An Assessment of Indoor and Outdoor Air Quality in a University Environment: A Case of University of Limpopo, South Africa

by

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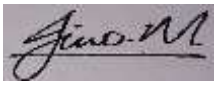
DECLARATION

I, Antony Jino Mundackal (student number: 61953687), registered for this PhD in Environmental Science declare that this thesis titled "**An assessment of indoor and outdoor air quality in a university environment: A case of University of Limpopo, South Africa"** is my own work and that all the sources that I have used or quoted have been indicated and acknowledged by means of complete references. I also declare that the originality of this thesis has been verified by the Turnitin software and is within the conventional guidelines for originality. I further declare that I have not previously submitted this work, or part of it, for examination at Unisa for another qualification or at any other institution for any purpose whatsoever.



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02/03/2020

Date

DEDICATION

This work is dedicated to my late mother, Mrs. Clara Mundackal (Kunjamma), whose inspiration, motivation and undying love made this thesis possible.

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LIST OF ABBREVIATIONS

ANOVA → Analysis of variance

API → Air pollution index

AQ → Air quality

AQI → Air quality index

ASHRAE → American Society of Heating, Refrigerating, and Air Conditioning Engineers

BIOD → Biodiversity

BMBT → Biochemistry, Microbiology and Biotechnology

DEA → Department of Environmental Affairs

EPA → Environmental Protection Agency

FSA → Faculty of Science and Agriculture

hPa → Hectopascals

HVAC → Heating, ventilation and air conditioning

I/O → Indoor:outdoor ratio

IAP → Indoor air pollution

IAPs → Indoor air pollutants

IAQ → Indoor air quality

LAV → Linear air velocity

m/s → meters per second

NAAQS → National Ambient Air Quality Standards

NIEHS → National Institute of Environmental Health Sciences

NMHCs → Non-methane hydrocarbons

NO_x → Nitrogen oxides

OAP → Outdoor air pollution

OAPs → Outdoor air pollutants

OAQ → Outdoor air quality

PEH → Physiology and Environmental Health

PM → Particulate matter

PM_{10} → Particulate matter (10 microns)

$PM_{2.5}$ → Particulate matter (2.5 microns)

ppb → Parts per billion

ppm → Parts per million

RH → Relative humidity

SAWS → South African Weather Service

SMLS → School of Molecular and Life Sciences

SPSS → Statistical Package for the Social Sciences

T_{db} → Dry-bulb temperature

TSP → Total suspended particulate

UL → University of Limpopo

VOCs → Volatile organic compounds

WHO → World Health Organization

ABSTRACT

Air pollution of late has been the focus of many studies due to the detrimental health risks that it poses to individuals. University environments have several academic departments with peculiar activities that could be affecting the indoor and outdoor air quality (AQ) of these environments. University settings differ from other environments because of the variety of activities and different lines of work that go on inside buildings housing academic departments and their surroundings, which are likely to have an impact on indoor air quality (IAQ) and outdoor air quality (OAQ) in this environment. Only a few AQ studies have been done in university sites and surrounds worldwide and in these studies, IAQ was given primary importance; whereas, the outdoor environment was and is often neglected. A study comparing both IAQ and OAQ is critical to further understand the relationship between IAQ and OAQ within a university campus. The University of Limpopo (UL) in the Mankweng township of South Africa has been undergoing some refurbishments with numerous construction activities going on in addition to the academic activities of UL. These activities may be affecting the AQ in this unique environment. The main aim of this study was to determine differences between indoor and outdoor AQ in a university environment and to understand how AQ in this unique environment varies with seasons and building function. The study was carried out in three buildings housing three different academic departments in UL namely: Department of Physiology and Environmental Health (PEH), Department of Biochemistry, Microbiology, and Biotechnology (BMBT) and the Department of Biodiversity (BIOD). Twenty indoor and 20 outdoor measuring sites were identified per departmental building from where real-time measurements of 11 AQ parameters (linear air velocity (LAV), dry-bulb temperature (T_{db}), relative humidity (RH), carbon monoxide (CO), carbon dioxide (CO_2), ozone (O_3), sulphur dioxide (SO_2), nitrogen dioxide (NO_2), hydrogen sulphide (H_2S), non-methane hydrocarbons (NMHCs) and volatile organic compounds (VOCs)) were taken over three consecutive days per season. Thus, a total of 60 indoor and 60 outdoor measurements were taken for each parameter in each of the three buildings of interest per season, leading to 360 measurements per season and 1440 measurement per parameter over the one-year period of study across the study area. A hot-wire anemometer was used to measure LAV, whereas the Q-Trak indoor AQ monitor was used in the measurement of T_{db}, RH, CO and CO_2. Aeroqual AQ monitors were employed in the measurement of O_3, SO_2, NO_2, H_2S, NMHCs and VOCs. The Wilcoxon signed ranks test was used to

determine differences between indoor and outdoor environments. Significant differences were found between the indoor and outdoor environments for LAV (all three buildings), T_{db} (PEH and BMBT), RH (BIOD), O_3 (all three buildings), NO_2 (all three buildings), CO (all three buildings), CO_2 (all three buildings), NMHCs (BMBT and BIOD), and VOCs (all three buildings) ($p < 0.05$). Linear air velocity, O_3, SO_2, CO, CO_2, and H_2S values/concentrations across the indoor/outdoor environments were within the ASHRAE/DEA/WHO guidelines/standards, whereas T_{db}, RH and NO_2 values/concentrations were not. Air quality in the study area varied with building, with the best AQ across both the indoor and outdoor environments being within the BIOD building, whilst the worst AQ across both environments was encountered in the PEH building. Seasonal differences between buildings were also identified between indoor and outdoor environments among the PEH, BMBT and BIOD buildings ($p < 0.008$). Across the indoor environment, the winter season was found to be the season with the best AQ, since all the pollutants were found at minimum concentrations. Factors affecting AQ in the study area included thermal comfort, occupant densities, building function, laboratory emissions, renovation activities, generators, vehicular emissions, among others. The best AQ across the outdoor environment occurred during the autumn season, since all the air pollutants were present at minimal concentrations during this time. The best predictors of LAV, T_{db}, CO, CO_2, NO_2, and NMHCs were seasons ($R^2 = 1.000$, $p < 0.01$). For the parameters RH, H_2S, and VOCs, the best predictor was building type ($R^2 = 1.000$, $p < 0.01$). The indoor and outdoor environment were the best predictors for SO_2 ($R^2 = 0.999$, $p < 0.01$). Ozone had no single predictor that was found to significantly influence its concentration in this study. In relation to an air pollution index (API), generally all pollutant indices fell within the fair, good to very good range when using mean and maxima concentrations, whereas, corresponding NO_2 concentrations throughout the study fell within the poor to very poor range (105.660–250.000). University management should take into consideration ventilation in laboratories, occupant densities and location of standby generators and car parks in the management of AQ on the university campus. All heating, ventilation and air conditioning (HVAC) systems need to be upgraded and work in tandem with natural ventilation when having high occupant densities within buildings. Future studies in this sector could incorporate larger sample sizes, be designed as a longitudinal study, and make use of questionnaires and sample more AQ parameters to get a detailed understanding of a university site and its surrounds.

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CHAPTER ONE
INTRODUCTION

1.1. Introduction

This chapter presents information on the background of this study, the problem statement, the aim, objectives and hypotheses for this study. Thereafter the rationale and the significance of carrying out a study of this nature is deliberated upon. The chapter ends with a conclusion that summarizes what was discussed in the various sections that make up the chapter.

1.2. Background

There are several factors that interact to impact on environmental pollution. The combined influence of economic and population growth results in an increased demand for supplies such as food, water, and energy. These, subsequently lead to an upsurge in the number of pollutants in the environment, which is a major concern worldwide due to their potentially damaging effects on large areas and the resultant human health implications (Leung, 2015; Hsu *et al*., 2013). Economic growth is associated with a deterioration of atmospheric quality (Luo *et al*., 2014). Air pollution may result from economic growth and expansion due to natural resource exploitation (Luo *et al*., 2014; Yang *et al*., 2012). Taghvaee and Parsa (2015) documented the relationship between high income and high energy consumption which accompanies economic growth with the added deterioration in the air quality (AQ) of an environment.

Air pollution can be described as the presence of a mixture of harmful, natural and anthropogenic substances in the atmosphere, and it is typically differentiated into two categories, that is, indoor and outdoor air pollution (OAP) (NIEHS, 2017). Indoor air pollution (IAP) involves the presence of carbon oxides, particulates and other indoor air pollutants (IAPs) within a building, whereas OAP describes air pollution which takes place outside the built environment (NIEHS, 2017). The World Health Organization (WHO) ascertained that exposure to air pollution was responsible for approximately eight million deaths worldwide in 2012, 4.3 million occurring from exposure to polluted or contaminated indoor air, and the remaining 3.7 million from diseases emanating from polluted outdoor air (Vegter, 2016). Earlier findings from WHO, pinpointed urban air pollution as being responsible for more than two million mortality cases per year in developing countries.

WHO (2002) established that both IAP and OAP caused damaging health impacts and were responsible primarily for a greater burden of disease. This was further supported by Dholikia *et al*. (2013) and Gulia *et al.* (2015) who found both IAP and OAP contribute to millions of deaths worldwide.

Outdoor air pollution can be influenced by location, season and the pollution source in an area (ELF, 2013), including emissions from vehicles, manufacturing or power generating facilities and the burning of fossil fuels for various purposes (WHO, 2017). Outdoor air pollutants (OAPs) include ozone (O_3), sulphur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO), carbon dioxide (CO_2) and particulate matter (PM) (10 microns in diameter (PM_{10}) and 2.5 microns in diameter ($PM_{2.5}$)) (WHO, 2017; Leung, 2015). The concentrations of these pollutants determine the quality of outdoor air. Outdoor air pollution is also linked to adverse health outcomes including high mortality rates, increased rates of hospital admissions due to chronic respiratory infections such as asthma and unhealthy lung functioning (Huang *et al*., 2009).

Indoor air pollution can be affected by the surrounding outdoor pollution levels. Studies have shown that pollutant concentrations are much higher in the indoor environment compared to an outdoor environment due to the closed nature of the environment and lack of enough air to dilute the pollutants (Alhakbani and Kanjo, 2013). Moreover, inadequate airflow rates, cleaning products and chemicals emanating from building materials and furnishings could also contribute to rising levels of IAP (Jovanovic *et al*., 2014). While completely sealing buildings may address energy efficiency concerns, they are problematic in that they may in turn result in elevated levels of indoor pollutants, subsequently, bringing about health risks (Spiru and Simona, 2017). However, outdoor pollutants can infiltrate the indoor environment through open windows, doors and even ventilation intakes (Wangchuk *et al*., 2015).

Indoor air quality can be referred to as the AQ in the nearest vicinity in and around buildings and structures, specifically attributed to the health, comfort, and well-being of building occupants (Badea *et al*., 2015; USEPA, 2015a). Most individuals spend their time indoor, including, homes, offices, schools and universities (Lee and Chan, 1998). Consequently, IAQ is imperative for the maintenance of human health, comfort, and well-being (Di Giulio *et al*., 2010). According to Klepeis *et al*. (2001), people on average spend

87 % of their time indoors. Spiru and Simona (2017) and Al horr *et al*. (2016) also established that individuals spend 80 to 90 % of their time in an enclosed environment. A substantial percentage of total personal exposure to air pollution therefore occurs in the indoor setting, implicating poor IAQ as a cause of health complications (Habermann *et al*., 2015; WHO, 2002). Indoor air quality has become an area of focus among researchers and the public due to the surplus amount of time spent by an individual in this environment (Na and Cocker*,* 2005; Klepeis *et al*., 2001) in addition to the associated occupational risks.

Generic factors that are known to affect IAQ are generally pollution sources across the outside environment, poorly constructed and maintained buildings, mismanaged cleaning schedules and improper natural ventilation provisioning (Jovanovic *et al*., 2014). Indoor air quality is also influenced by physical and biological factors, as well as chemical agents. Physical factors emanate from both natural and human sources and include temperature, humidity, ventilation rates, air movement (wind speed and direction), lighting (natural and artificial), noise, ionizing and non-ionizing radiation (Badea *et al*., 2015; Shendell, 2011; Di Giulio *et al*., 2010). Biological factors mainly refer to microorganisms such as bacteria (inclusive of metabolised products and cell-wall constituents), mould (spores, mildew, and fungi) and pollen (originating from plants, flowers, and trees) (Badea *et al*., 2015; Shendell, 2011). These biotic factors may contribute to the accumulation of pollutants such as volatile organic compounds (VOCs) originating from numerous metabolic pathways (Shendell, 2011). Chemical agents influencing IAQ can be categorised as organic and inorganic (Shendell, 2011) and include tobacco smoke, furniture, paints, cosmetics, carpets and drapes, examples of which include: VOCs, formaldehyde and carbon oxides (Di Giulio *et al*., 2010). Innumerable studies have related other indoor factors like building dampness, ventilation flow rate, microbial and mould contamination, dry-bulb temperature (T_{db}) and relative humidity (RH) with issues like sick building syndrome (Lu *et al*., 2016). Ventilation has been reported to play a major part in IAQ. Jurado *et al*. (2014) found that rooms which had natural ventilation provisioned for, had a higher concentration of mould than air-conditioned rooms. Airborne dust concentrations were in exceedance in both types of rooms, and CO_2 levels were almost three times more in air-conditioned rooms as compared to naturally ventilated rooms. Houri *et al*. (2009) established that in non-ventilated rooms, CO_2 levels exceeded levels specified by standards; whereas, in fan-

ventilated rooms, the CO_2 levels dropped to below or borderline levels, highlighting the importance of ventilation in relation to IAQ, within closed environments.

University environments differ from other environments because of the variety of activities that go on inside buildings within the university and its surroundings. These activities are likely to have an impact on IAQ and OAQ in a university environment. In a study conducted by Budiakova (2017) in Slovak University, issues relating to thermal comfort, lack of ventilation, unacceptable CO_2 levels, difficulty in breathing and the lack of concentration came to the fore with regard to IAQ. Alves *et al*. (2013) reported high-temperature values in IAQ studies within universities. Yau *et al*. (2011) indicated that only one of the six lecture halls investigated within the University of Malaya, in Kuala Lumpur, had thermal conditions conforming to the American Society of Heating, Refrigerating, and Air Conditioning Engineers (ASHRAE) guidelines. Yau *et al*. (2011), additionally found RH values in half of the lecture halls to be in exceedance of the ASHRAE guideline, whilst the air velocity in one of the six lecture halls was deemed to be below the ASHRAE guideline. Kalwasinska *et al*. (2012) pointed out that bacteria and mould fungi levels attained in a university library in Poland were consistent with levels expected in areas such as the library, laboratory, reading room, storeroom, main hall cafeteria, and toilet. Most of these studies carried out on AQ looked at IAQ and OAQ within schools, offices, hospitals, houses, and high traffic areas (Cetin and Sevik, 2016). In South Africa, however, no studies which investigate both IAQ and OAQ in tandem, particularly in a university setting, have been conducted. The status quo of the AQ in these environments needs to be investigated to determine whether staff, personnel and students working in these environments are exposed to any health risks.

1.3. Problem statement

Poor IAQ and OAQ are associated with health effects such as increased hospital admission rates, increased emergency visits to hospitals, aggravated respiratory conditions and in some cases, mortality (Huang *et al*., 2009). Research carried out by the National Institute of Environmental Health Sciences (NIEHS) has publicised that continued exposure to air pollutants may increase the prevalence of illnesses, such as asthma, pulmonary diseases and extreme health endpoints such as lung cancer (NIEHS, 2011). There is a need to monitor AQ in any environment to understand the health risks to which occupants are exposed. It has been established that areas which rapidly undergo

transformation and development are more prone to issues pertaining to poor AQ in their immediate surroundings (Leung, 2015). Within the Limpopo province, the township of Mankweng and its surroundings have been rapidly transforming and heading towards a highly-urbanised environment, thereby increasing the potential for air pollution and subsequently, compromising the AQ in the surrounding environments. The University of Limpopo (UL) is situated within Mankweng and its AQ is of interest due to the renovations and infrastructural development initiatives currently being done (Ramoroka *et al*., 2016), and the purpose for which the various buildings are being used. The UL houses a mixture of both old, newly renovated as well as newly constructed buildings. Understanding the indoor and OAQ will provide information on how these renovations are affecting AQ in the university.

Air conditioning within the university has proven to be a major challenge, and the same could be said for buildings conducting different lines of work. Universities, especially old university buildings tend to have incorrect architectural design, old windows and walls which are not sufficiently insulated, inefficiently controlled and regulated heating systems and in most cases are missing mechanical ventilation systems (Budiakova, 2017). Lecture halls have airtight steel-made doors that are equipped with an occasional ventilator fan and are dependent upon artificial ventilation (Houri *et al*., 2009). In UL, however, there is no consistency or standardisation regarding fan ventilation and the usage thereof is left to room users. Studies by Budiakova (2017), Mohammadyan *et al*. (2017), Widder and Haselbach (2017), Lu *et al*. (2016), Yu *et al*. (2015), Seppanen (2007) and Einberg (2005) have highlighted the role of ventilation in IAQ. This research will determine whether IAQ in a university environment varies with the function of a building. One would expect vast differences in IAQ and OAQ in a university environment due to the wide array of academic departments with buildings customised for the various activities performed within the university. University environments are also generally characterised by a high density of students (Budiakova, 2017) and differ from other environments such as homes or factories due to the presence of numerous building types such as lecture halls, offices, meeting, seminar, and conference rooms in a specific area.

Only a few AQ studies (Budiakova, 2017; Alves *et al*., 2013; Yau *et al*., 2011; Jurado *et al*., 2014; Houri *et al*., 2009; Kalwasinska *et al*., 2012; Di Giulio *et al*., 2010) have been done in university environments worldwide. In these studies, precedence was given to

IAQ and not IAQ and OAQ in tandem. Furthermore, only a few parameters were considered as determinants of IAQ in the above-mentioned studies, some of which were: ventilation, linear air velocity (LAV), CO_2, thermal conditions, RH, bacteria and mould fungi. Studies on IAQ and OAQ in university buildings in South Africa are scarce and need to be done to determine the current AQ in these unique environments. It is pertinent that other pollutant concentrations such as O_3, SO_2, CO, and VOCs are taken into consideration since the indoor environment in a university environment is also affected by these chemical factors and not solely by physical factors (Badea *et al*., 2015; Di Giulio *et al*., 2010). A study comparing both IAQ and OAQ is imperative to establish and further delineate the link between IAQ and OAQ in a university environment.

1.4. Aim, objectives, and hypotheses of the study

1.4.1. Aim

The main aim of this study is to determine differences, if any, between the indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa with a view of deciphering the state of air quality within these two environments across buildings performing different functions. In addition, it also aimed at identifying seasonal variations in the indoor and outdoor air quality in these different buildings which house different academic departments.

1.4.2. Objectives

The following specific objectives guided this study

- To determine the levels and/or concentrations of linear air velocity, dry-bulb temperature, relative humidity, ozone, sulphur dioxide, nitrogen dioxide, carbon monoxide, carbon dioxide, hydrogen sulphide, non-methane hydrocarbons and volatile organic compounds in selected buildings within the University of Limpopo;
- To evaluate differences between indoor and outdoor air quality amongst the selected academic buildings in the university;
- To evaluate conformity and compliance of indoor and outdoor air quality with the American Society for Heating, Refrigerating and Air Conditioning Engineers guidelines, the South African Department of Environmental Affairs standards and the World Health Organization standards;
- To evaluate seasonal differences between indoor and outdoor air quality amongst the selected buildings in the university.

- To determine which buildings have the worst air quality among university buildings in a Science faculty.

1.4.3. Hypotheses

H_{01}: There are no differences between indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa.

H_{02}: There are no seasonal variations in indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa.

1.5. Rationale

Epidemiological studies have linked air pollution with numerous human health outcomes (AAP, 2004; WHO, 2008). However, limited quantitative AQ studies have been done in South Africa (Table 1.1). Shirinde *et al*. (2014) investigated air pollution sources and the risk of wheezing. Wichmann and Voyi (2012) investigated pollutant concentrations and their associations with respiratory, cardiovascular and cerebrovascular mortality. Brits (2011) investigated indoor environmental conditions and sick-building syndrome. Lourens *et al*. (2011) focussed on the distribution and progression of some air pollutants within the Highveld region of South Africa. Josipovic *et al*. (2010) investigated the concentrations, dispersal, transport, and exceedance of SO_2, NO_2, and O_3 in South Africa. White *et al*. (2009) concentrated on asthma prevalence and refinery emissions. Maluleke and Worku (2009) surveyed environmental air pollution (smoke) and asthma in children. The multidisciplinary study conducted by Naidoo *et al*. (2007) concentrated on ambient pollutant concentrations and lung function in children. Sanyal and Maduna (2000) focussed on acute respiratory health and outdoor air pollutant measurements. Terblanche *et al*. (1998), Terblanche *et al*. (1994), Terblanche *et al*. (1993) and Terblanche *et al*. (1992) focussed on gaseous pollutants and their health effects. Klopper *et al*. (1988) looked into OAPs and the prevalence of respiratory health. Coetzee *et al*. (1986) assessed the relationship between OAPs and respiratory health and lung function. For all the above-mentioned studies, their respective study area, study design, and results are reflected in Table 1.1 below.

Table 1.1: Some quantitative studies related to air quality conducted in South Africa.

Area of the study	Study design	Result	Reference
Tembisa and Kempton Park (Gauteng).	Cross-sectional.	• Environmental tobacco smoke, truck emissions and paraffin usage for residential heating were all associated with wheezing in children.	Shirinde *et al*., 2014.
Cape Town (Western Cape).	Case-crossover.	• NO_2 was associated with cerebrovascular, respiratory and cardiovascular diseases; • SO_2 was associated with cardiovascular disease mortality; • PM_{10} was associated with respiratory and cardiovascular disease during warm periods.	Wichmann and Voyi, 2012.
Pretoria (Gauteng).	Cross-sectional.	• Insignificant associations between indoor environmental conditions and sick building syndrome were found.	Brits, 2011.
Vanderbijlpark (Gauteng); Delmas, Witbank, Ermelo, Carolina, Amersfoort, Standerton and Balfour (Mpumalanga).	Cross-sectional.	• Highest NO_2 and SO_2 concentrations were found in areas with local origin points; peaks during winter were encountered; however, all concentrations were beneath South African standards across all sites; • O_3 concentrations were miniscule in comparison to corresponding NO_2 and SO_2 maxima; O_3 peaks were found during spring and were also below the South African standards across all sites.	Lourens *et al*., 2011.
Witbank (Mpumalanga).	Cross-sectional.	• SO_2 peaked in winter; • NO_2 peaked during summer, late spring and early autumn; • O_3 peaks were seen during late winter and spring.	Josipovic *et al*., 2010.
Cape Town (Western Cape).	Cross-sectional.	• The prevalence of asthma and associated symptoms were higher in children situated nearby refinery emission points.	White *et al*., 2009.
Polokwane (Limpopo).	Cross-sectional.	• Persistent cough, smoke exposure and lack of access to flush toilets were significant precursors of asthma in children.	Maluleke and Worku, 2009.
South Durban (KwaZulu-Natal).	Cross-sectional and Multi-disciplinary	• Ambient NO_2, nitrogen oxide (NO_x), PM_{10} and SO_2 were associated with decreased lung function in children.	Naidoo *et al*., 2007.

	longitudinal (panel study).		
Eastern Cape.	Prospective cohort.	• Acute respiratory infections were associated with low income households using wood/coal for heating purposes.	Sanyal and Maduna, 2000.
Vaal Triangle (Gauteng and Free State).	Multi-disciplinary longitudinal.	• Upper and lower respiratory tract infections were associated with wood/coal usage for cooking and heating; • Air pollution levels were associated with coughing and phlegm production.	Terblanche *et al*., 1998; Terblanche *et al*., 1994; Terblanche *et al*., 1993 and Terblanche *et al*., 1992.
Western Cape.	Cross-sectional.	• Insignificant associations amidst OAPs and respiratory health outcomes were found.	Klopper *et al*., 1988.
Free State.	Cross-sectional.	• Significant differences in forced expiratory volumes for boys were found; • Lung functioning was affected by air pollution (associated with socio-economic status in girls).	Coetzee *et al*., 1986.

There have been no studies done with respect to AQ within a university setting, amongst different buildings; consequently, highlighting the need to do a study of this nature. Numerous studies address the issue of IAQ in addition to IAP and their detrimental effects (Choo *et al*., 2015; Madureira *et al*., 2015; Kumar *et al*., 2014; Di Giulio *et al*., 2010; Bruce *et al*., 2000); however, in a South African context, a study focussing on IAQ and OAQ amidst buildings, has not been done to ascertain the status quo of AQ and whether differences exist between IAQ and OAQ among buildings, in addition to seasonal variations in AQ between buildings.

Most university personnel spend a greater part of their day within office buildings of the university, including offices, lecture rooms, laboratories, conference and/or seminar rooms (within buildings) and to a lesser extent outside the buildings as well. Even though it is widely believed that outdoor air is more contaminated than indoor air, numerous studies have showed that indoor air pollutant concentrations can be in exceedance of outdoor pollutant concentrations (Chen and Zhao, 2011; Delgado-Saborit *et al*., 2011). Therefore, it is imperative that a study of this nature is done to identify whether differences exist in IAQ and OAQ among university buildings. The buildings within the Faculty of Science and Agriculture (FSA) in UL are predominantly old buildings and are devoid of extractor fans and centrally controlled air conditioning. Each building is unique since it has been constructed specifically for the type of work carried out within a specific school or department. These different buildings could be influencing the AQ of the university environment in unique ways.

1.6 Significance of the study

The results of this study provide insight into the current AQ status within a university setting across both indoor and outdoor environments, buildings and seasons in South Africa. Individuals spend more time indoors than outdoors. In areas where AQ is poor, people have been advised to stay indoors. The general belief is that the indoor environment is safer than the outdoor environment. The results of this study provide information that will indicate whether this is the case in university environments. In addition, the use of different chemicals in various academic departments in a university, may be presenting a health risk to staff and students which is not usually considered in the management of AQ.

The results of this study will present information that is valuable to university facilities managers in the maintenance of a healthy environment for both staff and students in university environments. The university management and various stakeholders will be informed and educated on the overall AQ within the University of Limpopo by this study. Additionally, the university management and other personnel would get a better idea on the exposure risks to which their staff are exposed to, and the overall health risk linked with this exposure. The study also contributes towards the making of decisions on the basis of the air pollution indices determined across the different university environments and its surrounds. Furthermore, AQ management initiatives can be designed on the basis of the current study findings.

1.7 Conclusion

Air quality in a given environment is severely affected and impacted upon by both indoor and outdoor pollutant sources in the environment itself. This chapter has highlighted the gaps that exist in literature with regards to AQ studies especially in South Africa and how this study contributes towards addressing such gaps. The problems that the research addresses, in addition to the main objectives of the study, have been presented.

CHAPTER TWO
LITERATURE REVIEW

2.1 Introduction

This chapter reviews literature on AQ, indoor and OAPs in addition to parameters that determine IAQ and OAQ. Furthermore, issues pertaining to AQ standards are explored in relation to human exposure and exposure pathways to air pollutants.

2.2. Air quality

Air quality can be described as the condition or the quality of air in a specific place and is dependent on the prevalent atmospheric conditions. Air quality is affected by both natural and anthropogenic factors including volcanic eruptions, wind erosion, particulate matter (from dust), biogenic emissions from plants and trees (natural factors), fossil fuel combustion power stations, vehicular and traffic emissions, aerosols and landfills (anthropogenic factors) (Fisk *et al*., 2009; Kojima and Lovei, 2001). In built-up environments, AQ could be affected by factors such as a building site and design, renovations that take place in the building, occupant densities within the building, and maintenance of air-cleaning devices or systems (OSHA, 2011). Indoor and OAQ including that of a university environment are influenced by physical, chemical and biological factors (Badea *et al*., 2015) such as vehicular and generator emissions, other power generating equipment and the burning of fossil fuels (WHO, 2017) in the environment. As AQ deteriorates, the burden it becomes to people within the nearest vicinity or surrounds increases. Poor AQ is indicative of an elevated pollutant concentration with detrimental health impacts whereas, good AQ is characterised by air that is pollution-free and clean, which is crucial for the comfort, health, and well-being of individuals (Thomson *et al.,* 2003). Air quality can be ascertained by the presence of unpleasant odours or stuffy air.

2.3. Indoor and outdoor air pollutants

Air pollution comprises of both IAP and OAP. Indoor air pollution emanates predominantly from the burning of fossil fuels such as wood, coal, and paraffin for purposes of cooking, space heating, and lighting (Wright and Oosthuizen, 2009). Common IAPs include O_3 (Koponen *et al*., 2001), SO_2 (Nkosi *et al*., 2017; Koponen *et al*., 2001), NO_2 (Meier *et al*., 2015; Rivas *et al*., 2014; Kousa *et al*., 2001), CO_2 (Gao *et al*., 2014; Mahyuddin *et al*., 2014), non-methane hydrocarbons (NMHCs) (Krugly *et al*., 2014; Masih *et al*., 2012;

Masih *et al*., 2010; Menichini *et al*., 2007), VOCs (Adgate *et al*., 2004; Brickus *et al*., 1998) and PM (Nkosi *et al*., 2017; Meier *et al*., 2015; Krugly *et al*., 2014; Rivas *et al*., 2014). Some of these IAPs are widely used as ingredients in fuels, paints, varnishes, aerosol sprays, building material and furnishes, office equipment (printers and copiers), graphic and craft material (glues, photographic solutions, and permanent markers) and other household products (cleaning, disinfecting, cosmetic, moth deterrents, air fresheners, pesticides, laundered clothing and degreasing products) (USEPA, 2017a).

In contrast, in the outdoor environment, the main sources of OAP are vehicular emissions, power generation plants, industrial furnaces, biomass burning and others (Balajee *et al*., 2017; BC, 2017; Bo *et al*., 2017; Radaideh, 2017). Common OAPs include: O_3 (Balajee *et al*., 2017; PPAH, 1998), SO_2 (BC, 2017), NO_2 (OAP, 2016; PPAH, 1998), CO (Radaideh, 2017), VOCs (PPAH, 1998) and PM (Bo *et al*., 2017; OSHA, 2011). The Environmental Protection Agency (EPA) has recognised OAPs that are harmful to humans and thus need to be monitored regularly; these pollutants are known as criteria air pollutants. The seven criteria pollutants deemed by the EPA to be significant in OAP are O_3, SO_2, NO_2, CO, PM_{10}, $PM_{2.5}$ and lead (Pb) (EPA, 2012). The concentration of criteria air pollutants within an enclosed building is a good indicator of whether a building is devoid of pollutants or not. Additionally, the WHO has formulated guidelines that make it possible to determine if AQ would be injurious to human health and at what levels. Exposure to high levels of O_3, SO_2, NO_2, PM_{10}, and $PM_{2.5}$ have been ascertained by the WHO to have pertinent health outcomes and their concentrations in any given environment should comply with standards (WHO, 2006). Some of the above-mentioned OAPs are briefly discussed in the following sections.

