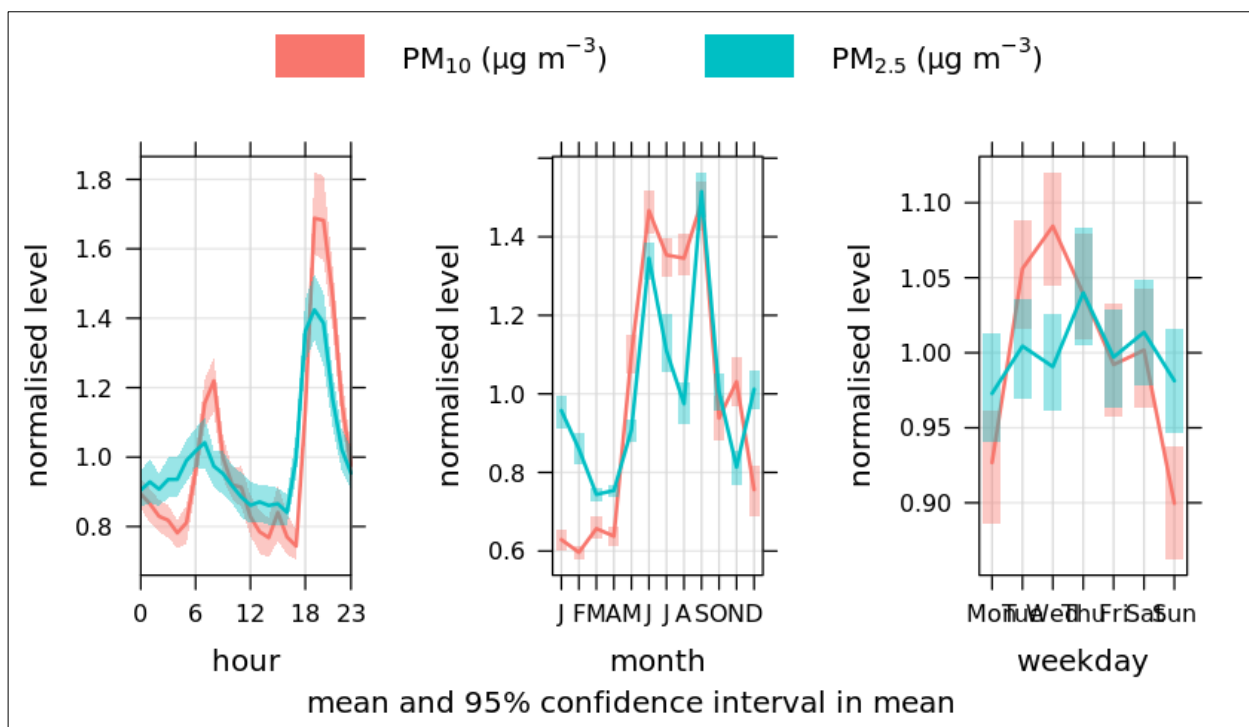


Figure 5.1-2. Normalised ambient temporal variations in concentrations as measured at the Zamdela monitoring site.

5.1.2 Evidence of possible sources from literature

Despite the intensive national electrification programme implementation, a large population within township continues to use solid fuels to meet a portion or all their energy needs (Nova Institute, 2017). The main fuels with air pollution potential used by households within the VTAPA are coal, wood and paraffin (Thomas, 2009). Zamdela has been identified as one of the townships within the VTAPA with continued use of coal and wood for space heating and cooking (Scorgie *et al.*, 2003, Muyemeki *et al.*, 2021). These fuels continue to be used due to growing informal settlements which have exacerbated backlogs in the distribution of basic services such as electricity and waste removal (Werneck, 2018). More so, some of the electrified households continue using coal due to its cost-effectiveness for space heating and cooking (Scorgie *et al.*, 2003; Nkosi, 2018).

Table 5.1-1 shows the estimated household use of the common fuel sources in Zamdela (Nova Institute, 2017) and Wedela (Stats SA, 2016). It is evident from the table that paraffin usage is high in both townships in comparison to the available electricity alternatives.

Table 5.1-1. Households' fuel usage by township (Stat SA, 2016, Nova Institute, 2017).

Energy Source	Cooking		Heating		Lighting	
	Zamdela	Wedela	Zamdela	Wedela	Zamdela	Wedela
Electricity	74%	76,6%	63,8%	50%	63,8%	50%
Paraffin	34%	21,4%	8,3%	25,9%	8,3%	25,9%
Wood	19%	0,1%	1,8%	0,9%	0%	0%
Coal	3%	0,1%	5,7%	0,2%	0%	0%

A qualitative waste burning survey conducted in Sharpeville highlighted that there is large scale local waste burning and veld fires in low-income settlements (DEFF, 2020). Wiedinmyer *et al.* (2014), Nagpure *et al.* (2015) and Das *et al.* (2018) affirm the fact that waste burning is a method to dispose of uncollected garbage, and is becoming a major concern, especially in densely populated low-income areas of developing countries.

PM emissions from vehicle tailpipes pose a serious threat to urban air quality (Colville *et al.*, 2001). South Africa's Road transport sector contributed about 11% of national emissions in 2012 and therefore presents a key source of particulates and gases (Vosper & Mercure, 2016). Scorgie *et al.* (2003), Thomas (2009) and DEFF (2020) identified vehicles as one of the major sources of emissions in the VTAPA.

Africa is the biggest continental source of biomass emissions (Roberts *et al.*, 2009; Bauer *et al.*, 2019; Ramo *et al.*, 2021), with South Africa estimated to be the largest contributor in Africa (Kituyi *et al.*, 2005; Bauer *et al.*, 2019). Scorgie *et al.* (2003) and DEFF (2020) identified biomass burning as one of the key sources of emissions within the VTAPA, which is inclusive of Zamdela township.

Windblown dust deposition is also one of the significant problems faced in South African urban and outskirt areas owing to the prevailing dry climatic conditions. Mpanza (2020) identified dust as the major source of particulates in the Wedela township, while DEFF (2020) and Thomas (2009) found fugitive dust to also be a significant contributor over the VTAPA.

Figure 5.1-3 shows the source apportionment results for sampling conducted during two weeks of July and two weeks of September 2016 at Zamdela. Domestic combustion of coal contributed the most to PM_{2.5} as burning events increase in the winter season (Nkosi, 2018). Dust, fly ash and secondary sulphate each accounted for approximately 10 % of the aerosol loading, while waste burning accounted for at least 4% of the particulate loading. A large portion of the aerosol (33 %) could not be resolved by the model with the existing profiles.

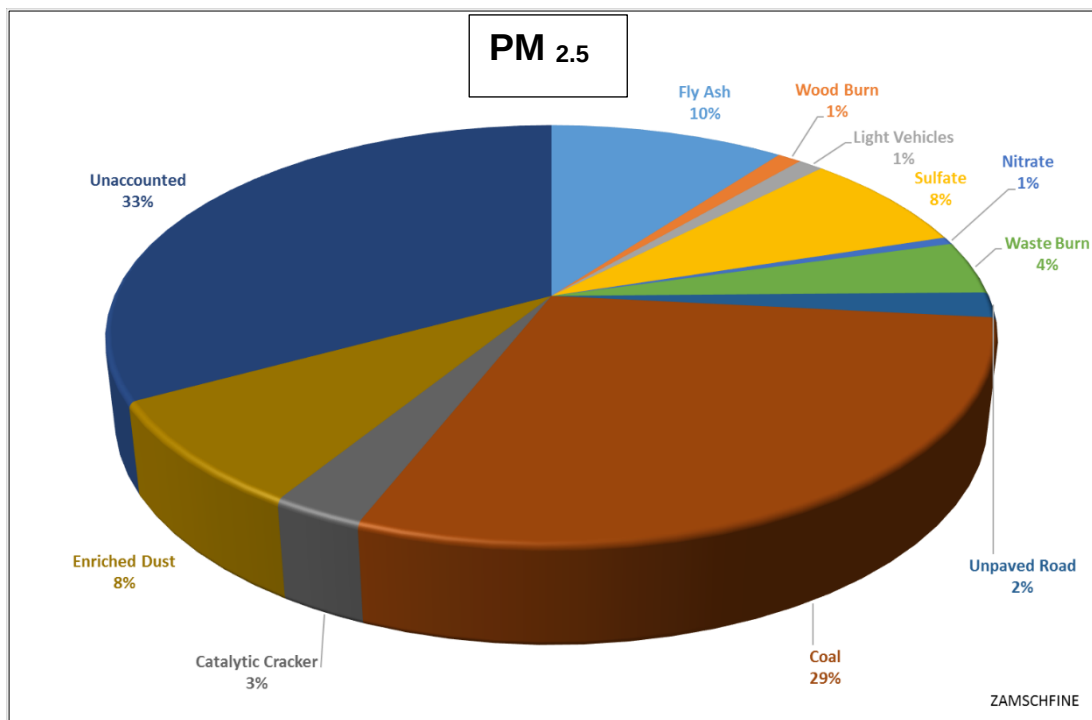


Figure 5.1-3. Source contributions to the total PM_{2.5} in Zamdela Township during July (15-30) and September (12-23) 2016. (Piketh et al, 2016).

The dominant source identified contributing to PM₁₀ in Zamdela is waste burning (44 %) followed by wood burning (21 %) (Figure 5.1-4). These sources were detected in the fine fraction but did not constitute the dominant sources. Dust and unpaved road (10 % combined) also make a notable contribution to the aerosol loading in the Township in the coarse fraction.

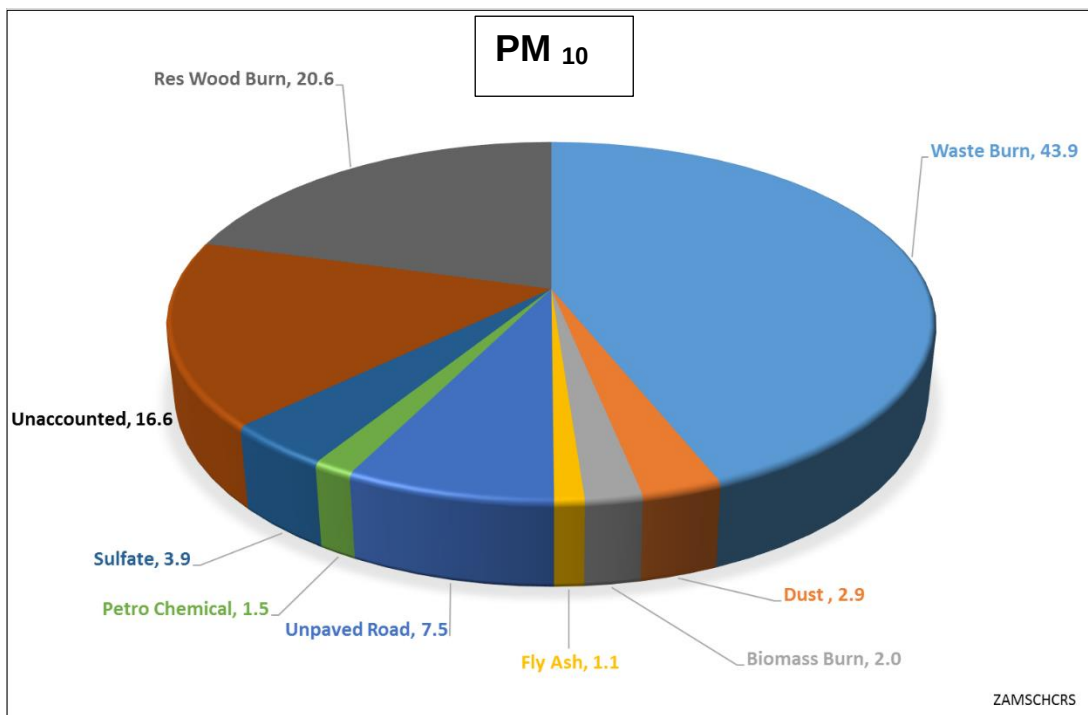


Figure 5.1-4. Source apportionment results PM₁₀ in Zamdela Township during July (15-30) and September (12-23) 2016.

5.1.3 Evidence of possible sources from satellite data

Satellite data was used to identify biomass burning spots and fugitive dust sources. The Modis data showed that approximately 0.43 km² and 0.33 km² in Wedela and Zamdela respectively, were exposed to fires in the period 2018 to 2020, which given the combustion efficiency and fuel load of grass it is a noteworthy contributor of PM. Wedela is likely to yield to the effects of mine tailings as several of them were observed around the township.

5.1.4 Evidence of possible sources from site surveys

Figure 5.1-5 shows examples of some of the activities observed within the settlements. Biomass burning was observed to be due to anthropogenic activities such as land clearing, municipal fire setting to control grass encroachment, and occurrences of waste burning fires extending to unintended plants and grassland (Figure 5.1-5). In Wedela, waste burning is an activity that is often controlled as the fires are started by municipal workers, while in Zamdela it is the community members that start the fires. Domestic solid fuel burning is not as common in Wedela, as most of the houses are electrified with just a few unelectrified houses in the new extension. Whereas in Zamdela it is a common practice that can be attributed to the number of informal houses and the accessibility and availability of solid fuels such as coal and wood. This agrees with the Stats SA (2016) and Nova Institute (2017) findings (Table 5.1-1). Wind erosion was also deemed to be an important source because of unpaved roads, bare open fields, and pile stocking, among other activities.



Figure 5.1-5. Images of some of the sources observed within the two townships, A: biomass burning, B: waste burning, C: domestic burning (evidenced by gases coming out through a chimney), D: exposed roads.

During the car counts done in both townships, it was observed that mobile sources in these townships are a major concern. Zamdela especially has busy roads with vehicles frequenting the roads due to Zamdela's proximity to the Sasol plant, as well as the links provided to other towns. The most common vehicle type in both Zamdela and Wedela are sedans, as individuals prefer to use their vehicles rather than public transport. Figure 5.1-6 illustrate the percentage frequencies of the different vehicle types; buses, mini-buses, light-duty vehicles (LDVs), heavy-duty vehicles (HDVs) and sedans.

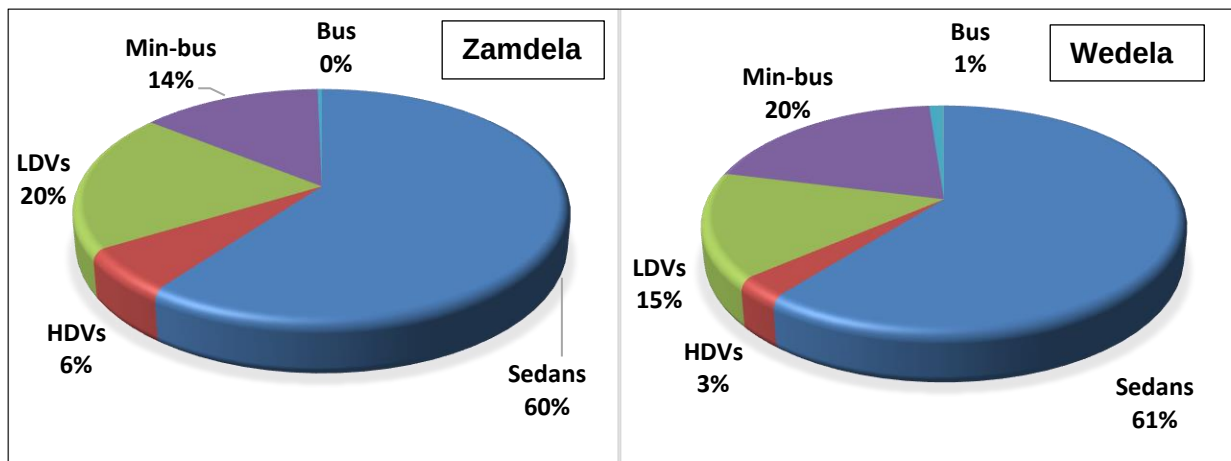


Figure 5.1-6. Vehicles type average activity as a percentage of the total flow observed in Zamdela and Wedela respectively.

Sources that were quantified and modelled during this study within the respective townships include vehicle exhausts, domestic solid fuel combustion (particularly coal, wood, gas and paraffin), waste burning, biomass burning and dust from mine dumps and other fugitive sources. The aforementioned source contributions estimates are discussed in the next section.

5.2 Emissions estimations for the identified sources

Sources of PM identified within the study areas were quantified with the aim to understand the contribution of each in terms of cumulative impact potentials.

5.2.1 Domestic fuel burning

The main fuels with air pollution potential used by households within Zamdela and Wedela are coal, wood, paraffin (Stats SA, 2016; Nova Institute, 2017). According to Langerman *et al.* (2018) and Nkosi (2018), households continue to use these fuels due to rapid urbanisation coupled with increasing informal housing, and affordability and accessibility of the fuels. Subsequently, even electrified households continue to use these fuels for space heating purposes amongst other uses. The estimates of domestic emissions were based on Stats SA (2016) and Nova Institute (2017) survey results and emission factors verified among various literature (Table 5.1-1). The distribution patterns of fuel use were randomly generated in ArcGIS as no definite data was available for the identification of burning households to which estimated emissions were assigned. It is evident from Figure 5.2-1 that domestic fuel usage in Zamdela is still a priority when it comes to mitigation strategies, as compared to Wedela where electricity is mostly used. Coal and wood are comparatively cheap and easily accessible in Zamdela due to its proximity to coal mines, compared to Wedela. Paraffin is most used source of energy after electricity in both townships. The former fuel type is known to emit a large quantity of gaseous particulates compared to the latter.

