AMBIENT AIR QUALITY MONITORING: A COMPARISON BETWEEN TWO URBAN PARKS IN SOWETO, SOUTH AFRICA

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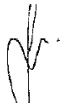
Declaration

I declare that this research is my own unaided work. This dissertation is submitted in fulfillment of the requirements for the degree of Master of Science in the School of Geography, Archaeology and Environmental Studies at the University of the Witwatersrand. I declare that the work presented in this report has not been submitted previously for any degree or examination at any other University.



(Sophia Valsamakis)

Signed on this __14____ day of ____August____2015

Abstract

Soweto is identified as an air pollution hot spot area which is characteristic of poor air quality where ambient air pollutant concentrations frequently exceed the South African Ambient Air Quality Standards. Urban greening programmes are seen as a way for cities to work towards reducing air pollution, offsetting greenhouse gas emissions and improve ambient air quality. The City of Johannesburg embarked on the Greening Soweto project in 2006 where many degraded open spaces were transformed into urban green parks and 6000 trees were planted. The urban parks and trees are believed to serve several environmental benefits; one of which includes the improvement in local ambient air quality. The aim of this research was to assess and compare the local ambient air quality situation at two different urban park types in close proximity, Thokoza Park (older trees) and Petrus Molefe Eco-Park (young trees), in Soweto and establish whether the air pollutants measured at the urban parks were lower compared to the urban background conditions. Furthermore, this study assessed whether the ambient concentrations of the selected criteria air pollutants were within the South African National Ambient Air Quality Standards.

Three ambient air quality monitoring campaigns were conducted during the spring (October) and winter (June and July) seasons of 2013 and 2014 with the use of a mobile air quality monitoring station. The findings of this research suggest that urban trees in Thokoza Park and Petrus Molefe Eco-Park has the greatest potential to improve air quality in Soweto mainly through changes in local meteorological conditions, specifically for temperature and wind fields, rather than direct removal of air pollutants. Differences in the concentrations of the air pollutants at the different sites showed a strong relationship with changes in temperature, wind speed and direction and emission source types. A significant difference in air pollutant concentrations between the two urban park types was only found for particulate matter (PM_{10} and $PM_{2.5}$) and CO_2. In general, lower air pollutant concentrations were recorded at the urban parks compared to the urban background site, particularly during the spring season. This study also suggests that the urban trees could represent a potential O_3 sink during the spring and winter seasons and for NO_x during the spring season. Exceedances of the South African Ambient Air Quality Standards at the two urban parks were only observed for PM_{10} and $PM_{2.5}$ during the winter season of 2014. PM_{10} and $PM_{2.5}$ and NO_x were identified to be air pollutants of concern at the urban parks in comparison to other criteria air pollutants assessed in this study. Maximum daily concentrations of 255 $\mu g/m^3$ for PM_{10} and 126 $\mu g/m^3$ for $PM_{2.5}$ and a maximum hourly concentration of 92 ppb for NO_x were recorded at the parks during the winter season.

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Table of Contents

List of Figures

List of Tables

List of Abbreviations

APPA	Air Pollution Prevention Act
AQMP	Air Quality Management Plan
BVOCs	Biogenic Volatile Organic Compounds
CDMPs	Clean Development Mechanism Programs
CO	Carbon Monoxide
CO_2	Carbon Dioxide
COJ	City of Johannesburg
GHG	Greenhouse Gas
HP	High Pressure
IPCC	Intergovernmental Panel on Climate Change
NEMAQA	National Environmental Management Air Quality Act
NO	Nitrogen Oxide
NO_2	Nitrogen Dioxide
NO_x	Oxides of Nitrogen
O_3	Ozone
PM_{10}	Particulate Matter aerodynamic diameter equal to or less than 10μm
$PM_{2.5}$	Particulate Matter aerodynamic diameter equal to or less than 2.5μm
Ppb	Parts per billion
Ppm	Parts per million
SO_2	Sulphur Dioxide
UNFCCC	United Nations Framework Convention on Climate Change
VOCs	Volatile Organic Compounds

1. Chapter One: Overview

Key air pollutants considered in this study and factors influencing their ambient concentrations in the City of Johannesburg are discussed in this chapter. The interaction between urban trees and air pollutants and their potential for mitigating air pollution are discussed. The basis for this research as well as the aims and objectives of this study are also outlined in this chapter.

1.1. Introduction

Air pollution in cities has a negative impact on both the environment and human health. Repeated human exposure to air pollutants over a long period of time can cause several respiratory, cardiovascular, reproductive and gastrointestinal health problems (Last *et al.,* 1994; Elsom, 1996; WHO, 2000; Brauer *et al.,* 2002; Wright *et al.,* 2014). Similarly, as seen in vegetation, continuous exposure to high concentrations of air pollutants can cause various plant deficiencies such as photosynthesis inhibition, genetic mutations, protein synthesis inhibition and chlorosis (Bamniya *et al.*, 2011). Poor air quality in cities is a result of several factors that collectively promote the presence of air pollutants in the atmosphere and this is generally driven by local meteorology, political and socioeconomic structures and their subsequent influence on human activities that are associated with the release of pollutants into the atmosphere (Held *et al.*, 1996; Wu *et al.*, 2011).

Key anthropogenic drivers of air pollution are as a consequence of rapid urbanisation, development and population growth, increases in economic activity and demand, agriculture, transport and industrial activities and expansion. These have an important role in the distribution and diffusion of air pollutants in a city (Clarke, 2002; Ryu *et al*., 2012). Areas susceptible to poor air quality are subject to the right combination of meteorological conditions, terrain characteristics and emission sources that collectively promote air pollution.

In the larger context air pollution has an imperative role in global climate change. Since the industrial revolution in the 19th century greenhouse gases (GHG) have increased and enhanced the greenhouse effect. Changes in trends of global air temperatures and sea surface temperatures have been observed and modelled. The impact of changing climate conditions on the environment was noted in the IPCC fifth assessment report, Working Group I (IPCC, 2013). South Africa, having been a signatory to the Kyoto Protocol, was obligated to invest in long term solutions that worked towards reducing GHG emissions and establish acceptable levels of GHG in the atmosphere (Scholes and Van de Merwe, 1996).

To this end urban greening programmes have since been seen as a cost-effective strategy that can be implemented to offset GHG emissions and air pollution (Jim and Chen, 2009). Urban forests act as an air pollution sink. Trees remove pollutants by intercepting particles and absorbing certain gases through stomata apertures (small openings on the leaf surface) and lenticels (porous membrane found on roots or bark on plant components) (Nowak, 2002; Nowak *et al*., 2006). Urban forests are able to store carbon for long periods of

time as well as other pollutants. Urban forests in this respect can be an advantage in cities where air pollution is apparent.

South Africa, in preparation for the 2010 Soccer World Cup, had launched the Greening Johannesburg Project, which initiated the Greening Soweto Project in 2006. In this year 6,000 trees were planted and by 2010 a total of 200,000 trees were planted across Johannesburg (www.jhbcityparks.com, 2012). The Greening Soweto project marked a significant change, transforming the Dustbowl perception of the area into an aesthetically pleasing green area. The project was designed to rehabilitate the Klipriver/Klipspruit area and plant thousands of trees along streets, homesteads and open spaces. Many of the open degraded areas were rehabilitated and transformed into vibrant urban parks that offered recreational facilities to the residents. The aim of the project was to provide citizens with a space to be proud of while working towards mitigating climate change and tackling environmental issues such as air pollution (www.jhbcityparks.com, 2012).

According to the South African Department of Environmental Affairs, Soweto is considered an air pollution hot spot area (South Africa, 2008). An air pollution hot spot area is considered to have a degraded air shed which is characteristic of poor air quality where ambient air pollutant concentrations frequently exceed the South African air quality standards. Since the Greening Soweto project was launched in 2006 many degraded open spaces have been transformed into urban green, vibrant parks and thousands of trees have been planted. The urban parks are believed to serve several environmental benefits; one of which includes the improvement in local ambient air quality. However, there has been no air quality monitoring conducted on a small scale in the urban parks of Soweto to investigate if urban parks are associated with lower air quality pollutants compared to the surrounding urban environment through these initiatives. Much of the research is based on international modelling studies with little justification of the modelling simulation results. It is in this respect that this study was undertaken.

1.1.1. Aims and Objectives

The objective of this research was to assess the local ambient air quality situation at two different urban park types, Thokoza Park (older trees) and Petrus Molefe Eco-Park (young trees) in Soweto and establish whether the air pollutants measured at the urban parks were lower compared to the urban background parameters. This research also intended to examine the differences in air pollutant concentrations between two urban park types by comparing older treed and younger treed sites in close proximity. Furthermore, this study wished to assess whether ambient concentrations of the selected criteria air pollutants (CO, NO_2, O_3, SO_2, PM_{10} and $PM_{2.5}$) were within the South African National Air Quality Standards for the period of monitoring.

In order to conduct this research the following research questions were proposed:

1.1.2. Research Questions

I. What are the ambient concentrations of O_3, CO, CO_2, NO_x, SO_2, PM_{10} and $PM_{2.5}$ measured at Thokoza Park (older trees) and Petrus Molefe Eco-Park (younger trees) during the spring and winter seasons of 2013 and 2014.

II. What are the observed differences in ambient concentrations of O_3, CO, CO_2, NO_x, SO_2, PM_{10} and $PM_{2.5}$ at Thokoza Park and Petrus Molefe Eco-Park compared to urban background concentrations?

III. How do the observed local meteorological conditions differ and compare with measured ambient air pollutant concentrations at Thokoza Park and Petrus Molefe Eco-Park?

IV. How do the ambient concentrations of the chosen criteria air pollutants (CO, NO_2, O_3, SO_2, PM_{10} and $PM_{2.5}$) compare with South African National Air Quality Standards?

The work presented herein is divided into five chapters. **Chapter 1** provides an introduction into this study and a literature review of the key air pollutants of concern in the City of Johannesburg, air quality management in South Africa, meteorological influences on air pollution over the City of Johannesburg and the interaction between urban trees and air pollutants and their role in mitigating air pollution in a City. The aims and objectives of the study are also provided. **Chapter 2** gives the methodological approach adopted in this study, including the choice of sampling sites and methods for data collection and analysis. The ambient air pollutant concentrations and meteorological conditions recorded at the two urban parks are presented in **Chapter 3** and compared against the urban background air quality conditions. **Chapter 4** provides a detailed discussion of the findings of this study with the main conclusions and summary of findings presented in **Chapter 5**.

1.2. Literature Review

1.2.1. Sources and Indicators of poor air quality in the City of Johannesburg

Anthropogenic activities in the City of Johannesburg (COJ) are key contributors of air pollutants, for example, sulfur dioxide (SO_2), particulate matter (PM), carbon monoxide (CO), oxides of nitrogen (NO_x), tropospheric ozone (O_3) and carbon dioxide (CO_2). Poor air quality is not evenly distributed over the COJ but is rather localised in several areas identified as hotspot areas (Venter *et al.*, 2012; Beukes, *et al.,* 2013).

Positioned within the COJ municipality is Soweto, which, according to the South African Department of Environmental Affairs is considered an air pollution hotspot area which falls within the Vaal Triangle Air-Shed Priority Area (South Africa, 2008). This suggests that Soweto often experiences poor air quality where ambient air pollutant concentrations frequently exceed the South African National Ambient Air Quality Standards which may be detrimental to human health. Activities within the energy, industrial and transport sectors as well as domestic biomass burning are significant contributors to air pollutants within this region (Table 1-1) (South Africa, 2003; City of Johannesburg, 2009).

Table 1-1: Sources of Air Pollutants in the City of Johannesburg (South Africa, 2003: 5-8).

AIR POLLUTANT	ANTHROPOGENIC SOURCE
PM_{10} and $PM_{2.5}$	• Domestic biomass/fuel burning • Industrial – fuel combustion (e.g. boilers) • Industrial – sand handling processes (e.g. sand reclamation) • Vehicle exhaust emissions • Mining, construction and agricultural activities (e.g. tailings) • Waste incineration and waste burning • Vehicle dust emissions from unpaved roads • Windblown dust from exposed soil and surfaces • Long range transport of particulate matter ($PM_{2.5}$)
NO_x and O_3	• Vehicle exhaust emissions • Domestic biomass/fuel burning • Industrial fuel burning operations • Power generation (long range transport)
SO_2 and CO	• Domestic biomass/fuel burning • Industrial fuel burning operations • Petrochemical and chemical plants (long range transport)
CO_2 (not criteria air pollutant)	• Power generation (primary source) • Industrial fuel burning industries • Domestic biomass/ fuel burning • Vehicle exhaust emissions

Air pollutants by definition are identified substances that exist in the atmosphere in concentrations which negatively impact on human health and environmental quality. Components of poor air quality are considered in two categories, primary and secondary pollutants. Primary pollutants are emitted directly into the atmosphere, mostly from combustion sources such as vehicle exhaust emissions, industrial stacks and mining activities (Harrison *et al.,* 2014). Secondary pollutants are formed in the atmosphere as a result of complex atmospheric chemical reactions amongst primary precursors, for example the photo-oxidation of NO_x to form O_3 (Saini *et al.,* 2008; Harrison *et al*., 2014).

1.2.2. Primary and Secondary Pollutants

Sulphur Dioxide (SO_2)

SO_2 is a primary pollutant which is commonly emitted from industrial activities. Combustion of sulphur containing fuels, such as coal and crude oils, is a key anthropogenic source of SO_2 emissions into the atmosphere (Venter *et al.,* 2012; Harrison *et al.,* 2014). Industrial sources of SO_2 emissions within the COJ originate mostly from small industries which burn sulphur containing fuels as an energy source, particularly with boilers. Furthermore the production, utilisation and processing of sulphur containing materials such as smelting of sulphide ores and production of sulphuric acid are associated with the release of SO_2 emissions (City of Johannesburg, 2008; Venter *et al.,* 2012; Beukes *et al.,* 2013; Harrison *et al.,* 2014).

In the COJ SO_2 emissions are also sourced from domestic biomass burning activities (Naidoo *et al.,* 2014). Domestic biomass burning occurs when households make use of alternative fuels such as coal, wood or paraffin for cooking and/or heating purposes as the accessibility of electricity is either limited or too expensive for frequent use (Kornelius *et al.,* 2012). In the COJ area there are a number of informal settlements such as, Soweto, Orange Farm, Alexandria and Diepsloot, which make use of alternative fuels for cooking and heating. Coal, for example, is cheap and easily accessible in the COJ and thus is frequently used amongst people as an alternative fuel source. Other forms of alternative fuels that are used in the COJ include paraffin and wood (Mrubata *et al.,* 2008; City of Johannesburg, 2008; Kornelius *et al.,* 2012; Naidoo *et al.,* 2014).

SO_2 is acidic in nature and is easily absorbed in the mucous membranes of the nose and upper respiratory tract where it reacts with moisture to form an acid (Hill, 2010). Short to long term exposure to high concentrations of SO_2 may cause changes in respiratory functioning whereby exposed individuals may suffer from irritation to the ears, nose and throat and experience symptoms associated with bronchoconstriction and dyspnoea including chronic coughing and wheezing (Katsouyanni *et al.,* 1997; Kampa and Castanas, 2008). Long term exposure to SO_2 could also result in cardiac problems (WHO, 2000; Hill, 2010).

One of the important reactions in atmospheric chemistry is gas to particle conversion of SO_2 to sulphate. Anthropogenic emissions of SO_2 are key contributors of sulphate particles (Eastwood, 2008; Kagawa and Ishizaka, 2014). A large portion of sulphate particles in the atmosphere is formed by the oxidation of SO_2 with the hydroxyl radical to form hydrogen sulphite (HSO_3). HSO_3 then reacts with oxygen to form sulphur trioxide (SO_3) and the hyperoxyl radical (HO_2). SO_3 then reacts with water vapour to produce sulphuric acid (H_2SO_4) which then forms sulphate particles (Khoder, 2002; Eastwood, 2008; Squizzato *et al.,* 2013). H_2SO_4 will then undergo nucleation and condensation processes where H_2SO_4 will condensate onto pre-existing particles or into new particles. H_2SO_4 also reacts with gaseous ammonium (NH_3) to form neutralised ammonium sulphate particles. Sulphate particles are small in size and generally have an aerodynamic diameter equal to or less than 1μm. They will grow in size as relative humidity increases as they are able to absorb water (Khoder, 2002).

The oxidation of SO_2 to sulphate depends largely on the availability of the hydroxyl radical which is a product of photolysis of O_3 in the presence of sunlight. Thus the concentration of hydroxyl radicals, relative humidity, temperature and solar radiation are key factors influencing the rate of oxidation of SO_2 in the atmosphere (Khoder, 2002; Eastwood, 2008; Kagawa and Ishizaka, 2014). Higher rates of oxidation occur during the day and warmer periods when temperature, relative humidity and solar radiation are higher and there is greater availability of OH radicals for reaction (Squizzato *et al.,* 2013).

Particulate Matter (PM_{10} and $PM_{2.5}$)

PM exists in the atmosphere as either solid or liquid particles varying in chemical composition and size (Beckett *et al.,* 1998; Kampa and Castanas, 2008; Harrison *et al*., 2014). Particles can be considered as either a primary or a secondary pollutant. Primary particles are emitted directly into the atmosphere from the source without any variation in the chemical composition. Secondary particles are formed in the atmosphere through oxidation reactions of gases, such as SO_2 and NO_2, during gas to particle conversion and condensation of vapours (Held *et al.,* 1996; Hueglin *et al.,* 2005; Beckett *et al.,* 1998).

SO_2 and NO_2 have a key role in secondary particle formation. SO_2 and NO_2 undergo oxidation reactions with radicals in the atmosphere to form sulphate and nitrate particles (Squizzato *et al.,* 2013). In a non-marine urban environment, these substances are often associated with direct anthropogenic emissions from industrial and vehicular activities as well as domestic biomass burning (Eastwood, 2008). Oxidation of SO_2 and NO_2 is significantly influenced by temperature and solar radiation, thus higher sulphate and nitrate production rates are recorded during the daytime and spring and summer seasons (Khoder, 2002). Other factors such as relative humidity, concentration of precursor gases, characteristic of pre-existing secondary particles, oxidants present in the air, the concentration of gaseous ammonium species and atmospheric stability are also key factors influencing production rates of sulphate and nitrate particles (Squizzato *et al.,* 2013). Higher production rates of sulphate and nitrate particles often occur when relative humidity, temperature and solar radiation are high, the availability of the OH radical for reaction is high and atmospheric conditions are stable which is associated with an accumulation of precursor gases (i.e. SO_2 and NO_x) (Squizzato *et al.,* 2013). Sulphate and nitrate also react with gaseous ammonium to form ammonium nitrate and sulphate particles (Eastwood, 2008).

The size distribution of particles is characterised by ultrafine, fine or coarse particles, each with an aerodynamic diameter equal to or less than 0.1µm, 2.5µm and 10µm respectively (Beckett *et al.,* 1998; Seinfeld and Pandis, 2006; Harrison *et al.*, 2014). It is important to consider the size fraction of the particle as this will provide as an indication of the source of the particle, the atmospheric lifespan of the particle, the mechanism through which the particle is removed from the atmosphere and most importantly, will determine where the particle will deposit in the lungs of exposed individuals (Beckett *et al.,* 1998; Beckett *et al.,* 2000; Brauer *et al.,* 2002; Kampa and Castanas*,* 2008).

Ultrafine particles ranging in size from 0.01 - 0.1µm are formed in the atmosphere as secondary pollutants during gas to particle conversion. Particles of this size range are effectively removed through coagulation with other particles and thus have a short atmospheric life span. Formation of fine particles (size fraction equal to or less than 2.5µm in diameter) occurs as a result of the coagulation of ultrafine particles and the

condensation of vapours on existing particle surfaces (Beckett *et al.,* 1998). Fine particles tend to grow in size and accumulate in the atmosphere and thus are removed less efficiently accounting for a longer atmospheric life span.

Fine particles can also be directly emitted into the atmosphere during combustion processes. Black carbon for instance is emitted during incomplete combustion processes associated mostly with industrial activity, vehicle emissions, domestic biomass burning and power stations (Eastwood, 2008; Lydia, 2010). Black carbon particles are formed through a series of complex reactions with polycyclic aromatic hydrocarbons and radicals, which eventually form molecular structures that can act as nuclei for particle formation (Lydia, 2010).

Coarse particles (size fraction equal to or greater than 2.5µm in diameter) are mostly emitted directly into the atmosphere from sources such as industrial stacks, mine dumps and vehicle entrainment of dust. Coarse particles consisting mainly of earth crust materials and windblown dust are larger in size and mass and thus they are efficiently removed through sedimentation and deposition processes near the source (Seinfeld and Pandis, 2006; Oguntoke *et al.,* 2013; Harrison *et al.*, 2014). Not all coarse particles are emitted directly into the atmosphere. Nitrate particles, for example, can make up a portion of coarse particles and as mentioned before are formed during gas to particle conversion (Lydia, 2010).

Particles with a diameter equal to or less than 10 µm and 2.5 µm, commonly referred to as PM_{10} and $PM_{2.5}$ respectively, are of particular importance as they are able to reach the upper and lower portions of the respiratory tract of exposed individuals and can cause cardiovascular and respiratory illness (Kampa and Castanas, 2008). The chemical composition of particles is complex as they contain a variety of substances which are easily absorbed onto the surface of the particle, some of which are toxic to the human body. Heavy metals, for example, can be transferred into the human system when particles are inhaled and are able to bio-accumulate in body tissues. Heavy metals can exceed a certain threshold in which the substance becomes toxic to the system, causing adverse health effects such as kidney damage and neuropathies (sleep disorder, memory loss, blurred vision, etc.) (Brauer *et al.,* 2002; Kampa *et al.,* 2008; Peel *et al.,* 2013; Shirinde *et al.,* 2014). Particles of this nature may also be an environmental nuisance as they are able to accumulate in the atmosphere and on artificial surfaces resulting in reduced visibility and corrosion of materials (Oguntoke *et al.,* 2013; Wang *et al.,* 2013).

In the COJ, biomass burning (veld fires), vehicle exhaust emissions, industrial activity, domestic biomass burning, crushing and screening of materials, material handling operations, vehicle dust emissions on unpaved roads and wind erosion of exposed areas such as tailings dams, mine dumps and ash heaps are key anthropogenic sources of particulate matter (City of Johannesburg, 2008; Oguntoke *et al.,* 2013).

Carbon Monoxide (CO)

CO is a primary air pollutant which has an atmospheric lifespan of approximately one to three months. In an urban setting, CO is mostly emitted from anthropogenic emissions, of which the incomplete combustion of hydrocarbon fuels and biomass burning is key (Ocak and Turalioglu, 2008; Venter *et al.,* 2012; Beukes *et al.,* 2013). CO emissions within the COJ are associated with industrial and domestic biomass burning and

vehicular activities (City of Johannesburg, 2008). Due to the relatively long atmospheric lifespan of CO, the gas is able to travel long range distances and accumulate in the atmosphere to reach relatively high background concentrations (~ 50 ppbv in the Southern Hemisphere) (Ocak and Turalioglu, 2008).

Exposure to CO can be detrimental to human health in that it binds to haemoglobin to form carboxy-haemoglobin, thus reducing the oxygen carrying capacity of blood. This impairs the ability of organ tissues to extract oxygen from the haemoglobin, negatively affecting organs such as the brain, heart and lungs (Kampa and Castanas*,* 2008; Hill, 2010; Chen *et al.,* 2011). Acute exposure to high concentrations of CO may result in CO poisoning with an onset of symptoms including nausea, vomiting, headaches, shortness of breath, confusion, and can quickly lead to death (WHO, 2000; Hill, 2010). The effects of long term exposure to elevated ambient concentrations of CO are often associated with cardiovascular problems amongst exposed individuals (Hill, 2010).

Nitrogen Oxides (NOx) and Tropospheric Ozone (O_3)

NO_x represents the sum of nitrogen oxide (NO) and NO_2. NO is considered a primary air pollutant as it is directly emitted during combustion processes under high temperature conditions. Power stations, biomass burning, vehicular emissions and agricultural activity are key anthropogenic sources of NO in the COJ area (Peel *et al.,* 2013). NO_2 can be considered a secondary air pollutant and is either created or destroyed during photo-oxidation and chemical reactions of NO, O_3, VOCs and radicals (e.g. peroxy radical RO_2, hydroxyl radical OH and the nitrate radical NO_3) (Zhang *et al.,* 2004). Short to long term exposure to high ambient concentrations of NO_x can cause decreased lung performance resulting in asthma, inflammation of the lungs and sensitivity to respiratory infections (Hill, 2010). Continuous exposure to high concentrations of NO_x may result in symptoms of chronic coughing and wheezing in vulnerable receptors (Last *et al.,* 1994, Hill, 2010).

Tropospheric O_3 is a secondary air pollutant which acts as both a hazardous air pollutant and a GHG. Studies have shown that chronic exposure to O_3 can affect the respiratory system through inflammation of the nasal and air way epithelial cells and may result in reduced lung performance, respiratory bronchiolitis, pulmonary fibrosis and lung tumours (Last *et al.,* 1994; Hill, 2010).

Ground level O_3 is formed in the atmosphere as a result of photo-chemical reactions amongst O_3 precursor gases in the presence of sunlight. VOCs and NO_x, mostly sourced from vehicle exhaust emissions, industrial processes and domestic biomass burning in the COJ, are key O_3 precursor gases, which play an important role in the accumulation and distribution of surface O_3 (Saini *et al.,* 2008).

A key reaction in tropospheric chemistry is the reaction of O_3 with ultraviolet light to form electronically excited atoms. These atoms react with water vapour in the air to form hydroxyl radicals (OH) which then react with trace gases to form peroxy radicals (RO_2) or hydro-peroxy radicals (HO_2) (Ball, n.d.). When considering O_3 and NO_x chemistry, the OH radical plays an important role influencing day and night time ground level O_3 and NO_x concentrations. During the daytime, the radicals react with NO to form NO2, which then reacts with ultraviolet light to form O_3 (Zhang *et al.,* 2004; Ball, n.d.). Higher solar radiation intensity and higher ambient temperatures will increase photochemical reaction rates forming O_3. As such maximum O_3 concentrations are

generally observed midday to early afternoon (12:00 – 14:00) when solar radiation and ambient temperatures are highest (Elminir, 2005; Saini *et al.,* 2008). Efficient oxidation of NO_2 to form O_3 during the day time will consequently result in lower NO_x concentrations during this period. Furthermore, higher temperatures during the daytime period facilitates the vertical dilution of NO_x as the surface boundary layer weakens resulting in lower ground level NO_x concentrations (Zhang *et al.,* 2004).

During the late afternoon and night time periods, the O_3 concentration generally rapidly decreases as the amount of ultraviolet light for reaction becomes limited. During the night time, the titration of O_3 occurs as photo-chemical reactions of O_3 precursor gases cease. NO reacts with O_3 to form NO_2 and NO_2 reacts with O_3 to form the nitrate radical NO_3 (Zhang *et al.,* 2004; Ball, n.d.). The concentration of NO_x will also decrease during the night time as it is scavenged during the titration of O_3 and the availability of the OH radical decreases for reaction with NO to form NO_2. NO_x concentrations generally peak twice in a 24 hour period, once in the early morning before sunrise and once in the afternoon as a result of the development of the surface boundary layer and an increase in vehicular emissions of NO during peak traffic times (Zhang *et al.,* 2004; Jose *et al.,* 2005).

Carbon Dioxide (CO_2)

CO_2 is of importance in that it represents GHG's which play an important role in influencing the thermodynamic state of the atmosphere and contributing to global warming (IPCC, 2013; Tyson and Preston-Whyte, 2013). The atmosphere maintains a thermodynamic steady state through the relationship between radiation cooling and warming. In general GHG's, including CO_2, have the ability to absorb long wave terrestrial radiation in various wavelengths (mostly out of the range of 8 - 13µm) and re-emit it back to the earth's surface, thus having an overall warming effect on the atmosphere (Tyson and Preston-Whyte, 2013). Whereas radiation cooling occurs as terrestrial radiation passes through the atmosphere without being absorbed and re-emitted back to the earth's surface thus having an overall cooling effect on the atmosphere. Increases in GHG concentrations allow for an enhanced absorption of infrared terrestrial radiation, thereby disturbing the stable state in which cooling and heating naturally occurs and is commonly referred to as global warming (Tyson and Preston-Whyte, 2013).

A steady increase in carbon dioxide concentrations and the relative impact of anthropogenic emissions have been noted in the fifth assessment report of the IPCC, working group I, issued in 2013 (IPCC, 2013). Fossil fuel burning, industrial processes, deforestation, mobile activity and waste disposal were some of the key anthropogenic contributors of CO_2 emissions identified in the fifth synthesis report (IPCC, 2013).

In terms of air quality, global warming may potentially exacerbate the adverse effects of air pollutants on human health through changes in meteorological conditions which are favourable to air pollution, thus making CO_2 an important atmospheric gas to consider in air pollution studies (Amato *et al.,* 2010). Air pollution is highly sensitive to changes in meteorological conditions. Climate change has the potential to impact air quality through changes in meteorological conditions which influences natural emissions of air pollutants and their precursors; atmospheric chemistry within the boundary layer, air pollution dispersion patterns and the deposition of air pollutants (Jacob and Winner, 2009; Trail *et al.,* 2014).

Changes in surface temperature and precipitation due to climate change are two key meteorological variables that will influence air quality. Other meteorological parameters that are of importance include cloud cover, solar radiation, relative humidity, wind patterns and mixing height (Jacob and Winner, 2009; Athanassiadou *et al.,* 2010). Athanassidou *et al.* (2010) and Trail *et al.* (2014) for instance show that a combination of higher temperatures, less cloud cover, greater relative humidity and higher emissions of biogenic volatile organic compounds such as isoprene are projected due to climate change and can result in an increase in photochemical oxidation rates forming secondary air pollutants such as O_3. Higher rates of photochemical oxidation reactions can result in a greater number of exceedance days of the air quality standards for O_3.

The IPCC also reports shifts and changes in global rainfall patterns (IPCC, 2013). A greater number of exceedance days of the PM_{10} and $PM_{2.5}$ standards could occur due to reduced wet depositional processes. Particulate matter is effectively removed from the air during rainfall events and a lower frequency of rainfall events due to climate change will result in higher particulate matter concentrations in several regions such as Africa. On the other hand, some areas, such as parts of America, could expect lower particulate matter concentrations due to a higher frequency of rainfall events (Jacob and Winner, 2009; Trail *et al.,* 2014). In this respect it is important to note that the impact of climate change on air quality is very much location dependent and will vary in different areas and may even be positive depending on the projected local air quality situation and climatic conditions.

Different air pollutants will also respond differently to changes in climate and meteorological conditions depending on their source and their behavioural characteristics. For instance, black carbon concentrations could decrease due to stricter emission control regulations. However, in the case of sulphate particles, higher temperatures could result in higher oxidation rates of SO_2 and the production of sulphates. Alternatively, organic carbon and ammonium particulate matter concentrations could decrease under higher temperature conditions due to higher gas phase conversion rates (Trail *et al.,* 2014). In the case of O_3, stricter emission control regulations of precursor gases such as NO_x and SO_2 could result in lower O_3 concentrations. However, higher temperatures can cause an increase in natural emissions of biogenic volatile organic compounds and an increase in photochemical oxidation rates which would increase O_3 concentrations. Furthermore, a reduction in vehicle emissions of NO_x due to stricter control regulations could subsequently result in slightly higher O_3 concentrations due to less availability of NO_x for titration (Jacob and Winner, 2009; Trail *et al.,* 2014).

Ravishankara *et al.,* (2012) highlight the importance of understanding the linkage between climate change and air quality for effective air quality management practices. Current air quality management practices and emission reduction strategies may not be sufficient to ensure a non-degraded airshed in the future. A strategic reduction in anthropogenic emissions and the use of cleaner alternative fuels and technology may be offset by the impacts of climate change on air pollutant concentrations in some regions (Jacob and Winner, 2009; Athanassidou *et al.* 2010*;* Ravishankara *et al.,* 2012; Trail *et al.,* 2014). However, the potential future responses of different air pollutants to changing climatic conditions and the spatial differences in these responses are very complex. Due to these complexities there is a high level of uncertainty in predicting future air quality impacts due to climate change. Uncertainty is based around our little knowledge of future land use

changes, future climate conditions and future emission sources that are location specific. Furthermore, the modelling tools and data that are used as a basis for future projections are sometimes limited and may oversimplify the complexity in atmospheric processes (Athanassidou *et al.* 2010*;* Ravishankara *et al.,* 2012; Trail *et al.,* 2014). This makes it very difficult for policy makers to mitigate and predict the impacts of climate change on air quality. Nevertheless, there is general agreement amongst studies that climate mitigation measures through stricter controls on GHG emitters have the potential to indirectly improve ambient air quality through reduced emissions of other air pollutants from the same sources (Kahn *et al.,* 2012).

1.2.3. Air Quality Management and Air Quality Standards

In the past air pollution in South Africa was controlled by the Air Pollution Prevention Act (No.45 of 1965) (APPA). The APPA regulated the release of harmful gases, mostly from industrial sources. However, there were a number of issues around the APPA at the time.

The APPA did not take into account fugitive emissions nor did it consider the cumulative effect of multiple emission sources on ambient air quality. Secondly, the APPA was not able to implement acceptable standards as the legal framework for this did not exist (Held *et al.*, 1996). In this respect the APPA was considered outdated and did not fulfil the requirements for effective air quality management in South Africa (South Africa, 2003).In contrast to the APPA the National Environmental Management: Air Quality Act (No. 39 of 2004) (NEMAQA), is now implemented in South Africa to manage air pollution which reflects an effect based approach.

Through the implementation of the NEMAQA, National ambient air quality standards have been developed specifically for South Africa (Table 1-2). Source specific air pollution control measures have also been developed for different industrial emitters which has allowed for more effective air quality management in South Africa (South Africa, 2003). The Department of Environmental Affairs and municipalities have key responsibilities in ensuring the development and implementation of air quality management policies, local guidelines and standards of emissions and ambient concentrations, air quality database, licensing provisions, public awareness campaigns and a framework to enforce legal compliance. These responsibilities are necessary in ensuring effective air quality management in South Africa (South Africa, 2003).

Unfortunately South Africa is limited by human, financial and technological resources. Thus, a priority area approach was adopted to focus available resources in areas that are considered to have very poor air quality and are in need of immediate attention. In keeping with the requirements of NEMAQA, the COJ has developed and is in the process of implementing the Air Quality Management Plan for Johannesburg (AQMP). The main objectives of the AQMP are firstly to achieve acceptable air quality standards within COJ, secondly to minimise the negative impacts of air pollution on human health and well-being, thirdly, to provide citizens with a clean environment and finally to support climate change protection programmes by reducing GHG's (South Africa, 2003).

The NEMAQA has developed ambient air quality standards for eight criteria pollutants namely, SO_2, NO_2, PM_{10} and $PM_{2.5}$, O_3, CO, benzene and lead (Table 1-2). These are substances that have been identified to

negatively affect human health when they exist in the atmosphere above a certain threshold. The ambient air quality standard provides the acceptable maximum concentration that a criteria air pollutant can exist in the atmosphere without threatening human health.

Table 1-2: South African National Ambient Air Quality Standards (Government Gazette, 2009).

POLLUTANT	AVERAGING PERIOD	CONCENTRATION (µg/m^3)	CONCENTRATION (ppb)	FREQUENCY OF EXCEEDANCE (5)
Sulphur dioxide (SO_2)	10 minutes	500	191	526
	1 hour	350	134	88
	24 hours	125	48	4
	1 year	50	19	0
Nitrogen dioxide (NO_2)	1 hour	200	106	88
	1 year	40	21	0
Particulate Matter (PM_{10})	24 hours	120 75(1)	-	4
	1 year	50 40(1)	-	0
Particulate Matter ($PM_{2.5}$)	24-hour average	65(2) 40(3) 25(4)	-	0
	Annual average	25(2) 20(3) 15(4)	-	0
Ozone (O_3)	8 hours	120	61	11
Carbon monoxide (CO)	1 hour	30 000	26 000	88
	8 hours (calculated on 1 hourly averages)	10 000	8 700	11
Lead (Pb)	1 year	0.5	-	0

Notes:
(1) Compliance required by 1 January 2015
(2) Immediate compliance required by the 31 December 2015
(3) Compliance required by 1 January 2016 - 31 December 2029
(4) Compliance required by 1 January 2030
(5) Frequency of exceedance refers to the number of times the pollutant is allowed to exceed the relevant standard in a year

Ambient air quality monitoring forms an important part of the AQMP and can allow for a continuous assessment of ambient concentrations of criteria air pollutants against South African National Ambient Air Quality Standards. A well-developed air quality monitoring network helps the COJ work towards achieving the AQMP objectives by allowing the government to identify temporal and spatial trends in air pollution over the COJ which is essential for effective air quality management (South Africa, 2003).

1.2.4. Meteorological influences on air pollution over the Highveld Region

Ambient concentrations of air pollutants are influenced by meteorological conditions and atmospheric processes. In general, air pollutants tend to accumulate near ground level over the COJ when local weather conditions are stable (i.e. clear, dry, calm winds) and strong surface inversions are prevalent. Air pollutants tend to disperse over the COJ when unstable (i.e. cloudy, moist, strong winds) weather conditions prevail which are associated with weaker surface inversions (Held *et al.,* 1996; Ryu *et al.,* 2012). In this respect it is important to consider the South African climate and local weather conditions which influence the dispersion and dilution of air pollution and subsequently the influence on air pollutant concentrations over the COJ.

The COJ is located in the Highveld region, which is within the subtropical high pressure belt where three high pressure systems, namely the South Atlantic High Pressure (HP) cell, the South Indian HP cell and the Continental HP cell persist. The high pressure systems are associated with strong subsidence of air in the troposphere and divergence of the air towards the earth's surface. Temporal variations in the intensity and positioning of the HP cells characterise the climate and weather over the COJ (Held *et al.,* 1996; Ryu *et al.,* 2012; Tyson and Preston-Whyte, 2013).

During the summer season (December - February), the HP cells weaken and move towards the south, thus allowing for easterly waves and low pressure systems to occur over the interior plateau in which the COJ is positioned. Unstable atmospheric conditions persist as a result of the weakening of the HP cells accounting for stronger wind fields, precipitation and greater vertical and horizontal movement of the air mass over the COJ (Held *et al.,* 1996; Ryu *et al.,* 2012; Tyson and Preston-Whyte, 2013). Relatively better air quality is evident over the COJ during the summer season as there is greater potential for air pollutants to disperse and dilute due to unstable atmospheric conditions, increased precipitation events and an increase in the height of the planetary boundary layer (Held *et al.,* 1996; Ryu *et al.,* 2012).

During the winter season (June - August), the HP cells intensify and move towards the north allowing for westerly waves associated with cold front cyclones and ridging anti-cyclones to occur (Tyson and Preston-Whyte, 2013). The intensification of the HP cells during the winter season results in stable atmospheric conditions associated with strong subsidence inversions, calmer winds, little to no rainfall and reduced, vertical and horizontal movement of the air mass over the COJ (Zunckel *et al.,* 1996; Ryu *et al.,* 2012; Tyson and Preston-Whyte, 2013).

Subsidence inversion layers occur as the air descends towards the surface and warms adiabatically creating stable layers. Stable layers exist in the atmosphere over South Africa at various depths of 800 hPa (seasonal), 700 hPa, 550 hPa (most persistent) and 350 hPa. Subsidence inversions layers limit the vertical dispersion of air pollution by reducing the mixing depth and thus reducing the dilution of the plume allowing for favourable conditions under which pollutants may accumulate (Held *et al.*, 1996; Tyson and Preston–Whyte, 2013). In comparison to the summer season, the occurrence of air pollution episodes over the Highveld region are more intense and frequent during the winter season due to the intensification of the HP cell, colder temperatures and less precipitation (Zunckel *et al.,* 1996; Tyson and Preston-Whyte, 2013).

Air pollution is transported both horizontally and vertically, thus the extent to which air pollutants will dilute and disperse in the atmosphere is significantly dependent on meteorological processes. The vertical dispersion of air pollutants is a function of atmospheric stability and the depth of the mixing layer which are both governed by temperature, moisture and the development of surface inversion layers. Horizontal dispersion of air pollutants is defined by prevailing wind fields which determine the direction and distance of downward transport and the dilution potential of the plume (Held *et al.,* 1996; Zunckel *et al.,* 1996; Elminir, 2005).

Therefore ambient concentrations of air pollutants will continuously fluctuate due to variations in meteorological conditions. Stable atmospheric conditions associated with calm winds and less turbulence will reduce the degree to which pollutants are able to diffuse and dilute giving rise to higher ambient concentrations of pollutants. Conversely, unstable atmospheric conditions associated with stronger wind fields and turbulence in the boundary layer will allow for greater diffusion and dilution potential of air pollutants, thus accounting for lower ambient concentrations (Elminir, 2005; Tyson and Preston-Whyte, 2013). Seasonal variations in precipitation will also influence ambient concentrations through atmospheric wet depositional processes.

1.2.5. Atmospheric Deposition Processes

Air pollutants are removed from the atmosphere through depositional processes. Atmospheric deposition is considered in two components, namely, wet deposition and dry deposition. Wet deposition is defined as the process where gaseous and particulate air pollutants are removed from the atmosphere via precipitation. There are three mechanisms in which air pollutants are removed through wet depositional processes. Firstly, pollutants, especially particulates, can act as condensation nuclei contributing to cloud formation. The gases and particulates are then incorporated into the water droplet and thus are removed during rainfall. Secondly, particulates may collide with water droplets during rainfall without contributing to cloud formation. Thirdly, gaseous pollutants can dissolve in water droplets during cloud formation and rainfall (Lovett, 1994; Beckett *et al.,* 2000).

Ambient concentrations of air pollutants will decrease with increases in the intensity and frequency of rainfall (Lovett, 1994; Beckett *et al.,* 2000). Therefore, over the COJ concentrations of air pollutants will decline during the summer months (December - February) when the COJ receives most of its rainfall (Elminir, 2005). Although wet removal processes of air pollutants may be beneficial during the summer season in terms of air quality, it can also negatively impact on the environment. Gases such as SO_2 and NO_x easily dissolve and mix in water droplets, thus increasing the acidity of the water droplet. Precipitation that contains a high level of contaminants and acidity is referred to as acid rain and causes significant corrosion of materials and negatively affects the quality of plants and fresh water systems (Lovett, 1994; Tyson and Preston-Whyte, 2013).

Dry deposition occurs when gaseous and particulate pollutants are removed from the atmosphere in the absence of precipitation and transferred to soil, vegetation and surface water bodies. Vegetation and surface water bodies can act as either a sink or source of gases through gaseous absorption and volatilisation

processes (www.environment.ucla.edu, 2006; www.epa.gov, 2014). The transfer of gases between air, water and vegetation will depend on interactions amongst several factors. These include the difference in concentration of the substance in air versus the median, meteorological conditions (wind speed, temperature, humidity, etc.), the physical and chemical properties of the substance (characteristics of the pollutants) and the physical and chemical properties of the species (trees) in those ecosystems (www.environment.ucla.edu, 2006; Jacob and Winner, 2009; www.epa.gov, 2014).

Deposition of particulates will depend mostly on the size fraction of the particle. Larger particles (particle diameter size ≥ 10 μm) tend to deposit closer to the emission source mainly through gravitational settling and interception and impaction processes. On the other hand finer particles (particle diameter size ≤ 10 μm) are less easily deposited through dry deposition processes and may persist in the atmosphere for longer periods of time (Lovett, 1994; Beckett *et al.,* 2000). Studies have shown that dry deposition of gases and particulates to vegetation can act as a notable sink for air pollutants (Nowak *et al.,* 2000; Yang *et al.,* 2005; Nowak *et al.,* 2006).

1.2.6. Benefits of Urban Trees

The benefit of air pollutant removal processes through urban greening was investigated as part of the development of Clean Development Mechanism Programmes (CDMPs). In 1997 the Conference of Parties to the United Nations Framework Convention on Climate Change (UNFCCC) adopted the Kyoto Protocol on Climate Change to enable signatory countries to reduce GHG emissions in an effort to tackle climate change. The Kyoto Protocol on Climate Change provided ways in which countries could reduce GHG emissions or offset them to reach allowable caps (Bloomfield and Pearson, 2000). As defined in the Kyoto Protocol, CDMPs could be used to encourage countries to invest in GHG reduction programmes. Urban greening programmes were one of the CDMPs that countries could invest in. By investing in urban greening programmes, countries are able to offset their GHG emissions and earn carbon credits (Bloomfield and Pearson, 2000).

Urban greening programmes involve planting trees, developing urban green areas (e.g. parks) or conserving urban green biomass (trees, shrubs, flora, soils and grass lawns) and rehabilitating degraded areas. Several environmental and social benefits of urban greening programmes have been considered (Table 1-3) (Coder, 1996; Escobedo *et al*., 2011). In terms of air quality the long term benefits of investing in urban greening programmes are not limited to offsetting GHG emissions but are also seen as a way that cities could reduce air pollution and improve ambient air quality (Coder, 1996; Nowak *et al*., 2000; Nowak, 2002; Yang *et al*., 2005; Jim and Chen, 2009; Escobedo *et al*., 2011). Urban trees improve air quality, mainly through temperature reduction, microclimate regulation, air pollutant removal processes and influencing the energy use of buildings (Nowak, 2002; Jim and Chen, 2009; Leung *et al*., 2011).

Table 1-3: Environmental and Social Benefits of Trees and Parks within an Urban Environment (Coder, 1996: 1).

ENVIRONMENTAL AND SOCIAL BENEFITS	GOODS AND SERVICES
Reduce temperature and energy use	• Provision of shade • Influence incoming solar radiation/ surface heating processes • Wind control/ microclimate regulation • Evaporation/transpiration processes
Improve air quality	• Pollutants uptake and storage • Carbon sequestration • Oxygen production
Hydrological benefits	• Reduce runoff and sedimentation • Store water • Reduce erosion • Improve water quality • Uptake water pollutants
Economic benefits	• Increase property values • Product production (e.g. medicinal plants)
Social and psychological benefits	• Aesthetically pleasing/ visual screening • Recreation facilities • Improve health (reduce stress) • Reduce noise pollution

1.2.7. Characteristics that Influence Air Pollution Removal by Trees on a Small Scale

On a small scale a direct interaction between the tree components, including leaf surface area, stomata aperture and lenticels, with the ambient air exists. The ability of trees to remove air pollutants on a small scale depends on complex interactions between the tree components (stomata, lentils, leaf properties, etc.), pollutants and exposure conditions. Gases and particles are removed from the atmosphere, mostly via absorption, impaction and interception processes (Beckett *et al.,* 2000). When pollutants are exposed to the tree surface, one of three things could happen. Firstly, particles can intercept and be temporarily stored on the tree surface and then be re-suspended back into the atmosphere. Secondly, particulates can accumulate on the tree and leaf surfaces and then be removed via precipitation or leaf fall. Thirdly, very fine particles (≤ 1 μm in diameter) and gases can be absorbed by stomata during photosynthesis, transpiration and respiration processes where gases are integrated and fixed into the plant tissues, either by diffusing into the intracellular spaces or dissolving into water films (Nowak, 2002; Nowak *et al.,* 2006; Yin *et al.*, 2011). Removal of air pollutants by trees will differ for particles and gaseous species.

Interactions between Trees and Gaseous Pollutants

Absorption of gaseous pollutants by trees largely depends on the efficiency of stomatal uptake. Therefore the plant stomata size and density as well as the size of the leaf surface area are key factors influencing the stomatal uptake of gaseous pollutants. The density and size of stomata influences the rate at which gases are able to diffuse across the leaf surface area. The inability for gases to diffuse across leaf components is expressed as stomata resistance which is often higher amongst trees growing in a polluted city environment (Wuytack, *et al.,* 2010). Trees in a polluted environment are often associated with smaller stomatal pores compared to trees in a non-polluted environment. Smaller stomatal pores in trees growing in a polluted environment, reduce the rate of intake of air pollutants which could be detrimental to plant health. This is a defence mechanism that is observed in some trees that are often exposed to air pollutants. The smaller the size of the stomatal pores the higher the stomata resistance is, therefore reducing the amount of pollutant uptake by the leaf (Wuytack, *et al.,* 2010).

Smaller stomata size is potentially counterbalanced by higher stomata density on a larger leaf surface area where the higher the density of stomata pores the greater the flow of pollutants across the leaf surface boundary layer will be (Wuytack *et al*., 2010). The stomata resistance is therefore influenced by the interaction between stomata size, stomata density and leaf surface area which is determined by the tree species. The interaction between these factors influences the amount of gaseous pollutants absorbed and removed from the air by different tree species (Cavanagh *et al*., 2009; Escobedo and Nowak, 2009).

Interactions between Trees and Particulate Pollutants

Particulates are removed from the atmosphere via a combination of processes known as wet, occult and dry deposition. Wet and occult deposition are the processes whereby particulates are collected and deposited on the ground surface via precipitation such as rainfall, snow, mist and fog. Dry deposition is the process whereby particulates are removed from the atmosphere via gravity, Brownian motion, interception, impaction and sedimentation processes (Beckett *et al.,* 2000). Interception, impaction and sedimentation are the main mechanisms through which particles are removed via trees and occur as the wind moves over and around the tree. The higher the levels of interception, impaction and sedimentation of particles, the more efficient the trees will be at removing particles from the atmosphere (Beckett *et al.,* 2000).

Removal of particulates by trees will depend on the size of the particle, tree species, tree leaf surface properties and meteorological conditions such as wind speed and humidity (Beckett *et al.,* 2000; Litschke and Kuttler, 2008). Depositional processes vary as a function of the size of the particle. Brownian motion accounts for the removal of very fine particles (diameter size < 0.1 µm) where a concentration gradient is established resulting in diffusion of particules across the leaf surface boundary layer. Brownian motion can be accelerated when the surface of the leaf is wet, stomata size is larger and there is a higher stomata density (Beckett *et al.,* 2000; Wuytack *et al*., 2010).

Particles with a diameter size between 0.1 µm and 10 µm are deposited mainly through interception and impaction processes. These processes occur as the wind passes over or around a tree resulting in the interception or impaction (collision) of particles against the leaf or tree surface. Surface properties of the leaf and the structural arrangement of the tree can increase the amount of particulates deposited on the tree surface. A hairy and/or sticky leaf surface area will increase interception and decrease re-suspension of particulates. Similarly, the more complex the structural arrangement of the tree the more efficient the tree is at removing particulates, as interception and impaction processes will increase as the wind curves around a more complex obstacle (Beckett *et al.,* 2000; Litschke and Kuttler, 2008). Gravity is the key process acting on particles larger than PM_{10} therefore as the mass of the particle increases, the greater the deposition rate will be. Thus, larger particles tend to deposit near the source as opposed to smaller particles that tend to deposit further away from the emission source (Litschke and Kuttler, 2008).