2.3.1. Ozone

Ozone is a colourless compound that comprises of three oxygen (O_2) atoms and it occurs both at ground level and the upper atmosphere. It is one of the most important photochemical oxidants in the troposphere and is formed when NO_x and VOCs are present in the air. It emanates from aerochemical reactions between NO_x, VOCs and other gases as a secondary product. Indoor O_3 concentrations can differ quite markedly from one room or structure to another because of its short half-life of 7 to 10 minutes. According to Weschler (2000), indoor O_3 concentrations are affected by air exchange rates, outdoor ambient O_3 concentrations, rates of surface removal and other chemical and compound

reactions in the atmosphere. Within a university environment, motor vehicles are the primary anthropogenic sources of O_3 and its precursors NO_2 and VOCs (PPAH, 1998) but in other areas, power plants, refineries, chemical plants, and boilers are the main sources (Balajee *et al*., 2017). Secondary sources of O_3 may include organic solvents from laboratories, fossil fuel combustion (PPAH, 1998), photocopying and air purifiers (Britigan *et al*., 2006). Ozone presents some health risks. It may cause pulmonary function related problems and induce inflammation in one's airways (Hubbard *et al*., 2005). Daily mortality increases have been documented through time-series studies ranging between 0.300 to 0.500 %, every 10.000 $\mu g/m^3$ augmentation in 8-hour O_3 concentrations above the appraised baseline of 70.000 $\mu g/m^3$ (WHO, 2006).

A few studies have focussed on indoor and outdoor concentrations of O_3 (Kalimeri *et al*., 2016; Jovanovic *et al*., 2014; Poupard *et al*., 2005). Findings of these studies indicate that indoor O_3 concentrations tend to be lower than outdoor O_3 concentrations due to its short half-life and reactivity with other air pollutants. In both environments, various levels of compliance with permitted values of O_3 have been reported. Fadeyi *et al*. (2014) and Radaideh (2017) reported O_3 concentrations exceeding maximum acceptable levels whereas Chao (2001) reported indoor O_3 concentrations to be below the IAQ guideline for office premises. Meteorological variables are also reported to affect O_3 concentrations in indoor and outdoor environments. Radaideh (2017) and Pudasainee *et al*. (2006) found positive correlations between temperature, RH and O_3 concentrations. According to Elampari and Chithambarathanu (2011), temperature peaks were found to be correlated with O_3 concentration peaks, whereas rainfall had an inversely proportional relation with O_3 concentrations. Dikty *et al*. (2010), Pudasainee *et al*. (2006) and Wang *et al*. (2001) all highlighted the relationship between daytime O_3 concentration and temperature, solar radiation and light winds. The formation of tropospheric O_3 is triggered by the presence of heat and the intensity of solar radiation, with O_3 concentrations also being influenced by long-range transport (Derwent *et al*., 2003; Kato *et al*., 2001). Furthermore, O_3 concentrations are greatly affected by wind speed, temperature and RH (Ocak and Turalioglu, 2008).

Results from studies focusing on the roles of seasons in O_3 concentrations have been contradictory. In the outdoor environment, O_3 displays a strong seasonal pattern and variation with meteorological conditions (Balajee *et al*., 2017; GreenFacts, 2017; Koponen

et al., 2001). Elampari and Chithambarathanu (2011) found wind speed to have a negative correlation with O_3 concentrations in winter, whereas in summer positive correlations were evident. Furthermore, Elampari and Chithambarathanu (2011) established a negative correlation between RH and O_3 concentrations across all four seasons. Wheida *et al*. (2018), Bloomer *et al*. (2010), El-Tahan (2018) and Balajee *et al*. (2017) all reported minimum O_3 concentrations during winter and maximum concentrations during summer. Generally, maximum O_3 concentrations encountered during summer months were as a result of stratospheric intrusions, the release of NO_x and other VOCs, synoptic wind patterns, solar radiation differences, biomass burning and prevailing air mass types. Minimum values attained for O_3 concentrations during winter were attributed to negligible photochemical production, and dry deposition. In contrast, Chen *et al*., (2014) observed O_3 maxima concentrations during winter and minima O_3 concentrations during the late spring and summer seasons. David and Nair (2011), Suthawaree *et al*. (2008) and Lal *et al*. (2000) all reported similar findings. Be that as it may, Kalimeri *et al*. (2016) and Poupard *et al*. (2005) in their studies on IAQ found no correlations between indoor O_3 concentrations and seasons.

2.3.2. Sulphur dioxide

Sulphur dioxide is another common pollutant that may have damaging health impacts upon exposure to high concentrations. It is a pungent, colourless, toxic gas, present in the air and it is associated with discomfort in individuals (Lippmann *et al.,* 2003). Some harmful health effects of SO_2 exposure are changes in pulmonary function and both upper and lower respiratory health endpoints (Nkosi *et al*., 2017). Additionally, average SO_2 concentrations are linked with a surge in cardiovascular and respiratory mortality in elderly populations (Chen *et al*., 2012). Furthermore, elevated SO_2 concentrations have also been linked with throat irritation, bronchoconstriction and dyspnoea, being evident in asthmatics predominantly (Kampa and Castanas, 2008; Tunnicliffe *et al*., 2001). Once released, SO_2 serves as a surrogate for other substances (Chen *et al*., 2012) in addition to being reactive with other contaminants and chemicals in the air and can lead to the formation of fine PM (BC, 2017). In general, the main sources of SO_2 are oil and gas industries, metal smelting, and paper production (BC, 2017). Other sources of SO_2 are massive ships and outdoor equipment burning high sulphur comprising fuels (BC, 2017). Within the university, SO_2 may originate from surrounding power plants burning coal with high sulphur content, emissions from domestic coal burning and vehicular emissions (Costabile *et al*., 2006;

PPAH, 1998). Annual guidelines for SO_2 are not needed since daily guidelines will ensure that low annual averages are attained. However, it is quite difficult for most countries to conform to the 24-hour guideline. Transition-type approaches with conformation dates being enforced are the norm (WHO, 2006) as is the case with the National Ambient Air Quality Standards (NAAQS) of South Africa.

Sulphur dioxide indoor concentrations are consistently lower than outdoor concentrations (Nkosi *et al*., 2017; Benson *et al*., 1972). Due to the limited number of studies done in line with indoor and outdoor SO_2 concentration, differences, and compliance to standards, one cannot conclude that these are the normal patterns encountered in indoor and outdoor environments. Furthermore, due to just one study finding pointing towards ambient SO_2 concentration exceedance of the NAAQS, this cannot be established as the norm. A few studies have documented the relationship between meteorological conditions and SO_2 concentrations. Datta *et al*. (2010) found ambient temperature and rainfall to have significant negative correlations with SO_2 concentrations. Likewise, Radaideh (2017), found outdoor SO_2 concentrations to be negatively correlated with temperature and positively correlated with RH. Tayanc (2000) and Akpinar *et al*. (2009) both reported an association between airflow rate and decreased SO_2 concentrations. The general trend that emerges from the literature regarding the relationship between meteorological factors and SO_2 concentrations are the following: ambient SO_2 concentrations tend to be negatively correlated with both temperature and airflow rate, whilst having a positive correlation with RH. Generally higher SO_2 concentrations are encountered in winter, whereas lower SO_2 concentrations are seen in summer (Jones and Harrison, 2011; Costabile *et al*., 2006; Bari *et al*., 2003; Koponen *et al*., 2001).

2.3.3. Nitrogen dioxide

Nitrogen dioxide is introduced into the air through the burning of vehicular fuel emissions, power plants, and acid rain (USEPA, 2016a; Perraud *et al*., 2012). Residential combustion of fossil fuels and microbial activity in soils are other sources of NO_2 (OAP, 2016). Within the university, the main sources of NO_2 are from heat and electricity generating areas, laboratories harbouring nitrogenous chemicals and gases, an arc-welding workshop and a small proportion emanating from motor vehicle exhausts (PPAH, 1998). Direct toxic effects have not been documented for long-term exposure to NO_2. However, there is increasing evidence of airway irritation and other respiratory symptoms and disorders

attributed to OAP mixtures of which NO_2 is a major component (Gilbert *et al*., 2006; WHO, 2006; Sakai *et al*., 2004).

Numerous studies have focused on NO_2 concentrations in indoor and outdoor environments, their compliance with standards and indoor:outdoor (I/O) ratios. Radaideh (2017), who investigated the influence of climatic factors on air pollutant differences in arid climates in Saudia Arabia, found mean outdoor NO_2 concentration values ranging between 0.41 to 0.88 parts per million (ppm) whereas, in the indoor environment, the study conducted by Lu *et al*. (2016) found mean indoor NO_2 concentrations to be approximately 0.02 ppm in China. Meier *et al*. (2015), Pegas *et al*. (2011) and Pegas *et al*. (2010) also reported higher outdoor NO_2 concentrations than in the indoor environment across Swiss homes and elementary schools in Lisbon. On the contrary, Jovanovic *et al*. (2014) reported higher mean concentrations of indoor NO_2 in comparison to the outdoor environments within schools in Serbia. Another viewpoint was established by Wichmann *et al*. (2010) and El-Hougeiri and El Fadel (2004) who found indoor and outdoor NO_2 concentrations to be correlated. Wheida *et al*. (2018), Pegas *et al*. (2011) and Pegas *et al*. (2010) all found indoor NO_2 concentrations to be within the WHO guideline value of 40.00 $\mu g/m^3$. Different ratios of I/O NO_2 concentrations have also been reported in literature; ranging from 0.29 to 1.25 (Meier *et al*., 2015; Pegas *et al*., 2011; Pegas *et al*., 2010; Chao, 2001).

The consensus from literature points towards higher outdoor NO_2 concentrations compared to indoor NO_2 concentrations, with indoor NO_2 concentrations generally complying with standards (Kalimeri *et al*., 2016; Rivas *et al*., 2014; Alves *et al*., 2013; Gilbert *et al*., 2006). Studies focussing on NO_2 concentrations with specific attention to urbanisation, spatial distribution, and seasonal differences are limited. Studies on NO_2 concentrations and seasonal variations have also been deciphered briefly, in literature. Kalimeri *et al*. (2016) investigated the IAQ in primary school environments in Greece and found higher indoor NO_2 concentrations in summer compared to winter. Similar seasonal patterns of NO_2 have been reported by Ni *et al*. (2016), Cyrys *et al*. (2000), Wangchuk *et al*. (2015) and Kornartit *et al*. (2010). In contrast, however, Hargreaves *et al*. (2000) studied the confined and seasonal differences in atmospheric NO_2 levels in the UK and observed winter highs and summer lows to be the norm. The consensus seems to be

increased NO_2 concentrations in summer compared to the winter season, with just a single study opposing this notion.

2.3.4. Carbon monoxide

Carbon monoxide is categorised as an odourless, tasteless and non-irritant gas of central importance in the earth's atmosphere, especially in the troposphere (Radaideh, 2017). Natural sources of CO include oxidation of natural hydrocarbons and methane, and emissions from both vegetation and the ocean (Radaideh, 2017). Anthropogenic emissions usually emanate from vehicular exhausts, biomass burning, industrial activities and fossil fuel burning (Radaideh, 2017). Carbon monoxide combines with haemoglobin in the blood, leading to the inhibition of the body's ability to transport and exchange O_2 readily, eventually bringing about mortality (ASHRAE, 2013). On exposure to elevated concentrations of CO, extremely detrimental health effects may occur (Ole, 2013; Leung *et al.,* 2002). Some health-related effects of CO exposure include headaches, nausea, dizziness, and upon prolonged exposure even mortality (Weaver *et al*., 2002). Carbon monoxide is unreactive and indoor concentrations are expected to approximate outdoor concentrations after a certain lag time (Benson *et al*., 1972).

A few researchers have studied CO as a parameter of interest in both the indoor and outdoor environments. These researchers have reported various levels in the different environments, with indoor concentrations being generally less than outdoor concentrations. Indoor CO concentrations were found to range between 0.01 and 8.10 ppm, whereas outdoor CO concentrations were ranging between 1.16 and 30.40 ppm (Radaideh, 2017; Fazlzadeh *et al*., 2015; Wangchuk *et al*., 2015; Fadeyi *et al*., 2014; Alves *et al*., 2013; El-Hougeiri and El Fadel, 2004; Baek *et al*., 1997). Contrastingly, Choo *et al*. (2015) and Lee and Chan (1998) reported elevated indoor CO concentrations. The overall picture is that CO tends to comply with regulations and standards in both the indoor and outdoor environments. However, occasional indoor peak concentrations are encountered, depending on the usage and burning of fossil fuels.

A few studies relating to CO concentrations and seasonal differences have been conducted. Average indoor CO concentrations in literature for summer ranges were found to be between 0.50 and 4.70 ppm, whilst in winter the corresponding range is between 8.80 and 13.00 ppm (Kalimeri *et al*., 2016; Ni *et al*., 2016). Al-Rehaili (1999) found higher

indoor than outdoor CO concentrations during winter. Baek *et al*. (1997) found statistically significant correlations amongst CO concentrations across both the indoor and outdoor settings in the summer season within office environments. From the above literature about CO concentrations, and its relationship to seasonal changes, higher indoor CO concentrations are generally seen during winter in comparison to the summer season, whilst a disparity is encountered when trying to establish CO concentrations in indoor and outdoor environments. These differences may be due to the inherent activities and sources of CO emanating from both environments.

2.3.5. Carbon dioxide

Carbon dioxide is a product of respiration and other physiological processes. Carbon dioxide levels very rarely escalate to levels that can have deleterious health effects and, in the norm, can be tolerated by healthy individuals without any health-related problems arising (ASHRAE, 2013). However, CO_2 within an enclosed environment or building for extended periods potentiates its concentrations to levels that are toxic to humans (OSHA, 2011). Carbon dioxide levels can furthermore be used as a marker for the existing level of ventilation within a given environment (St-Jean *et al*., 2012) in tandem with it being indicative of high population densities within buildings. Its concentration increases proportionally with increase in population densities and can be filtered out from buildings based on the existing ventilation system (OSHA, 2011; Prill, 2000). Carbon dioxide in a university environment usually originates from human presence, through metabolism and respiration processes (Budiakova, 2017).

Carbon dioxide is predominantly a parameter of interest in several IAQ studies, whereas OAQ studies focussing on CO_2 concentrations are scarce. Indoor and outdoor concentrations, ranges and compliance to standards are quite well documented in literature, with contradictory findings commonly seen. Jovanovic *et al*. (2014) investigated the indoor and OAQ within classrooms in Serbia and found that two of the three classrooms studied had elevated indoor CO_2 concentrations. Studies relating to indoor CO_2 concentrations found ranges between 1000 and 2739 ppm (Budiakova, 2017; Peng *et al*., 2017; Cetin and Sevik, 2016; Fadeyi *et al*., 2014; St-Jean *et al*., 2012). Contrastingly, Choo *et al*. (2015), Alves *et al*. (2013) and Baek *et al*. (1997) all found indoor CO_2 concentrations to be below 1000 ppm. The overall trends observed from these studies are for indoor CO_2 concentrations to be higher than outdoor CO_2 concentrations,

due to increased occupant densities across the indoor environment. Furthermore, a disjuncture in terms of CO_2 concentrations and compliance with indoor standards are commonly encountered.

Concerning CO_2 concentrations and the type of ventilation incorporated in different environments; several studies have observed the use of heating, ventilation and air conditioning (HVAC) systems in addition to natural ventilation and the issue of occupant densities. These studies reported conflicting results. Budiakova (2017) and Gao *et al*. (2014) both found higher indoor CO_2 concentrations when natural ventilation was provisioned for. In stark contrast, Jurado *et al*. (2014) and Houri *et al*. (2009) found naturally ventilated classrooms to have lower CO_2 concentrations (408–520 ppm) as compared to classrooms with HVAC systems (497–1434 ppm). Several studies have found an association between occupant densities and CO_2 concentrations (Pereira *et al*., 2014; Fadeyi *et al*., 2014; Mentese *et al*., 2012; Pegas *et al*., 2011).

With regard to the seasonal trends of indoor and outdoor CO_2 concentrations, a few studies have looked into these relationships. Gao *et al*. (2014) and Mentese *et al*. (2012), all found average indoor CO_2 concentrations to be higher in winter than in summer, whereas outdoor summer CO_2 readings were higher than winter readings. In support of these indoor and outdoor differences, Baek *et al*. (1997) found statistically significant differences and correlations between both environments' CO_2 concentrations during summer and winter seasons in offices ($p < 0.01$). Contrastingly, Moriwaki and Kanda (2004) and Pataki *et al*. (2003) all observed lower outdoor CO_2 concentrations in summer as opposed to peaks attained in winter. From the available evidence, indoor CO_2 is found at minimal concentrations in summer in comparison to winter peak concentrations. In the outdoor environment, contradicting findings are seen; however, the consensus is for CO_2 concentrations to be higher in winter.

2.3.6. Particulate matter

Particulate matter is a combination of a wide array of liquid droplets and solid particulates within the air that could be deposited deep within the air passages of the respiratory system leading to respiratory diseases and in some instances, lung cancer and premature deaths. It has been associated with the increased mortality of infants and other sensitive populations (Estokova *et al*., 2010). Both PM_{10} and $PM_{2.5}$ differ only based on their size

(Ole, 2013). Particulate matter concentrations are mainly influenced by rates of air exchange, concentrations in the ambient environment, deposition and penetration factors, and mechanisms of re-suspension (Fromme *et al*., 2007).

A few studies are documented in literature which assess PM origination points in different environments. Road traffic, power generation plants, industries, smoking, cooking, heating systems, cleaning activities, re-suspension due to human presence, agriculture and domestic heating systems are documented as the main sources of outdoor PM (Bo *et al*., 2017). Fromme *et al*. (2007) found that walking, cooking, cleaning, and smoking may lead to the formation of PM in indoor air. Hence, from literature, the main indoor and outdoor sources of PM originate from a disparate number of sources, with heating systems being a commonality. Little evidence points to the correlation of outdoor and indoor PM concentrations, in addition to compliance with standards and seasonal PM concentration variations. Bo *et al*. (2017) and Al-Rehaili (1999) both found a direct correlation between indoor and outdoor total suspended particulate (TSP) levels. Furthermore, Al-Rehaili (1999) found that PM levels exceeded standards predominantly in winter as compared to summer. Fugas and De Koning (1991) conducted a study in Bombay, Toronto, and Zagreb and reported low summer TSP concentrations and similar I/O ratios, whereas, in winter, TSP concentrations were found to be higher. As seen with CO_2 and CO concentrations, a similar trend is evident with PM concentrations; higher concentrations are found in winter in comparison to summer.

2.4. Factors that determine indoor air quality

Ventilation or airflow rate, RH, and temperature are deemed to be good determinants of IAQ, due to their influence on the movement of air (RDH, 2015; CPCB, 2014; OSHA, 2011; Indoor Air Quality in Office Buildings, 1995; USEPA, 1991). Some other factors to take into consideration when assessing IAQ as stated by OSHA (2011) are building site and design, renovations if any, occupancy within the building, and the maintenance of air-cleaning devices or systems in the building. This section presents available literature on the factors that influence IAQ.

2.4.1. Linear air velocity

Linear air velocity or the airflow rate can be defined as the displacement rate of air or gas within an area along a straight line (ASHRAE, 2010). When monitoring air-conditioning

and ventilation systems, LAV is the best predictor to ascertain the airflow in a certain location within any given environment (Gunnarsen and Fanger, 1992). To minimise the prevalence of respiratory health endpoints, ample ventilation within an enclosed environment is paramount. A few studies have identified airflow rate as a marker to determine the quality of air. The focus has been on indoor LAV ranges, LAV differences in natural and air-conditioned environments and LAV compliance to standards. Budiakova (2017), Mohammadyan *et al*. (2017), Lu *et al*. (2016), Choo *et al*. (2015) and Yau *et al*. (2011) all found indoor LAV rates ranging between 0.00 and 3.00 meters per second (m/s) within universities, schools, and dwellings. Jurado *et al*. (2014) found the mean indoor LAV in naturally ventilated rooms within universities to be 0.14 m/s; whereas, in air-conditioned rooms, a corresponding value of 0.09 m/s was found. Both conform to the recommended Brazilian guideline of ≤ 0.25 m/s. Wangchuk *et al*. (2015) found the mean ambient LAV during the wet season (warm and wet weather-summer) to be 0.58 m/s, whilst in the dry season (cool and windy weather-winter) the corresponding value attained was 1.14 m/s. On the other hand, Al-Rehaili (1999) found outdoor LAV values in Saudia Arabia to range between 1.50 and 4.50 m/s in summer, and between 1.00 and 2.80 m/s in winter.

A few studies have examined the linkage between airflow and pollutant concentrations. Koponen *et al*. (2001) found strong correlations amongst the rate of ventilation and indoor particulate and gas concentrations. Widder and Haselbach (2017), Yu *et al*. (2015) and Seppanen (2007) and Einberg (2005) all found that increases in ventilation rates led to proportionate emission rate increases of specific VOCs, PM and other pollutants; whereas, lower ventilation rates resulted in diminished energy dissipation and a corresponding deterioration in IAQ. It is clear from these studies that low airflow rates are linked with increasing levels of pollutant concentrations.

2.4.2. Temperature

The heat intensity or thermal conditions within any given environment is paramount in governing the AQ in the surrounds. Temperature has been documented widely in literature as a parameter to determine AQ in both indoor and outdoor environments. Indoor temperature ranges reported in various studies vary from 2.7 and 30.9 °C (Budiakova, 2017; Mohammadyan *et al*., 2017; Peng *et al*., 2017; Jovanovic *et al*., 2014; Pereira *et al*., 2014; Tomic *et al*., 2014; Yau *et al*., 2011; Pegas *et al*., 2010). In the outdoor environment,

Lu *et al*. (2016), Brickus *et al*. (1998) and Chao (2001) reported a range of -3.0 to 32.5 °C. Exceedance in temperatures in comparison with guidelines are encountered; however, indoor environments generally tend to conform to their respective guidelines (Fadeyi *et al*., 2014; Jurado *et al*., 2014; Alves *et al*., 2013).

The relationship between temperature values and corresponding seasonal variations has also been investigated quite extensively, with higher temperatures observed in summer (wet) and winter (dry) seasons. Summer indoor and outdoor temperatures are usually higher than winter indoor and outdoor temperatures (Kalimeri *et al*., 2016; Wangchuk *et al*., 2015; Gao *et al*., 2014; Jurado *et al*., 2014). Temperature variations in an indoor environment are generally as a result of unregulated heating systems, the construction of new windows and the provisioning of natural ventilation (Budiakova, 2017). Furthermore, outdoor temperature infiltration, room occupant densities, solar radiation and the improper control of temperatures bring about temperature variations in the indoor environment (Fadeyi *et al*., 2014). Both Gao *et al*. (2014) and Pereira *et al*. (2014) established higher temperatures during building occupancy and natural ventilation (opening doors and window), whilst lower temperatures were recorded in unoccupied buildings and mechanically ventilated classrooms. In relation to the association between temperature variations and pollutant concentrations, the evidence in literature is limited. Radaideh (2017) found some associations between temperature and certain pollutants. According to his findings, changes in temperature had little or no effect on O_3 concentrations but an extremely weak negative correlation (R^2 = -0.036) between SO_2 concentrations and ambient temperature was reported. Radaideh (2017) also reported a relationship between NO_2 (R^2 = 0.648) and CO (R^2 = 0.096) concentrations and temperature.

2.4.3. Relative humidity

Relative humidity relates to the actual quantity of moisture within the air in connection to the highest amount of moisture the air can contain at a given temperature, thus acting as a precursor in determining thermal comfort (Mason *et al*., 2001). Relative humidity is mostly affected by local weather conditions. However, it is additionally affected by the use of a building and its prevalent thermal properties (Mason *et al*., 2001). Damp interior environments have been associated with serious health conditions including asthma, hypersensitivity, and sinusitis. Humidity and temperature levels within recommended standards or guidelines are imperative in maintaining healthy and conducive

environments (OSHA, 2011; EPD, 2003). Higher humidity levels may result in microbial growth, whereas temperature extremes may increase atmospheric moisture content thereby encouraging the growth of mould and mildew (OSHA, 2011; USEPA, 1991). Relative humidity affects thermal comfort by impeding the body's ability to get rid of excess heat through the physiological process of perspiration and the heat exchange mechanism of evaporation. Subsequently, an individual may feel wet and sweaty, which are ideal conditions for mould growth. These could aggravate allergies and asthmatic conditions (Weaver *et al*., 2002).

Studies relating to RH levels and conformity to guidelines have been documented extensively in literature. Relative humidity values ranging from 13.5 to 69.3 % were reported in several AQ studies (Budiakova, 2017; Jovanovic *et al*., 2014; Pereira *et al*., 2014; Tomic *et al*., 2014; Alves *et al*., 2013; Pegas *et al*., 2011; Yau *et al*., 2011; Pegas *et al*., 2010; Brickus *et al*., 1998). Generally, indoor RH values conform to the ASHRAE guideline of 30 to 60 % (Fadeyi *et al*., 2014; Alves *et al*., 2013; Pegas *et al*., 2011). Similarly, Jurado *et al*. (2014) found indoor RH levels were within acceptable ranges in air-conditioned classrooms, whilst in naturally ventilated classrooms some exceedance to the Brazilian guideline of 40 to 65 % was encountered. In the outdoor environment, RH values tend to range between 27 to 94 % (Mohammadyan *et al*., 2017; Lu *et al*., 2016; Choo *et al*., 2015; Jovanovic *et al*., 2014; Alves *et al*., 2013; Brickus *et al*., 1998). Outdoor RH values tend to be higher than indoor RH values.

Studies investigating the differences in RH values per season have also been reported. According to Kalimeri *et al*. (2016), Mentese *et al*. (2012), St-Jean *et al*. (2012) and Al-Rehaili (1999), indoor RH levels in summer tend to be slightly lower (14.9 to 65.0 %) than winter levels (23.0 to 64.4 %). A similar trend was observed in outdoor environments where RH summer levels were lower (18.0 to 46.0 %) than winter levels (18.6 to 72.0 %) (Mentese *et al*., 2012; Al-Rehaili, 1999). With regards to the relationship between RH values and pollutant concentrations, Bayer *et al*. (2002) showed that schools with functional and active humidity systems generally have higher pollutant emissions. Wolkoff (2018) documented correlations between low RH and O_3 with detrimental health effects. According to Wolkoff and Kjaergaard (2007), O_3 deposition rates on harder surfaces decreased substantially (up to 50 %) with increasing levels of RH. Radaideh (2017) however reported a weak positive correlation between O_3 concentrations and ambient RH

levels, an extremely weak positive correlation between SO_2 concentrations and ambient RH levels (R^2 = 0.029), a weak positive correlation between CO concentrations and RH (R^2 = 0.175), and a strong negative correlation between NO_2 concentrations and RH (R^2 = -0.781). Radaideh's observation of the relationship between RH and NO_2 was contrary to what Wolkoff and Kjaergaard (2007) found. They found an increase in the deposition rate of NO_2 with an increase in RH levels. An inversely proportional relationship is seen between RH and O_3 and NO_2 concentrations, whereas a directly proportional relationship is seen between RH and SO_2 and CO concentrations.

2.4.4. Building site, structure, design and renovation activities

The site and location where a building is erected can have numerous implications for IAQ. Proximity to highly industrialised areas, highways, and urban areas may be areas where PM and other criteria pollutants are harboured and disseminated to nearby buildings (OSHA, 2011). Additionally, buildings built upon land that underwent prior industrial activities may bring about pollutant intrusion into the building (OSHA, 2011). A building's structural design may affect its IAQ. Poor foundation, roofs, windows and door openings may facilitate the intrusion of air pollutants (OSHA, 2011).

Common pollutant sources in the surroundings of a building include outside air ventilation systems incorrectly positioned; for example, near idling vehicles, waste containers, and other products of combustion. Building exhaust fumes may also re-enter the building through these ventilation systems (OSHA, 2011). Factors which could potentially affect IAQ during the design of a building include siting (traffic, parking areas, soil emissions, moisture sources), building envelope (cooling and heating loads, unintended infiltration), waste service loading dock entrances (odours from waste, fossil fuels emitted from vehicles and particle intake), HVAC systems (filters, air ducts, unintended pathways), plumbing (leaking pipes provide moisture leading to microbial growth) electrical systems (electromagnetic fields), sanitation vents, exhaust fans and fume hoods (potential chemical and biological exposures), communicative wiring (excess wiring could lead to VOC emissions), and materials used for internal finishing and furnishing of buildings (sources of VOCs and nutrient source for microorganisms) (Spengler and Chen, 2000).

Renovation or construction activities in the vicinity of a building can further exacerbate pollutant concentrations within a building and its surroundings (OSHA, 2011). Painting, in

addition to other renovation or construction activities produce dust and other emissions that may intrude and/or circulate within a building (OSHA, 2011). Climatic parameters, along with atmospheric pollution play a role in the degradation of metallic construction materials, concrete materials and plasters of buildings around the world (Ivaskova *et al*., 2015). Building materials and furnishings with compromised thermal insulation material, wet or damp structural surfaces and certain pressed-wood products may also contribute to IAPs such as VOCs, in turn affecting the quality of air in that given environment (Senitkova, 2017; OSHA, 2011). Findings in literature tend to support the notion than pollutant concentrations are elevated in construction areas and in buildings undergoing renovations (Jia *et al*., 2008a; Schneider *et al*., 2001).

2.4.5. Occupancy

Several studies have ascertained the relationship between occupant densities and pollutant concentrations. Carbon dioxide concentrations increase proportionally with higher population densities, with seating position and occupancy periods being additional factors contributing to CO_2 concentrations (Mahyuddin *et al*., 2014; OSHA, 2011; Ponsoni and Raddi, 2010; Prill, 2000). Peng *et al*. (2017) found an association between PM levels and the occupancy of classrooms, with student activities such as cleaning the board, walking and running leading to the re-suspension of PM. Widder and Haselbach (2017) also highlighted cooking, cleaning and painting as precursors of IAPs and new furniture, cabinets, carpeting and drapes as emitting sources. Building occupants have been reported to contribute to bio-effluent emissions in the air, in addition to VOCs, PM, CO_2, bio-aerosols, and water vapour (Ponsoni and Raddi 2010). According to Poupard *et al*. (2005) and Holmberg and Chen (2003), occupants control the amount of natural ventilation and HVAC systems. Particulate matter, VOCs, CO_2, bioaerosols, water vapour, bacteria, temperature increases, occupant-related emissions and re-suspension of particulates seem to increase with increases in occupancy.