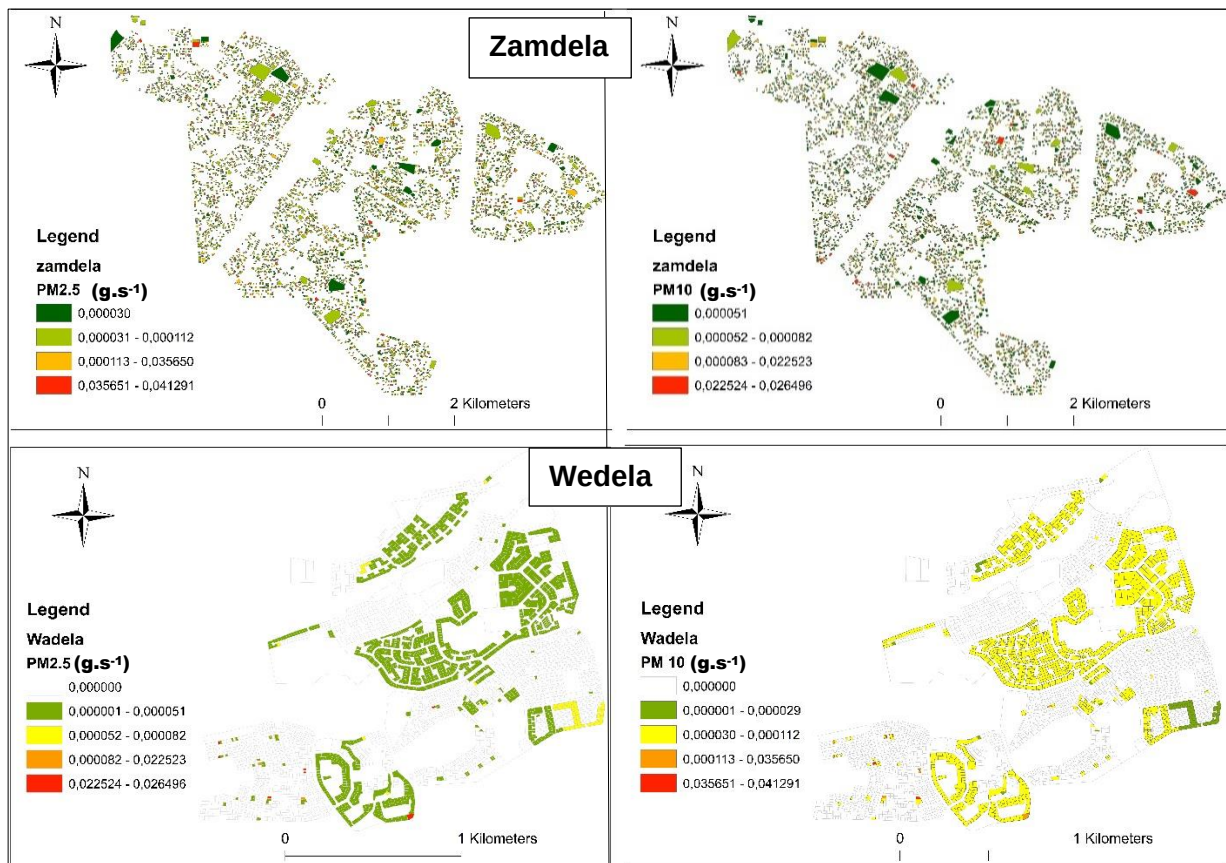


Figure 5.2-1. Spatial distribution of household fuel-burning simulated for the identification of dispersion modelling areas.

The diurnal trend in domestic fuel burning is illustrated in Figure 5.2-2 (Nkosi, 2018). Burning events tend to follow a bimodal distribution with peaks in the morning and evening. Nkosi (2018), did a study in KwaDela a township like the study areas. She observed that the morning peak happens between 05:00 and 08:00, averaging a duration of 3 to 4 hours, while the evening fires are ignited from around 16:30 to 17:30, lasting an average of 4 to 6 hours depending on the season. The demand for space heating is dependent on the temperature, hence increased burning events that happen in winter were considered in the modelling process.

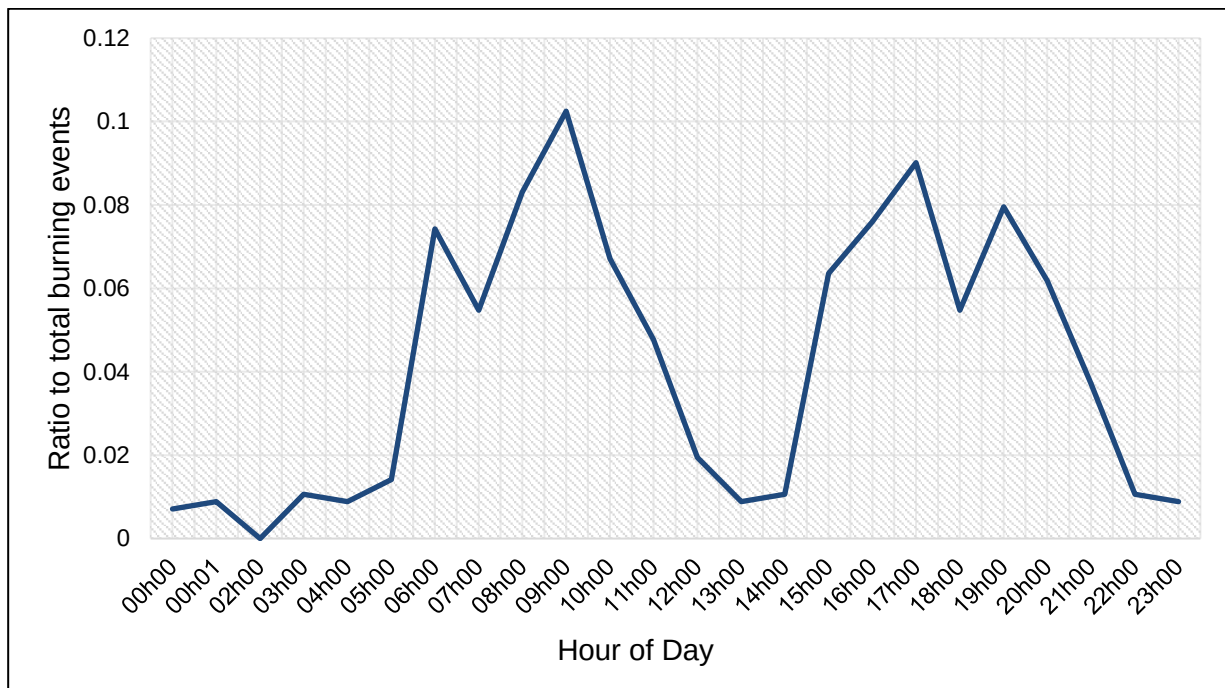


Figure 5.2-2. Diurnal trend in domestic fuel burning used during the simulation of concentrations from KwaDela (after Nkosi, 2018)

5.2.2 Waste burning

The extent of waste burning in Wedela is illustrated in Figure 5.2-3. The emissions were estimated centred on the total population of the township and the approximated total waste. This was further proportioned among the identified sites based on the size area of the activity as obtained from Google Maps. The distribution of the sites shows the effect of distance decay theory as individuals would prefer to walk short distances from their homes, hence many dumping points.

It was observed that burning events are varied with most burning events happening in the morning between 07:00 and 10:00. More so, the events are not long-lived as they last ~2 to 3 hours.

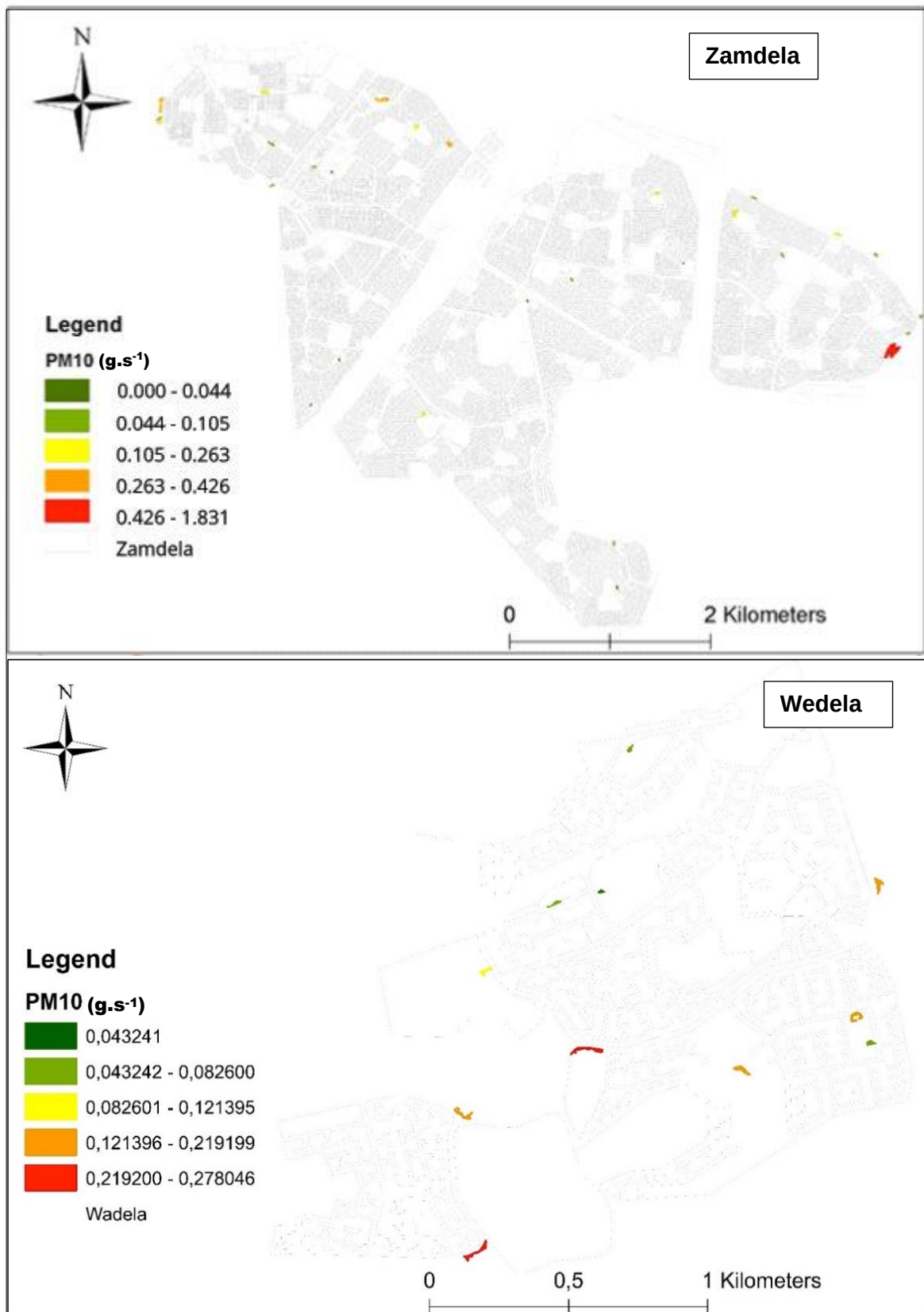


Figure 5.2-3. Spatial distribution of PM₁₀ estimated from open space waste burning within Zamdela and Wedela respectively.

5.2.3 Fugitive dust

Dust estimates were based on the Marticorena and Bergametti (1995) dust emission scheme for regional modelling systems. Unlike the other sources that are mainly governed by human activities, dust emissions depend on wind velocity (Liebenberg-Enslin, 2014). Also, dust emissions as a consequence of wind velocity are relative to the geological environments and the soil texture (the gravitational and cohesive forces between the particles) of the region (Lu & Shao, 2001).

At least 70% of the country's surface is exposed to varying intensities and types of soil emissions (Hoffman & Todd, 2000). The spatial distribution of soil erosion emission is generally inaccurate, owing to spatial and temporal variability in source activities (Jetten *et al.*, 2003; Le Roux, 2007). While soil erosion is considered a significant source in South Africa, the challenge in research remains the limited information on where the worst erosion problems are located (Mpumalanga DACE, 2002; Le Roux, 2007). Therefore, assumptions are high because of the indefinite input factors, hence apart from the recognisable mine tailings, intra-urban dust sources were assumed to be spatially distributed area sources of approximately 70% of the townships.

Figure 5.2-4 shows the assumed windblown dust source within Zamdela and Wedela respectively. Based on the area and the vertical dust flux estimation ($1.73213 \times 10^{-6} \text{ g.s}^{-1}$ – $\text{PM}_{2.5}$ and $4.87854 \times 10^{-6} \text{ g.s}^{-1}$ – PM_{10}) total dust contributions were approximated. This was obtained assuming a clay content of 6.9%, moisture content of 0.20%, friction velocity of 0.25 cm.s^{-1} and threshold velocity of 0.164310008 for $\text{PM}_{2.5}$ and 0.058130583 for PM_{10} . Figure 5.2-5 illustrates the areas assumed as mine tailing sources over Wedela. Table 5.2-1 summaries the key data inputs for ADDAS model.

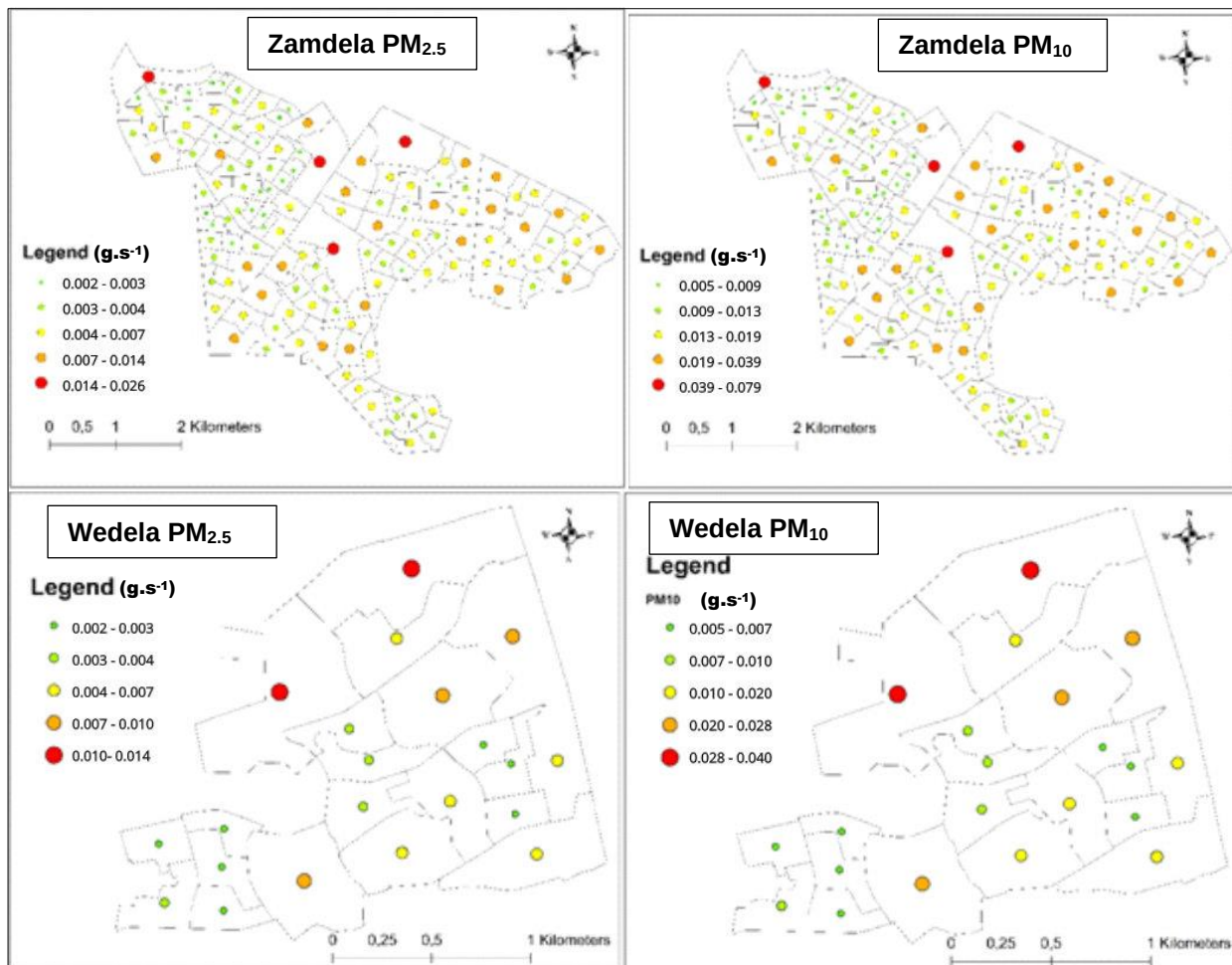


Figure 5.2-4. Spatial distribution of fugitive dust sources over Zamdela and Wedela respectively used as the layout of fugitive dust for simulation in dispersion modelling.