Depositional velocities of particles are also dependent on meteorological factors such as humidity and wind speed by influencing the mass of the particles and the ability of the tree to remove particles. The size and mass of particles will increase as a function of relative humidity. Therefore, higher depositional velocities could be expected as relative humidity increases due to the fact that particles have the ability to absorb and release water (Litschke and Kuttler, 2008). Variations in wind speed will mostly influence impaction and interception processes on a small scale. Particules are less likely to intercept or impact leaf components under higher wind speed (WS > 9 $m.s^{-1}$) conditions as the plant tends to streamline and re-suspension of particles is to be expected (Beckett *et al.,* 2000). The combination of the aforementioned factors, stomata density and size, tree structural arrangement, leaf surface properties, particulate size and meteorological conditions, will determine the filtration capacity of the tree on a small scale.

1.2.8. Characteristics that Influence Air Pollution Removal by Trees on Medium Scale

The interaction between the porosity of the tree canopy and the arrangement and positioning of the trees with ambient air will influence the ability for trees to act as a notable sink for air pollutants on a medium scale (Mensink *et al.,* 2012). Trees act as an obstacle as the wind moves over and through the trees subsequently influencing wind velocity and turbulence patterns. Dry depositional processes are enhanced with an increase in the aerodynamic roughness of the tree canopy. However, in urban areas, higher concentrations of air pollutants could also occur as a result of reduced ventilation and turbulence caused by decreased wind velocities associated with trees (Litschke and Kuttler, 2008). The higher the density of the tree canopy the more likely the prevailing wind velocity and turbulence will be reduced, allowing for less ventilation and vertical mixing of pollutants which could cause localised high pollutant concentrations (Litschke and Kuttler, 2008).

Furthermore, the structural arrangement of the trees can influence pollutant concentrations. Air pollutant concentrations may vary on different sides of the tree due to greater wind turbulence on the windward side of the tree. As such, higher concentrations of pollutants may occur on the leeward side of the tree and below the tree canopy due to reduced vertical mixing near the surface associated with decreased wind velocities (Gromke and Ruck, 2010; Buccolieri *et al.,* 2011).

Characteristics of Air Pollution Source

The type and number of emission sources and the distance of the source from the trees will also influence the pollutant concentrations in and around the tree canopy depending on the arrangement and positioning of the trees. Air pollution from sources emitting near ground level (e.g. road vehicle exhaust emissions) and within close proximity to trees, defined by a very tight arrangement and a dense tree canopy structure, are more likely to accumulate between trees due to reduced ventilation. As a result the trees could potentially have an overall negative effect on local air quality causing slightly higher pollutant concentrations (Litschke and Kuttler, 2008; Vos *et al.*, 2013).

On the other hand, emission sources emitting higher above ground level (e.g. industrial stacks) and further away from trees with a very tight arrangement and a dense tree canopy structure are less likely to have a negative effect on the local air quality (in and below the tree canopy) as the trees would act as a buffer inhibiting the air from above the tree canopy penetrating through to the surface (Litschke and Kuttler, 2008; Vos *et al.*, 2013).

The findings of Litschke and Kuttler (2008), Gromke and Ruck (2010) and Vos *et al.* (2013) suggest that the arrangement and positioning of trees is an important concept that needs to be considered when wanting to use trees to mitigate air quality related issues in an urban environment.

1.2.9. Air Pollution Removal by Trees Observed on Large Scale

In the larger context the distribution and extent of urban trees may have an overall positive effect on air quality within a city environment in several ways which may not necessarily be observed on a small to medium scale (Mensink *et al.,* 2012). Urban areas are often characterised by high energy commercial and industrial land use together with a modified artificial land cover and less vegetation which are associated with elevated average temperatures compared to surrounding non-urban areas. This phenomenon is commonly known as the urban heat island effect (Bolund and Hunhammer, 1999; Akbari *et al.*, 2001).

The urban heat island effect can be detrimental to air quality as it is associated with elevated ambient temperatures, which can subsequently cause an increase in temperature dependent atmospheric chemical reaction rates forming secondary pollutants (Akbari *et al.*, 2001). Studies have shown that urban trees have the potential to mitigate these impacts by reducing surface air temperatures through evapotranspiration, absorption of GHG's and the provision of shade (Akbari *et al.*, 2001; Nowak, 2002; Jim and Chen 2007; Jim and Chen, 2009). Evapotranspiration from trees can reduce air temperatures by absorbing latent heat and using the heat to evaporate water into the atmosphere (Jim and Chen 2007; Jim and Chen, 2009). The provision of shade from trees can modify the amount of incoming solar radiation reaching the surface and thus decreasing surface temperatures (Bolund and Hunhammer, 1999; Akbari *et al.*, 2001). In addition trees have the effect of reducing air temperatures over a long time period as they act as a sink for GHG such carbon dioxide, which absorb infrared radiation and have a general warming effect (Rosenfeld *et al.*, 1998).

The reduction in surface temperatures improves air quality as it reduces atmospheric chemical reactions which are temperature dependent (Nowak, 2002). The formation of secondary pollutants such as O_3 for example, is significantly influenced by temperature reductions. When O_3 precursors are exposed to light they undergo photochemical oxidation reactions to form ozone. A reduction in air temperature would reduce reaction rates amongst O_3 precursors in the atmosphere and thus decrease ambient O_3 concentrations (Saini *et al.,* 2008).

Furthermore, trees influence building energy use by reducing the extent in which air conditioners are needed to cool air (Nowak, 2002; Jim and Chen, 2009). A reduction in energy use can indirectly influence the amount of air pollutants emitted during the production of electricity. The amount of pollutants emitted at power stations will reduce as the demand for electricity reduces (Bolund and Hunhammer, 1999; Nowak, 2002). Urban parks also have the ability to initiate local park breezes which counteract the urban heat island effect. Urban parks are associated with cooler, more dense air, due to reduced surface temperatures, thus acting as a low pressure system where warmer less dense air from the surrounding environment subsides. The small scale park breeze can positively influence local air quality through the reduction of surface temperatures and an increase in the dispersion and dilution potential of air pollutants (Bolund and Hunhammer, 1999; Nowak, 2002).

Trees also act as a wind breaker. In the winter season urban parks and street trees could shield residential buildings from cool breezes and thus reduce the frequency in which domestic biomass burning appliances are used for heating purposes. The burning of alternative fuels, for example coal, paraffin or wood, for heating is a significant source air pollutants, including particulate matter, carbon monoxide and sulphur dioxide. Reduced domestic biomass burning for heating during the winter season may have a positive impact on local air quality in developing countries, however further research is needed in this regard (Akbari *et al.*, 2001).

Even though trees may act as a notable sink for air pollutants, thus having a positive impact on air quality, they are also known to have a negative impact on air quality (Leung *et al.,* 2011). Trees are a source of biogenic volatile organic compounds, commonly referred to as BVOCs, which can contribute to the formation of secondary aerosols and O_3. BVOCs are emitted in trace amounts, however, they are estimated to be 2 – 3 times more reactive than anthropogenic emissions of VOCs (Paolletti, 2009; Leung *et al.,* 2011). Studies have shown that BVOCs can contribute significantly to O_3 formation and can account for elevated O_3 levels (Betts *et al.,* 2008; Paolletti, 2009; Alonso *et al.,* 2011). Isoprene and monoterpenes are two examples of BVOCs which often make up a large portion of BVOCs emitted from trees. Other examples of BVOCs, which are emitted in lower amounts, include alkanes, alkenes, alcohols and carbonyls, to name a few (Paolletti, 2009). Trees in an urban environment are often associated with higher emissions of BVOCs compared to trees in non-urban environment. Urban trees tend to release higher amounts of BVOCs when exposed to certain stresses, which are typical of an urban environment, such as water shortages, polluted water, high air pollutant concentrations or high temperatures (Betts *et al.,* 2008; Alonso *et al.,* 2011). The amount and type of BVOCs emitted from trees will vary for different species. Lower BVOC emitting trees in particular lower Isoprene and monoterpene emitters, should be planted in an urban environment where they are potentially vulnerable to environmental stresses (Leung *et al.,* 2011).

2. Chapter Two: Methodology

The methodological approach adopted in this study is detailed in this chapter. A description of the two urban parks and the background site under study, as well as the locality of the sampling sites are provided. Methods for the collection and analysis of the air quality data are given.

2.1. Description of Sampling Sites

Soweto is located south-west of Johannesburg, Gauteng, South Africa (26 16' 00'' S, 27 52' 00'' E) and falls under the responsibility of the City of Johannesburg Metropolitan Municipality. The area is situated on the Highveld interior plateau (elevation ~ 1586m above sea level) forming part of the Highveld grassland which is characterised by a variety of grasses, small shrubs and few trees. The urban climate conditions of Johannesburg are characteristic of fairly cool, dry winter periods with well-established inversion layers and warm wet summer periods (Goldreich, 1992; City of Johannesburg, 2008). The average daily temperature ranges between 16^{o}C and 26^{o}C reaching a minimum around June during winter and a maximum in January during the summer season (City of Johannesburg, 2008). Rainfall is mostly observed during the summer period and averages around 710 - 818 mm per annum depending on the frequency of drought episodes (Goldreich, 1992; City of Johannesburg, 2008).

2.1.1. Thokoza Park and Petrus Molefe Eco-Park (Sampling Sites A and B)

As with the Greening Soweto Project many degraded open spaces have been transformed into green vibrant urban parks. In this study, ambient air quality monitoring took place in two of the rehabilitated urban parks, Thokoza Park and Petrus Molefe Eco-Park respectively (Figure 2-1 and 2-2).

Domestic biomass burning, tailings dams/mine dumps, mining and industrial activity, vehicle entrainment of dust on unpaved roads, vehicle exhaust emissions, small scale agricultural activity and wind borne dust from exposed land are identified as potential sources of air pollutants surrounding (< 10km) Thokoza Park and Petrus Molefe Eco-Park. Domestic biomass burning, vehicle exhaust emissions and vehicle dust emissions on unpaved roads are concentrated in the areas immediately (< 5km) surrounding the two parks which are characterised by residential land use consisting of a mixture of both formal and informal establishments (Figure 2-1). The latter emission sources are concentrated in the outer areas (5 - 10km) surrounding the two parks. A band of tailings dams and mine dumps which stretch from the east to the west are positioned north of Thokoza Park and Petrus Molefe Eco-Park (Figure 2-3).

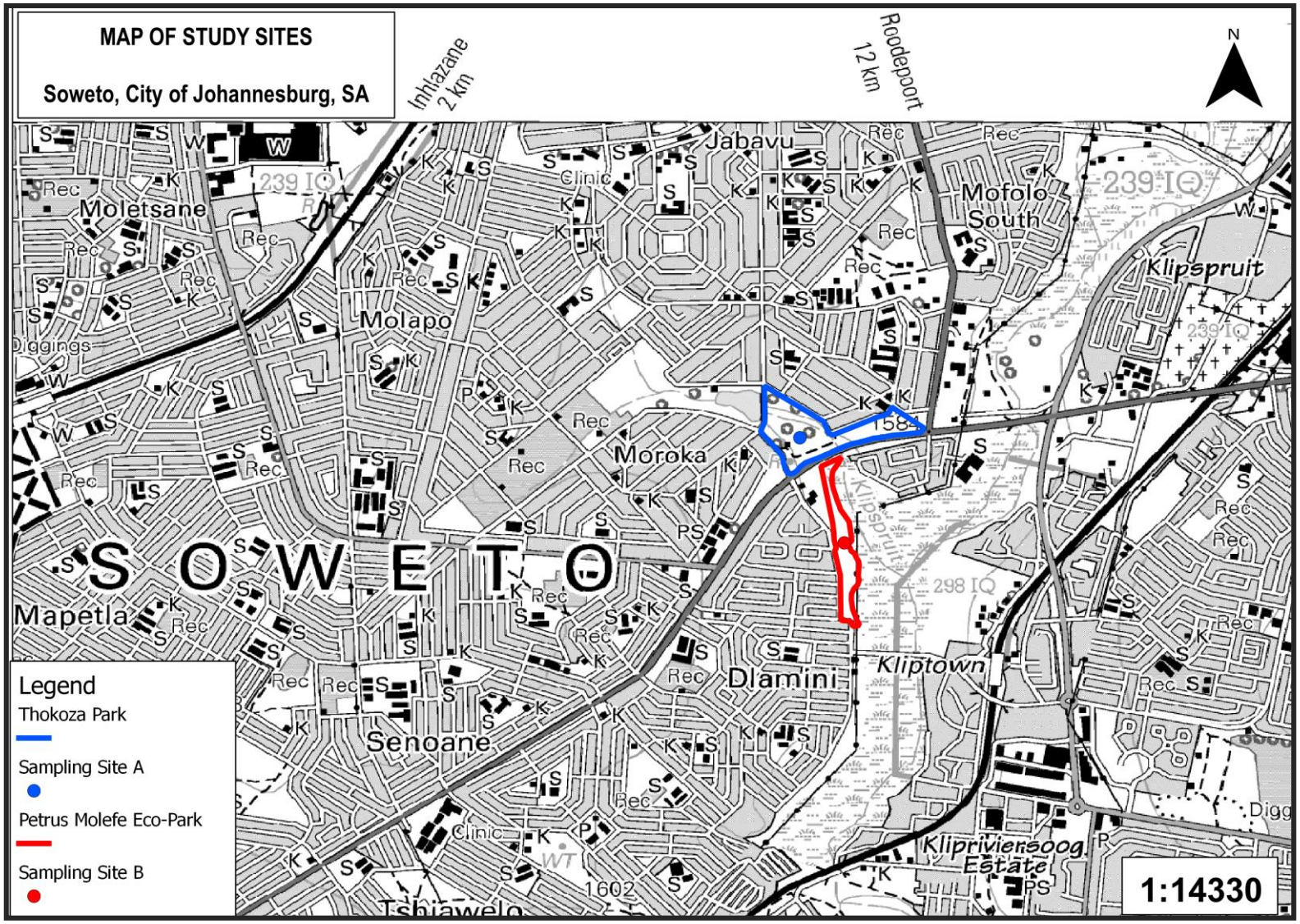


Figure 2-1: Map of study sites, Thokoza Park (26^0 15' 48'' S, 27^0 52' 44'' E) and Petrus Molefe Eco Park (26^0 16' 01'' S, 27^0 52' 50'' E).

Figure 2-2: Aerial representation of study sites, Thokoza Park (blue polygon) and Petrus Molefe Eco Park (red polygon) (Google Earth, 2014).

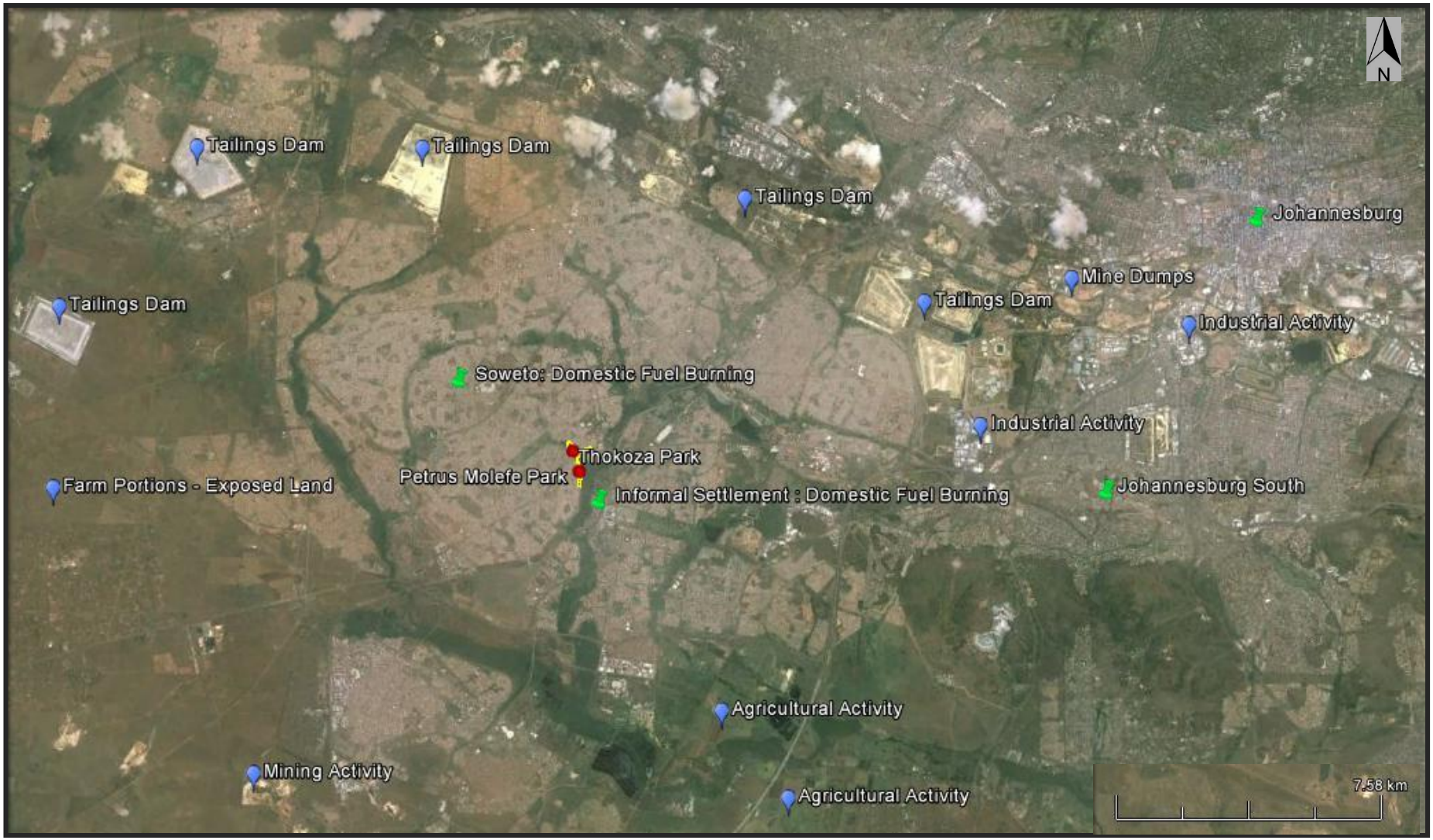


Figure 2-3: Stationary emission sources within a 10km radius from project study sites (Google Earth, 2014).

Thokoza Park (sampling site A) is located in Moroka, Soweto. The park currently consists of the rehabilitated Moroka dam, Klipspruit stream, older trees, recreational facilities, Rea Vaya bus station, rock features, toilets and green lawns (Figure 2-4).

Petrus Molefe Eco-Park (sampling site B) on the other hand is a much smaller park that is located just south of Thokoza Park in Dlamini, Soweto. This area has also been completely rehabilitated in different phases. The park consists of a soccer field, an outdoor green gym, art features, toilets, paved foot paths, jungle gyms, grass lawns, young trees and the background of the Klipspruit wetland/marshland area. Unlike Thokoza Park this park is more characteristic of an open space where several indigenous trees have been planted along Jacaranda Street and are younger compared to the trees in Thokoza Park (Figure 2-5) (www.jhbcityparks.com, 2013).

Figure 2-4: Locality of sampling site A (red pin) within the tree canopy at Thokoza Park (Google Earth, 2015).

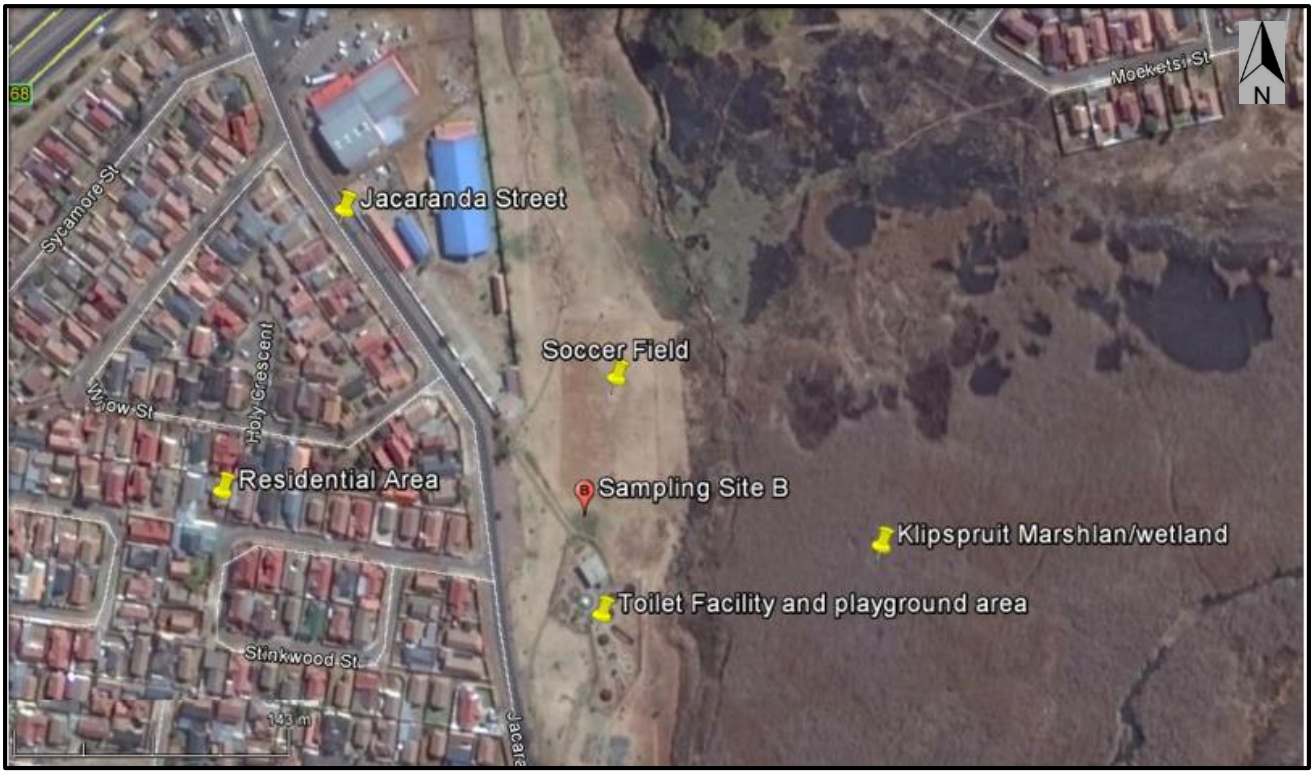


Figure 2-5: Location of sampling site B (red pin) at Petrus Molefe Eco-Park (Google Earth, 2015).

Sampling sites were chosen to best represent a well-developed older tree and an open space younger tree site which have similar environmental conditions and exposures to local and regional air pollution sources. On this basis sampling site A and sampling site B was chosen. Both sampling sites are within close proximity to each other (~500 m). In this respect both sampling sites are exposed to similar sources of air pollution and similar meteorological conditions (i.e. temperature, relative humidity, wind speed, wind direction, rainfall, etc.).

Furthermore both sampling sites are within similar distances from Chris Hanni main road (measured from Thokoza Park Rea Vaya bus station) which acts as a main source of traffic pollution into both parks. The choice of sampling sites was largely restricted by the position of a power outlet and the availability of security. Sampling site A represents a small cluster of tree canopies that extends approximately between 10-12m in radius (Figure 2-4). The tree canopy comprises of a small cluster of exotic Babylon weeping willow trees (*Salix babylonica*), White willow trees (*Salix alba*), and Peruvian pepper trees (*Schinus molle*). Sampling site B represents an open space situated between a soccer field and a toilet facility. The Klipspruit wetland area is positioned adjacent to sampling site B. Sampling was conducted approximately 3-4 m above the ground surface just below the tree canopy.

2.1.2. Diepkloof Air Quality Monitoring Station (Background Station)

Air quality monitoring at the two urban parks could not run concurrently due to the availability of only one set of instruments. A second set of instruments could not be obtained due to the financial costs associated with obtaining such equipment. In order to counter this limitation, air quality monitoring and meteorological data

were also obtained from the South African Weather Services from the Diepkloof air quality monitoring station (26.251° S; 27.956° E). Therefore, allowing for a comparison of air pollutant concentrations and meteorological variables measured at the two urban parks against the urban background conditions. The Diepkloof station is operated by the City of Johannesburg and is located 7.8 km and 7.7 km NE of sampling site A and B. Diepkloof monitoring station is the closest permanent air quality monitoring station to Thokoza Park and Petrus Molefe Eco-Park that was able to provide accurate air quality data that could be used for comparative purposes in this study (Figure 2-6).

Figure 2-6: Location of Diepkloof Station (yellow pin) in relation to sampling sites A and B (Google Earth, 2015).

The Diepkloof monitoring station is representative of urban background conditions as it is subject to similar meteorological conditions, surrounding emission sources and surrounding land use types as Thokoza Park and Petrus Molefe Eco-Park but is not characteristic of an urban park or open space. The Diepkloof monitoring station is located within a residential area and not within an open space or urban park with trees.

Domestic biomass burning, tailings dams/mine dumps, mining and industrial activity, vehicle entrainment of dust on unpaved roads, vehicle exhaust emissions, small scale agricultural activity and wind borne dust from exposed land are identified as potential sources of air pollutants surrounding (< 10km) Diepkloof monitoring station. A few tailings facilities are located directly north of Diepkloof station (< 2 km). Land use to the west and south-west of Diepkloof is characterised by residential land use consisting of a mixture of both formal and informal establishments. Industrial areas are located in close proximity (6 - 10 km) to the east and north-east of Diepkloof station. South and south-west of Diepkloof station consists mostly of residential and small agricultural holdings land use.

2.2. Sample Collection and Data Analysis

2.2.1. Sampling Duration

In this study the ambient concentrations of CO, CO_2, O_3, SO_2, NO_x, PM_{10} and $PM_{2.5}$ as well as meteorological parameters (rainfall, relative humidity, temperature, wind speed and wind direction) were continuously measured at Thokoza Park (sampling site A) and Petrus Molefe Eco-Park (sampling site B) over three monitoring campaigns conducted in 2013 and 2014 using a mobile air quality monitoring station.

The date and duration for each monitoring campaign was dictated by the availability of the air quality instruments and security, thus accounting for inconsistent sampling periods for each monitoring campaign. Collectively, monitoring took place for approximately 10 weeks, allowing for a comparison between the data collected at the two parks, background data as well as the South African National Ambient Air Quality Standards (Table 2-1).

Table 2-1: Sampling Period for Each Ambient Air Quality Monitoring Campaign Conducted at Thokoza Park and Petrus Molefe Eco-Park in 2013 and 2014.

MONITORING CAMPAIGN	SAMPLING PERIOD	EXPOSURE PERIOD	PARK
Campaign 1	Sampling Period 1	3 – 10 June 2013	Petrus Molefe
	Sampling Period 2	10 – 14 June 2013	Thokoza
Campaign 2	Sampling Period 3	7 – 21 October 2013	Petrus Molefe
	Sampling Period 4	21 – 31 October 2013	Thokoza
Campaign 3	Sampling Period 5	27 May 2014 - 13 June 2014	Petrus Molefe
	Sampling Period 6	13 - 23 June 2014	Thokoza

2.2.2. Sampling Instrumentation

Ground level ambient concentrations of the chosen gases were measured at sampling sites A and B with the use of a sheltered mobile air quality monitoring station, which contained air quality monitoring instruments developed by Met One Instruments, Horiba Ltd. and Thermo Environmental Instruments. The mobile monitoring station, including all the instruments was provided by the North West University. Due to limited resources all the instruments were calibrated by the North West University prior to each monitoring campaign.

The monitoring station was assembled at sampling site B first and then relocated to sampling site A for each campaign. To prevent the instruments from overheating the mobile station was sealed off with an air conditioner to maintain a cool temperature of 20 - 25^{o}C. The ambient air samples were drawn in via the inlet heads, which penetrated the roof of the mobile station and were located approximately 3m above ground level. A RM Young weather station, which contained conventional meteorological instruments (wind speed, wind direction, temperature, relative humidity, rainfall and atmospheric pressure) was attached to a 9 m lattice mast which was externally attached to the mobile monitoring station. Weekly visual checks were undertaken during each monitoring campaign to assess whether the instruments were operating efficiently and that a constant supply of electricity was provided to the station. The air conditioner was also checked weekly to see if it was operating properly. All the data were recorded in one minute intervals and were automatically downloaded to the CR 1000 data logger which was retrieved at the end of each monitoring campaign.

Ambient concentrations of PM_{10} and $PM_{2.5}$ were measured in units of micrograms per cubic meter of air ($\mu g/m^3$) with the use of the BAM-1020 Beta Mass Ambient Particulate monitor developed by Met One instruments. The BAM 1020 uses beta attenuation as its operating method to determine the concentration of PM_{10} and $PM_{2.5}$ particles in the sample of air (Tasic *et al.,* 2010). An external pump draws a sample of air in via the inlet head and through the instrument at a rate of 16.7 litres per minute (air flow rate) and then deposits the collected dust onto a filter tape. The filter tape is then exposed to beta rays which are emitted from a carbon 14 element. The extent of beta attenuation provides an indication of the mass of particles deposited on the filter tape and allows for the concentration of PM_{10} and $PM_{2.5}$ in the air sample to be determined (Met One Instruments, 2008; McNamara *et al.,* 2011).

Sampling of NO_x was measured in parts per billion (ppb) with the use of HORIBA APNA-370 ambient NO_x analyser developed by HORIBA Ltd. The general method for measuring NO_x by the HORIBA NO_x analyser is based on the theory of chemiluminescence. Sampling of O_3 was measured in ppb with the use of HORIBA APOA-370 ambient O_3 Monitor developed by HORIBA Ltd. This instrument uses the theory of dispersive ultraviolet absorption for its method of measuring the concentration of O_3. Sampling of CO was also measured in ppb with the use of HORIBA APMA-370 ambient CO analyser developed by HORIBA Ltd. The HORIBA CO analyser uses the theory of non-dispersive infrared analysis as its basis for measuring the concentration of CO in a sample (Lovett, 1994).

The general method adopted by the HORIBA APNA-370 series is similar to measuring each pollutant. The concentration of each gas in the air sample is determined by how much the molecules are able to emit/absorb either light, ultraviolet light or infrared radiation at different wavelengths (HORIBA$_{a, b, c}$, 2004). The HORIBA APMA-370 series design allows the instruments to measure very low concentrations with detection limits of 0.5 ppb for the O_3 and NO_x analysers and 0.1 ppm for the CO analyser, therefore allowing for an accurate measurement of the ambient concentrations of NO_x, O_3 and CO at even very low concentrations (Reddy *et al.,* 2012; Rasheed *et al.,* 2014).

The Model 43C pulsed fluorescence SO_2 analyser developed by Thermo Environmental Instruments was used to measure the concentration of SO_2 in ppb. The model 43C operating method is based on the principle that SO_2 molecules are able to absorb ultraviolet light (Nuwarinda, 2007). The amount of radiation absorbed by the molecules provides an indication of the concentration of SO_2 in the air sample (Thermo Electric Corporation$_a$, 2004; Nuwarinda, 2007).

The Model 41C high level gas filter correlation CO_2 analyser developed by Thermo Environmental Instruments was used to measure the concentration CO_2 in parts per million (ppm) at the two sampling sites. The model 41C operating method is based on the principle that CO_2 molecules are able to emit infrared radiation (Nuwarinda, 2007). The amount of radiation emitted by the molecules provides an indication of the ambient concentration of CO_2 in the air sample (Thermo Electric Corporation$_b$, 2004; Nuwarinda, 2007).

2.2.3. Data Analysis and Data Capture

Meteorological Data

Wind field data, recorded using an RM Young weather station at Petrus Molefe and Thokoza Park, were converted into hourly averages and represented as wind rose plots, using WR Plot software developed by Lakes Environmental USA, indicating the predominant wind direction and the frequency distribution of wind velocities in m/s. Temperature and relative humidity recorded at Petrus Molefe and Thokoza Park were converted into 24 hour (daily) averages and compared against the urban background conditions. At least 85% data recovery was required for both hourly and 24 hour averages in order to accurately represent the local meteorological conditions at Thokoza Park and Petrus Molefe Eco-Park. Urban background data for temperature, rainfall and humidity were also obtained from the South African Weather Services for the Diepkloof air quality monitoring station.

There was 100% data capture for all meteorological parameters recorded during all three campaigns. However, due to the poor data capture for air quality parameters recorded during campaign one, only the results for meteorological parameters measured during campaign two and campaign three were chosen to be presented in this study.

Air Quality Data

In order to assess the local ambient air quality at Thokoza Park and Petrus Molefe-Eco Park the ambient concentrations of the chosen criteria air pollutants measured during the study were compared against the South African National Ambient Air Quality Standards. The concentrations of the pollutants measured at the two parks were averaged into either eight hour, hourly or 24 hour averages depending on the averaging period required for each respective air pollutant standard. At least 85% data recovery was chosen for both one hour and 24 hour averages in order to accurately represent the local air quality conditions at Thokoza Park and

Petrus Molefe Eco-Park and to allow for a comparison of the monitored data against the South African National air quality standards. All data were assessed and analysed using the Microsoft Excel package.

Data Quality Control

All data collected at the two urban parks were manually checked for any inaccuracies before any analysis was conducted. Data were checked for any negative readings, extremely high values and missing data. If there were a series of negative values running for more than one hour, then the negative readings were removed from the dataset and were classified as missing data. If a single negative reading was found in between at least ten accurate readings, then the negative value was replaced with an average of the upper two values. The amount of missing data was also recorded.

In this study no extremely high values were recorded. However, during campaign one a large portion of the dataset contained negative values. This provided an indication that the instruments were disrupted during the monitoring period. Air quality instruments are very sensitive to changes in electricity supply, thus repeated power disruptions, as experienced during campaign one, would result in a high level of inaccuracy within the dataset. The series of negative values obtained during campaign one were discarded and treated as missing data. There were also several incidents where there were large gaps (12 - 24 hours) in the dataset where no data was collected. These data were treated as missing data.

The percentage data capture was calculated for each parameter during all three campaigns. Percentage data capture for all pollutant parameters recorded at each park is given in Table 2-2. Data recovery at both parks for the first monitoring period (3 – 14 June 2013) ranged from 0 – 83% and fell below 85% data capture per parameter for a large portion of the parameters under study. A high level of error was associated with the data captured during campaign one for many of the parameters due to instrument failure from repeated power disruptions. As a result the data cannot be considered an accurate representation of ambient air quality in the parks for the period and thus could not be presented in this study. Data capture at both parks for the second and third monitoring period, 7 – 31 October 2013 and 27 May - 23 June 2014, was above 85% for most of the parameters and thus was considered a better representation of ambient concentrations at Petrus Molefe Eco-Park and Thokoza Park compared to the first set of results.

It should be noted that the data obtained from the SO_2 analyser during the second monitoring period, were flawed due to instrument failure and consequently could not be presented. In addition the CO_2 analyser malfunctioned and could not be fixed prior to the third monitoring campaign and thus there was no monitoring of CO_2 for the period 27 May - 23 June 2014. Therefore, CO2 concentrations are only presented for campaign two in this study.

Table 2-2: Data Capture for Air Pollutants Measured during campaign one, two and three.

CAMPAIGN	DATE	DATA CAPTURE (%)							
		CO	CO_2	SO_2	NO	NO_2	O_3	PM_{10}	$PM_{2.5}$
1	3-10 JUN 2013	83	33	33	83	0	0	83	0
	10-14 JUN 2013	75	75	75	75	< 10	0	75	0
2	7-21 OCT 2013	0	93	0	93	93	93	93	93
	21-31 OCT 2013	0	100	0	100	100	100	100	100
3	27 MAY – 13 JUN 2014	100	0	100	100	100	100	100	50
	13-23 JUN 2014	100	100	100	100	100	100	100	100

Statistical Analysis

To investigate if there was a significant difference in air quality between the two park types and the urban background site a two tailed t-Test with unequal variances and a paired t-Test statistical analysis was undertaken (Lam *et al.,* 2005; Yin *et al.,* 2011).

A t-Test is a useful statistical analysis test that can be used to compare the means of two samples to determine if there is a significant difference between the samples means in a small sample size (McDonald, 2014). A two-tailed t-Test analysis with unequal variances was undertaken to compare the mean air pollutant concentrations between Petrus Molefe Eco-Park and Thokoza Park. A paired t-Test was undertaken to compare the mean air pollutant concentrations between the urban park and the urban background site (Lam *et al.,* 2005). The two tailed t-Test with unequal variance was chosen to compare the mean parameters between the two urban park types as they were not recorded over the same time frame. The paired t-Test analysis was chosen to compare the mean parameters between the urban parks and the urban background site because they were recorded over the same time frame.

The t-Test provides a p-value which represents the probability that a difference would occur within the two samples. In this study it was assumed that the difference in the samples represented a normal distribution and 0.05 was chosen as the significance level. The Null hypothesis assumed that the difference in means was not significantly greater than zero where the alternative hypothesis assumed that the difference in means was significantly greater than zero. If the p-value provided by the t-Test was smaller than the level of significance ($\alpha = 0.05$), then the Null hypothesis was rejected, which confirmed that there was a significant difference in mean air quality parameters between the two sites of interest (Lam *et al.,* 2005; Yin *et al.,* 2011). A 95% confidence interval about the mean difference was also calculated to show the range of values in which the statistically significant mean difference lies.

The results of the t-Test analysis were compared separately between the urban background site and the two park types and discussed to allow for an investigation into the difference in air quality between the different sites.

3. Chapter Three: Air Quality Monitoring Results

The meteorological variables and air pollutant concentrations recorded at both the urban park types are presented in the chapter. The differences in air pollutant concentrations between the two urban parks and the urban background site are given. The concentrations for the criteria air pollutants recorded at the urban parks are also compared to the relevant South African National Ambient Air Quality Standards.

3.1. Meteorological Overview

3.1.1. Local Wind Fields

Campaign Two: 8 – 30 October 2013

Wind fields that were observed at Petrus Molefe during the second campaign were characteristic of winds occurring predominantly from the north-west and north-north-west (Figure 3-1). Wind speeds were moderate to fast and frequently remained within the range of 2 – 6 m/s for approximately 74% of the time with wind speeds exceeding 6m/s also recorded for 6% of the time. Calm conditions, which are defined as wind speeds less than 1 m/s, occurred less frequently for 6% of the time (Figure 3-2). A different wind field was observed for the urban background site for the period 8 – 20 October 2013 with prevailing winds occurring from the north-north-east. Faster winds were recorded at the Diepkloof monitoring station and frequently remained within the range of 2 – 6 m/s for 91% of the time.

Wind fields observed at Thokoza Park during the second campaign were characteristic of winds occurring predominantly from the north-north-east (Figure 3-1). Wind speeds were significantly slow and frequently remained below 2 m/s. Calm conditions occurred with the highest frequency for 67% of the time (Figure 3-2). A similar predominant wind direction was observed for the urban background site for the period 22 – 30 October 2013 with winds observed mostly from the north-north-east direction. However, much faster wind speeds (4 – 6 m/s) were recorded at the background site in comparison to Thokoza Park.

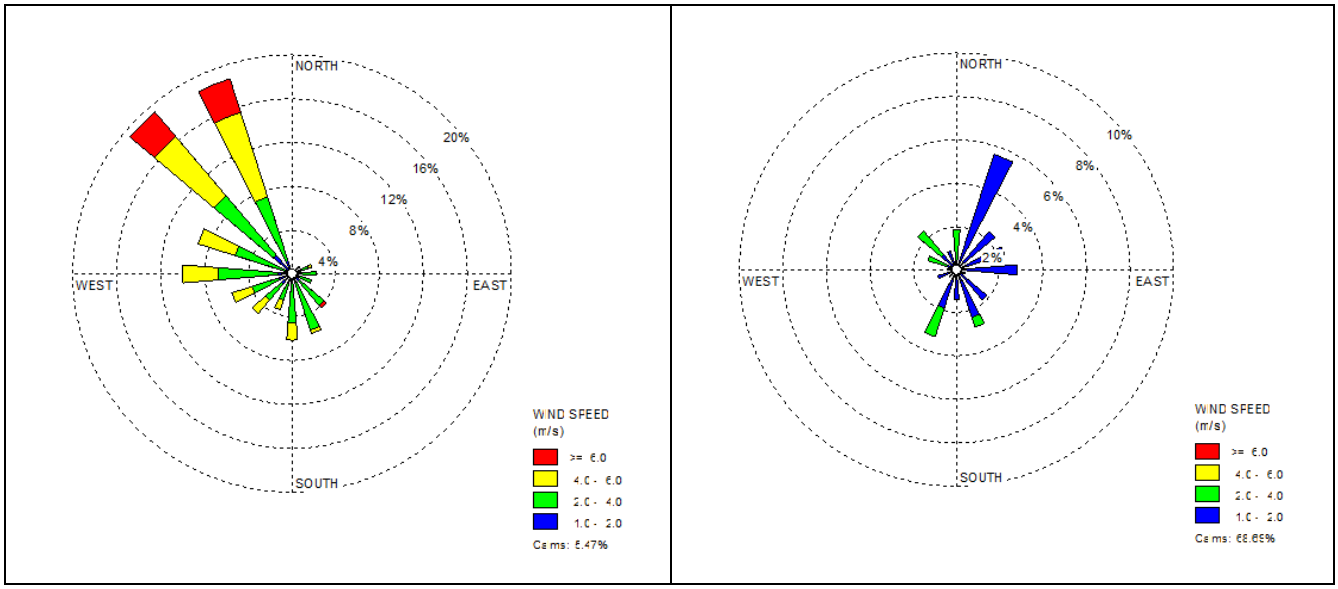


Figure 3-1: Campaign two wind rose plots for Petrus Molefe Eco-Park for 8 - 20 October 2013 (left) and Thokoza Park for 22 - 30 October 2013 (right).

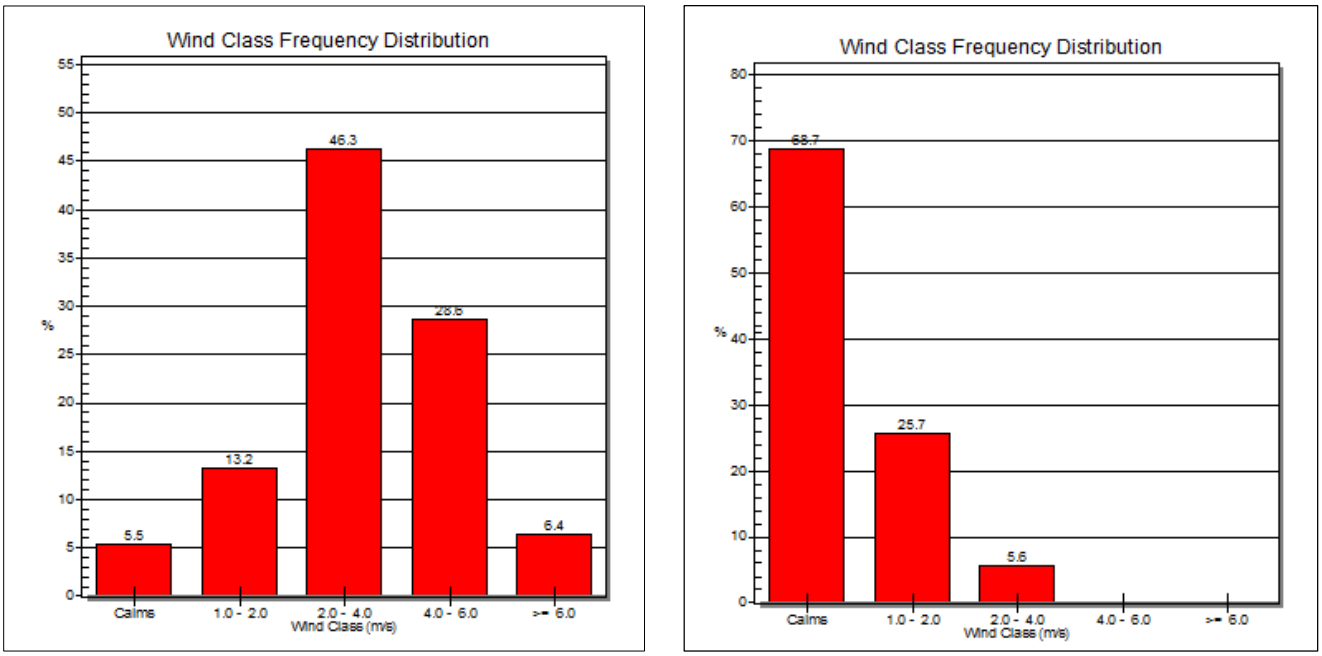


Figure 3-2: Campaign two wind class frequency distribution for Petrus Molefe Eco-Park for 8 - 20 October 2013 (left) and Thokoza Park for 22 - 30 October 2013 (right).

Slight diurnal variation in winds is observed in the meteorological dataset at Petrus Molefe Eco-Park for the period 8 – 20 October 2013 (Figure 3-3). Throughout the day and night a predominant north-westerly and north-north-westerly wind is evident. However, during the afternoon period (12:00-18:00), a south-westerly component is established. Moderate to fast winds are observed throughout the day and night at Petrus Molefe in campaign two.

Diurnal variation in winds is observed at Thokoza Park for the period 22 – 30 October 2013 (Figure 3-4). In the morning period (00:00 – 12:00) winds are observed predominantly from the north-north-east. In the afternoon a prevailing south-south-west wind is established with winds also observed from the north and north-west. In the evening, prevailing north-westerly and south-south-westerly winds are observed. Very slow winds are recorded throughout the day and night at Thokoza Park in campaign two.

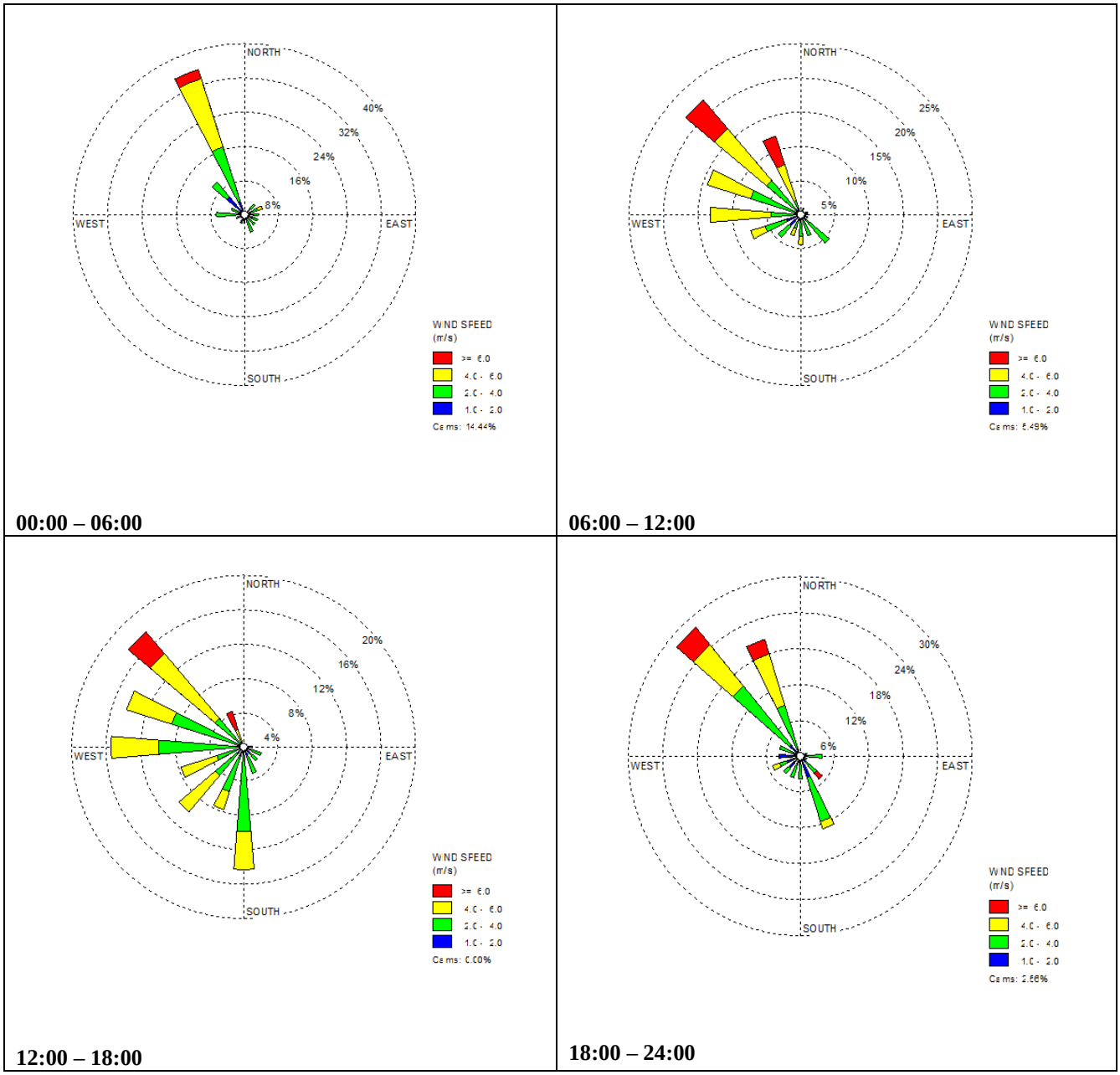


Figure 3-3: Diurnal variation in winds at Petrus Molefe Eco-Park in campaign two (8 – 20 October 2013).