2.4.6. Maintenance of heating, ventilation and air conditioning systems

The design and performance of HVAC systems determine the quality of indoor air to a great extent (Khan *et al*., 2016). A ventilation system's efficacy is primarily determined by the proficiency of the air delivery mechanism in eradicating IAPs from the aerated environment (Perreira *et al*., 2009). Moreover, the functionality of HVAC systems and their maintenance is imperative in ensuring a good supply of indoor air (OSHA, 2011).

Generally, four kinds of ventilation or air circulation systems are denoted: “conventional air circulation system with ceiling source and return (above and near the floor), the underfloor air distribution system and a split system” (Perreira *et al*., 2009). The split system has the highest concentration of large particles near the breathing zone, with the ventilation efficacy being excellent for small particulates (Perreira *et al*., 2009). Conversely, the underfloor air circulation system shows optimal efficacy for larger particles, with particulate concentrations proportionally increasing within the breathing area of an occupant as particulate diameter increases (Perreira *et al*., 2009). With the ceiling source and return system near the floor, particulate concentration and air circulation efficacy are both diminished with a corresponding reduction in particulate diameter (Perreira *et al*., 2009). For the ceiling source and return system above the floor, a directly proportional relationship was evident between particulate concentration and particulate diameter, with there being an inversely proportional relationship between ventilation effectiveness and particulate diameter (Perreira *et al*., 2009). The maintenance of local exhaust ventilation systems within a university environment (situated in laboratories, kitchens, parking areas, bathrooms, copy rooms, and other specialised areas) is paramount to ensuring limited pollutant generation (OSHA, 2011).

A non-functional HVAC system leads to the intrusion of OAPs and other pollutant sources such as PM, vehicular and generator exhaust emissions, humid air, and parking area contaminants (OSHA, 2011). Upon renovation or any construction activities arising, HVAC systems tend not to be maintained or upgraded to accommodate the type of occupancy (OSHA, 2011). Old university buildings are generally devoid of regulated heating systems and mechanical ventilation systems (Budiakova, 2017). In contrast, newer university buildings have more modern thermo-technical parameters, energy-saving regulated heating systems and mechanical ventilation systems, which contribute to thermal comfort (Budiakova, 2017). The best method of controlling thermal comfort is by utilising minimal energy dissipating processes such as localised and/or task orientated air conditioning systems (Al horr *et al*., 2016).

Lee *et al*. (2009) compared airflow and pollutant circulations in rooms with outdated displacement and underfloor air distribution systems and found that higher air distribution effectiveness was evident with each system as opposed to a combination of ventilation systems. Widder and Haselbach (2017) concluded that continuous exhaust ventilation

was not an effective means of reducing indoor VOC concentrations unless implemented below the breathing zone of people within a given environment. Opposing findings are encountered in connection to HVAC systems and their associations with pollutant concentrations. Some studies report using a single means of ventilation as being most effective, whereas other studies pointed out the fact that a singular mechanism of ventilation may not be the most efficient means in reducing indoor pollutant concentrations. A summary of some additional studies carried out worldwide to determine the factors affecting indoor AQ is presented below (Table 2.1).

Table 2.1: Additional indoor air quality studies conducted worldwide.

Study area	Environmental setting	Parameter investigated	Factors affecting air quality	Reference
Italy.	School buildings.	$PM_{2.5}$, metals, and polycyclic aromatic hydrocarbons.	Industry (steel plant), school equipment, paints, and dyes.	Di Gilio *et al*., 2017.
Lithuania.	Primary school buildings.	VOCs and NMHCs.	Motor vehicles, heating (fuel burning).	Krugly *et al*., 2014.
Spain.	School buildings.	$PM_{2.5}$, NO_2 and ultrafine particulates.	Indoor school sources, outdoor traffic emissions, limited building protection, clothing, cooking, organic emissions.	Rivas *et al*., 2014.
India.	Urban residential and roadside buildings.	VOCs and NMHCs.	Seasonal variations, cooking, smoking and incense burning, traffic and generator emissions.	Masih *et al*., 2012.
India.	Urban residential and roadside buildings.	VOCs and NMHCs.	Vehicular emissions, gas utilities, cooking, smoking and incense burning.	Masih *et al*., 2010.
Italy.	Residential buildings.	VOCs and NMHCs.	Seasonal variation, vehicle exhaust emissions, indoor VOC and NMHC sources.	Menichini *et al*., 2007.
China.	Urban residential buildings.	VOC and NMHCs.	Smoking, incense burning, and poor ventilation, building materials and furniture, industries, seasonal variations.	Wang *et al*., 2007.

USA.	School buildings.	VOCs.	Vehicle exhaust, smoking, consumer products.	Adgate *et al*., 2004.
Lebanon.	Urban commercial, residential and industrial buildings.	CO, NO_2, PM and selected priority metals.	Combustion sources, cleaning products, and other indoor pollutant sources.	El-Hougeiri and El Fadel, 2004.
USA.	Urban residential buildings.	VOCs and $PM_{2.5}$.	Automotive exhaust and smokestack emissions, cooking, smoking, consumer products, ventilation rates.	Sexton *et al*., 2004.
Germany.	Residential buildings.	VOCs and NMHCs.	Automobile exhaust emissions, outdoor VOC and NMHC concentrations.	Ilgen *et al*., 2001.
Switzerland, Finland and Czech Republic.	Residential and office buildings.	NO_2.	Gas appliances, workplace location.	Kousa *et al*., 2001.

2.5. Parameters that determine outdoor air quality

Numerous factors affect OAQ. The DEA (2017), established that the largest contributing anthropogenic factors affecting OAQ are vehicular emissions, fixed power generating plants, other industrialised and agricultural emissions, domestic fuel burning, biomass burning and landfill gas emissions. Additional factors affecting OAQ are natural factors such as volcanic eruptions, wind erosion and biogenic emissions from plants and trees. Some of these factors are discussed below.

2.5.1. Vehicular and industrial emissions

In urban areas, automobile exhaust fumes may be accountable for 90 to 95 % of CO emissions and 60 to 70 % of NO_2 emissions (DEA, 2017). These emissions furthermore contribute to the creation and proliferation of photochemical smog in traffic-dense areas (DEA, 2017). There have been several studies documented in the literature that have looked at the linkage between OAQ and vehicular and industrial emissions (DEA, 2017; Choo *et al*., 2015; Lu *et al*., 2006; Baez *et al*., 2003). Hydrogen sulphide (H_2S) and CO_2 release are associated with the type of incineration and the properties of the fuel being used by vehicles. Lourens *et al*. (2016) identified industrial emissions as the main outdoor sources within the Mpumalanga province in South Africa. Similarly, Guttikunda *et al*. (2014) elucidated that ambient emissions of pollutants mainly originated from industrial clusters, petrochemical refineries, and power plants. Power generation by coal contributes to the emission of PM, SO_2, NO_x, and mercury (DEA, 2017). Results from studies carried out by Lu *et al*. (2006), Baez *et al*. (2003) and Schneider *et al*. (2001) indicated that VOC and NMHC concentrations are of vehicular and industrial origin. Nitrogen oxides, PM, O_3, NO_2, and VOCs have also been associated with vehicle emissions (Bo *et al*., 2017; Raysoni *et al*., 2017; Lee, 2016; Lu *et al*., 2016). This was further supported by the OSHA (2011) that found common pollutant sources outside buildings to be idling vehicles. All the aforementioned studies have established the influence of vehicular and industrial emissions on the concentrations of PM, O_3, SO_2, NO_2, CO, H_2S, NMHCs and VOCs and in outdoor environments.

2.5.2. Agricultural emissions and biomass burning

Agricultural emissions and residue burning lead to the formation of greenhouse gases and other carcinogenic pollutants such as CO_2, CO, NMHCs, SO_2 and NO_x (Chen *et al*., 2017). Additionally, VOCs and O_3 precursors are pollutants commonly encountered upon

inception and completion of agricultural and biomass burning events (Chen *et al*., 2017). Jain *et al*. (2014) found agricultural crop residue burning to contribute to CO_2, NO and methane (CH_4) concentrations in the outside environment, in addition to pollutants such as CO, NO_x, NMHC, SO_2, VOCs, PM, ammonia, and smoke. The major non-traffic origin of pollutants in Bhutan was biomass burning. Chen *et al*. (2017), Zhao *et al*. (2017), Wangchuk *et al*. (2015), Chanduka (2013) and Satyendra *et al*. (2013) reported a 33.10 to 86.65 % increases in PM concentrations during crop residue burning. This was supported by Cusworth *et al*. (2018), Liu *et al*. (2018) and Chen *et al*. (2017) who all found elevated PM concentrations arising from biomass burning, with exceedance in the WHO standard regularly found. Arslan and Aybek (2012) found PM originating from pollen blown from plants, in addition to human activities such as pesticide and fertilizer usage. Agricultural activities such as tilling and clearing were also precursors of elevated PM concentrations in ambient air (Arslan and Aybek, 2012).

2.5.3. Meteorological parameters

Meteorological parameters such as wind speed, wind direction, RH, rainfall and temperature can also affect OAQ. An upsurge in surface temperatures and a decline in wind speed are both linked with higher pollutant concentrations due to their influence on the photochemical processes that are responsible for the formation of some air pollutants (Ramsey *et al*., 2014). High amounts of rainfall have also been associated with lower pollutant concentrations since rainfall removes certain pollutants from the lower atmosphere (Ramsey *et al*., 2014). Huang *et al*. (2017) reported a significant negative correlation between air pollutants (CO, NO_2, $PM_{2.5}$, and PM_{10}) and temperature, whereas Zeng and Zhang (2017) reported higher O_3 concentrations correlated with higher temperatures. Ozone was found to have a positive correlation with temperature, whilst a negative correlation was established between O_3 and pressure, RH and wind direction (Hosseinibalam and Hejazi, 2012). Ozone concentrations generally increase with higher temperature, lower humidity levels and decreases in pressure. Both Zeng and Zhang (2017) and Hosseinibalam and Hejazi (2012) found increased SO_2, NO_x, NO_2 and CO concentrations when wind speed, temperature and RH were at lower values and pressure was elevated.

2.5.4. Volcanic eruptions and wind erosions

Volcanic eruptions can generate surplus amounts of ash, which in turn are imperilled to erosive forces causing damaging effects to ecosystems. Volcanic gas emissions primarily constitute of water vapour, CO_2, SO_2, $PM_{2.5}$, CO, H_2S, hydrogen chloride, CH_4 and other gases and have been incriminated as the source of SO_2 and PM in the atmosphere across different countries (Canyon and Campbell 2017; Balsa *et al*., 2016; Tam *et al*., 2016; Schmidt *et al*., 2015). Wind erosion is envisaged to be the precursor of airborne particulates. In addition to specific pollutants, suspended dust can be transported for lengthy distances, whilst TSP, PM_{10}, and $PM_{2.5}$ are transported for relatively minuscule distances (Sullivan and Ajwa, 2011). Saxton *et al*. (2001) established that PM_{10} concentrations were three to five times the maximum national AQ levels during dust storm days. Wagenbrenner *et al*. (2011) had also associated wind erosion with elevated PM concentrations.

2.5.5. Biogenic emissions

Plants produce and emit a wide array of phytochemicals during photosynthesis, with the majority of the phytochemicals being reactive VOCs (GHASP, 1999). Biogenic VOC emissions from vegetation constitute approximately 90 % of the global land-dwelling non-methane VOC emissions annually (Baghi *et al*., 2012). Biogenic VOC emissions are exceedingly dependant on biomass and vegetation activity and are influenced by seasonal changes (Baghi *et al*., 2012). Edwards *et al*. (2001) explored personal and residential indoor, outdoor and workplace settings, to identify VOC sources in Finland and found elevated concentrations of some VOCs in areas with an abundance of pine trees and conifers. Abou Rafee *et al*. (2017), Sartelet *et al*. (2012) and Hogrefe *et al*. (2011) all found low pollutant concentrations (O_3 and PM) emanating from biogenic sources with anthropogenic sources being the main contributors. Biogenic emissions tend to be principally made up of O_3 and PM_{10}, with these emissions themselves contributing minimally to the overall concentrations of these pollutants in the outside environment. There are few studies pertaining to biogenic emissions and pollutant diurnal and seasonal variation. Baghi *et al*. (2012) investigated the influence of flowering trees on urban atmospheric biogenic VOC emissions, and found varying rates of emission based on species types, with strong correlations attained with temperature. Churkina *et al*. (2017) reported that biogenic VOCs in Berlin contributed approximately 12 % to ground level O_3 formation during summer.

2.6. Air quality standards

Due to the significance of AQ in human health, various countries (including South Africa) and other organisations have established AQ standards for different chemicals and compounds. However, standards have not been established for all the IAQ and OAQ parameters of interest in this study. The ASHRAE has formulated IAQ guidelines for some of these parameters to determine the comfort levels of an individual within an indoor environment (see Table 2.2).

Table 2.2: The American Society of Heating, Refrigerating, and Air Conditioning Engineers standards for indoor air quality parameters in this study.

Parameter	Limit/Range	Reference Standard
Linear air velocity	< 0.25 m/s	ASHRAE Standards 62.1:2007
Dry-bulb temperature	23.25–28.00 °C (summer/spring)	ASHRAE Standards 55:2010
	20.50–25.50 °C (autumn/winter)	
Relative humidity	30–60 %	ASHRAE Standards 55:1999
Carbon monoxide	9.0 ppm	ASHRAE Standards 62:2010
Carbon dioxide	700 ppm + corresponding ambient concentration	

South African OAQ standards are enforced by the Department of Environmental Affairs (DEA) (DEA, 2009), Air quality act no. 39 of 2004. Additionally, the WHO has also implemented OAQ standards for selected parameters (Table 2.3). In connection to NMHCs and VOCs, individual compounds, gases, and solvents have standards; currently, no enforceable standard has been set for total NMHCs and VOCs in non-industrial settings. Hence, the standards as stated by the DEA and the WHO in Table 2.3 below need to be complied to. The above-mentioned pollutants are all regularly monitored within South Africa by the DEA and the South African Weather Service (SAWS). From a South African perspective, AQ standards for NO_2 and O_3 are exceeded predominantly at areas with a high density of road traffic, in addition to industrial and domestic fuel burning areas (DEA, 2017). Regarding SO_2, the standard is rarely exceeded in South Africa. However, petrochemical reactions, metallurgy, power generation, and some industrial processes contribute to SO_2 levels (DEA, 2017).

Table 2.3: The South African Department of Environmental Affairs and World Health Organization standards for outdoor air quality parameters in this study.

Parameter	Limit/Range
Ozone	120 µg/m^3 (61 parts per billion (ppb)) (0.061 ppm) – 8 hour* 100 µg/m^3 – 8 hour**
Sulphur dioxide	350 µg/m^3 (134 ppb) (0.134 ppm) – 1 hour* 24 µg/m^3 – 24 hour**
Nitrogen dioxide	200 µg/m^3 (106 ppb) (0.106 ppm)– 1 hour* 200 µg/m^3 – 1 hour**
Carbon monoxide	30 mg/m^3 (26 ppm) – 1 hour* 10 mg/m^3 (8.7 ppm) – 8 hour*
Hydrogen sulphide	150 µg/m^3 (0.1 ppm) – 24 hour***

* DEA, 2009; ** WHO, 2017; *** WHO, 2000a.

Carbon monoxide levels are generally low in South Africa, with key indicators being traffic emissions and fuel combustion activities (DEA, 2017). Hydrogen sulphide emissions in South Africa primarily emanate from landfill sites, fuel combustion activities and fish industries, with concentrations predominantly being within standards (DEA, 2017). All the above-mentioned indoor and OAPs become detrimental to humans on exposure to them via various routes. It is therefore critical to decipher the route of exposure to ascertain the health endpoints attributed to these air pollutants.

2.7. Human exposure and exposure pathways to air pollutants

Individuals are exposed to different pollutants on a day-to-day basis. Exposure can be elucidated by several characteristics such as duration, magnitude, frequency and the route of exposure (Silins *et al*., 2011). The duration of exposure can be defined as the total time frame whereby a person is in contact with a specific pollutant; whereas, magnitude refers to the concentration of the pollutant or contaminant of interest to which the individual is exposed (Silins *et al*., 2011). Frequency, on the other hand, denotes the regularity of the exposure to a substance (Silins *et al*., 2011). Air pollutants enter the human body via three main routes of exposure (Figure 2.1.), i.e. inhalation, ingestion and dermal absorption (Kampa and Castanas, 2008).

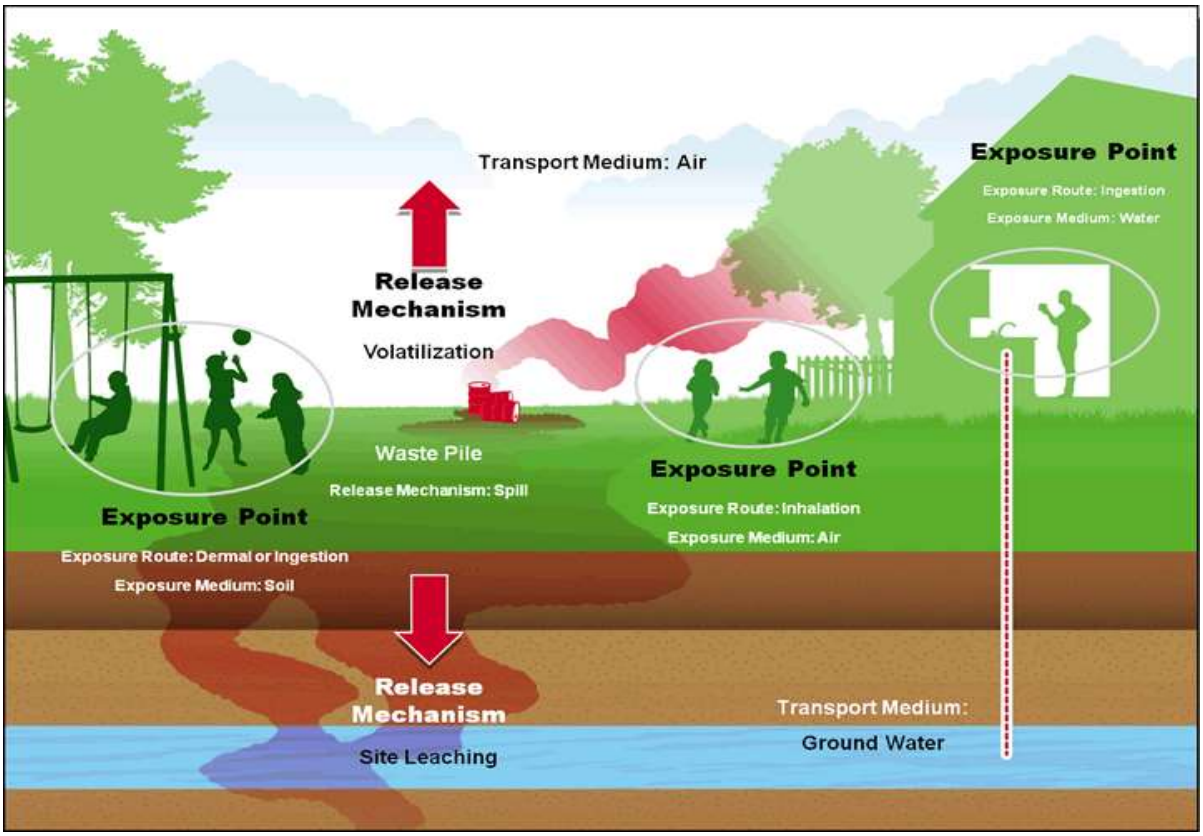

Figure 2.1: Different routes of exposure to air pollutants (USEPA, 2017b)**.**

Exposure through inhalation to air pollutants typically occurs when one breathes in contaminants in the form of vapours (VOCs), aerosols and PM (USEPA, 2016b), with this route presenting the most health risk. The ingestion route of exposure to air pollutants can occur through the consumption of contaminated water (pollutants settling in rivers or streams), food and chemical residues on surfaces or objects on which air pollutants have settled (USEPA, 2016b). Dermal exposure to air pollutants results from direct skin contact with pollutants such as indoor and outdoor dust (USEPA, 2016b). Children tend to have a greater exposure to air contaminants due to their lungs still being in the developmental stage (ALA, 2018). Furthermore, children tend to spend more time in the outdoor environment and are highly active in that environment, in addition to breathing in a greater volume of air as compared to adults (ALA, 2018).

2.8. Conclusion

The intention of this review was to collate studies done around the globe with reference to AQ, indoor and OAPs and parameters that determine IAQ and OAQ. Additionally, current IAQ and OAQ standards were identified that exist for various pollutants both locally and internationally and lastly various routes of exposure and exposure pathways to

pollutants are documented. It has been ascertained that most studies about AQ focus predominantly on IAQ, due to the extended periods of time spent indoors. It is clear from the research reviewed that AQ is quite a diverse entity that is affected by numerous factors. Indoor and OAQ are predominantly influenced by air pollutants in the surrounding environments emanating from natural and anthropogenic sources. Generally, as per studies done throughout the world, some of the most common parameters of air pollution studied in both environments are LAV, T_{db}, RH, O_3, SO_2, NO_2, CO, CO_2, PM_{10}, $PM_{2.5}$, and VOCs. Additional characteristics that influence IAQ, over and above the above-mentioned pollutants are: building characteristics, occupant densities, and the maintenance of HVAC systems. Contrastingly, OAQ is affected by factors such as vehicular, industrial and agricultural emissions, biomass burning, biogenic emissions and meteorological factors.

Generally, within an indoor environment, the ASHRAE guidelines are adopted as a comparative across all AQ studies worldwide. The parameters and pollutants that have been determined to comply with the ASHRAE guidelines of 1999, 2007 and 2010 are LAV, T_{db}, RH, CO, and CO_2. In the outdoor environment, the South African DEA (2009) standards and the WHO (2017) and WHO (2000a) standards have established standards for the following pollutants: O_3, SO_2, NO_2, CO, and H_2S. All the above-mentioned indoor and OAPs become detrimental to humans on exposure to them via various routes.

CHAPTER 3
RESEARCH METHODS AND ANALYTICAL PROCEDURES

3.1 Introduction

This chapter presents the study area, study design, study site, data collection tools used in the study, experimental and analytical procedures. Methods of data management and analysis are also presented and explained here. Lastly, the ethical considerations and the limitations of this study are presented.

3.2. Study area

3.2.1. Location and description

This study was carried out within the School of Molecular and Life Sciences (SMLS), FSA, UL, Mankweng, South Africa (Figure 3.1). The UL is situated in Mankweng (23.886° S, 29.718° E), midway between Polokwane and Magoebaskloof (UL, 2014). Ramoroka *et al*. (2016) described Mankweng as an example of a developing region because of its energy supply, dependable public services, and refined transport and communication development infrastructure. Within the university and in its surroundings, there are numerous projects currently being undertaken to construct hostels and other types of accommodation for students, erection of a mall and a sporting complex, in addition to an electricity generation plant/substation.

Figure 3.1: Map of the study area.

Three academic departments namely; The Department of Physiology and Environmental Health (PEH), The Department of Biochemistry, Microbiology and Biotechnology (BMBT), and The Department of Biodiversity (BIOD) were the specific study areas within UL. The PEH and BIOD buildings were constructed at the same time (about 30 years ago), with the BMBT building being constructed in the last two decades as an extension to the PEH building. The construction, structure, and design of the PEH and BMBT buildings were identical (four floors), whilst the BIOD building had a different structure (four floors and a basement/underground floor, with an internal cylindrical layout). The BMBT and BIOD buildings had office environments scattered throughout the various floors of the buildings, whereas in the PEH building, offices were based predominantly on the 1st floor of the building. All three buildings have tutorial, seminar, lecturing rooms, laboratories, and staff rooms. In the PEH building, computer-based practicals are generally carried out, with wet practicals being seldom undertaken within this building, whereas the BMBT building and to a lesser extent the BIOD building are predominantly used for wet practicals.

Concerning occupancy, regular occupants of all three buildings (staff and other personnel) were contacted to ensure their availability during sampling sessions. Hence, during the measurement taking processes, occupant densities were reflective of the normal day-to-day activities occurring within the three buildings of interest. In relation to the outdoor environment of the three buildings, the PEH building had functional generators situated on the south and east wings of the building and a car park in the north wing of the building. The BMBT building had functional generators present on the west wing of the building whilst a municipal waste bin was situated on the north side of the building. Car parks were found on the north, east and west wings of the BMBT building. The BIOD building had a functional generator on the north wing of the building; a municipal waste bin was situated on the south wing. There was no official demarcated parking area for the BIOD occupants, thus individual vehicles were parked across the south, east and west wings of the building.

Concerning HVAC systems within each building, the PEH building comprised primarily of air conditioners. Offices where no air conditioners were present, tended to use heaters and fans. Within the BMBT and BIOD buildings, all office spaces were equipped with air conditioners, with some offices having additional fans and heaters as well. Centralised air conditioners were the norm in offices and rooms adjacent to laboratories. Some offices across all three buildings had no provisioning of natural ventilation through windows and

as such made use of open doors as the only source of natural ventilation. All three buildings had exhaust fans situated in their laboratories to get rid of the vapours and gases produced indoors during practicals.

It is worthwhile to report that the south and east wings of the PEH building were adjacent to other buildings (BMBT and others which housed physics and chemistry laboratories). The BMBT building's south, east, and west (PEH) wings are occupied by chemistry, physics, BIOD and microbiology laboratories in close proximity. The BIOD building's north and south wings were also near other buildings (accountancy, geography buildings, and other lecture halls). The nearest weather station to the university is situated in Polokwane. The annual averages for selected meteorological parameters in Polokwane as per the statistics obtained from the SAWS are deliberated upon in the next section.

3.2.2. Climate

The highest temperature values were recorded during summer and the lowest during winter (SAWS, 2019a) as presented in Table 3.1. Similarly, the highest periods of rainfall were also seen across summer, with the driest spells recorded during winter (SAWS, 2019a) (Table 3.1). Relative humidity maxima were seen across summer with minima values recorded during the spring season (SAWS, 2019a). The prevalent wind directions in the study area were from the South and the South South-East as per the wind rose depicted in Figure 3.2. The atmospheric pressure at the site was 882.2 hectopascals (hPa). The climatological parameters were obtained from the SAWS (2019a) for the period 2011–2018 and are portrayed in Table 3.1 (summer and winter) and 3.2 (autumn and spring) below.

Table 3.1: Climatological parameter averages for the period 2011–2018 for summer and winter seasons (SAWS, 2019a).

Climatological parameters	Annual average	Summer						Winter					
		December		January		February		June		July		August	
		Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
T_{db} (8-hour) (°C)	5.0 (Minimum); 28.4 (Maximum)	15.9	27.7	16.2	27.9	16.3	28.4	5.0	21.7	4.9	21.2	7.0	23.9
Rainfall (mm)	0.0 (Minimum); 199.1 (Maximum)	27.2	199.1	20.2	157.8	6.2	170.5	0.0	2.0	0.0	2.4	0.0	19.2
RH (8-hour) (%)	35.5 (Minimum); 71.0 (Maximum)	49.6	66.7	52.3	71.0	55.1	67.2	44.5	56.5	46.2	60.4	36.8	52.0
Wind speed (m/s)	2.3 (Minimum) 4.6 (Maximum)	3.0	4.1	3.1	3.8	2.6	3.5	2.4	3.4	2.5	3.3	2.5	3.5

Table 3.2: Climatological parameter averages for the period 2011–2018 for autumn and spring seasons (SAWS, 2019a).

Climatological parameters	Annual average	Autumn						Spring					
		March		April		May		September		October		November	
		Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum	Minimum	Maximum
T_{db} (8-hour) (°C)	5.0 (Minimum); 28.4 (Maximum)	15.0	27.8	11.8	25.2	8.3	24.0	10.5	26.9	12.4	27.1	13.9	27.5
Rainfall (mm)	0.0 (Minimum); 199.1 (Maximum)	3.1	95.0	0.2	106.6	0.0	34.5	0.0	23.3	6.7	92.3	24.9	147.5
RH (8-hour) (%)	35.5 (Minimum); 71.0 (Maximum)	54.5	68.9	53.2	70.2	44.7	59.4	35.5	49.5	43.7	55.9	41.2	58.3
Wind speed (m/s)	2.3 (Minimum) 4.6 (Maximum)	2.4	3.2	2.4	2.8	2.3	2.7	3.3	4.2	3.8	4.5	3.8	4.6

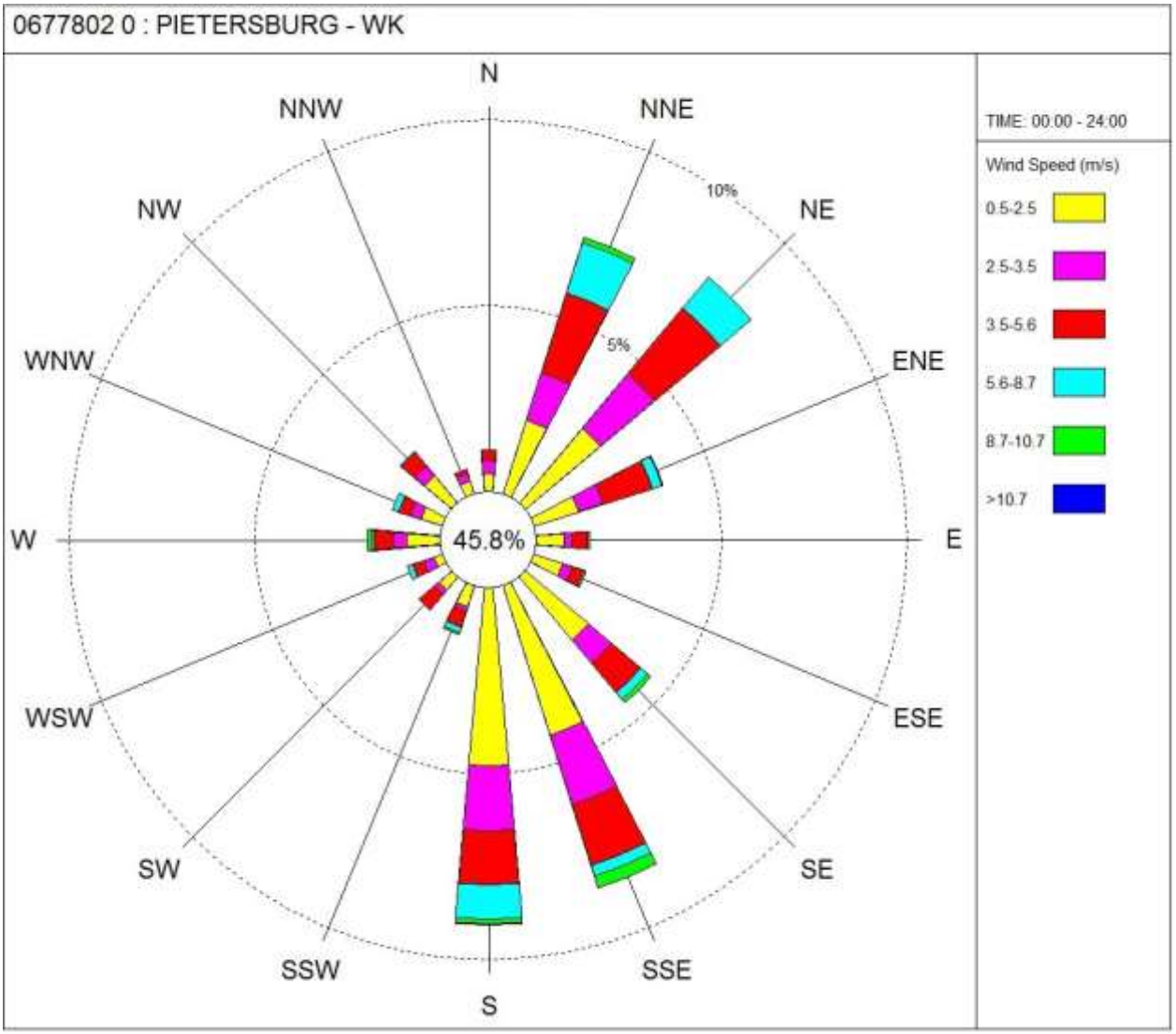

Figure 3.2: Wind rose for Polokwane depicting the average wind direction for the period 2011–2018(SAWS, 2019a).