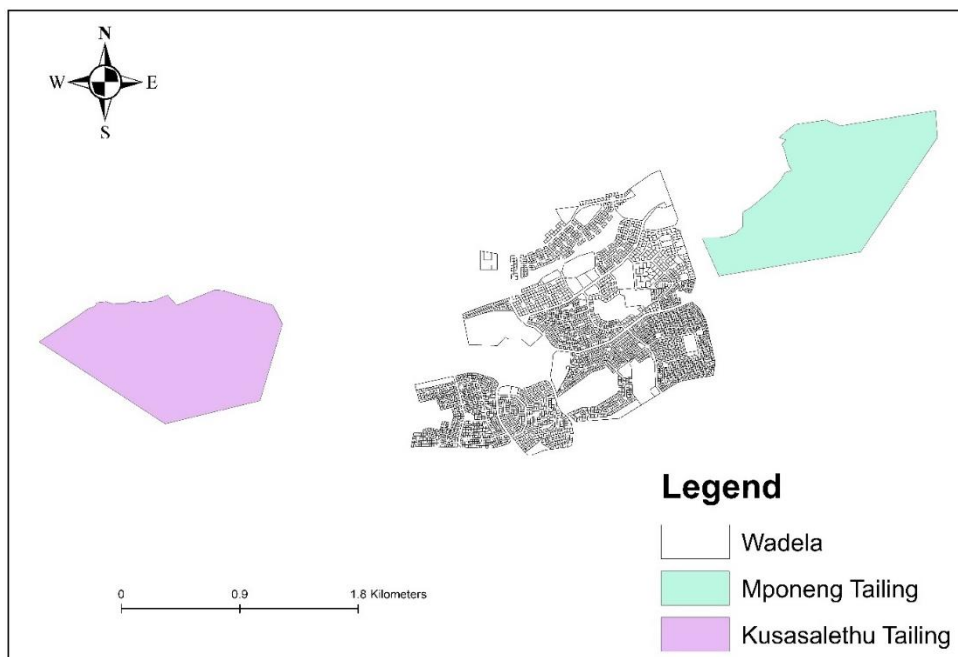


Figure 5.2-5. Map showing the extent of mine tailing dust sources within the proximity of Wedela

Table 5.2-1. ADDAS model key inputs for dispersion modelling, moisture content and surface cover.

Source	Area m ²	Clay %	Moisture content %	Particle density (kg.m ⁻³)	Surface Cover (%)
Kusasaletu TSF	1094344.8	6.9	0.2	2625	20
Mponeng TSF	1056230.5	6.9	0.2	2625	27

Temporal variation in activities was based on wind parameters measured at the monitoring stations within the two townships. The variables used for this study were extracted from Kon *et al.* (2007), as summarized in Table 5.2-2.

Table 5.2-2. Variable emission factor by wind speed m.s⁻¹ (Kon *et al.*, 2007).

Wind speed (m.s ⁻¹)	2	4	6	8	10	12	14	16
Factor	0.05	0.1	0.15	0.2	0.25	0.3	0.35	0.4

5.2.4 Vehicle emissions

Vehicle emissions were derived from an estimate of vehicle kilometres travelled (VKT). For this study, a method used by Colvile *et al.* (2001) and EEA (2019) was used, thus VKT was based on the product of the vehicle flow and length of the road the vehicle travelled on. Vehicle flow values were based on physical counts done in the townships and other study estimates, while road length was based on GIS road datasets.

Figure 5.2-6 provides the estimated spatial distribution of the total annual particulates resulting from vehicle emissions over Zamdela and Wedela. The emissions were estimated based on the road type and length, and the vehicle counts (<https://nextcloud.nwu.ac.za/index.php/s/R6xya86CXyYwnC>). Wedela is less congested compared to Zamdela resulting in the emissions contributions difference. The estimates range from 0.0024 to 0.38 t.a⁻¹ (PM_{2.5}) and 0.0034 to 0.55 t.a⁻¹ (PM₁₀) over Wedela, while Zamdela ranges between 0.0019 and 0.54 t.a⁻¹ (PM_{2.5}) and 0.0028 and 0.77 t.a⁻¹ (PM₁₀).

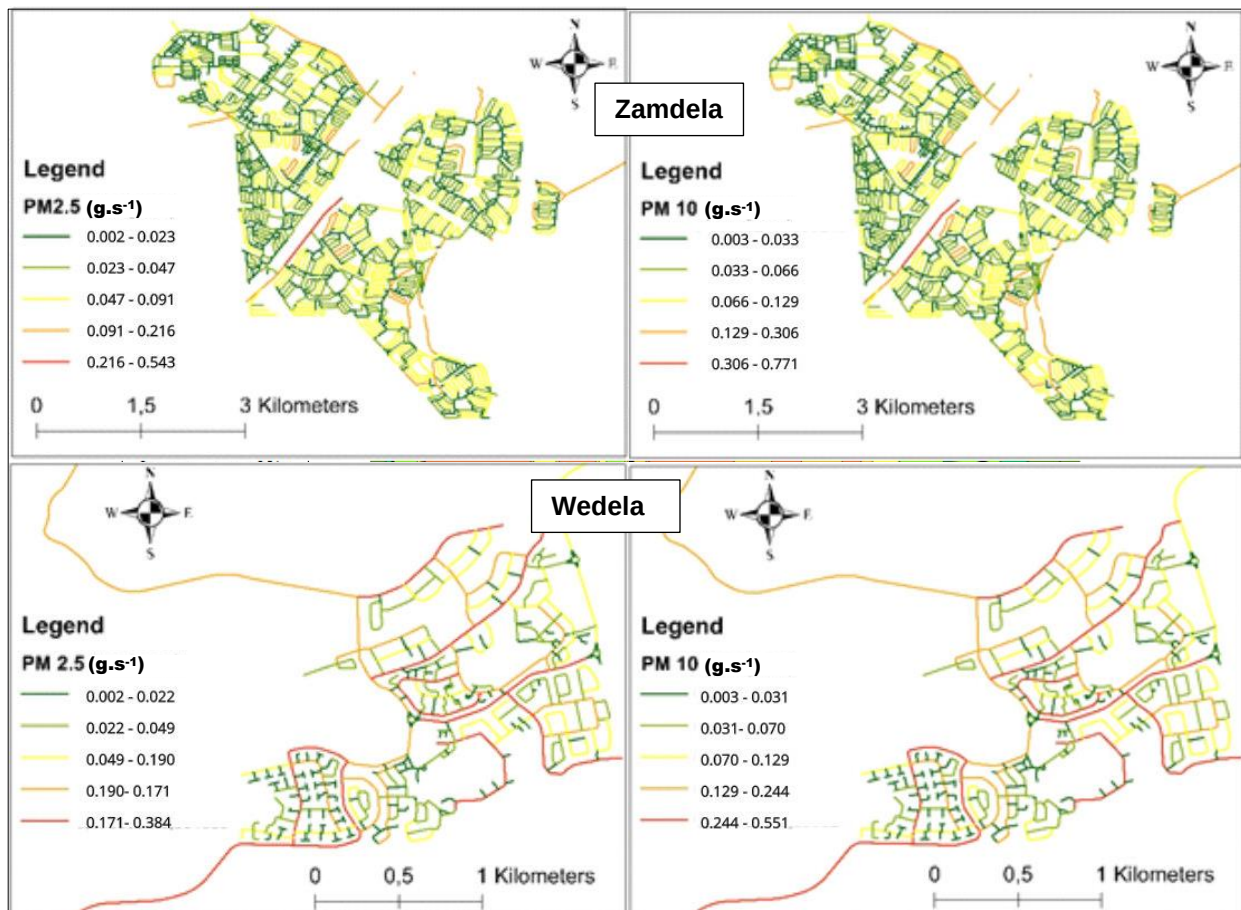


Figure 5.2-6. Spatial distribution of estimated vehicle PM emission over Zamdela and Wedela respectively used as the layout of the road sources for simulation in dispersion modelling.

Figure 5.2-7 and Figure 5.2-8 demonstrate the diurnal variations of average hourly traffic flow over the sampling periods for the Zamdela and Wedela respectively. It is evident that traffic volume has an impact on pollutant concentrations, as the traffic peaks align with PM highs observed at the monitoring sites (Figure 4.3-2). Both the curves show similar diurnal variations, with high values during the morning (6:00 - 8:00) and early evening period (16:00–18:00) as a result of a large amount of vehicle movement in the evening rush hours. In Wedela, the early afternoon peak can be attributed to the end of the school day, as well as lunchtime activities. Zamdela has a flow that is arguable constant in the afternoon, with a low around 13:00. This could be because sedans are the preferred public transport in comparison to minibuses used in Wedela, with the drivers commonly taking a break around 13:00 hence the dip inflow. The volume between 18:00 – 6:00 was assumed from the VTAPA report (2018) and Thomas (2009) since the physical counts were done for 12 hours during the day.

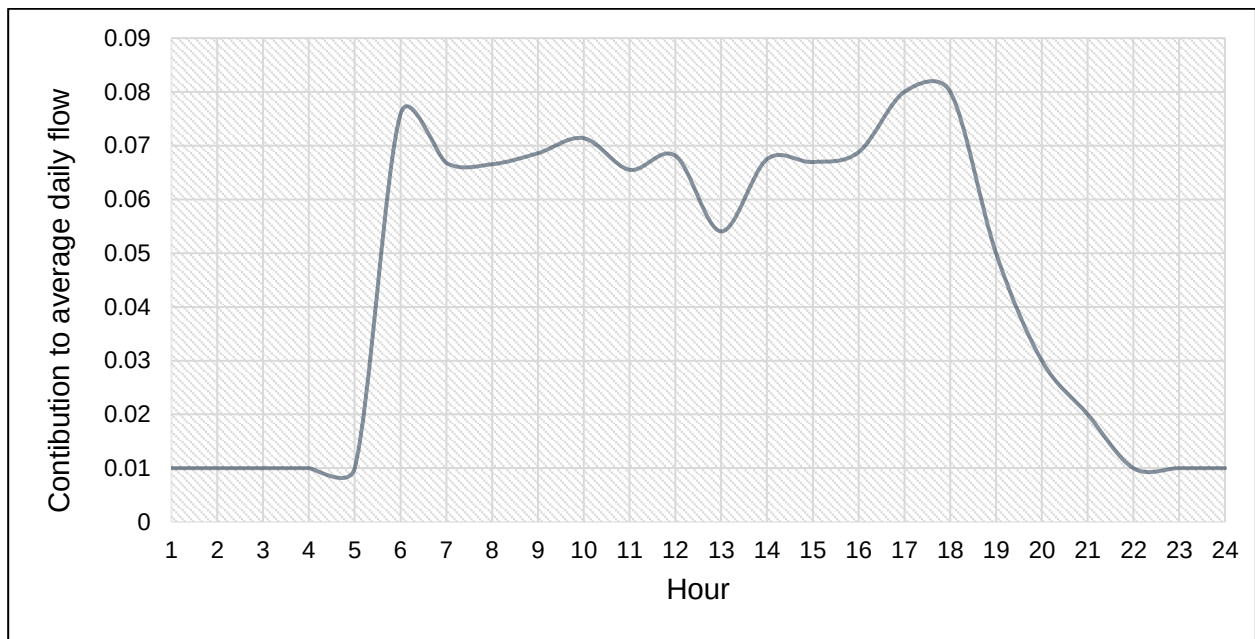


Figure 5.2-7. Diurnal profile of vehicle flow in Zamdela as obtained from vehicle counts.

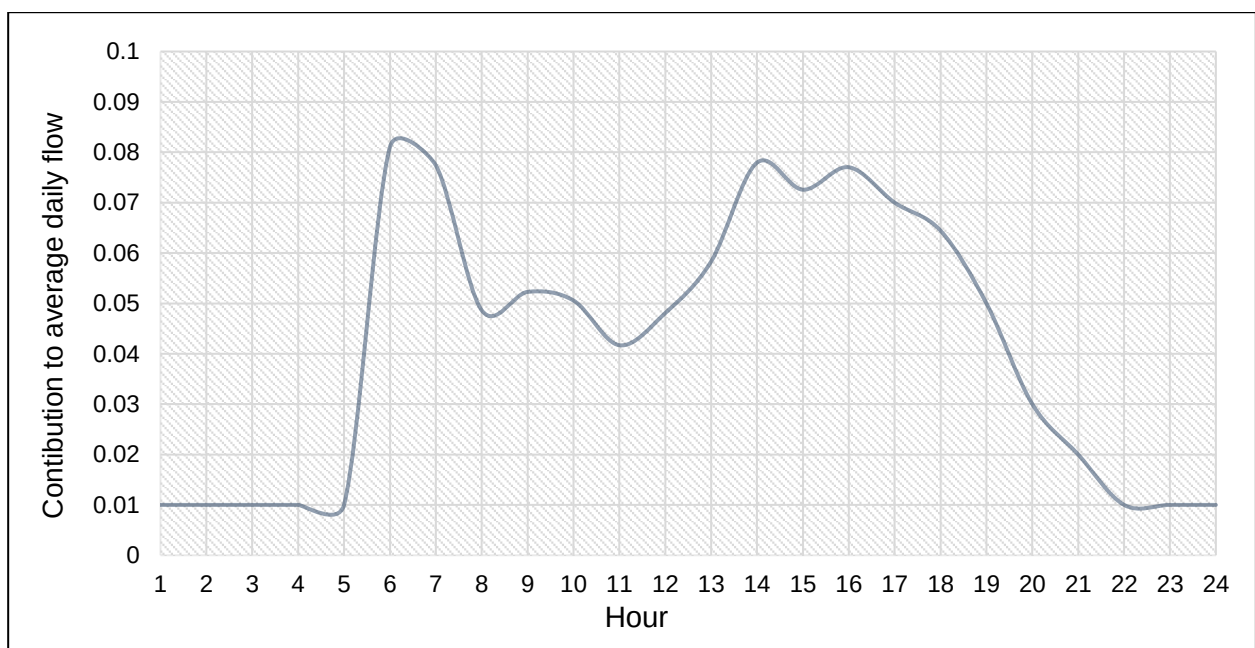


Figure 5.2-8. Diurnal profile of vehicle flow in Wedela as obtained from vehicle counts.

5.2.5 Biomass burning

Open biomass burning is an important component of the total global emissions of greenhouse gases, reactive trace gases, and PM (Wiedinmyer *et al.*, 2006). To identify fire exposed areas, the MODIS Burned Area Products instrument data was used. Data for the Southern Africa was obtained in shapefile format from (ba1.geog.umd.edu), which enabled the researcher to narrow it to the desired townships in ArcGIS (Figure 5.2-9). This allows for the estimation of emissions per fire detected for the period of study.

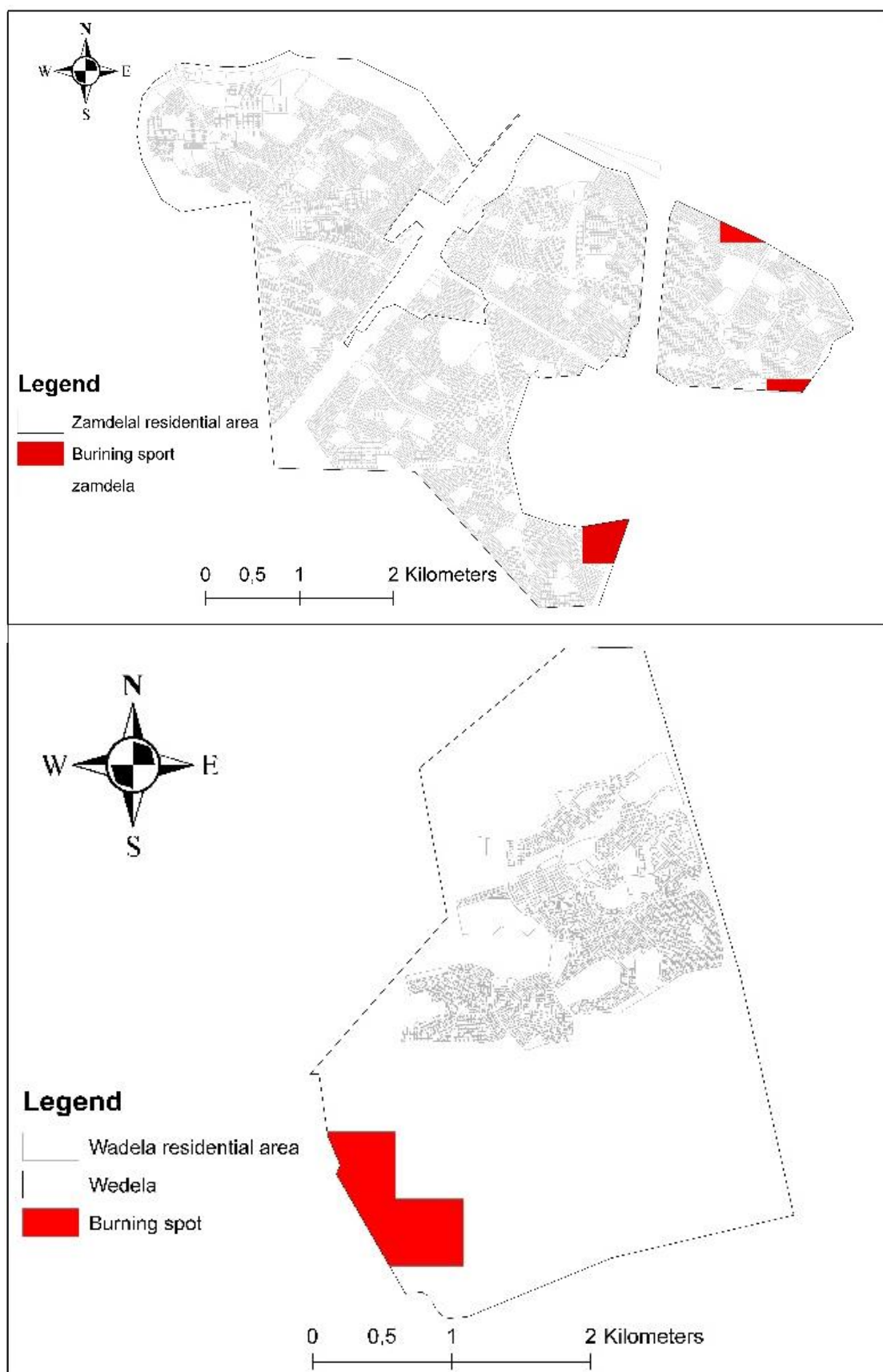


Figure 5.2-9. Maps show the extent of biomass fire burning polygons over Zamdela and Wedela respectively.