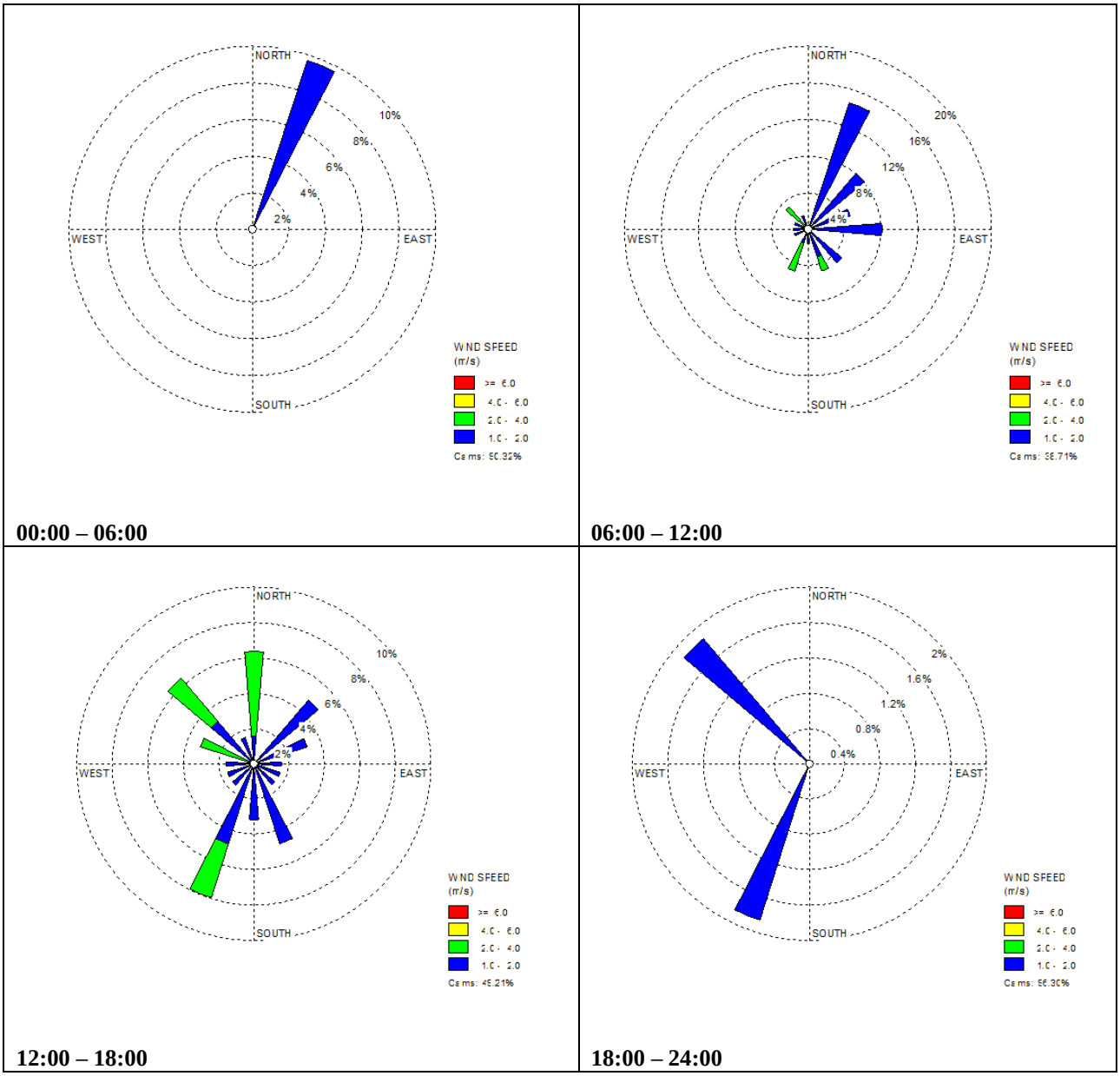


Figure 3-4: Diurnal variation in winds at Thokoza Park in campaign two (22 – 30 October 2013).

Campaign Three: 28 May – 22 June 2014

Wind fields at Petrus Molefe Eco-Park during campaign three were characteristic of winds occurring predominantly from the north-west and west (Figure 3-5). Wind speeds were generally slow to moderate and frequently remained within the range of 1 – 4 m/s. Wind speeds exceeding 6m/s were recorded for 2% of the time. Calm conditions occurred frequently for 31% of the time (Figure 3-6). A similar wind field was recorded for the urban background site with prevailing winds observed from the north-west and north-north-west for the period 28 May – 12 June 2014. Faster winds were observed at Diepkloof monitoring station and frequently remained within the range of 4 – 6 m/s for 45.6% of the time.

Wind fields recorded at Thokoza Park during campaign three were characteristic of winds occurring predominantly from the south-westerly and south-easterly directions (Figure 3-5). Wind speeds were again significantly slow and frequently remained below 2 m/s. Calm conditions occurred with the highest frequency for 52% of the time (Figure 3-6). Winds recorded for the urban background site for the period 14 – 22 June 2014 were similar to Thokoza Park with prevailing winds observed from the south and south-west for the period 14 – 22 June 2014. Yet again much faster winds (4 – 6 m/s) were recorded at the background site in comparison to Thokoza Park.

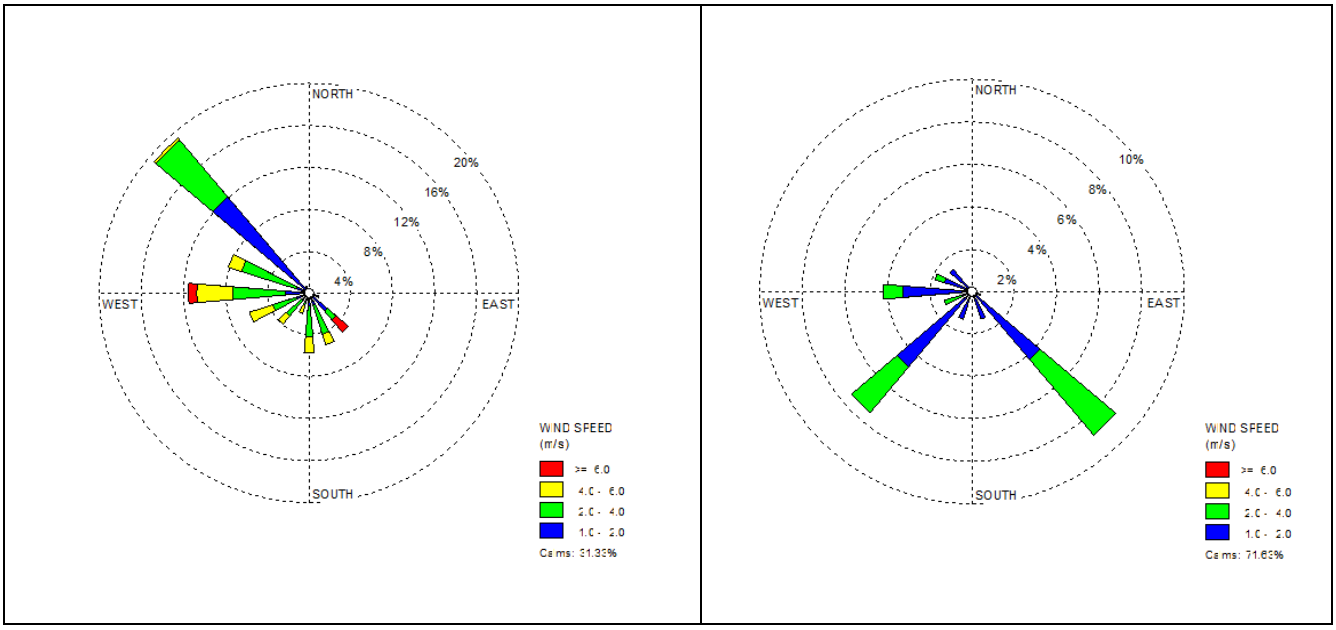


Figure 3-5: Campaign three wind rose plots for Petrus Molefe Eco-Park for 28 May - 12 June 2014 (left) and Thokoza Park for 14 - 22 June 2014 (right).

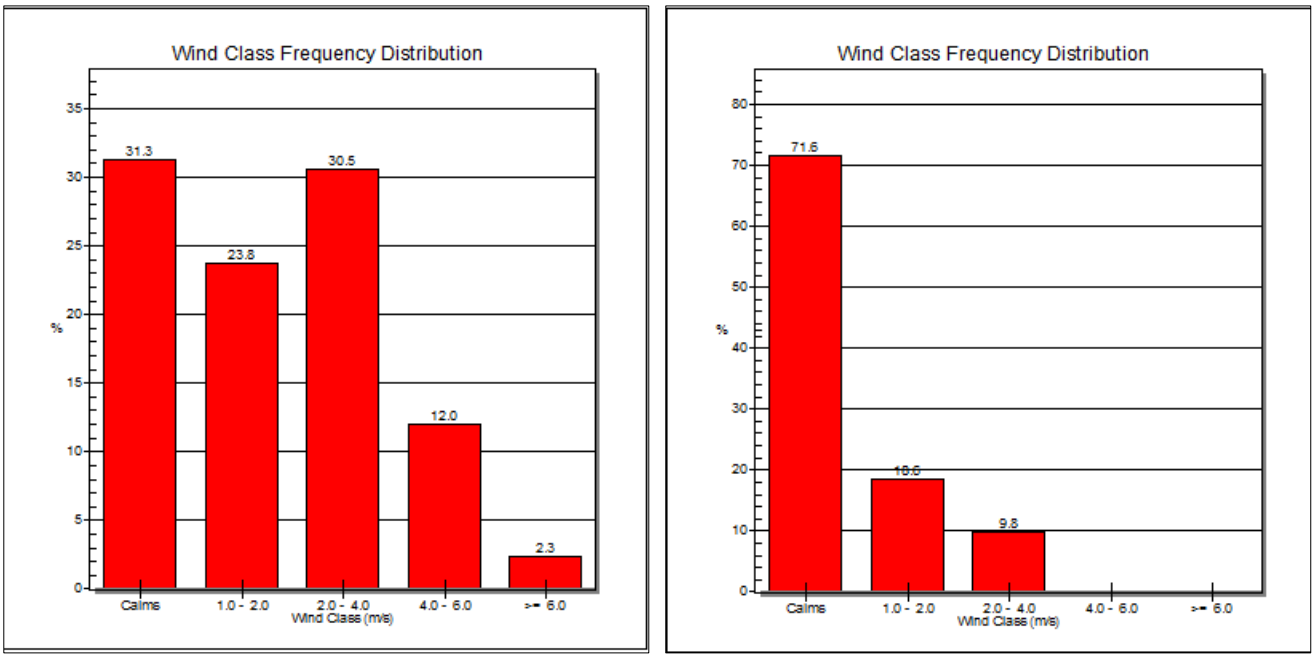


Figure 3-6: Campaign three wind class frequency distribution for Petrus Molefe Eco-Park for 28 May - 12 June 2014 (left) and Thokoza Park for 14 - 22 June 2014 (right).

Diurnal variation in winds is observed in the meteorological dataset at Petrus Molefe Eco-Park for the period 28 May to 12 June 2014 (Figure 3-7). In the early morning (00:00 – 06:00) and evening (18:00 -23:00) periods, prevailing winds are observed from the north-westerly direction. In the late morning (06:00 – 12:00) and afternoon (12:00 – 18:00) periods, a strong westerly wind is observed with additional prevailing winds also observed from the west-north-west and west-south-west during the late morning period. A higher frequency of calm winds occurs in early morning and evening periods.

Diurnal variation in winds is observed at Thokoza Park for the period 14 – 22 June 2014 (Figure 3-8). Throughout the day and night a high frequency of calm wind is observed from the south-easterly direction, however during the afternoon (12:00 – 18:00) a predominant south-westerly wind is established. Slightly faster winds are shown during the late morning and afternoon periods (06:00 – 18:00) with only calm winds observed during the early morning (00:00 – 06:00 and evening (18:00 – 24:00) periods.

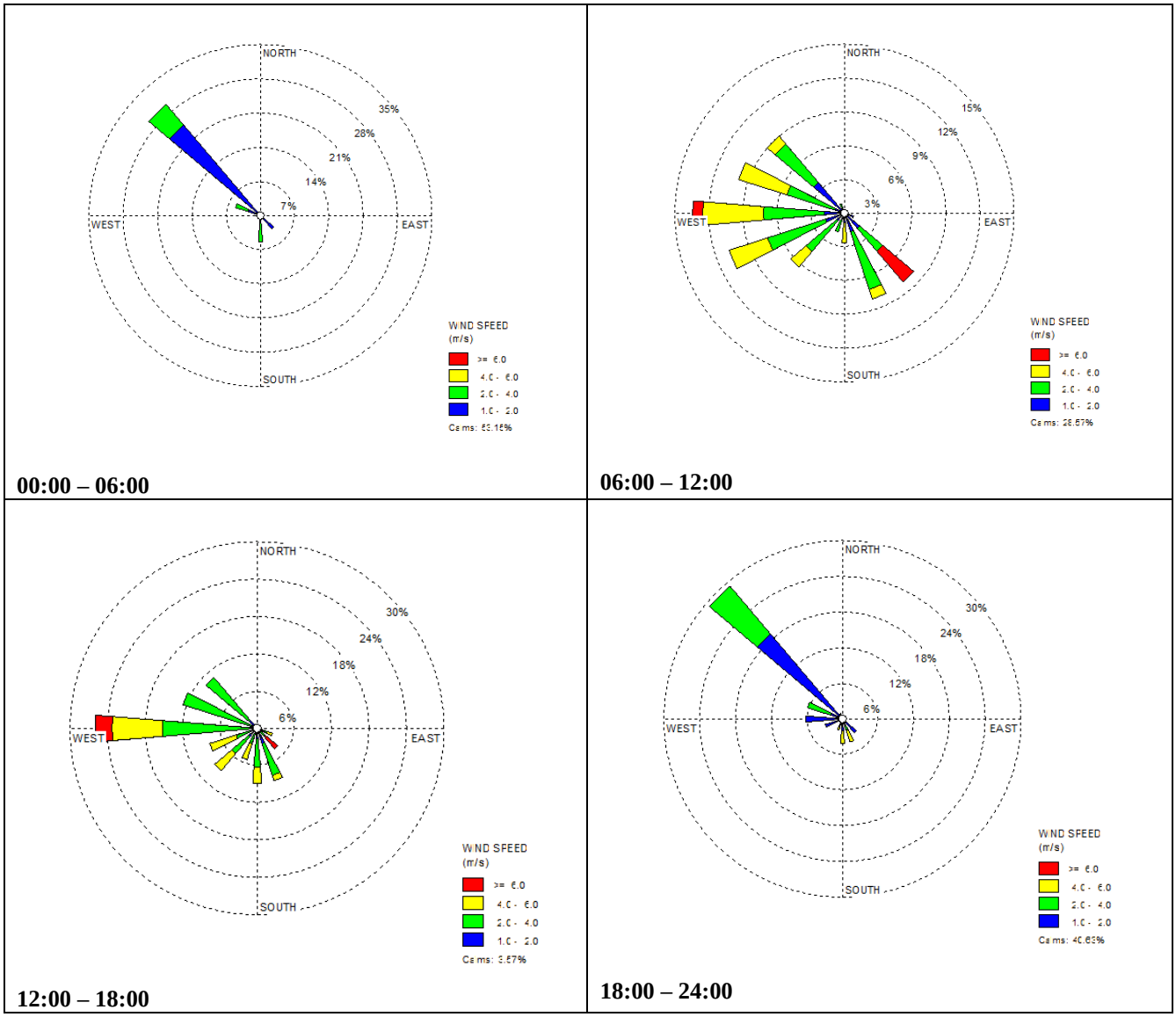


Figure 3-7: Diurnal variation in winds at Petrus Molefe Eco-Parkin campaign three (28 May – 12 June 2014).

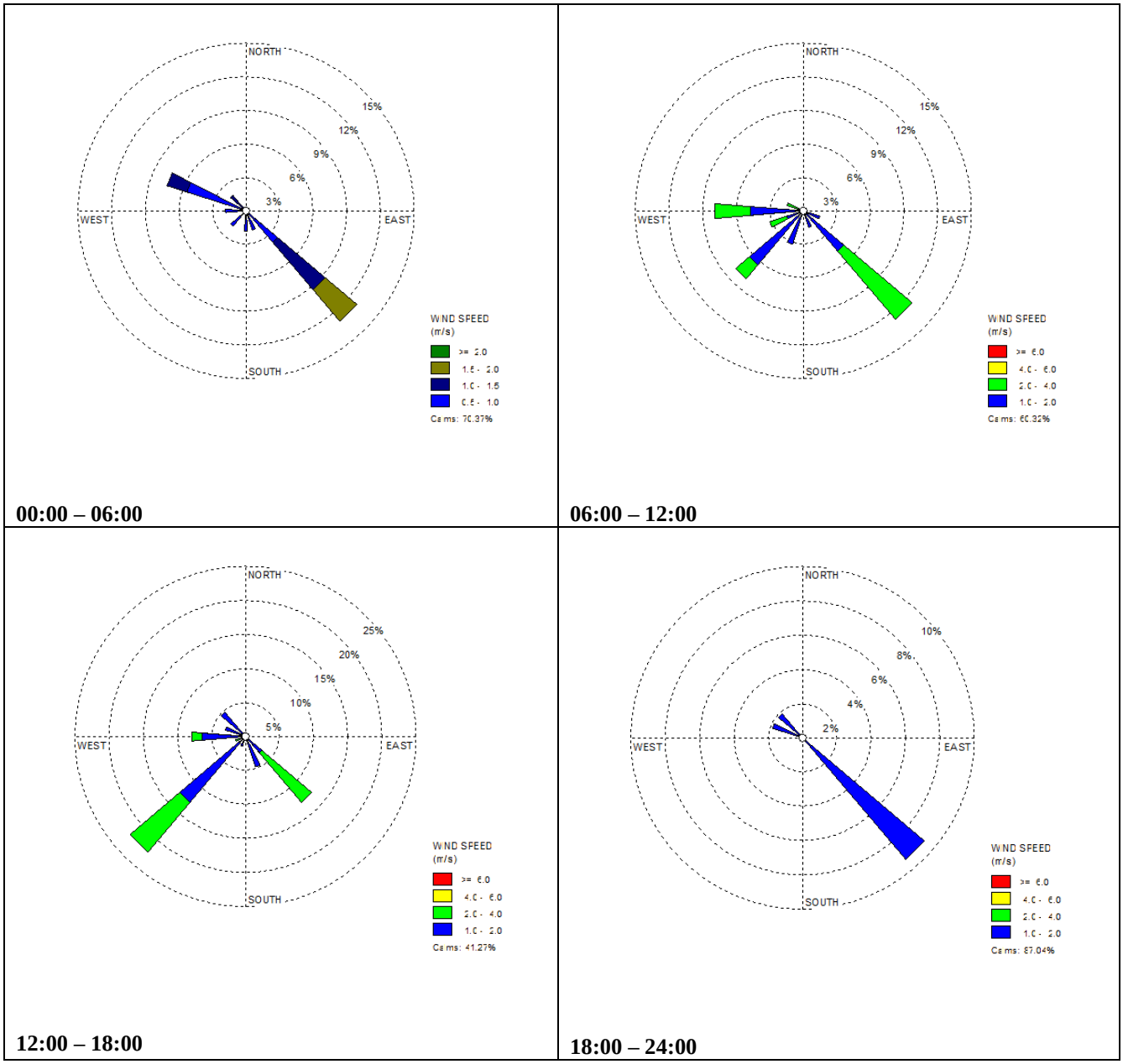


Figure 3-8: Diurnal variations in winds at Thokoza Park in campaign three (14 – 22 June 2014).

3.1.2. Temperature, Relative Humidity and Rainfall

Temperature

Daily average temperatures measured at Petrus Molefe Eco-Park during campaign two ranged from 13.6 - 23.7 ^{o}C with a mean temperature of 19.2 ^{o}C recorded for the period (Figure 3-9). Daily average temperatures measured at Thokoza Park during campaign two were slightly lower compared to Petrus Molefe Eco-Park and ranged between 14.2 - 20.2 ^{o}C with a mean temperature of 16.9 ^{o}C recorded (Figure 3-10).

Although the temperatures recorded at Thokoza Park were only slightly lower compared to Petrus Molefe Eco-Park during campaign two, the two tailed t-Test indicated a significant difference ($P < 0.05$) in the mean temperatures between the two park types (Table 3-2).

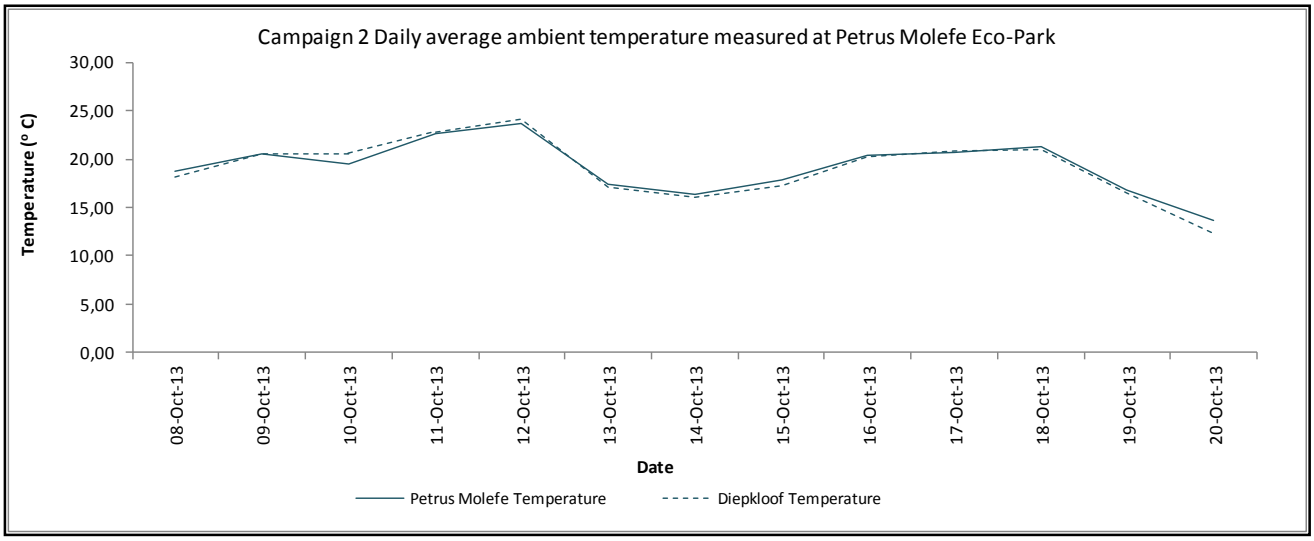


Figure 3-9: Campaign two daily average temperature measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

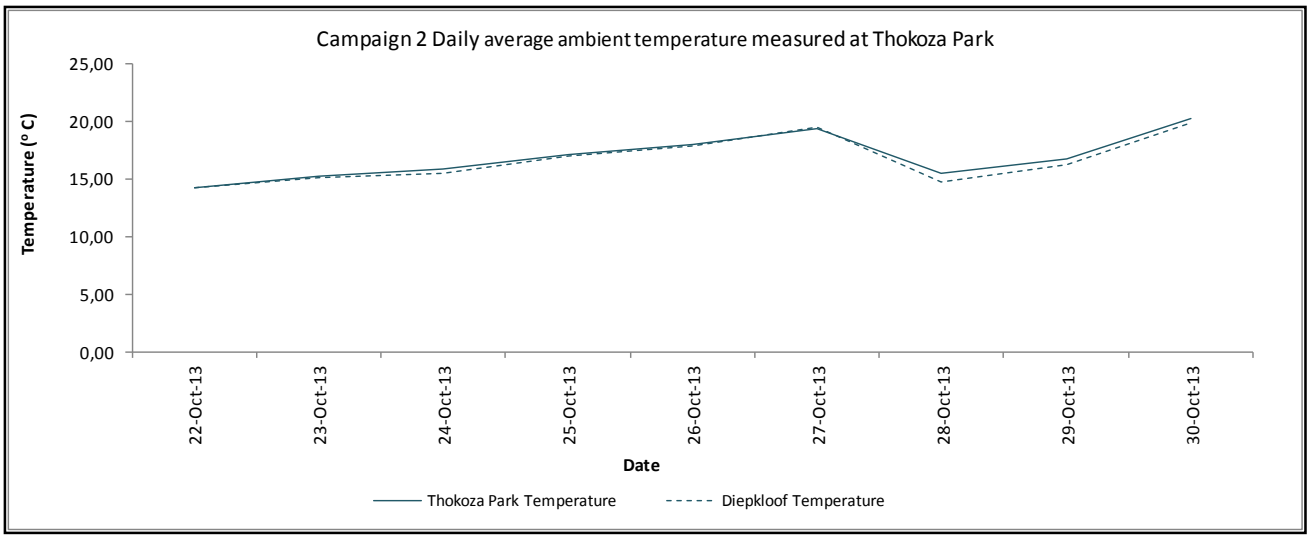


Figure 3-10: Campaign two daily average temperature measured at Thokoza Park (22 - 30 October 2013).

Air quality monitoring during campaign three was conducted in the early winter season (May - June). A significant temperature decline was observed on the 6 June 2014 at Petrus Molefe Eco-Park, which may likely have been caused by the onset of a cold front (Figure 3-11).

As a result daily average temperatures measured at Petrus Molefe Eco-Park and Thokoza Park during campaign three were cooler compared to campaign two ranging between 2.9 - 14.6 ^{o}C and 7.5 - 11.9 ^{o}C respectively (Figure 3-11 and 3-12). Unlike campaign two, the average temperatures measured at Thokoza Park appeared to be slightly warmer compared to Petrus Molefe Eco-Park with an overall mean temperature of 9.4 ^{o}C and 9.0 ^{o}C recorded at Thokoza Park and Petrus Molefe Eco-Park respectively (Table 3-1). However, the two tailed t-Test indicated that the difference in the mean temperatures between the two park types during campaign three was statistically insignificant (P > 0.05) (Table 3-2).

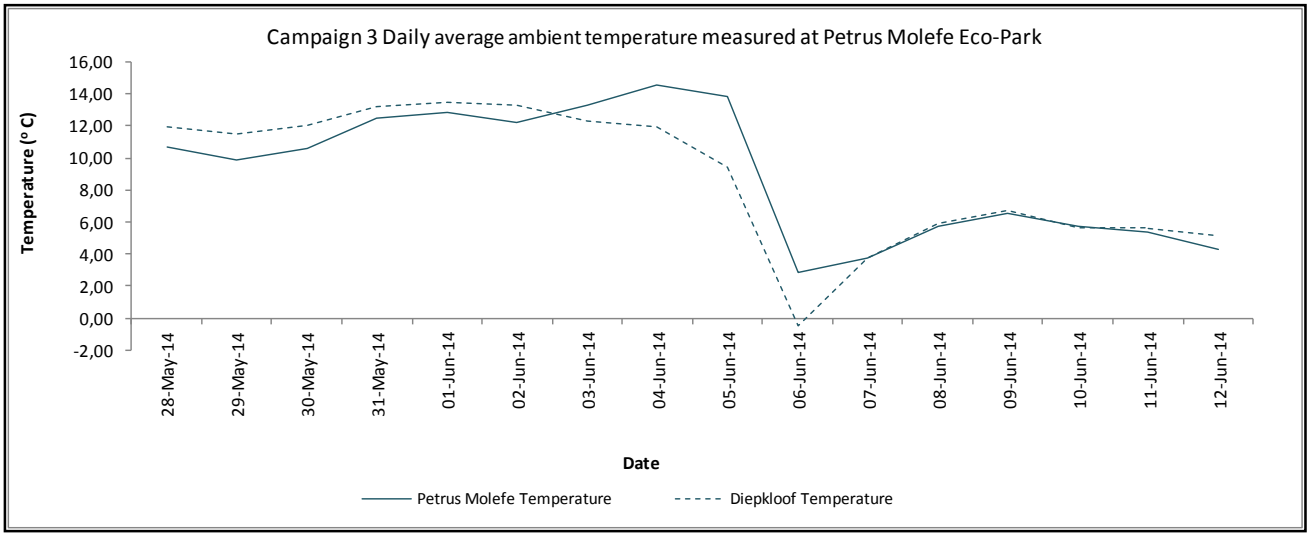


Figure 3-11: Campaign three daily average temperature measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

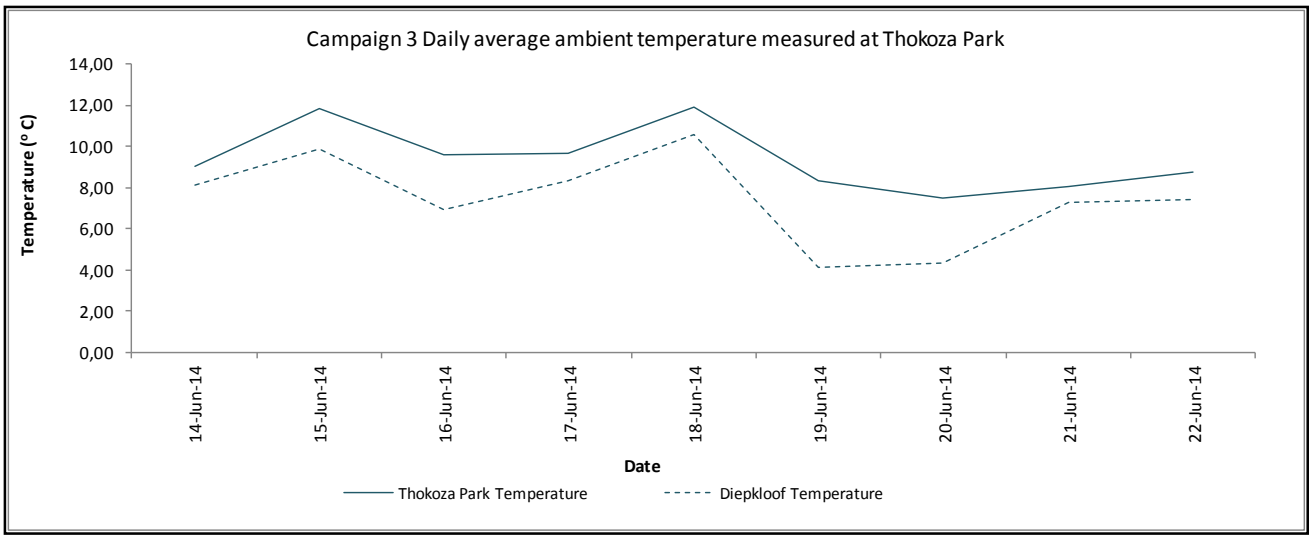


Figure 3-12: Campaign three daily average temperature measured at Thokoza Park (14 - 22 June 2014).

Table 3-1: Data summary of ambient temperatures (ºC) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	19.2	16.9
Median	19.5	16.7
Standard Deviation	2.8	2.0
Minimum	13.6	14.2
Maximum	23.7	20.2
Campaign Three		
Mean	9.0	9.4
Median	10.2	9.0
Standard Deviation	4.0	1.6
Minimum	2.9	7.5
Maximum	14.6	11.9

Table 3-2: Two tailed t-Test analysis with unequal variances of ambient temperatures at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	20	2.22	0.04	P < 0.05
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	21	-0.32	0.75	P > 0.05

A comparison of ambient temperatures between the urban parks and the urban background site was conducted and the results of the paired t-Test analysis is given in Table 3-3 below. The paired t-Test analysis indicated that there was no significant difference in the mean temperatures between Petrus Molefe Eco-Park and the urban background site for both monitoring campaigns (P > 0.05) (Table 3-3). On the other hand the paired t-Test analysis showed that the difference in the mean temperatures between Thokoza Park and the urban background site was statistically significant during both monitoring campaigns (P < 0.05) (Table 3-3).

Ambient temperatures measured at Thokoza Park during both campaigns were mostly higher compared to the urban background site (Figure 3-10 and 3-12). However, a more statistically significant mean difference in ambient temperatures between Thokoza Park and the urban background site was observed for campaign three compared to campaign two (Table 3-3). The temperature profile for Thokoza Park and the urban background site in campaign two suggest that very similar ambient temperatures were recorded at both sites. Slightly higher mean temperature differences are observed for Thokoza Park compared to Petrus Molefe Eco-Park, which suggests that the ambient temperatures at Petrus Molefe Eco-Park are more closely related with the urban background site compared to Thokoza Park (Table 3-3).

Table 3-3: Two tailed paired t-Test analysis of urban park ambient temperatures (°C) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	0.88	0.4	P > 0.05	0.13	-1	0.47
Thokoza Park (older trees)	8	2.72	0.03	P < 0.05	0.24	0.03	0.45
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	0.47	0.64	P > 0.05	0.2	-0.7	1.2
Thokoza Park (older trees)	8	5.12	0.0009	P < 0.05	2	1.1	2.9

Rainfall and Relative Humidity

There was no rainfall recorded at both the urban parks and the urban background site during both monitoring campaigns therefore the influence of rainfall on air quality was absent during campaign two and three.

Relative humidity measured at Petrus Molefe Eco-Park during campaign two ranged between 20 to 89 % with a mean percentage of 49 % recorded for the period (Figure 3-13). Relative humidity measured at Thokoza Park during campaign two was slightly higher compared to Petrus Molefe Eco-Park ranging between 52 to 78 % with a mean percentage of 68 % recorded (Figure 3-14). The two tailed t-Test analysis showed that the mean difference in relative humidity between the two urban park types was statistically significant ($P < 0.05$) (Table 3-5).

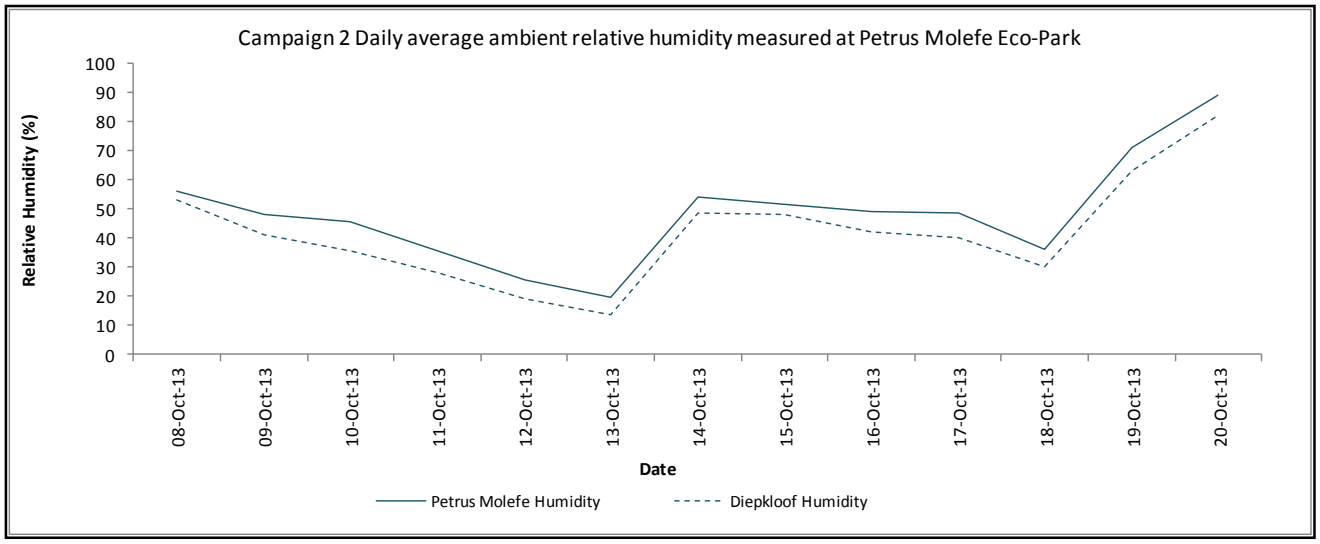


Figure 3-13: Campaign two daily average relative humidity measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

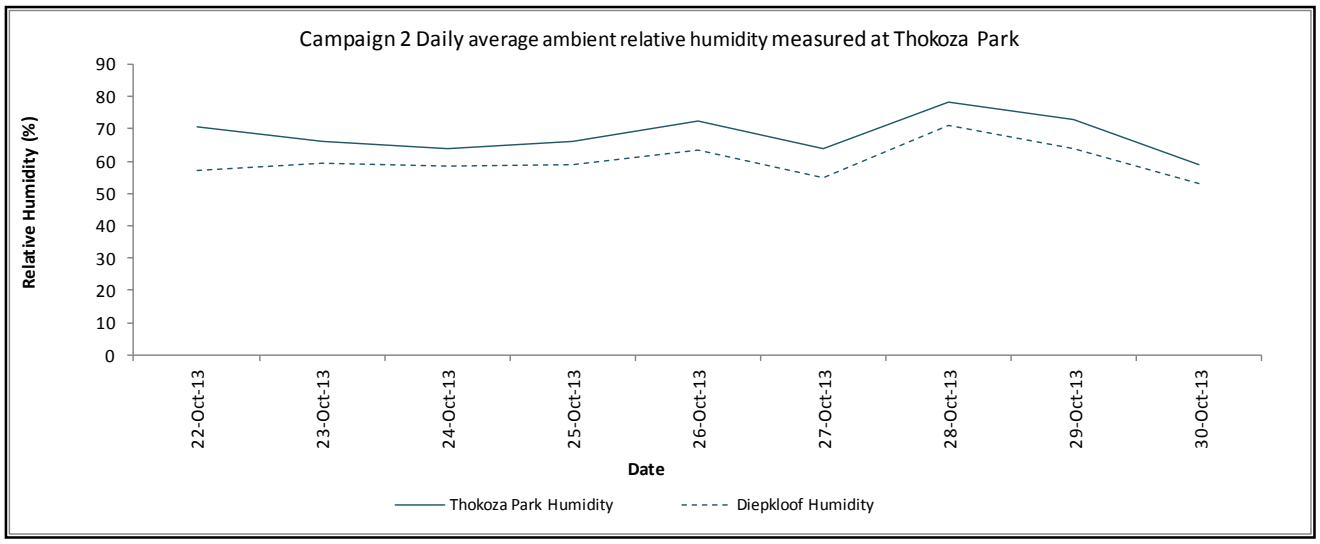


Figure 3-14: Campaign two daily average relative humidity measured at Thokoza Park (22 - 30 October 2013).

Relative humidity measured at Petrus Molefe Eco-Park and Thokoza Park during campaign three ranged between 37 to 61 % and 41 to 56 %, respectively (Figure 3-15 and 3-16). Unlike campaign two, relative humidity measured at Thokoza Park was slightly lower, although statistically insignificant (P > 0.05), compared to Petrus Molefe Eco-Park with an overall mean percentage of 49% recorded for Thokoza Park and 54% recorded for Petrus Molefe Eco-Park (Table 3-4 and 3-5).

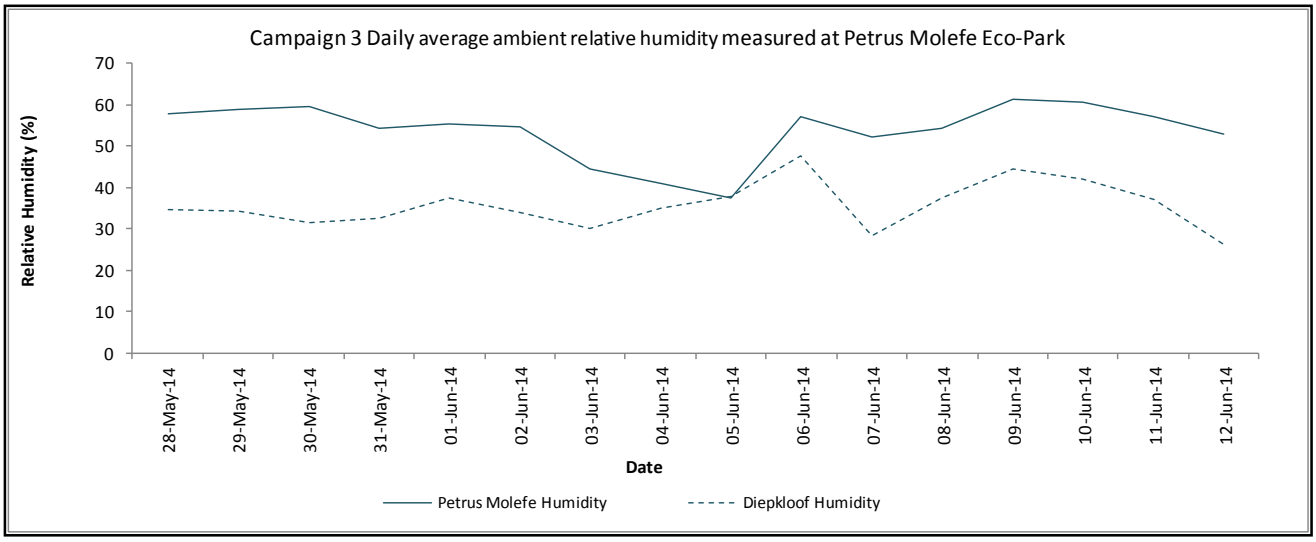


Figure 3-15: Campaign three daily average relative humidity measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

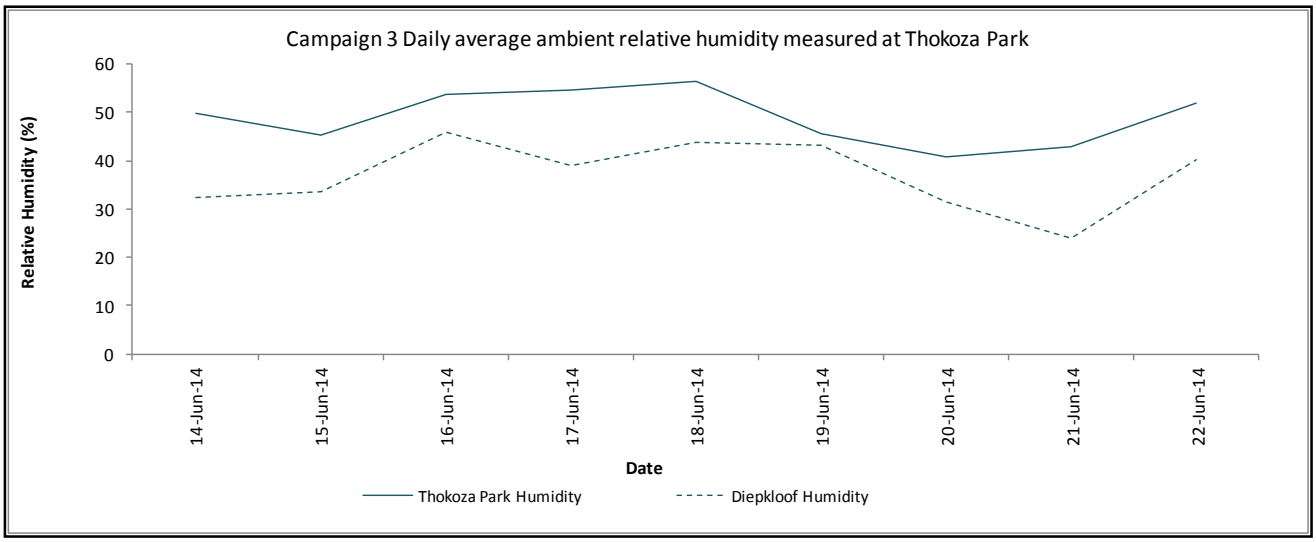


Figure 3-16: Campaign three daily average relative humidity measured at Thokoza Park (14 - 22 June 2014).

Table 3-4: Data summary of ambient relative humidity (%) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	48.5	68.2
Median	48.6	66.2
Standard Deviation	18.0	6.0
Minimum	19.9	58.8
Maximum	88.9	78.4
Campaign Three		
Mean	53.6	48.9
Median	54.9	49.7
Standard Deviation	7.0	5.6
Minimum	37.2	40.8
Maximum	61.3	56.4

Table 3-5: Two tailed t-Test analysis with unequal variances of relative humidity at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	15	-3.65	0.002	P < 0.05
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	20	1.84	0.08	P > 0.05

The paired t-Test analysis indicated that relative humidity at both the urban parks was statistically significantly different to the urban background site during both monitoring campaigns ($P < 0.05$), except at Petrus Molefe Eco-Park during campaign two ($P > 0.05$) (Table 3-6). Relative humidity at both the urban parks was higher compared to the urban background site during both monitoring campaigns.

The mean differences in relative humidity between the urban background site and the two urban parks were fairly higher during campaign three, particularly at Petrus Molefe Eco-Park, compared to campaign two (Table 3-6). This implies that the local relative humidity conditions at the two urban parks during campaign three were not as closely related to the urban background conditions as observed during campaign two.

Table 3-6: Two tailed paired t-Test analysis of urban park the urban background sitedity (%) compared against urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	13.05	1.9	P > 0.05	6.56	5.46	7.65
Thokoza Park (older trees)	8	10.39	6.3×10^{-6}	P < 0.05	8.20	6.38	10.00
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	9.39	1.1×10^{-7}	P < 0.05	18.00	13.93	22.11
Thokoza Park (older trees)	8	6.99	0.0001	P < 0.05	12.00	8.00	15.85

3.2. Air Pollutant Concentrations

3.2.1. Particulate Matter (PM_{10} and $PM_{2.5}$)

3.2.1.1 Particulate Matter with an Aerodynamic Diameter equal to or less than 10 μm (PM_{10})

Ground level PM_{10} concentrations were measured at Petrus Molefe Eco-Park and Thokoza Park during the second and third monitoring campaigns. Concentrations of PM_{10} measured during the second campaign were below the South African daily average PM_{10} standards of 120μg/m^3 (pre-2015) and 75μg/m^3 (2015) (Figure 3-17 and 3-18). Concentrations of PM_{10} measured during the third monitoring campaign were much higher compared to the second monitoring campaign and exceeded both the standards of 120μg/m^3 (pre-2015) and 75μg/m^3 (2015) at both parks for most of the campaign (30 – 31 May 2014, 7 June 2014, 10 – 12 June 2014, 14 – 18 June 2014 and 20 – 22 June 2014) (Figure 3-19 and 3-20).

The two tailed t-Test analysis indicated a significant difference in mean PM_{10} concentrations between the two park types (P < 0.05) for both monitoring campaigns (Table 3-8). Daily average PM_{10} concentrations recorded at Thokoza Park were higher compared to Petrus Molefe Eco-Park for campaign two and ranged from 9.3 – 33.7 μg/m^3 at Petrus Molefe Eco-Park and 33.6 – 66.8 μg/m^3 at Thokoza Park (Table 3-7).

Similarly daily average PM_{10} concentrations recorded at Thokoza Park during campaign three were considerably higher compared to Petrus Molefe Eco-Park and ranged from 21.0 – 117.6 µg/m^3 at Petrus Molefe Eco-Park and 55.2 – 255.2 µg/m^3 at Thokoza Park (Table 3-7).

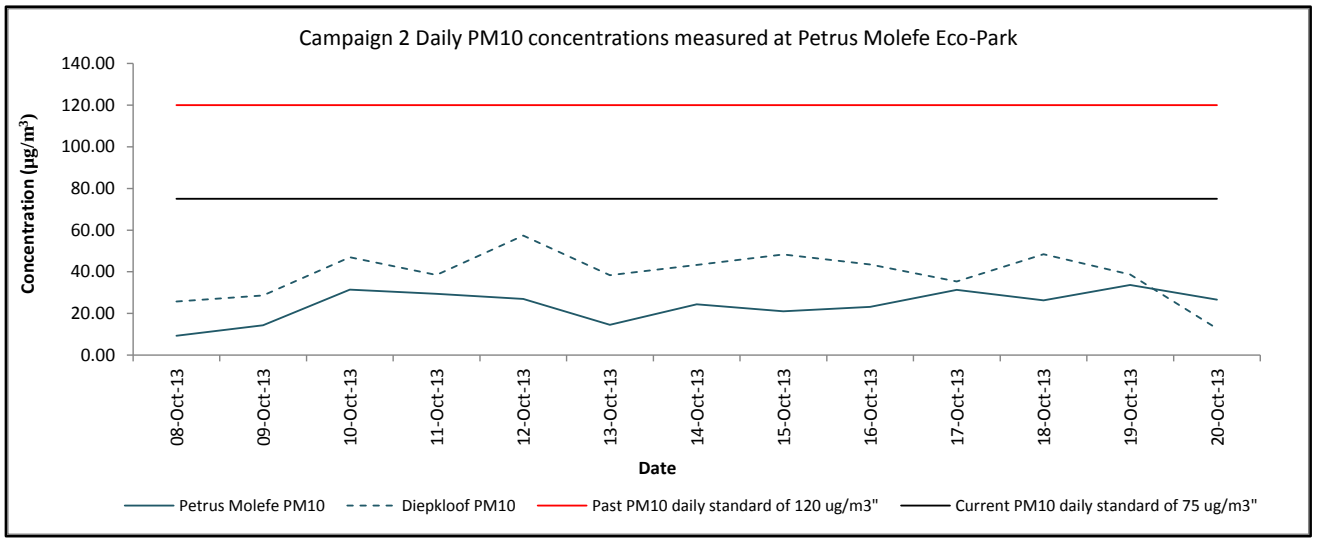


Figure 3-17: Campaign two daily average PM_{10} concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

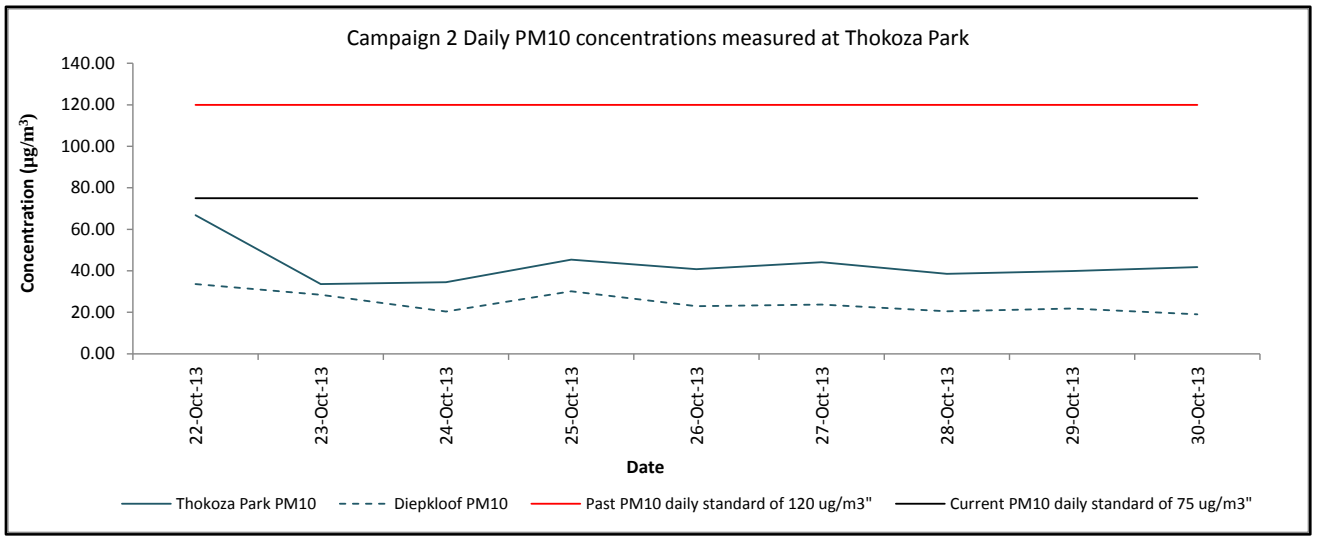


Figure 3-18: Campaign two daily average PM_{10} concentrations measured at Thokoza Park (22 - 30 October 2013).