3.3. Study design

This study was conceptualised as a quantitative, cross-sectional study, to evaluate the IAQ and OAQ of selected buildings at UL. The following 11 AQ parameters were measured both in the indoor and outdoor university environment: LAV (m/s), T_{db} (˚C), RH (%), O_3 (ppm), SO_2 (ppm), NO_2 (ppm), CO (ppm), CO_2 (ppm), H_2S (ppm), NMHCs (ppm) and VOCs (ppm).

3.4. Sampling sites

All departmental buildings within the FSA, in SMLS, namely: PEH, BMBT, and BIOD were included in this study. The study sites comprised of offices, lecture and tutorial/seminar

rooms in these buildings (Figure 3.1). As determined by Yau *et al*., (2012) laboratory-based environments tend to have relatively high NMHC and VOC concentrations that affect AQ. Laboratories within the buildings were however excluded from the study due to the presence of flammable chemicals present in the laboratories. The use of a hot-wire anemometer in the presence of these chemicals could cause a fire incident. The number of sampling points needed in each environment was calculated based on a sampling time frame from 7h30 to 16h30 (9 hours-inclusive of a one-hour lunch break). The study was initially piloted to ascertain the number of sites that could be sampled during the course of a day. Based on the pilot study, a maximum of 20 indoor and 20 outdoor environments were sampled daily since sampling at each point took approximately 12 minutes. Hence, a total of 40 indoor and outdoor points around each building were sampled. Since each sampling point took 12 minutes, a grand total of 480 minutes was required for sampling these 40 sites. Occupants of the different buildings were only available between 7h30 and 16h30 (9 hours) and so the maximum number of sampling points across both environments that could be sampled was 40, to adhere to these specific time frames. The 20 indoor sampling sites included office, lecture and tutorial/seminar rooms in each of the buildings that were included in the study in each department.

A total of 60 rooms were therefore sampled in each department per season. The outdoor environment also comprised of an equivalent number of 20 measuring points per department (five measurements per cardinal direction taken outside the departmental building), leading to a total of 60 outdoor measurements across the three departments of interest per season. Hence for the three buildings, 360 measurements were made in total per season, leading to a grand total of 1440 measurements per parameter over the four seasons in both the indoor and outdoor environments across the three buildings. During the measurement of various AQ parameters, offices were occupied by their respective staff members who were engaged in their typical day-to-day duties under regular conditions as prescribed by the ASHRAE Standard 62-2001, (ASHRAE, 2010). This was also adhered to when measurements were taken in the lecture and tutorial/seminar rooms. The direct reading Q-Trak IAQ monitor, three handheld portable Aeroqual AQ monitors, and a hot-wire anemometer were made use of for the measurements of the 11 AQ parameters of interest in this study. Measurements of the AQ parameters commenced in January 2018 till September 2018 [inclusive of all four seasons, i.e. Summer (29th January–14th February 2018); Autumn (9th to 25th April 2018); Winter (2nd to 18th July 2018)

and Spring (3rd to 19th September 2018)]. The southern hemisphere has the following seasonal calendar, which is what was utilised in this study (SAWS, 2019b):

- 1 December to 28/29 February: Summer;
- 1 March to 31 May: Autumn;
- 1 June to 31 August: Winter and
- 1 September to 30 November: Spring.

Measurements were taken in triplicate (i.e. over three consecutive days) in both the indoor and outdoor environments of the buildings. Data gathering was carried out over the four seasons. Each sampling session per season consisting of three consecutive days, across all three departments in both the indoor and outdoor environments.

3.5. Measurement of air pollutants in the study area.

All the air pollutants of interest in the current study, (O_3, SO_2, NO_2, CO, CO_2, H_2S, NMHCs, and VOCs) were measured using a direct reading Q-Trak IAQ monitor (model number: 7575-X, serial number: 7575X1318008), and three handheld portable Aeroqual AQ monitors (series 500, serial numbers: S500L 1310171-3747; S500L 1310171-3729 and S500L 0812171-3899). Due to the storage of several laboratory chemicals across BMBT and BIOD buildings, it was deemed imperative to measure both NMHC and VOC concentrations to try and isolate points of origin. The direct reading Q-Trak IAQ monitor is a multi-function, handheld test instrument that uses a probe to simultaneously measure T_{db}, RH, CO and CO_2. It provides quick and accurate information to measure and assess these AQ parameters (TSI, 2018a) (Figure 3.3). The handheld, direct-reading portable Aeroqual series 500 monitor (Figure 3.4) allows precise instantaneous measurements of air contaminants, commonly incorporated for short-spanned AQ studies and checking air pollution “hot spots” (Aeroqual, 2018). The series 500 Aeroqual monitor has a unique sensor head format that allows sensor heads for any pollutant to be detached and switched within a short time frame, allowing an individual to quantify several gases as needed (Aeroqual, 2018). Each sensor head contains a fan that works continuously or periodically to draw a sample of air into the sensor (Aeroqual, 2018).

Figure 3.3: Indoor and outdoor parameter sampling using the Q-Trak IAQ monitor.

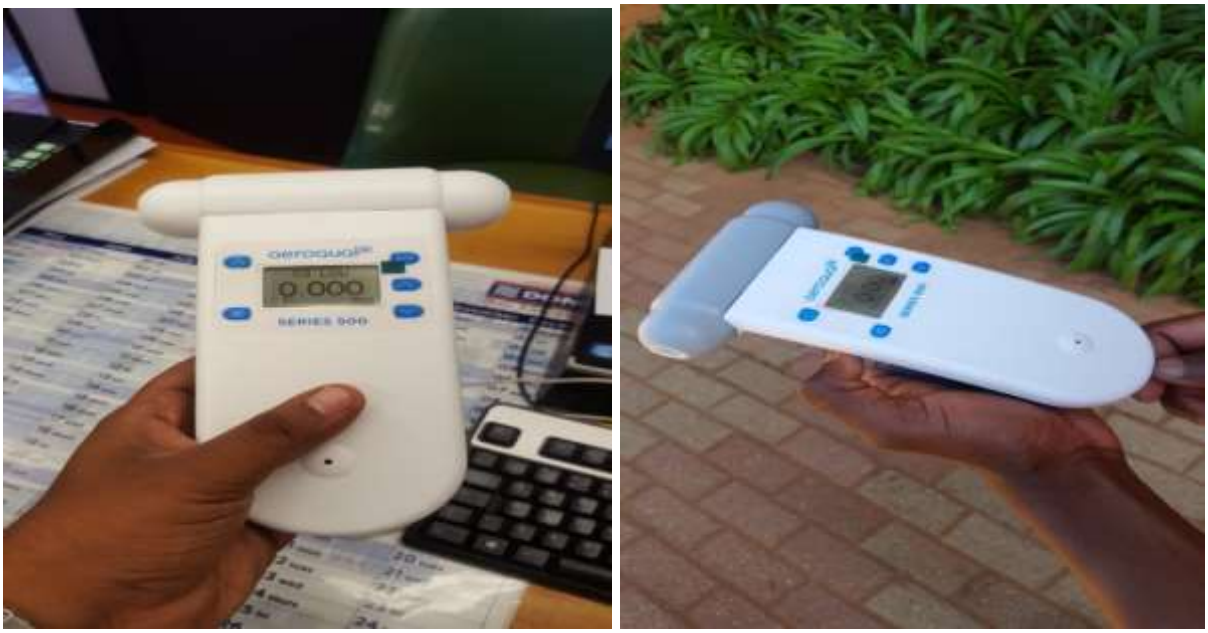

Figure 3.4: Indoor and outdoor pollutant sampling using the Aeroqual portable handheld meter.

Once sampled, the air is exhausted from the opposite side of the sensor head, ensuring representative sampling, measurement validity, and accuracy (Aeroqual, 2018). Both the Q-Trak IAQ monitor and the portable Aeroqual handheld AQ meters have manual and automatic logging capability; however, this function was not used, due to it draining the battery fairly rapidly on both instruments once activated. The real-time raw measurements depicted on the display screen for each of the instruments were recorded onto datasheets (Appendix A1, Table A1.1). The following pollutants were quantified with the direct-reading portable Aeroqual series 500 monitors: O_3, SO_2, NO_2, H_2S, NMHCs, and VOCs.

Furthermore, the sensor heads have active fan sampling. Since the measurements taken in the indoor environment were area measurements, monitors were placed approximately 152.4 to 177.8 cm above the floor in areas having airflow. Measurements were taken around the occupant's desk or workstation as per regulatory operating procedures described in the IAQ Handbook (2013) (Figure 3.5). The above-mentioned procedures were adhered to and followed also when taking measurements in the outdoor environment (TSI, 2018a; IAQ Handbook, 2013), due to no current national or international guidelines and/or protocols available on the quantification of the parameters of interest in the outdoor environment.

Figure 3.5: Linear air velocity indoor and outdoor sampling using the ALNOR TSI velometer.

Precautionary and preventative measures were taken prior to and during measurements to ensure that the sensor heads were not obstructed or hindered in any way during the measurements. Outdoor AQ parameter measurements were taken in tandem with IAQ parameter measurements across all the three departments. Observations made during sampling included the prevalent weather conditions, the extent of airflow across both environments, the existence of fans, heaters and/or air conditioners and other electrical appliances, the number of people present in the different rooms sampled, movement of people and vehicles in and around the sampling area and the presence of any potential source of pollutants. These observations were documented during the measurements of all AQ parameters for all three departments throughout the entire sampling period (Appendix A1, Table A1.1).

3.5.1. Measurement of linear air velocity

Linear air velocity or air flow rate was evaluated using the TSI ALNOR velometer thermal anemometer (model number: AVM410, serial number: AVM411146006). The TSI ALNOR velometer thermal anemometer is a digital instrument with good accuracy and precision, including a telescopic straight probe (TSI, 2018b). Several studies have made use of the ALNOR velometer to ascertain airflow rate in different environments (Chandler *et al*., 2017; Halloran *et al*., 2014; Funk, 2011; Cameron, 2006). The anemometer measures air velocity real-time manually and accurately, with a range of 0.0 to 20.0 m/s, in addition to having a digital display (TSI, 2018b). The hot-wire anemometer has a built-in heated element and the airflow rate is measured based on the drop in temperature of the heated element caused by the colder airflow within the surrounding environment (TSI, 2018b). The anemometer was positioned parallel to the work area of the occupant while ensuring that no obstacles or barriers were present between the device and the airflow to the desk or workstation. For purposes of ensuring the validity, precision, and accuracy of each measurement using the anemometer, the sensor head was always in direct contact with the incoming airflow.

3.5.2. Measurements of temperature and relative humidity

Dry-bulb temperature and RH measurements were also taken with the aid of the direct reading Q-Trak IAQ monitor as per the ASHRAE standards (ASHRAE, 2013; ASHRAE, 2010). This instrumentation has a relatively higher sensitivity to temperature deviations than to RH deviations. To overcome this sensitivity, the instrumentation is equipped with temperature-compensating circuits. This instrumentation has been used in other studies (Chandler *et al*., 2017; Funk, 2011) to measure the same parameters. The monitor was placed at the desk or workstation in the indoor environment. In instances where desks or workstations were absent, measurements were taken at approximately 152.4 to 177.8 centimetres above the floor in the open space available as stated in the IAQ Handbook (2013). In the outdoor environment, the same measurement technique used indoors were adopted to ensure uniformity and accuracy (TSI, 2018a). The air temperature was measured in degrees Celsius, whilst RH was measured in percentage.

3.6. Quality control measures

The direct-reading IAQ monitor and the velometer thermal anemometer were calibrated by the TSI Company, whereas the three handheld portable AQ monitors were calibrated

by the supplier of the equipment itself, Aeroqual in New Zealand (see attached calibration certificates in Appendix C). Although calibration certificates are usually valid for a year, it was deemed by the suppliers (three handheld portable AQ monitors) that, calibration of the instrumentation be done depending on how vigorously the equipment was being used, since the monitors would automatically display a “sensor head aging”, followed by a “sensor head failure” pop up message when re-calibration was deemed necessary. Data were collected in triplicate (over three consecutive days) per season, per sampling point, to increase the validity and precision of the data collected. The areas, positions, and instrumentation setup used for measurement were as recommended by the IAQ Handbook (2013) and the guidelines given by the instrument suppliers (Aeroqual, 2018; TSI, 2018a; TSI, 2018b).

3.7. Data management and analysis

All AQ parameter measurements, taken both inside and outside the buildings were captured and thereafter double-entered into Microsoft Excel 2015. Once verified and cleared, the data was exported to the Statistical Package for the Social Sciences (SPSS), version 25. Daily averages for the different AQ parameters were computed by averaging the triplicate readings attained across both environments of each building. This was repeated for all three departments and across all four seasons.

As per the second objective, to evaluate differences between IAQ and OAQ amongst the selected buildings in the university, the daily averages over the three days of measurement across both the indoor and outdoor environments were used in calculating the mean of each parameter for the specific environment. The means of indoor and outdoor measurements in the different buildings were compared using the Friedman and Wilcoxon signed ranks tests to determine differences between the two environments (indoor and outdoor environments) and among the three buildings (BMBT, BIOD, and PEH). All 11 AQ parameter seasonal averages per building were compared with one another in both the indoor and outdoor environments. Due to multiple comparisons being made, i.e. PEH-BMBT; PEH-BIOD and BMBT-BIOD, the Bonferroni adjustment was applied (Lee and Lee, 2018), resulting in a new level of statistical significance (i.e. 0.05/3 = 0.017). Details of statistical tests carried out are presented in Appendix A2, Tables A2.1 and 2.2 at the end of this report.

In line with the third objective, compliance of indoor and outdoor AQ parameter concentrations with various AQ standards were evaluated by comparing the means of the respective AQ parameters to their corresponding guidelines and standards. Indoor measurements were compared against the ASHRAE guidelines to determine conformity, whilst the outdoor measurements were compared with the DEA and WHO standards. Both comparisons were made visually with the aid of graphs constructed in Microsoft Excel 2015.

For all statistical analyses, a p-value of < 0.05 was set to be statistically significant, with exceptions seen in calculating the differences between IAQ and OAQ amongst the selected buildings and the seasonal differences between IAQ and OAQ parameters. The distribution of each parameter was presented using a bar graph in addition to the Shapiro-Wilk test computed to test for normality. Furthermore, the calculated mean, median and mode of the different parameters were compared to one another to further ascertain whether the distribution was a normal Gaussian distribution or a non-Gaussian distribution. Since all the AQ parameters displayed a non-Gaussian distribution, the non-parametric Spearman's correlation coefficient test (to identify statistically significant correlations between indoor and outdoor readings for each AQ parameter) was carried out to substantiate the findings of the Wilcoxon signed ranks test for significant differences between indoor and outdoor readings for each AQ parameter.

The fourth objective was to evaluate seasonal differences between IAQ and OAQ parameters. The Friedman test was also computed to determine whether differences obtained from measurements of the different AQ parameters during the different seasons were significant. All the AQ parameters that were found to have significant differences (i.e. $p < 0.05$) between the four seasons were taken forward for further analyses, by applying a post hoc test, the Wilcoxon signed ranks test to identify the specific seasons where the differences were found to occur. In the Wilcoxon signed ranks test, the following seasonal comparisons were made to ascertain statistically significant seasonal differences in both the indoor and outdoor environments for each AQ parameter: summer vs. autumn, summer vs. winter, summer vs. spring, autumn vs. winter, autumn vs. spring and lastly spring vs. winter (i.e. six comparisons). Due to multiple comparisons being made, the Bonferroni adjustment was again applied to the outputs attained from the Wilcoxon signed ranks test to account for the Type I error that arises when declaring a result as statistically

significant (Lee and Lee, 2018). The initial p-value that was set as being statistically significant ($p < 0.05$) is divided by the number of comparisons being run (i.e. six comparisons), thereby resulting in a new level of statistical significance (i.e. 0.05/6 = 0.008). All the results attained from the Wilcoxon signed ranks test for purposes of determining seasonal differences were deemed to be statistically significant once the p-value was less than 0.008. All the above-mentioned analyses were conducted in SPSS version 25. Details of statistical tests can also be found in Appendix A2, Tables A2.3 and A2.4 at the end of this report.

To identify the best predictors (the factor with the greatest influence) of AQ, a multiple regression analysis was run. The effect of environments (indoor and outdoor), buildings (PEH, BMBT, and BIOD) and season (summer, autumn, winter, spring) on the different AQ parameters was carried out in SPSS version 25, by running a multiple regression analysis. The model summary initially excluded AQ parameters based on the correlation strength, i.e. partial correlations. In this study, an R^2 value of less than 0.996 was considered by the model to render a partial correlation. An analysis of variance (ANOVA) was the next step in the model to identify statistically significant differences between the annual AQ averages and the various predictors per parameter. Lastly, the coefficient of determination was calculated to determine the variable that was deemed to be the best predictor of the annual AQ average per parameter. See Appendix A2, Tables A2.5 to 2.7 for results of the best predictor statistical analyses.

The last aspect of data analysis was the determination of an air quality index (AQI) / air pollution index (API) for the university environment. A pollutant's index is defined as its concentration expressed as a percentage of the relevant air pollutant standard (USEPA, 2015b). Refer to the formula below:

$$API = (pollutant\ concentration \div pollutant\ standard\ level) \times 100$$ (USEPA, 2015b)

The USEPA (2015b) computes an AQI by firstly calculating pollutant averages, thereafter calculating a pollution index and lastly by calculating an AQI summary. The AQI was determined only for the following parameters: O_3, NO_2, SO_2, CO, and H_2S. An outdoor guideline or standard does not exist for LAV, T_{db}, RH, CO_2, NMHCs, and VOCs and as such, no AQI was determined for these parameters. According to USEPA (2015b), an

index value of 100 reflects a pollutant concentration level corresponding to the AQ standard of the respective pollutant. The AQI for the different seasons and buildings were calculated and the AQ was classified as directed by the USEPA (2015b).

3.8. Ethical considerations

This study received ethical clearance from the UNISA CAES General Ethics Review Committee (reference number: 2017/CAES/132 – Appendix B1), in addition to informing the Turfloop Research Ethics Committee of UL, where the study was conducted. In this study, the participants were informed about the purpose of the study and how the data will be collected. On the consent form, (Appendix B2: Participant information leaflet and informed consent documents) the purpose of the study was explained in detail.

Permission to carry out the study in UL, within the FSA, in SMLS was obtained from the university Registrar, the Vice-chancellor and Principal, the Dean of the Faculty, the Facilities Manager, the Occupational Health and Safety Officer, the Director of the School, as well as the Head of Departments. With regards to offices, only staff members who gave consent for measurements to be taken within their offices were included in the study with the remainder of indoor environments being seminar/tutorial rooms. The approval and support letters attained from the above-mentioned personnel are included in Appendix B3.

Measures were put in place to ensure the confidentiality of the participants so that they could not be identified from the study results through the implementation of unique identifier codes for measurements collected from office, lecture and tutorial/seminar rooms across the three departments. All personal information was secured and locked in a filing cabinet, only accessible to the principal researcher.

3.9. Limitations of the study

Due to complications arising from the unavailability of personnel within their office spaces when taking measurements in triplicate (over three consecutive days), sampling could not be done at the same time over the three days in some sites. Air pollutant concentrations may sometimes vary at different times of the day, hence taking these measurements at different times of the day may lead to variant pollutant concentrations or possible outliers that may affect the calculated averages.

The sensor heads incorporated to measure VOCs and NMHCs measured these pollutants holistically and as a whole as opposed to measuring each VOC or NMHC individually. The results of VOCs and NMHCs therefore reflect total concentrations of all the compounds classified as such in the study area and not the concentration of individual VOC and NMHC compounds.

A common aspect deciphered in literature concerning the effects on air pollutant concentrations was wind direction, which was not included in this study; hence, the effect of long-range transport and the prevalent wind direction per season could not be documented. The wind direction prevalent in Polokwane during the study period was obtained from data from the SAWS obtained from a weather station which was quite far from the study site.

CHAPTER 4
PATTERNS OF INDOOR, OUTDOOR AND BUILDING AIR QUALITY IN THE UNIVERSITY ENVIRONMENT

4.1. Introduction

The IAQ and OAQ in a university environment is influenced by several factors. Some of these factors include building occupancy, proximity of buildings to sources of air pollutants, air exchange between the indoor and outdoor environments, proximity to other buildings and the activities being carried out in those buildings. In this chapter, differences in some factors which affect IAQ and human comfort in the indoor environment of selected buildings in the university studied are presented. The patterns of each AQ parameter across both environments in the selected university buildings are then presented and discussed. An explanation of the differences and correlations observed between and among these parameters is also presented.

4.2 Site observations during measurements in the various buildings

During measurements, observations of the surroundings were made. Overall there were fans, heaters and/or air conditioners used by staff within their respective office spaces. Electrical appliances such as laptops, desktops, tablets, cell phones, lights and kettles were used within the indoor environments. The watering of plants in offices and the presence and use of water baths in laboratories during practicals were also noted. The number of people within the indoor environments generally ranged from about two to ten people, with occasionally larger groups of people encountered during practical timeslots. The source of indoor air flow was primarily from open doors, windows and functional fans, heaters and/or air conditioners. There were instances across the three buildings where no ventilation was observed.

The presence of solvents and other chemical reagents within the laboratories and in nearby storage rooms was also observed in these buildings. Air pollutant generating activities such as cooking, cleaning and painting initiatives were going on in some buildings. In general, during sampling, there were instances when the weather conditions varied. Weather conditions included clear, few clouds, cloudy and overcast conditions accompanied by sunny, misty, foggy and rainy days. Air movements (winds) could be described as slightly windy to extremely windy. Normal daily activities including student,

personnel and vehicular movement across the building surrounds were commonly encountered. Potential sources of emissions such as the exhaust/stacks of laboratory fans were also present around the buildings where laboratories were located. In the vicinity of the sampling sites were municipal garbage bins and soil and vegetation environments. Furthermore, renovation initiatives were also noted in some buildings across the study sites.

4.3. Characteristics of the indoor and outdoor atmosphere in the study area

4.3.1 Linear air velocity

Mean indoor LAV values in the PEH building ranged from 0.000 to 0.243 m/s, with a mean of 0.022 m/s. Indoor LAV values for both the BMBT and BIOD buildings ranged from 0.000 to 0.320 m/s with mean values of 0.021 and 0.018 m/s respectively. The highest outdoor LAV values in the study area were obtained outside the BIOD building (0.789 m/s) and the lowest outside the BMBT building (0.446 m/s) (Figure 4.1). The mean outdoor LAV readings for the current study varied between 0.113 to 2.460 m/s across all three buildings of interest. Details of the LAV results can be found in Appendix A1, Tables A1.2 to 1.4.

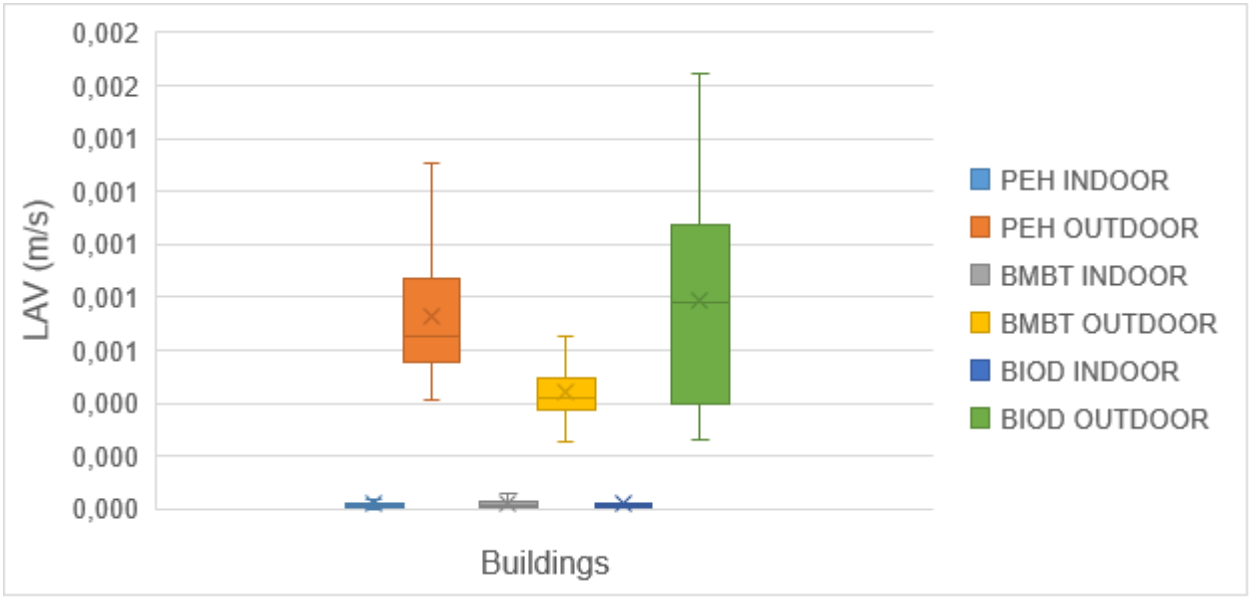

Figure 4.1: Mean indoor and outdoor linear air velocity values across the three buildings studied.

The results of the Wilcoxon signed ranks test comparing indoor and outdoor LAV values (Table 4.1) show that there were significant differences in LAV values between both environments across all three buildings ($p < 0.01$). This difference is further substantiated

by the extremely low I/O ratios calculated for LAV (mean PEH I/O ratio: 0.034, mean BMBT I/O ratio: 0.047 and mean BIOD I/O ratio: 0.022). The LAV values obtained in this study are consistent with results from indoor (Budiakova, 2017; Choo *et al*., 2015; Jurado *et al*., 2014; Yau *et al*., 2011) and outdoor (Budiakova, 2017; Mohammadyan *et al*., 2017; Lu *et al*., 2016; Choo *et al*., 2015; Wangchuk *et al*., 2015; Yau *et al*., 2011; Al-Rehaili, 1999) AQ studies in similar environments.

Table 4.1: Wilcoxon signed ranks and Spearman correlation test comparing indoor and outdoor linear air velocity values in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-3.920[a]	0.000	-0.098	0.681
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.920[a]	0.000	0.261	0.266
Biodiversity (BIOD)	20	-3.921[a]	0.000	-0.147	0.536

a. Based on negative ranks.

Across the indoor environment, the factors most likely contributing to indoor LAV values are the use of HVAC systems and the provisioning of natural ventilation. Outdoor LAV values are likely to have been influenced by the prevailing wind speed and the tunnelling effect created as a result of the proximity of buildings on campus to each other. A building's proximity to other buildings and structures, the height of a building and the wind direction affect and contribute to the tunnelling effect (Kuo *et al*., 2015). This effect increases wind speeds and cause strong air movements between buildings and other structures (Kuo *et al*., 2015) which might have resulted in the higher outdoor LAV values obtained in this study compared to the indoor values.

Though LAV readings across the outdoor environment were higher than indoor LAV values, these values are much lower than the average wind speed for the area. The SAWS (2019a) station data reports a minimum of 2.3 to a maximum of 4.6 m/s for the period 2011–2018 in Polokwane. This difference is however explained by the distance between the weather station where data is collected and the study area. It is expected that buildings with a high outdoor LAV would have high indoor LAV values due to outdoor air infiltration

into the indoor environment during times of natural ventilation provisioning. However, in this study, the Spearman's correlation test to identify this relationship depicted no significant correlations between indoor and outdoor LAV in the three buildings studied ($p > 0.05$) (Table 4.1). Hence, the differences observed between both environments could be due to the inherent characteristics of the respective environments rather than air exchange between the indoor and the outdoor environments.

Linear air velocity readings within a building are mostly affected by the availability of ventilation systems which may include open doors and/or windows, building entrances, mechanical ventilation systems including fans, heaters, and air conditioners. The three buildings assessed in this study are characterized by different structures and fittings and are used for different purposes. The PEH building generally functioned as an office building but also accommodated tutorials/seminars, lectures and computer-based practicals. The BMBT and BIOD buildings' primary functions were for laboratory practical sessions. However, tutorial/seminar rooms and staff offices were also present in these two buildings. These differences in building function did not seem to have any effect on the indoor LAV readings as no differences in indoor LAV values were observed between buildings ($p = 0.504$).