It is challenging to define the fire duration as well as to ascribe a diurnal profile for an emissions simulation. Therefore, this study used hourly smoke emissions suggested by WRAP (2005) (Figure 5.2-10). It is assumed that biomass burning exhibits a distinct diurnal cycle with peak emissions during the early afternoon (12:00 -14:00) and considerably low emissions through the night (WRAP, 2005).

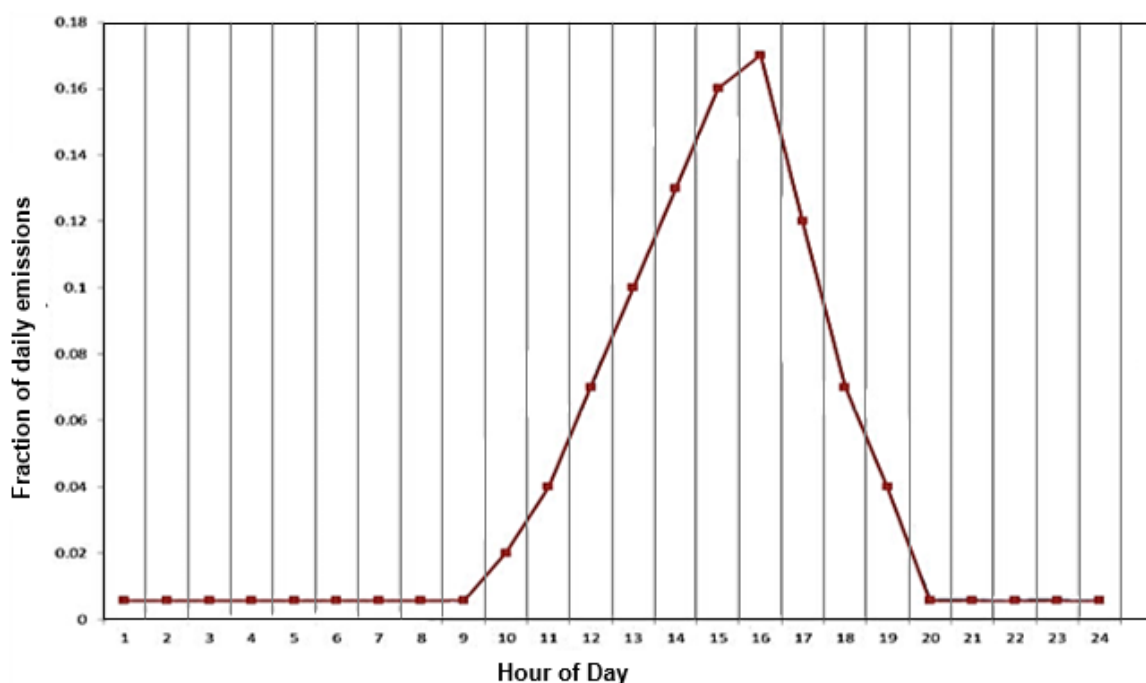


Figure 5.2-10. The diurnal variation used for biomass burning events (WRAP, 2005).

5.3 Summary of emissions quantified

The contribution of PM_{2.5} and PM₁₀ emissions from all quantified sources within Zamdela and Wedela are presented in Table 5.3-1. Vehicle emissions showed to be a significant source in both townships with the estimates comparable to studies by Thomas (2009) and DEFF (2020).

Table 5.3-1. Quantified emissions (ton per annum) per sector within Zamdela and Wedela.

Source	Zamdela		Wedela	
	PM _{2.5} (t.a ⁻¹)	PM ₁₀ (t.a ⁻¹)	PM _{2.5} (t.a ⁻¹)	PM ₁₀ (t.a ⁻¹)
Vehicles	29.1	42.9	8.6	12.4
Domestic fuel burning	74.7	44.7	1.6	1.1
Fugitive dust	25.9	73	3.3	9.5
Tailings	-	-	114.1	321.2
Waste burning	4.5	5.4	1.4	1.7
Biomass burning	3.1	3.3	5.6	6.1

Domestic burning has the highest PM_{2.5} estimates in Zamdela, as there is a rapid growth of informal housing in the area, easy availability and accessibility of solid fuels, and specifically due to the cost-effectiveness associated with coal and wood for space heating purposes. Wedela has no coal nor wood with higher paraffin use compared to Zamdela, hence domestic fuel emissions are very low. However, vehicles proved to be a significant source in both Zamdela and Wedela. This is because vehicle ownership is increasing in the country and individuals prefer to use private transport to public transport as evident in Figure 5.1-6. Compared to domestic fuel and waste burning, biomass burning is a more significant source in Wedela since there are larger vegetation-covered areas than in Zamdela, which is more urbanised. Fugitive dust from mine tailings was estimated to contribute significantly to particulate emissions, therefore rendering the other sources less noteworthy in Wedela, which is understandable given the proximity to mine tailings.

5.4 Chapter conclusions

This chapter depicted the results and discussions for the first two objectives. The researcher identified and characterised the local sources of emissions within Wedela and Zamdela, and quantified emissions for the different sources identified within the settlements.

An emission inventory was developed for identified sources. Domestic burning is a more distributed source and showed considerably high quantities which may have a wider impact on ambient air quality. On-road vehicles, a mobile source, was estimated to contribute significant PM, which, based on its spatial coverage, may affect larger areas. Fugitive dust is a significant source in Wedela due to the town's proximity to mine tailings. Biomass burning and waste burning exhibit considerable emissions, however, these emissions are localised and are likely short-lived, hence their total contribution is minimal compared to the former sources. The temporal variability in source events is also an important consideration given the impact of meteorology, radiation and the inversion layer on pollutant dispersion. The sources' input data can be accessed on <https://nextcloud.nwu.ac.za/index.php/s/R6xya86CXyYwnC>.

CHAPTER 6

This chapter presents the simulated results to determine PM concentrations within Zamdela and Wedela. Daily and annual averaging periods' isopleths plots were simulated over the period 2018 – 2020. Hourly averaged output was discretely modelled based on the location of the monitoring sites and was used to compare the simulated results to the recorded data.

6 DISPERSION SIMULATION

6.1 Atmospheric Dispersion Results

Daily and annual average simulations were done to determine the PM concentrations across Zamdela and Wedela due to the identified quantifiable sources of emissions. Plots were analysed for the daily and annual averages for which the NAAQS are available. It should be noted that, even though a high daily concentration is predicted in certain locations, it is possible that this can only be true for a single day of the entire period.

The modelled averages for daily and annual $PM_{2.5}$ over Zamdela are illustrated in Figure 6.1-1 and Figure 6.1-2. The NAAQS is highlighted in red as defined in Table 4.3-1. The simulated maximum average predictions range between 50 – 650 $\mu g.m^{-3}$ (daily) and 10 –110 $\mu g.m^{-3}$ (annual). These values fall above the stipulated NAAQS over the entire township for both averaging periods, with the highest values predicted over the central parts of the township. Given the impacts of $PM_{2.5}$ on human health, such values can have great implications and thus call for urgent interventions.

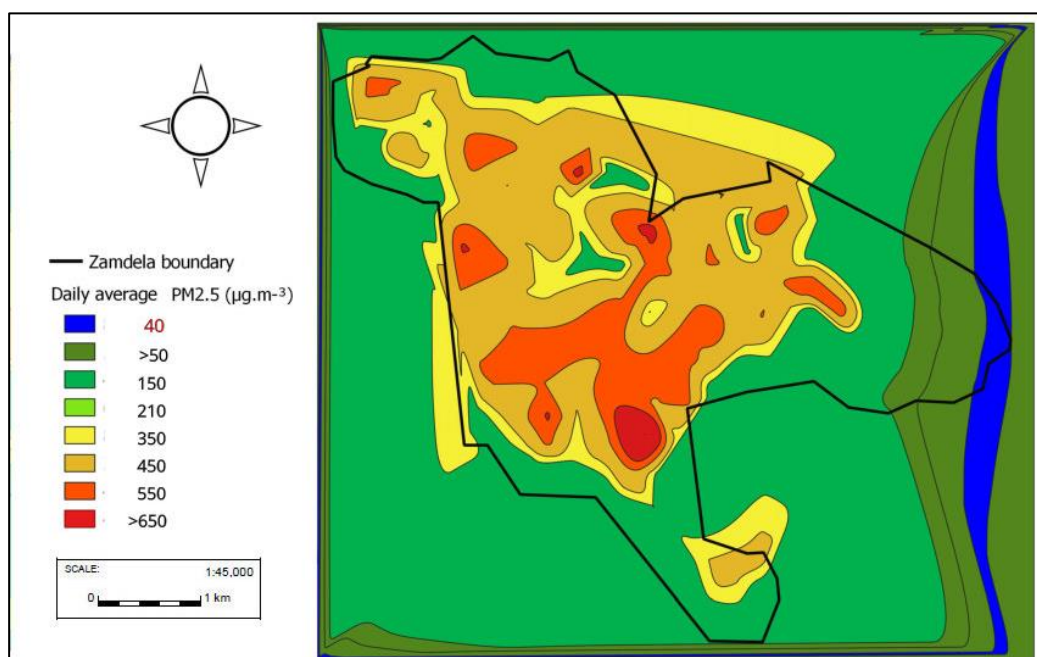


Figure 6.1-1. The spatial distribution of maximum daily average modelled $PM_{2.5}$ ($\mu g.m^{-3}$) over Zamdela due to all estimated sources. The NAAQS is highlighted in red.

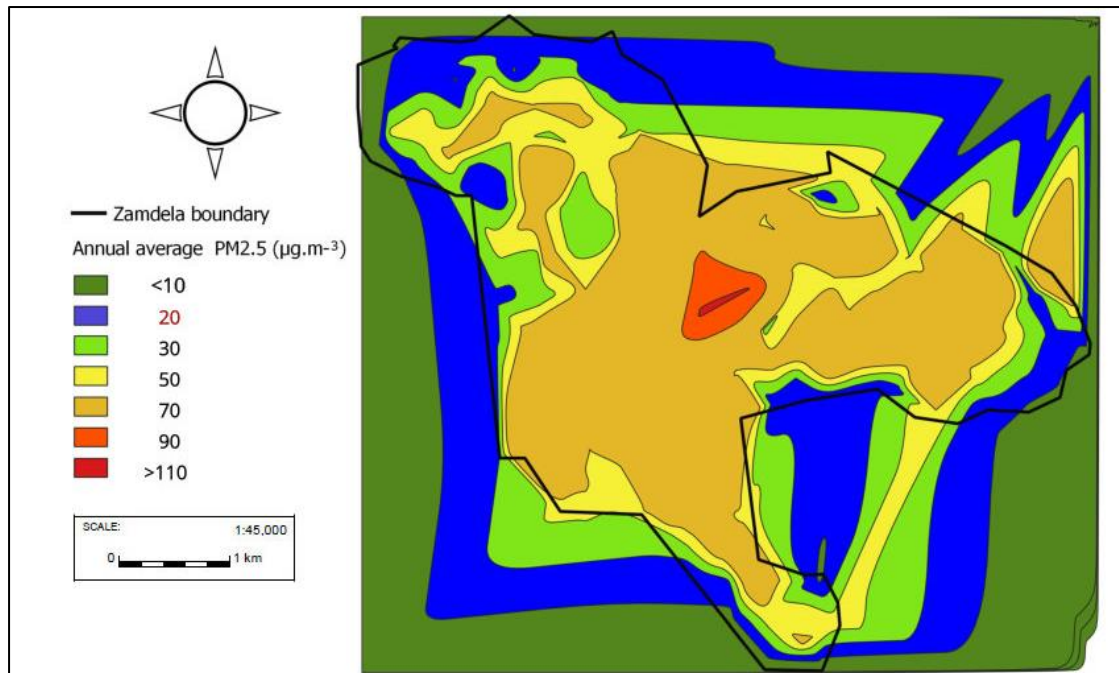


Figure 6.1-2. The spatial distribution of the annual average modelled PM_{2.5} (µg.m⁻³) over Zamdela due to all estimated sources. The NAAQS is highlighted in red.

The maximum daily and annual average PM₁₀ concentrations over Zamdela are shown in Figure 6.1-3 and Figure 6.1-4 respectively. Exceedances of NAAQS exist throughout the township, with predicted concentrations rising as high as 650 µg.m⁻³ for the 24-hour average and a maximum of 230 µg.m⁻³ for the annual average. The impact of the sources is seen to be more concentrated in the easterly direction, which is potentially due to emissions dispersing to the elevated ground more gradually under more stable or neutral conditions.

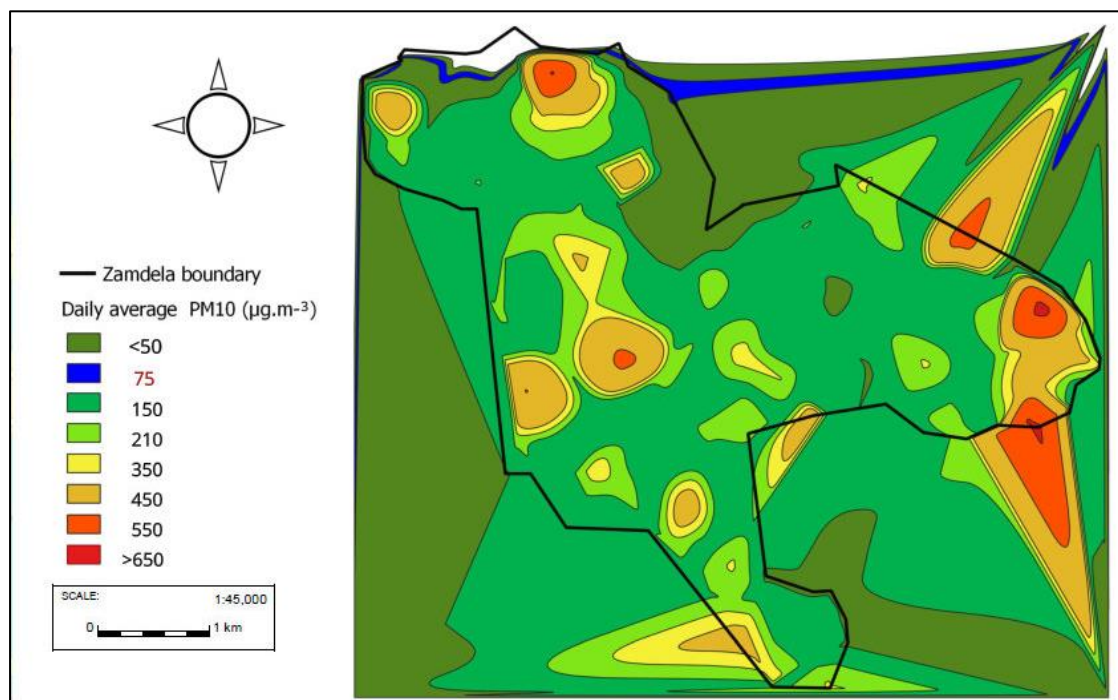


Figure 6.1-3. The spatial distribution of the maximum 24-hour average modelled PM₁₀ (µg.m⁻³) over Zamdela due to all estimated sources. The NAAQS is highlighted in red.