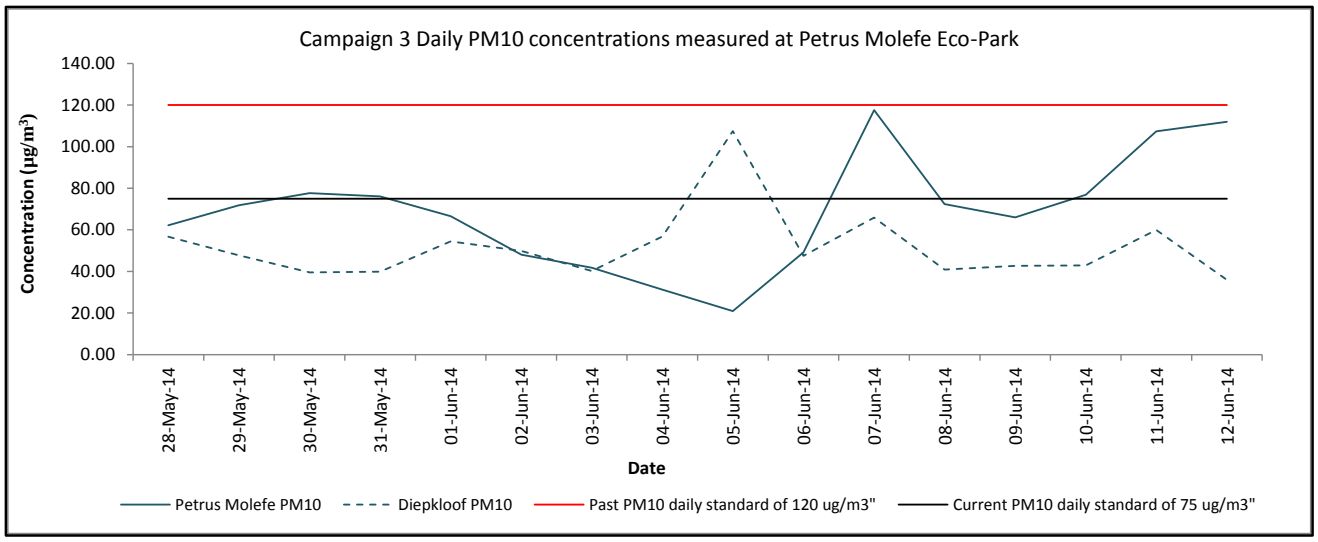


Figure 3-19: Campaign three daily average PM_{10} concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

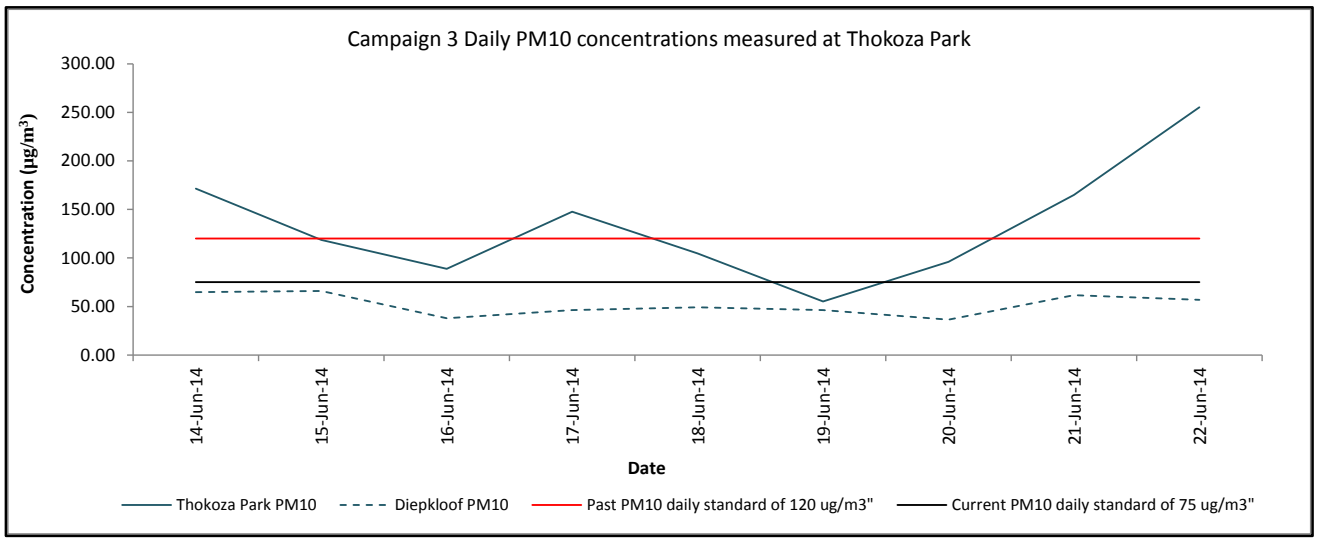


Figure 3-20: Campaign three daily average PM_{10} concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-7: Data summary of ambient PM_{10} concentrations (µg/m3) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	24.01	42.85
Median	26.29	40.83
Standard Deviation	7.43	9.81
Minimum	9.26	33.61
Maximum	33.69	66.83
Campaign Three		
Mean	68.59	133.6
Median	69.15	118.6
Standard Deviation	27.4	59.2
Minimum	20.9	55.2
Maximum	117.6	255.2

Table 3-8: Two tailed t-Test analysis with unequal variances of PM_{10} concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	14	-4. 86	P < 0.05	0.0003
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	10	-3.11	P < 0.05	0.01

The paired t-Test analysis suggests that there was a significant difference in mean PM_{10} concentrations between the two urban parks and the urban background site for both monitoring campaigns ($P < 0.05$), with the exception of Petrus Molefe Eco-Park during campaign two ($P > 0.05$) (Table 3-9).

The difference in mean PM_{10} concentrations between the background site and the two urban parks shows that higher concentrations of PM_{10} were recorded at Thokoza Park compared to the urban background site during campaign two and three. On the other hand the concentrations of PM_{10} recorded at Petrus Molefe Eco-Park were only higher compared to the urban background site during the winter season over campaign three (Table 3-9). The concentrations of PM_{10} recorded at both the urban parks during the second campaign appeared to have followed a similar pattern to that of the background site, however this was not apparent during the third campaign. The concentrations of PM_{10} recorded at both the urban parks during campaign three fluctuated over the period and followed a different pattern to that observed for the urban background site (Figure 3-19 and 3-20).

Table 3-9: Two tailed paired t-Test analysis of urban park ambient PM_{10} concentrations ($\mu g/m^3$) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	-4.58	0.0006	P < 0.05	-15.00	-21.97	-7.80
Thokoza Park (older trees)	8	7.38	0.00008	P < 0.05	18.31	12.56	24.04
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	1.82	0.09	P > 0.05	16.83	-2.89	36.56
Thokoza Park (older trees)	8	4.55	0.002	P < 0.05	81.92	40.40	123.45

Diurnal variation in air pollution studies can be described as the change in air pollutant concentrations over a 24-hour time period. A typical diurnal signature, for example, would indicate a rise in air pollutant concentrations in the early morning and evening periods as a result of increased vehicle emissions during peak traffic times. Diurnal variation in PM_{10} concentrations during the second monitoring campaign was not observed at the two urban parks or the urban background site. The PM_{10} concentrations recorded at both the urban parks were relatively consistent throughout the day and night with only a very slight increase in the concentrations during the early morning and evening periods (Figure 3-21 and 3-22).

Two distinct differences in the PM_{10} diurnal variation plots are observed between Petrus Molefe Eco-Park and Thokoza Park for campaign two. PM_{10} concentrations recorded at Petrus Molefe Eco-Park gradually increased over a longer time frame during the early morning (3:00 – 7:00) and evening (18:00 – 22:00) periods, whereas at Thokoza Park they appeared to have increased slightly over a shorter time frame during the morning (5:00 – 7:00) and evening (19:00 – 21:00) periods (Figure 3-21 and 3-22). The diurnal variation plots for campaign two furthermore clearly show that the concentrations of PM_{10} at Thokoza Park are significantly higher compared to Petrus Molefe Eco-Park and the urban background site.

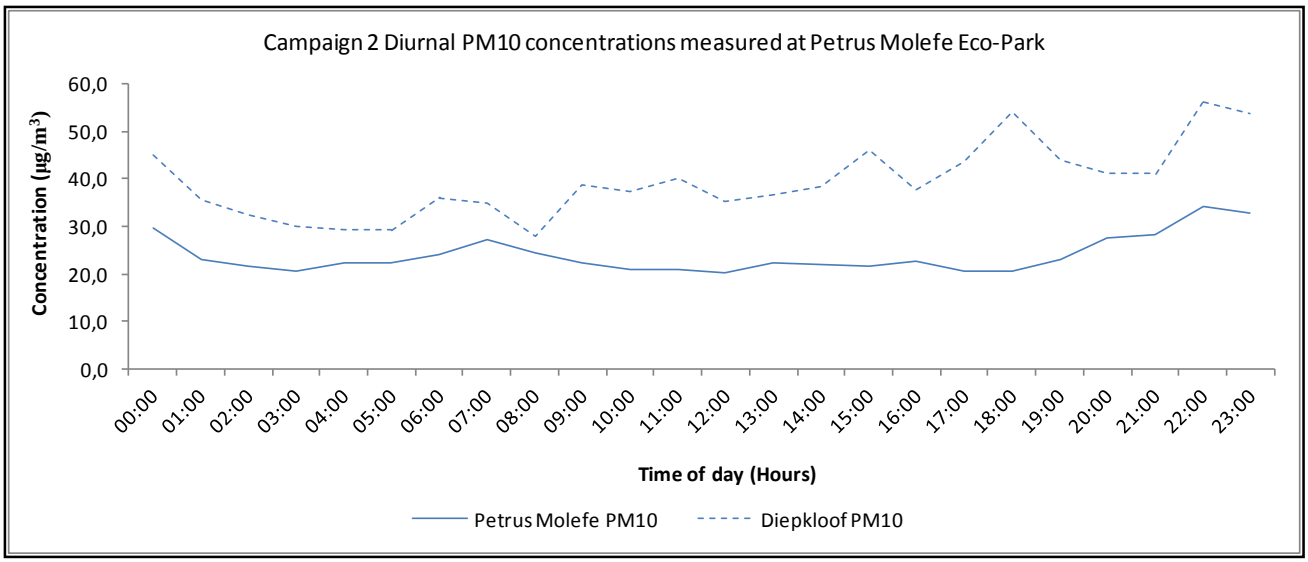


Figure 3-21: Campaign two diurnal PM_{10} concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

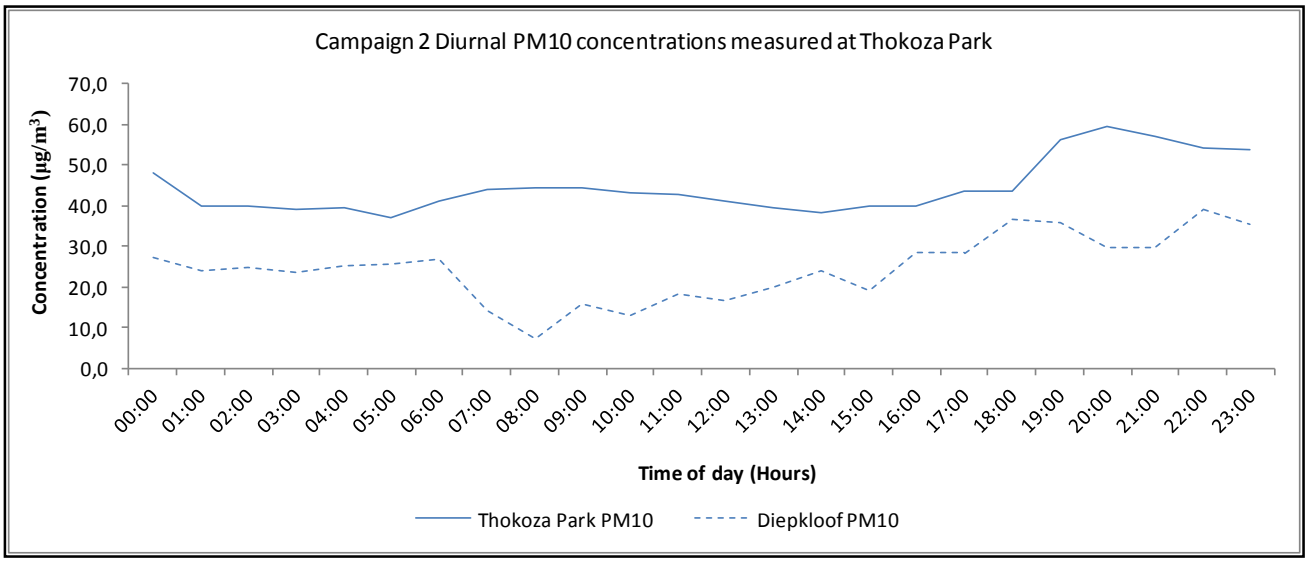


Figure 3-22: Campaign two diurnal PM_{10} concentrations measured at Thokoza Park (22 - 30 October 2013).

Diurnal variation in PM10 concentrations is observed at Petrus Molefe Eco-Park during campaign three. An increase in PM_{10} concentrations is observed in the morning period (5:00 – 9:00) and again in the evening period (17:00 – 21:00) (Figure 3-23). The PM_{10} concentrations recorded at Thokoza Park also increased in the morning and evening periods during campaign three, but they also increased around midday between 11:00 – 14:00 (Figure 3-24). Like campaign two, no diurnal variation is observed in PM_{10} concentrations at the urban background site during campaign three.

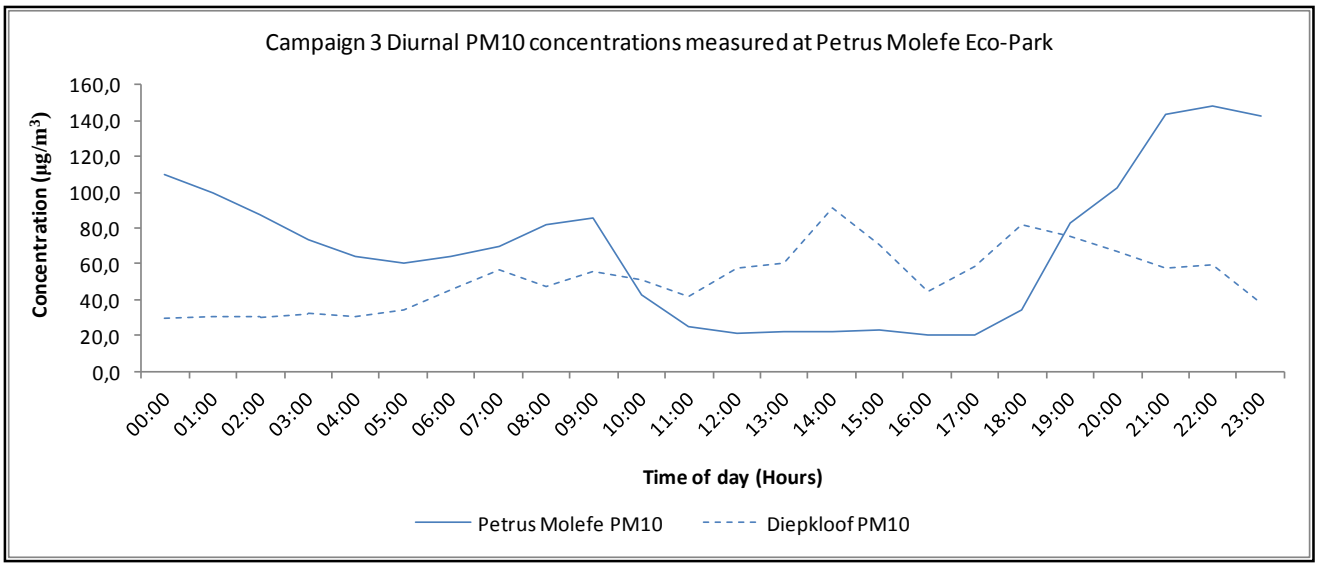


Figure 3-23: Campaign three diurnal PM_{10} concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

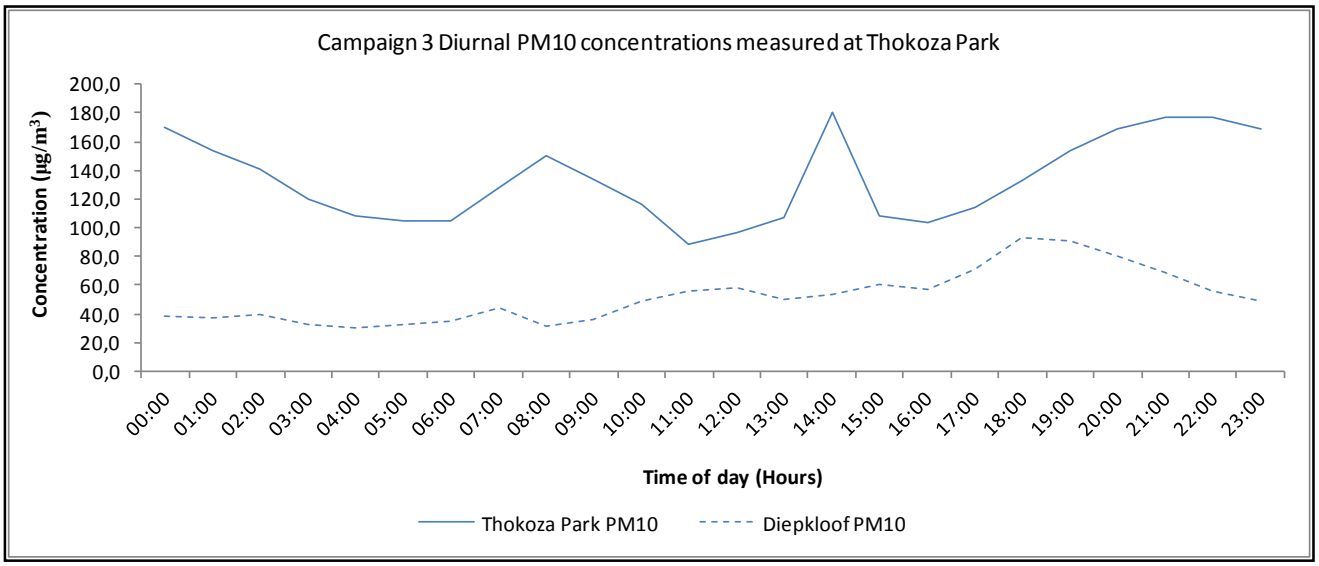
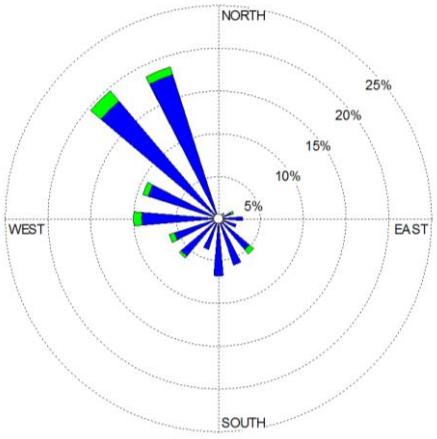


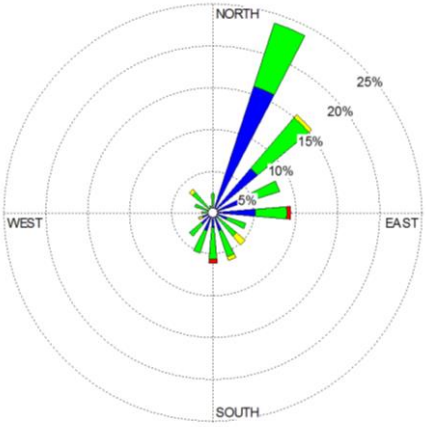
Figure 3-24: Campaign three diurnal PM_{10} concentrations measured at Thokoza Park (14 - 22 June 2014).

A pollution rose is a means of illustrating the frequency distribution of wind direction temporally correlated with an air pollutant, in this case PM_{10}. The pollution rose for PM_{10} during campaign two and three again shows that higher concentrations of PM_{10} were associated with Thokoza Park compared to Petrus Molefe Eco-Park. The pollution rose plots for PM_{10} also show that higher concentrations of PM_{10} were recorded during the winter season (campaign three) compared to the spring season (campaign two).

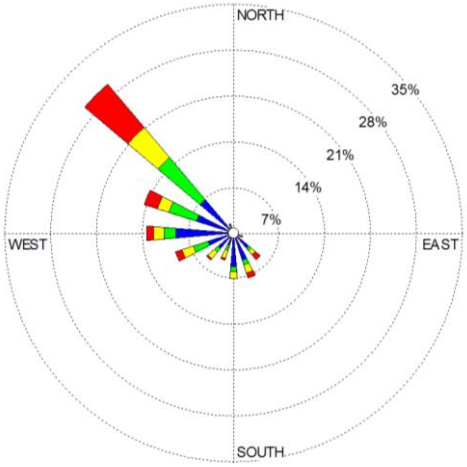
At Petrus Molefe higher PM_{10} concentrations were shown from the north-westerly component during both campaigns. At Thokoza Park higher concentrations of PM_{10} were shown from the north-easterly component during campaign two and from the westerly component during campaign three (Figure 3-25).



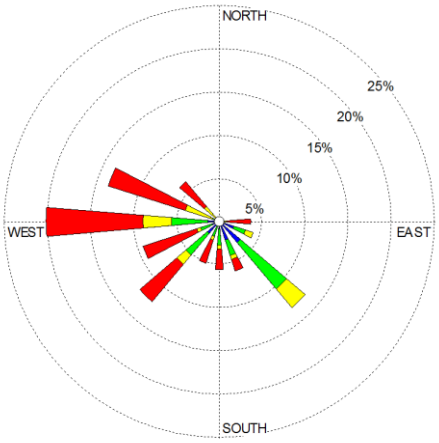
Petrus Molefe: PM_{10} Pollution Rose (8 – 20 OCT 2013)



Thokoza Park: PM_{10} Pollution Rose (22 – 30 OCT 2013)



Petrus Molefe: PM_{10} Pollution Rose (28 MAY – 12JUN 2013)



Thokoza Park: PM_{10} Pollution Rose (14 – 22 JUN 2013)

Concentration (µg/m³):
Blue: 0 - 40
Green: 40 – 80
Yellow: 80 – 120
Red: > / = 120

Figure 3-25: Pollution rose diagrams of PM_{10} concentrations recorded at Petrus Molefe (left) and Thokoza Park (right) for campaigns two (top) and three (bottom).

3.2.1.2. Particulate Matter with an Aerodynamic Diameter equal to or less than 2.5 µm ($PM_{2.5}$)

Ground level $PM_{2.5}$ concentrations were recorded at Petrus Molefe Eco-Park and Thokoza Park during the second and third monitoring campaigns. However, during the third monitoring campaign there was a problem with the filter tape of the BAM 1020 $PM_{2.5}$ Sampler which resulted in no data capture of $PM_{2.5}$ concentrations at Petrus Molefe Eco-Park from the 28 May – 05 June 2014. During the site visit the filter tape was replaced, which resulted in 100% data capture from the 06 June 2014 onwards.

$PM_{2.5}$ concentrations recorded at both the urban parks only exceeded the current $PM_{2.5}$ daily standard of 65µg/m³ during the third monitoring campaign (Figure 3-26 to 3-29). A distinct difference between the two

urban parks is evident when comparing the $PM_{2.5}$ concentrations measured at the parks with the South African daily standard. The $PM_{2.5}$ concentrations measured at Petrus Molefe Eco-Park from 7 – 12 June were in a constant state of exceedance with the South African daily standard of 65µg/m^3 (Figure 3-28). However the concentrations of $PM_{2.5}$ measured at Thokoza Park only exceeded the standard of 65µg/m^3 three times on the 14 June 2014 and 21 – 22 June 2014 (Figure 3-29). Furthermore, the background $PM_{2.5}$ concentrations did not exceed the standard of 65µg/m^3 throughout both campaigns.

The two tailed t-Test analysis confirmed that there was a significant difference in mean $PM_{2.5}$ concentrations between Petrus Molefe Eco-Park and Thokoza Park for both monitoring campaigns ($P < 0.05$) (Table 3-11). Contrasting to what was observed for PM_{10}, the concentrations of $PM_{2.5}$ recorded at Thokoza Park were significantly lower compared to Petrus Molefe Eco-Park during campaign two and three. $PM_{2.5}$ concentrations recorded during campaign two ranged from 18.1 – 61.2 µg/m^3 at Petrus Molefe Eco-Park and 18.5 – 38.4 µg/m^3 at Thokoza Park. $PM_{2.5}$ concentrations recorded during campaign three ranged from 62.9 – 126.4 µg/m^3 at Petrus Molefe Eco-Park and 25.1 – 71.9 µg/m^3 at Thokoza Park (Table 3-10). Similar to that of PM_{10}, the concentrations of $PM_{2.5}$ recorded during the winter season (May – June) over the third monitoring campaign were much higher compared to the second monitoring campaign.

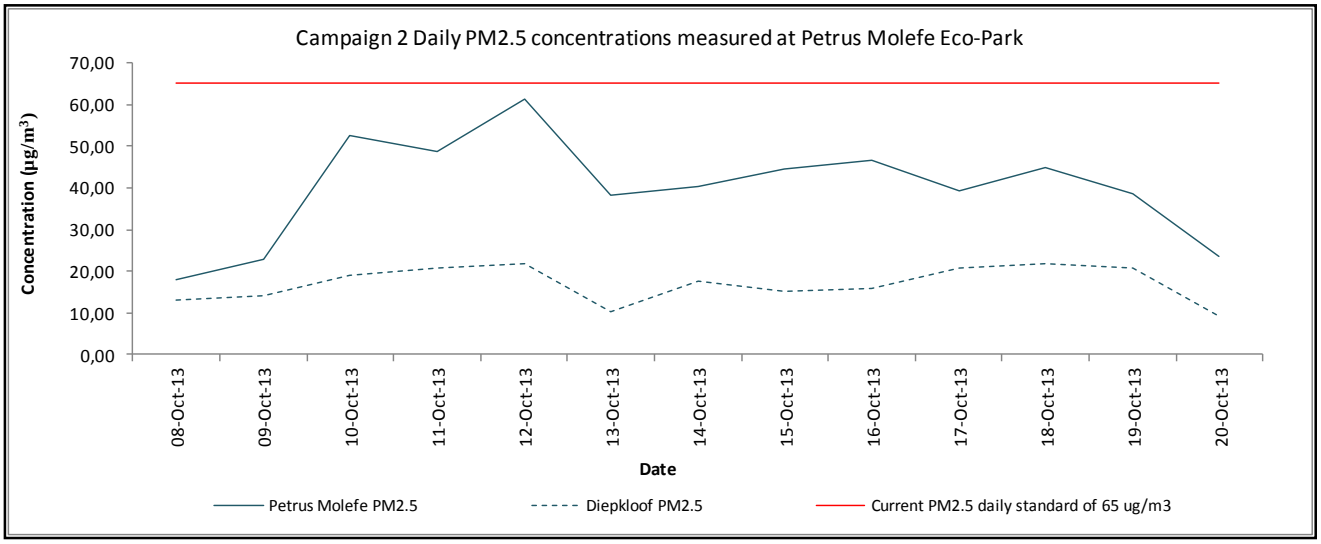


Figure 3-26: Campaign two daily average $PM_{2.5}$ concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

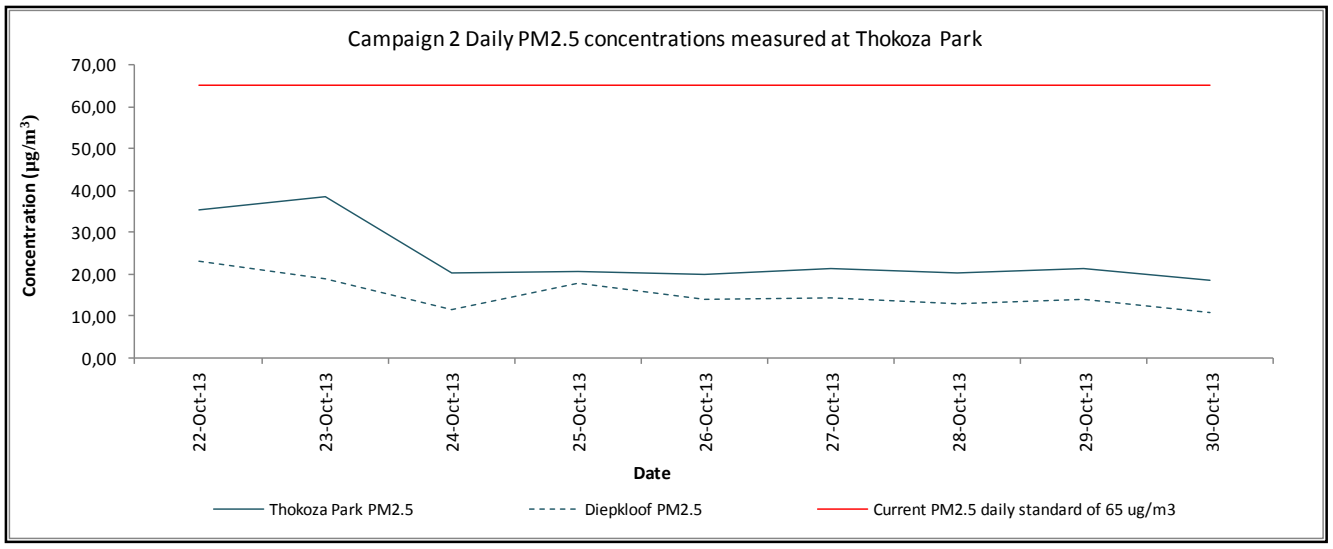


Figure 3-27: Campaign two daily average $PM_{2.5}$ concentrations measured at Thokoza Park (22 - 30 October 2013).

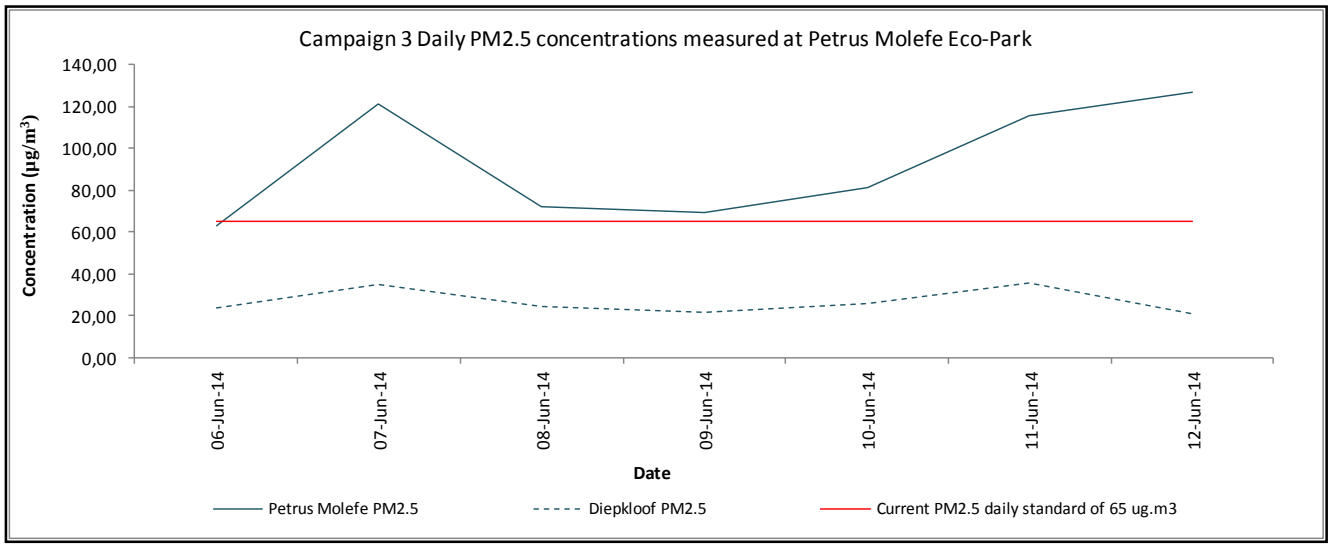


Figure 3-28: Campaign three daily average $PM_{2.5}$ concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

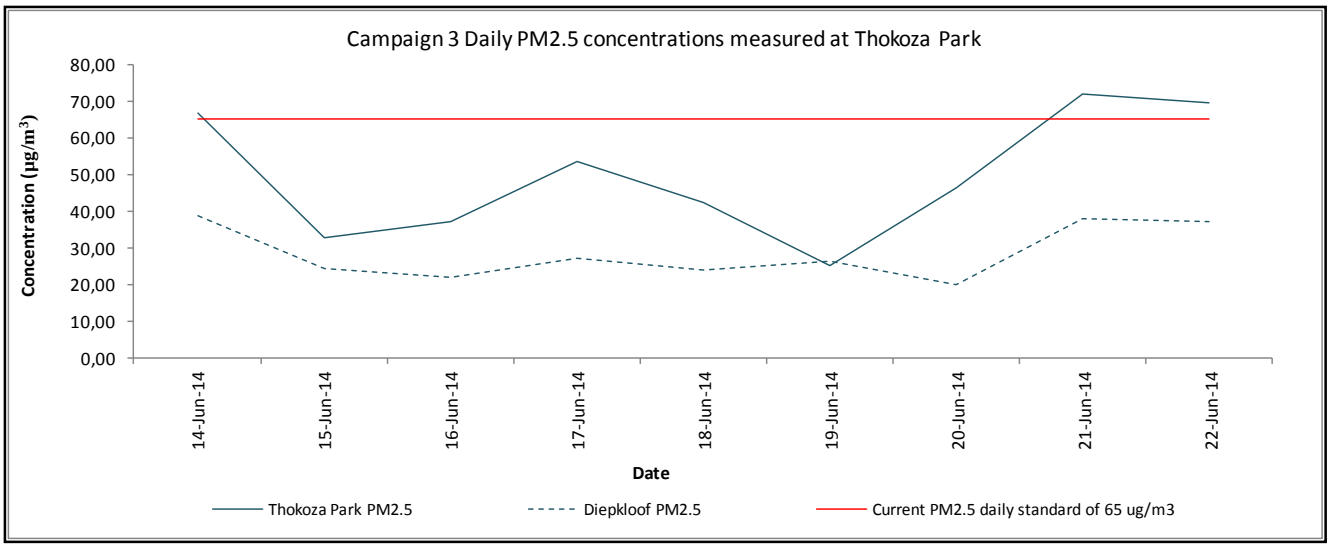


Figure 3-29: Campaign three daily average $PM_{2.5}$ concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-10: Data summary of ambient $PM_{2.5}$ concentrations (µg/m^3) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	39.9	24.0
Median	40.3	20.6
Standard Deviation	12.3	7.3
Minimum	18.1	18.5
Maximum	61.2	38.4
Campaign Three		
Mean	92.6	49.4
Median	81.0	46.3
Standard Deviation	27.2	17.0
Minimum	62.9	25.1
Maximum	126.4	71.9

Table 3-11: Two tailed t-Test analysis with unequal variances of $PM_{2.5}$ concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	20	3.80	P < 0.05	0.001
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	10	3.68	P < 0.05	0.004

Although the $PM_{2.5}$ concentrations recorded at both the urban parks followed a very similar pattern to the urban background site, the two tailed paired t-Test analysis showed that the difference in mean $PM_{2.5}$ concentrations between the urban parks and background site was significantly different (P < 0.05) (Table 3-12).

The $PM_{2.5}$ concentrations recorded at Petrus Molefe Eco-Park were distinctly higher compared to the urban background site during both monitoring campaigns (Figure 3-26 and 3-28). Similarly the $PM_{2.5}$ concentrations recorded at Thokoza Park were also higher compared to the urban background site for both monitoring campaigns (Figure 3-27 and 3-29). However, higher mean differences of 23 µg/m^3 and 66 µg/m^3 in $PM_{2.5}$ concentrations are shown between Petrus Molefe Eco-Park and the urban background site compared to Thokoza Park and the background site (Table 3-12).

Table 3-12: Two tailed paired t-Test analysis of urban park ambient $PM_{2.5}$ concentrations (µg/m^3) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	8.40	2.29×10^{-6}	P < 0.05	23.00	17.04	28.98
Thokoza Park (older trees)	8	5.60	0.0005	P < 0.05	8.81	5.17	12.44
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	6	7.05	0.0004	P < 0.05	66.08	43.13	89.00
Thokoza Park (older trees)	8	5.35	0.0007	P < 0.05	20.83	11.89	29.81

Diurnal variation in $PM_{2.5}$ concentrations is only observed at both urban parks and the urban background site during campaign three (Figure 3-32 and 3-33). In campaign three the $PM_{2.5}$ concentrations at both the urban parks and urban background site increased in the morning (5:00 – 9:00) and evening periods (16:00 – 22:00). The $PM_{2.5}$ diurnal plots show significant increases in the concentrations during the evening period, particularly for the urban parks. During campaign two the $PM_{2.5}$ concentrations recorded at both urban parks remain relatively consistent throughout the day and night with a gradual increase in the concentrations between 06:00 – 16:00 (Figure 3-30 and 3-31).

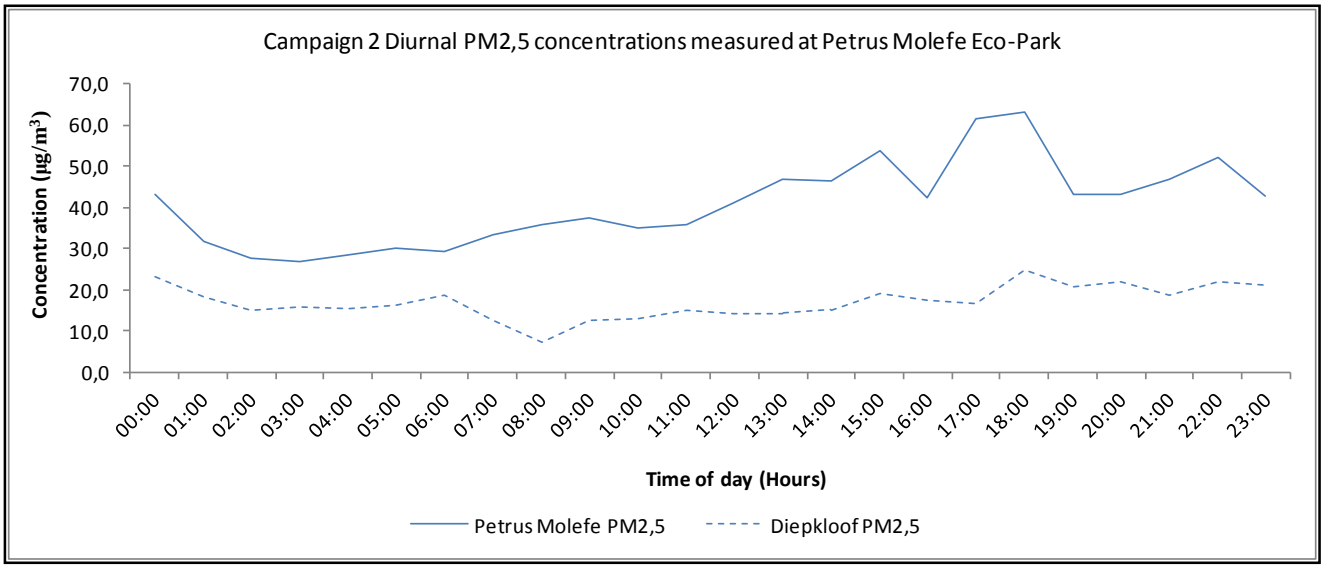


Figure 3-30: Campaign two diurnal $PM_{2.5}$ concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

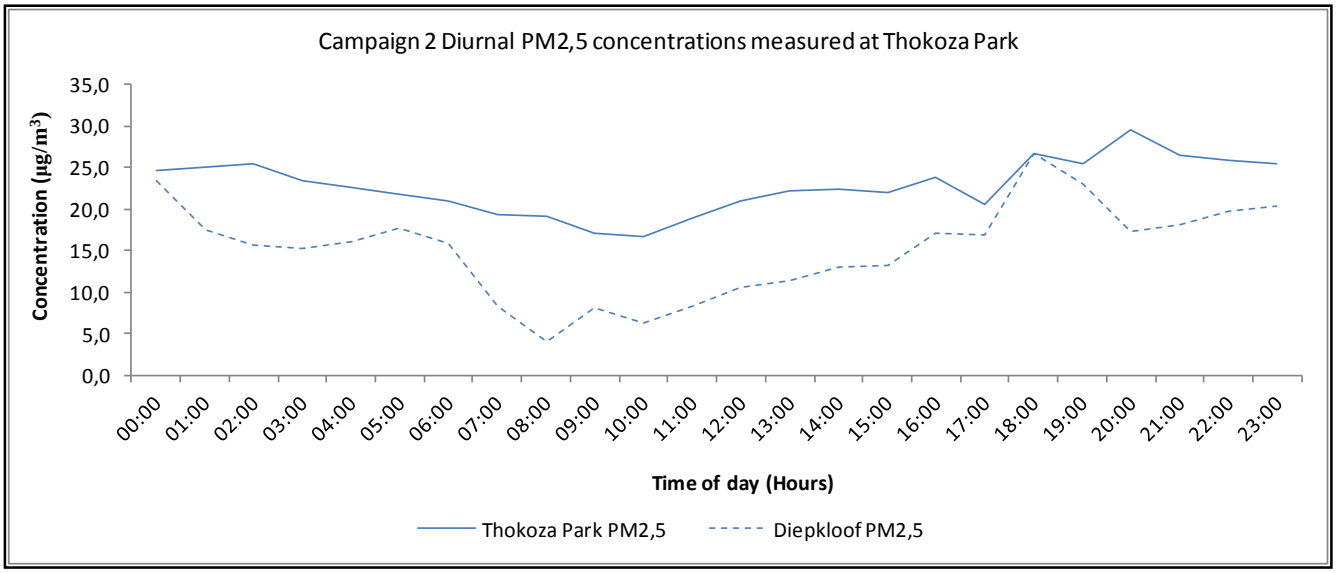


Figure 3-31: Campaign two diurnal $PM_{2.5}$ concentrations measured at Thokoza Park (22 - 30 October 2013).

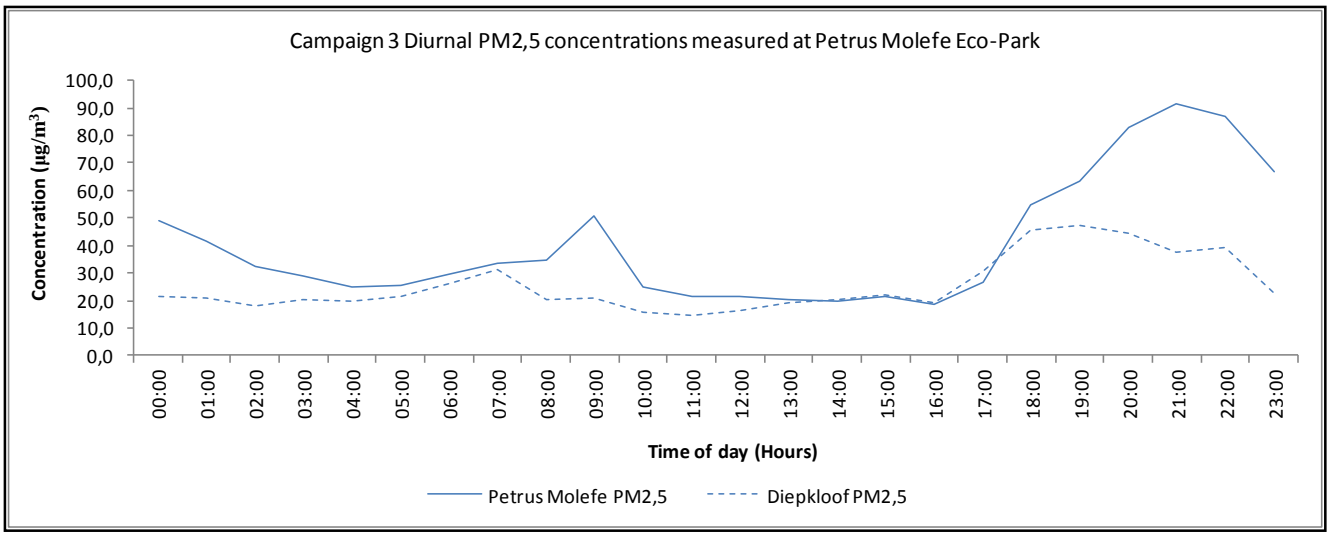


Figure 3-32: Campaign three diurnal $PM_{2.5}$ concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

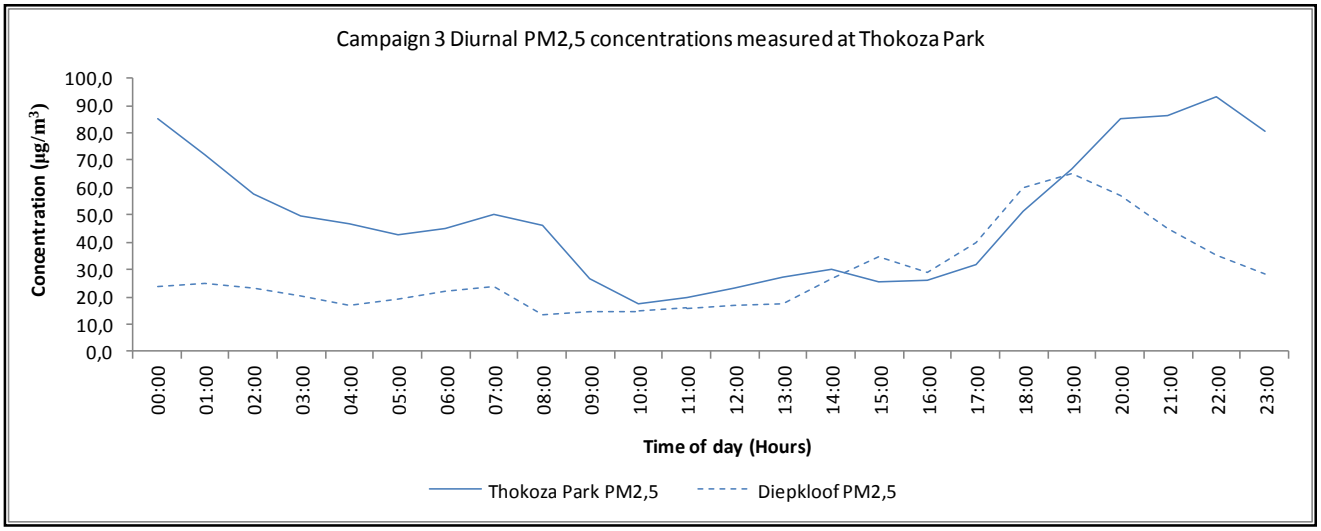
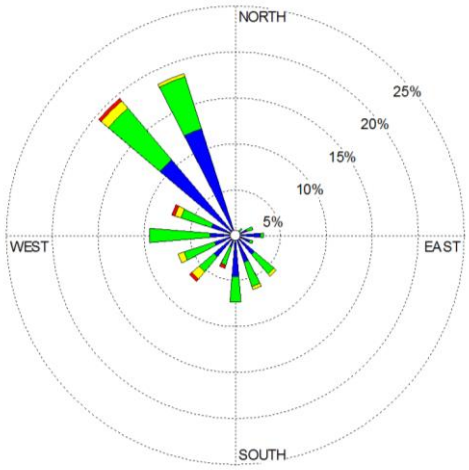


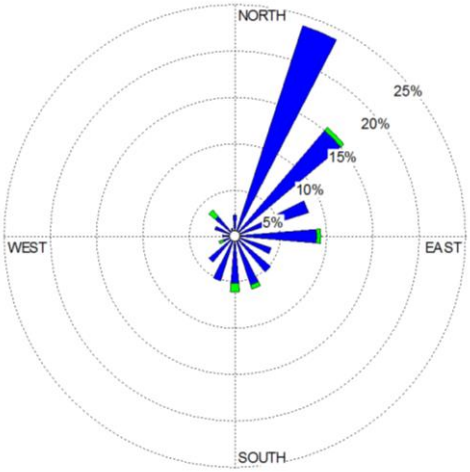
Figure 3-33: Campaign three diurnal $PM_{2.5}$ concentrations measured at Thokoza Park (14 - 22 June 2014).

A pollution rose plot for $PM_{2.5}$ is given in Figures 3-34 for Petrus Molefe Park and Thokoza Park for the second and third monitoring campaign. The pollution rose plot shows that higher concentrations of $PM_{2.5}$ were associated with Petrus Molefe Eco-Park compared to Thokoza Park. The pollution rose also shows that higher concentrations of $PM_{2.5}$ were recorded during the winter season (campaign three) compared to the spring season (campaign two).

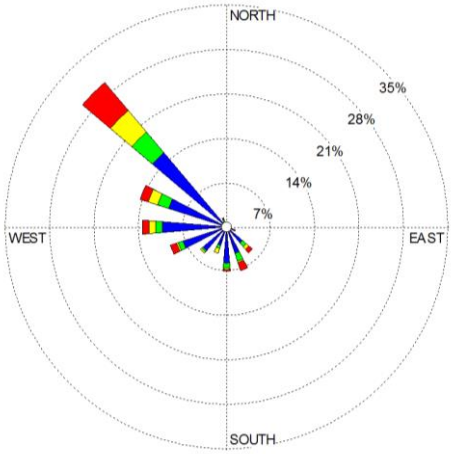
At Petrus Molefe Eco-Park higher $PM_{2.5}$ concentrations were mostly shown from the north-westerly direction during both monitoring campaigns. Higher concentrations of $PM_{2.5}$ were not as clearly defined at Thokoza Park during campaign two, however, higher concentrations of $PM_{2.5}$ are observed from a westerly direction during campaign three (Figure 3-34).