Outdoor LAV, on the other hand, is affected by factors which include, but are not limited to the proximity of buildings to one another and the structure of the buildings themselves. Tan *et al*. (2002) indicated that a building's structure affects the outdoor LAV, an effect which Vallero (2008) attributed to the turbulence which is created around the building as the wind comes into contact with it. Significant differences were observed in the mean annual outdoor LAV between the PEH and BMBT buildings and between the BMBT and BIOD buildings ($p < 0.01$). The mean annual outdoor LAV values in the PEH and BIOD buildings were quite high compared to what was obtained in the BMBT building. This might have been caused by the closeness of these buildings to surrounding buildings, which could have created a tunnelling effect, subsequently increasing the outdoor LAV values especially in the south and east wings of the PEH building and the south and north wings of the BIOD building. The BIOD building also had a tower-like and cylindrical structure in addition to being the tallest building amongst the three buildings studied which might have caused an increase in the outdoor LAV. According to Boeker and van Grondelle (2001), wind speed increases proportionally with the height of a building which may explain the

higher LAV values observed around the BIOD building. The BMBT building, on the other hand, had numerous abrasive surfaces and supporting structures on its exterior compared to the PEH and BIOD buildings which could have reduced the outdoor LAV for the building.

4.3.2 Dry-bulb temperature

Temperature has been documented widely in literature as a parameter that affects the chemistry and behaviour of air pollutants. Mean indoor temperature readings in the PEH building ranged from 15.2 to 26.1 °C, with an average of 21.9 °C. Similar mean indoor temperature readings were observed in the BMBT and BIOD buildings (i.e. 21.2 and 21.1 °C respectively). Literature reports a range of 13.0 to 26.0 °C for indoor temperature readings within school classrooms and university lecture halls in Europe and Asia (Budiakova, 2017; Kalimeri *et al*., 2016; Choo *et al*., 2015; Fadeyi *et al*., 2014; Gao *et al*., 2014), which is comparable to the current study findings. Outdoor T_{db} values, according to SAWS (2019a) data from 2011–2018 in Polokwane, ranged from 5.0 to 28.4 °C. Most of the outdoor T_{db} values obtained in the current study fell within the values reported by the SAWS (2019a) data, but some sites had values ranging between 7.5 and 33.8 °C, which was slightly higher than the average values for the region (SAWS, 2019a). Outdoor T_{db} readings were generally lower than indoor T_{db} readings, except in the BIOD building (Figure 4.2).

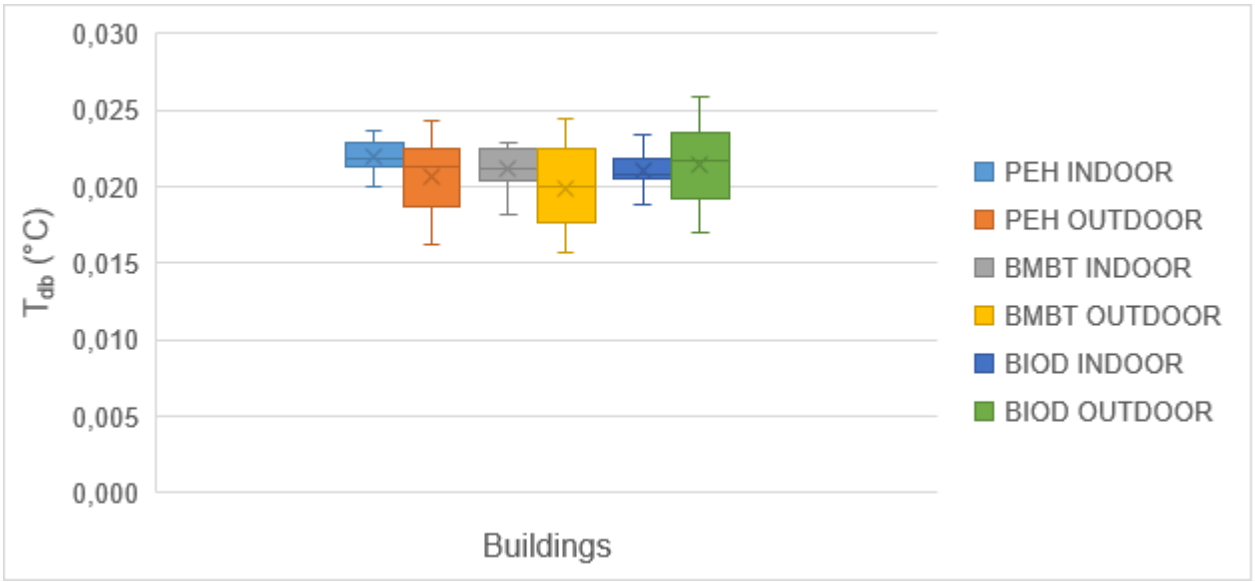

Figure 4.2: Mean indoor and outdoor dry-bulb temperature readings across the three buildings studied.

Other studies have found an outdoor T_{db} range of 1.0 to 28.0 °C in school, university and home environments (Mohammadyan *et al*., 2017; Lu *et al*., 2016; Wangchuk *et al*., 2015; Jovanovic *et al*., 2014; Alves *et al*., 2013), which is lower than the current study findings. Most of the aforementioned studies were however conducted in colder regions and countries, which may explain the difference. Significant differences in T_{db} readings were found between both environments in the PEH and BMBT buildings whereas, in the BIOD building, the differences between the indoor and outdoor T_{db} readings were insignificant.

Table 4.2: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor dry-bulb temperature values in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-2.764[a]	0.006	0.912*	0.000
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-2.912[a]	0.004	0.830*	0.000
Biodiversity (BIOD)	20	-1.120[b]	0.263	0.903*	0.000

a. Based on negative ranks;
b. Based on positive ranks;
* Correlation is significant at the 0.01 level (2-tailed).

Factors that affect indoor and outdoor T_{db} levels include natural ventilation provisioning (Budiakova, 2017; Gao *et al*., 2014; Pereira *et al*., 2014), occupant densities (Fadeyi *et al*., 2014; Gao *et al*., 2014; Pereira *et al*., 2014) and the use of HVAC systems by building occupants (Budiakova, 2017; Fadeyi *et al*., 2014). An occupant's individual preference may lead to them using fans or heaters or air conditioners in their respective indoor environments creating a gradient between indoor and outdoor temperatures (Lee and Lee 2015). The personal preferences of occupants with regard to their individual levels of comfort might have caused differences observed between the indoor and outdoor environments. The Spearman's correlation test indicated strong positive correlations between both environments ($p < 0.01$). Generally, the movement of air is caused by a temperature or pressure difference (Enviropedia, 2019). Similarly, as is the case across pressure gradients, the movement of air occurs from a high-temperature region to a low-temperature region (Enviropedia, 2019). Correlation between indoor and outdoor T_{db} could be explained by the differences in pressure and temperature between these environments. The warmer indoor environment causes warm air to rise, creating a

pressure gradient at the bottom of the building which draws outside cold air into the building. Hence, air movement would occur from the outdoor environment toward the indoor environment. This is further supported by the calculated I/O ratios for the T_{db} readings which were all close to one for all three buildings, indicating a move towards equilibrium temperature conditions. Details of the T_{db} results can be found in Appendix A1, Tables A1.2 to 1.4.

Indoor T_{db} readings were found to be significantly different between the PEH and BMBT building and between the PEH and BIOD buildings ($p < 0.01$). Indoor T_{db} tends to be affected by the function of a building, the thickness of the walls of the building and the materials used in its construction (Boeker and van Grondelle, 2001). The PEH building is primarily used to host lectures and therefore accommodates a huge number of students. The higher indoor T_{db} readings for this building would have been caused by high occupant densities, the physiological processes of human respiration, body heat dissipation and perspiration (Amasuomo and Amasuomo, 2016; Lim *et al*., 2008) which all give off heat, contributing to the indoor T_{db} values measured. In the BIOD building, outdoor louvers covering the windows in addition to the thicker walls of the building would have reduced the direct heating of the indoor environment, resulting in the lower values observed. Across the outdoor environment, T_{db} readings were significantly different between the PEH and BMBT buildings, the PEH and BIOD buildings, and the BMBT and BIOD buildings ($p < 0.01$). The PEH and BMBT buildings were in close proximity to one another and to other buildings as well. This meant that the building is shaded most of the time, a condition that would have contributed to the lower T_{db} values measured across PEH and BMBT compared to the BIOD building.

4.3.3. Relative humidity

The role of RH in thermal comfort and atmospheric pollutant behaviour has been well documented. Mean indoor RH readings in the PEH building ranged from 26.9 to 70.4 %, with a mean of 46.7 %. Similar mean indoor RH readings were observed in the BIOD (a range of 15.9 to 65.4 %, with a mean of 46.3 %) and the BMBT (a range of 24.5 to 70.5 % with a mean of 49.3 %) buildings. The highest annual mean outdoor RH value was obtained outside the BMBT building (50.0 %) and the lowest outside the BIOD building (41.6 %) (Figure 4.3). Details of the RH results can be found in Appendix A1, Tables A1.2 to 1.4. Outdoor RH levels are known to fluctuate between 18.0 to 89.7 % across school

classrooms, university environments, offices, and homes (Mohammadyan *et al*., 2017; Lu *et al*., 2016; Wangchuk *et al*., 2015; Jovanovic *et al*., 2014; Alves *et al*., 2013; Mentese *et al*., 2012; Al-Rehaili, 1999).

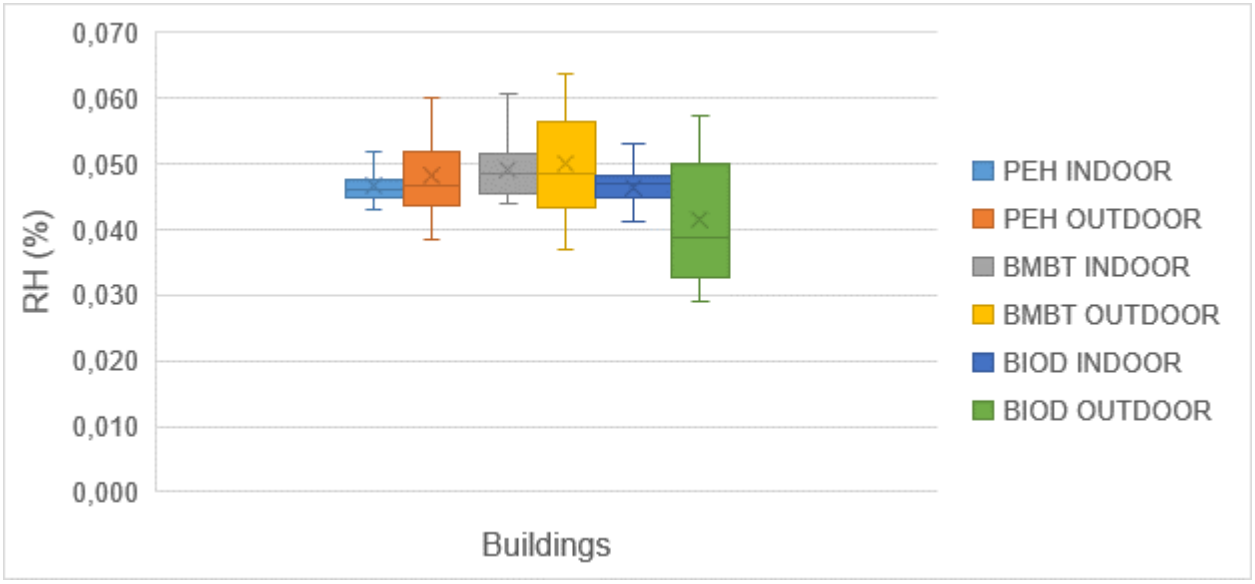

Figure 4.3: Mean indoor and outdoor relative humidity levels across the three buildings studied.

The norm is for outdoor RH readings to be higher than indoor RH readings and this is consistent with the findings of this study for the PEH and BMBT buildings whereas, in the BIOD building, outdoor RH values were lower than indoor RH values. Relative humidity has an inverse relationship with temperature because warm air tends to hold more moisture than cold air. When temperature increases, RH falls if no moisture is added (Dec *et al*., 2018; Money, 1988). The inverse relationship between temperature and RH is also quite evident in this study where the R^2 and p-values computed between RH and T_{db} in both the indoor and outdoor environments depicted significant inverse relationships (Table 4.3). Though there were differences in indoor and outdoor RH values recorded across all buildings, these differences were insignificant (Table 4.4), except for the difference between indoor and outdoor RH in the BIOD building (Table 4.4). Values of I/O ratios of RH were found to be around or just above 1.0. Results obtained for the Spearman's correlation test (Table 4.4) show strong positive significant correlations between RH values in both environments across all three buildings ($p < 0.01$). The correlation between indoor and outdoor RH may be a result of the strong atmospheric mixing between indoor and outdoor spaces through vapour diffusion which causes water vapour to move from regions of high to regions of low concentrations (Tamerius *et al*., 2013).

Table 4.3: Spearman's correlation test comparing relative humidity and dry-bulb temperature values across the indoor and outdoor building environments.

Building comparatives	N	Spearman	
		R^2	p-value
PEH (RH) - PEH (T_{db}) indoor*	20	-0.737	0.000
BMBT (RH) - BMBT (T_{db})	20	-0.875	0.000
BIOD (RH) - BIOD (T_{db})	20	-0.941	0.000
PEH (RH) - PEH (T_{db}) outdoor	20	-0.971	0.000
BMBT (RH) - BMBT (T_{db})	20	-0.983	0.000
BIOD (RH) - BIOD (T_{db}) outdoor	20	-0.961	0.000

* Physiology and Environmental Health indoor relative humidity in comparison to the Physiology and Environmental Health indoor dry-bulb temperature;
** Biochemistry, Microbiology and Biotechnology indoor relative humidity in comparison to the Biochemistry, Microbiology, and Biotechnology indoor dry-bulb temperature;
*** Biodiversity indoor relative humidity in comparison to the Biodiversity indoor dry-bulb temperature.

Table 4.4: Wilcoxon signed ranks and Spearman correlation test comparing indoor and outdoor relative humidity values in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-1.419[a]	0.156	0.735*	0.000
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-0.653[a]	0.514	0.860*	0.000
Biodiversity (BIOD)	20	-2.763[b]	0.006	0.896*	0.000

a. Based on negative ranks;
b. Based on positive ranks;
* Correlation is significant at the 0.01 level (2-tailed).

Occupant densities have also been highlighted as a possible factor affecting indoor RH because of the breathing and perspiration of occupants which contributes to water vapour in the indoor environment. Furthermore, the usage of kettles by staff within their office spaces in addition to water baths utilised during practicals within the indoor environment of these buildings could also contribute to the elevated indoor RH levels observed. Use of fans, heaters and air conditioners have also been highlighted as factors affecting RH levels by Alves *et al*. (2013), Mentese *et al*. (2012), St-Jean *et al*. (2012) and Pegas *et al*.

(2011). In the BIOD building, indoor sources of moisture are possibly the main factors influencing the indoor RH levels detected.

The RH values in the different buildings also varied. Significant differences in indoor RH were observed between PEH and BMBT buildings and between BMBT and BIOD buildings ($p < 0.01$) (Table A2.2). For outdoor RH values, significant differences were observed between PEH and BIOD and between BMBT and BIOD ($p < 0.01$) (Table A2.2). These differences could be attributed to the functions of the various buildings which may be contributing to indoor moisture levels as well as the building structure. The primary function of the BMBT building was to host laboratory practicals during which water baths and numerous solvents and reagents used may have increased indoor RH levels as opposed to PEH which was used mainly for lectures. Furthermore, improper provisioning of natural ventilation, use of kettles (for boiling water) and renovation activities (cement and paint not drying) may all have contributed to higher indoor RH levels observed especially in the BMBT and to a lesser extent across PEH buildings. In addition, most of the indoor environments and the newly renovated offices across the BMBT building and a few across the PEH building had dry walling as dividers between offices.

A study by Dedesko and Siegel (2015) indicated that dry walling is a ubiquitous material that is not intended to get wet but is hygroscopic and very slow to dry out. Elevated levels of RH are commonly seen in office spaces that have dry walling due to the wallboard absorbing and retaining the moisture over time (Pepper *et al*., 2006). This may also explain the high RH values observed in the buildings which had this material. The temperature of the buildings as measured by T_{db} may also have a role to play in the observed RH values. The trend for outdoor T_{db} readings was BIOD > PEH > BMBT whereas the trend for outdoor RH values was BMBT > PEH > BIOD, further highlighting the inverse relationship between T_{db} and RH.

4.4. Indoor and outdoor air quality in the study area

4.4.1. Ozone

There were minimal variations in O_3 concentrations across the three buildings. The mean indoor O_3 concentrations in the three buildings followed the order PEH (0.002 ppm) > BIOD (0.001 ppm) > BMBT (0.000 ppm) (Figure 4.4). Indoor O_3 concentrations in the current study were very low in comparison with those found in literature for school

classrooms (Kalimeri *et al*., 2016; Fadeyi *et al*., 2014; Jovanovic *et al*., 2014; Poupard *et al*., 2005). Outdoor O_3 concentrations were higher than indoor O_3 concentrations across all three buildings in the current study (Figure 4.4). The mean outdoor O_3 concentrations across all three buildings were found to comply with the DEA (2009) standard of 0.061 ppm. Details of the O_3 results can be found in Appendix A1, Tables A1.2 to 1.4. Literature shows outdoor O_3 concentrations ranging between 0.009 and 0.065 ppm (Radaideh, 2017; Jonson *et al*., 2006; Chao, 2001; Lee and Chan, 1998) which are much higher than the range observed in this study.

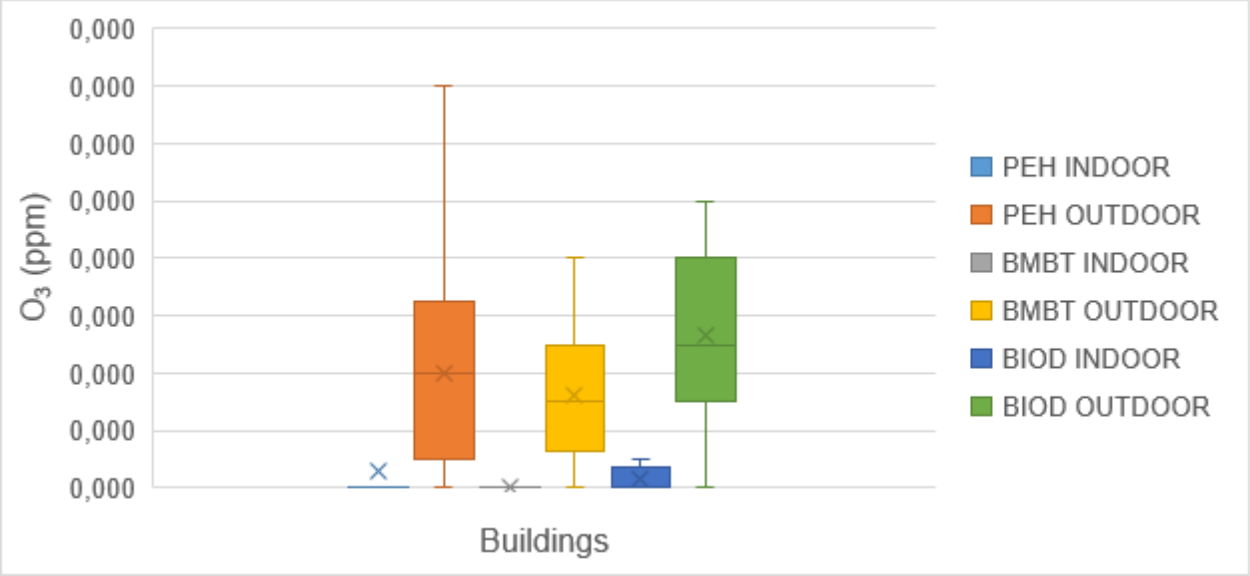

Figure 4.4: Mean indoor and outdoor ozone concentrations across the three buildings studied.

During this study, some sites especially across the indoor BMBT environment rendered zero (0) for O_3 concentrations (Figure 4.4). In the indoor environment, very few O_3 sources exist which may explain the low indoor O_3 values obtained; subsequently, the readings were found to be below the detection limit of the instrumentation. However, printers, photocopiers (Tipayarom and Tipayarom, 2011) and laboratory-based emissions have been incriminated as sources of indoor O_3 and may have contributed to the levels of O_3 detected in the indoor environments of the buildings studied as these are commonly found in university environments. Differences in O_3 concentrations between both environments were statistically significant for all three buildings ($p < 0.01$) (Table 4.5). Monn (2001) similarly found indoor O_3 concentrations to be much lower than outdoor O_3 concentrations.

Ozone occurs naturally in the stratosphere but its presence in the tropospheric layer is an indication of anthropogenic inputs. In the outdoor environment, it is produced from chemical reactions involving organic compounds, CO, NO_x and sunlight (Weschler, 2000). Across the current study area, the most likely outdoor origins of O_3 were motor vehicle exhaust emissions and functional generators. Balajee *et al*. (2016), Britigan *et al*. (2006) and PPAH (1998) all ascertained the primary sources of O_3 to be motor vehicles, power plants, refineries, chemical plants and boilers, fossil fuel combustion areas, photocopying areas, and air purifiers. These all release NO_2, VOCs and CO which are essential in the photochemical production of O_3 as per equations 4.1–4.4 below (Tiwary and Colls, 2010; Vallero, 2008; Pepper *et al*., 2006; de Nevers, 2000).

$N_2 + O_2 \rightarrow 2NO$ (Equation 4.1);

$2NO + O_2 \rightarrow 2NO_2$ (Equation 4.2);

$NO_2 + \hbar\upsilon^* \rightarrow NO + O$ (Equation 4.3);

$O + O_2 + M^{**} \rightarrow O_3 + M^{**}$ (Equation 4.4).

* $\hbar\upsilon$: photon of light energy;

**M: usually being O_2 or N_2.

The I/O ratios of O_3 (Appendix A1, Table 1.4) were all close to zero indicating an outdoor influence rather than an indoor source. A correlation between indoor and outdoor O_3 concentrations is therefore unlikely. The Spearman's correlation test for O_3 concentrations between the indoor and outdoor environments resulted in insignificant correlations between indoor and outdoor O_3 concentrations across all three buildings (Table 4.5).

Table 4.5: Wilcoxon signed ranks and Spearman's correlation tests comparing indoor and outdoor ozone concentrations in the different buildings.

Building	N	Wilcoxon Test		Spearman correlation	
		Z	p-	R^2	p-value
Physiology and Environmental Health (PEH)	20	-2.878[a]	0.004	-0.290	0.215
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.734[a]	0.000	-*	-*
Biodiversity (BIOD)	20	-3.833[a]	0.000	-0.060	0.802

a. Based on negative ranks;

* Due to the strong reactive and deposition properties of ozone, the concentrations at those sampling points may have been zero ppm/ measurements taken may have been below the detection limit of the instruments and as such rendered zero readings.

Indoor O_3 concentrations did not render any differences between buildings ($p = 0.066$), whereas across the outdoor environment, O_3 concentrations in the BIOD building were much higher than those measured across the BMBT building ($p < 0.01$) (Table A2.2). The primary source of O_3 in the BIOD building was probably vehicular emissions due to the proximity of the building to vehicles parked in the immediate surroundings of the building entrance and all around the building as compared to BMBT building which is situated further away from the parking lot.

4.4.2. Sulphur dioxide

Mean SO_2 indoor concentrations in both the PEH and BMBT buildings were the same (0.012 ppm) whereas mean indoor SO_2 concentrations in the BIOD building were 0.003 ppm. Koponen *et al.* (2001) found indoor SO_2 concentrations ranging between 0.0005 and 0.008 ppm within office buildings in Finland, which is lower than the SO_2 concentration ranges detected in the PEH and BMBT buildings but slightly higher than that of the BIOD building. The lowest outdoor SO_2 concentration of 0.010 ppm was found outside the BIOD building, whilst the highest SO_2 concentration of 0.016 ppm was found outside the PEH building (Figure 4.5). Nkosi *et al*. (2017) reported an outdoor mean SO_2 concentration of 0.01–0.02 ppb outside school classrooms in South Africa, which is markedly less than the outdoor SO_2 concentrations in this study. Outdoor SO_2 concentrations were higher than indoor SO_2 concentrations across all three buildings (Figure 4.5). Details of the SO_2 results can be found in Appendix A1, Tables A1.2 to 1.4. Lee and Chan (1998) and Brauer *et al*. (1991) also found outdoor SO_2 concentrations to be higher than indoor SO_2 concentrations in Hong Kong and the USA respectively. The differences between both the environment's SO_2 concentrations in the current study were however insignificant ($p > 0.05$) (Table 4.6).

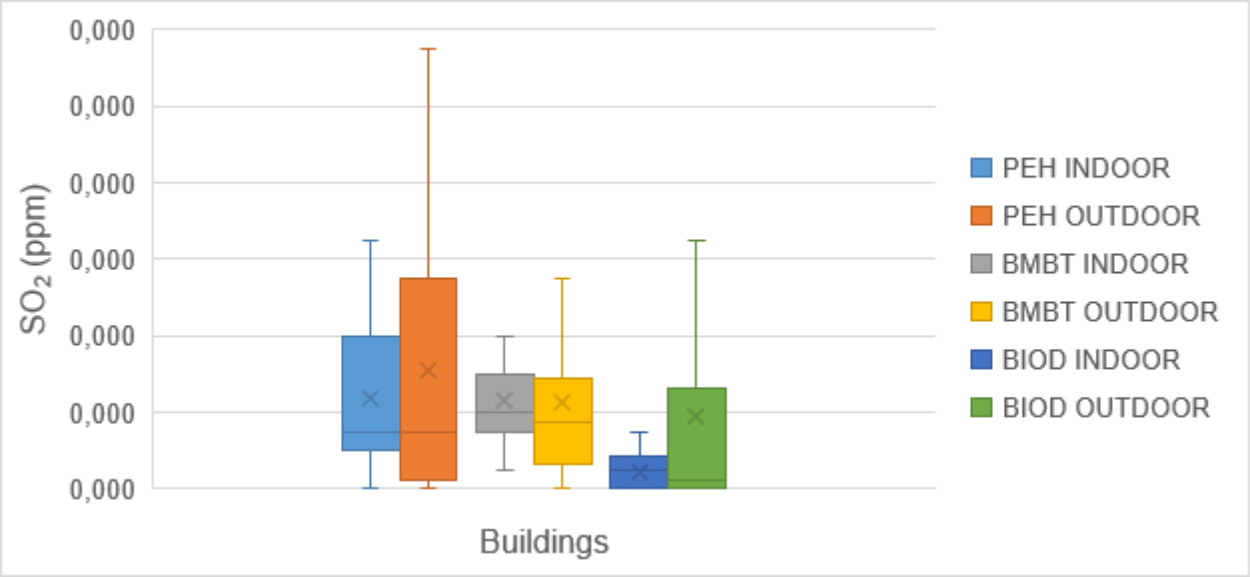

Figure 4.5: Mean indoor and outdoor sulphur dioxide concentrations across the three buildings studied.

Table 4.6: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor sulphur dioxide concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-0.222[a]	0.825	-0.600*	0.005
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-0.343[b]	0.731	-0.401	0.080
Biodiversity (BIOD)	20	-1.863[a]	0.062	0.230	0.330

a. Based on negative ranks;
b. Based on positive ranks;
* Correlation is significant at the 0.01 level (2-tailed).

Sulphur dioxide is one of the gasses formed when sulphur-containing fuels (oil and coal) are burned for electricity generation, heating purposes, metal smelting and other industrial processes (USEPA, 2018a). Sulphur is oxidised to SO_2 during the combustion of sulphur-based fuels. The main indoor and outdoor sources of SO_2 in this study may have been the construction industries and power plants situated approximately two to three kilometres away and situated southeast of the study area. The prevalent wind directions during the sampling period were from the south and south south-east directions (Figure 3.2) (SAWS, 2019a), thus validating the notion of long-range transport as a source of SO_2. Some other sources of SO_2 within the university environment could have been vehicular

emissions around the campus, functional generators, and the surrounding laboratory environments. Most laboratories store and make use of sulphur-based compounds such as sulphuric acid which are used in fume hoods during practicals. This compound releases pungent fumes into the atmosphere which may contribute towards the presence of SO_2 in the indoor environment of the buildings in a university environment.

Literature indicates that outdoor SO_2 concentrations generally have a positive correlation with RH and temperature and poor correlations with LAV values (Goverdhan *et al*., 2015; Jayamurugan *et al*., 2013; Tripathi *et al*., 2013; Akpinar *et al*., 2009). The norm is for SO_2 concentrations to increase with corresponding increases in temperature (Akpinar *et al*., 2009; Beard, 2013; Davis and Masten, 2004). Rogalski *et al*. (2014) found an inversely proportional relationship between outdoor SO_2 and RH. The inverse relationship between RH and SO_2 concentrations is due to SO_2 concentrations changing under different RH ranges and temperatures. Kollikho (1995) found that at 5 °C, SO_2 concentrations start to increase slightly at an RH range of 40–80 %; whereas, at 30 °C, SO_2 concentrations increases even more markedly at an RH value of greater than 90 %. On the other hand, higher LAV readings are indicative of increased wind speeds which may cause diffusion and dispersion of SO_2. In this study as well, lower SO_2 concentrations were found in the BIOD building which was the building characterised by the highest outdoor LAV values.

The results attained from Spearman's correlation test (Table 4.6) found a negative significant correlation between indoor and outdoor SO_2 concentrations in the PEH building (R^2 = -0.600; $p < 0.01$) whereas the correlations for the BMBT and BIOD buildings were insignificant (Table 4.6). For indoor SO_2 concentrations, significant differences were observed between the PEH and BIOD buildings as well as between the BMBT and BIOD buildings ($p < 0.01$). Building function and location seem to play a role in the concentration of SO_2 in the indoor and outdoor environments in this study. The PEH building is surrounded by a car park from where SO_2 could originate. Sulphur dioxide in the outdoor environment is likely to have diffused into the indoor environment through building entrances, open doors and windows across a concentration gradient, i.e. from elevated concentrations to minimal concentration areas (Lan *et al*., 2004). The diffusion of SO_2 occurs during periods of intense solar radiation, elevated temperatures and low RH levels (Beard, 2013; Davis and Masten, 2004), whereby SO_2 moves from the higher concentrations outdoors to the lower concentrations indoors. Within the PEH building, the

principal source of indoor SO_2 concentrations could have been outdoor air infiltration into the indoor environment. The BIOD building did not have any justifiable source of indoor SO_2, hence, the low indoor SO_2 concentrations. Differences in SO_2 concentrations between the BIOD and PEH buildings and between the BIOD and BMBT buildings are therefore not unexpected. Insignificant differences were found in outdoor SO_2 concentrations between buildings. All three buildings are likely to have had similar outdoor sources of SO_2.