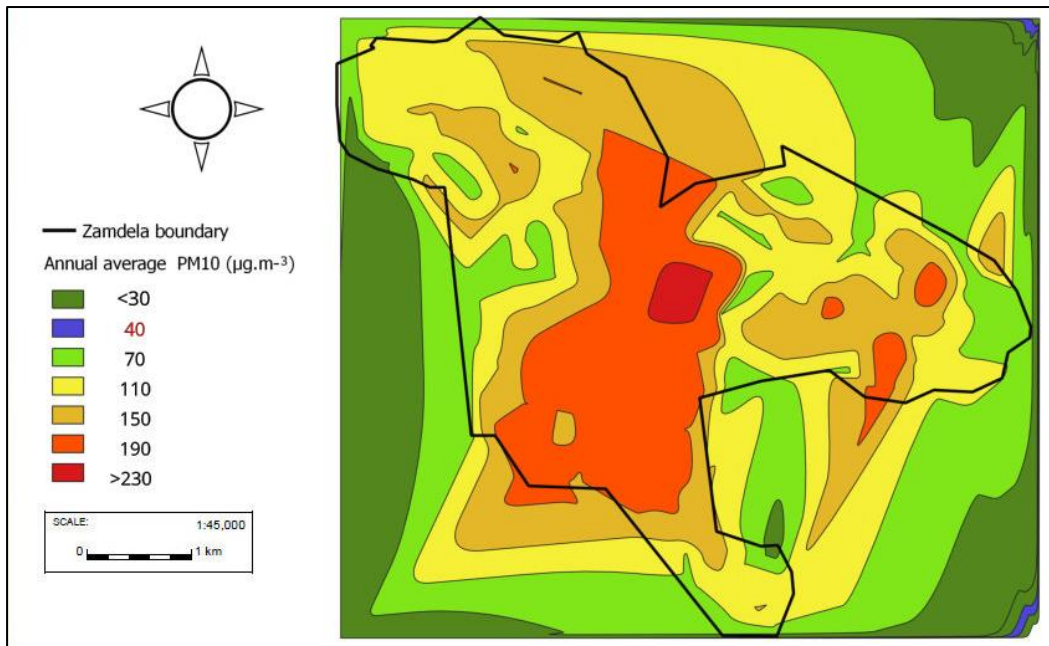


Figure 6.1-4. The spatial distribution of the annual average modelled PM_{10} ($\mu\text{g.m}^{-3}$) over Zamdela due to all estimated sources. The NAAQS is highlighted in red.

Modelled $PM_{2.5}$ over Wedela for maximum daily and annual averages are provided in Figure 6.1-5 and Figure 6.1-6 respectively. Ground-level daily concentrations are predicted to be within the NAAQS over a greater area, except for a slight exceedance in the central part of Wedela where the majority of the activities are expected to occur. Fairly low concentrations were predicted annually with a maximum of $11 \mu\text{g.m}^{-3}$ simulated, thus adhering to the NAAQS over the modelling period.

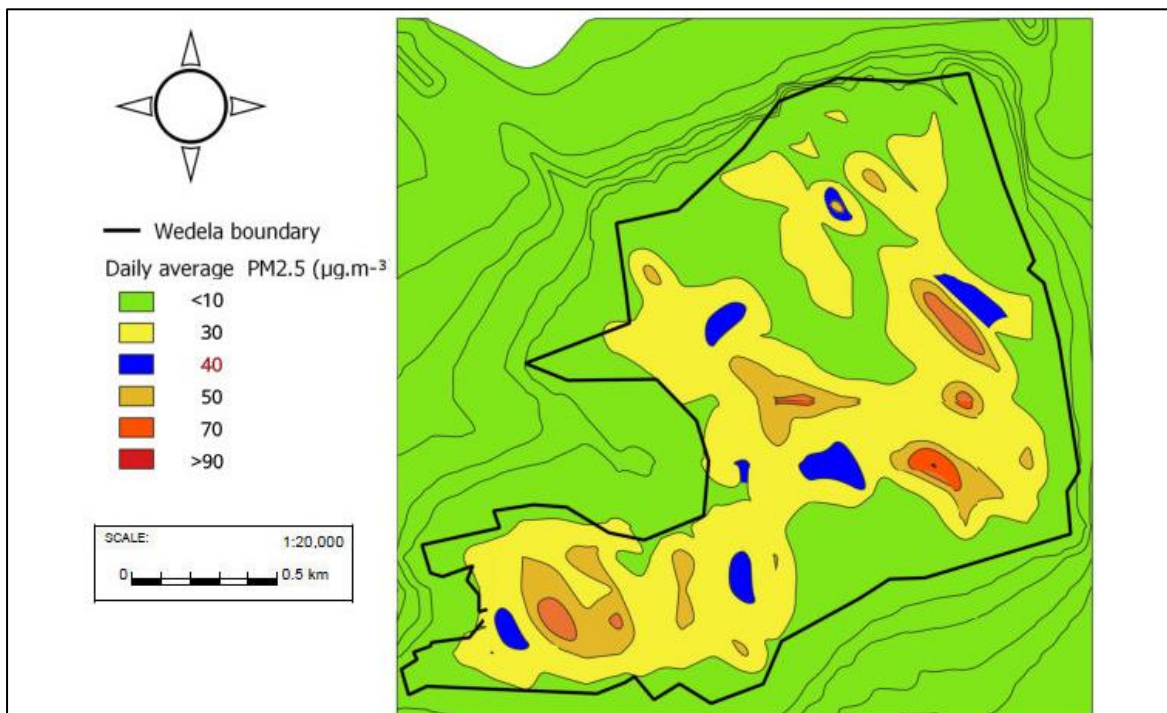


Figure 6.1-5. The spatial distribution of maximum daily average modelled $PM_{2.5}$ ($\mu\text{g.m}^{-3}$) over Wedela due to all estimated sources. The NAAQS is highlighted in red.

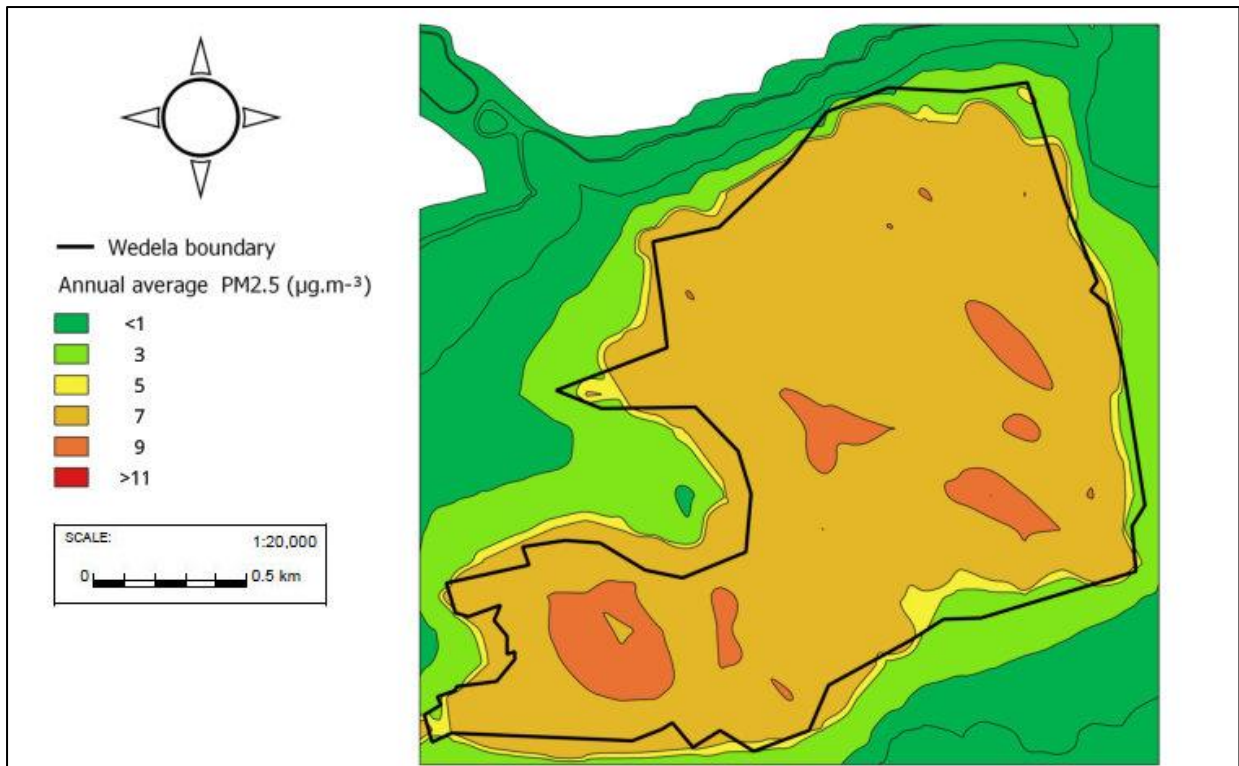


Figure 6.1-6. The spatial distribution of the annual average modelled PM_{2.5} (µg.m⁻³) over Wedela due to all estimated sources.

Figure 6.1-7 and Figure 6.1-8 illustrate the 24-hour and annual maximum modelled PM₁₀ contributions of the identified sources to concentrations over Wedela. The model predicts that concentrations over a greater area within the township fall between 90 and 180 µg.m⁻³ on an extreme day, which is over the NAAQS. On the contrary, annual simulations fall within the NAAQS over the township, even during the extreme year. The main affected areas are downwind of the mine tailings, on the northern side of the township. The cumulative impact of the tailings is confirmed by the annual averages where exceedances are predicted on the north-western side of Wedela. A large part of the predicted concentrations over Wedela is also within the same range, suggesting that it is from a single source.

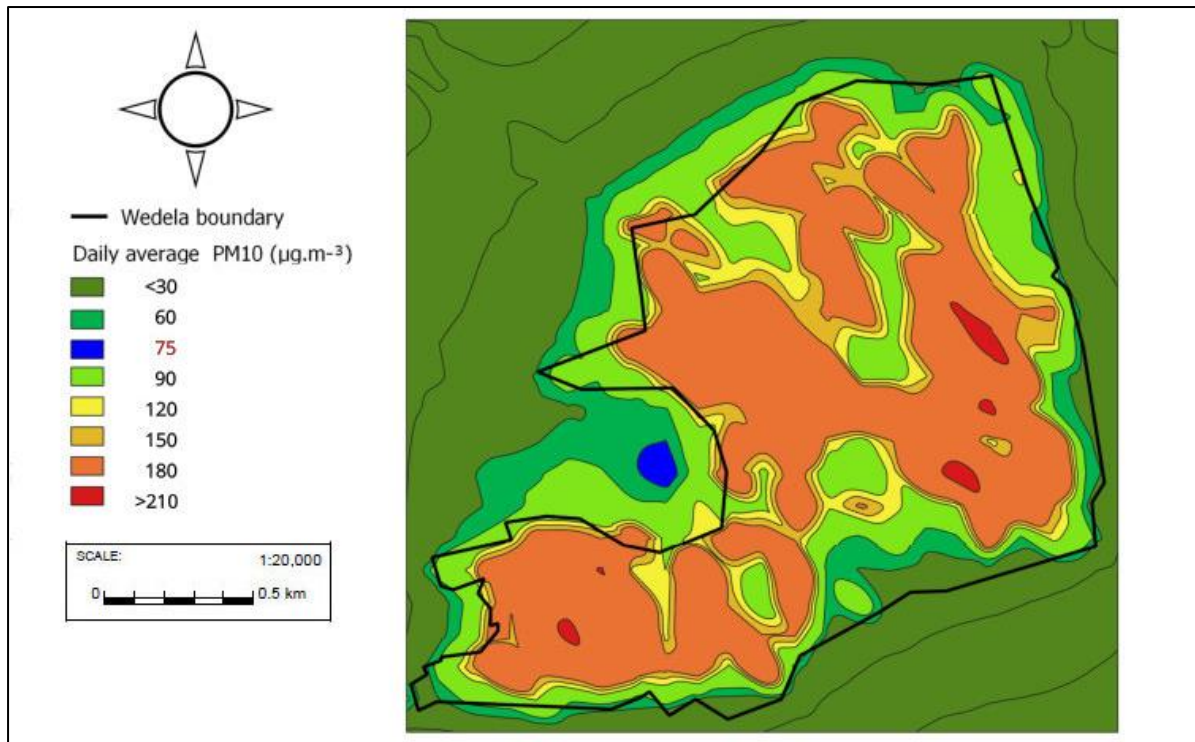


Figure 6.1-7. Spatial distribution of predicted highest daily PM_{10} ground level concentrations ($\mu\text{g.m}^{-3}$) within Wedela. The NAAQS is highlighted in red.

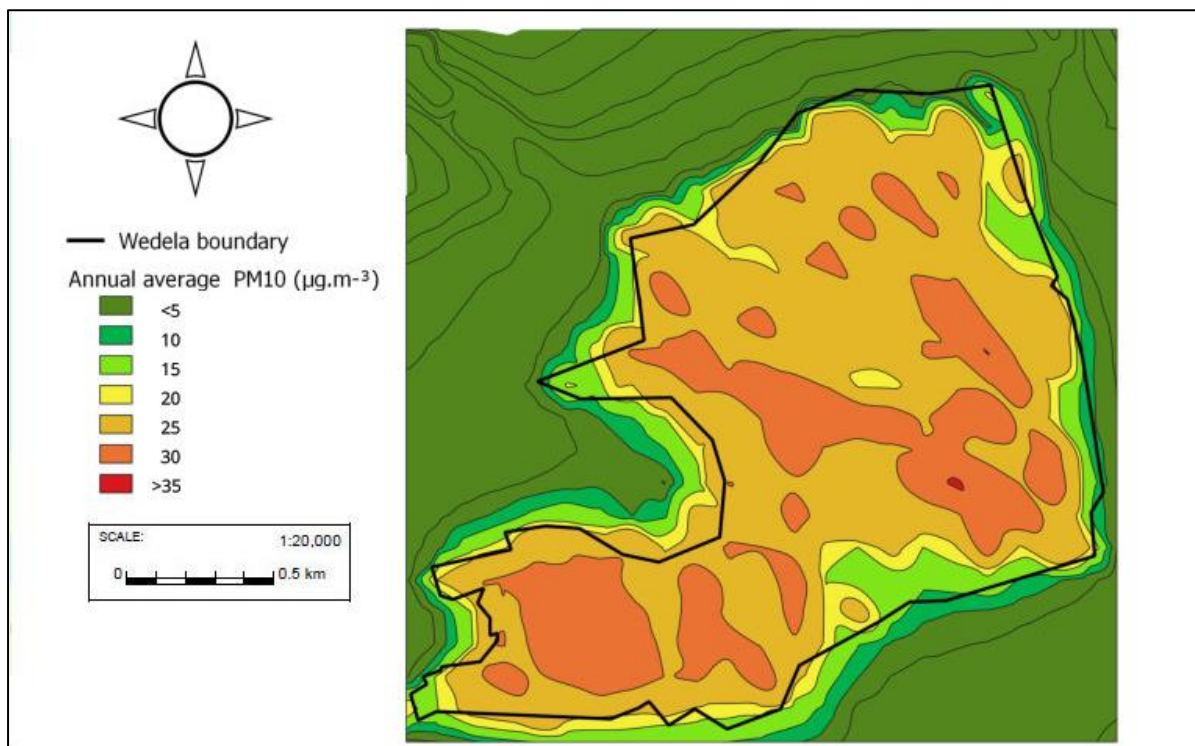


Figure 6.1-8. Spatial distribution of the predicted highest annual PM_{10} ground level concentrations ($\mu\text{g.m}^{-3}$) within Wedela.

To emphasize the significance of varying concentrations across townships, box plots were done to compare the predictions at different areas of concern. According to DEA (2014), areas of concern such as schools and public places should be considered as discrete receptors when

doing a modelling project. Figure 6.1-9 shows the location of the areas of concern used as discrete receptors across Zamdela.