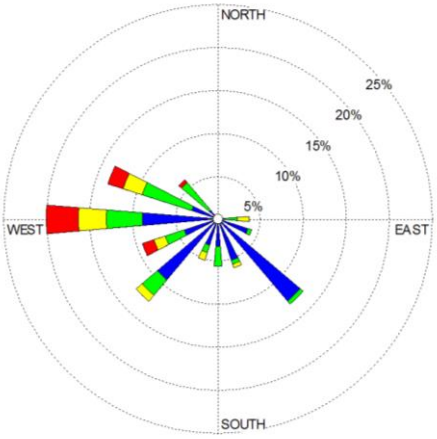
Petrus Molefe: $PM_{2.5}$ Pollution Rose (8 – 20 OCT 2013)



Thokoza Park: $PM_{2.5}$ Pollution Rose (22 – 30 OCT 2013)



Petrus Molefe: $PM_{2.5}$ Pollution Rose (28 MAY – 12 JUNE 2013)



Thokoza Park: $PM_{2.5}$ Pollution Rose (14 – 22 JUNE 2013)

Concentration ($\mu g/m^3$):
Blue: 0 - 40
Green: 40 – 80
Yellow: 80 – 120
Red: > / = 120

Figure 3-34: Pollution rose diagrams of $PM_{2.5}$ concentrations recorded at Petrus Molefe (left) and Thokoza Park (right) for campaigns two (top) and three (bottom).

3.2.2. Nitrogen Oxides (NO + NO_2)

Ground level concentrations of NOx were measured at Petrus Molefe Eco-Park and Thokoza Park during the second and third monitoring campaigns (Figures 3-35 to 3-38).

Hourly concentrations of NOx measured at both the urban parks were below the South African National air quality standard of 106 ppb (expressed as NO_2) during both campaigns. Maximum hourly concentrations of 92.00 ppb and 65.00 ppb were recorded at Petrus Molefe Eco-Park during campaign two and three respectively, and slightly lower maximum hourly concentrations of 86.00 ppb and 56.00 ppb were recorded at Thokoza Park during campaign two and three respectively. Much higher concentrations were observed at both the urban parks over the winter season during the third monitoring campaign compared to the second monitoring campaign.

Daily average concentrations of NOx measured at Petrus Molefe-Eco Park and Thokoza Park over the second monitoring campaign were similar with mean concentrations of 13.1 and 14.6 ppb recorded respectively (Table 3-13). The daily average concentrations of NOx recorded during campaign three at Thokoza Park appeared to be slightly lower compared to Petrus Molefe Eco-Park with mean concentrations of 68.00 ppb and 83.7 ppb recorded respectively (Table 3-13). However the two tailed t-Test analysis with unequal variances indicated that there was no significant difference in the mean NOx concentrations between the two urban parks for both monitoring campaigns (P > 0.05) (Table 3-14).

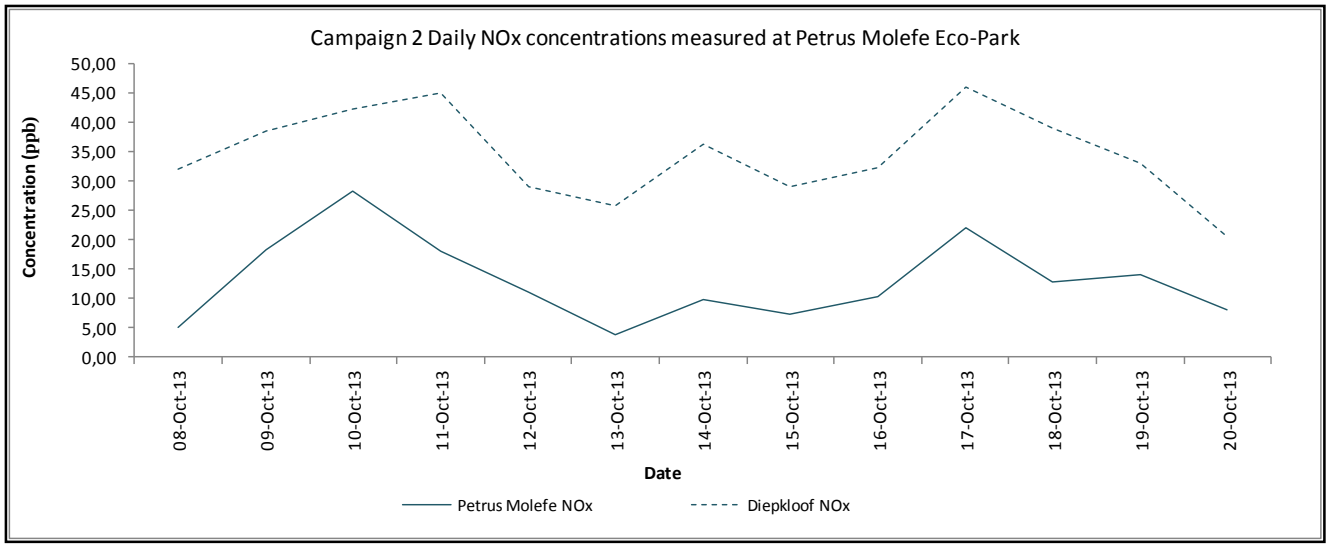


Figure 3-35: Campaign two daily average NOx concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

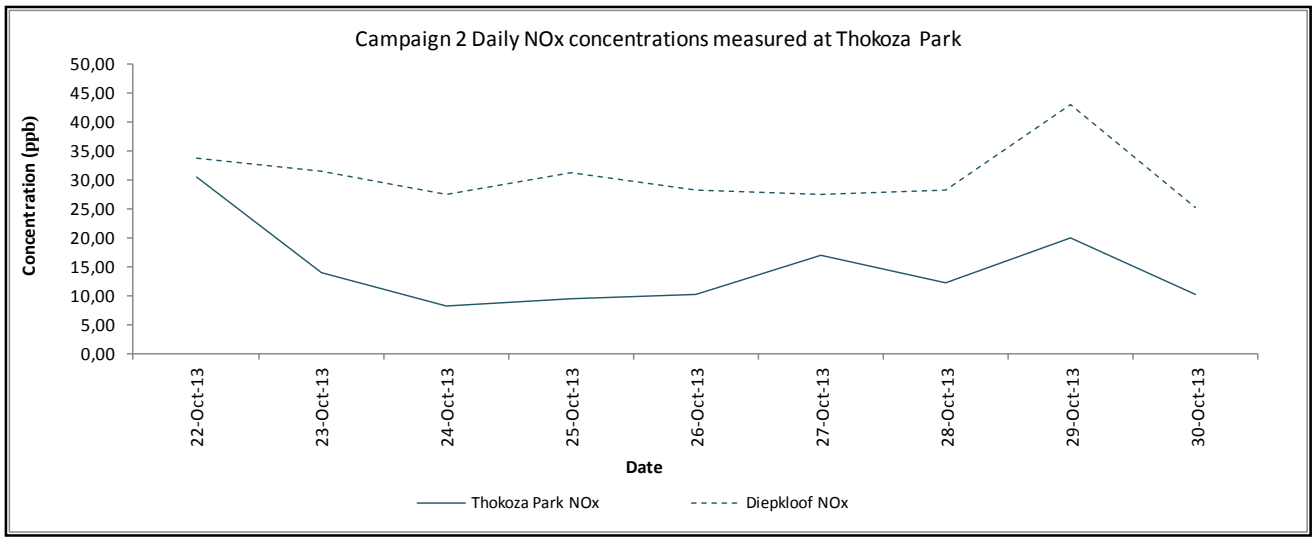


Figure 3-36: Campaign two daily average NOx concentrations measured at Thokoza Park (22 - 30 October 2013).

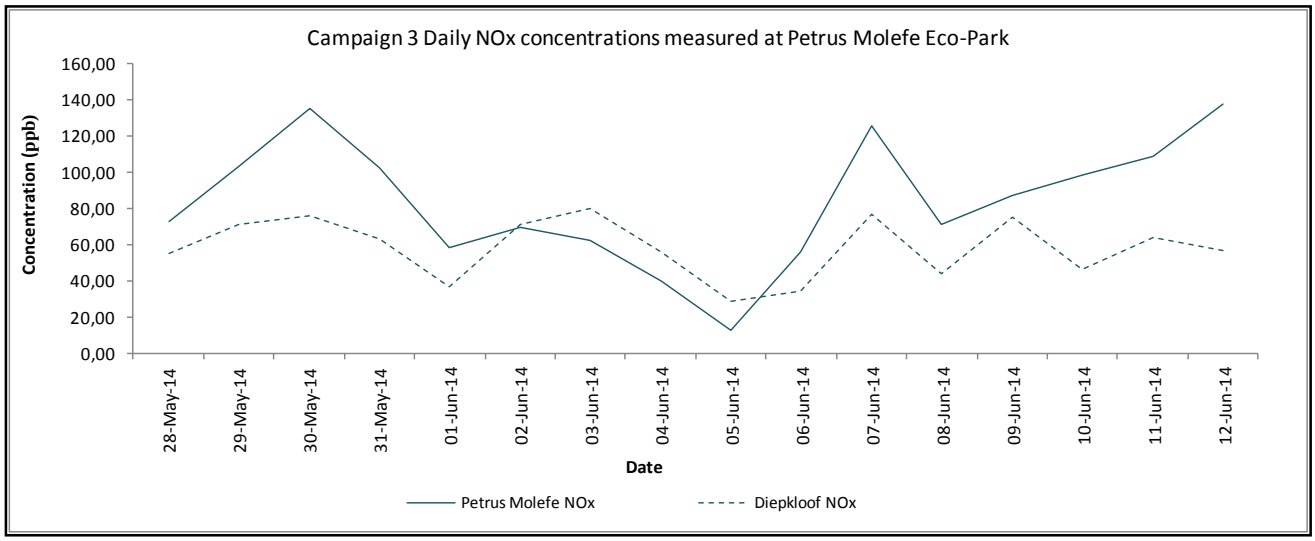


Figure 3-37: Campaign three daily average NOx concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

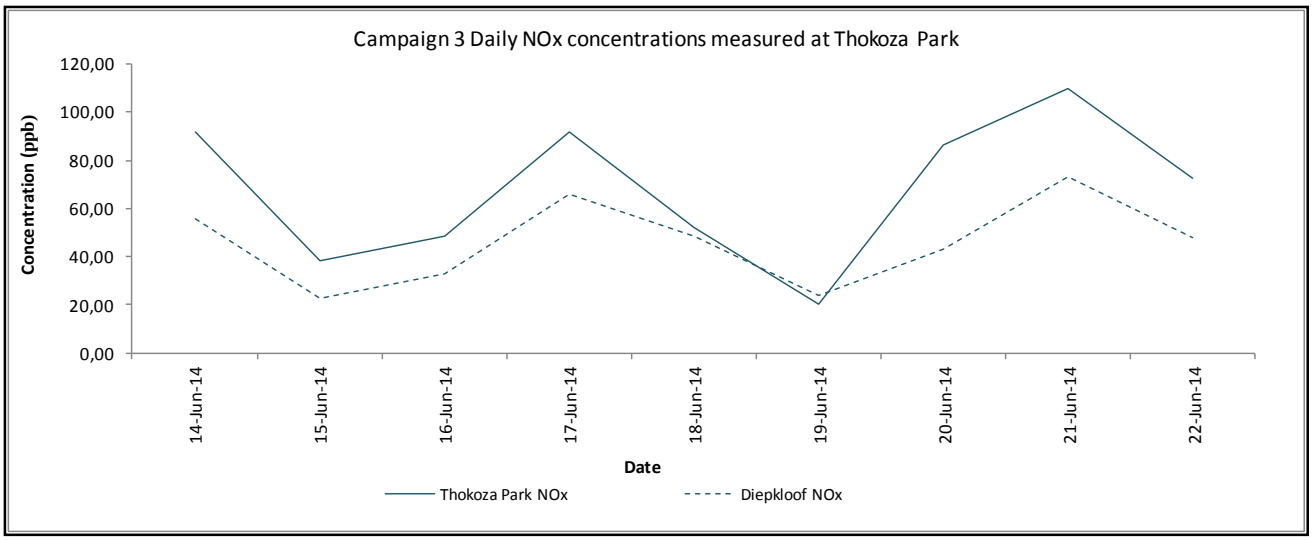


Figure 3-38: Campaign three daily average NOx concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-13: Data summary of ambient NOx concentrations (ppb) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	13.1	14.6
Median	11.2	12.2
Standard Deviation	7.0	7.1
Minimum	3.9	8.1
Maximum	28.2	30.5
Campaign Three		
Mean	83.7	68.0
Median	79.8	72.5
Standard Deviation	34.9	29.6
Minimum	12.5	20.4
Maximum	137.2	109.8

Table 3-14: Two tailed t-Test analysis with unequal variances of NOx concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	17	-0.51	P > 0.05	0.613
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	19	1.19	P > 0.05	0.249

Although there was no significant difference in the NOx concentrations between the two urban parks, the two tailed paired t-Test analysis showed that there was a significant difference in NOx concentrations between the urban background site and both the urban parks particularly during campaign two (P<0.05) (Table 3-15). During campaign two the NOx concentrations measured at both the urban parks were significantly lower compared to the urban background site and a higher mean difference of -21.48 ppb was calculated for Petrus Molefe Eco-Park compared to Thokoza Park (Figure 3-35 and 3-36) (Table 3-15). Unlike campaign two, the NOx concentrations measured at both the urban parks during campaign three were generally higher compared to the urban background site (Figure 3-37 and 3-38). Similar mean differences of 25.35 ppb and 21.99 ppb were calculated for Petrus Molefe Eco-Park and Thokoza Park respectively (Table 3-15). Even though a statistically significant difference in NOx concentrations between the two urban parks was observed, the concentrations of NOx measured at both the urban parks appeared to have followed a similar pattern to that of the urban background site (Figure 3-35 to 3-38).

Table 3-15: Two tailed paired t-Test analysis of urban park ambient NOx concentrations (ppb) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	-16.19	1.6×10^{-9}	P < 0.05	-21.48	-24.36	-18.59
Thokoza Park (older trees)	8	-7.94	0.00005	P < 0.05	-16.02	-20.68	-11.37
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	3.55	0.003	P < 0.05	25.35	10.11	40.59
Thokoza Park (older trees)	8	4.23	0.003	P < 0.05	21.99	10.00	34.00

Diurnal variation is observed in the concentrations of NOx measured at both the urban parks and the urban background site during campaign two and three (Figures 3-39 to 3-42). At Petrus Molefe Eco-Park the concentration of NOx gradually increased in the morning (4:00 – 8:00) and in the evening (16:00 – 22:00) during both campaigns (Figure 3-39 and 3-41). A similar pattern is observed at Thokoza Park for both campaigns where NOx concentrations gradually increased in the morning (4:00 – 8:00) and evening (16:00 – 21:00) periods (Figure 3-40 and 3-42). However, during campaign two a quicker rise in the NOx concentrations is observed at Thokoza Park in the evening period around 16:00 which is not observed at Petrus Molefe Eco-Park.

The diurnal variation plots for NOx also display that there was greater diurnal variation in the urban background NOx concentrations compared to the urban parks especially during the morning period. A rapid increase in the urban background NOx concentrations is observed between 3:00 – 7:00 in the morning throughout both campaigns. A more distinct rise in the urban background NOx concentrations, although not as distinct as the morning period, is observed again in the evening between 15:00 – 19:00 (Figure3-39 to 3-42).

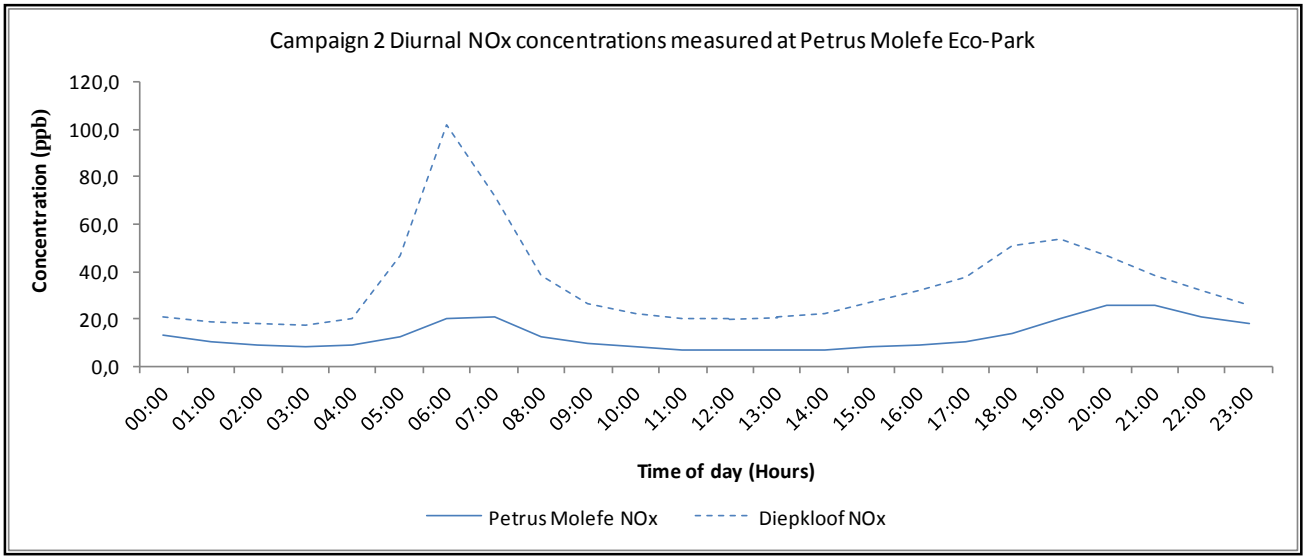


Figure 3-39: Campaign two diurnal NOx concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

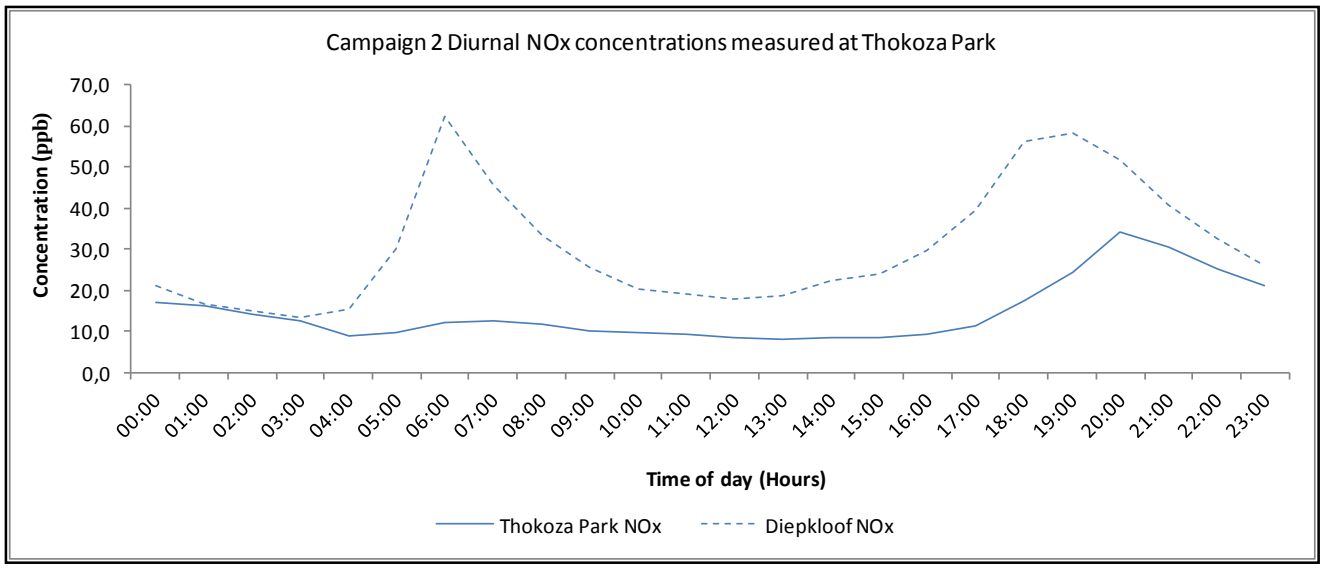


Figure 3-40: Campaign two diurnal NOx concentrations measured at Thokoza Park (22 - 30 October 2013).

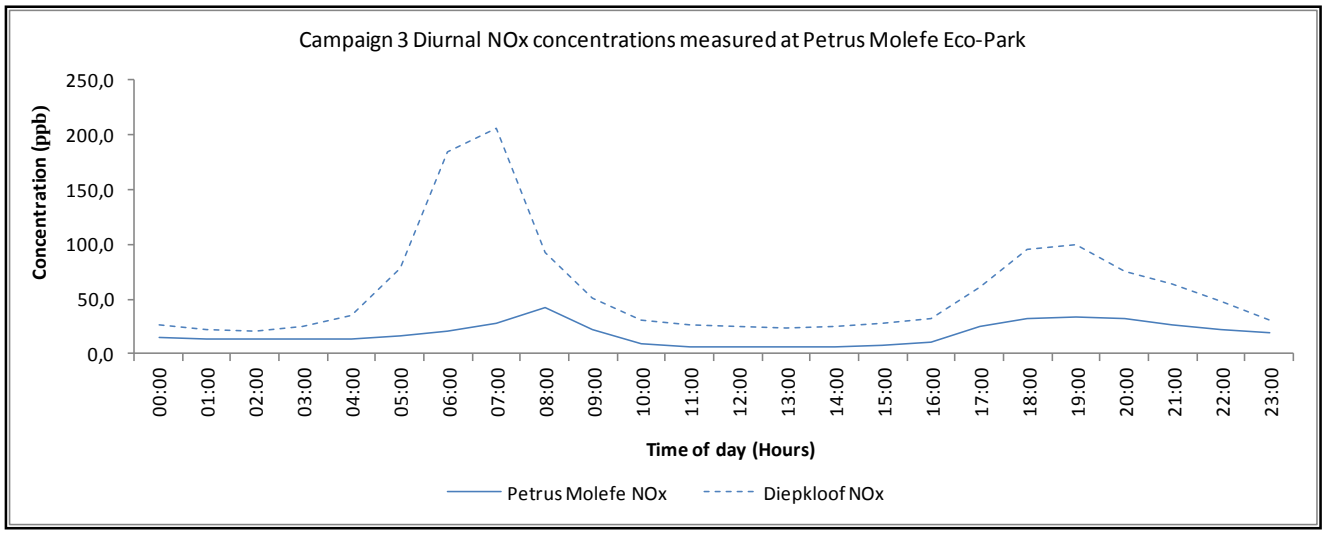


Figure 3-41: Campaign three diurnal NOx concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

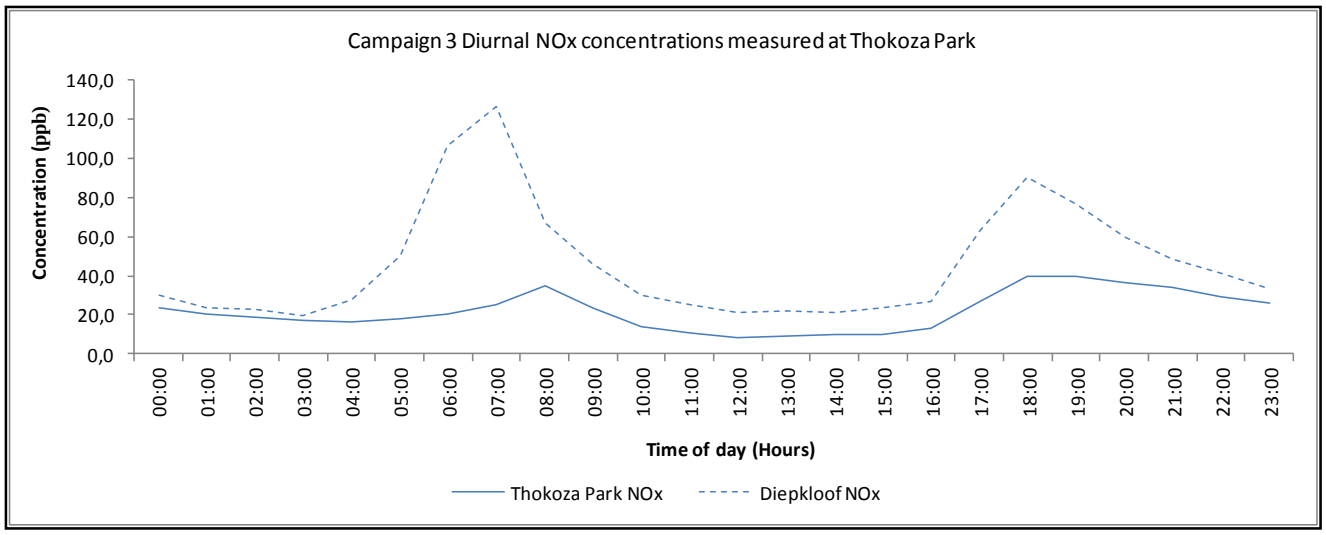


Figure 3-42: Campaign three diurnal NOx concentrations measured at Thokoza Park (14 - 22 June 2014).

3.2.3. Ozone

Ground level O_3 concentrations were measured at Petrus Molefe Eco-Park and Thokoza Park during the second and third monitoring campaigns (Figure 3-43 to 3-50). Eight-hour average concentrations of O_3 measured at both the urban parks were below the South African National air quality standard of 61 ppb throughout both campaigns (Figure 3-43 to 3-46). Higher concentrations of O_3 were observed during the second monitoring campaign over the spring season however the eight-hour average O_3 concentrations did not exceed 50 ppb (Figure 3-43 and Figure 3-44). In campaign three the eight-hour average O_3 concentrations did not exceed 40 ppb (Figure 3-45 and Figure 3-46). Eight hour average O_3 concentrations measured at Diepkloof monitoring station exceeded the standard of 61 ppb during both monitoring campaigns.

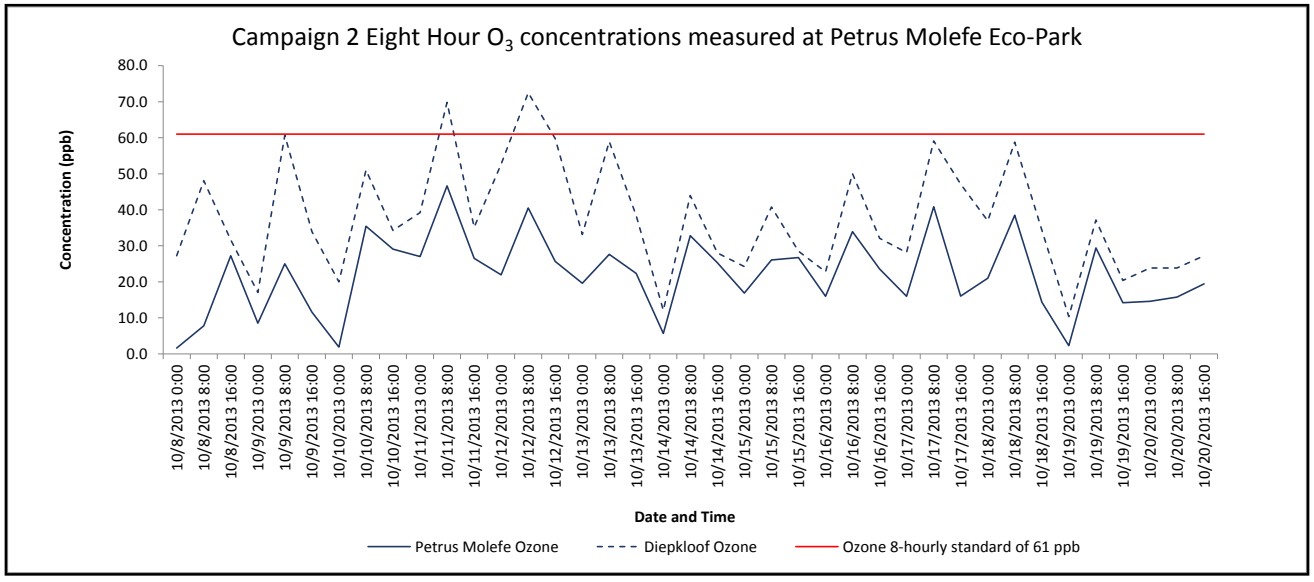


Figure 3-43: Campaign two 8-hour average O_3 concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

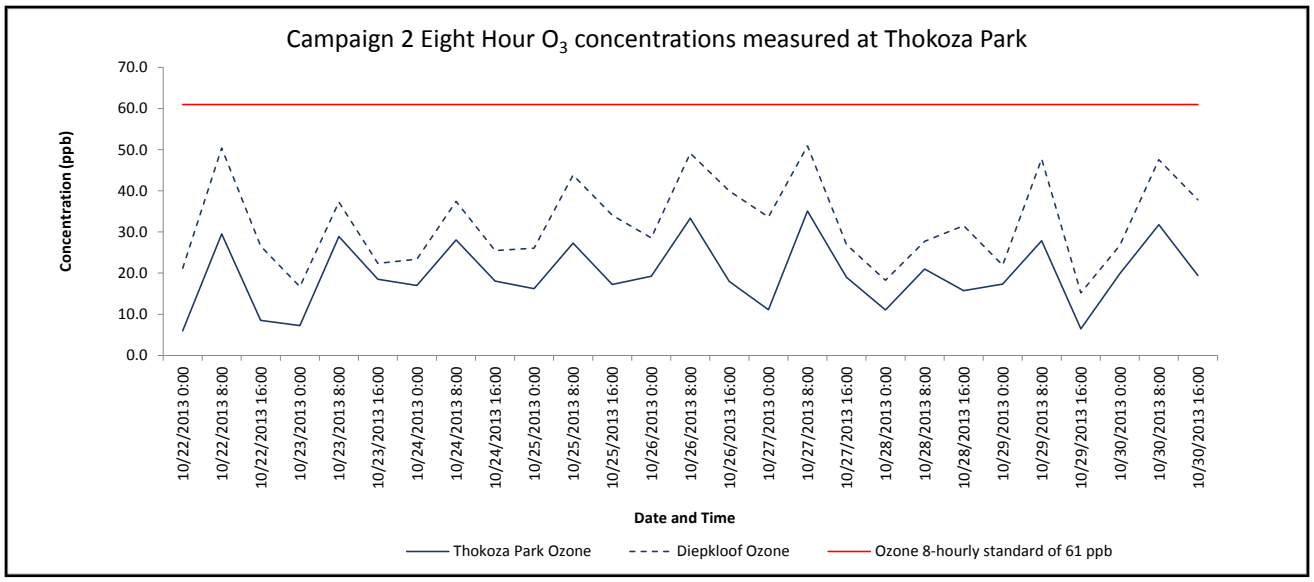


Figure 3-44: Campaign two 8-hour average O_3 concentrations measured at Thokoza Park (22 - 30 October 2013).

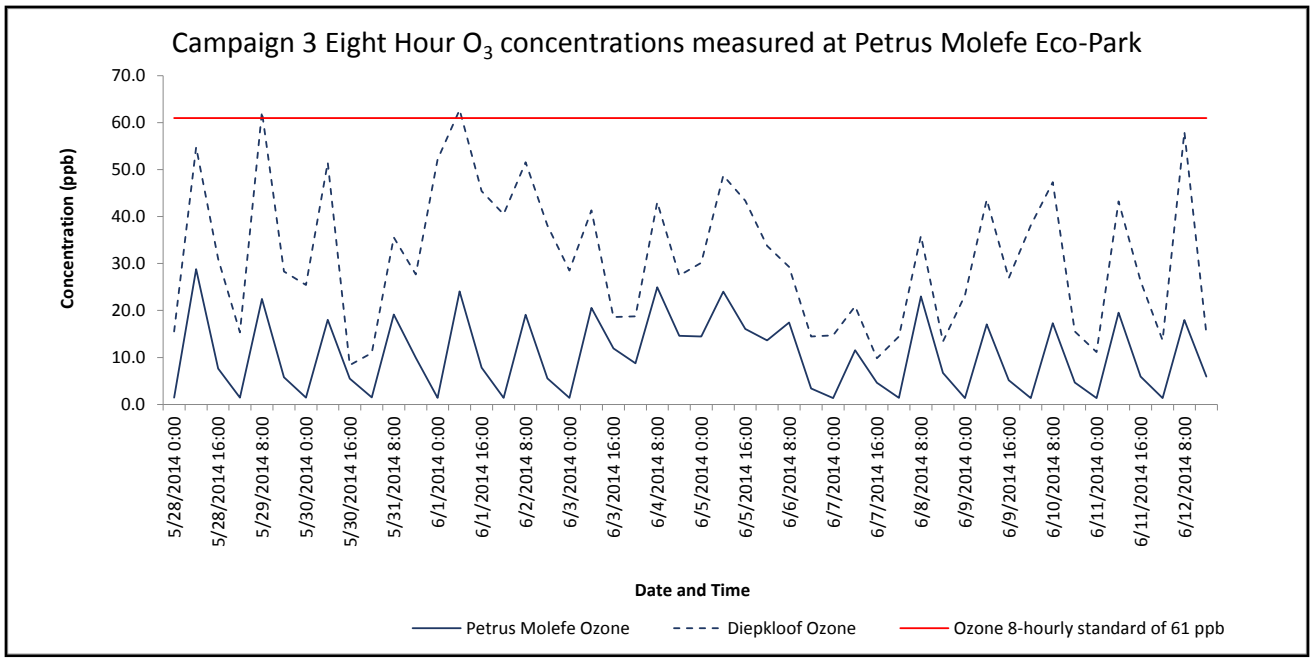


Figure 3-45: Campaign three 8-hour average O_3 concentrations measured at Petrus Molefe Eco-Park (28 May – 12 June 2014).

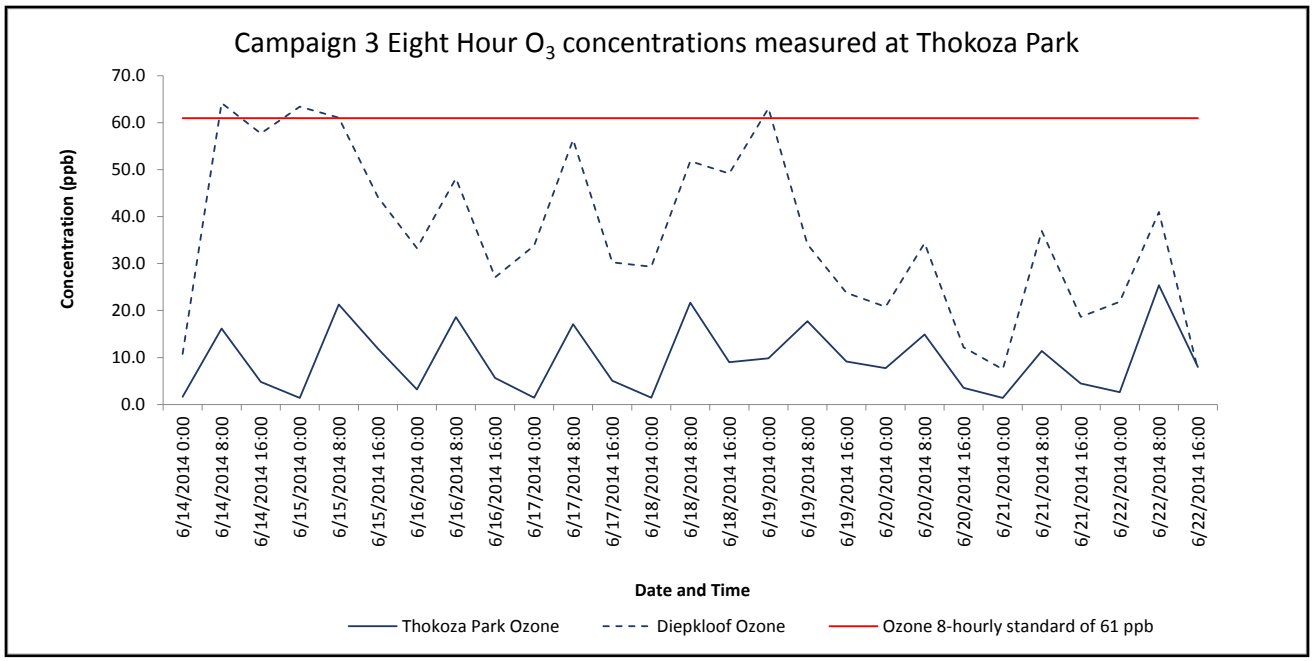


Figure 3-46: Campaign three 8-hour average O_3 concentrations measured at Thokoza Park (14 - 22 June 2014).

Daily average O3concentrations were slightly higher at Petrus Molefe Eco-Park compared to Thokoza Park during both monitoring campaigns. In campaign two the mean O_3 concentration recorded at Petrus Molefe Eco-Park was 21.9 ppb (12.2 – 33.5 ppb) and 19.5 ppb (13.9 – 23.9 ppb) at Thokoza Park (Figure 3-47 and 3-48) (Table 3-16). In campaign three the mean O_3 concentration recorded at Petrus Molefe Eco-Park was 18.1 ppb (5.8 – 18.1 ppb) and 12.2 ppb (5.7 – 12.2 ppb) at Thokoza Park (Figure 3-49 and 3-50) (Table 3-16). However the t-Test analysis with unequal variances suggested that there was no statistically significant difference in O_3 concentrations between the two urban park types ($P > 0.05$) (Table 3-17).

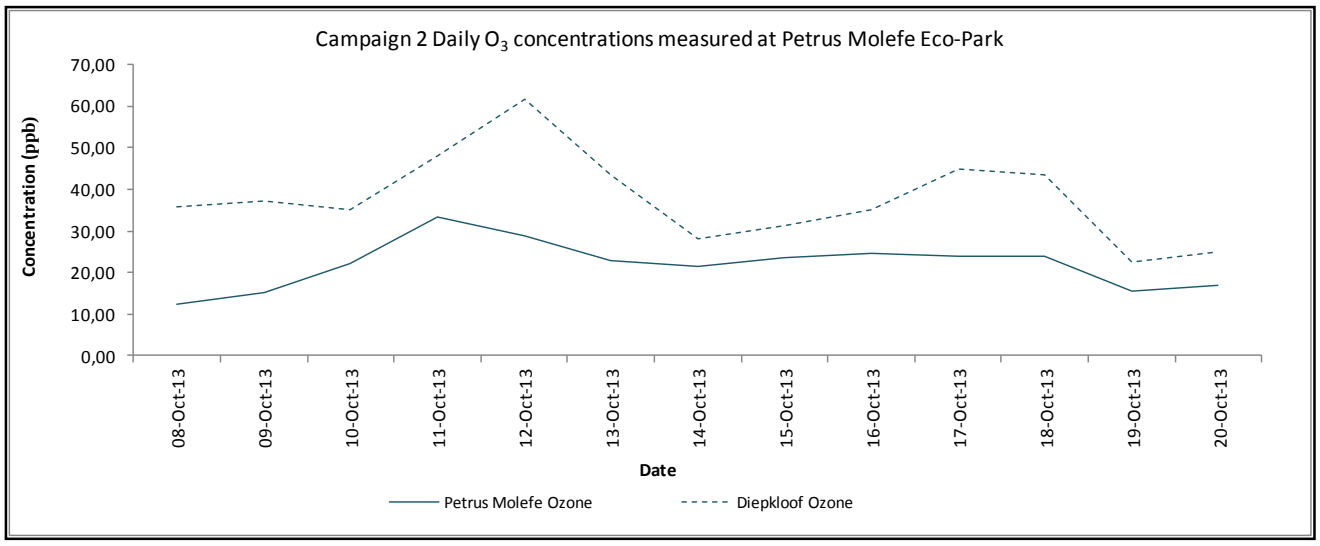


Figure 3-47: Campaign two daily average O_3 concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

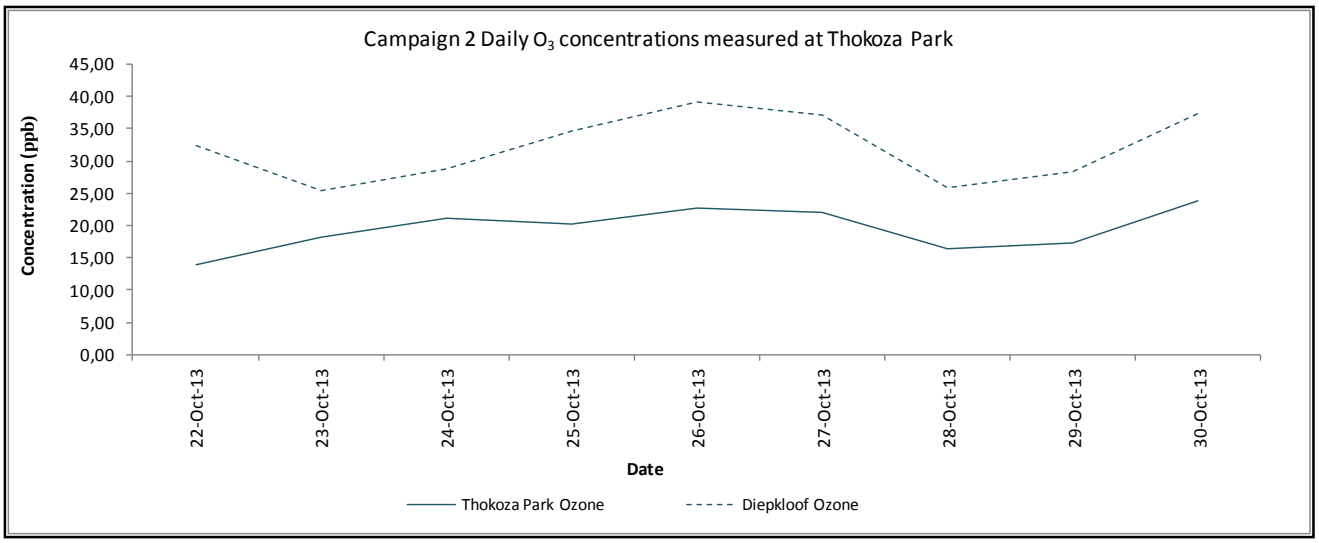


Figure 3-48: Campaign two daily average O_3 concentrations measured at Thokoza Park (22 - 30 October 2013).

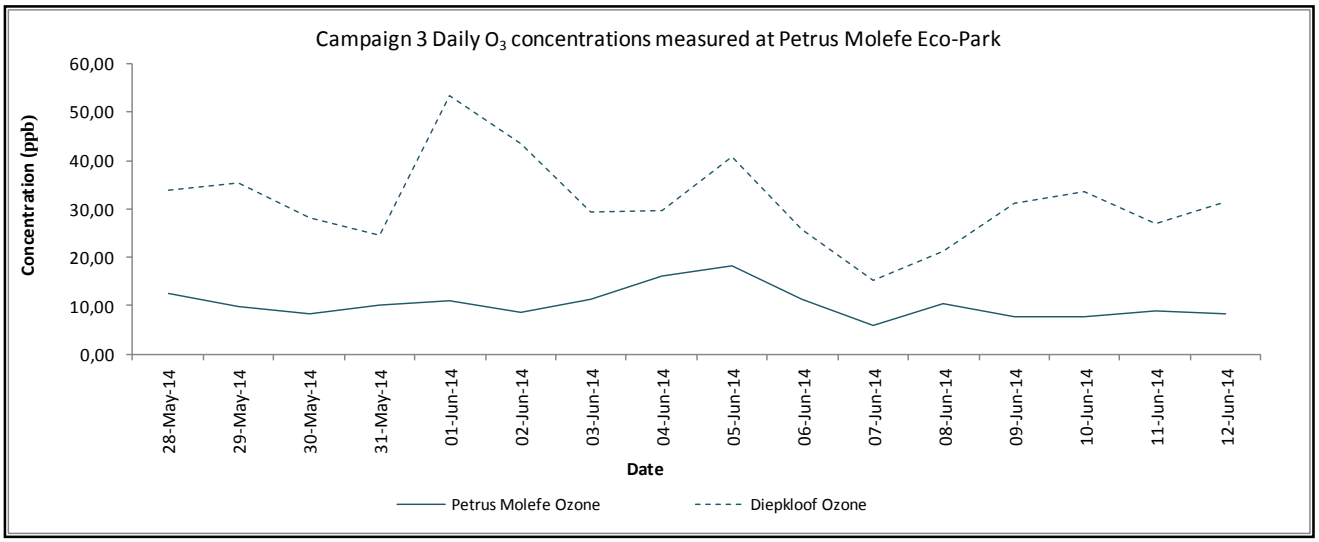


Figure 3-49: Campaign three daily average O_3 concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

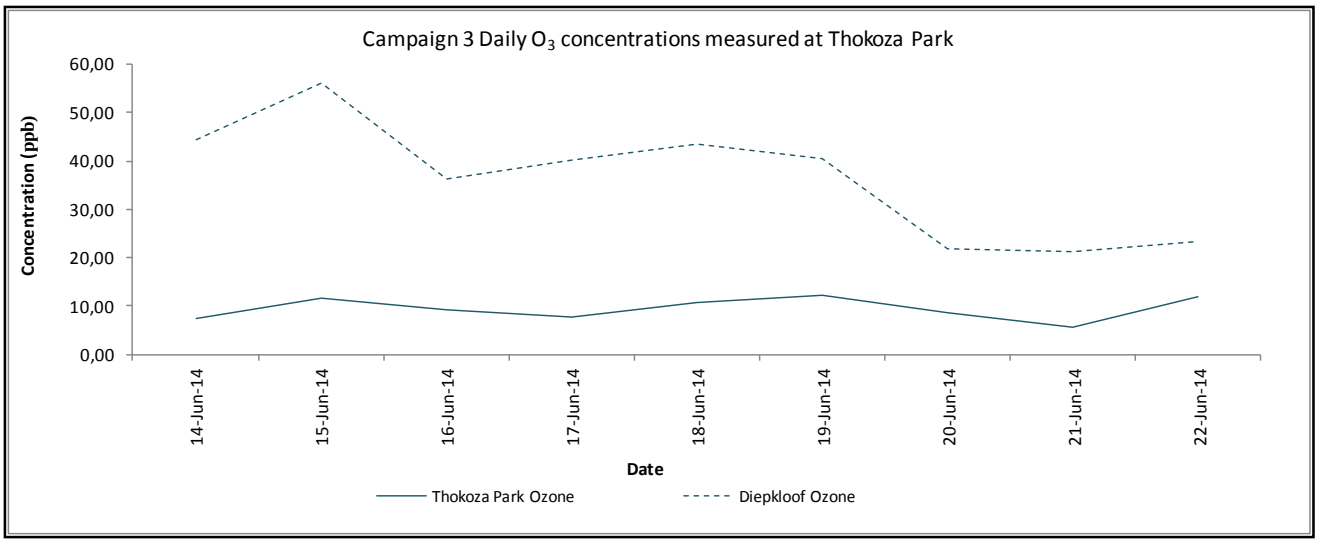


Figure 3-50: Campaign three daily average O_3 concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-16: Data summary of ambient O_3 concentrations (ppb) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (8 - 30 October 2013) and campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	21.9	19.5
Median	22.8	20.2
Standard Deviation	5.8	3.3
Minimum	12.2	13.9
Maximum	33.5	23.9
Campaign Three		
Mean	10.4	9.5
Median	10.0	9.1
Standard Deviation	3.1	2.3
Minimum	5.8	5.7
Maximum	18.1	12.2

Table 3-17: Two tailed t-Test analysis with unequal variances of O_3 concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	19	1.22	P > 0.05	0.238
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	21	0.86	P > 0.05	0.399

A significant difference in mean O_3 concentrations is observed between the two urban parks and the urban background site ($P < 0.05$) (Table 3-18). The calculated mean difference in O_3 concentrations as well as the O_3 concentration plots show that significantly higher concentrations of O_3 were recorded at the urban background site compared to the two urban parks (Figure 3-43 to 3-46) (Table 3-18). As seen with NOx concentrations, the daily average concentrations of O_3 also followed a similar pattern to that of the urban background site.

Table 3-18: Two tailed paired t-Test analysis of urban park ambient O_3 concentrations (ppb) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Two						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	12	-7.06	0.00001	P < 0.05	-15.89	-20.79	-10.99
Thokoza Park (older trees)	8	-9.53	0.00001	P < 0.05	-12.61	-15.67	-9.56
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	-9.85	6.1×10^{-8}	P < 0.05	-21.07	-25.64	-16.52
Thokoza Park (older trees)	8	-7.08	0.0001	P < 0.05	-26.84	-35.59	-18.10

Similar diurnal variation is observed in O_3 concentrations recorded at the two urban parks and the urban background site for both monitoring campaigns (Figure 3-51 to 3-54). The concentration of O_3 gradually increased in the morning from approximately 06:00 onwards, reaching a peak around midday/early afternoon between 12:00 – 14:00 and then decreased rapidly from around 17:00 onwards. However, a significant

increase in the O_3 concentration was observed at the Diepkloof monitoring station during the night time period, but not at the two urban parks (Figure 3-51 to 3-54).

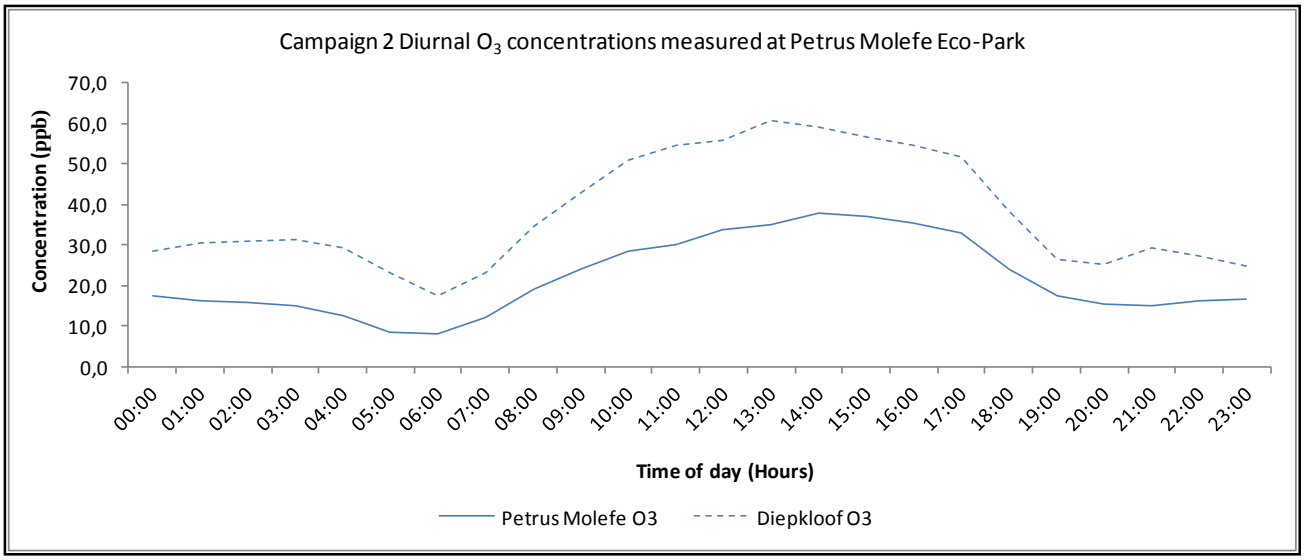


Figure 3-51: Campaign two diurnal O_3 concentrations measured at Petrus Molefe Eco-Park (8 - 20 October 2013).