4.4.3. Nitrogen dioxide

Indoor NO_2 concentrations in the PEH building ranged from 0.026 to 0.307 ppm, with a mean of 0.100 ppm. These values were similar to the indoor NO_2 concentrations at the BMBT building (with a range of 0.005 to 0.449 ppm and a mean of 0.096 ppm) but higher than what was obtained for the BIOD building (a range of 0.013 to 0.202 ppm, with a mean of 0.082 ppm). These are not unusual as numerous studies have established similar ranges in school classrooms, offices, restaurants, and homes (Villanueva *et al*., 2018; Kalimeri *et al*., 2016; Lu *et al*., 2016; Ni *et al*., 2016). Kornartit *et al*. (2010), Kousa *et al*. (2001) and Lee *et al*. (2000) have all showed that gas appliances are a major source of NO_2 emissions. These appliances including Bunsen burners commonly used in the laboratory make use of liquefied petroleum gas or gas which produces NO_2 during combustion. In addition, some laboratory equipments make use of nitrous oxide gas which could contribute to the emissions of NO_2 in the laboratory environment. This would have contributed to the elevated indoor NO_2 concentrations encountered across PEH and BMBT buildings. The lowest outdoor NO_2 concentration of 0.140 ppm was found outside the BMBT building, whilst the highest NO_2 concentration of 0.152 ppm was found outside the BIOD building (Figure 4.6). Details of the NO_2 results can be found in Appendix A1, Tables A1.2 to 1.4. The outdoor NO_2 mean concentrations in the current study are much lower than the maximum value established for outdoor NO_2 concentrations in literature (Villanueva *et al*., 2018; Wheida *et al*., 2018; Radaideh, 2017; Meier *et al*., 2015). Natural outdoor NO_2 concentrations normally range from 0.01–0.05 ppm (WHO, 2000b) and are usually formed by the reaction of NO and O_2 or O_3 in the atmosphere. Alves *et al*. (2013) and PPAH (1998) both ascertained heat and electricity generating areas, laboratories harbouring nitrogenous chemicals and nitric acid, arc-welding workshops, and biogenic emissions as anthropogenic sources of NO_2, with a small proportion emanating from motor vehicle and generator exhaust emissions.

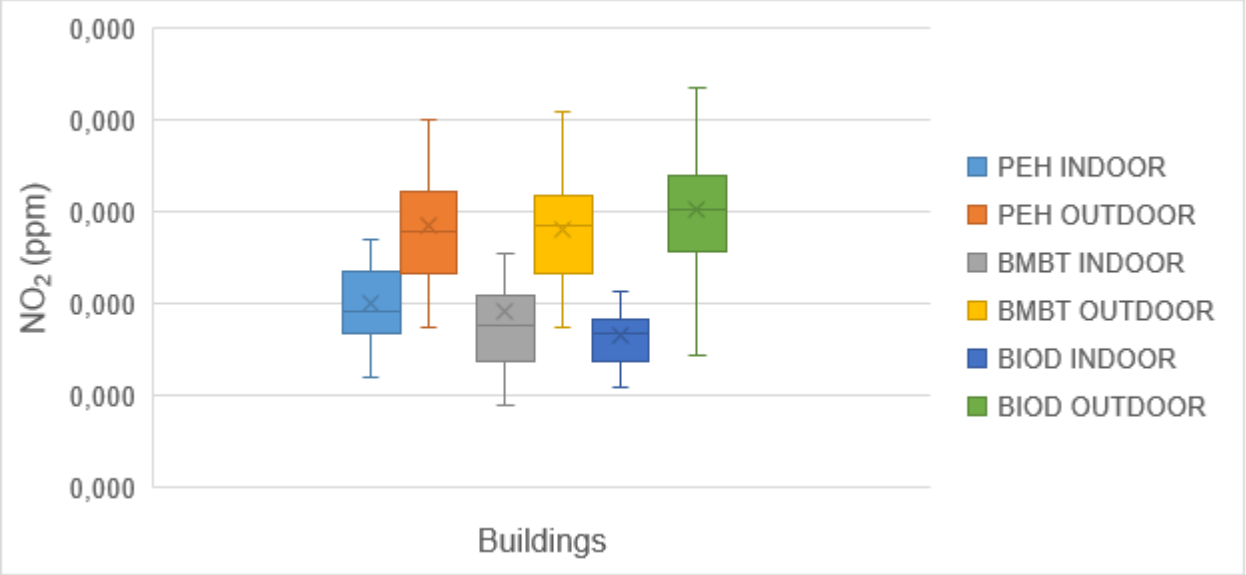

Figure 4.6: Mean indoor and outdoor nitrogen dioxide concentrations across the three buildings studied.

The trend of outdoor NO_2 concentrations being consistently higher than indoor NO_2 concentrations (Figure 4.6) are aligned with the findings of Kornartit *et al*. (2010), Kalimeri *et al*. (2016), Meier *et al*, (2015) and Rivas *et al*. (2014). The differences between indoor and outdoor NO_2 concentrations across all three buildings in this study were significant, as reflected by the p-values in Table 4.7 ($p < 0.01$).

Table 4.7: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor nitrogen dioxide concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-2.951[a]	0.003	-0.309	0.184
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.139[a]	0.002	0.273	0.245
Biodiversity (BIOD)	20	-3.735[a]	0.000	0.353	0.127

a. Based on negative ranks.

In this study, the PEH and BMBT buildings had functional Bunsen burners which made use of gas in the indoor environment during practicals, which could explain the higher concentrations of NO_2 in the indoor environment of these buildings compared to the BIOD

building. Outdoor air infiltration as always is a factor that cannot be discounted in the level of indoor NO_2 concentrations. Kalimeri *et al*. (2016); Poupard *et al*. (2005) and Chao (2001) all found that very little building protection could result in outdoor NO_2 infiltrating into the indoor environments. Infiltration may also have contributed to the indoor NO_2 concentrations. The results from the Spearman correlation of indoor and outdoor NO_2 concentrations depicted insignificant correlations across all three buildings ($p > 0.05$) (Table 4.7). Chan (2002) also found insignificant correlations between indoor and outdoor environments. Contrastingly, Kalimeri *et al*. (2016), Wichmann *et al*. (2010) and El-Hougeiri and El Fadel, (2004) all found strong positive correlations between indoor and outdoor NO_2 concentrations. The observations in this study are therefore not new.

Pegas *et al*. (2011), Hazenkamp-von Arx *et al*. (2004), Chan (2002) and Rijnders *et al*. (2001) all established traffic emissions as a good indicator and marker of outdoor NO_2 pollution. The PEH building had an arc-welding workshop situated adjacent to the car parking area at the north side of the building, which may have contributed to the high outdoor NO_2 concentrations observed around this building. PPAH (1998) established that NO_2 emissions were by-products of submerged arc-welding processes. Vehicular emissions have been incriminated as sources of NO_2. Large car parking areas present on the north side of the PEH and BMBT buildings, and cars parked all around the BIOD building may also have contributed to outdoor NO_2 concentrations. Throughout the entire duration of the study, there were several time periods during which load shedding was implemented, which led to the use of diesel-powered generators to supply power to the various buildings. These functional generators may also have contributed to the outdoor NO_2 concentrations in the study area. The NO_2 concentrations did not vary across buildings, as evident in Figure 4.6 above. Nitrogen dioxide concentrations seemed to have been more dependent on outdoor sources. There were no significant differences in NO_2 concentrations in the indoor ($p = 0.271$) and outdoor ($p = 0.087$) environments across the three buildings.

4.4.4. Carbon monoxide

Indoor CO concentrations in the PEH building fluctuated between 0.000 to 0.133 ppm, with a mean of 0.029 ppm. Mean indoor CO concentrations of BMBT and BIOD buildings were 0.037 and 0.033 ppm respectively. Values obtained for the current study are much lower than mean indoor CO concentrations reported by Choo *et al*., (2015), Fadeyi *et al*.,

(2014), Alves *et al*., (2013) and Al-Rehaili, (1999). Common indoor sources of CO include cooking, heating and smoking sources (WHO, 2010). High concentrations of CO in the indoor environment have been associated with anthropogenic activities that include the use of gas appliances that emit CO and other gases as by-products (WHO, 2010). In university environments, Bunsen burners used in the laboratory environments could also be a source of CO in the indoor environment.

The lowest outdoor CO concentration of 0.107 ppm was found outside the BMBT building, whilst the highest CO concentration value of 0.130 ppm was found outside the BIOD building (Figure 4.7), which are lower than values reported in literature (Radaideh, 2017; Kalimeri *et al*., 2016; Fazlzadeh *et al*., 2015; El-Hougeiri and El Fadel, 2004). Under natural and unpolluted atmospheric conditions, the mean CO concentrations are around 0.20 ppm (DEH, 2005). Details of the CO results can be found in Appendix A1, Tables A1.2 to 1.4.

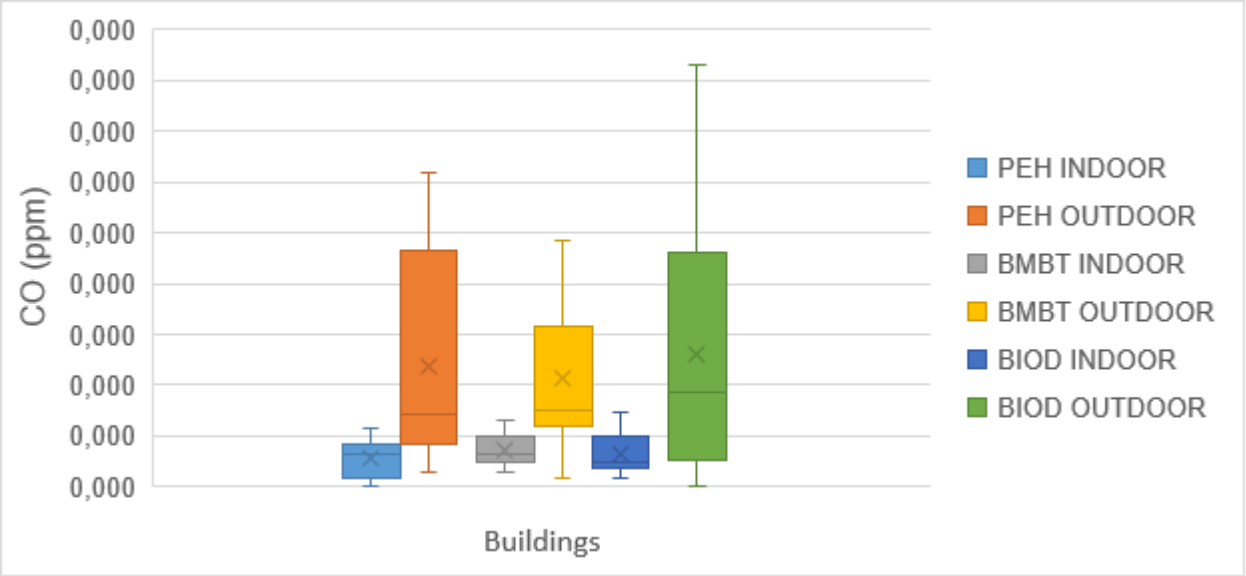

Figure 4.7: Mean indoor and outdoor carbon monoxide concentrations across the three buildings studied.

Carbon monoxide is a product of incomplete combustion of carbon-containing material with vehicular traffic emissions, domestic fuel burning (gas appliances and coal usage), industrial sources and environmental tobacco smoke being the main environmental sources (Choo *et al*., 2015; Hill, 2007; Jo and Lee, 2006; de Bruin *et al*., 2004). In this study, elevated outdoor concentrations of CO were encountered in areas close to car parks and generator stations which could be the source of CO. Fadeyi *et al*. (2014) found

vehicular traffic to be the major contributing factor to CO concentrations within school environments. The USEPA (2018b) ascertained 95 % of all CO emissions to be emanating from motor vehicular exhausts. Some additional outdoor sources of CO emissions may have been the power plants on the outskirts of the university that burn fossil fuels. This might have produced by-products including CO, CO_2, and SO_2, that may have been transported to the study area by means of long-range transport mechanisms. This explanation is justified by the fact that the building was located south/south-east of the power plant and the prevalent wind direction as per the SAWS (2019a) data was south and south south-east as well. Another source of CO in the study area may have been CO originating naturally within the soil and vegetation environments. Potter *et al*. (1996) established that soils could release between 16.1 to 50.6 kg of CO per year. Carbon monoxide can be produced from soils during the oxidation of CH_4 as depicted below in equations 4.5 and 4.6 (Davis and Masten, 2004). Methane usually exists in the soil as a by-product of decomposing organic matter and wastes.

$CH_4 + OH^{\cdot} \rightarrow CH_3. + H_2O$ (the $OH^{\cdot}$ radical serves as the initial oxidising agent, combining with CH_4 to form an alkyl radical CH_3.) (Equation 4.5);

$CH_3 + O_2 + 2\hbar\upsilon^* \rightarrow CO + H_2 + OH$ (the alkyl radical and the O_2 are both exposed to photons of light energy ($2\hbar\upsilon^*$)) (Equation 4.6)

* $\hbar\upsilon$: photon of light energy.

The overall trend was higher outdoor CO concentrations compared to indoor CO concentrations across all three buildings (Figure 4.7), consistent with the finding of Chaloulakou *et al*. (2003), Chaloulakou and Mavroidis (2002). The Wilcoxon signed ranks test revealed significant differences between indoor and outdoor CO concentrations ($p < 0.01$) (Table 4.8).

Table 4.8: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor carbon monoxide concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-3.543[a]	0.000	0.174	0.462
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.249[a]	0.001	-0.302	0.196
Biodiversity (BIOD)	20	-2.979[a]	0.003	-0.253	0.281

a. Based on negative ranks.

Insignificant correlations were observed between indoor and outdoor CO concentrations across all three buildings ($p > 0.05$) (Table 4.8). The lack of correlation found in this study is indicative of different indoor and outdoor CO sources. The norm is for gases to diffuse along a concentration gradient, but in this study, CO seemed not to have infiltrated to the indoor environment, though the concentration was higher outdoors. Furthermore, diminished indoor LAV values may lead to the build-up of CO, provided an inherent indoor source of CO is present. Differences in CO concentrations among the three buildings, both in the indoor and outdoor environments were insignificant ($p = 0.354$ and 0.951 respectively). Building type and building function both did not seem to have any effect on CO concentrations across the three buildings.

4.4.5. Carbon dioxide

Indoor CO_2 concentrations across the three buildings followed the order PEH > BMBT > BIOD (Figure 4.8). Indoor CO_2 concentrations in the PEH building ranged from 481 to 1022 ppm, with a mean of 625.6 ppm. In comparison, the mean indoor CO_2 concentrations in BMBT and BIOD buildings were 581.2 and 530.6 ppm respectively. Details of the CO_2 results can be found in Appendix A1, Tables A1.2 to 1.4. Several studies (Budiakova, 2017; Peng *et al*., 2017; Choo *et al*., 2015; Fadeyi *et al*., 2014; Gao *et al*., 2014; Jurado *et al*., 2014) have reported a CO_2 concentration range of 408 to 2739 ppm within indoor environments. However, the upper limit of the range (i.e. 2739 ppm) was much higher than the maximum value (i.e. 1022 ppm) observed in the current study. The main sources of CO_2 in the outdoor setting are vehicle exhaust emissions, fossil fuel-burning activities and industrial emissions (Widder and Haselbach, 2017). In the indoor environment, the

contributors to CO_2 concentrations are mainly human respiration and the burning of different types of fossil fuels (Widder and Haselbach, 2017; Knížatová *et al*., 2010; Koerner and Klopatek, 2002) for cooking purposes in kitchens. The higher indoor CO_2 concentrations in comparison to the outdoor CO_2 concentrations found in this study can be explained by the high occupant densities within the buildings which might have contributed to the high CO_2 concentrations through breathing. Budiakova (2017), OSHA (2011) and Prill (2000) also found a positive correlation between CO_2 concentrations and occupant density within a given environment.

Indoor CO_2 concentrations were higher than outdoor CO_2 concentrations across all three buildings (Figure 4.8). The outdoor CO_2 concentrations ranged from 338 to 507 ppm. A possible source of outdoor CO_2 concentrations in the study area may be CO_2 emanating from soil environments and municipal waste bins, which were near the outdoor sampling points. Carbon dioxide is released from the soil, through microbial, root and faunal respiration at the soil surface or the upper layer (Rastogi *et al*., 2002). The differences recorded between indoor and outdoor CO_2 concentrations in each of the three buildings studied were significant ($p < 0.05$) (Table 4.9). The plants and trees in the outdoor environment and their role in absorbing CO_2 for purposes of photosynthesis could also play a role in explaining the lower outdoor CO_2 concentrations across all three buildings (Suwanmontri *et al*., 2013).

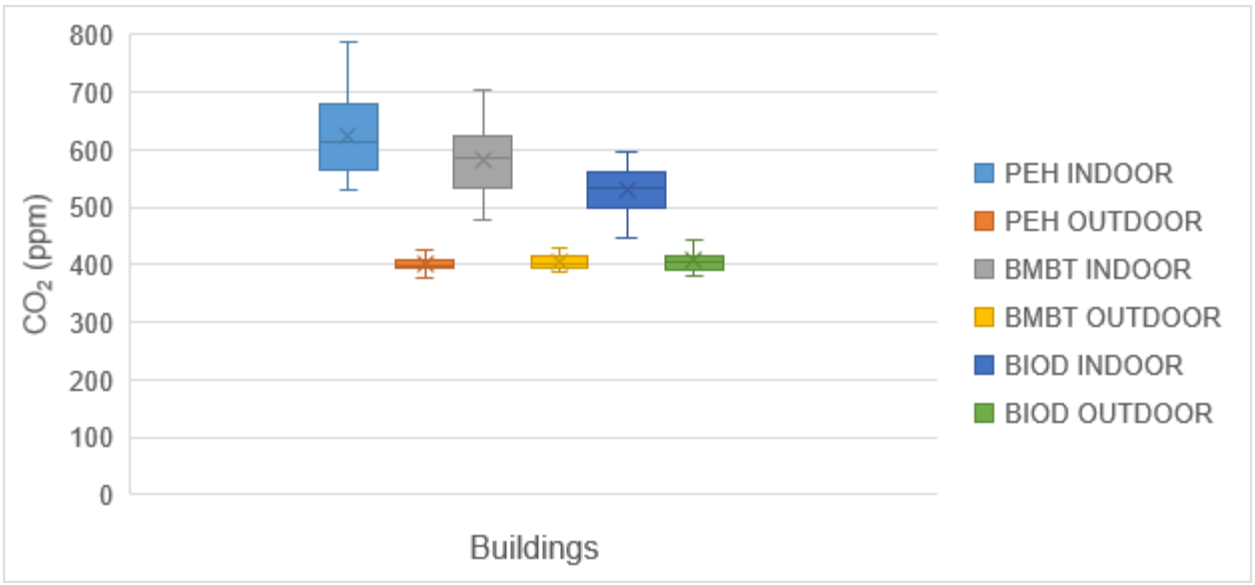

Figure 4.8: Mean indoor and outdoor carbon dioxide concentrations across the three buildings studied.

Table 4.9: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor carbon dioxide concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-3.920[a]	0.000	-0.113	0.636
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.920[a]	0.000	0.108	0.649
Biodiversity (BIOD)	20	-3.920[a]	0.000	-0.131	0.582

a. Based on positive ranks.

Insignificant correlations between indoor and outdoor CO_2 concentrations across all three buildings were found as indicated by results from Spearman's correlation test in Table 4.9. This is also reflected in the I/O ratios for CO_2 which were greater than 1.2. This I/O ratio suggests the extensive contribution of the indoor environment to the overall CO_2 concentration in the area. The insignificant correlations found are contrary to the findings of Baek *et al*. (1997), who had established correlations between indoor and outdoor CO_2 concentrations across office environments ($p < 0.01$).

Carbon dioxide concentrations varied within buildings. Indoor CO_2 concentrations in the PEH building were found to be significantly different from the BIOD building ($p < 0.01$). The factors that could have contributed to the differences observed among buildings could be the structure and design of the building, occupant densities, ventilation systems, and the main function of the buildings themselves. The space allocation of the PEH building was such that it had a higher density of occupants because it was primarily used for lectures, computer-based practicals and for staff offices. The BMBT and BIOD buildings, in contrast, had just practical laboratories, tutorial/seminar rooms and staff offices distributed across the buildings, thus reducing the occupant density relative to that of the PEH building. This may account for the differences observed in the CO_2 concentrations across the different buildings. All three buildings had HVAC systems present. Natural ventilation provisioning varied across the three buildings, due to the innate behaviour and the perception of thermal comfort by occupants themselves. Outdoor CO_2 concentrations of the PEH building were found to be significantly different from the BMBT building ($p =$

0.015). The larger number of students entering and exiting the PEH building in comparison to the BMBT building might have been responsible for these differences.

4.4.6. Hydrogen sulphide

The lowest indoor H_2S concentrations were found in the PEH building followed by the BMBT building and finally the BIOD building. Indoor H_2S concentrations in PEH ranged from 0.000 to 0.093 ppm, with a mean of 0.026 ppm. In comparison, the mean indoor H_2S concentrations in BMBT and BIOD were 0.020 and 0.017 ppm respectively. Pertaining to the outdoor environment, the lowest outdoor H_2S concentration of 0.019 ppm was found outside the BIOD building, whilst the highest outdoor H_2S concentration of 0.024 ppm was found outside the BMBT building (Figure 4.9). Details of the H_2S results can be found in Appendix A1, Tables A1.2 to 1.4. Outdoor H_2S concentrations were higher than indoor H_2S concentrations in the BMBT and BIOD buildings but not in the PEH building (Figure 4.9). Very few studies have looked at H_2S concentrations across indoor and outdoor environments. Hence, indoor and outdoor H_2S concentration mean values and ranges are not common in literature. Neither any significant differences nor significant correlations were found between indoor and outdoor H_2S concentrations across all three buildings ($p > 0.05$) (Table 4.10). The indoor H_2S sources could be laboratories, whereas outdoor H_2S sources could be municipal waste bins.

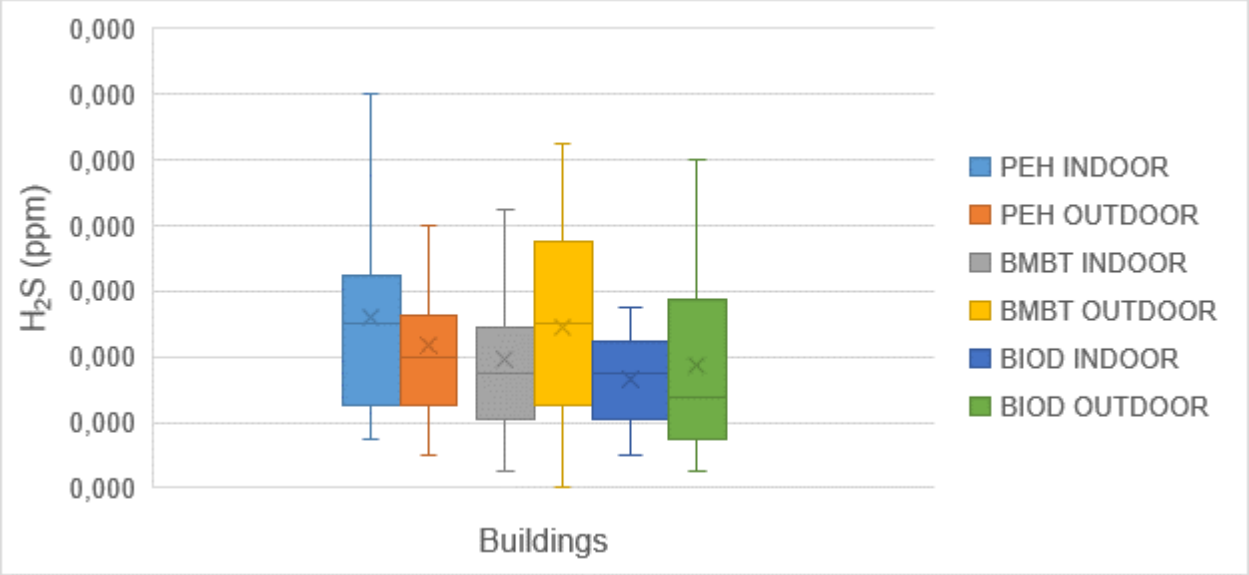

Figure 4.9: Mean indoor and outdoor hydrogen sulphide concentrations across the three buildings studied.

Table 4.10: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor hydrogen sulphide concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-0.850[b]	0.395	-0.187	0.431
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-0.806[a]	0.420	0.205	0.386
Biodiversity (BIOD)	20	-0.141[a]	0.888	0.134	0.574

a. Based on negative ranks;
b. Based on positive ranks.

Within the indoor setting, the predominant source of H_2S must have been the laboratory environments and infiltration of H_2S from the outdoor environment. Hydrogen sulphide moves by means of diffusion from the outdoor environment to the indoor environment. Generally, H_2S is a highly toxic and corrosive gas identified easily by its rotten-egg odour (Botkin and Keller, 2011) and it is released naturally as a product of the decomposition of dead animal and plant material among other sources (Kourtidis *et al*., 2008; Skrtic, 2006). Hydrogen sulphide is also produced from the action of bacteria on organic material containing proteins and reflected in equation 4.7 below (Ko *et al*., 2015):

$$SO_4^{2-} + 2C + 2H_2O \xrightarrow{SRB^*} H_2S + 2HCO_3^- \quad \text{(Equation 4.7).}$$

* SRB: Sulphate-reducing bacteria

Under anaerobic conditions, waste-containing organic sulphur compounds can be decomposed through a desulphurisation process that generates H_2S in a gaseous form (Ko *et al*., 2015) (equations 4.8–4.10), subsequently contributing to the outdoor concentrations of H_2S (Kourtidis *et al*., 2008; Skrtic, 2006, Ko *et al*., 2015).

$$SO_4^{2-} + ATP \xrightarrow{APS\,sulphurlase} APS + PP \quad \text{(Equation 4.8);}$$

$$SO_4^{2-} + ATP + H_2 \rightarrow HSO_3^- + AMP + H^+ \quad \text{(Equation 4.9);}$$

$$HSO_3^- + 3H_2 \rightarrow HS^- + 3H_2O \quad \text{(Equation 4.10).}$$

Hence, outdoor sources of H_2S across the university environment were most likely the municipal garbage bins in addition to the laboratory environments.

Indoor H_2S concentrations in the PEH and BIOD buildings were found to be significantly different from one another ($p = 0.010$). All three buildings had dry-walling commonly used between offices, especially across the PEH and BMBT buildings. These structures have been known to increase indoor H_2S concentrations as they release sulphate ions and other organic compounds that sulphate-reducing bacteria use to generate H_2S (Yang *et al*., 2006; ATSDR, 2012). Outdoor H_2S concentrations did not show any differences between buildings ($p = 0.154$).

4.4.7. Non-methane hydrocarbons

Non-methane hydrocarbons are a group of natural and anthropogenic aliphatic, aromatic and alkyl moieties containing only hydrogen and carbon atoms (ranging from C_2–C_{12}), coupled with a low molecular weight (Kumar *et al*., 2017). They form a major group of VOCs that are highly reactive due to their strong tendency of getting oxidised by the OH^- radical and reacting with NO_x, eventually forming O_3 (Kumar *et al*., 2017; Tan *et al*., 2012; Tang *et al*., 2009). Compounds commonly grouped as NMHCs include alkanes, alkenes, ethyne, aromatics, aldehydes, ketones halocarbons and other organic compounds (Kumar *et al*., 2017). Mean indoor concentrations of NMHC across the PEH, BMBT and BIOD buildings were 0.211, 0.664 and 0.521 ppm respectively which indicate that the PEH building had the highest indoor NMHC concentrations. Outdoor NMHC concentrations were lower than the indoor values with the lowest outdoor NMHC concentration of 0.188 ppm recorded outside the BMBT building, and the highest outdoor NMHC concentration of 0.260 ppm recorded outside the BIOD building (Figure 4.10). There are both natural and anthropogenic sources of NMHCs in both environments. Natural sources include biogenic emissions, whereas some anthropogenic sources are biomass burning, vehicular exhaust emissions, industrial emissions (gasoline evaporation, liquefied petroleum gas usage, burning of biofuel and coal), evaporation of fossil fuels, organic solvents (consumer products, paints) and other stationary sources (chemical solvent-use businesses) (Kumar *et al*., 2017; Jaimes-Palomera *et al*., 2016; Ou *et al*., 2015; Li and Wang, 2012).

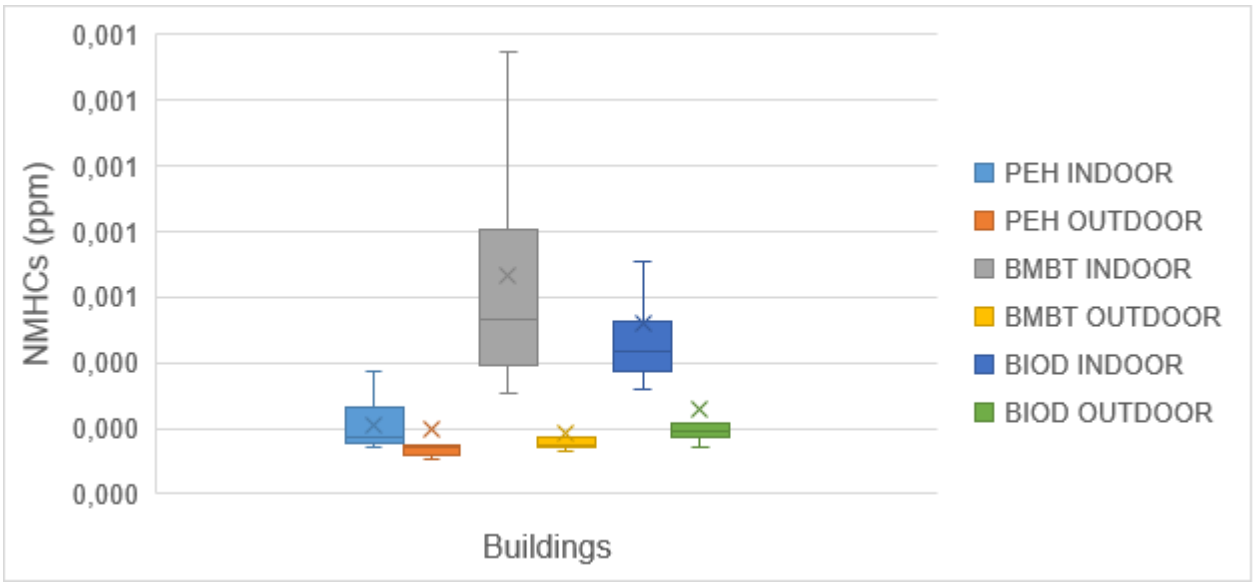

Figure 4.10: Mean indoor and outdoor non-methane hydrocarbon concentrations across the three buildings studied.