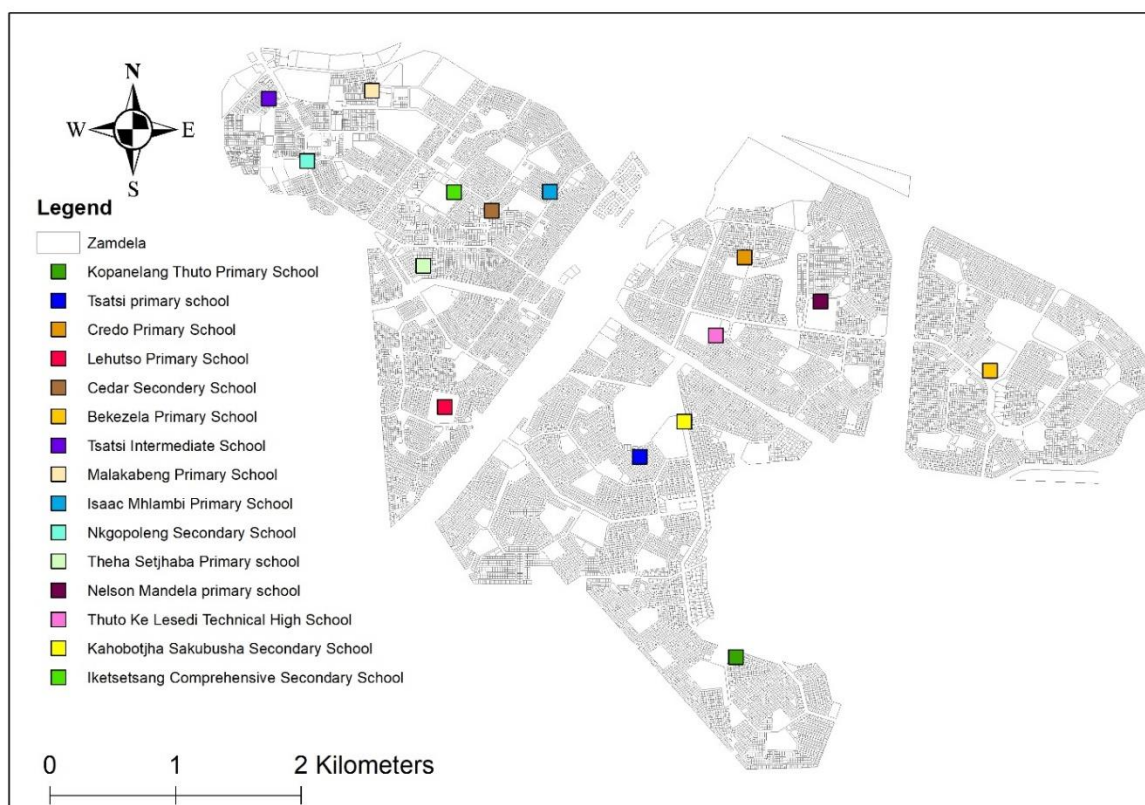


Figure 6.1-9. Map illustrating the geographic location of the areas of concern used as receptors across Zamdela.

The modelled spatial distribution of daily PM_{10} concentrations across Zamdela is shown in Figure 6.1-10. The median daily averages varied between $10 \mu g \cdot m^{-3}$ and $62 \mu g \cdot m^{-3}$ across Zamdela, which is evident of how concentrations vary significantly within a township. A study done in KwaZamokuhle using E-Bam instruments came to a common conclusion that there is large variability within townships (Moletsane, 2021). The highest levels were observed at Malakabeng Primary School (MPS) and Thuto Ke Lesedi Technical High School (TLTHS) located on the far north-west side and the centre respectively. The lowest median predicted concentrations were at Kopanelang Primary School (KPS) [$10 \mu g \cdot m^{-3}$] followed by $18 \mu g \cdot m^{-3}$ at Nkgopoleng Secondary School (NSS).

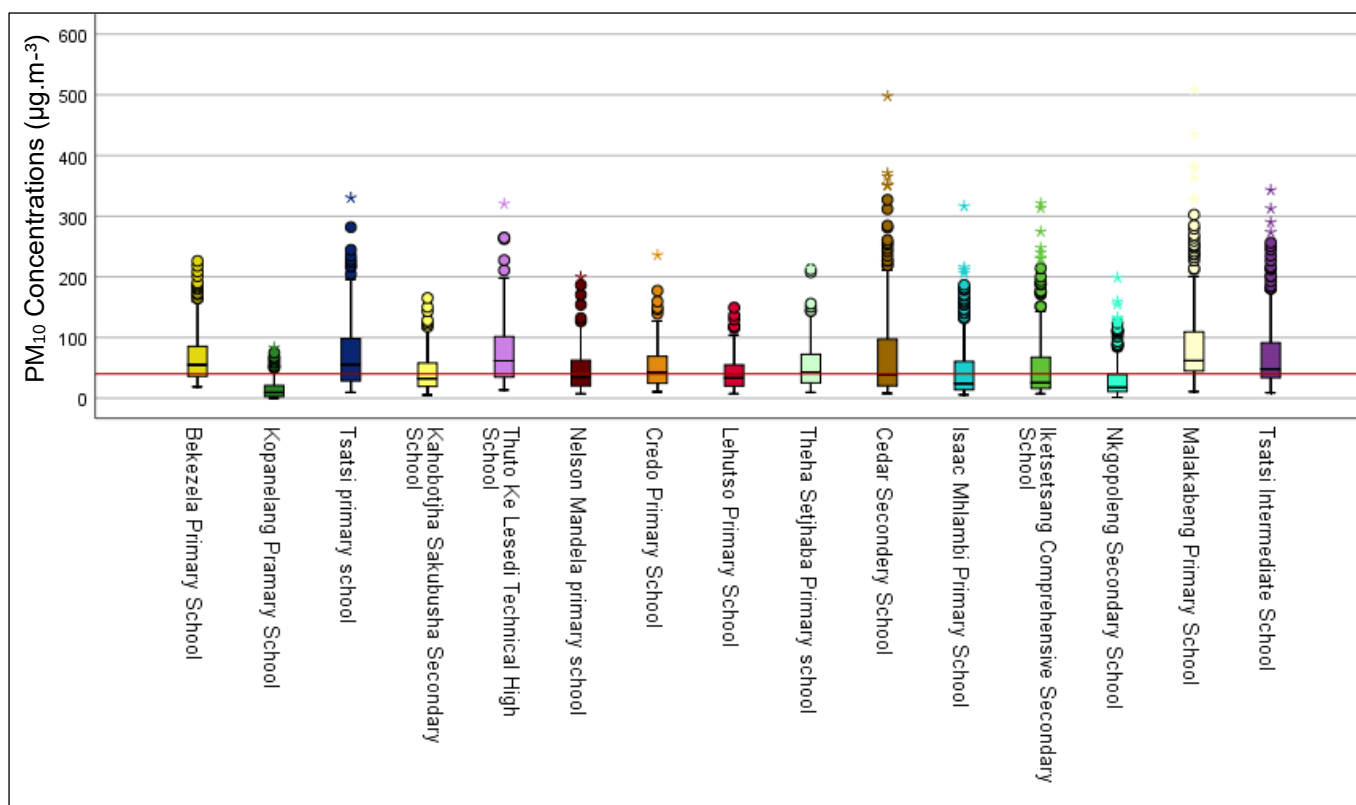


Figure 6.1-10. Spatial variation of the daily averaged ambient PM_{2.5} concentrations across Zamdela

Figure 6.1-11 shows the location of the areas of concern used as discrete receptors across Wedela.

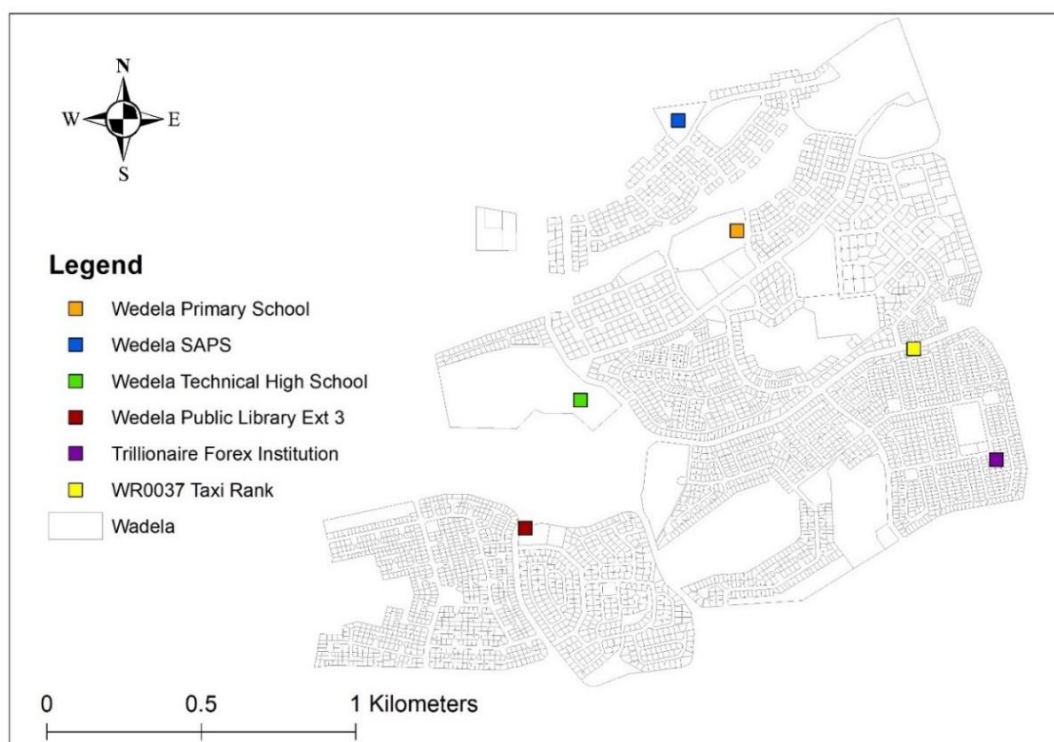


Figure 6.1-11. Map illustrating the geographic location of the areas of concern used as receptors across Wedela.

The simulated spatial distribution of the daily average ambient PM₁₀ across Wedela is shown in Figure 6.1-12. The daily PM₁₀ predicted concentrations show considerable spatial variability with the highest medians observed at Trillionaire Forex Institution (TFI) – (232 µg.m⁻³) followed by Wedela Public library (WPL) – (147 µg.m⁻³), Taxi Rank WR0037 (TRW) – (113 µg.m⁻³), Wedela Technical High School (WTHS) – (109 µg.m⁻³), Wedela Primary School (WPS) – (56 µg.m⁻³) and Wedela SAPS (WS) – (22 µg.m⁻³) (Figure 6.1-12). The variation is justified given the distance of the receptors from the major sources identified as mining tailings.

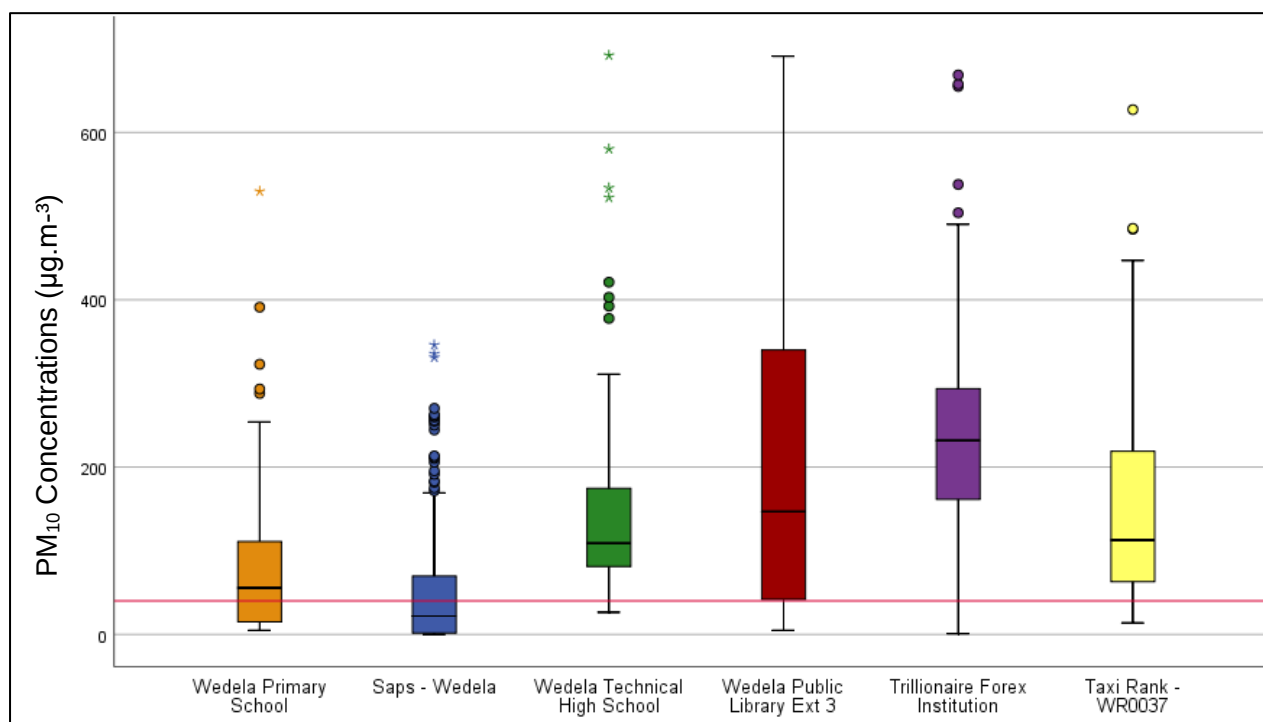


Figure 6.1-12. Spatial variation of the daily averaged ambient PM₁₀ concentrations across Wedela.

Figure 6.1-13 and Figure 6.1-14 show the contribution of the individual identified sources as simulated at the discrete receptors, which are located at the same position as the Zamdela monitoring station. As expected for a low-income settlement, domestic burning shows the highest contribution to both PM size fractions, followed by vehicle emissions contributing at least a quarter to the total measurements.

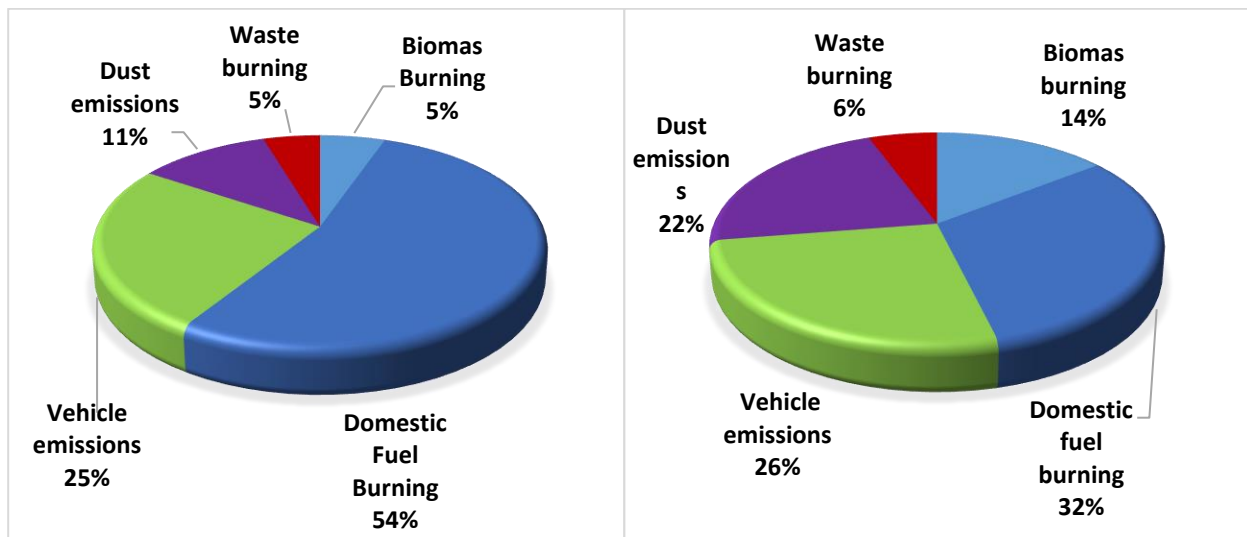


Figure 6.1-13. Modelled PM_{2.5} and PM₁₀ concentrations from individual identified sources simulated at Zamdela monitoring station.

Figure 6.1-14 illustrates the PM contributions of the identified sources to concentrations at the Wedela monitoring station. As anticipated, dust emissions contribute significantly at over 40% in the simulations, due to Wedela being surrounded by mine tailings and bare areas. Vehicles have the second-highest PM contribution, as vehicle ownership is on the rise in South Africa and individuals prefer to use their cars over public transport.

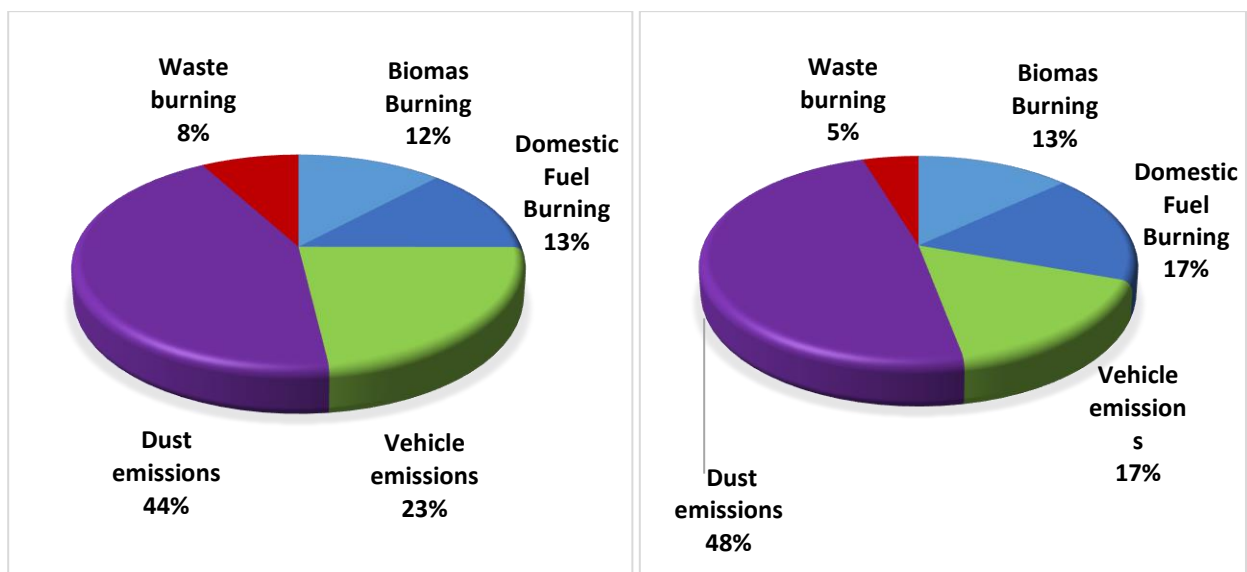


Figure 6.1-14. Modelled PM_{2.5} and PM₁₀ concentrations from individual identified sources simulated at Wedela monitoring station.