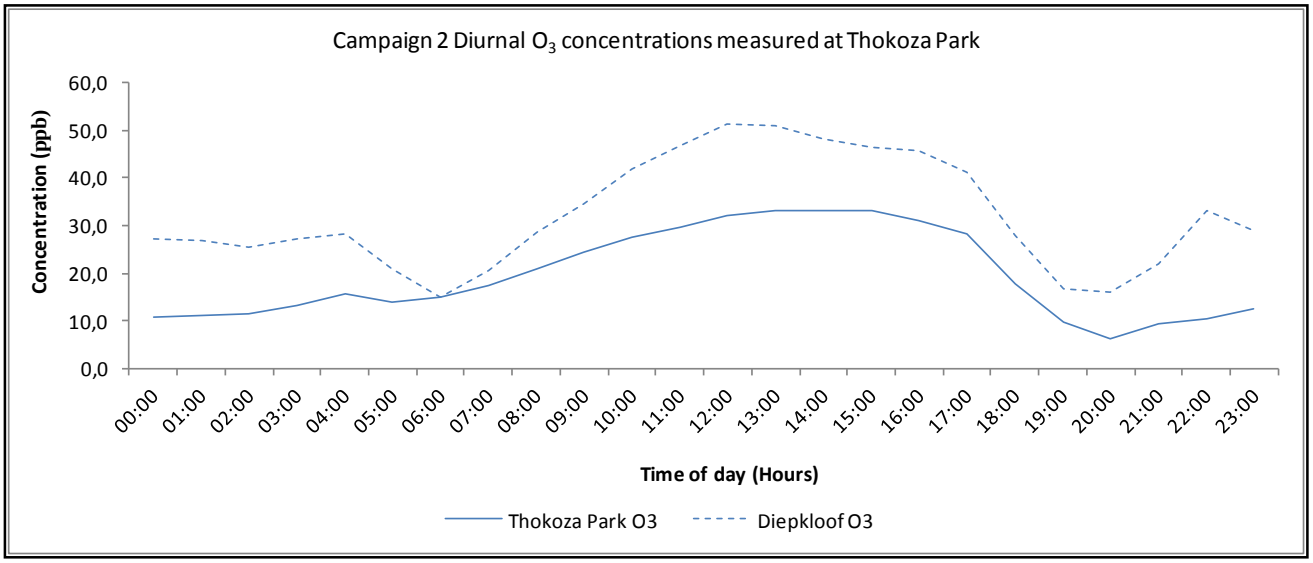


Figure 3-52: Campaign two diurnal O_3 concentrations measured at Thokoza Park (22 - 30 October 2013).

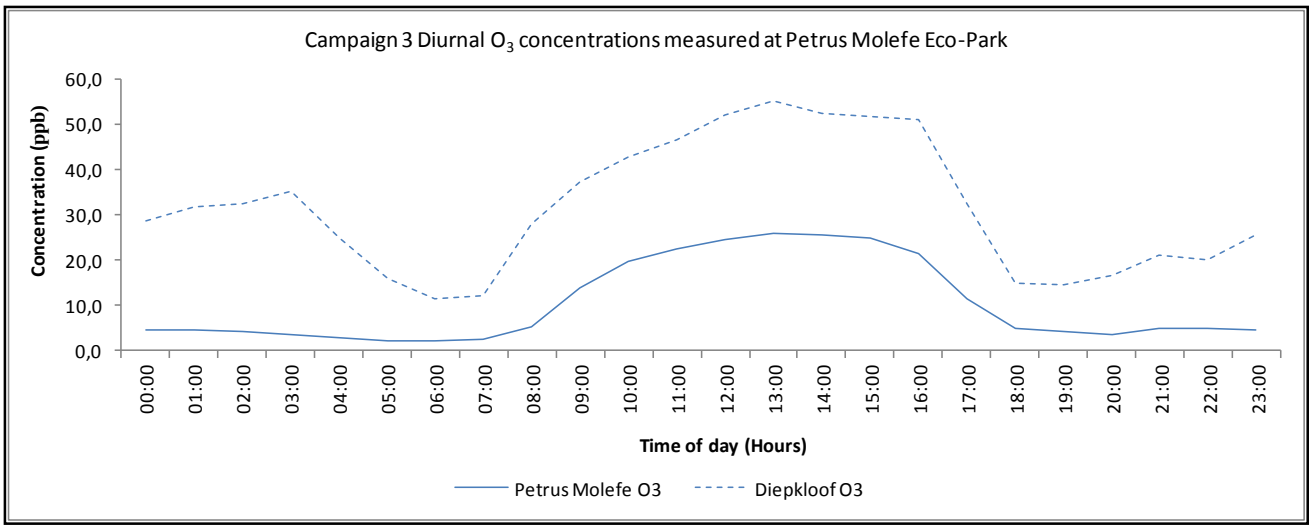


Figure 3-53: Campaign three diurnal O_3 concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

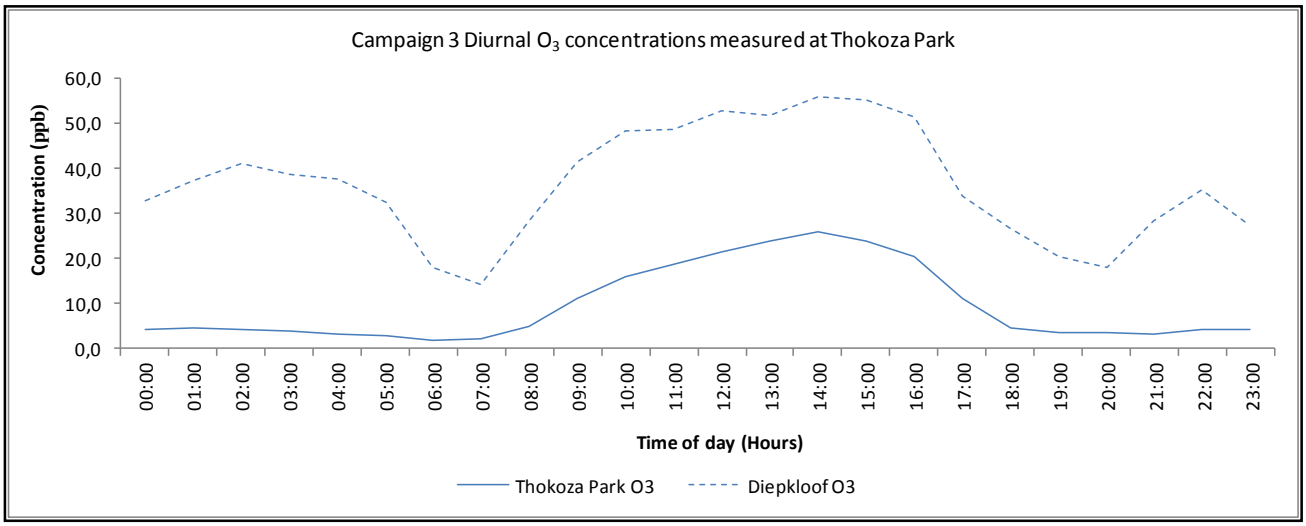


Figure 3-54: Campaign three diurnal O_3 concentrations measured at Thokoza Park (14 - 22 June 2014).

3.2.4. Carbon Monoxide

Due to instrumental failure the ambient concentrations of CO were only measured at Petrus Molefe Eco-Park and Thokoza Park during the third monitoring campaign. In campaign three the daily average concentrations of CO were very low at both the urban parks and ranged from 0.2 – 2.4 ppb at Petrus Molefe Eco-Park and 0.4 – 2.2 ppb at Thokoza Park (Figure 3-55 and 3-56). Both hourly and eight-hourly average concentrations of CO were well within the South African National air quality standard of 26 000 ppb and 8 700 ppb respectively and did not exceed 10 ppb throughout the campaign.

Very similar CO concentrations were observed at both urban park types with mean concentrations of 1.6 ppb and 1.5 ppb recorded for Petrus Molefe Eco-Park and Thokoza Park respectively (Table 3-19). The two tailed

t-Test analysis with unequal variances further showed that there was no significant difference in mean CO concentrations between the two urban park types (P > 0.05) (Table 3-20).

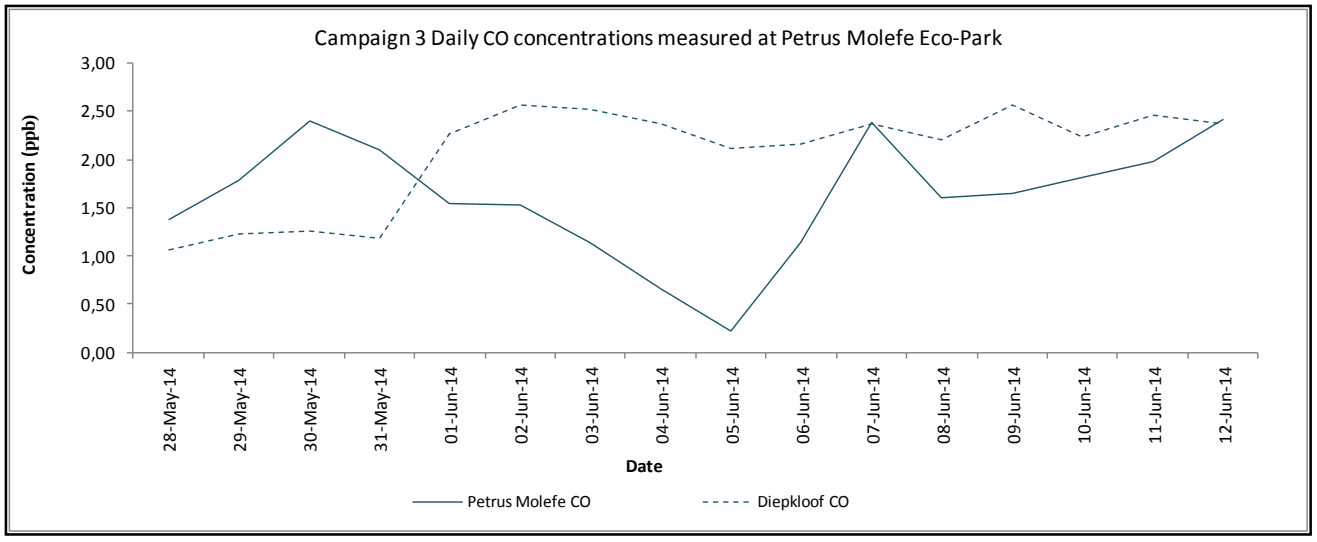


Figure 3-55: Campaign three daily average CO concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

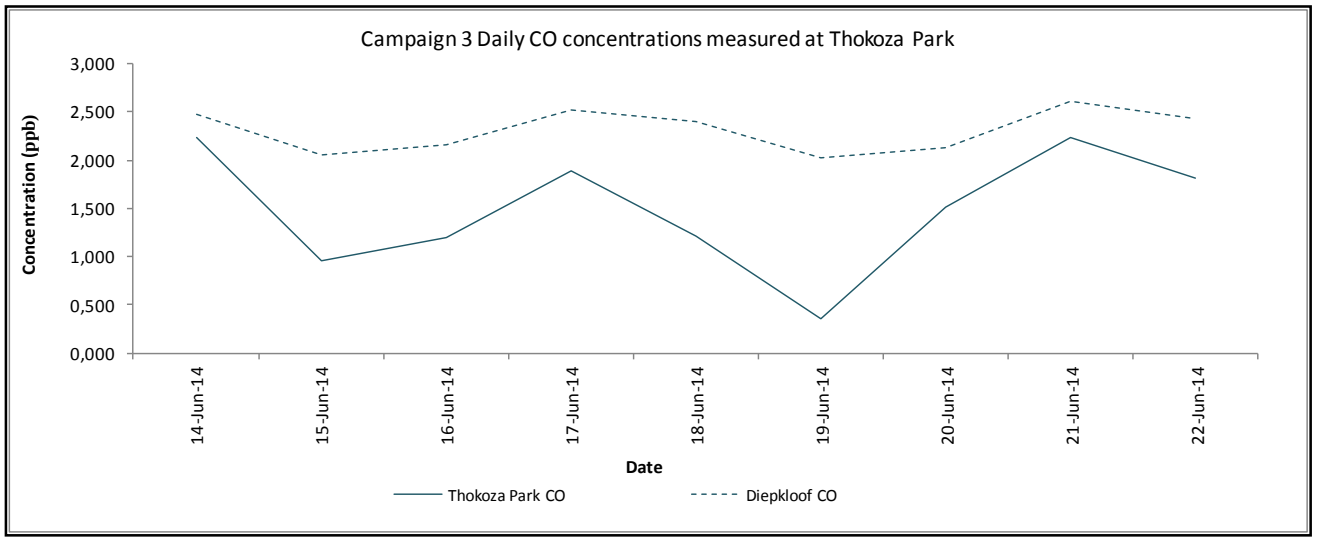


Figure 3-56: Campaign three daily average CO concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-19: Data summary of ambient CO concentrations (ppb) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Three		
Mean	1.6	1.5
Median	1.6	1.5
Standard Deviation	0.6	0.6
Minimum	0.2	0.4
Maximum	2.4	2.2

Table 3-20: Two tailed t-Test analysis with unequal variances of CO concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	16	0.46	P > 0.05	0.65

CO concentrations measured at both the urban parks appeared to have an inverse relationship to ambient temperature (Figure 3-57 and 3-58). On the 6 June 2014 the daily average ambient temperature at Petrus Molefe Eco-Park dropped significantly from approximately 14 ^{o}C to 3 ^{o}C which coincided with a significant increase in CO concentrations (Figure 3-57). Lower CO concentrations were observed from the 1 – 5 June 2014 when the daily average ambient temperature increased above 12 ^{o}C. Similarly, lower CO concentrations at Thokoza Park were also observed on days when the daily average temperature increased with the exception of the 19 June 2014 (Figure 5-58).

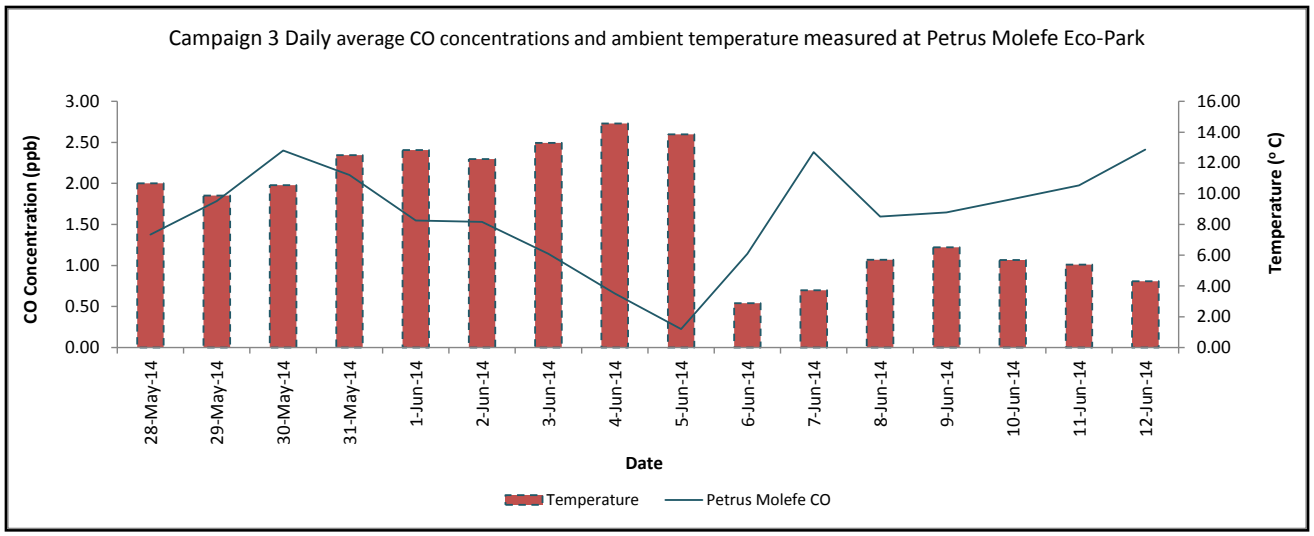


Figure 3-57: Campaign three daily average CO concentrations and temperature measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

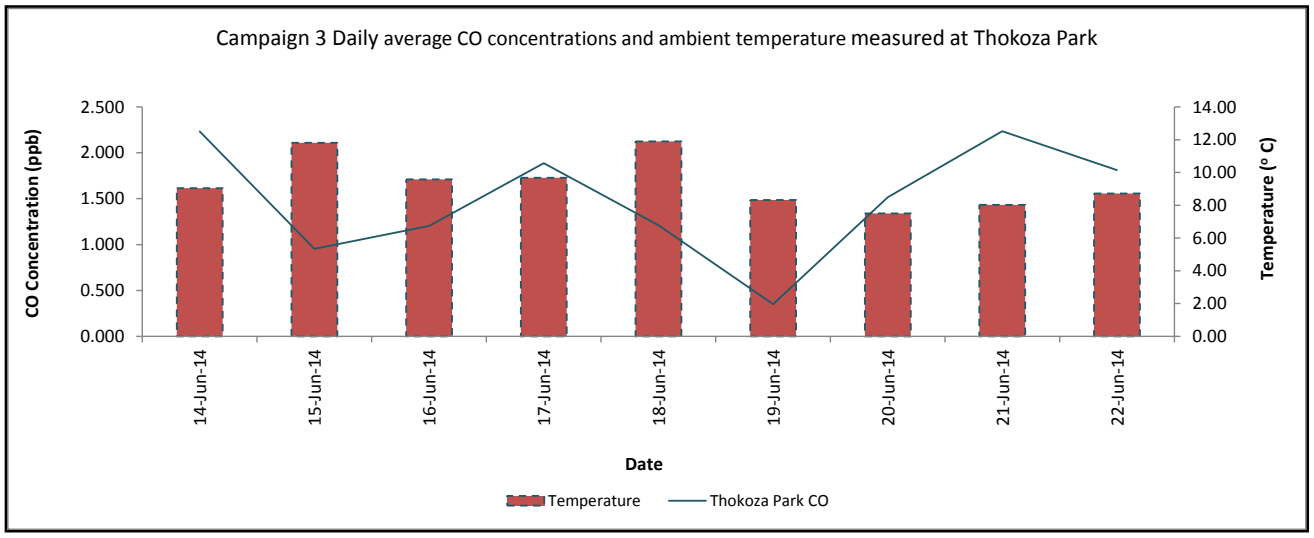


Figure 3-58: Campaign three daily average CO concentrations and temperature measured at Thokoza Park (14 - 22 June 2014).

The two tailed paired t-Test analysis suggested that there was a statistically significant difference, although very narrow, in the mean CO concentrations between Thokoza Park and the urban background site ($P < 0.05$) (Table 3-21). Even though the concentrations of CO recorded at both the urban park types were very similar, a significant difference in the mean CO concentrations was not observed between Petrus Molefe Eco-Park and the urban background site ($P > 0.05$) (Table 3-21).

The CO concentrations recorded at Thokoza Park were lower than the urban background site, however they also followed the same general patterns as the CO concentrations measured at the urban background site (Figure 3-56). The CO concentrations recorded at Petrus Molefe Eco-Park on the other hand were higher than

the urban background concentrations for 28 – 31 May 2014 and appeared to have followed a different pattern to the urban background site.

Table 3-21: Two tailed paired t-Test analysis of urban park ambient CO concentrations (ppb) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	-2.01	0.062	P > 0.05	-0.45	-0.93	0.03
Thokoza Park (older trees)	8	-5.48	0.0006	P < 0.05	-0.82	-1.17	-0.47

Diurnal variation in CO concentrations is observed at both the urban park types and the urban background site (Figure 3-59 and 3-60). An increase in the CO concentrations is observed at Thokoza Park and Petrus Molefe Eco-Park in the morning (04:00 – 07:00) and evening (16:00 – 20:00) periods. Similarly, an increase in CO concentrations is observed at the urban background site in the morning (05:00 – 07:00) and evening (16:00 – 18:00) periods. However, a significant decline in CO concentrations is observed throughout the day (08:00 – 16:00) at the two urban parks but not at the urban background site.

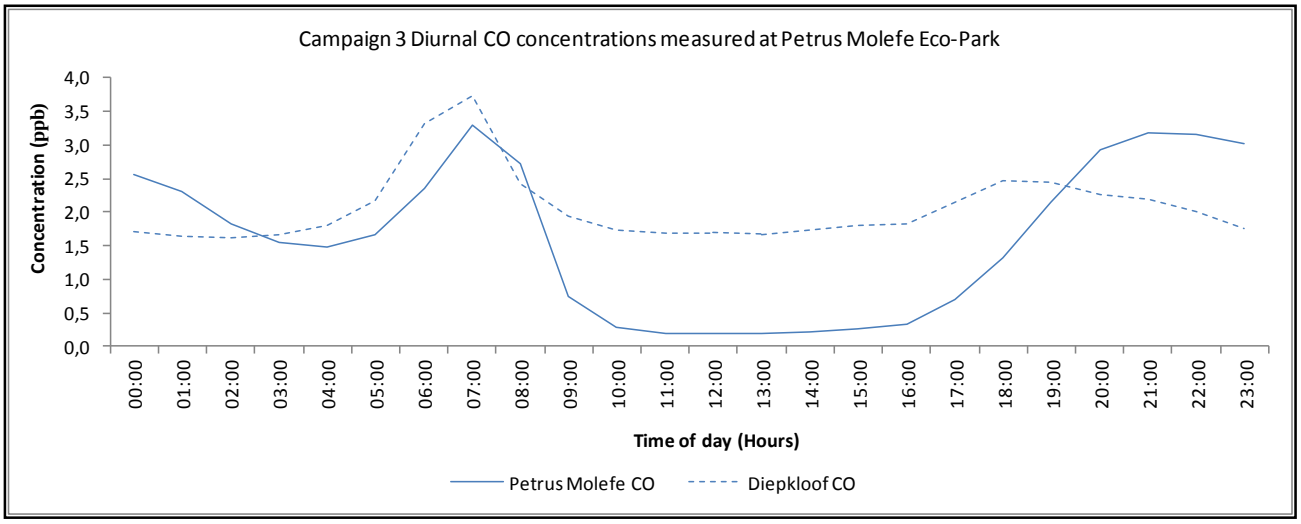


Figure 3-59: Campaign three diurnal CO concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

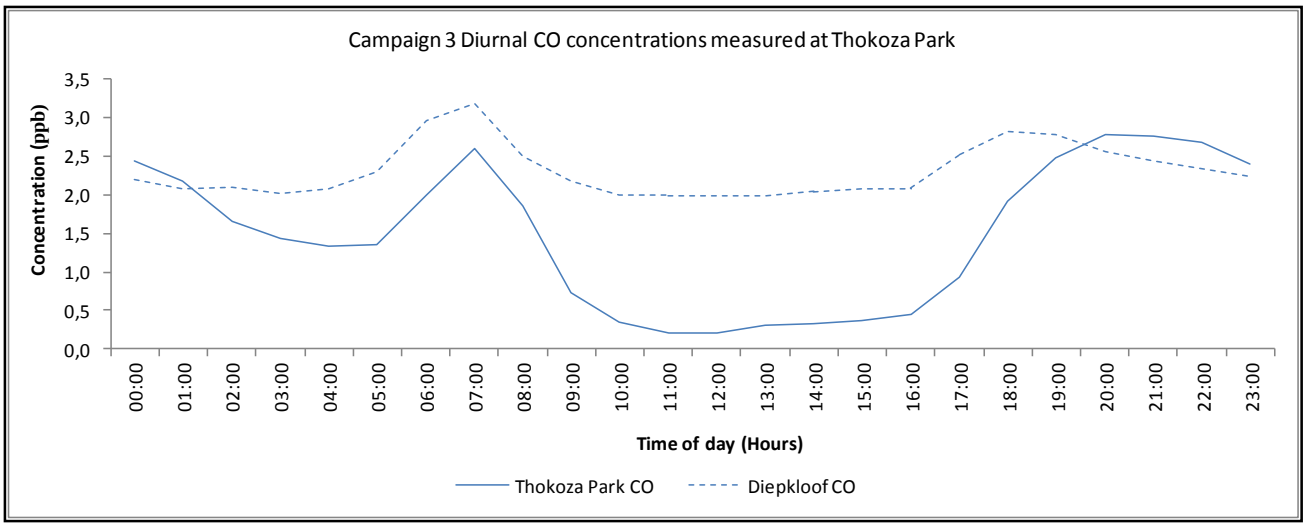


Figure 3-60: Campaign three diurnal CO concentrations measured at Thokoza Park (14 - 22 June 2014).

3.2.5. Sulphur Dioxide

Ground level SO_2 concentrations were only measured at Petrus Molefe Eco-Park and Thokoza Park during the third monitoring campaign as the SO_2 analyser was broken for the duration of campaign two. The daily average SO_2 concentrations recorded at both urban parks during campaign three were well below the South African National air quality standard of 48 ppb (Figure 3-61 and 3-62). Similarly the hourly average and 10 minute average SO_2 concentrations recorded at both urban parks were well below the South African National air quality standard of 134 ppb and 191 ppb throughout the third monitoring campaign.

Very low daily average SO_2 concentrations were recorded at both urban park types with mean concentrations of 3.6 ppb (0.1 – 7.8 ppb) and 4.6 ppb (0.02 – 13.8 ppb) recorded at Petrus Molefe Eco-Park and Thokoza Park respectively (Table 3-22). Statistically, no significant difference in the mean SO_2 concentrations was observed between the two urban park types ($P > 0.05$) (Table 3-23). However, slightly higher SO_2 concentrations were recorded at Thokoza Park (Figure 3-61 and 3-62) (Table 3-22).

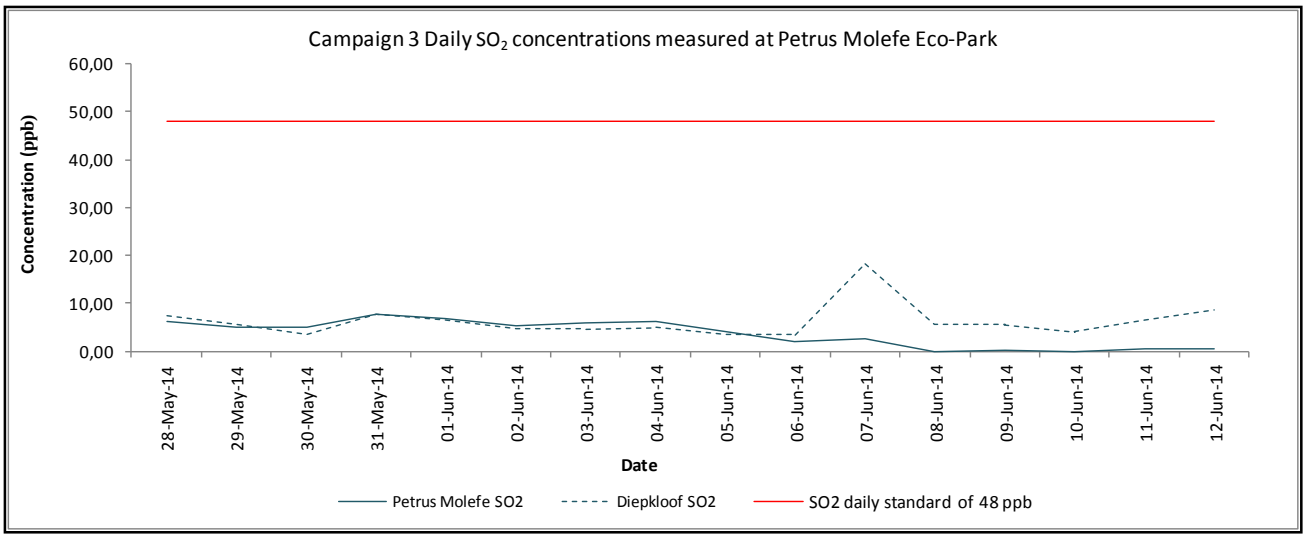


Figure 3-61: Campaign three daily average SO_2 concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

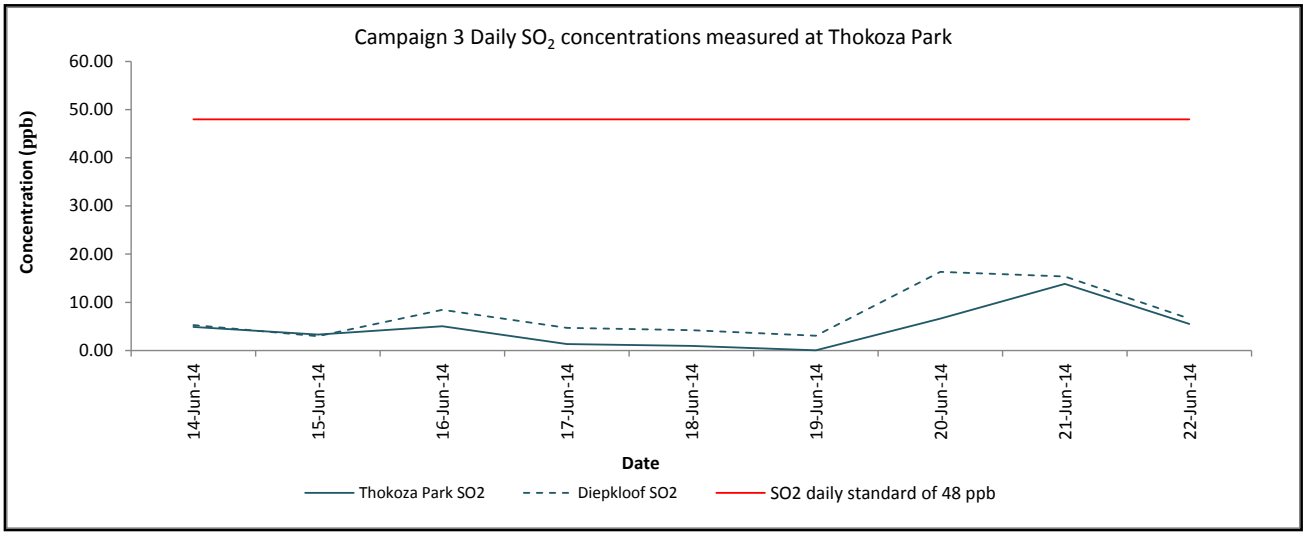


Figure 3-62: Campaign three daily average SO_2 concentrations measured at Thokoza Park (14 - 22 June 2014).

Table 3-22: Data summary of ambient SO_2 concentrations (ppb) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign three (28 May - 22 June 2014).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Three		
Mean	3.6	4.6
Median	4.5	4.9
Standard Deviation	2.8	4.1
Minimum	0.1	0.02
Maximum	7.8	13.8

Table 3-23: Two tailed t-Test analysis with unequal variances of SO_2 concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Three				
Petrus Molefe Eco-Park and Thokoza Park	12	-0.63	P > 0.05	0.54

A statistically significant difference in the mean SO_2 concentrations between the two urban parks and the urban background site was determined from the paired t-Test analysis (P < 0.05) (Table 3-24). Similar mean differences of -2.67 ppb and -2.82 ppb were calculated for Petrus Molefe Eco-Park and Thokoza Park respectively, and show that slightly higher concentrations of SO_2 were recorded at the urban background site (Table 3-24).

The SO_2 concentration plots for both the urban parks depict very similar concentrations and pattern compared the urban background site except on the 7 June 2014 where a significant rise in SO_2 concentrations was observed at the urban background site (Figure 3-61 and 3-62).

Table 3-24: Two tailed paired t-Test analysis of urban park ambient SO_2 concentrations (ppb) compared against the urban background site.

PARK TYPE	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)		MEAN DIFFERENCE	95% CONFIDENCE INTERVAL ABOUT MEAN DIFFERENCE	
Campaign Three						**lower**	**upper**
Petrus Molefe Eco-Park (younger trees)	15	-2.32	0.035	P < 0.05	-2.67	-5.12	-0.22
Thokoza Park (older trees)	8	-2.89	0.020	P < 0.05	-2.82	-5.07	-0.57

Two differences in the diurnal variation plots for SO_2 are seen between the two urban park types. At Petrus Molefe Eco-Park the concentration of SO_2 increased slightly in the morning at 7:00 and then remained within the range of 3 – 6 ppb throughout the rest of the day and evening (Figure 3-63). On the other hand, at Thokoza Park the concentration of SO_2 increased significantly at 7:00 reaching a maximum of 14.7 ppb at 11:00 and then steadily decreased thereafter (Figure 3-64).

It appears as if the SO_2 concentrations measured at both the urban parks follow a very similar diurnal pattern to the urban background site with the exception of a slight lag time. A steeper increase in the PM10 concentrations at the urban background station is observed during the early morning (04:00 – 9:00) and evening (16:00 – 19:00) periods however this is not observed at the two urban parks. The diurnal plots also show that higher concentrations of SO_2 were recorded at the urban background site compared to the two urban parks (Figure 3-63 and 3-64).

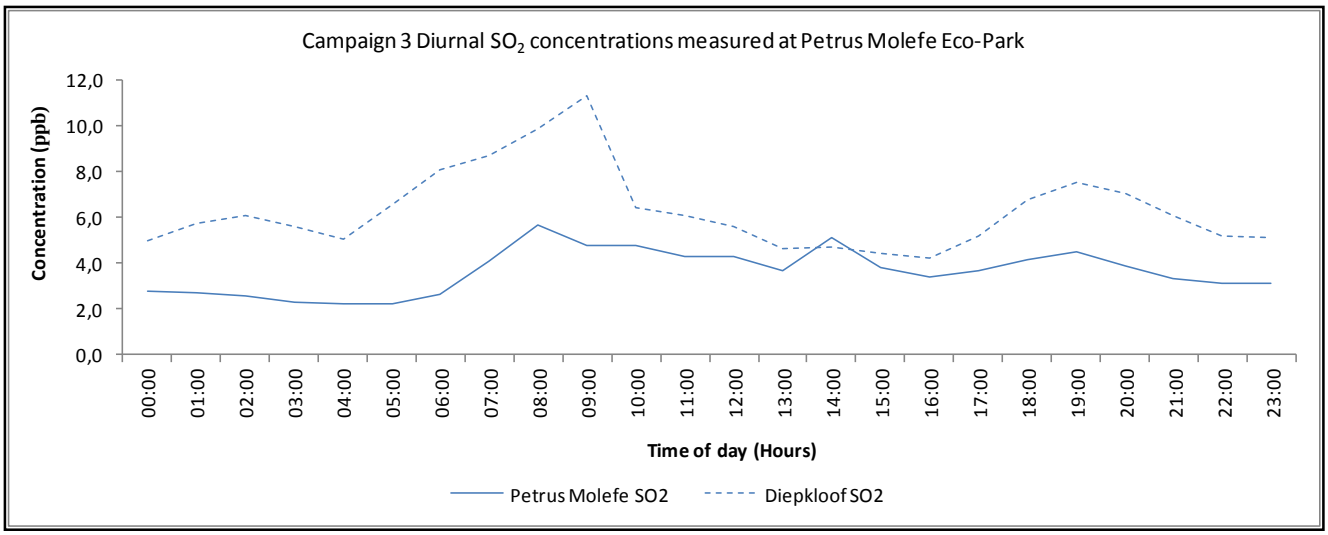


Figure 3-63: Campaign three diurnal SO_2 concentrations measured at Petrus Molefe Eco-Park (28 May - 12 June 2014).

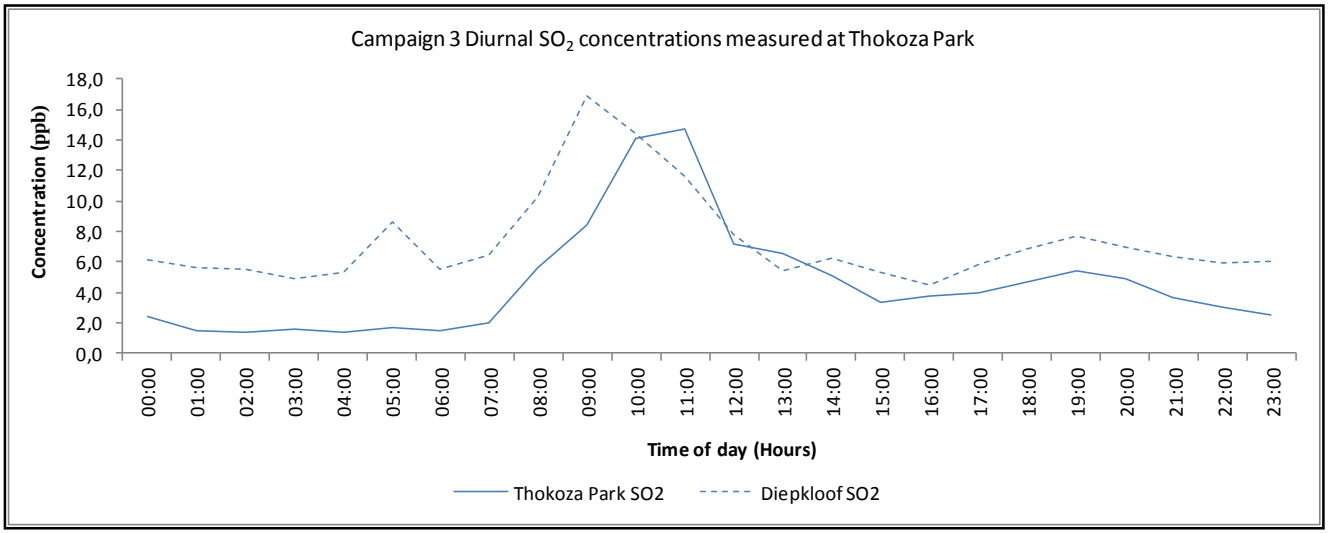


Figure 3-64: Campaign three diurnal SO_2 concentrations measured at Thokoza Park (14 - 22 June 2014).

3.2.6. Carbon Dioxide

Ambient CO_2 concentrations were only measured at Petrus Molefe Eco-Park and Thokoza Park during the second monitoring campaign as the CO_2 analyser was only made available for this period

There was greater variation in the CO_2 concentrations recorded at Thokoza Park compared to Petrus Molefe Eco-Park. At Petrus Molefe Eco-Park the daily average CO_2 concentrations were relatively consistent and ranged from 250 – 310 ppm over the campaign (Figure 3-65). At Thokoza Park the daily average CO_2 concentrations steadily decreased over the monitoring period and ranged from 347 – 362 ppm (Figure 3-66).

A significant difference in the mean CO_2 concentrations between the two urban park types was found from the two tailed t-Test analysis ($P < 0.05$) (Table 3-26). Contrary to what was expected, higher concentrations of CO_2 were observed at Thokoza Park compared to Petrus Molefe Eco-Park with mean concentrations of 354 ppm and 297 ppm recorded respectively (Table 3-25). The concentrations of CO_2 measured at the two urban parks could not be compared to the urban background site as these data were not available.

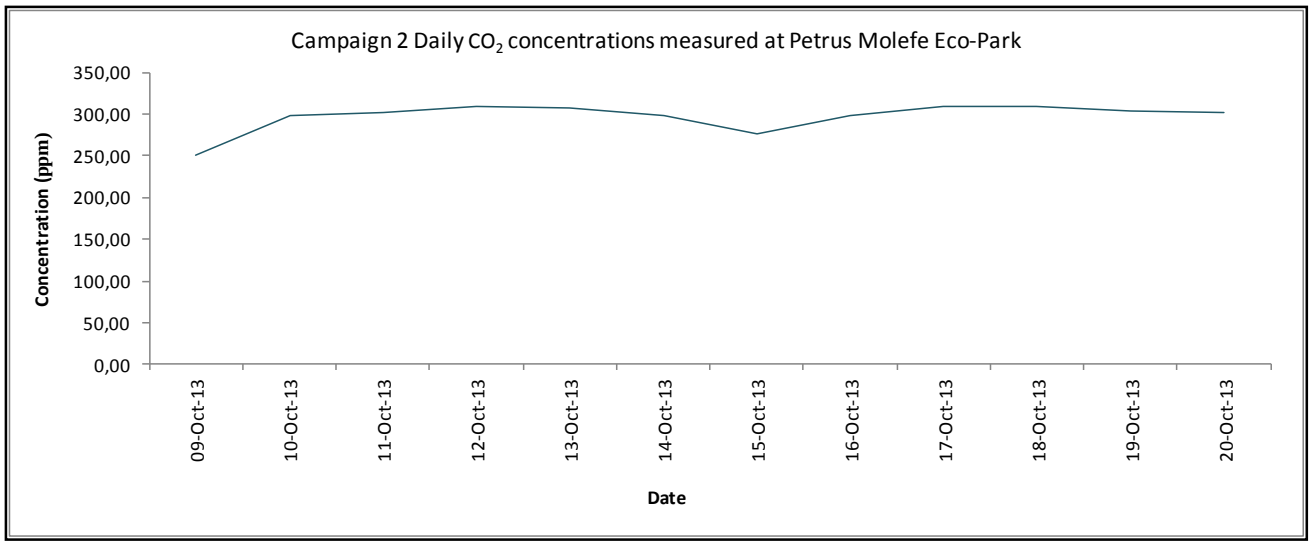


Figure 3-65: Campaign two daily average CO_2 concentrations measured at Petrus Molefe Eco-Park (9 - 20 October 2013).

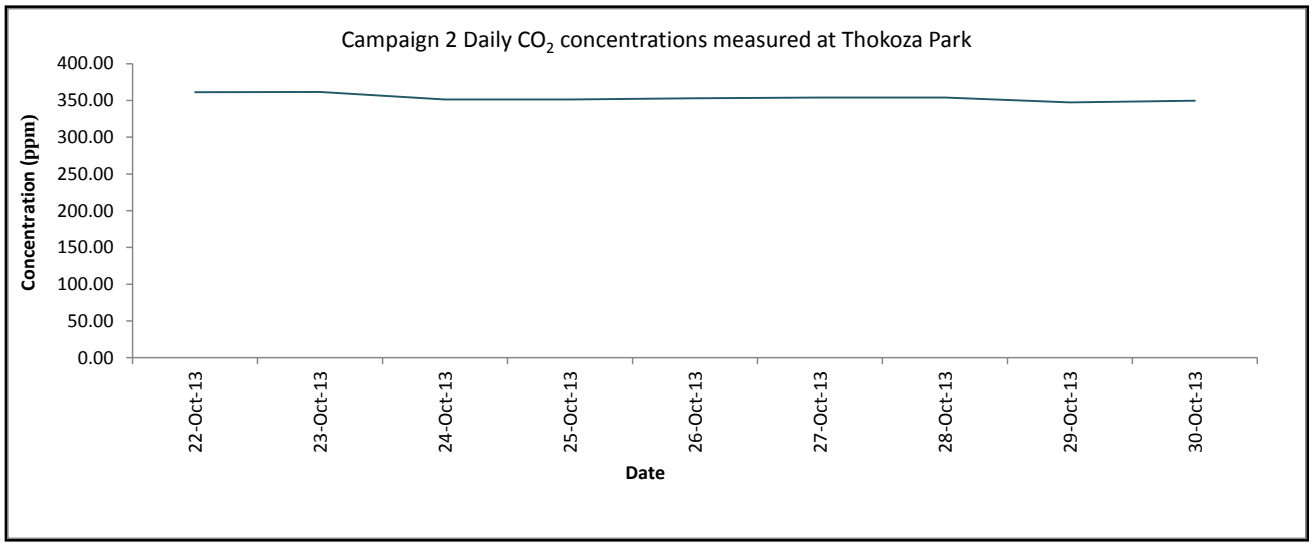


Figure 3-66: Campaign two daily average CO_2 concentrations measured at Thokoza Park (22 - 30 October 2013).

Table 3-25: Data summary of ambient CO_2 concentrations (ppm) measured at Petrus Molefe Eco-Park and Thokoza Park during campaign two (9 - 30 October 2013).

	PETRUS MOLEFE ECO-PARK	THOKOZA PARK
Campaign Two		
Mean	297.2	353.7
Median	301.8	352.9
Standard Deviation	17.4	4.8
Minimum	250.0	347.4
Maximum	310.8	361.5

Table 3-26: Two tailed t-Test analysis with unequal variances of CO_2 concentrations at both urban park types, Petrus Molefe Eco-Park (younger trees) and Thokoza Park (older trees).

	DEGREES OF FREEDOM (*d.f.*)	T-VALUE	P-VALUE (α = 0.05)	
Campaign Two				
Petrus Molefe Eco-Park and Thokoza Park	13	-10.71	P < 0.05	8.07×10^{-8}

No diurnal variation in CO_2 concentrations is observed at Petrus Molefe Eco-Park and Thokoza Park during campaign two (Figure 3-67). However, two differences are observed between the two urban parks types. The diurnal variation plot for CO_2 clearly depicts that lower concentrations of CO_2 were recorded at Petrus Molefe Eco-Park compared to Thokoza Park.

The plot also shows that the concentrations of CO_2 recorded at Thokoza Park were more consistent over the day and evening periods compared to Petrus Molefe Eco-Park. Diurnal concentrations at Thokoza Park ranged from 349 – 359 ppm and from 251 – 308 ppm at Petrus Molefe Eco-Park (Figure 3-67).

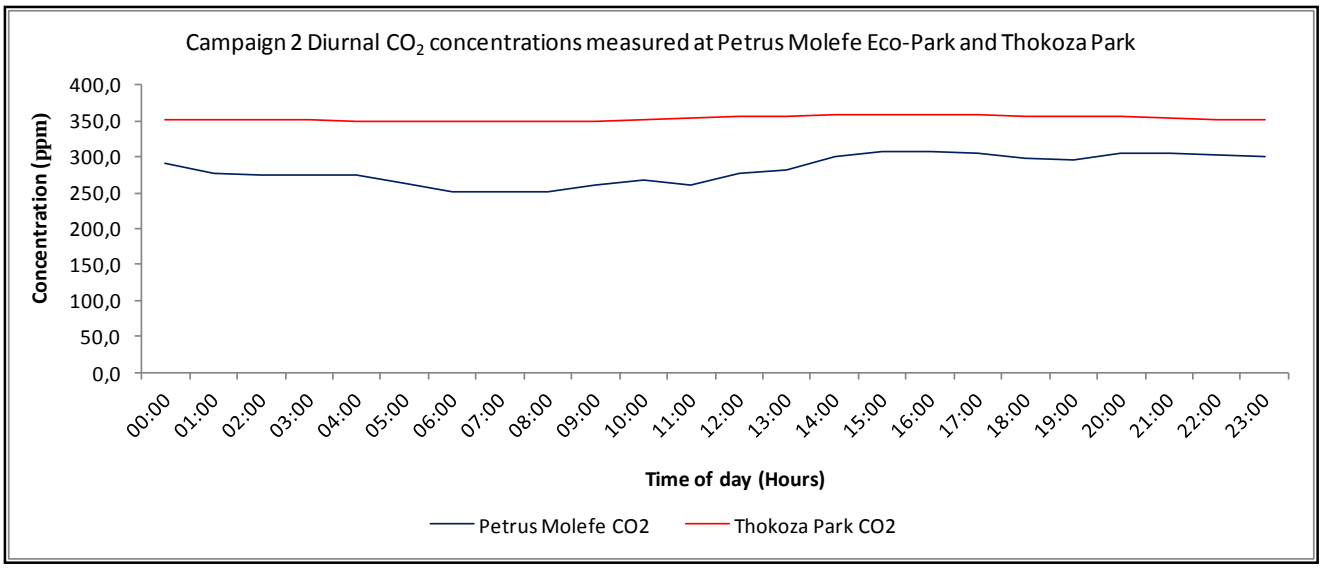


Figure 3-67: Campaign two diurnal CO_2 concentrations measured at Petrus Molefe Eco-Park (9-20 October 2013) (blue line) and Thokoza Park (22 - 30 October 2013) (red line).

4. Chapter Four: Discussion

This chapter includes a discussion of the findings obtained during this study. The main focus of the study was to compare air quality between two urban parks types, namely Petrus Molefe Eco-Park characteristic of an open space park with young trees and Thokoza Park characteristic of a treed park consisting of older trees with well-developed tree canopies. The study further aimed to compare the concentrations of key criteria air pollutants recorded at the two urban parks types with urban background concentrations obtained from a nearby air quality monitoring station as well as the South African National air quality standards.

4.1. Meteorological Variables

Air quality is significantly influenced by prevailing meteorological conditions. Air pollutants tend to increase with calmer winds and stable atmospheric conditions and air pollutant concentrations tend to decrease with higher wind speeds and unstable atmospheric conditions (Held *et al.,* 1996; Cavanagh *et al.,* 2009; Ryu *et al.,* 2012). Secondary air pollutants, such as O_3, will increase during warmer periods due to the increase in temperature dependent photo-oxidation reactions amongst O_3 precursor gases (Yang *et al.,* 2005; Paoletti, 2009). On the other hand primary pollutants can decrease during the warmer period due to increased convectional uplift and turbulence facilitating the dilution and dispersion of pollutants as well as reduced domestic biomass burning activities (Cavanagh *et al.,* 2009; Yin *et al.,* 2011).

Urban trees can influence air quality through changes in local meteorological conditions which can result in either direct or indirect reductions in air pollutant concentrations (Nowak, 2002; Yang *et al.,* 2005; Leung *et al.,* 2011). In this study five meteorological variables, namely wind speed, wind direction, temperature, relative humidity and precipitation were recorded at both Thokoza Park and Petrus Molefe Eco-Park during the spring season of 2013 and the winter season of 2014. Meteorological data were also obtained from the Diepkloof monitoring station to represent urban background meteorological conditions. Interestingly, very similar meteorological conditions were observed between the two urban park types and the urban background site with a few notable differences in wind speed, temperature and relative humidity observed.

Wind Speed

A significant difference in wind speed was observed between Thokoza Park and Petrus Molefe Eco-Park and the urban background site. A higher frequency of calmer winds was recorded at Thokoza Park. Calmer winds can be attributed to the aerodynamic roughness of the surface of the older trees with well-developed tree canopies. As the air mass comes into contact with the tree surface, it will flow over and move across the leaves and plant components in which the wind velocity decreases significantly (Litschke and Kuttler, 2008;

Pullman, 2009; Mensink *et al.,* 2012). Dry deposition of particles and gases will occur as the air mass moves across and through the tree canopy. Dry deposition processes are enhanced with an increase in the aerodynamic roughness of the tree, greater variation in the tree configuration and structure and the larger the size of the tree canopy cover (Nowak *et al.,* 2000; Nowak, 2006).