Elevated indoor NMHC concentrations observed in this study could be as a result of occupants housing solvents, and reagents, such as ethanol (C_2H_6O), methanol (CH_4O), formaldehyde (CH_2O) and acetone (C_3H_6O) within their office spaces especially in the BMBT and BIOD buildings as opposed to storing them in a laboratory environment. Some of these reagents are highly volatile and can influence indoor AQ in areas where they are stored. Though storing reagents and solvents in the office is not an approved practice, practical instructors tend to store reagents in their offices to prevent others from using them without permission. Other indoor sources of NMHCs in the study area include consumer products (cleaning products, nail polish), paints and printing. According to Kumar *et al*. (2017) and Ou *et al*. (2015), these products contain NMHCs such as ethane, hexane, heptane, ethylene, acetylene, propane, butane, propylene, dibutyl phthalate, butyl and/or ethyl acetate, benzene, toluene, ethylbenzene and CH_2O.

Vehicular emissions and industrial emissions contain certain NMHCs such as ethane, ethene, ethyne, propene, *i*-pentane, *i*-butane, *i*-butene, ethylbenzene, (*m*+*p*+*o*)-xylene, toluene, benzene and C_3–C_5 alkanes (Ou *et al*., 2015; Arsene *et al*., 2009; Chan *et al*., 2006). Other NMHCs, namely: isoprenes, monoterpenes, and sesquiterpenes are common biogenic emissions from vegetation including trees and may contribute to outdoor NMHC concentrations (Kumar *et al*., 2017; Ou *et al*., 2015; So and Wang, 2004). Another source of minimal NMHC concentrations across the outdoor environment were

microbial emissions in the soil and from decomposing waste (Kumar *et al*., 2017; Ou *et al*., 2015; Kansal, 2009) found within the municipal waste bins.

The Wilcoxon test indicated that the difference in indoor and outdoor NMHC concentrations between the BMBT and BIOD buildings were significant, whereas that of the PEH building was insignificant (Table 4.11). Details of the concentrations of NMHC can be found in Appendix A1, Tables A1.2 to 1.4. Insignificant correlations in NMHC concentrations in the indoor and outdoor environments were found across the PEH and BIOD buildings, whereas in the BMBT building, indoor and outdoor NMHC concentrations had significant correlations.

Table 4.11: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor non-methane hydrocarbon concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-1.867[a]	0.062	0.379	0.100
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.883[a]	0.000	-0.566*	0.009
Biodiversity (BIOD)	20	-3.304[a]	0.001	-0.244	0.299

a. Based on positive ranks;
* Correlation is significant at the 0.01 level (2-tailed).

Correlations between indoor and outdoor NHMC concentrations are not expected, given the nature of NMHC sources in the two environments (Table 4.11). The I/O ratio across PEH building rendered values around one (Appendix A1, Table A1.4) which indicates a shift towards equilibration between indoor and outdoor NMHC levels.

Indoor NMHC concentrations were significantly different between the PEH and BMBT buildings and between the PEH and BIOD buildings ($p < 0.01$) (Table A2.2). The function of the BMBT and BIOD buildings which was mainly laboratory use for practical-related purposes would have contributed to the higher indoor NMHC concentrations measured in comparison to those of the PEH building. Some of the most common organic solvents used during the practicals held in this building include acetone, alkanes, benzene, butane, butene, ethane, ethene, ethanol, ethylbenzene, formaldehyde, methanol, phenol,

propane, propene, propanol, toluene, and xylene. The higher concentrations of NMHC observed in the BMBT and BIOD buildings are therefore not unexpected. Outdoor NMHC concentrations were also significantly different between the PEH and BIOD buildings and between the BMBT and BIOD buildings ($p < 0.01$).

4.4.8. Volatile organic compounds

Volatile organic compounds are compounds that easily become vapours or gases (Toxtown, 2017; Cunningham and Saigo, 2001). Similar mean values of 0.041 ppm and 0.035 ppm were found for indoor VOC concentrations in the PEH and BIOD buildings respectively. In stark contrast, the indoor VOC concentrations in the BMBT building were exceptionally higher with a mean of 0.427 ppm. Alves *et al*. (2013) had found indoor total VOC concentrations reaching levels of 2.3 ppm, which is much higher than the values encountered across the three buildings in the current study. Some indoor VOC sources in the current study were floors, ceilings, walls, recently renovated environments (stripping, painting and construction which release formaldehyde which is a primary VOC), consumer products such as nail polish and remover, perfumes and deodorants (both used by staff members and other personnel), detergents, floor wax and polish, solvents (adhesives, welding, inks, chlorinated tap water), other building materials (plastics, coatings, foam insulators, varnish, paint remover, plywood, phenolic resins, furniture polish), moth repellents, cigarette smoke and gasoline-related VOCs (burning of fossil fuels in kitchen environments and cooking emissions). In addition to the above-mentioned sources above, possible sources of indoor VOCs in the current study area were the chemicals and reagents stored within office spaces across the BMBT and BIOD buildings. Generally, VOC concentrations are found to increase proportionally with increases in temperature and RH (Shrubsole *et al*., 2019; Markowicz and Larsson, 2015) but are inversely proportional to LAV increases (Rastan *et al*., 2005). Temperature increases tend to cause rapid evaporation and diffusion of VOCs in a given environment (Lin *et al*., 2009). Floors, ceilings, walls and other building materials tend to be sources of VOCs in indoor environments. Their moisture content changes with changes in RH. The higher moisture content of these building materials tends to make VOCs diffuse (Huang *et al*., 2016; Rösch *et al*., 2014; Lin *et al*., 2009). The process of VOC diffusion can be explained as follows: as the water from these materials evaporates, heat is absorbed and that increase in heat, in turn, increases the diffusion of VOCs in an environment already having known sources of VOCs (Huang *et al*., 2016; Rösch *et al*., 2014; Lin *et al*., 2009). Results from other

studies indicate that high outdoor LAV tends to dilute the VOC concentrations through dispersion mechanisms. This could be the case in the current study.

The lowest outdoor VOC concentration of 0.015 ppm was found outside the BIOD building, whilst the highest VOC concentration of 0.029 ppm was found outside the PEH building (Figure 4.11). The maximum value obtained at the PEH building can be attributed to some renovation activities that were on-going during the entire sampling period. Paints usually comprise of alcohols, esters, texanol, cellosolve, and glycols, which are all primary VOCs (Chang *et al*., 2011).

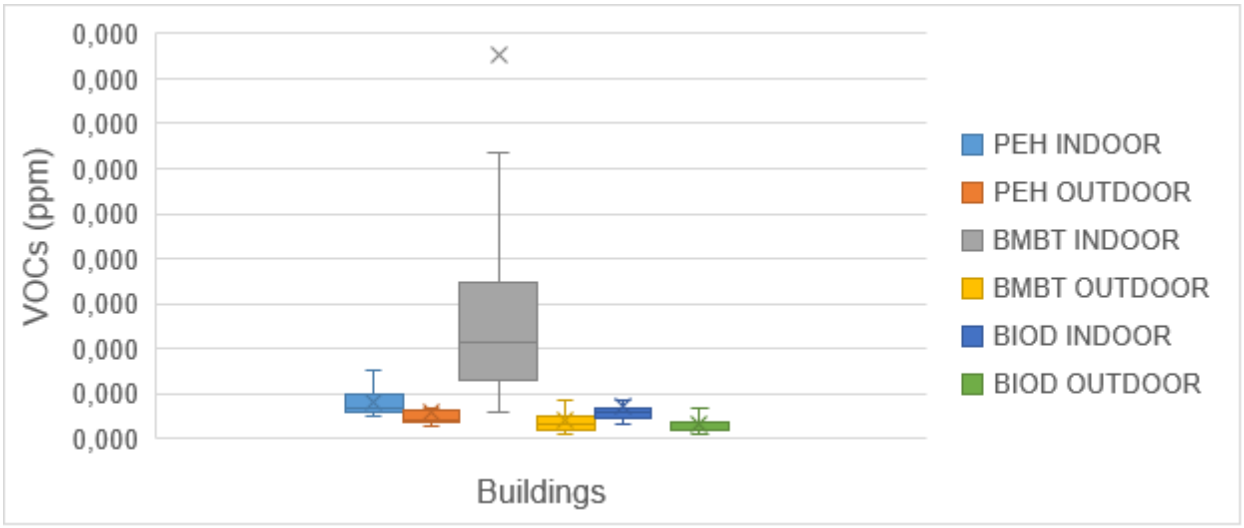

Figure 4.11: Mean indoor and outdoor volatile organic compound concentrations across the three buildings studied.

In the current study, the major sources of VOCs across the outdoor environment were possibly vehicular and generator exhaust emissions (benzene, toluene, n-decane/hexane/heptane/nonane/octane, o-xylene and 1,2,4-trimethylbenzene) and emissions from laboratory exhaust fans nearby (benzene, cyclohexane, and styrene). Additional VOC sources in this study could be the presence of municipal garbage skips that have decomposing wastes, and biogenic emissions (styrene and terpenes) (Pekey and Arslanbaș, 2008). The most common VOCs in some of the above-mentioned sources are benzene, carbon tetrachloride, ethylbenzene, tetrachloroethylene, toluene, xylene, methylene chloride, styrene, trichloroethylene, *p*-Dichlorobenzene, *d*-Limonene, α-Pinene, *m*/*p*-Xylene, and *o*-Xylene as determined by Jia *et al*. (2008a), Jia *et al*. (2008b) and Adgate *et al*. (2004).

The overall trends were for indoor VOCs to be higher than outdoor VOC concentrations across all three buildings (Figure 4.11), which is comparable to the findings of Jia *et al*. (2008a), Kim *et al*. (2001) and Fischer *et al*. (2000). Details of the VOC results for the study can be found in Appendix A1, Tables A1.2 to 1.4. Differences in VOC concentrations observed between the indoor and outdoor environments in each of the three buildings were significant (Table 4.12), whilst a moderate positive correlation in VOC concentrations were found only between the indoor and outdoor environments in PEH ($R^2 = 0.577$; $p < 0.01$).

Table 4.12: Wilcoxon signed ranks and Spearman's correlation test comparing indoor and outdoor volatile organic compound concentrations in the different buildings.

Buildings	N	Wilcoxon Test		Spearman correlation	
		Z	p-value	R^2	p-value
Physiology and Environmental Health (PEH)	20	-2.501[a]	0.012	0.577*	0.008
Biochemistry, Microbiology, and Biotechnology (BMBT)	20	-3.920[a]	0.000	-0.309	0.185
Biodiversity (BIOD)	20	-3.083[a]	0.002	-0.135	0.571

a. Based on positive ranks;
* Correlation is significant at the 0.01 level (2-tailed).

Within BMBT and BIOD buildings, the laboratory environments and the office spaces themselves had high VOC concentrations due to the solvents housed there, whereas in the PEH building, solvents of such nature were not found. Indoor VOC concentrations had statistically significant differences between the PEH and BMBT building and between the BMBT and BIOD buildings ($p < 0.01$) (Table A2.2). The major factor affecting the indoor VOC concentrations in the buildings was the storage of chemicals. Occupants within the BMBT building tended to store organic solvents, chemicals and reagents within their office spaces as opposed to within laboratory storage cabinets thus rendering high VOC concentration readings as illustrated in Figure 4.11. A difference in VOC concentrations between PEH and BMBT building is therefore expected due to the inherent differences in VOC sources in the indoor environment across the two buildings. The PEH building's VOC concentrations peaked during periods when the floors were polished and a portion of the building was painted. In contrast, the BMBT building's VOC concentrations were

consistently high across the entire study period due to large quantities of chemicals, solvents and reagents present within the indoor environments. The overall BIOD building's VOC concentrations were much less than that of the BMBT building due to the nature of practicals conducted in the BIOD building and the reagents used. The BIOD building tended to preserve numerous specimens in alcohol and formalin-based solutions that contain lower numbers and concentrations of VOCs compared to the solvents used in the BMBT building. Outdoor VOC concentrations had significant differences between PEH-BMBT, PEH-BIOD and BMBT-BIOD ($p < 0.01$). The BIOD building's outdoor VOC concentrations were the lowest compared to the PEH and BMBT buildings due to the activities going on in this building.

4.5. Summary

The results from this study indicate that there are significant differences in LAV (all three buildings), T_{db} (PEH and BMBT), RH (BIOD), O_3 (all three buildings), NO_2 (all three buildings), CO (all three buildings), CO_2 (all three buildings), NMHCs (BMBT and BIOD), and VOCs (all three buildings) between indoor and outdoor environments within the university. For CO_2, NMHCs, and VOCs, indoor sources outweighed outdoor sources and as such, significant differences between indoor and outdoor environments were found.

The type of building significantly affected AQ on the university campus. Results were indicative of significant differences in the AQ between the PEH and BMBT buildings for outdoor LAV, indoor RH, indoor and outdoor T_{db}, indoor CO_2, indoor NMHCs and indoor and outdoor VOCs. Statistically significant differences in AQ between the PEH and BIOD buildings were evident for outdoor RH, indoor and outdoor T_{db}, indoor SO_2, indoor CO_2, indoor H_2S, indoor and outdoor NMHCs, and outdoor VOCs. Lastly, significant differences in AQ between the BMBT and BIOD buildings were identified for indoor LAV, indoor and outdoor RH, outdoor T_{db}, outdoor O_3, indoor SO_2, outdoor NMHCs, and indoor and outdoor VOCs.

CHAPTER 5
SEASONAL PATTERNS OF AIR QUALITY IN THE STUDY AREA

5.1. Introduction

This chapter discusses seasonal patterns and differences of indoor and OAQ in the study area. A discussion on the differences observed and justifications for these differences are also included. Furthermore, the level of compliance/conformity of AQ in the study area to different AQ guidelines/standards are highlighted.

5.2. Seasonal variations of factors affecting air quality

5.2.1 Linear air velocity

Indoor LAV values for all four seasons were lower than the outdoor LAV values in all three buildings with all indoor LAV values being below 0.04 m/s (Figure 5.1). Significant differences were found between summer and winter and between autumn and winter outdoor LAV values ($p < 0.008$). The seasonal patterns of LAV across all three buildings varied, with patterns of outdoor LAV values for the BIOD building being winter > spring > autumn > summer, whereas the pattern for the BMBT and PEH buildings followed the order autumn > spring > summer > winter and autumn > spring > winter > summer respectively. Values for seasonal outdoor LAV obtained in this study are similar to what has been obtained in other studies. For example, Wangchuk *et al*. (2015) and Zhang *et al*. (2015a) found lower outdoor LAV values during summer and higher values in winter and spring in school environments in Bhutan and China respectively. In contrast to the current findings, Al-Rehaili (1999) found higher outdoor LAV values during summer (1.5–4.5 m/s), in comparison to winter (1.0–2.8 m/s) across buildings in Saudi Arabia. The day-to-day activities taking place in these buildings such as lectures, practicals, tutorials/seminars, workshops, and meetings occur irrespective of the prevalent season. The only factor that could affect LAV readings across the indoor setting during the various seasons is the thermal comfort of occupants which vary according to individual preferences. During autumn and winter, air conditioners are set at high temperatures and heaters are used, whereas during summer and spring, air conditioners are set at lower temperatures, in addition to the use of fans. These conditions may have contributed to the seasonal differences observed in indoor LAV values across buildings. Across the indoor environment of BMBT, during autumn, a zero value was encountered for LAV, due to the

majority of the indoor sampled environments not having any source of ventilation, as doors and windows mostly remained closed.

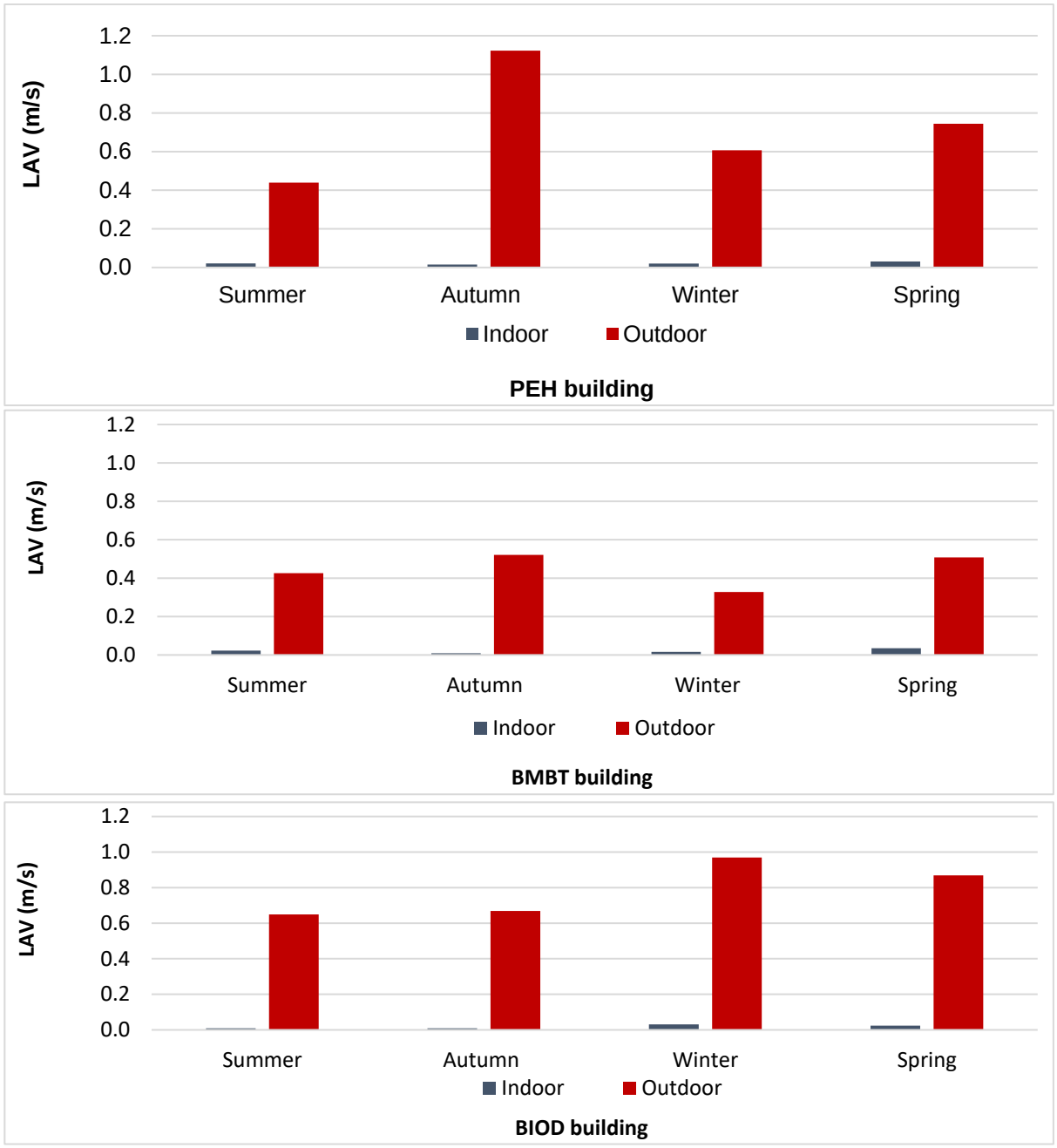

Figure 5.1: Seasonal variations of linear air velocity for all three buildings.

Despite these differences, all indoor LAV values conformed to the ASHRAE guideline of < 0.25 m/s for three of the four seasons except for the two exceedances encountered during the spring season (Figure 5.2). The two exceedances and other peaks across the other seasons were attributed to LAV readings taken in offices where air conditioners and fans were operational most of the time.

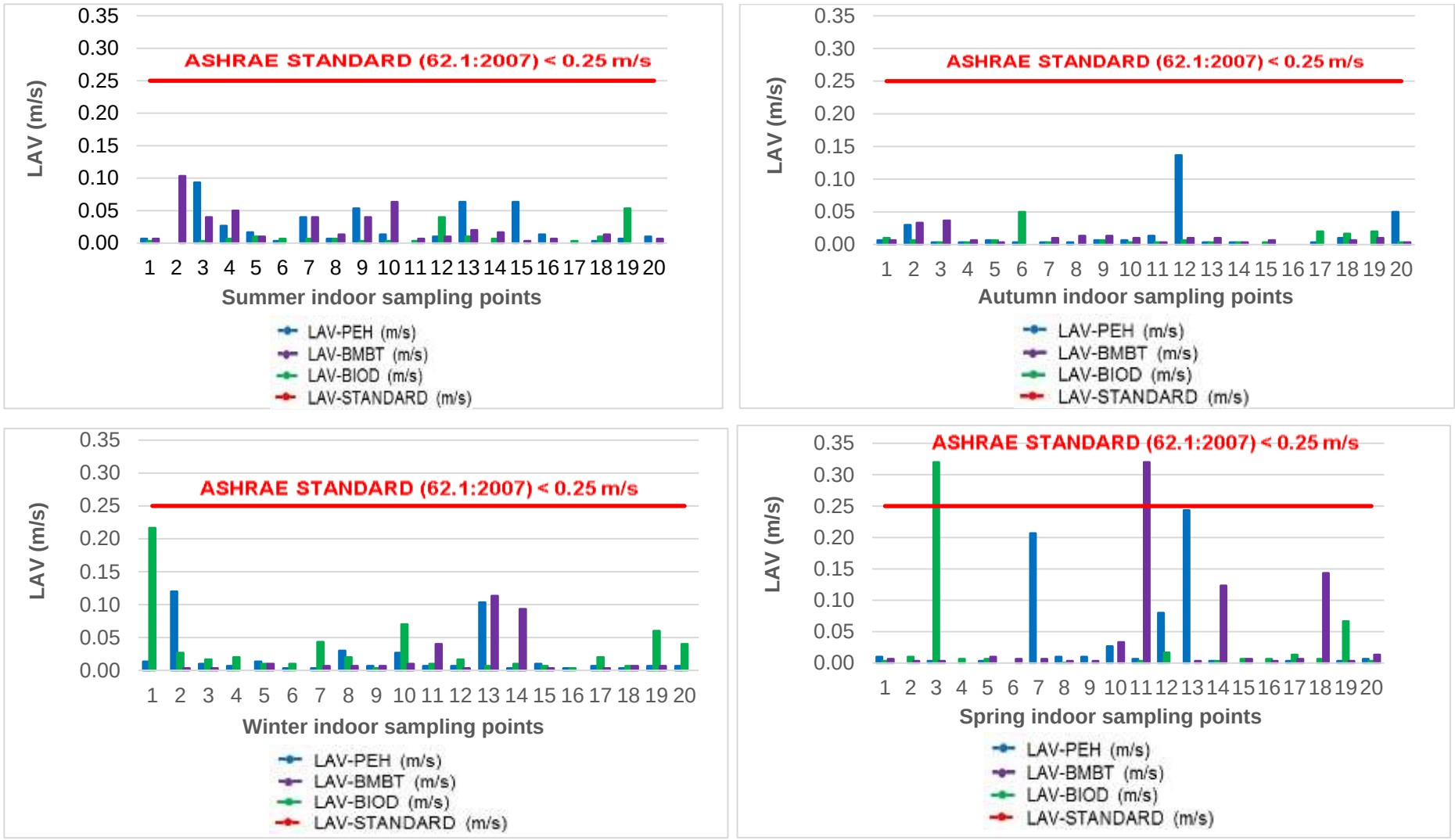

Figure 5.2: Indoor linear air velocity rates for all seasons within all three departments in comparison to the ASHRAE standard of 2007.

During all four seasons, there were numerous times when the indoor LAV readings rendered zero values (dips) (Figure 5.2), which may have come about due to poor ventilation mechanisms within the rooms.

There were no seasonal differences in outdoor LAV values in the BIOD building ($p = 0.069$), but significant differences were observed between autumn and winter in both the PEH and BMBT ($p < 0.008$) buildings, and between summer and autumn and summer and spring ($p < 0.008$) seasons in the PEH building. Mean seasonal outdoor LAV values recorded for the study area were 0.77, 0.71, 0.51 and 0.64 m/s for autumn, spring, summer, and winter respectively. Wind generally flows from an area of high pressure to that of low pressure during colder temperatures (Sciencing, 2019). Summer months are characterised by lower outdoor LAV readings, whilst colder months have higher outdoor LAV readings due to colder temperatures creating areas of high pressure, which in turn increase wind speed (Sciencing, 2019).

5.2.2. Dry-bulb temperature

Indoor temperatures across all three buildings showed the same pattern during all four seasons with the highest T_{db} values recorded during summer and spring and the lowest during the winter season (Figure 5.3). Outdoor T_{db} values were the highest during summer and spring and the lowest during winter in the PEH and BMBT buildings. The findings in this study are in conjunction with the findings of Kalimeri *et al*. (2016); Zhang *et al*. (2015a); Gao *et al*. (2014) and Mentese *et al*. (2012) who all found summer highs and winter lows for T_{db} values. In contrast, the BIOD building had the highest outdoor T_{db} value in spring and autumn, and the lowest values in winter. The results are also in agreement with the prevailing weather conditions during these seasons where T_{db} average values followed the seasonal pattern: summer (22.1 °C) > spring (19.7 °C) > autumn (18.7 °C) > winter (14.0 °C) (SAWS, 2019a). Significant seasonal differences in indoor T_{db} values were found between summer and the other three seasons, between autumn and winter, and between winter and spring in all three buildings ($p < 0.008$). Differences in temperature between autumn and spring were significant in the BIOD and BMBT buildings ($p < 0.008$) but not in the PEH building ($p = 0.727$). The prevalent meteorological conditions during sampling could have led to temperature variances within the study area. Fadeyi *et al*. (2014), Gao *et al*. (2014) and Pereira *et al*. (2014) all found meteorological conditions such as solar radiation to play a role in observed T_{db} values.

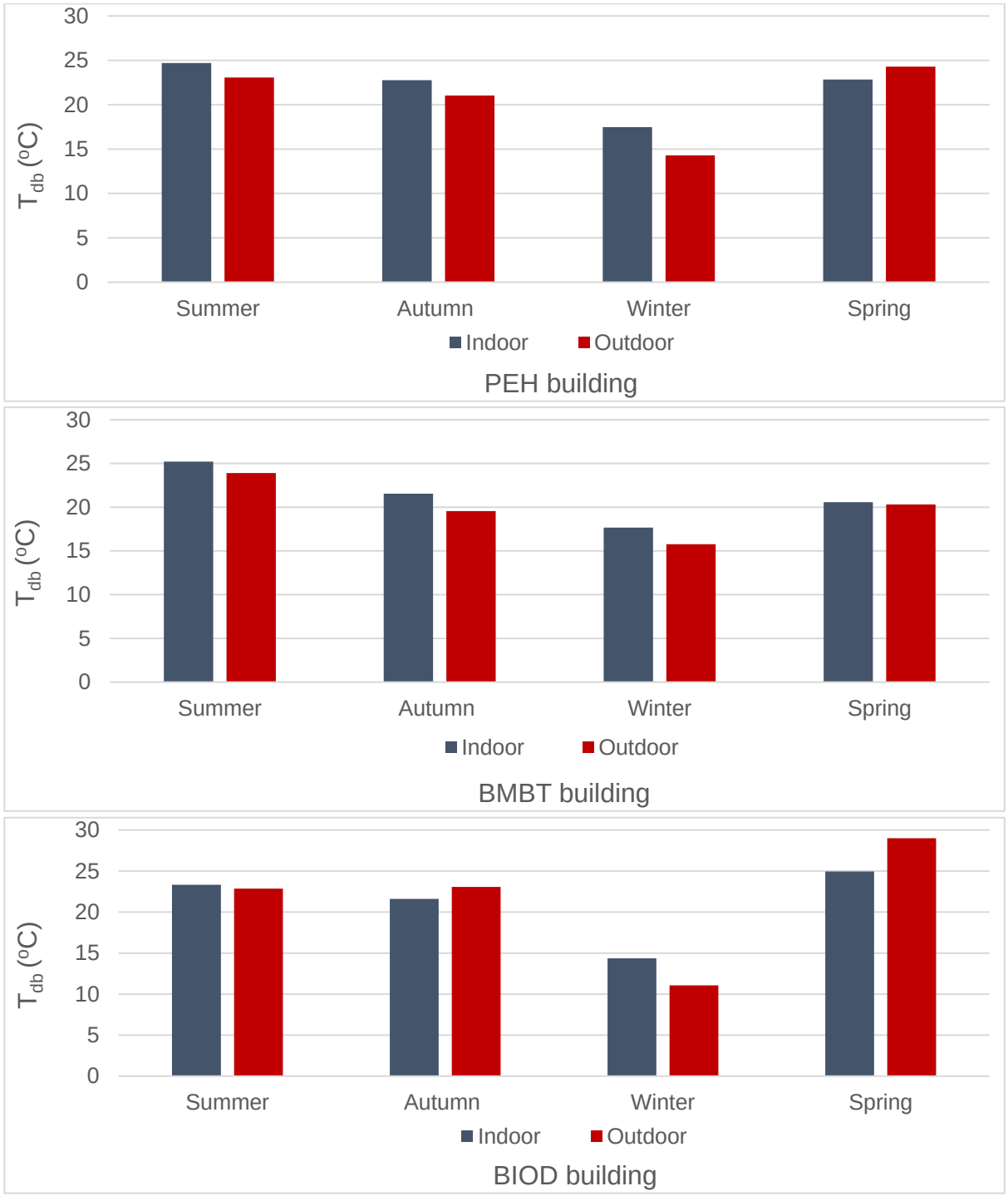

Figure 5.3: Seasonal variations of dry-bulb temperature for all three buildings.

During summer and spring, more sunlight would increase outdoor T_{db} values. The contrary would be true during autumn and winter seasons when the intensity and duration of solar radiation is lower. The main factors affecting outdoor atmospheric temperature gradients are energy transfer from warmer regions to colder regions (Hanrahan, 2012; de Nevers, 2000). During autumn and winter periods the ground surface is cool, subsequently cooling

the layer of air above. Lower T_{db} values during the colder months are therefore not unexpected. In contrast, during warmer seasons (summer and spring), the ground surface is heated by solar radiation which in turn heats the layer of air above the ground by the same heat exchange mechanisms as in the winter period (de Nevers, 2000). The higher outdoor T_{db} values measured during the summer and spring seasons for the current study are explained by heat transfer from the ground surface to adjacent layers.