6.2 Dispersion modelling results validation

Concentrations recorded at the respective monitoring stations (assumed as discrete sensitive receptors in the model setup) were used to validate the simulated data. The simulated results are generally based on constant source emission rates (Thomas, 2009; Hertwig *et al.*, 2018),

however, in this study, variable emission factors were based on hours and seasons to emulate reality.

The measured and modelled concentrations were compared for the highest 95th percentile hourly, daily, and annual averages (Table 6.2-1). The comparison was made at the 95th percentile since the model generally overpredicts along the plume centreline (Hertwig *et al.*, 2018). The ratio between measured and monitored PM is also presented, considering the range of uncertainty recommended by the US EPA of >0.5 <2.0 (modelled to monitored ratio).

The simulated ground level concentrations compared relatively well with measured ambient PM throughout the averaging periods with the ratios falling within the recommended range. The lowest ratio of 0.75 (10 $\mu\text{g.m}^{-3}$ - modelled and 13 $\mu\text{g.m}^{-3}$ - monitored) and the highest was 1.86 (99 $\mu\text{g.m}^{-3}$ - modelled and 53 $\mu\text{g.m}^{-3}$ - monitored) were estimated over Wedela for PM_{2.5}.

The general model bias is to underpredict the long-term averaging periods, i.e., annual averages, which was evident for this study. There is also a suggestion of overprediction of the 24-hour averages, which for Zamdela could be attributed to the threshold assumed for the monitoring sensors. There is also a possibility that the emissions' zones of impact differ between monitored and modelled data due to the upper air meteorological parameters that were acquired from an alternative source owing to the unavailability of the data for the study areas.

Table 6.2-1. Comparison of measured and predicted PM averages within Zamdela and Wedela.

			Hourly	Daily	Annual
Zamdela	PM _{2.5}	Monitored	51053 $\mu\text{g.m}^{-3}$	140 $\mu\text{g.m}^{-3}$	54 $\mu\text{g.m}^{-3}$
		Modelled	870 $\mu\text{g.m}^{-3}$	207 $\mu\text{g.m}^{-3}$	64 $\mu\text{g.m}^{-3}$
		Ratio	1.71	1.48	1.19
	PM ₁₀	Monitored	801 $\mu\text{g.m}^{-3}$	206 $\mu\text{g.m}^{-3}$	101 $\mu\text{g.m}^{-3}$
		Modelled	1220 $\mu\text{g.m}^{-3}$	279 $\mu\text{g.m}^{-3}$	94 $\mu\text{g.m}^{-3}$
		Ratio	1.52	1.35	0.93
Wedela	PM _{2.5}	Monitored	53 $\mu\text{g.m}^{-3}$	41 $\mu\text{g.m}^{-3}$	13 $\mu\text{g.m}^{-3}$
		Modelled	99 $\mu\text{g.m}^{-3}$	48 $\mu\text{g.m}^{-3}$	10 $\mu\text{g.m}^{-3}$
		Ratio	1.86	1.16	0.75
	PM ₁₀	Monitored	192 $\mu\text{g.m}^{-3}$	97 $\mu\text{g.m}^{-3}$	38 $\mu\text{g.m}^{-3}$
		Modelled	219 $\mu\text{g.m}^{-3}$	104 $\mu\text{g.m}^{-3}$	36 $\mu\text{g.m}^{-3}$
		Ratio	1.14	1.08	0.94

The monitored and modelled PM concentrations results were compared to one another in Figure 6.2-1 to Figure 6.2-4 using scatter plots. The analysis showed a positive correlation between the

modelled and monitored concentrations for both townships. A weak correlation is seen between modelled and monitored PM_{2.5} for Wedela. This could be because only primary emissions were considered, and secondary particulates were excluded (Figure 6.2-1).

The model simulated the lower values well compared to higher concentrations, where the model overestimated as it continuously shows higher than the monitored values (Figure 6.2-1 to Figure 6.2-4). This however is ideal, as the expected performance of the model is to overestimate rather than underestimate, and consequently, the simulations would be conservative, thus ensuring compliance with ambient air quality regulatory frameworks and policy standards (DEA, 2014).

The overall performance of the model is positive; thus, it can be argued that the model can make a valuable contribution to air quality predictions for policy and regulatory purposes.

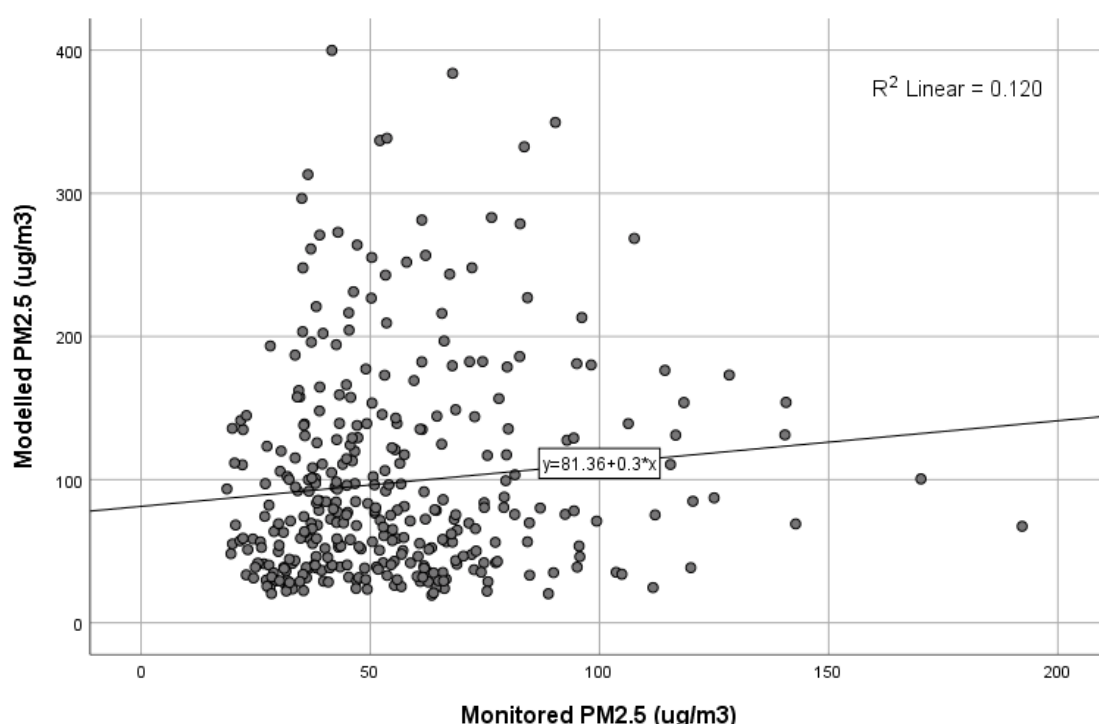


Figure 6.2-1. Scatter plot of monitored against modelled daily average PM_{2.5} concentrations for Zamdela.

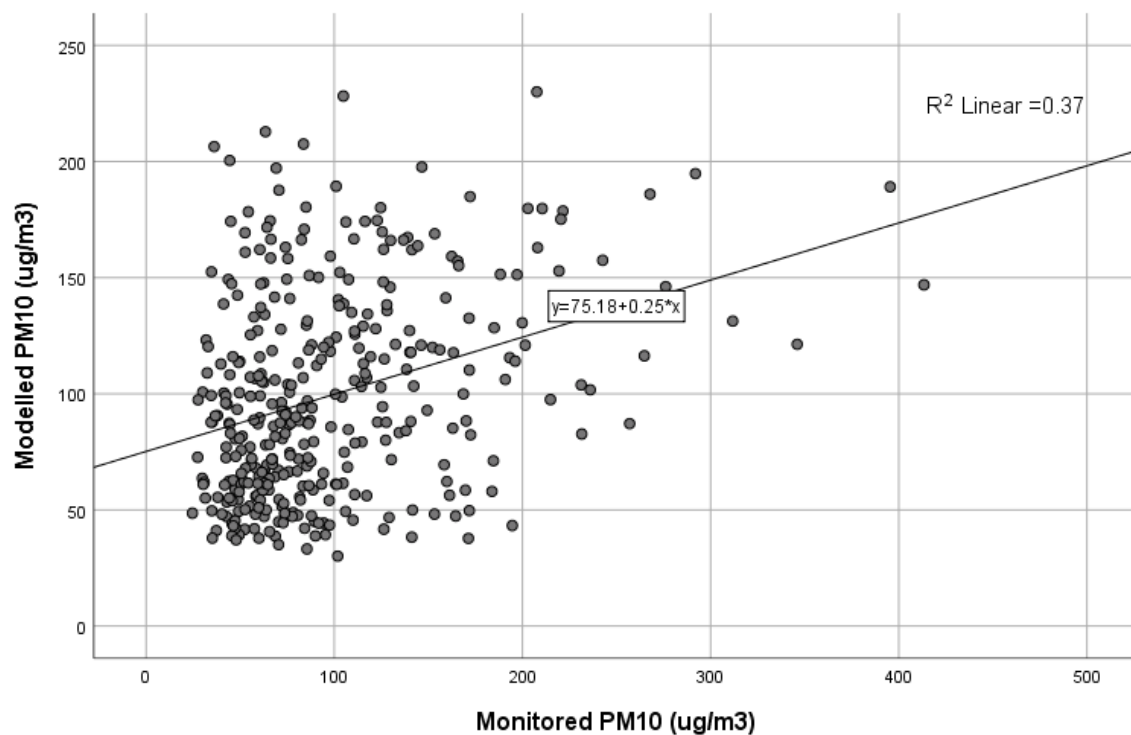


Figure 6.2-2. Scatter plot of monitored against modelled daily average PM₁₀ concentrations for Zamdela.

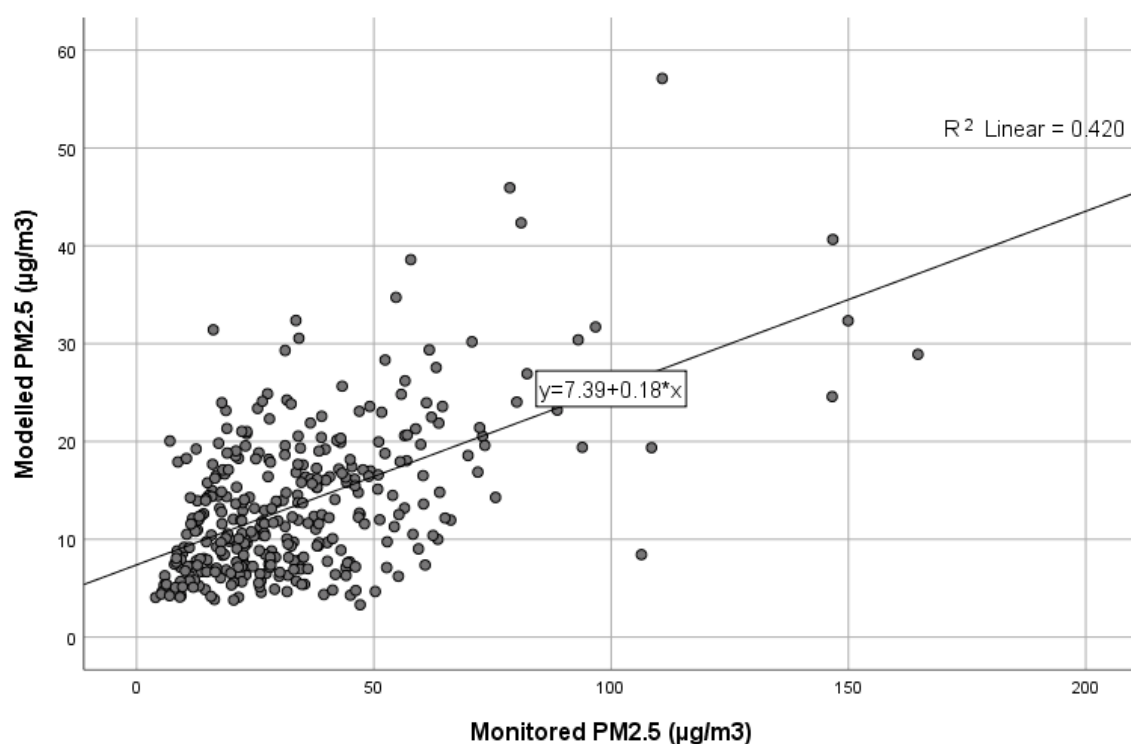


Figure 6.2-3. Scatter plot of monitored against modelled daily average PM_{2.5} concentrations for Wedela.

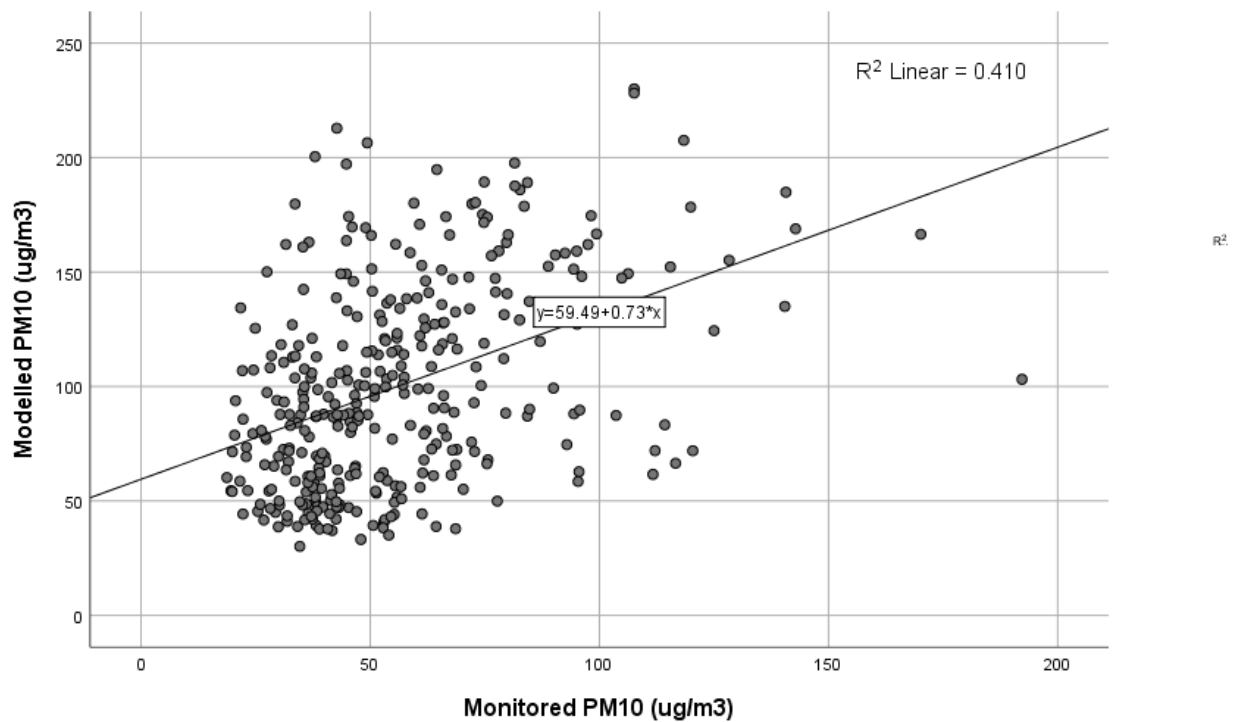


Figure 6.2-4. Scatter plot of monitored against modelled daily average PM₁₀ concentrations for Wedela.

According to Carslaw (2015), variations of pollution by time can be useful in revealing likely sources. The plots were generated at a 95% confidence interval in the mean, applying the normalised option. Normalisation requires adjusting values with different scales to an estimated common one (Carslaw, 2015), which is useful in comparing the patterns of datasets that had distinct concentration ranges.