The higher frequency of calm winds observed at Thokoza Park in comparison to Petrus Molefe Eco-Park and the urban background site provides as an indication of the interaction between the air mass and the tree canopy which could facilitate dry depositional processes. This is particularly true for the deposition of particles which mostly depends on impaction and interception processes (Beckett *et al.,* 1998). Lower concentrations of pollutants could result due to enhanced dry depositional processes associated with aerodynamic roughness of the urban tree canopies as indicated by several studies (Nowak *et al.,* 2000; Nowak *et al.,* 2006; Yang *et al.,* 2005; Alonso *et al.,* 2011; Yin *et al.,* 2011). On the other hand the findings of Litschke and Kuttler (2008) and Gromke and Ruck (2010) show that reduced wind speeds can limit the dispersion of air pollutants which may result in localised higher concentrations of pollutants.

Temperature

A difference in ambient temperatures between the two urban park types and the urban background site was also observed in this study. In campaign two over the spring season ambient temperatures recorded at Thokoza Park were lower compared to Petrus Molefe Eco-Park. Lower temperatures recorded at Thokoza Park during campaign two could be attributed to the presence of older trees with well-developed tree canopies. Trees can reduce local ambient temperatures through the provision of shade and evapotranspiration processes. Well established tree canopies are able to absorb incoming solar radiation, store heat and also reduce the amount of heat and solar radiation intercepting the ground surface thus reducing ambient temperatures. Furthermore, trees can utilise latent heat during the evapotranspiration process, thus reducing ambient temperatures (McPherson *et al.,* 1994; Alonso *et al.,* 2011; Leung *et al.,* 2011).

However, ambient temperatures recorded at both the parks and the urban background site were very similar during campaign two with very small (< 0.5) mean temperature differences observed. Slightly lower temperatures recorded at Thokoza Park during the spring season (campaign two) compared to Petrus Molefe Eco-Park is probably due to the fact that the sampling periods differed. For the period that sampling took place at Thokoza Park, 22 – 30 October 2014, lower temperatures were also recorded at the Diepkloof monitoring station compared to the previous sampling period. Thus, it is unlikely that the trees at Thokoza Park can account for the lower temperatures recorded at Thokoza Park compared to Petrus Molefe Eco-Park during the spring season.

It was found that ambient temperatures recorded at Petrus Molefe Eco-Park were very similar to the urban background site especially during the spring season. This is most likely due to the fact that Petrus Molefe Eco-Park is characteristic of an open space park with younger trees, therefore temperature reduction due to shade and evapotranspiration is less prevalent. Interestingly, the ambient temperatures at Petrus Molefe Eco-Park were slightly warmer compared to the urban background site from the 2 - 6 June 2014 during the onset of a

cold front indicated by a rapid decline in the ambient temperatures at both sites. The slightly warmer temperatures observed at Petrus Molefe Eco-Park may be attributed to the slow release of heat from the Klipspruit wetland which is located directly next to the park. Wetlands have a larger heat storage capacity compared to terrestrial and artificial surfaces, thus the slow release of heat during the cold front may have accounted for the slightly higher temperatures observed at Petrus Molefe Eco-Park for the period 2 – 6 June 2014 (Souch *et al.,* 1998).

Ambient temperatures recorded at Thokoza Park during both monitoring campaigns (spring and winter) were slightly warmer compared to the urban background site, but unlike Petrus Molefe Eco-Park these were shown to be statistically significant. However, as mentioned earlier, during the spring season (campaign two) the mean temperature difference between Thokoza Park and the Diepkloof monitoring station was very small (0.24 ^{o}C) and may even be considered negligible. On the other hand, during the winter season a slightly greater mean temperature difference of 2.0 ^{o}C was observed between Thokoza Park and the Diepkloof monitoring station. Slightly warmer temperatures, although statistically insignificant, were also observed at Thokoza Park in comparison to Petrus Molefe Eco-Park during the winter season.

Warmer ambient temperatures observed at Thokoza Park in comparison to the Diepkloof monitoring station and Petrus Molefe Eco-Park (only during winter season) could be as a result of reduced mixing of heat emitted from the ground below the tree canopies and the presence of a higher frequency of calmer winds. The trees at Thokoza are relatively scattered with open spaces between the clusters of trees. Incoming solar radiation is able to reach and heat the exposed ground surface. The clusters of trees could limit the mixing and convectional uplift of the heat that is emitted from ground surface and may explain for slightly warmer temperatures observed at Thokoza Park (Nowak, 2000). Furthermore, a high frequency of calmer winds observed at Thokoza Park could also limit the mixing and vertical uplift of heat thus reducing heat loss. This is interesting as it indicates that the trees at Thokoza Park may act as a buffer for cold winds as well as limit heat loss during the winter season (Leung *et al.,* 2011).

In summary, the temperature differences between the three sites during the spring season (campaign two) can be considered negligible with only very small temperature differences observed. The difference in temperatures between the two urban parks observed in this study is most likely as a result of changes in background temperature conditions over the different sampling periods and not the influence of the older urban trees at Thokoza Park.

In the winter season (campaign three) higher temperatures were observed at Thokoza Park compared to the urban background site and Petrus Molefe Eco-Park. The temperature difference is likely as a result of reduced heat loss from the trees at Thokoza Park (Nowak, 2000). Studies have shown that trees may be indirectly beneficial in terms of air quality by acting as a buffer for cold winds (Nowak, 2000; Cavanagh *et al.,* 2009; Leung *et al.,* 2011; Yin *et al.,* 2011). This could have a potential indirect positive impact on local air quality as warmer temperatures may result in reduced domestic biomass burning activity in those households during the winter season. Cavanagh *et al.,* (2009) showed that significantly higher PM_{10} concentrations were recorded at an urban park in New Zealand due to increased wood oven burning in surrounding residential areas when a combination of low temperatures and higher wind speeds persisted. Further research would need to be

conducted to establish whether the urban trees in Soweto have the potential to buffer neighbouring houses from cold wind breezes and if this would impact air quality in terms of domestic biomass burning activities.

Rainfall

No difference in precipitation was observed between the urban parks and the urban background site. Throughout both monitoring campaigns there was no precipitation, thus the influence of wet deposition on air pollutants concentrations at the two urban parks and the urban background site was absent as all days monitored were no-rain-days. Wet deposition is the removal of air pollutants through precipitation (i.e. rainfall or snow). Gases and particulates are removed from the atmosphere via wet deposition either through the formation of clouds and rain droplets/snowflakes or they are scavenged during rain or snow fallout (Lovett, 1994). Wet depositional processes tend to have a cleansing effect on air through the efficient removal of air pollutants, therefore resulting in lower concentrations of air pollutants. In this study dry deposition was the main depositional process influencing air pollutants concentrations as no rainfall was recorded for each monitoring campaign.

Relative Humidity

A distinct difference in relative humidity was only observed between the two urban park types once in campaign two during the spring season in which Thokoza Park was associated with generally higher relative humidity conditions. Evapotranspiration by trees could have accounted for the slightly higher relative humidity conditions observed at Thokoza Park (Betts *et al.,* 2008; Leung *et al.,* 2011). However, there was a significant increase in urban background relative humidity conditions during the changeover period from Petrus Molefe Eco-Park to Thokoza Park. Therefore the higher relative humidity conditions observed at Thokoza Park are more likely as a result of an increase in background humidity conditions rather than evapotranspiration from the trees. This suggests that relative humidity conditions at both the urban park types are similar which is likely due to evapotranspiration processes from the Klipspruit wetland located next to Petrus Molefe Eco-Park.

Relative humidity conditions observed between the two urban parks and the urban background site were significantly different during both monitoring campaigns. The urban parks were associated with much higher relative humidity conditions with higher mean humidity differences observed during the winter season (campaign three). Evapotranspiration processes due to the urban trees and the Klipspruit wetland might account for higher relative humidity conditions observed at the two urban parks in comparison to the urban background site (McPherson *et al.,* 1994; Souch *et al.,* 1998; Nowak, 2000; Leung *et al.,* 2011).

Higher relative humidity conditions (~ > 40%) observed at the two urban parks can have a positive impact on air quality. Lower concentrations of particulate matter can occur due to increased occult depositional processes. Higher relative humidity conditions can facilitate the sedimentation of particles due to the growth in size/weight of the particles as they absorb water (Beckett *et al.,* 1998). Higher concentrations of pollutants such as NO_2 can also occur as the availability of the hydroxyl radical OH^- for reaction with gases increases

under higher relative humidity conditions (Elminir, 2005; Zhang *et al.,* 2004). On the other hand, high humidity conditions (~ > 80%) can cause an increase in air pollutant concentrations. For instance, higher oxidation reactions can allow for a greater availability of precursor gases for the formation of secondary pollutants such as O_3 (Elminir, 2005).

4.2. Air Pollutant Concentrations

4.2.1. Particulate Matter (PM_{10} and $PM_{2.5}$)

Out of all the air pollutants monitored in this study, particulate matter was the only pollutant that exceeded the South African National air quality standards at Petrus Molefe Eco-Park and Thokoza Park. Exceedances of the 120 µg/m^3 and 75 µg/m^3 24-hour standards for PM_{10} and the 65 µg/m^3 24-hour standard for $PM_{2.5}$ were observed at both urban parks types during campaign three over the winter season. Interestingly, for the urban background site only one exceedance of the PM_{10} standard of 75 µg/m^3 was observed on the 5 June 2014 during campaign three.

Over the winter season higher concentrations of particulates are often recorded in Johannesburg due to the intensification of the high pressure belt over the interior plateau, which is associated with inversion layers and calm winds limiting the dilution and dispersion potential of particulates in the atmosphere during the season (Held *et al*., 1996; Ryu *et al.,* 2012). Furthermore, during the winter season there is little to no rainfall and lower humidity conditions over the interior, which allows for higher concentrations of particulates to persist as a result of reduced wet depositional processes (Lovett, 1994; Tyson and Preston-Whyte, 2013). Reduced precipitation over the winter period also results in lower soil moisture content, thus facilitating dust emissions from exposed soils, unpaved roads, tailings, etc. (Betts *et al.,* 2008).

Domestic biomass burning is also a significant contributor to particulate matter emissions during the winter season as a result of increased biomass burning activity in the informal residential areas. In the warmer seasons domestic biomass burning is mostly used for cooking and lighting purposes whereas during the winter season biomass burning is also used for heating purposes (Kornelius *et al.,* 2012; Naidoo *et al.,* 2014).

Comparison of PM_{10} and $PM_{2.5}$ between Thokoza Park and Petrus Molefe Eco-Park

Two notable differences in PM_{10} and $PM_{2.5}$ concentrations were observed between the two urban park types. Higher concentrations of PM_{10} were recorded at Thokoza Park compared to Petrus Molefe Eco-Park whereas lower concentrations of $PM_{2.5}$ were recorded at Thokoza Park compared to Petrus Molefe Eco-Park during both monitoring campaigns. As monitoring was not conducted over the same time frame at the two urban parks, differences in the prevailing local wind fields observed over the different time periods could explain the significant difference in PM_{10} and $PM_{2.5}$ concentrations between the two urban park types.

The direction of downward transport of particulates from its source is a function of the prevailing wind direction (Elminir, 2005).The predominant wind direction for Thokoza park was observed from the north-north-east, south-west and south-east during campaign two and three respectively. A cluster of tailings facilities and industrial areas are concentrated north-north-east of Thokoza Park and agricultural small holdings are located south-west and south-east of Thokoza Park. Downwind transport of particulates from the tailings, industrial areas and exposed soil associated with small agricultural holdings are potentially key emission sources of PM_{10} accounting for elevated PM_{10} concentrations at Thokoza Park (Ogunntoke, *et al.,* 2013).

On the other hand, at Petrus Molefe Eco-Park the predominant wind direction during both monitoring campaigns was from the north-west and north-north-west which was different to that observed at Thokoza Park. Towards the north-west and north-north-west is located a large portion of Soweto, including several formal and informal residential houses. Fewer tailings and small agricultural holdings are located north-west of Petrus Molefe Eco-Park. Domestic biomass burning and vehicle exhaust emissions may be key emission sources of $PM_{2.5}$ accounting for higher $PM_{2.5}$ concentrations at Petrus Molefe Eco-Park.

Studies have shown that particles of different size fractions and elemental constituents are associated with different emission source types (Lovett, 1994; Beckett *et al.,* 1998; Hueglin, *et al.,* 2005). The smaller fraction of particles (i.e. $PM_{2.5}$) is more likely to be sourced from anthropogenic emissions associated with combustion such as domestic biomass burning activities, industrial combustion processes and vehicle exhaust emissions. Contrastingly, the larger fraction of particles (i.e. PM_{10}) is generally made of organic and natural earth crust materials from sources such as tailings, unpaved roads and exposed bare soils (Lovett, 1994; Beckett *et al.,* 1998). It is a possibility that the higher concentrations of PM_{10} observed at Thokoza Park and higher concentrations of $PM_{2.5}$ observed at Petrus Molefe Eco-Park may be attributed to the transport of PM_{10} and $PM_{2.5}$ from different downwind sources due to different prevailing wind directions. This observation agrees with the findings of Cavanagh *et al.* (2009) who showed that changes in wind direction significantly influenced PM_{10} concentrations at different urban parks located in close proximity to each other.

Diurnal variation in $PM_{2.5}$ concentrations at Petrus Molefe Eco-Park over the winter season also suggests that household combustion and vehicle activity occurring north-west and north-north-west of the sites may be key emissions of $PM_{2.5}$. Diurnal variation in $PM_{2.5}$ concentrations at Petrus Molefe Eco-Park showed an increase in the early morning and evening periods which is likely as a result of increased domestic biomass burning and traffic movement during these periods (Hueglin *et al.,* 2005; Venter *et al.,* 2012). Diurnal variation in PM_{10} concentrations was only observed when the prevailing wind direction was from the north-west and north-north-west. There was no diurnal variation in PM_{10} concentrations at Thokoza Park when different prevailing wind directions were observed thus again suggesting different sources. The difference in diurnal patterns suggests that differing prevailing wind directions could transport particles from different downwind emissions sources accounting for the different concentrations of particulate matter observed at the two urban parks.

There was also a significant difference in wind speeds observed between the two urban parks. Higher wind speeds were observed at Petrus Molefe Eco-Park compared to Thokoza Park. Calmer winds observed at Thokoza Park are likely due to the interaction of the air mass with the tree surface resulting in lower wind speeds (Litschke and Kuttler, 2008; Pullman, 2009; Mensink *et al.,* 2012). Lower wind speeds can limit the

dispersion of particles which may result in localised higher concentrations of PM_{10} and $PM_{2.5}$ (Litschke and Kuttler, 2008; Gromke and Ruck, 2010). While lower wind speeds could partly account for higher PM_{10}, the observation of lower $PM_{2.5}$ suggests other processes are also operating.

Higher humidity conditions were observed at Thokoza Park compared to Petrus Molefe Eco-Park during campaign two but not during campaign three. Higher humidity readings were also recorded at Thokoza Park compared to the urban background site. PM_{10} concentrations decrease with higher relative humidity conditions. Particulates easily absorb water and thus would increase in weight under higher humidity conditions which would facilitate the gravitational settling of the particles. Therefore, in theory, higher humidity conditions observed at Thokoza Park should facilitate deposition of PM_{10} and allow for lower ambient concentrations to be detected (Beckett *et al.,* 1998; Betts *et al.,* 2008). However, significantly higher concentrations of PM_{10} were observed at Thokoza Park compared to the Petrus Molefe Eco-Park and the urban background site. Therefore, it is unlikely that higher humidity conditions would account for higher PM_{10} concentrations observed at Thokoza Park.

Even though sampling at the two urban parks was not conducted over the same time period, the results suggest localised influences on PM_{10} and $PM_{2.5}$. This is supported by the significant differences in PM_{10} and $PM_{2.5}$ concentrations between the parks and the Diepkloof monitoring station, which are located in close proximity to each other.

Comparison of PM_{10} and $PM_{2.5}$ between the Two Urban Parks and the Urban Background Site

Throughout this study lower concentrations of PM_{10} and $PM_{2.5}$ were unexpectedly recorded at the urban background site with the exception of Petrus Molefe during campaign two. Similar meteorological conditions were observed at Diepkloof monitoring station and the urban parks during both monitoring campaigns and both the urban parks and the urban background site are surrounded by similar emission source types. Therefore, it would have been expected that over the same time period and under similar meteorological conditions that similar particulate matter concentrations would have been recorded at the parks and the background site (Lam *et al.,* 2005).

Faster winds recorded at Diepkloof monitoring station compared to the two urban parks are likely to explain for the lower PM_{10} and $PM_{2.5}$ concentrations. Faster wind speeds observed at the Diepkloof monitoring station could have facilitated the dispersion and dilution of particles, thus accounting for lower concentrations (Lam *et al.,* 2005; Litschke and Kuttler, 2008). Differences in surrounding emission sources could also explain for lower PM_{10} and $PM_{2.5}$ concentrations at the Diepkloof station. It appears as if less domestic biomass burning activity took place surrounding Diepkloof station. This is indicated on the diurnal plots for PM_{10} and $PM_{2.5}$ at the parks. The PM_{10} and $PM_{2.5}$ concentrations at the urban parks show a clear rise in the concentrations in the morning and evening periods which could be associated with increased domestic fuel burning (Venter *et al.,* 2012). At the Diepkloof station there appears to be greater variability in the concentrations throughout the day, which provides an indication of differing emission source types.

The only time particulate matter concentrations were higher at Diepkloof monitoring station compared to the two urban parks was during campaign two. Higher concentrations of PM_{10} were recorded at Diepkloof compared to Petrus Molefe Eco-Park during campaign two. During this period, 22 – 30 October 2013, the prevailing wind direction at Diepkloof monitoring station differed significantly from the urban parks. The prevailing wind direction at Petrus Molefe Eco-Park was observed from the north-west and north-north-west, whereas the prevailing wind direction of the Diepkloof station was north-north-east. A couple of tailings facilities are located in close proximity to the north-east of Diepkloof monitoring station. Downwind emissions of PM_{10} from the tailings located north-north-east of Diepkloof monitoring station is likely to explain for higher concentrations during campaign two.

Even though higher concentrations of PM_{10} and $PM_{2.5}$ were recorded at the two urban parks it should be emphasised that urban trees still act as a noteworthy sink for particles. Reduced wind speeds observed at Thokoza indicate that there is an interaction between the air mass and the tree canopy. The aerodynamic roughness of the tree canopy creates turbulence which will facilitate the dry deposition of particles. Dry deposition of particulates via trees occurs through impaction and interception processes between the particle and the leaf and tree exterior surfaces (Lovett, 1994; Beckett *et al.,* 1998; Cavanagh *et al.,* 2009). The interception and impaction of particles will increase with higher wind turbulence due to the tree canopy surface roughness. Greater variation in the tree configuration and canopy structure would increase wind turbulence and facilitate impaction and interception processes in the tree canopy (Cavanagh *et al.,* 2009; Yin *et al.,* 2011). Interception and impaction will also increase if the tree leaf or bark surface is either more sticky or wet (Beckett *et al.,* 1998; Leung *et al.,* 2011). Lower wind speeds would also allow for a longer exposure period in which the particles could be deposited onto the leaf and tree surface components as shown by Yin *et al.* (2011).

The PM_{10} and $PM_{2.5}$ results obtained in this study indicate that wind direction has a significant influence on local particulate matter concentrations. The results also suggest that spatial variation in surrounding emission sources may influence particulate matter concentrations differently through localised impacts even for areas located within close proximity (< 10 km) to each other.

4.2.2. Nitrogen Oxides (NOx)

Hourly average concentrations of NO_x fell below the South African air quality standard of 106 ppb at both urban parks and the urban background site during both monitoring campaigns. Although the concentrations of NO_x recorded in this study were considered acceptable they were also shown to be relatively high in relation to the standard of 106 ppb, reaching a maximum hourly average of 92ppb at Petrus Molefe Eco-Park. This implies that NO_x is a key air pollutant in the area and has the potential of exceeding the standards.

A distinct difference in NO_x concentrations was not observed between Thokoza Park and Petrus Molefe Eco-Park, however slightly higher concentrations of NO_x were recorded at Thokoza Park during the spring season (campaign two) and lower concentrations recorded during the winter season (campaign three). This may be due to the fact that monitoring at the two urban parks was not conducted over the same time frame. At the

Diepkloof monitoring station higher concentrations of NO_x were observed over the period, 22 – 30 October 2013, and lower concentrations were observed for the period 14 – 22 June 2014. These were the same time periods that sampling took place at Thokoza Park. Therefore the slight differences in NO_x concentrations between the two urban park types are likely due to slight changes in background NO_x concentrations.

A significant difference in NO_x concentrations was observed between the two urban parks and the urban background site. Higher concentrations of NO_x were recorded at the two urban parks during the winter season, whereas lower concentrations were recorded during the spring season. Higher concentrations observed at the parks during the winter season may be due to slight differences in relative humidity conditions observed between the urban parks and the background site. During both campaigns slightly higher relative humidity conditions were observed at the urban parks. Higher concentrations of NO_x can occur under higher relative humidity conditions as NO reacts with the hydroxyl radical OH to form NO_2 (Zhang *et al.,* 2004; Jose *et al.,* 2005). Increases in NO_2 are directly proportional to increases in NO_x. Higher concentrations of NO_x at the urban parks during the winter season could be explained by higher oxidation rates of NO to form NO_2 due to higher humidity conditions as well as less photo-oxidation of NO_2 to form O_3. During the winter season lower photo-oxidation reaction rates of NO_x occur as solar radiation is less intensive and lower temperatures persist (Zhang *et al.,* 2004; Jose *et al.,* 2005). Furthermore, faster winds were recorded at the Diepkloof monitoring station compared to the parks. Faster winds could have facilitated the dispersion and dilution of NO_x emissions at the background site, thus contributing to lower concentrations of NO_x observed during the winter season (Elminir, 2005; Saini *et al.,* 2008).

During the spring season lower concentrations of NO_x observed at the urban parks could be due to efficient removal of NO_x by the urban trees. Studies have found that higher removal rates of NO_2 and other gaseous pollutants by trees occur during the spring and summer seasons (Yang *et al.,* 2005; Betts *et al.,* 2008; Yin *et al.,* 2011). This is mainly due to the growth in leaves and tree components, thus facilitating stomatal uptake of gaseous pollutants (Nowak *et al*., 2006; Wuytack *et al*., 2010; Yin *et al.,* 2011). In this respect the lower concentrations of NO_x observed at the two urban parks over the spring season may be due to efficient removal of NO_x by the urban trees. Furthermore, unlike the winter season, higher relative humidity conditions at the urban parks under warmer temperature and higher solar radiation conditions during the spring season could have facilitated photo-oxidation of NO_x. Efficient photo-oxidation of NO_2 could result in lower NO_x concentrations observed at the urban parks during the spring season (Elminir, 2005; Saini *et al.,* 2008). This is supported by the fact that in this study higher O_3 concentrations were observed at the urban parks during the spring season compared to winter. This provides an indication of efficient photo-oxidation of NO_x to form O_3.

In this study vehicle exhaust emissions and domestic biomass burning were identified to be key emissions sources of NO_x in the area. The diurnal plots for NO_x at both the urban parks and the urban background site show a distinct increase in concentrations in the morning and evening periods. This suggests that vehicle exhaust emissions and domestic biomass burning emissions are key contributors of NO_x (Elminir, 2005; Azmi *et al.,* 2010; Venter *et al.,* 2012). In most instances lower concentrations of NO_x were recorded at the urban background site and the urban parks over the weekends, which is likely due to reduced vehicle activity and vehicle exhaust emissions (Venter *et al.,* 2012). However, in the winter season a slight increase in NO_x

concentrations was also observed at the urban parks over some weekends when the ambient temperature decreased. It is possible that domestic biomass burning practices in residential areas surrounding Thokoza Park and Petrus Molefe Eco-Park in addition to vehicle exhaust emissions may account for the higher concentrations of NOx observed at the parks during the winter season (Venter *et al.*, 2012).

4.2.3. Tropospheric Ozone (O_3)

In this study, eight-hour average O_3 concentrations measured at both the urban parks were below the South African National air quality standard of 61 ppb for both monitoring campaigns. In terms of the South African standard people were thus exposed to ambient concentrations of O_3 at Thokoza Park and Petrus Molefe Eco-Park that would not negatively impact on human health. On the other hand the eight-hour average O_3 concentrations measured at the Diepkloof monitoring station exceeded the South African National air quality standard of 61 ppb during both monitoring campaigns.

Higher concentrations of O_3 were recorded at the urban background site compared to the two urban parks during both monitoring campaigns. Higher concentrations at the urban background site may be due to local industrial emissions of VOCs (Zhang *et al.,* 2004; Jose *et al.,* 2005). Unlike the two urban parks, the Diepkloof monitoring station is located in close proximity (< 6 km) to an industrial area (east and north-east of Diepkloof station). VOCs such as alkenes (e.g. C_2H_4) are important precursor gases for O_3 formation. VOCs react to form organic peroxy radicals (RO_2) which react with NO to form O_3. NO_x emissions act as a catalyst for O_3 formation when there is enough VOCs available for reactivity (Zhang *et al.,* 2004). Higher VOC reactivity and higher NO_x concentrations at the Diepkloof monitoring station could possibly account for higher O_3 concentrations observed during this study. Higher concentrations of CO were also recorded at the background site. CO is a precursor gas for O_3 formation and thus could have also contributed to higher concentrations of O_3 observed at the urban background site.

Ground level O_3 concentrations were measured at Thokoza Park and Petrus Molefe Eco-Park during the spring and winter seasons of 2013 and 2014 respectively. No significant difference in mean O_3 concentrations was observed between the two urban parks, however slightly higher mean concentrations were recorded at Petrus Molefe Eco-Park during both seasons. Petrus Molefe Eco-Park is characteristic of an open space park, which consists of younger trees, whereas Thokoza Park consists of older trees with well-developed tree canopies.

Trees are most efficient at removing O_3 through stomatal uptake rather than impaction and interception processes on leaf and tree exterior surface components. As O_3 is transported across the leaf stomata it is then fixed into the intercellular membranes as it undergoes biogenic reactions with solutes and chemical compounds within the leaf (Paoletti, 2009). Factors such as the greater size of the tree canopy, larger leaf surface area, size of stomatal pores and the higher stomata density will promote stomatal uptake of O_3 and other gaseous pollutants (Nowak *et al.*, 2006; Escobedo and Nowak, 2009). Other factors, including the ambient concentrations of O_3, duration of in leaf season and the tree species physiology in response to continuous exposure to O_3 concentrations will also influence the efficiency of stomatal uptake of O_3 via urban

trees (Nowak *et al*., 2006; Alonso *et al.,* 2011; Bamniya, *et al.,* 2012). It may be possible that the combination of the type of species of trees and the larger tree canopies present at Thokoza Park may be beneficial in promoting stomatal uptake of O_3 in comparison to Petrus Molefe Eco-Park accounting for the slightly lower concentrations observed at Thokoza Park (Nowak *et al.,* 2006; Yin *et al.*, 2011).

Meteorological conditions will also significantly influence ground level O_3 concentrations as well as stomatal uptake. O_3 is a secondary air pollutant which is formed in the atmosphere as a result of photochemical reactions amongst ozone precursor gases in the presence of sunlight (Paoletti, 2009; Azmi *et al.,* 2010). Solar radiation, temperature and relative humidity are key meteorological parameters influencing the rate of O_3 production. Higher solar radiation intensity, ambient temperatures and relative humidity will increase photochemical reaction rates amongst O_3 precursor gases (Saini *et al.,* 2008). Trees may mitigate O_3 concentrations through changes in local meteorological conditions. Trees decrease ambient temperatures through the provision of shade and evapotranspiration processes and in turn can reduce photochemical reaction rates amongst O_3 precursor gases, thus reducing O_3 formation (Coder, 1996; Cavanagh *et al.,* 2009; Paoletti, 2009). Yet, this study was limited in that sampling of the two urban parks could not be conducted over the same time frame, thus the slight difference in O_3 concentrations observed may be due to temporal differences in meteorological conditions.

In campaign two a difference in mean temperatures was found between the two parks with higher mean temperatures observed at Petrus Molefe Eco-Park. This finding suggests that O_3 formation may have been facilitated at Petrus Molefe Eco-Park due to higher temperatures resulting in slightly higher ambient concentrations of O_3 during campaign two. In campaign three relative humidity was higher at Petrus Molefe Eco-Park compared to Thokoza Park. As relative humidity increases, so does the production of hydroxyl radicals (OH). The OH radicals react with NO_x in the presence of sunlight to form O_3 (Ball, n.d.). Therefore, slightly higher concentrations of O_3 at Petrus Molefe Eco-Park during campaign three may likely be due to the slightly higher relative humidity conditions.

The diurnal variation plots for O_3 at both urban park types and the urban background station show very similar diurnal patterns. Diurnal variation in O_3 concentrations at both the urban parks and the urban background site showed a rapid increase in concentrations from 6:00 onwards reaching a peak between 12:00 – 14:00. An increase in O_3 formation is facilitated by an increase in solar radiation and temperature where maximum O_3 concentrations are generally observed around midday when solar radiation and ambient temperatures are highest (Saini *et al.,* 2008; Azmi *et al.,* 2010). Higher emission rates of O_3 precursor gases such as NO_x due to increased vehicle activity during peak traffic times will also result in increased O_3 production due to the availability of O_3 precursor gases for photochemical oxidation (Saini *et al.,* 2008; Paolleti, 2009).

One difference in the diurnal plots for O_3 was observed between the urban background site and the two urban parks. Unlike the two urban parks, the concentration of O_3 increased during the night time period at the Diepkloof monitoring station. This is could be due to a lower availability of NO_x emissions for chemical reaction with O_3. During the night time, the titration (destruction) of O_3 occurs as photo-chemical reactions of O_3 precursor gases seize. NO reacts with O_3 to form NO_2 and NO_2 reacts with O_3 to form the nitrate radical NO_3 (Zhang *et al.,* 2004; Ball, n.d.). Vehicle emissions from the Chris Hanni main road may act as a constant

supply of NO_x during the night time period, which could facilitate the titration of O_3 at the two urban parks during the evening.

Although in this study a significant difference in O_3 concentrations between the two park types was not observed studies have shown that urban trees have the potential to mitigate O_3 pollution. The findings of Nowak *et al.* (2000) reports maximum pollutant removal via urban trees for O_3 in comparison to other key air pollutants such as SO_2 and CO. The model predictions of Alonso *et al.* (2011) also show that the influence of land cover change from bare soil to trees in an urban setting resulted in a net decrease in ground level O_3 concentrations and a net increase in dry deposition of O_3 over that area. The study also showed that during the spring season higher depositional rates of O_3 (2%) were recorded due to increased stomatal conductance. Furthermore the findings of Harris and Manning, (2010) also suggest that lower concentrations of O_3 may occur due to the release of NO_2 from below soils which oxidises with O_3 thus facilitating the destruction of O_3 in the tree canopy as well as stomatal uptake of O_3 precursor gases.

4.2.4. Carbon Monoxide (CO)

Ambient air quality monitoring of CO was undertaken at both Thokoza Park and Petrus Molefe Eco-Park over the winter season during campaign three. Concentrations of CO recorded at both the urban park types were very low and fell well below the South African hourly and 8-hourly average air quality standards of 26 000 ppb and 8 700 ppb respectively for the period of monitoring. Similarly, hourly CO concentrations observed at the urban background site were very low and fell well below the South African air quality hourly standard of 26 000 ppb for CO. Similar to SO_2, no exceedances of the standard suggests that CO is not an air pollutant of concern in the area.

The difference in CO concentrations between the two urban park types was found to be negligible with very similar concentrations recorded at both parks. Nonetheless, slightly lower concentrations were recorded at Thokoza Park compared to Petrus Molefe Eco-Park. The marginal difference in CO concentrations between the two urban park types may be attributed to the fact that sampling at the urban parks was conducted over different time frames. During the winter season Thokoza Park was associated with warmer temperatures compared to Petrus Molefe Eco-Park. This was mainly attributed to a general increase in the background temperatures. Higher temperatures observed at Thokoza Park may have facilitated photo-oxidation of CO compounds. CO is considered an important primary precursor gas of O_3 thus warmer temperatures may result in the scavenging of CO and the formation of CO_2 and O_3 (Elminir *et al.,* 2005; Harrison *et al.,* 2014).

In this study an inverse relationship between ambient temperature and the concentration of CO recorded at the two urban parks was found. At both park types a distinct increase in CO concentrations coincided with a significant decline in ambient temperature. Similarly, a distinct decrease in CO concentrations coincided with a significant rise in ambient temperature at both urban parks. This relationship suggests that domestic biomass burning activities in the residential areas surrounding Thokoza Park and Petrus Molefe Eco-Park may be a key source of CO in the area. As the temperature decreases people tend to burn more fuel for heating purposes (Venter *et al.,* 2012; Feig *et al.,* 2014; Naidoo *et al.,* 2014). Distinct diurnal variation in CO concentrations was

also observed at the two urban parks. A significant increase in CO concentrations was observed in the early morning and evening periods. This further suggests that domestic biomass burning activities are a key emission source of CO in the area (Venter *et al.,* 2012).

In contrast to the two urban parks, no relationship between temperature and the CO concentrations was observed at the urban background site. Furthermore, diurnal variation in CO concentrations was less pronounced at the urban background site compared to the two urban parks. The concentration of CO only slightly increased in the morning and afternoon periods. Throughout the day the concentrations remained fairly stable, whereas at the urban parks the concentration decreased significantly after the morning period. Similar concentrations and diurnal patterns were reported by Azmi *et al.* (2011) who found that vehicle exhaust emissions were a key source of CO emissions. The increase in CO concentrations at the Diepkloof station in the morning and evening periods coincided with peak traffic times. This suggests that vehicle exhaust emissions are likely a key source of CO at the urban background site. The continuous supply of CO from vehicle exhaust emissions may also account for the higher CO concentrations observed at the urban background site compared to the urban parks.

The differences in CO concentrations and diurnal patterns observed between the urban background site and the two urban parks are likely due to different CO emissions sources influencing the area (Azmi *et al.,* 2011). Domestic biomass burning appears to have the greatest impact on CO concentrations at Thokoza Park and Petrus Molefe Eco-Park. Venter *et al.* (2012) found similar results where an inverse relationship between temperature and CO concentrations existed due to increased household fuel burning activity. In this study the concentration of CO appears to be sensitive to changes in ambient temperature. Trees have the ability to influence temperature through the provision of shade and evapotranspiration processes (Nowak, 2000). The relationship between CO and temperature at the two parks as indicated in this study implies that trees have the potential to be indirectly beneficial by influencing ambient temperatures.

4.2.5. Sulphur Dioxide (SO_2)

Ambient air quality monitoring of SO_2 was undertaken at both Thokoza Park and Petrus Molefe Eco-Park over the winter season during campaign three. Similar to CO, SO_2 concentrations recorded at both the urban park types were very low and fell well below the South African 24-hour standard of 48 ppb, the hourly standard of 134 ppb and the 10 minute standard of 191 ppb. Daily and hourly average concentrations recorded at the urban background site also did not exceed the relevant South African ambient air quality standards for SO_2. The lack of exceedances of the SO_2 short term standards and the low concentrations of SO_2 recorded at both the urban parks and the urban background site suggest that SO_2 is not necessarily an air pollutant of concern in Soweto in comparison to PM_{10} and $PM_{2.5}$ which exceeded the relevant standards on several occasions over the winter season in this study (Fenger, 1999).

No distinct difference in SO_2 concentrations between the two urban park types was found. Very similar concentrations within the low ranges were detected at both Thokoza Park and Petrus Molefe Eco-Park. Slightly higher SO_2 concentrations were recorded at Thokoza Park however this coincided with an increase in background SO_2 concentrations. Lam *et al.* (2005) found similar results and showed that the SO_2

concentrations at several different urban parks located within Hong Kong were relatively low and did not differ significantly amongst the different urban park types or from the urban background. Similar findings were also found by Yin *et al.* (2011) who detected comparable outside and inside urban park SO_2 concentrations and showed that the concentrations of SO_2 at the urban parks were more representative of urban background concentrations.

The similar concentrations measured at both the urban parks could suggest that the older trees with well-developed tree canopies at Thokoza Park may not be more efficient at removing SO_2 compared to the younger trees at Petrus Molefe Eco-Park. The rates of SO_2 removal through stomatal uptake by trees might also not be high enough for short term field measurements conducted in this study to detect influences on local ground level SO_2 concentrations. Model applications by Paoletti (2011) and Yang *et al.* (2005) illustrate the lowest rates of removal of SO_2 as well as CO by various urban trees in comparison to other key air pollutants such as PM_{10} and O_3.

In contrast to the findings of Lam *et al.* (2005) and Yin *et al.* (2011), higher concentrations of SO_2 were, however observed at the urban background site. The higher concentrations of SO_2 observed at the urban background site may be due lower relative humidity conditions at the site. Higher relative humidity conditions were recorded at the two urban parks in comparison to the urban background site. Evapotranspiration processes due to the urban trees and the Klipspruit wetland likely could account for the higher relative humidity conditions at the two urban parks (McPherson *et al.,* 1994; Souch *et al.,* 1998; Nowak, 2000; Leung *et al.,* 2011). Subsequently, lower concentrations of SO_2 could have resulted due to the greater availability of the hydroxyl radical OH^- for reaction with SO_2 under higher relative humidity conditions (Elminir, 2005; Harrison *et al.,* 2014). In this respect the urban trees at Thokoza Park and Petrus Molefe Eco-Park could indirectly have a positive impact on SO_2 concentrations through evapotranspiration processes. On the other hand, the Diepkloof monitoring station was located in close proximity to an industrial area compared to the two urban parks. Higher emissions of SO_2 from nearby industrial activity could account for higher concentrations of SO_2 observed at the background site.

In this study a relationship between temperature and SO_2 concentrations seemed to have existed at the two urban parks and the urban background site. Two distinct increases in the daily average SO_2 concentrations were observed from the 6 – 7 June 2014 at Petrus Molefe Eco-Park and 19 – 21 June 2014 at Thokoza Park. A distinct increase in urban background SO_2 concentrations was also observed for the same time periods. The sudden rise in SO_2 concentrations coincided with a rapid decrease in ambient temperature over these periods. An increase in SO_2 may have been attributed to an increase in domestic biomass burning for warming purposes. This is suggested by the findings of Yang *et al.* (2005) who found that an increase in SO_2 concentrations during the winter season corresponded with a significant decrease in temperature and attributed this to the increased coal burning activity. However, the diurnal variation plots for SO_2 in this study do not show two distinct increases in SO_2 concentrations in the early morning and evening periods which would have been evident if domestic biomass burning were the main emitter of SO_2. Therefore, it is unlikely that increased fuel burning activity in surrounding residential areas due to lower temperatures alone can account for the higher SO_2 concentrations observed from the 6 – 7 June 2014 and 19 – 21 June 2014 (Venter

et al., 2012). However, higher temperatures facilitate oxidation reactions of SO_2. Thus a decrease in SO_2 concentrations during higher temperature conditions could be due to enhanced oxidation and conversion of SO_2 (Xu *et al.,* 2014)

The sudden increase in SO_2 concentrations associated with a decline in temperature may be also be a result of the lowering of the boundary layer height and the strengthening of the inversion layers (Xu *et al.,* 2014). Inversion layers are common over the Highveld during the winter season and are often associated with increased air pollutant concentrations due to limited dispersion and dilution of the pollutants (Ryu *et al.,* 2012; Tyson and Preston-Whyte, 2013). The rapid decline in temperature could have facilitated the lowering of the boundary layer height and the strengthening of the inversion layers due to reduced convectional uplift and turbulence subsequently resulting in higher concentrations of SO_2 (Elminir, 2005; Xu *et al.,* 2014).

The diurnal variation plots for SO_2 for the period 14 – 22 June 2014 further supports the significant influence of inversion layers and boundary layer height on ground level SO_2 concentrations. For the period 14 – 22 June 2014, maximum SO_2 concentrations at both the urban background site and at Thokoza Park were observed between 7:00 – 11:00 which is possibly due to the break-up of the inversion layers resulting in higher ground level SO_2 concentrations. As the inversion layers break up they release pollutants that have accumulated between the inversion layers throughout the evening and morning periods (Venter *et al.,* 2012). The significant rise in SO_2 concentrations in the late morning period suggests that industrial sources such as metallurgical industries, foundries, smelters, etc. may be significant contributors of SO_2 emissions impacting on the study area (Elminir, 2005; Venter *et al.,* 2012; Harrison *et al.,* 2014). Emissions from industrial activities are generally released from stacks and thus can be released at heights above the inversion layer. Industrial stack emissions also have the potential to travel longer distances thus industrial sources located further away from Soweto could have an impact on SO_2 concentrations observed at the study site. Venter *et al.* (2012) observed the same diurnal pattern in SO_2 concentrations at Marikana in Rustenburg which is located near to a well-developed industrial area (Industrialised Western Bushveld Igneous Complex).

On the other hand, diurnal variation in urban background SO_2 concentrations for the period 28 May – 12 June 2014 showed an increase in concentrations in both the early morning and evening periods with maximum concentrations observed in the later morning period. This implies that a combination of industrial emitters and ground level emitters (vehicle exhaust emissions/domestic fuel burning) were contributing to SO_2 emissions. The persistence of inversion layers during the evening and early morning periods will limit the dispersion and dilution potential of ground level SO_2 emissions, which would result in higher SO_2 concentrations in the morning and evening periods (Elminir, 2005). For the same period (28 May – 12 June 2014), a less pronounced late morning peak in SO_2 concentrations was observed at Petrus Molefe Eco-Park in comparison to Thokoza Park. The slightly different diurnal variations observed between the two sampling periods and the two urban parks could be attributed to differing prevailing wind directions observed during these periods, thus transporting emissions from different downwind sources (Elminir, 2005; Xu *et al.,* 2014).

An increase in morning SO_2 concentrations at the urban parks also showed a slight lag time compared to the Diepkloof monitoring station. The prevailing wind direction for the morning time period, 04:00 – 10:00, differed between the parks and the background site. The Diepkloof monitoring station is also located in close proximity

to industrial areas compared to the urban parks. The slight lag time may be representative of the movement of different SO_2 plumes transporting SO_2 from different downwind emissions sources (Xu *et al.,* 2014). The earlier increase in SO_2 concentrations at the Diepkloof monitoring station in the morning period could also be due to the close proximity of the Diepkloof monitoring station to industrial areas (Venter *et al.,* 2012).

The results for SO_2 obtained in this study present an interaction amongst meteorological factors (i.e. temperature, relative humidity and wind direction), boundary layer heights and inversion layers as well as different emission source types. The interaction amongst these factors have been shown to influence ground level SO_2 concentrations at Thokoza Park and Petrus Molefe Eco-Park.

4.2.6. Carbon Dioxide (CO_2)

An unexpected difference in concentrations of CO_2 was observed between the two urban park types over the spring season (campaign two) where significantly higher concentrations were recorded at Thokoza Park compared to Petrus Molefe Eco-Park. In theory, urban trees with well-developed tree canopies would act as a notable sink for CO_2 as they are able to remove and sequestrate CO_2 via stomatal uptake during photosynthesis and transpiration processes (Nowak, 2002; Jim and Chen, 2009; Escobedo *et al.,* 2011). Therefore, it would have been expected that lower concentrations of CO_2 would be associated with Thokoza Park compared to Petrus Molefe Eco-Park due to the presence of older trees with well-developed and larger tree canopies.

In an urban environment the density and continuous nature of emission sources could outweigh the ability for a few scattered urban trees to act as a significant sink for air pollutants such as CO_2 on a small local scale and over a short term period (Escobedo *et al.,* 2008; Litschke and Kuttler, 2008; Contini *et al.,* 2012; Vos *et al.,* 2013). In this respect the main Chris Hanni road and the Thokoza Bus Rapid Transit station could account for higher concentrations of CO_2 observed at Thokoza Park as vehicle exhaust emissions are a key source of CO_2 during the combustion of carbon containing fuels (Harrison *et al.,* 2014). Although the two urban parks are situated near to each other (~ 500m), the close proximity of vehicle exhaust emissions to Thokoza park compared to Petrus Molefe Eco-Park may explain for higher ambient concentrations of CO_2 at Thokoza Park (Kumar and Nagendra, 2015). This observation agrees with the findings of Lam *et al.* (2005) who found that concentrations of air pollutants were 2 – 22% higher at the edge of several parks which were situated closer to road vehicle exhaust emissions. Similar findings were also found by Yin *et al.* (2011) who showed that there was an overall decrease in pollutant concentrations as a function of distance from the roadside to the interior of the urban park.

However, the diurnal plots for CO_2 concentrations measured at both the urban parks did not show an increase in CO_2 in the morning and evening periods. If traffic activity was a dominant source of CO_2 at Thokoza Park a clear increase in the morning and evening CO_2 concentrations would have been observed due to increased traffic activity during these periods (Kumar and Nagendra, 2015). The constant diurnal pattern in CO_2 at both urban parks therefore suggest that CO_2 concentrations are more representative of background conditions (Lietzke and Vogt, 2013). As sampling at the parks was not conducted over the same period, the significant

increase in CO_2 concentrations at Thokoza Park could be representative of an increase in background CO_2 over the different sampling periods.

On the other hand, atmospheric stability is also a key factor that influences ambient CO_2 concentrations Lietzke and Vogt, 2013). During campaign two significantly calmer winds were recorded at Thokoza Park compared to Petrus Molefe Eco-Park. Calmer wind conditions at Thokoza Park could allow for slightly more stable conditions in which CO_2 could accumulate. Therefore, higher concentration of CO_2 recorded at Thokoza Park could also be attributed to calmer wind conditions, thus reducing the dilution potential of CO_2 at Thokoza Park (Lietzke and Vogt, 2013; Kumar and Nagendra, 2015).

Several studies have shown that differences in tree characteristics (i.e. tree age and expected life span, tree size, tree canopy cover, height of tree, leaf characteristics, tree diameter, tree biomass, etc.) and subsequently, differences in tree species can significantly influence removal rates of air pollutants (Nowak *et al.,* 2002; Yang *et al.,* 2005; Alonso *et al.,* 2011; Paoletti *et al.,* 2011; Bamniya *et al.,* 2012; Wang and Lin, 2012). The difference in the tree ages between Thokoza Park (older trees) and Petrus Molefe Eco-Park (younger trees) may be a factor accounting for the difference in CO_2 concentrations. Nowak *et al.* (2002) found that tree life span together with tree size and growth rate were the most important factors influencing net CO_2 uptake. They found that faster growth rates accounted for quicker CO_2 uptake in comparison to trees with slower growth rates. Younger trees have faster growth rates in comparison to older trees. It is a possibility that the faster growth rates associated with the younger trees at Petrus Molefe Eco-Park could account for lower CO_2 concentrations observed over the short term monitoring periods. However, on a small scale in an urban setting the difference in ambient CO_2 concentrations detected in short term field measurements are more likely to be explained by changes in background conditions, variations in meteorological conditions and differences in contributions of emissions sources (Contini *et al.,* 2012).

It should be emphasised though that the older trees at Thokoza Park may be able to sequestrate more CO_2 due to increased tree biomass, larger tree size and tree canopy size which may not be noticeable through short term field measurements of CO_2 (Nowak *et al.,* 2002; Wang and Lin, 2012). Even though higher tree growth rates are associated with quicker CO_2 uptake, over the long term trees that are larger in size and can live for longer periods of time will sequestrate more CO_2 in the long run compared to smaller trees with a shorter lifespan (McPherson *et al.,* 1994; Nowak *et al.,* 2002). Therefore, tree size and lifespan is significant factors influencing overall net CO_2 uptake regardless of tree age and growth rate (McPherson *et al.,* 1994; Nowak *et al.,* 2002; Wang and Lin, 2012).

On a different note both aquatic and terrestrial ecosystems are able to sequestrate CO_2 therefore the Klipspruit wetland, which is located directly next to Petrus Molefe Eco-Park, may be of significance. Wetlands have the ability to absorb CO_2 in a similar way to that of terrestrial ecosystems. Studies have shown that in some cases freshwater microorganisms have the ability to sequestrate CO_2 more efficiently (10 - 50%) compared to terrestrial plants (Kativu, 2011). Therefore, CO2 sequestration by freshwater microorganisms in the natural Klipsruit wetland as well as the younger trees at Petrus Molefe Eco-Park may act as a notable sink for CO_2 in the short term.

This difference in CO_2 concentrations observed between the two park types provides room for further research. Through model applications the net CO_2 sequestration by trees and the wetland could be compared between the wetland and the two urban park types identified in this study. Furthermore, additional long term ambient air quality monitoring is needed for CO_2. There are limited data available for urban background CO_2 concentrations in Johannesburg and thus the concentrations of CO_2 recorded in this study could not be compared against urban background concentrations.

Ambient monitoring of CO_2 is undertaken by SAWS at Cape Point in Cape Town. The air quality monitoring results indicate concentrations of CO_2 within the range of 390 - 400 ppm for 7 - 31 October 2013 which is the same period of monitoring as that of campaign two in this study (www.saaqis.org.za, 2014). Concentrations of CO_2 measured at the urban parks over campaign two were in the range of 250 -361 ppm, which is lower than that shown at Cape Point. Although these concentrations are not necessarily comparable it does still provide an indication of background concentrations of CO_2 in South Africa. The Cape Point station is considered a background station as it is not located in close proximity to any cities or towns and thus is not influenced by any nearby anthropogenic emission sources.

Higher CO_2 concentrations at Cape Point could be attributed to a source of CO_2 from the near-shore coastal eco-system. Oceans are both a source and sink of CO_2 (Majorek, 2007). A release of larger amounts of CO_2 from the ocean could account for higher concentrations of CO_2 recorded at Cape Point for the period 7 – 31 October 2014 (Majorek, 2007). On the other hand, CO_2 has a long atmospheric life span and is able to accumulate in the atmosphere and travel over long distances. Therefore, higher concentrations of CO_2 measured at the Cape Point station could be due to long range transport of CO_2 from a wider variety of sources across South Africa or other areas (Aalto *et al.,* 2015).

Several key anthropogenic sources of CO2, such as vehicle emissions, industries, landfills, etc. would significantly influence local CO_2 concentrations in the COJ. The concentrations of CO_2 at the parks are most likely representative of local contributions of CO_2 emission sources and not regional background concentrations. Ambient CO_2 concentrations on a local scale will vary significantly due to variability in local meteorological conditions (i.e. wind speed and direction, atmospheric stability, atmospheric pressure, height of mixing layer, etc.) and anthropogenic emission sources (Lietzke and Vogt, 2013; Kumar and Nagendra, 2015). It is also possible that over the short sampling period, meteorological conditions at the parks were favorable for increased dilution and dispersion of CO_2 thus accounting for lower CO_2 concentrations.

5. Chapter Five: Summary and Conclusions

Anthropogenic activities in the City of Johannesburg are key contributors of air pollutants such as SO_2, PM_{10} and $PM_{2.5}$, CO, NO_x, O_3 and CO_2 to name a few. Poor air quality is not evenly distributed over Johannesburg but is rather localised in several areas recognised as hotspot areas. According to the South African Department of Environmental Affairs, Soweto is identified as an air pollution hot spot area which is characteristic of poor air quality where ambient air pollutant concentrations frequently exceed the South African air quality standards (South Africa, 2008).

Urban greening programmes are seen as a way for cities to work towards reducing air pollution, offsetting GHG emissions and improve ambient air quality (Coder, 1996; Nowak *et al*., 2000; Nowak, 2002; Yang *et al*., 2005; Jim and Chen, 2009; Escobedo *et al*., 2011). Urban trees improve air quality, mainly through temperature reduction, microclimate regulation and air pollutant removal processes (Nowak, 2002; Jim and Chen, 2009; Leung *et al*., 2011).

The City of Johannesburg initiated the Greening Soweto project in 2006 where many degraded open spaces were transformed into urban green parks and 6000 trees were planted. The urban parks and trees are believed to serve several environmental benefits; one of which includes the improvement in local ambient air quality. However, there has been no air quality monitoring conducted on a small scale inside the urban parks of Soweto to investigate if the air quality is relatively "*cleaner"* compared to the surrounding urban environment through these initiatives. Much of the research is based on international modelling studies with little justification of the modelling simulation results. It is in this respect that this study was undertaken to assess the local ambient air quality situation at two different urban park types, Thokoza Park (older trees) and Petrus Molefe Eco-Park (young trees) in Soweto and establish whether the air pollutants measured at the urban parks were lower compared to the urban background conditions. This research also examined the differences in air pollutant concentrations between two urban park types by comparing older treed and younger treed sites in close proximity. Furthermore, this study assessed whether ambient concentrations of the selected criteria air pollutants (CO, NO_2, O_3, SO_2, PM_{10} and $PM_{2.5}$) were within the South African National Air Quality Standards for the period of monitoring. The following conclusions are made based on the findings of this research work.

5.1. Difference in Ambient Concentrations of Air Pollutants and Meteorological Conditions between Thokoza Park and Petrus Molefe Eco-Park.

A significant difference in air pollutant concentrations was found for particulate matter. Higher PM_{10} concentrations were associated with Thokoza Park throughout both monitoring campaigns. Mean PM_{10} concentrations of approximately 43µg/m^3 and 134µg/m^3 were recorded at Thokoza Park for campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. Mean PM_{10} concentrations of approximately 24µg/m^3 and 69µg/m^3 were recorded at Petrus Molefe Eco-Park for campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. On the other hand, higher $PM_{2.5}$ concentrations were associated with Petrus Molefe Eco-Park. Mean $PM_{2.5}$ concentrations of approximately

24µg/m^3 and 49µg/m^3 were recorded at Thokoza Park for campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. Mean $PM_{2.5}$ concentrations of approximately 40µg/m^3 and 93µg/m^3 were recorded at Petrus Molefe Eco-Park for campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively.

The pollution rose plots for PM_{10} and $PM_{2.5}$ suggest that the transport of PM_{10} and $PM_{2.5}$ from different downwind sources, due to different prevailing wind directions over the different sampling periods, is likely to explain the significant difference in concentrations observed between the two urban park types throughout this study (Cavanagh *et al.,* 2009). Surrounding tailings facilities, vehicle dust emissions from unpaved roads and windblown dust from small agricultural holdings are most likely key emission sources of PM_{10} at Thokoza Park. Vehicle exhaust emissions and domestic biomass burning activities are most likely notable emission sources of $PM_{2.5}$ at Petrus Molefe Eco-Park.

A significant difference in CO_2 concentrations was also found between the two urban park types where much higher concentrations of CO_2 were unexpectedly recorded at Thokoza Park. In campaign two (8 – 30 October 2013) mean CO_2 concentrations of 297 ppm and 354 ppm were recorded at Petrus Molefe Eco-Park and Thokoza Park respectively. Thokoza Park is characteristic of an older treed site consisting of well-developed tree canopies. Therefore, it was expected that lower concentrations of CO_2 would have been recorded at Thokoza Park in comparison to Petrus Molefe Eco-Park. The difference in concentrations is likely attributed to the close proximity of Thokoza Park to the Chris Hanni main road and the Rea Vaya bus station, as vehicle and bus exhaust fumes are key emission sources of CO_2. However, the diurnal plots also suggest that the CO_2 concentrations recorded at the urban parks are more representative of background concentrations. As sampling was not conducted over the same time period, the difference in CO_2 concentrations may be due to a difference in background CO_2 concentrations over the two sampling periods.

On the other hand the Klipsruit wetland, located directly next to Petrus Molefe Eco-Park, could represent an efficient sink for CO_2. A combination of the closer proximity of Thokoza Park to the Rea Vaya Bus Station and the Chris Hanni main road and CO_2 sequestration by the Klipspruit wetland might account for the lower CO_2 concentrations observed at Petrus Molefe Eco-Park. Nevertheless the older, larger trees at Thokoza Park have the potential to sequestrate more CO_2 over the long term compared to the younger, smaller trees at Petrus Molefe Eco-Park due to increased tree biomass, larger tree size and tree canopy size which may not be noticeable through short term field measurements of CO_2 (Nowak *et al.,* 2002; Wang and Lin, 2012). However, this study was limited in that monitoring was conducted over short campaigns and monitoring could not be conducted at the two parks over the same time period. Therefore the difference in CO_2 concentrations between the two parks could be due to changes in the background CO_2 concentrations.

No significant differences in O_3, CO, SO_2 and NO_x concentrations were found between the two parks with only slight differences detected. Slightly higher concentrations of O_3 were recorded at Petrus Molefe Eco-Park during both campaigns which suggests that the older trees at Thokoza Park possibly could represent a sink for O_3 through efficient stomatal uptake (Nowak *et al.,* 2000; Azmi *et al.,* 2010; Alonso *et al.,* 2011). During the spring season (8 – 30 October 2013) mean O_3 concentrations of approximately 22 ppb and 19.5 ppb were recorded at Petrus Molefe Eco-Park and Thokoza Park respectively. During the winter season (28 May – 22

June 2014) mean O_3 concentrations of approximately 10.4 ppb and 9.5 ppb were recorded at Petrus Molefe Eco-Park and Thokoza Park respectively.

The lower temperatures recorded at Thokoza Park during the spring season proposes that the trees at Thokoza Park could reduce photo-chemical production rates forming O_3 through the provision of shade and evapotranspiration processes therefore accounting for slightly lower concentrations of O_3 at Thokoza Park (Saini *et al.,* 2008; Paolleti, 2009). However, Petrus Molefe Eco-Park was associated with slightly higher temperatures during the spring season and slightly higher relative humidity conditions during the winter season. Higher temperature and relative humidity conditions facilitate photo-chemical production rates of O_3 and thus could account for higher O_3 concentrations at Petrus Molefe Eco-Park during both campaigns (Elminir, 2005; Saini *et al.,* 2008). As sampling at the two parks could not be conducted over the same time period, the slightly higher O_3 concentrations recorded at Petrus Molefe Eco-Park is likely due to temporal differences in temperature and relative humidity conditions between the two parks. The very similar concentration plots and diurnal patterns between the two urban parks suggests that the O_3 concentrations at both parks were subject to similar surrounding influences including precursor gases and meteorological conditions.

The difference in CO concentrations between the two parks was found to be insignificant with only a slight difference observed. Slightly higher concentrations of CO were recorded at Petrus Molefe Eco-Park during the winter season. The difference in CO concentrations between the two urban parks is likely due to the fact that sampling was conducted over different time frames. Slightly warmer temperatures were recorded at Thokoza Park during the winter season, which was mainly attributed to an increase in background temperatures. Warmer temperatures are associated with higher photo-oxidation rates of CO which could account for slightly lower CO concentrations observed at Thokoza Park (Elminir, 2005; Harrison *et al.,* 2014). A strong relationship between CO concentrations and temperature was evident at the urban parks. Furthermore, a strong diurnal pattern was observed in CO concentrations at the parks. An increase in the concentration was observed in the morning and evening periods. This implies that domestic biomass burning practices are likely key emission sources of CO in the area (Venter *et al.,* 2012).

Similar to CO, the difference in SO_2 between the two park types was very small and considered negligible. Slightly higher concentrations of SO_2 recorded at Thokoza Parks is most likely attributed to a slight increase in the background SO_2 concentrations. The diurnal variation plots for SO_2 suggest that industrial activities are the main emission source impacting on SO_2 at the two urban parks (Venter *et al.,* 2012).

A distinct difference in NO_x concentrations was not observed between the two park types either. Higher concentrations of NO_x were recorded at Thokoza Park during the spring season and lower concentrations were recorded at Thokoza Park during the winter season. The difference in NO_x concentrations was mostly attributed to general changes in background NO_x concentrations. The findings in this study showed that the concentrations of NO_x at the urban parks had a strong relationship with O_3. This implies that the trees have the potential to influence O_3 concentrations. Trees act as a key sink for NO_x (Nowak *et al.,* 2006; Betts *et al.,* 2008). Lower NO_x concentrations could reduce day time photo-chemical production rates of O_3 as the availability of NO_2 for reaction decreases (Wuytack *et al.,* 2010; Yin *et al.,* 2011). However at night time, higher

O_3 concentrations could occur as the availability of NO_x for reaction decreases thus reducing the destruction of O_3. Trees are a source of water vapour through evapotranspiration processes. High humidity conditions to facilitate the production of hydroxyl radicals, which is key during O_3 formation as well as titration (Zhang *et al.,* 2005; Jose *et al.,* 2005).

Notable differences in meteorological parameters, including wind fields and temperature, were found between the two urban park types. Much calmer winds were observed at Thokoza Park compared to Petrus Molefe Eco-Park. At Thokoza Park calm winds (≤ 1m/s) occurred for 67% and 72% of the time during campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. At Petrus Molefe Eco-Park calm winds occurred less frequently for 6% and 31% of the time during campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. This provides an indication of the interaction between the air mass and the tree canopy as the wind moves over and through the trees. Dry depositional processes of air pollutants occur as the air mass interacts with the tree canopy. Characteristics of the tree canopy including, aerodynamic roughness, tree structure and configuration and leaf and tree surface properties will influence the dry deposition of particles (Litschke and Kuttler, 2008; Pullman, 2009; Mensink *et al.,* 2012). Dry deposition of gaseous pollutants will mostly depend on the efficiency of stomatal conductance which will increase as a function of leaf surface area, stomata size and stomata density. Nevertheless, calmer winds will increase the exposure period in which gaseous and particulate air pollutants can interact with the tree canopy and thereby facilitate dry deposition of air pollutants (Nowak *et al.,* 2000; Nowak, 2006). On the other hand, in an urban environment calmer winds may account for localised higher concentrations of pollutants by limiting the dispersion and dilution of emissions, particularly in the case of ground level emitters such as vehicle exhaust emissions (Litschke and Kuttler, 2008).

A difference in temperature was observed between the two urban parks. In the spring season lower ambient temperatures were recorded at Thokoza Park, while during the winter season slightly warmer temperatures were recorded at Thokoza Park compared to Petrus Molefe Eco-Park. Mean temperatures of 16.9 ^oC and 19.2 ^oC were recorded during the spring season (8 – 30 October 2013) at Thokoza Park and Petrus Molefe Eco-Park respectively. Contrastingly, during the winter season (28 May – 22 June 2014) mean temperatures of 9.4 ^oC and 9 ^oC were recorded at Thokoza Park and Petrus Molefe Eco-Park respectively. These findings suggest that the older trees with well-developed tree canopies located at Thokoza Park could reduce temperatures during the spring season through the provision of shade and evapotranspiration processes. This can have a positive impact on air quality by decreasing photochemical and oxidation reactions of air pollutants. In the winter season the trees could limit heat loss which could also have a positive impact on air quality by facilitating the dispersion of ground level air pollutants (Nowak, 2000; Cavanagh *et al.,* 2009; Leung *et al.,* 2011; Yin *et al.,* 2011). However, the temperature differences between the urban parks and background site were very small. It is likely that the difference in temperature between the two parks could be due to the difference in sampling time frames and changes in background temperatures.

5.2. Difference in Ambient Concentrations of Criteria Air Pollutants at Thokoza Park and Petrus Molefe Eco-Park Compared to the Urban Background Site.

Very similar meteorological conditions (i.e. relative humidity, temperature, no precipitation and wind fields) were observed between the urban parks and the urban background site. However, differences in the concentrations of particulate matter, O_3, NO_x, CO and SO_2 were found between the urban parks and the Diepkloof monitoring station. Generally, lower concentrations of air pollutants were observed at the urban parks in comparison to the urban background site, particularly during the spring season.

Higher concentrations of O_3 were recorded at the urban background site in comparison to the urban parks. Mean differences of -16 ppb and -21 ppb in O_3 concentrations were observed between Petrus Molefe Eco-Park and the urban background site during campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. Mean differences of -13 ppb and -27 ppb in O_3 concentrations were observed between Thokoza Park and the urban background site during campaigns two (8 – 30 October 2013) and three (28 May – 22 June 2014) respectively. The similar surrounding influences, both meteorological and precursor gases, suggest that the trees at the parks potentially represent an O_3 sink. However, higher concentrations of precursor gases (namely CO and NO_x) were also recorded at the Diepkloof station which could have accounted for the higher concentrations of O_3 at the urban background site (Saini *et al.,* 2008). Furthermore, the Diepkloof monitoring station is located in close proximity to an industrial area. VOC emissions from the industrial area could also account for higher O_3 concentrations at the urban background site (Zhang *et al.,* 2004).

Seasonal differences in NO_x concentrations were observed between the two urban parks and the urban background site. Higher concentrations of NO_x were recorded at the Diepkloof station during the spring season, while lower concentrations were recorded at the Diepkloof station during the winter season. Mean differences of -21 ppb and 25 ppb in NO_x concentrations were observed between Petrus Molefe Eco-Park and the urban background site during spring (8 – 30 October 2013) and winter (28 May – 22 June 2014) respectively. Mean differences in NO_x concentrations of -16 ppb in the spring season (8 – 30 October 2013) and 22 ppb in the winter season (28 May – 22 June 2014) were observed between Thokoza Park and the urban background. These findings suggest that the parks may act as a notable sink for NO_x during the spring season (Yang *et al.,* 2005; Betts *et al.,* 2008; Yin *et al.,* 2011). Slightly higher humidity conditions observed at the urban parks during the spring season could have facilitated the photo-oxidation of NO_x which could have also accounted for lower NO_x concentrations (Elminir, 2005; Saini *et al.,* 2008). During the winter season, faster winds recorded at the Diepkloof station could have facilitated the dispersion and dilution of NO_x thus accounting for lower NO_x concentrations at the urban background site during winter (Elminir, 2005; Saini *et al.,* 2008). Furthermore, higher humidity conditions and less photo-oxidation of NO_2 at the parks during the winter season could have facilitated the formation NO_2 (Zhang *et al.,* 2005; Jose *et al.,* 2005).

On the other hand the findings in this study also suggest that domestic biomass burning emissions may have a greater impact on NO_x concentrations at the two parks in comparison to the urban background site. Vehicle exhaust emissions appeared to have been a key source of NO_x at the urban background site. Therefore, less domestic biomass burning during the spring season could have accounted for the lower NO_x concentrations

observed in the parks whereas during the winter season the contribution of domestic biomass burning in addition to vehicle exhaust emissions may have accounted for higher NO_x concentrations observed in the urban parks (Venter *et al.*, 2012).

Slightly higher concentrations of CO were observed at the urban background site compared to the urban parks during the third monitoring campaign. A mean difference of -0.45 ppb in CO concentrations for the period 28 May – 22 June 2014 was observed between Petrus Molefe Eco-Park and the urban background site. Similarly, a mean difference of -0.82 ppb in CO concentrations for the period 28 May – 22 June 2014 was observed between Thokoza Park and the urban background site. Vehicle exhaust emissions appeared to be a key source of CO at the urban background site. Higher concentrations of CO observed at the urban background site compared to the urban parks were probably due to the continuous supply of emissions from vehicles throughout the day with slight increases during peak traffic times (Azmi *et al.,* 2011). The differences in CO concentrations and diurnal patterns between the urban parks and the urban background site suggests that domestic biomass burning may have a greater impact on CO concentrations at the parks compared to the urban background site. This is indicated by a strong inverse relationship between temperature and CO concentrations observed at the parks, but not at the urban background site. A distinct increase in CO concentrations coincided with a decrease in temperature during the winter season, which corresponds with surrounding residential areas burning more fuel for heating purposes as the temperature decreases (Azmi *et al.,* 2011; Venter *et al.,* 2012).

Higher concentrations of SO_2 were observed at the urban background site compared to the urban parks. Higher humidity conditions at the urban parks due to evapotranspiration processes from the wetland and the urban trees could explain the lower SO_2 concentrations as the availability of the hydroxyl radical for oxidation of SO_2 increases (Elminir, 2005). On the other hand, the Diepkloof monitoring station is located in close proximity to industrial areas compared to the parks. Higher emissions of SO_2 due to nearby industrial activity could also account for higher SO_2 concentrations observed at the background site. Furthermore, warmer temperatures were observed at the urban parks compared to the Diepkloof station during the winter season. Higher temperatures facilitate oxidation reactions of SO_2 and could account for lower SO_2 concentrations observed at the urban parks (Xu *et al.,* 2014).

The diurnal plots for SO_2 for the urban parks and the urban background site suggest that industrial emitters are the main source of SO_2 impacting the area. This is indicated by a late morning peak in SO_2 concentrations (Venter *et al.,* 2012; Xu *et al.,* 2014). There were also slight variances in the diurnal plots between the two parks and the background site. This was found to be mainly attributed to variations in the prevailing wind directions and possible downwind transport of emissions from different downwind sources (Lam *et al.,* 2005; Yin *et al.,* 2011; Xu *et al.,* 2014). The lag period between the SO_2 peaks at the parks and the Diepkloof station also provides an indication of the movement of different SO_2 plumes (Venter *et al.,* 2012; Xu *et al.,* 2014).

Significantly higher concentrations of particulate matter (PM_{10} and $PM_{2.5}$) were observed at both urban parks in comparison to the urban background site during the winter season (28 May – 22 June 2014). A higher frequency of calmer winds at both the urban parks in comparison to the urban background site could have

reduced the dispersion and dilution of particulate matter which may have resulted in higher concentrations of PM_{10} and $PM_{2.5}$ (Lam *et al.,* 2005; Litschke and Kuttler, 2008). However, this was only evident when the prevailing wind directions at the urban parks and the Diepkloof station were very similar. In campaign two, when the prevailing wind direction at Diepkloof station differed significantly to the urban park, higher concentrations of particulate matter were observed at the urban background site. Downwind emissions of particles from the tailings located in close proximity to Diepkloof station could explain for the higher concentrations observed at the background site for the period 8 -30 October 2013.

5.3. Comparison of the Ambient Concentrations of Criteria Air Pollutants at Thokoza Park and Petrus Molefe Eco-Park against the South African National Ambient Air Quality Standards.

Ambient concentrations of O_3, CO, CO_2, NO_x, SO_2, PM_{10} and $PM_{2.5}$ recorded at Thokoza Park (older trees) and Petrus Molefe Eco-Park (younger trees) were analysed for the period 8 - 30 October 2013 and 28 May - 22 June 2014. Out of all the criteria air pollutants measured, PM_{10} and $PM_{2.5}$ were the only pollutants that exceeded the South African National Ambient Air Quality Standards at Thokoza Park and Petrus Molefe Eco-Park. Exceedances of the standards were only observed during the winter season. Winter meteorological conditions are likely to account for the exceedances observed as there is less wet deposition of particles, lower humidity conditions, lower soil moisture content, cooler temperatures and the persistence of inversion layers limiting the dispersion and dilution of atmospheric particles (Lovett, 1994; Tyson and Preston-Whyte, 2013). Increased domestic biomass burning is likely to also contribute to higher particulate matter concentrations during the winter season (Kornelius et al., 2012).

The ambient concentrations of O_3, CO and SO_2 were low at both the urban parks and fell well below the South African National Ambient Air Quality Standards throughout this study. The concentrations of NO_x at Thokoza Park and Petrus Molefe Eco-Park did not exceed the applicable South African National ambient air quality standard, but were shown to be relatively high in comparison to the other gaseous air pollutants monitored in the study. A maximum hourly NO_x concentration of 92 ppb was recorded at Petrus Molefe Eco-Park during the spring season, which fell just below the air quality standard of 106 ppb. Furthermore, eight hour average O_3 concentrations at the Deipkloof monitoring station exceeded the eight hour standard of 61 ppb during campaign two and three. This also suggests that O_3 may be a pollutant of concern around the Diepkloof area.

The findings of this study suggest that particulate matter and NO_x are two criteria air pollutants of concern at Thokoza Park and Petrus Molefe Eco-Park in comparison to the other criteria air pollutants monitored in this study. The pollution rose plots and diurnal variation plots for PM_{10} and $PM_{2.5}$ suggest that surrounding tailings facilities, small agricultural holdings, domestic biomass burning and vehicle dust emissions on paved and unpaved roads as well as vehicle exhaust emissions may be key emissions of particulate matter at the two urban parks. Diurnal variation in NO_x concentrations, slight changes in the concentrations of NO_x over the weekends and as a function of temperature suggests that domestic biomass burning practices and vehicle exhaust emissions may be key emission sources of NO_x at the two urban parks.

5.4. Summary

In summary, the findings of this study suggest that the urban trees have the greatest potential to improve air quality in Soweto mainly through changes in local meteorological conditions, specifically for temperature and wind fields, rather than direct removal of air pollutants. Differences in the concentrations of the air pollutants at the different sites showed a strong relationship with changes in temperature, wind speed and direction and emission source types.

A significant difference in air pollutant concentrations between the two urban park types was only found for particulate matter (PM_{10} and $PM_{2.5}$) and CO_2. Differences in the prevailing wind direction over the different sampling periods appeared to be an important factor influencing the PM_{10} and $PM_{2.5}$ concentrations recorded at the two urban parks types. The difference in PM_{10} and $PM_{2.5}$ concentrations under differing wind directions emphasises the significance of differing downwind emissions sources impacting on local particulate matter concentrations. Surrounding tailings, dust emissions from paved and unpaved roads and windblown dust from small agricultural holdings surrounding the parks are potentially key emission sources of PM_{10}. Domestic biomass burning and vehicle exhaust emissions are potential key emissions sources of $PM_{2.5}$. The differences in particulate matter concentrations between the urban background site and the parks indicated that the surrounding emissions sources have the potential to have a localised impact.

The difference in CO_2 concentrations between the urban parks suggested that the distance between the urban park and the main road might be an important factor influencing CO_2 concentrations. Furthermore, the Klipspruit wetland possibly represents an efficient CO_2 sink. However, the diurnal plots don't show the influence of traffic related activities and further suggest that CO_2 concentrations at the urban parks are more representative of background concentrations over the different sampling periods.

The air quality at the two urban parks appeared to be generally better compared to the urban background site. This is most likely due to differing surrounding influences of emissions sources. However, the results of this study do suggest that the urban trees could act as a potential sink for O_3 during the spring and winter seasons and for NO_x during the spring season. The findings also suggest that local changes in meteorological conditions, namely humidity, induced by the trees and the wetland could also account for lower concentrations of pollutants through increased occult depositional processes and increased oxidation rates. Furthermore, the significantly calmer winds observed at Thokoza Park provides an indication of the interaction between the air mass and the tree canopies. Deposition of pollutants occurs as the air mass interacts with tree components. Therefore, enhanced deposition of pollutants could result in lower pollutant concentrations and have an overall positive impact on air quality over the long term. Lastly, particulate matter and NO_x were identified to be air pollutants of concern at the parks in comparison to the other criteria air pollutants assessed in this study. Maximum concentrations were recorded during the winter season. Particulate matter, NO_x and O_3 were identified to be key air pollutants of concern at Diepkloof.

This study was limited by financial resources and equipment and thus only allowed for an indirect comparison between the two urban park types over short term field measurements, since only one site could be monitored at any given time. Nevertheless the findings presented in this study still provide an indication of the various

interactions amongst the urban trees, different air pollutants, meteorological conditions and emissions sources and their subsequent influence on local ambient air quality in the two urban parks in Soweto. Key air pollutants of concern are identified and potential emission sources at Thokoza Park and Petrus Molefe Eco-Park is also acknowledged. The findings of this study provide room for further research. The information provided in this study can be used to define future air quality management procedures in Soweto as well as assisting the City of Johannesburg in successfully implementing the use of urban trees in mitigating air quality issues in Soweto.

6. Reference List

Aalto, T., Hatakka, J., Kouznetsov, R. and Stanislawska, K., 2015: Background and anthropogenic influences on atmospheric CO_2 concentrations measured at Pallas: comparison of two models for tracing air mass history, *Boreal Environment Research*, 20, 213-226.

Akbari, H., Pomerantz, M. and Taha, H., 2001: Cool surfaces and shade trees to reduce energy use and improve air quality in urban areas, *Solar Energy,* 70(3), 295–310.

Alonso, R., Vivanco, M.G., Gonzalez-Fernandez, I., Bermejo, V., Palomino, I., Garrido, J.L., Elvira, S., Salvador, P. and Artinano, B., 2011:Modelling the influence of peri-urban trees in the air quality of Madrid region (Spain), *Environmental Pollution*, 159, 2138-2147.

Amato, G.D., Cecchi, L., Amato, M.D. and Liccardi, G., 2010: Urban Air Pollution and Climate Change as Environmental Risk Factors of Respiratory Allergy: An Update, *Journal of Investigational Allergology and Clinical Immunology*, 20(2), 95-102.

Athanassiadou, M., Baker, J., Carruthers, D., Collins, W., Girnary, S., Hassell, D., Hort, M., Johnson, C., Johnson, K., Jones, R., Thomson, D., Trought, N. and Witham, C., 2010: An assessment of the impact of climate change on air quality at the two UK sites, *Atmospheric Environment*, 44, 1877-1886.

Azmi, S.Z., Latif, M.T., Ismail, A.S., Juneng, L., Jemain, A.A., 2010: Trend and status of air quality at three different monitoring stations in the Klang Valley, Malaysia, *Air Quality Atmospheric Health*, 3, 53-64.

Bamniya, B.R., Kapoor, C.S., and Kapoor, K., 2012: Searching for efficient sink for air pollutants: studies on *Mangifera indica L*, *Clean Technologies and Environmental Policy*, 14, 107-114.

Ball, S., n.d.: Atmospheric chemistry at night, *ECG Environmental Briefs*, ECGEB No.3.

Beckett, K.P., Freer-Smith, P.H. and Taylor, G., 1998: Urban woodlands: their role in reducing the effects of particulate pollution, *Environmental Pollution,* 99, 347-360.

Beckett, K.P., Freer-Smith, P.H. and Taylor, G., 2000: Particulate pollution capture by urban trees: effect of species and wind speed, *Global Change Biology*, 6, 995-1003.

Betts, R., Sanderson, M. and Woodward, S., 2008: Effects of large scale Amazon forest degradation on climate and air quality through fluxes of carbon dioxide, water, energy, mineral dust and isoprene., *Philosophical Transactions Royal Society Biological Sciences*, 363, 1873-1880.

Beukes, J.P., Vakkari, V., van Zyl, P.G., Venter, A.D., Josipovic, M., Jaars, K., Tiitta, P., Kulmala, M., Worsnop, D., Pienaar, J.J., Virkkula, A. and Laakso, L., 2013: Source region plume characterisation of the interior of South Africa as observed at Welgegund, *National Association for Clean Air, The Clean Air Journal,* 23(1), 7-10.

Bloomfield, J., and Pearson, H.L., 2000: Land use, land-use change, forestry, and agricultural activities in the clean development mechanism: Estimates of greenhouse gas offset potential, *Mitigation and Adaptation Strategies for Global Change*, 5, 9-24.

Bolund, P. and Hunhammar, S., 1999: Ecosystem services in urban areas, *Ecological Economics,* 29, 293–301.

Brauer, M., Hoek, G., Van Vliet, P., Meliefste, K., Fischer, P.H., Wijga, A., Koopman, L.P., Neijens, H.J., Gerritsen, J., Kerkhof, M., Heinrich, J., Bellander., T. and Brunekreef., B., 2002: Air pollution from traffic and the development of respiratory infections and asthmatic and allergic symptoms in children, *American Journal of Respiratory and Critical Care Medicine*, 166, 1092-1098.

Buccolieri, R., Salim, M.S., Leo, L.S., Sabatino, S.D., Chan, A., Lelpo,P., de Gennaro, G. and Gromke, C., 2011: Analysis of local scale tree atmosphere interaction on pollutant concentration in idealized street canyons and application to a real urban junction, *Atmospheric Environment,* 45, 1702-1713.

Cavanagh, J.A.E, Reza, P.Z., and Wilson, J.G., 2009: Spatial attenuation of ambient particulate matter air pollution within an urbanised native forest patch, *Urban Forestry and Urban Greening*, 8, 21-30.

Chen, R., Pan, G., Zhang, Y., Xu, Q., Zeng, G., Xu, X., Chen, B. and Kan, H., 2011: Ambient carbon monoxide and daily mortality in three Chinese cities: The China Air Pollution and Health Effects Study (CAPES), *Science of the Total Environment,* 409, 4923-4928.

City of Johannesburg, 2008: *State of the air report*
Available from:
{http://www.joburg.org.za/index.php?option=com_content&task=view&id=3959&Itemid=114.}
(Accessed 01 January 2014)

City of Johannesburg, 2009: *176 Air Quality-Joburg*.
Available from: {http:// www.joburg-archive.co.za/2009/pdfs/report.../enviro-air-quality.pdf.}
(Accessed 24 February 2013)

Clarke, J., 2002: Coming back to earth-South Africa's changing environment, Jacana (pty) Ltd, Houghton.

Coder, R.D., 1996: *Identified Benefits of Community Trees and Forests*, University of Georgia, United States of America.

Contini, D., Donateo, A., Elefante, C. and Grasso, F.M., 2012: Analysis of particles and carbon dioxide concentrations and fluxes in an urban area: correlation with traffic rate and local micrometeorology, *Atmospheric Environment*, 46, 25-35.

Eastwood, P., 2008: *Particulate Emissions from Vehicles*, John Wiley & Sons, West Sussex England.

Elminir, H.K., 2005: Dependence of urban air pollutants on meteorology, *Science of the Total Environment*, 350, 225-237.

Elsom, D., 1996: *Smog alert: Managing urban air quality*, Earthscan Publications Limited, London.

Escobedo, F.J., Kroeger, T., and Wagner, J.E., 2011: Urban forests and pollution mitigation: Analyzing ecosystem services and disservices, *Environmental Pollution*, 159, 2078-2087.

Escobedo, F.J., Wagner, J.E., Nowak, D.J., De la Maza, C.L., Rodriguez, M., and Crane, D.E., 2008: Analyzing the cost effectiveness of Santiago, Chile's policy of using urban forests to improve air quality, *Journal of Environmental Management*, 86, 148-157.

Escobedo, F.L., and Nowak, D.J., 2009: Spatial heterogeneity and air pollution removal by an urban forest, *Landscape and Urban Planning*, 90, 102-110.

Feig, G., Ncipha, X., Vertue, B., Naidoo, S., Mabaso, D., Ngcukana, N., Tshehla, C. and Masuku, N., 2014: Analysis of a period of elevated ozone concentrations reported over the Vaal Triangle on 2 June 2013, *National Association for Clean Air, The Clean Air Journal*, 24 (1), 10-16.

Fenger, J., 1999: Urban air quality, *Atmospheric Environment,* 33, 4877-4900.

Government Gazette, 2009: *Department of Environmental Affairs: National Environmental Management: Air Quality Act, 2004 (Act No. 39 of 2004): National Ambient Air Quality Standards*, Report No. 32816, Government Printer, Pretoria.

Gromke, C. and Ruck, B., 2010: Interaction of traffic pollutant dispersion with trees in urban street canyons, in: Proceedings of the CLIMAQS Workshop 'Local Air Quality and its Interactions with Vegetation' 21-22 January 2010, Antwerp Belgium, 93 - 97.

Harris, T.B. and Manning W.J., 2010: Nitrogen dioxide and ozone levels in urban tree canopies, *Environmental Pollution*, 158, 2384-2386.

Harrison, R.M., Pope, F.D., and Shi, Z., 2014: Air pollution, *Earth Systems and Environmental Sciences*, 1-17.

Held, G., Gore, B.J., Surridge, A.D., Tosen, G.R., Turner, C.R., and Walmsley, R.D., 1996: *Air Pollution and Its Impacts On The South African Highveld*, Environmental Scientific Association, Cleveland.

Hill, M.K., 2010: *Understanding Environmental Pollution*, Cambridge University Press UK, United Kingdom.

HORIBA, 2004_a: *Ambient O_3 monitor, APOA-370, Operation Manual*, Code GZ0000051248A.

HORIBA, 2004_b: *Ambient CO monitor, APMA-370, Operational manual*, Code GZ0000051249A.

HORIBA, 2004_c: *Ambient NO_x monitor, APNA-370, Operational manual,* Code GZ9100497232B.

Hueglin, C., Gehrig, R., Baltensperger, U., Gysel, M., Monn, C. and Vonmont, H., 2005: Chemical characterisation of PM2.5, PM10 and coarse particles at urban, near city and rural sites in Switzerland, *Atmospheric Environment,* 39, 637-651.

IPCC, 2013: *Climate Change 2013*. *The Physical Science Basis. Contributions of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Stocker, T.F., Quin, D., Plattner, G.K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex V. and Midgley, P.M.* (eds.)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Jacob, D.J. and Winner, D.A., 2009: Effect of Climate Change on air quality, *Atmospheric Environment*, 43, 51-63.

Jim, C.Y. and Chen, W.J., 2007: Assessing the ecosystem service of air pollutant removal by urban trees in Guangzhou (China), *Journal of Environmental Management,* 88, 665–676.

Jim, C.Y., and Chen, W.Y., 2009: Ecosystem services and valuation of urban forests in China, *Cities*, 26, 187-194.

Jose, R.S., Stohl, A., Karatzas, K., Bohler, T., James, P. and Perez, J.L., 2004: A modelling study of an extraordinary night time ozone episode over Madrid domain, *Environmental modelling and software,* 20(2005), 587-593.

Kagawa, M. and Ishizaka, Y., 2014: Conversion of SO_2 to particulate sulfate during transport from Chine to Japan, Assessment by Selenium in aerosols, *Aerosol and Air Quality Research*, 14, 269-279.

Kampa, M. and Castanas, E., 2008: Human health effects of air pollution, *Environmental Pollution*, 151, 362-367.

Kan, H., Chen, R. and Tong, S., 2012: Ambient air pollution, climate change, and population health in China, *Environmental International*, 42, 10-19.

Kativu, E., 2011: *Carbon dioxide absorption using fresh water algae and identifying potential uses of algal biomass*, University of the Witwatersrand, Johannesburg, South Africa.

Katsouyanni, K., Touloumi, G., Spix, C., Schwartz, J., Balducci, F., Medina, S., Rossi, G., Wojtyniak, B., Sunyer, J., Bacharova, L., Schouten, J.P., Ponka, A. and Anderson, H.R., 1997: Short term effects of ambient sulphur dioxide and particulate matter on mortality in 12 European cities: results from time series data from APHEA project, *BMJ*, 314, 1658-1663.

Khoder, M.I., 2002: Atmospheric conversion of sulphur dioxide to particulate sulfate and nitrogen dioxide to particulate nitrate and gaseous nitric acid in an urban area, *Chemosphere*, 49(2002), 675-684.

Kornelius, G., Kruger, S., Fouchee, R. and van Wyk, H., 2012: A wood gasification stove for domestic use: Design, performance and emission factors, *National Association of Clean Air*, *The Clean Air Journal,* 22(2), 14-16.

Kumar, M.K. and Nagendra, S.M.S., 2015: Characteristics of ground level CO_2 concentrations over contrasting land uses in a tropical urban environment, *Atmospheric Environment*, 115, 286-294.

Lam, K.C., Ng, S.L., Hui, W.C. and Chan, P.K., 2005: Environmental quality of urban parks and open spaces in Hong Kong, *Environmental Monitoring and Assessment*, 111, 55-73.

Last, J.A., Sun, W.M. and Witschi, H., 1994: Ozone, NO, and NO2: Oxidant air pollutants and more, *Environmental Health Perspectives,* 102(10), 179-184.

Leung, D.Y.C., Tsui, J.K.Y., Chen, F., Yip, W.K., Vrijmoed, L.L.P., and Liu, C.H., 2011: Effects of Urban Vegetation on Urban Air Quality, *Landscape Research*, 34(2), 173-188.

Litschke, T. and Kuttler, W., 2008: On the reduction of urban particle concentration by vegetation – a review, *Meteorologische Zeitschrift,* 17(3), 229-240.

Lietzke, B. and Vogt, R., 2013: Variability of CO_2 concentrations and fluxes in and above an urban street canyon, *Atmospheric Environment*, 74, 60-72.

Lovett, G.M., 1994: Atmospheric deposition of nutrients and pollutants in North America: an ecological perspective, *Ecological Applications*, 4, 629-650.

Lydia, A.O., 2010: *Characteristics of particulate matter over the South African industrialised Highveld*, University of the Witwatersrand, South Africa.

Majorek, S., 2007: *The coastal ocean: A source or sink of atmospheric CO_2*?, Institute of Biogeochemistry and Pollutant Dynamics, Department of Environmental Sciences, Zurich.

McDonald, J.H., 2014: *Handbook of Biological Statistics, Third edition,* Sparky House Publishing, Baltimore, Maryland.

McNamara, M.L., Noonan, C.W. and Ward, T.J., 2011: Correction factor for continuous monitoring of wood smoke fine particulate matter, *Aerosol and Air Quality*, 11, 315-322.

McPherson, E.G., Nowak, D.J. and Rowntree, R.A., 1994: *Chicago's urban forest ecosystem: results of the Chicago urban forest climate project*, *General Technical report NE – 186*, United States Department of Agriculture, Forest Service, Northeastern Forest Station, United States.

Mensink, C., de Maerschalck, B., Maiheu, B., Janssen, S. and Vankerkom, J., 2012: Chapter 3: The role of vegetation in local and urban air quality, *Air Pollution Modelling and its Application XXI,* 15-20.

Met One Instruments, Inc., 2008: *BAM 1020 Particulate Monitor, Operational Manual,* BAM-1020-9800*,* Washington Boulevard, America.

Mrubata, M., Balmer, M. and Kruger, T., 2008: I*mpact of the removal of paraffin flame stoves from the market*, Programme for Basic Energy and Conservation (ProBEC), AFRECA.
Available from: {http://www.active.cput.ac.za/energy/past_papers/DUE/2008?PDF/Paper%20-%20Mrubata%20M.pdf.}
(Accessed 01 January 2014).

Naidoo, S., Piketh, S.J. and Curtis, C., 2014: Quantification of emissions generated from domestic burning activities from townships in Johannesburg, *National Association for Clean Air*, *The Clean Air Journal*, 24 (1), 34-40.

Nowak, D.J., Civerolo, K.L., Rao, S.T., Sistla, G., Luley, C.J., and Crane, D.E., 2000: A modeling study of the impact of urban trees on ozone: *Atmospheric Environment*, 34, 1601-1613.

Nowak, D.J., 2002: *The effects of urban trees on air quality*, USA Forest Service, New York.

Nowak, D.J., Crane, D.E., and Stevens, J.C., 2006: Air pollution removal by urban trees and shrubs in the United States, *Urban Forestry and Urban Greening*, 4, 115-123.

Nowak, D.J., 2006: Institutionalizing urban forestry as a biotechnology to improve environmental air quality, Urban Forestry and Urban Greening*,* 5, 93-100.

Nuwarinda, H., 2007: *Air pollution study of a Highveld township during A Basa Njengo Magogo rollout*, Department of Geography, Environmental Management and Energy Studies, University of Johannesburg, South Africa.

Ocak, S. and Turalioglu, F.S., 2008: Effect of meteorology on the atmospheric concentrations of traffic related pollutants in Erzurum, turkey, *International Journal of Environmental Application and Science*, 3(5), 325-335.

Oguntoke, O., Ojelede, M.E. and Annegarn, H.J., 2013: Frequency of mine dust episodes and the influence of meteorological parameters on the Witwatersrand area, South Africa, *Hindawi, International Journal of Atmospheric Sciences*, 2013, Article ID 128463.

Paoletti, E., 2009: Ozone and urban forests in Italy, *Environmental Pollution,* 157, 1506-1512.

Paoletti, E., Bardelli, T., Giovannini, G. and Pecchioli, L., 2011: Air quality impact of an urban park over time, *Procedia Environmental Sciences,* 4, 10-16.

Peel, J.L., Haeuber, R., Garcia, V., Russell, A.G. and Neas, L., 2013: Impact of nitrogen and climate change interactions on ambient air pollution and human health, *Biochemistry,* 114, 121-134.

Pullman, M.R., 2009: *Conifer PM2.5 deposition and re-suspension in wind and rain events*, *Cornell University*, America.

Rasheed, A., Aneja, V.P., Aiyyer, A. and Rafique, U., 2014: Measurements and analysis of air quality in Islamabad Pakistan, *Earths Future*, 2, 303-314.

Ravishankara, A.R., Dawson, J.P. and D.A. Winner, 2012: New directions: Adapting air quality management to climate change: a must for planning, *Atmospheric Environment*, 50, 387-389.

Reddy, B.S.K., Kumar, K.R., Balakrishnaiah, G., Gopal, K.R., Reddy, R.R., Sivakumar, V., Lingaswamy, A.P., Arafath, S.M., Umadevi, K., Kumari, S.P., Ahammed, Y.N. and Lal, S., 2012: Analysis of diurnal and seasonal behaviour of surface ozone and its precursors (NOx) at a semi-arid rural site in southern India, *Aerosol and Air Quality Research,* 12, 1081-1094.

Rosenfeld, A.H., Akbari, H., Romm, J.J. and Pomerantz, M., 1998: Cool communities: Strategies for heat island mitigation and smog reduction, *Energy and Buildings,* 28, 21-62.

Ryu, Y.H., Baik, J.J., Kwak, K.H., Kim, S., and Moon, N., 2012: Impacts of urban land-surface forcing on air quality in Seoul metropolitan area, *Atmospheric chemistry and physics discussions*, 12, 25791-25832.

Saini, R., Satsangi, G.S. and Taneja, A., 2008: Concentrations of surface O3, NO2 and CO during winter seasons at semi-arid region - Agra, India, *Indian Journal of Radio & Space Physics*, 37, 121-130.

Scholes, R.J., and Van de Merwe, M.R., 1996: Greenhouse gas emissions from South Africa, *South African Journal of Science*, 92, 220-222.

Seinfeld, J.H. and Pandis, S.N., 2006: *Atmospheric Chemistry and Physics*: *From Air Pollution to Climate Change*, John Wiley and Sons, Inc., Hoboken, New Jersey

Shirinde, J., Wichmann, J. and Voyi, K., 2014: Association between wheeze and selected air pollution sources in an air pollution priority area in South Africa: a cross-sectional study, *Environmental Health: A Global Access Science Source*, 13, 32.

Souch, C., Susan, C., Grimmond, B. and Wolfe, C.P., 1998: Evapotranspiration rates from wetlands with different disturbance histories: India dunes National lakeshore, *Wetlands,* 18(2), 216-229.

South Africa, 2003: *Air Quality Management Plan for the City of Johannesburg*, Department of Development Planning, Transportation and Environment and the Department of Environmental health, Report No. MTX/03/JHB-01d, Government Printer, Pretoria.

South Africa, 2008: *Executive Summary of the Vaal Triangle Airshed Priority Area Air Quality Management Plan*, Department of Environmental Affairs and Tourism, report No. 31615, Government printer, Pretoria.

South African weather Services, 2014: *Air quality monitoring: Plot air quality data on graphs, carbon dioxide*.
Available from: {http://www.saaqis.org.za/AQLineGraph.aspx.}
(Accessed 30 November 2014)

Squizzato, S., Masiol, M., Brunelli, A., Pistollato, S., Tarabotti, E., Rampazzo, G. and Pavoni, B., 2013: Factors determining the formation of secondary inorganic aerosol: a case study in the Po Valley (Italy), *Atmospheric Chemistry and Physics*, 13, 1927-1939.

Stolzenbach, K.D., 2006: *Atmospheric Deposition*
Available from: {http://www.environment.ucla.edu/reportcard/article1497.html.}
(Accessed 01 January 2014)

Tasic, V., Milosevic, N., Kovacevic, R. and Petrovic, N., 2010: The analysis of air pollution caused by particle matter emission from the copper smelter complex Bor (Serbia), *Chemical Industry and Chemical Engineering Quarterly,* 16(3), 219-228.

Thermo Electron Corporation Environmental Instruments$_a$, 2004: *Model 43C, Pulsed fluorescence SO_2 analyzer, Instruction manual*, P/N 100100-00, Massachusetts, America.

Thermo Electron Corporation Environmental Instruments$_b$, 2004: *Model 41C, High level gas filter correlation CO_2 analyzer, Instruction manual*, P/N 100041-00, Massachusetts, America.

Trail, M., Tsimpidi, A.P., Tsigarisis, K., Rudokas, J., Miller, P., Nenes, A., Hu, Y. and Russell, A.G., 2014: Sensitivity of air quality to potential future climate change and emissions in the United States and major cities, *Atmospheric Environment*, 94, 552-563.

Tyson, P.D. and Preston-Whyte, R.A., 2013: *The weather and Climate of Southern Africa*, Oxford University Press, New York.

Unknown, 2012: *Greening Soweto-transforming dustbowls and landfill sites to award winning parks and eco-services, Johannesburg City Parks.*
Available from: {http://www.jhbcityparks.com/pdfs/greening-soweto12.pdf.}
(Accessed 24 February 2013)

Unknown, 2014: *Atmospheric Deposition 101*
Available from: {http://www.epa.gov/air/oaqps/gr8water/handbook/airdep-sept-2.pdf.}
(Accessed 01 January 2014)

Venter, A.D., Vakkari, V., Beukes, J.P., Van Zyl, P.G., Laakso, H., Mabaso, D., Tiitta, P., Josipovic, M., Kuimala, M., Pienaar, J.J., and Laakso, L., 2012: An air quality assessment in the industrialised western Bushveld Igneous Complex, South Africa, *South African Journal of Science*, 108, 84-94.

Vos, P.E.J., Maiheu, B., Vankerkom, J., and Janssen, S., 2013: Improving local air quality in cities: To tree or not to tree?, *Environmental Pollution,* 183, 113-122.

Wang, S., Zhang, J., Zeng, X., Zeng, Y. and Chen, S., 2009: Association of traffic related air pollution with children's neurobehavioral functions in Quanzhou, China, *Environmental Health Perspectives*, 117, 1612-1618.

Wang, T., Wei, X.L., Ding, A.J., Poon, C.N., Lam, K.S., Li, Y.S., Chan, L.Y. and Anson, M., 2009: Increasing surface ozone concentrations in the background atmosphere of Southern China, 1994-2007, *Atmospheric Chemistry and Physics*, 9, 6217-6227.

Wang, Y.C. and Lin, J.C., 2012: Air quality enhancement zones in Taiwan: A carbon reduction benefit assessment, *Forest Policy and Economics*, 23, 40-45.

Wang, J., Hu, Z., Chen, Y., Chen, Z. and Xu, S., 2013: Contamination characteristics and possible sources of PM10 and PM2.5 in different functional areas of Shangai, China, *Atmospheric Environment*, 68, 221-229.

World Health Organisation, 2005: *WHO air quality guidelines for particulate matter, ozone, nitrogen dioxide and sulphur dioxide: Global update 2005*, WHO Press, Switzerland.

Wright, C.Y., Matooane, M., Oosthuizen, M.A. and Phala, N., 2014: Risk perceptions of dust and its impacts among communities living in a mining areas of the Witwatersrand, South Africa, *National Association for Clean Air, The Clean Air Journal,* 24 (1), 22-27.

Wu, S., Mickley, L.J., Kaplan, J.O., and Jacob, D.J., 2011:Impacts of changes in land use and land cover on atmospheric chemistry and air quality over the 21st century, *Atmospheric chemistry and physics discussions*, 11, 15469–15495.

Wuytack, T., Verheyen, K., Wuts, K., Kardel, F., Adriaenssens, S. and Samson, R., 2010: The potential of bio monitoring of air quality using leaf characteristics of white willow (Salix alba L.), in: Proceedings of the CLIMAQS Workshop 'Local Air Quality and its Interactions with Vegetation' 21-22 January 2010, Antwerp, Belgium, 36-40.

Xu, W.Y., Zhao, C.S., Ran, L., Lin, W.L., Yan, P. and Xu, X.B., 2014: SO_2 noontime –peak phenomenon in the North China plain, *Atmospheric Chemistry and Physics,* 14, 7757-7768.

Yang, J., McBride, J., Zhou, J., and Sun, Z., 2005: The urban forest in Beijing and its role in air pollution reduction, *Urban Forestry and Urban Greening*, 3, 65-78.

Yin, S., Shen, Z., Zhou, P., Zou, X., Che, S. and Wang, W., 2011: Quantifying air pollution attenuation within urban parks: An experimental approach in Shanghai, China, *Environmental Pollution*, 159, 2155-2163.

Zhang, R., Lei, W., Tie, X. and Hess, P., 2004: Industrial emissions cause extreme urban ozone diurnal variability, *The National Academy of Sciences of the USA*, 101(17), 6346-6350.

Zhou, W., Yuan, D., Ye, S., Qi, P., Fu, C. and Christiani D.C., 2001: Health effects of occupational exposures to vehicle emissions in Shanghai, *International Journal of Occupational Environmental Health,* 7, 23-30.

Zunckel, M., Turner, C.R. and Wells, R.B., 1996: Dry deposition of sulphur on the Mpumalanga Highveld: a pilot study using the inferential method, *South African Journal of Science*, 92, 485-491.