Figure 6.2-5 show the temporal variation of the modelled results in comparison to the measured data from the Zamdela monitoring station. The diurnal plots for both PM sizes show an agreement between the monitored and modelled results, with both peaking during the early morning (06:00–09:00) and early evening (17:00–20:00), a pattern generally associated with domestic fuel burning events and vehicle emissions (Nkosi, 2018). The same is seen throughout the week with the weekends showing relatively lower variations suggesting the slowing down of events as people are generally resting. The monthly plots show peaks during the winter season, which are aligned to the monitored maxima, but failed to show the spring peaks. The peaks can be attributed to increased fuel burning events for space heating and can occur when there is little to no rainfall, resulting in easier entrainment of dust into the atmosphere. The failure to show the September peak demonstrates that the simulated values prove to be more sensitive to time variables in source activities, and unsuccessfully simulated wind impact.

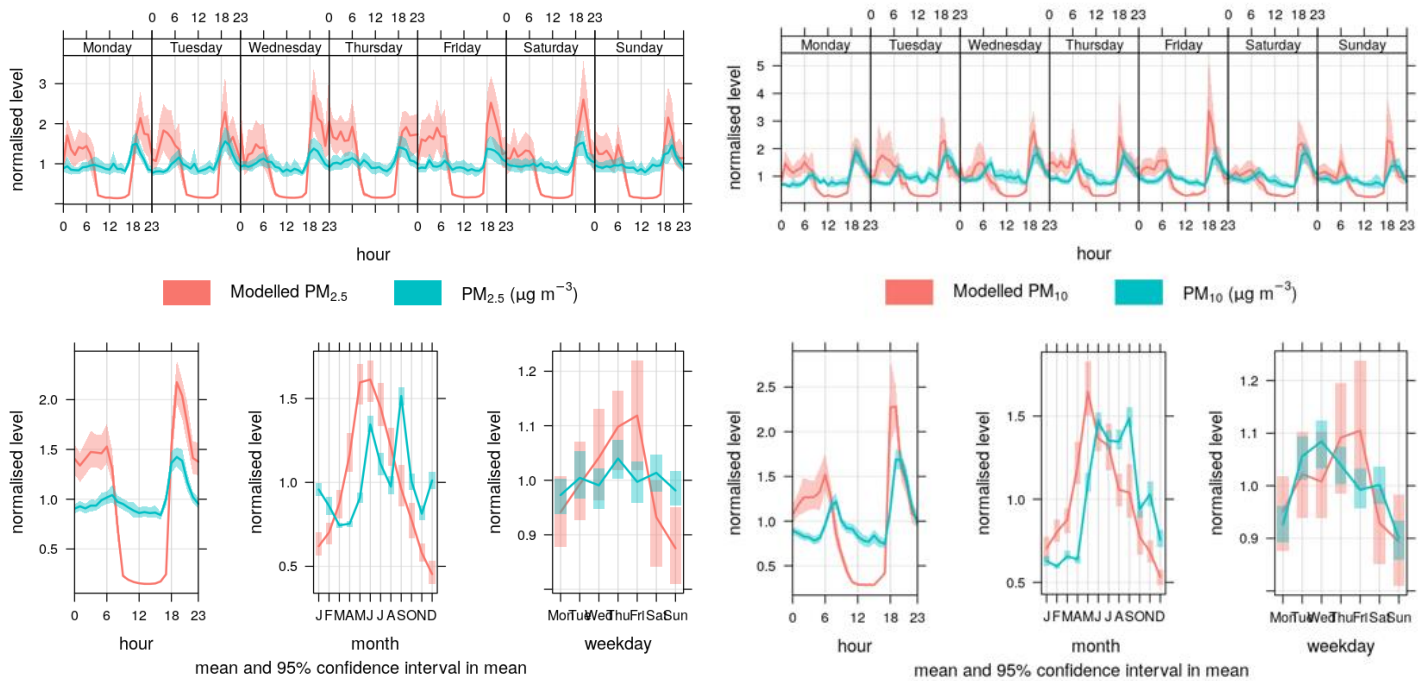


Figure 6.2-5. Temporal variation plots for normalised PM_{2.5} and PM₁₀ concentrations respectively comparing measured and simulated data over Zamdela.

Figure 6.2-6 shows the temporal variation at the Wedela site over the study period. Unlike at Zamdela, the diurnal patterns for PM_{2.5} and PM₁₀ in Wedela are distinct, however, the variations between the monitored and modelled concentrations for both particulate sizes show agreement. PM_{2.5} depicts a bimodal pattern that is associated with domestic fuel burning, waste burning and traffic events. On the contrary, PM₁₀ reaches a maximum during the afternoon around 14:00 which can be attributed to the breaking of the inversion layer leading to downward mixing of particles. This is also the time when the wind impact is at its peak, therefore the peaks are suggested to be a result of dust sources. The variation is uniform throughout the week with low measurements observed during the weekends as activity slows down. The monthly variations showed little agreement. For both PM sizes, monitored observations recorded the highest concentrations during the windy periods (August and September). This is the time when fugitive dust emissions are at their peak. The simulated values proved to be more sensitive to variations in source activities than the measured data for the diurnal profile and failed to simulate wind impact. This is a general AERMOD bias as it underpredicts the impact of winds.

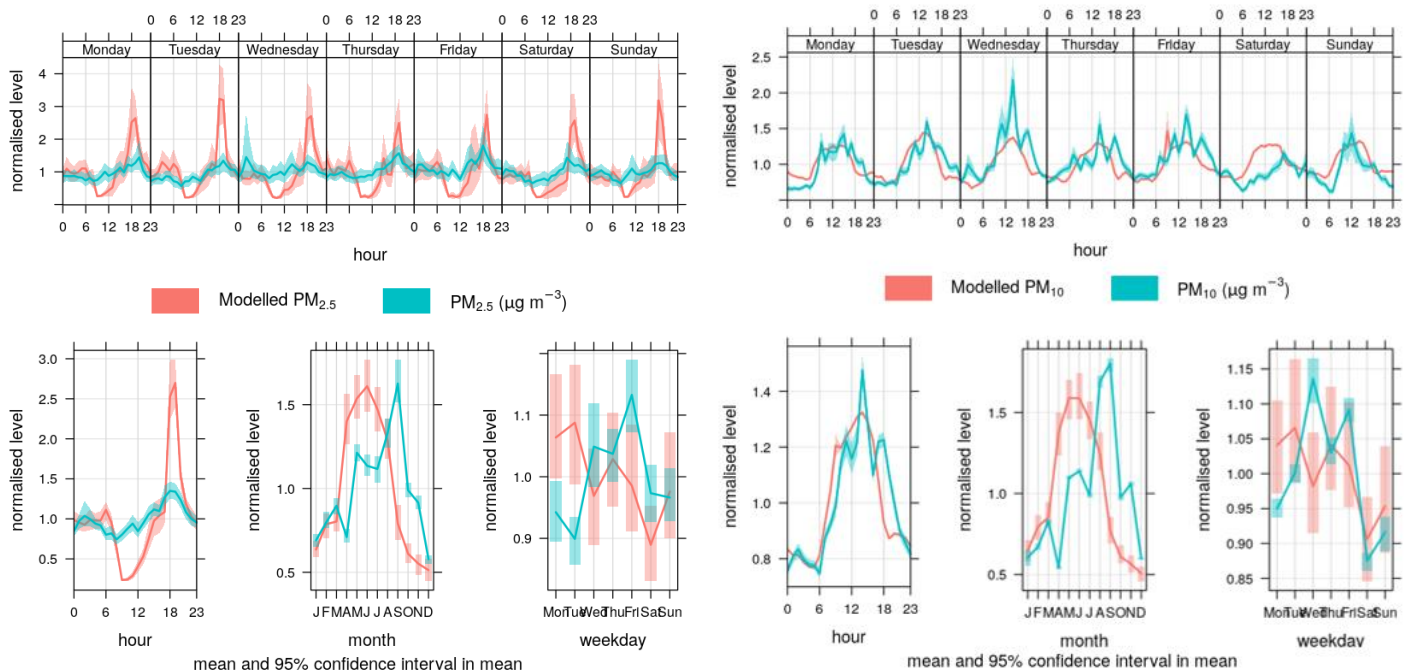


Figure 6.2-6. Temporal variation plots for normalised PM_{2.5} and PM₁₀ concentrations respectively comparing measured and simulated data over Wedela.

6.3 Chapter conclusion

The ability to accurately model and predict the concentration is crucial for the development of effective air quality management and regulations. This chapter presented the spatial distribution of PM over Zamdela and Wedela, as simulated by AERMOD, as well as made a comparison between monitored and modelled PM concentrations. Maximum PM concentrations over Zamdela were predicted in the centre of the township with simulations exceeding the NAAQS over the entire township for both averaging periods. Wedela, on the contrary, shows PM_{2.5} adherence for both averaging times over a greater part of the township, while PM₁₀ tends to exceed the daily averages. When compared to the monitored data, modelled data showed a positive agreement with the ratios falling between the US EPA recommended range. The time variations also showed a correlation, suggesting that the modelled sources are major contributors to measured data. Therefore, it is imperative to conclude that with proper inputs and procedures, dispersion modelling can aid as a regulatory tool in managing air quality.

CHAPTER 7

This chapter concludes the main findings of this study, indicating the applicability of dispersion modelling as a suitable regulatory tool. The aim of the research is recapitulated and a link between the findings to the objectives is given.

7 CONCLUSIONS

This study successfully demonstrates that with appropriate parameters and appropriate inputs, it is conceivable to use dispersion models to simulate intra-urban emissions and to guide regulations and interventions.

PM has acknowledged impacts on human health and generally on the environment. The impacts mostly affect the low-income residents who make use of dirty fuels for affordability reasons, coupled with poor service delivery, resulting in waste burning and dust emissions from unpaved roads and bare, open spaces. Consequently, airsheds in such environments are heterogeneous with concentrations varying significantly over space and time. This study, therefore, evaluated the use of dispersion models as regulatory tools in modelling emissions within low-income settlements to better understand the contributions of local sources in intra-urban environments. The first part of the study analysed the ambient data over the study areas, and subsequently, the potential sources of PM in the respective townships were identified and characterised. This included an estimation of emissions from the identified sources, followed by an evaluation of the impact of the sources on the ambient air over the townships. Finally, a comparison of modelled results to the monitored concentrations was completed.

7.1 Measured ambient data

Based on the available monitored data, Zamdela in comparison to Wedela is a settlement in need of imperative interventions as the PM levels exceed the NAAQS almost all the time for both the daily and annual averaging periods. Wedela on the other hand showed elevated PM₁₀ readings especially in the dry seasons, indicating why the areas are significant to research. The main findings in this section indicate that:

- I. PM concentrations are elevated over Zamdela owing to the household use of solid fuels, the ever-rising vehicle use, and its proximity to industrial operations, amongst other possible sources. This is evident as the measured particulates are over the NAAQS more than half the time for both PM concentrations over the daily and annual set maximum allowed concentrations.

- II. Wedela monitoring station showed adherence to the NAAQS almost all the time for PM_{2.5} with the observed 24-hour exceedances just meeting the allowed exceedances (4) per year, while PM₁₀ exceeded 4 times more than the acceptable exceedance standard.
- III. A bimodal diurnal pattern is observed at Zamdela with early morning (6:00 – 10:00) and evening (17:00 -21:00) peaks detected, with a prominent low-income settlement's seasonal variation that peaks in winter attributed to seasonally influenced burning events.
- IV. A similar pattern to Zamdela is observed for PM_{2.5} over Wedela, while distinct peaks during the mid-day (10:00 – 16:00) are observed, potentially due to wind-governed sources. This is confirmed by the seasonal variations that peak during the windy seasons.
- V. Meteorological parameters showed little correlation to the PM concentration showing that the main potential sources are likely anthropogenically influenced.

7.2 Identification and characterisation of local sources

A PM emissions inventory for this study was established for sources identified within the study areas (intra-urban). A combination of primary and secondary evidence led to the conclusions below:

- I. The various evidence agreed that the main sources likely to contribute to cumulative PM within the townships are anthropogenic activities.
- II. Domestic fuel burning remains a noteworthy source in Zamdela as it remains significant despite the different intervention programs.
- III. In Wedela fugitive dust, especially from tailing facilities, was identified as the major source of PM over the township.
- IV. Due to poor service delivery, residents in both towns resort to burning their waste as garbage disposal.
- V. Vehicles were also identified as sources that continue to increase with modernisation, and significantly, in low-income towns, the vehicles are usually not well maintained resulting in high emissions.
- VI. Biomass burning was also acknowledged to cause elevated concentrations in the towns, especially in the spring season.

7.3 Estimation of emissions

PM contributions were quantified for the identified sources within the study areas for the period 2018 – 2020. Emissions were estimated based on the product of observed and assumed activity rates and emission factors acquired from previous studies. The inferences drawn from the results include:

- I. Domestic fuel burning contributes the highest estimated PM_{2.5} over Zamdela, given its accessibility in the township.
- II. Fugitive dust was estimated to contribute the most to PM₁₀ concentrations over the two towns as well as PM_{2.5} over Wedela where it is a major source owing to its proximity to mining activities.
- III. Vehicles were estimated to be the third-highest contributor in both towns, followed by waste burning and biomass burning over Zamdela, while biomass burning was greater than waste burning in Wedela.
- IV. It was observed that source activities vary significantly diurnally and seasonally owing to factors such as the impact of wind, change in burning events due to seasons, as well as the start and end of workday hence vehicle peaks.

7.4 Simulated impact of the sources on the ambient air

AERMOD was used to model the dispersion of PM_{2.5} and PM₁₀ over Zamdela and Wedela as a result of the identified sources. Dispersion modelling is a useful tool that provides a link between the environment and emissions to the atmosphere, hence giving an almost true depiction of reality. Based on simulated dispersion modelled results, the main findings indicate that:

- I. The concentrations of PM are highest around the centre of the domain as sources are more clustered in the centre of the townships.
- II. Given that the assumed sources are mostly low-level sources, such concentrations can pose health risks.
- III. Predicted PM₁₀ and PM_{2.5} concentrations over Zamdela exceed the NAAQS over the entire township for both averaging periods.
- IV. Maximums of 1262 µg.m⁻³ daily and 115 µg.m⁻³ annually for PM_{2.5}, and 3496 µg.m⁻³ 24-hour and 334 µg.m⁻³ annual PM₁₀ averages were predicted over Zamdela.
- V. PM_{2.5} and PM₁₀ were predicted to have maximums of 96.69 µg.m⁻³ daily and 13.2 µg.m⁻³ annually, and 653 µg.m⁻³ 24-hour and 343.7 µg.m⁻³ 1-year averages respectively in Wedela.

7.5 Comparison of modelled to monitored results

The modelling scenarios were compared to the measured PM data from the monitoring stations within Zamdela and Wedela. This was done to validate the simulated results, gauging with the acceptable US EPA modelled to monitored ratios and the temporal variations. The main findings are, therefore:

- I. Simulations provided PM concentrations in good agreement with monitored concentrations, thus demonstrating that the regulatory dispersion model can be a useful tool for intra-urban assessments.
- II. The good agreement with the modelled results indicates that the modelled source groups are the main contributors to air quality within the townships masking the contribution from other “background” sources
- III. The ratios between the modelled and monitored data fell within the US EPA recommendation ranging between 0.58 to 1.83.
- IV. AERMOD allows for time-variable emission factors, hence after simulations, the researcher could compare the temporal variations between monitored and modelled data. The data sets compared well confirming that the sources assumed as potential are the major contributors to ambient PM.
- V. The effect of meteorological conditions is evident as the modelled concentrations field mimic the study areas’ prevailing winds.

With all considered, the AERMOD modelling system can be an imperative regulatory tool for assessing the spatial distribution of priority pollutants on an intra-urban scale. Nevertheless, the capacity to accurately simulate source contributions to ambient pollutant concentrations is heavily dependent upon the model input data, particularly, the emission rates of fugitive sources that are complex to estimate.

7.6 Assumptions

The following assumptions were considered in this research:

- I. Ambient monitoring data was mainly limited to fine and coarse particulate pollutants given their health impacts with 3 years’ worth of data from monitoring stations used within Zamdela and Wedela.
- II. There was no available rainfall data for Wedela, they for representative data from Potchefstroom station. This can be a limitation as meteorology tend to vary significantly over space due to topography and altitude among other factors
- III. The quantification of emission sources was restricted to sources within the townships. Windblown dust from mine tailings in the proximity of Wedela was also quantified as it was assumed to be a major contributor to PM.
- IV. High spatial variability contributes to uncertainties and challenges when quantifying and modelling low-income settlements as most characteristics significantly differ, for example, domestic fuel use, vehicle flow, land use, service delivery and income level which to a greater extend influence behaviour. Therefore, this study aimed to account for these

characteristics by integrating several sources such as surveys, literature, satellite evidence among others.

- V. Given the above, assessing the applicability of AERMOD in predicting intra-urban air quality, would have been more effective if there were multiple monitoring stations to compare with simulated concentrations. This study was limited to one monitoring station per township, however, due to available statistical protocols, it could be concluded that the models perform relatively well.
- VI. A point of concern was that the concentrations needed to be measured within the same location as the source, for example in the case of domestic solid fuel burning polluting, it would be interesting to simulate the direct environment of the residents. However, due to computing time, the receptor resolution was limited to the recommended 50 meters.

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APPENDIX

Log sheet used to record traffic flow during the car count surveys

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