Except for summer and spring in the PEH building ($p = 0.211$), autumn and spring in the BMBT building ($p = 0.093$) and summer and autumn in the BIOD ($p = 0.501$), all seasonal differences in outdoor T_{db} levels were significant ($p < 0.008$). Once again both conformity and non-conformity of the T_{db} readings with the ASHRAE (55 of 2010) guidelines were observed (Figure 5.4). During autumn a similar trend to summer is seen with dips in the early morning and then conformity as the day progresses. The non-conformity throughout winter could be attributed to the cold winds and fog encountered throughout the season in the outdoor environment, which led to lower outdoor T_{db} levels. The prevailing temperatures in the buildings are likely to have had an effect on air pollutants in the building. Ozone, SO_2, NO_2, CO, CO_2, H_2S, NMHCs, and VOCs tend to increase in concentration during periods of higher temperatures because the reactions producing these air pollutants occur at a faster rate. Subsequently, peaking in summer and spring and dipping during autumn and winter. Thus one would expect autumn and winter concentrations for these pollutants to be low, whereas during summer and spring their concentrations would be high.

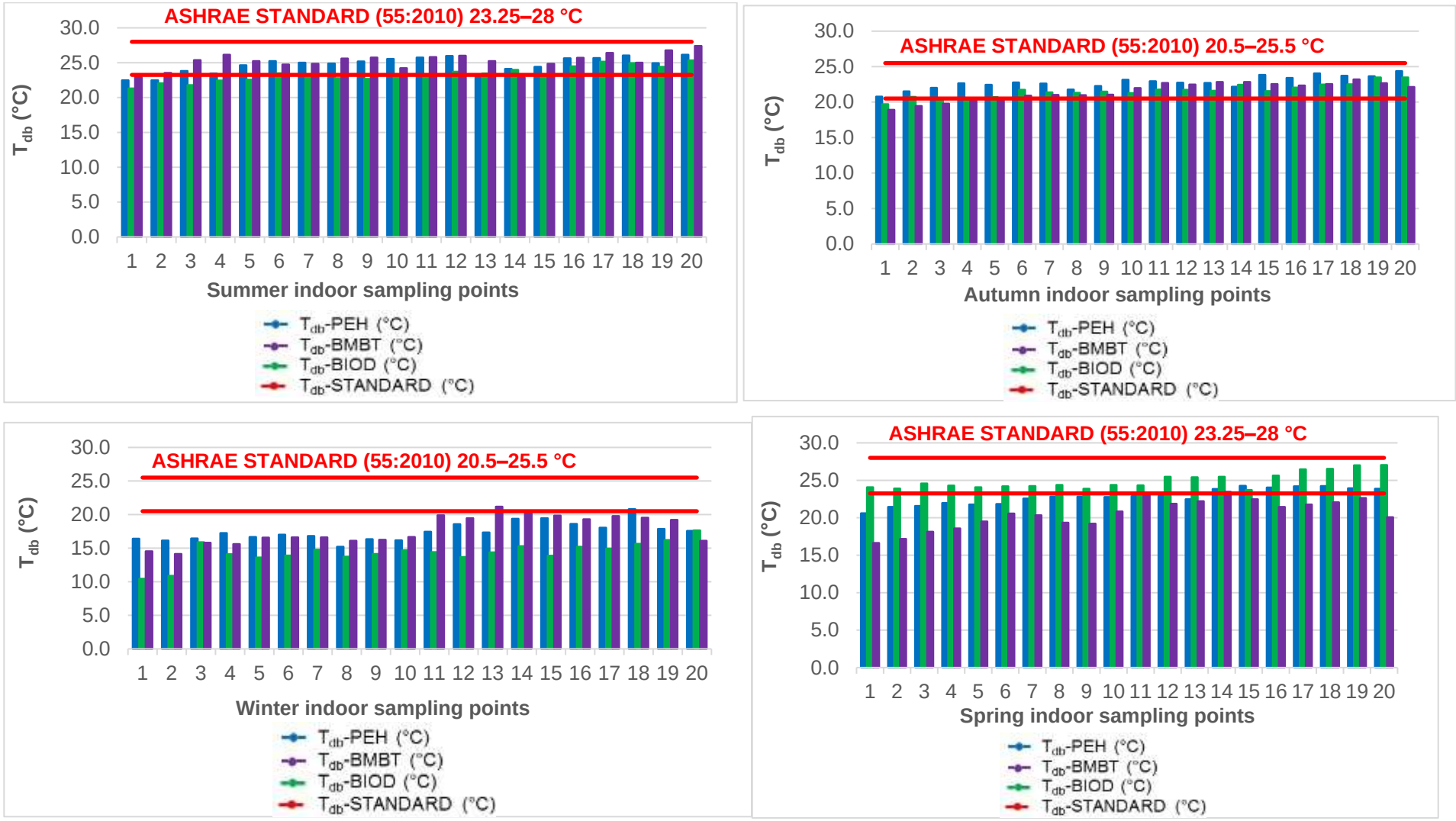

Figure 5.4: Indoor dry-bulb temperatures for all seasons across the three departments in comparison to the ASHRAE standard of 2010.

5.2.3. Relative humidity

The highest indoor RH levels were recorded during autumn, followed by summer, winter and then spring (Figure 5.5). Contrary to the current study findings, Kalimeri *et al*. (2016) reported indoor RH summer highs and winter lows within primary schools in Greece. Significant seasonal differences in indoor RH were found across all three buildings ($p < 0.008$), with exceptions seen across the BMBT building between summer and autumn ($p = 0.117$), summer and winter ($p = 0.295$) and the autumn and winter season ($p = 0.011$). A similar trend was seen in the outdoor environment across the PEH and BMBT buildings as well (Figure 5.5).

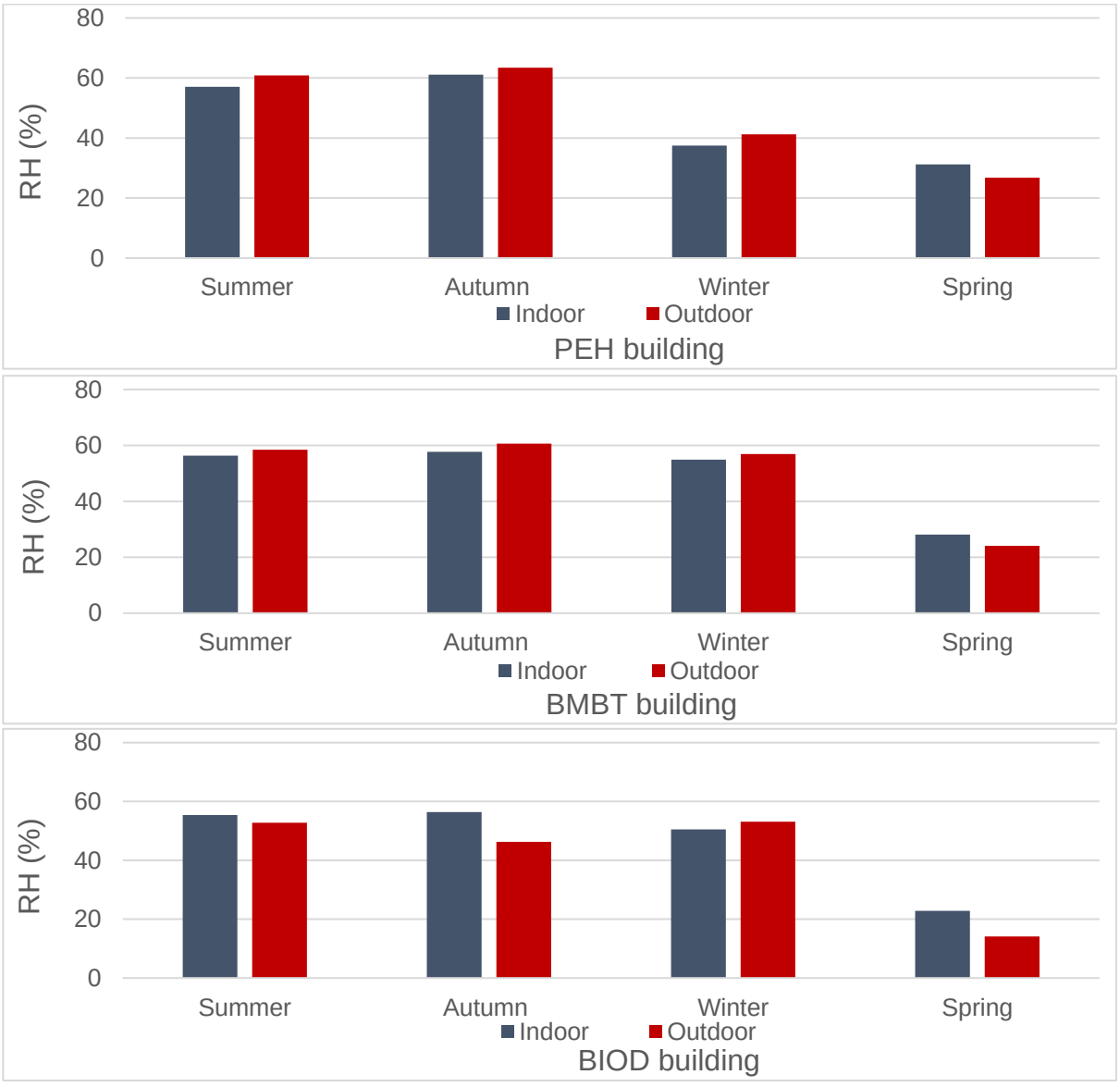

Figure 5.5: Seasonal variations of relative humidity for all three buildings.

The outdoor RH levels across the BIOD building had a different pattern with winter highs and spring lows. The outdoor RH seasonal patterns seen within the PEH and BMBT buildings are similar to the findings of Zhang *et al*. (2015a) and Wangchuk *et al*. (2015) who also reported autumn highs and spring lows in Beijing, China. In contrast, both Mentese *et al*. (2012) and Al-Rehaili (1999) found summer lows and winter highs for outdoor RH readings, similar to the RH levels encountered in the BIOD building. The general trend of outdoor RH levels as per the SAWS (2019a) data, were summer and autumn peaks with dips encountered in spring, as was the case in the current study. Significant seasonal differences in indoor RH were found across all three buildings ($p < 0.008$), with exceptions seen across the BMBT building between summer and autumn ($p = 0.117$), summer and winter ($p = 0.295$) and autumn and winter ($p = 0.011$) seasons. There were also significant differences in seasonal outdoor RH values in the three buildings ($p < 0.008$). Exceptions were found across the PEH and BMBT buildings between summer and autumn ($p = 0.021$ and $p = 0.064$) and in the BMBT and BIOD buildings between the summer and winter seasons ($p = 0.341$ and $p = 0.926$).

Non-conformity of indoor RH readings with the ASHRAE guideline 55 of 1999 were encountered during all four seasons. The summer, autumn and winter seasons (Figure 5.6) exceeded the upper limit of the ASHRAE guideline (1999) of 60 % whereas, during spring, the RH values were below the lower limit of the ASHRAE guideline (1999) of 30 %. Warm air can hold more moisture than cool air. Hence, an increase in temperature would increase the saturated moisture content and decrease RH (Mason *et al*., 2001). During the spring season, the T_{db} values are higher (16.6 to 27.0 ˚C) than in the winter season (10.4 to 21.2 ˚C). Lower RH is therefore expected within the indoor environments due to higher temperatures. In winter, building occupants tend to switch on their heaters and set their air conditioners at higher temperatures, subsequently raising the air temperatures and reducing the amount of moisture that the air can hold resulting in lower RH levels across indoor environments. During autumn (18.9 to 24.4 ˚C), RH levels increase as a result of the changes in temperature (Dec *et al*., 2018; Money, 1988). Hence, differences in outdoor RH values between seasons across all three buildings are expected as per the inverse relationship of temperature with RH (Dec *et al*., 2018; Money, 1988).

Figure 5.6: Indoor relative humidity percentages for all seasons across the three departments in comparison to the ASHRAE standard of 1999.

High RH values observed in summer could have been caused by rainfall during this season. Summer and early autumn are generally periods characterised by high precipitation levels in South Africa which increases the moisture content in the air (SAWS, 2019a). Another factor that may have contributed to indoor RH peak levels during the autumn and winter seasons could have been fog because of its high moisture content. Pollutants are synthesized from other compounds more readily during periods of higher T_{db}, which usually coincide with lower RH. The dispersion of pollutants including O_3, SO_2, NO_2, CO, and CO_2 tends to increase with a decrease in RH. One would expect to find summer and spring peaks for these pollutants and minimal concentrations during autumn and winter.

5.3 Seasonal variation of air quality parameters in study area

5.3.1 Ozone

Even with the minimal O_3 concentrations in the study area, a seasonal pattern could be identified. Indoor O_3 concentrations in the BMBT building were negligible during all four seasons but in the BIOD building, some level of O_3 was detected indoors during spring and winter (Figure 5.7). Indoor O_3 concentrations in the PEH building were detectable during spring, winter and summer with the highest concentration values recorded in summer (Figure 5.7). There were insignificant seasonal differences in indoor O_3 concentrations between the PEH and BMBT buildings ($p = 0.328$ and $p = 0.112$ respectively), whilst significant seasonal differences were found in the indoor environment of the BIOD building ($p < 0.01$) between the summer and winter and autumn and winter seasons ($p < 0.008$).

Seasonal outdoor O_3 concentrations followed the order spring > winter > autumn > summer for the BMBT and BIOD buildings but in the PEH building, the pattern was slightly different being spring > winter > summer > autumn. These outdoor seasonal patterns for O_3 concentrations are similar to those reported by Kalimeri *et al*. (2016) and Josipovic *et al*. (2010) in Greece and South Africa. Contradictory findings were seen across the studies conducted by Wheida *et al*. (2018); Masiol *et al*. (2017) and Zhao *et al*. (2016) who all found winter lows and summer highs. In contrast to the current study findings, Zheng *et al*. (2010) and Yamaji *et al*. (2006) both found autumn highs and summer lows concerning O_3 concentrations.

Across the indoor environment, zero O_3 concentration readings were encountered during winter in the PEH and BMBT buildings. Additionally, across the indoor environment of the BMBT building, zero O_3 concentration values were also found for the summer, autumn and spring seasons. The BIOD building also rendered zero O_3 concentration readings across the summer and autumn seasons. No inherent sources of O_3 were present across these indoor environments within the respective buildings, thus explaining the zero readings obtained as seen in Figure 5.7 below.

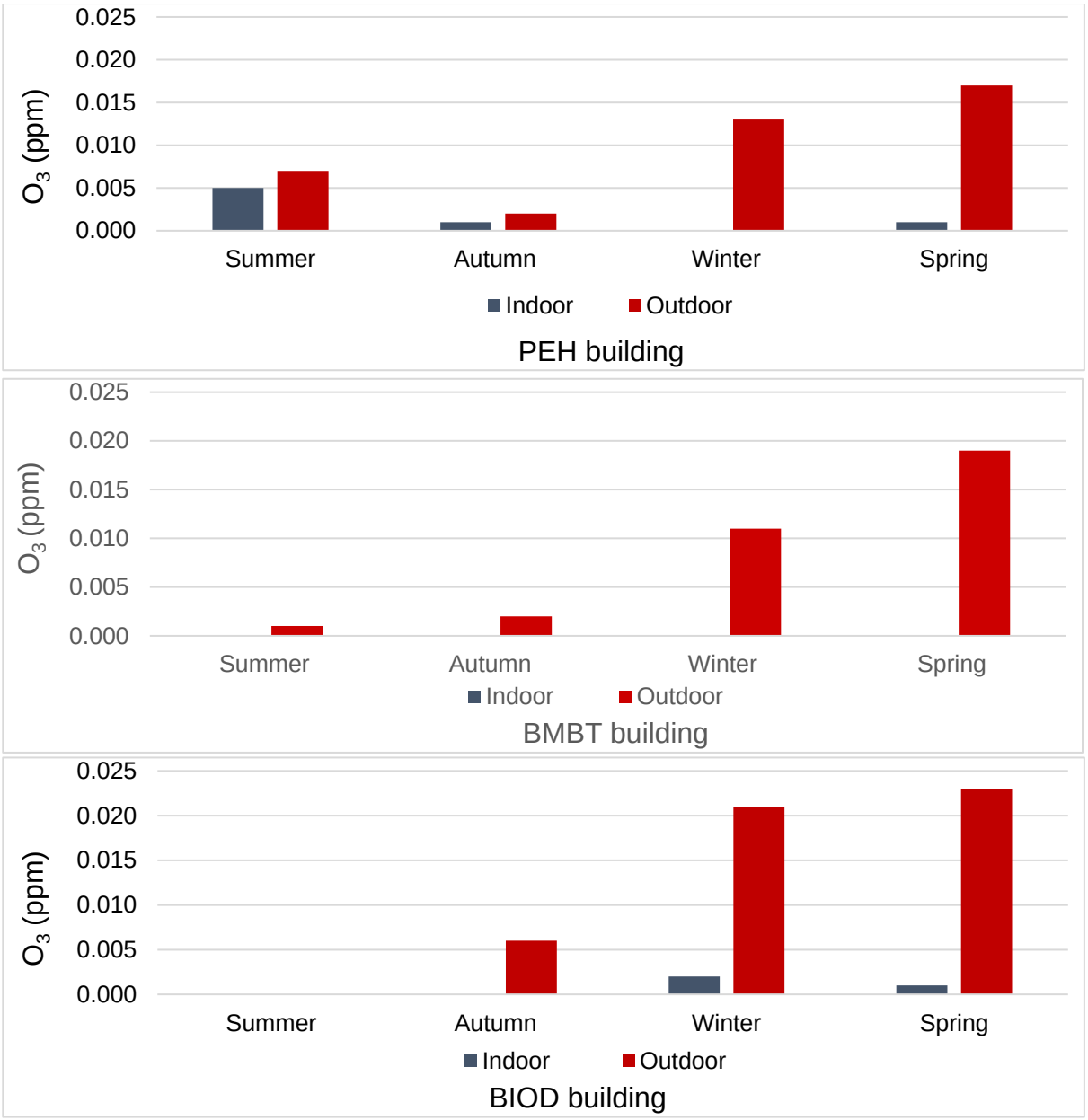

Figure 5.7: Seasonal variations of ozone for all three buildings.

Across the outdoor environment, significant differences were found across all three buildings ($p < 0.01$). Ozone concentrations in the outdoor environment of the PEH building showed seasonal differences between summer and spring, autumn and winter and autumn and spring ($p < 0.008$). The BMBT building's outdoor O_3 concentrations showed seasonal differences between summer and winter, summer and spring, autumn and winter, autumn and spring and winter and spring. Lastly, outdoor O_3 concentrations in the BIOD building showed seasonal differences between the summer and autumn, summer and winter, summer and spring, autumn and winter and autumn and spring ($p < 0.008$ for all pairs respectively). The norm for O_3 concentrations, are to peak during spring and summer months due to favourable meteorological conditions (temperatures beyond 32 ˚C, low LAV readings, extreme radiation, and truncated rainfall) for photochemical reactions in the atmosphere (Pepper *et al*., 2006). Furthermore, high O_3 concentrations encountered in spring can be substantiated by the corresponding peaks of NMHCs during the same season (Figure 5.25). Non-methane hydrocarbons play a critical role in O_3 formation, whereby for every NMHC molecule oxidised, two O_3 molecules and carbonyl compounds are formed. These carbonyl compounds can also produce more O_3 through the production of H^+ radicals as indicated in equations 5.1–5.5 below (Kumar *et al*., 2017):

$NMHC + OH + O_2 \rightarrow R$ (hydrocarbon radical) $+ NO + O_2$ (Equation 5.1);

$RO_2 + NO + O_2 \rightarrow NO_x + HO_2$ + carbonyl compounds (Equation 5.2);

$HO_2 + NO_x \rightarrow NO_2 + OH$ (Equation 5.3);

$2(NO_2) + \hbar\upsilon^* + O_2 \rightarrow NO + O_3)$ (Equation 5.4);

Net: $NMHC + 4O_2 + \hbar\upsilon^* \rightarrow 2O_3$ + carbonyl compounds (Equation 5.5).

* $\hbar\upsilon$: photon of light energy.

During winter months, low solar radiation intensity tends to reduce O_3 concentrations (Elminir, 2005) because there are fewer photons present to react with NO_2 to produce O_3. The high precipitation levels in summer could also have contributed to the lower O_3 concentrations observed as a result of dissolution, washout (Mohtar *et al*., 2018) and wet deposition (Pearce *et al*., 2011) of O_3 by rainwater. Rainfall tends to cleanse the atmosphere by washing out air pollutants and their precursors (Pudasainee *et al*., 2006; Khemani *et al*., 1995; Ravindra *et al*., 2003; Lal *et al*., 2000). Zheng *et al*. (2010), Yamaji *et al*. (2006) and So and Wang (2003) all attributed O_3 concentration fluctuations during different seasons to differing meteorological parameters. According to literature, summer O_3 concentrations are usually elevated due to higher solar radiation intensity, higher

temperatures, and lower RH that promote the photo-chemical reactions that produce O_3 when rain is not present (Jhun *et al*., 2015; Zhang *et al*., 2015a; Zhang *et al*., 2015b; Levy *et al*., 2014). The difference in summer patterns of O_3 observed in this study could be attributed to rainfall during summer. During winter, O_3 concentrations decline due to stable vertical structures in the lower atmosphere, lower solar radiation intensities, lower temperatures, higher RH values, higher LAV values and the scavenging effects of NO_x (Jhun *et al*., 2015; Zhang *et al*., 2015a; Zhang *et al*., 2015b; Levy *et al*., 2014). From a South African context, both Lourens *et al*. (2011) and Josipovic *et al*. (2010) attributed similar O_3 concentration patterns found in this study to an increase in photochemical activity and elevated CO concentrations (an important precursor of O_3). All outdoor O_3 concentration values were within the DEA standard of 2009 (Figure 5.8). Compliance was seen throughout the four seasons of sampling, with a single exceedance in summer (Figure 5.8). The dips and zero values in O_3 concentrations were aligned with periods characterised by lower temperatures and higher precipitation days as found during the summer season. Similarly, when looking at the diurnal variation of O_3 concentrations across the entire sampling period in Figure 5.9, all concentrations were found to comply with the DEA standard (2009). Additionally, the calculated 8-hour running mean of O_3 (0.011 ppm) was also within the DEA standard (2009) of 0.106 ppm. Figure 5.9 is indicative of O_3 concentrations increasing steadily during the course of the day. The early morning lows are indicative of no inherent sources of O_3 being present, as the temperature increases, so too does the corresponding O_3 concentrations. The peak encountered at 16h30 can be attributed to vehicular emissions originating from university personnel leaving the premises, in addition to functional generator emissions in close proximity to the sampling points.

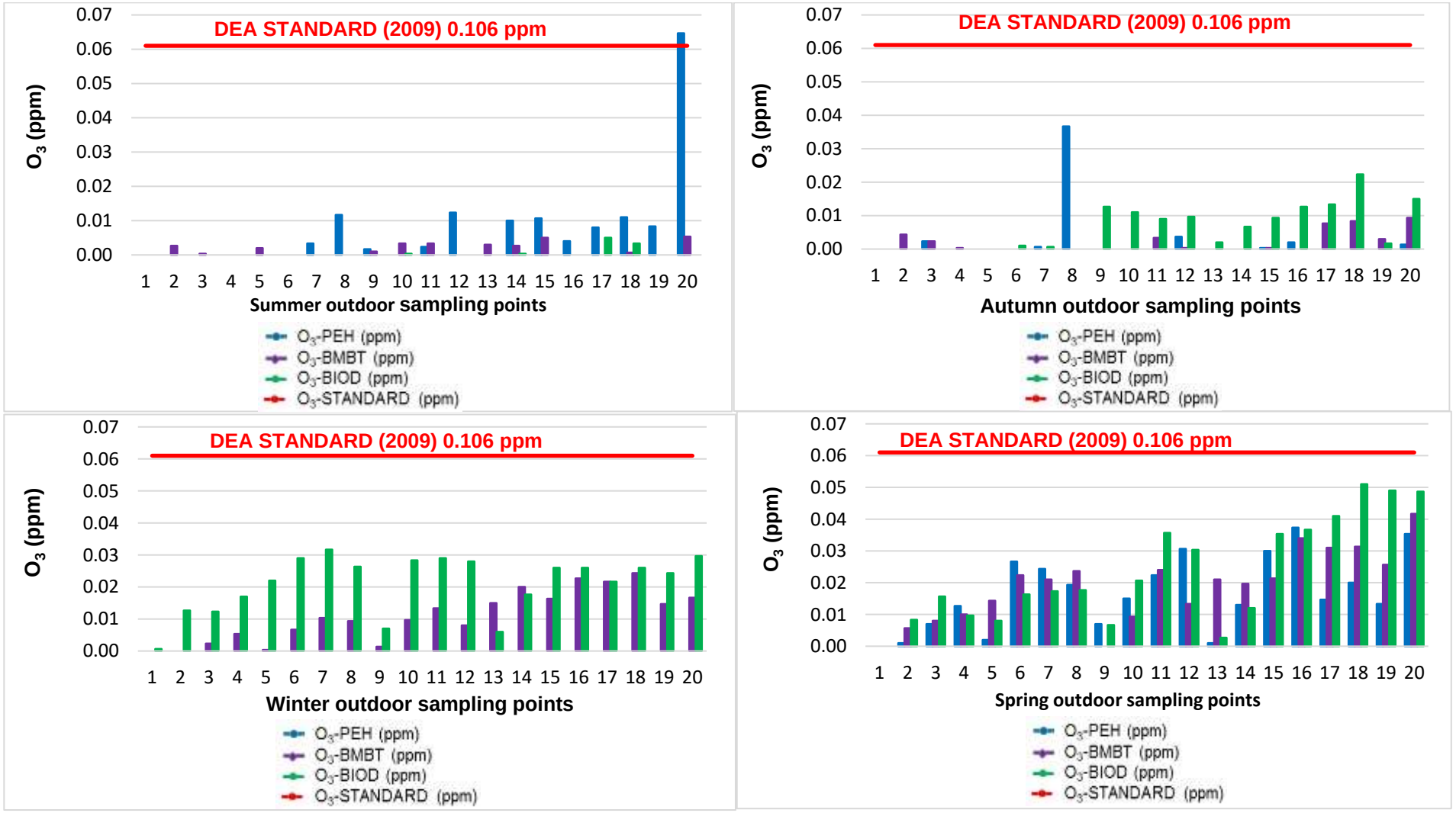

Figure 5.8: Outdoor ozone concentrations for all seasons across the three departments in comparison to the DEA standard of 2009.

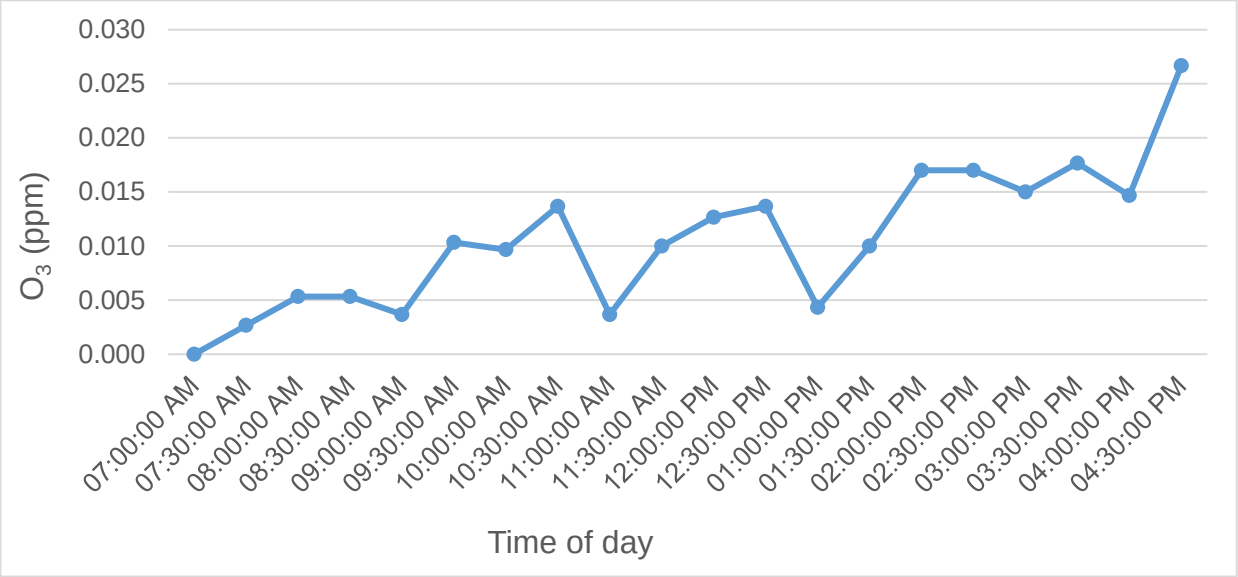

Figure 5.9: Diurnal ozone concentration variation during the entire sampling period.

5.3.2. Sulphur dioxide

Seasonal patterns of indoor SO_2 concentrations indicate a decrease from summer to spring to autumn and then to winter for both the PEH and BMBT buildings, whereas for the BIOD building, indoor SO_2 concentrations followed a different pattern (Figure 5.10). Differences in indoor SO_2 concentrations between summer and autumn, summer and winter and the summer and spring seasons in the PEH and BMBT buildings were significant ($p < 0.008$). No significant seasonal differences were recorded between indoor SO_2 concentrations in the BIOD building ($p > 0.008$). The normal pattern for SO_2, are for elevated concentrations to be found during winter due to domestic fossil fuel burning initiatives for purposes of heating (Masiol *et al*., 2017; Poberžnik and Štrumbelj, 2016; Zhao *et al*., 2013; Lourens *et al*., 2011). However, elevated temperatures are also found to favour SO_2 production as reported by Beard (2013) and Davis and Masten (2004), which are consistent with the current study findings during the summer season. Additionally, the use of generators due to load shedding may have acted as a confounder (using gasoline and diesel fuels) during summer, contributing toward the elevated SO_2 emissions measured. Due to load shedding during summer, several generators were used across the PEH and BMBT building surrounds which would have contributed to the elevated concentrations of SO_2 observed. The BIOD building rendered several zero readings for SO_2 across the indoor environment and as such insignificant differences between seasons were found. The distribution of SO_2 in the outdoor environment was slightly different from what was observed in the indoor environment during the different

seasons. The highest SO_2 concentrations occurred during summer around the PEH building and lowest concentrations were found in winter around the BMBT building (Figure 5.10). Indoor winter SO_2 concentrations across both BMBT and BIOD were found to be zero. This may be due to the higher RH levels encountered in winter that leads to the oxidisation of SO_2 (Zeng and Zhang, 2017; Hosseinibalam and Hejazi, 2012), in addition to no inherent indoor SO_2 source being present during the winter season across these buildings.

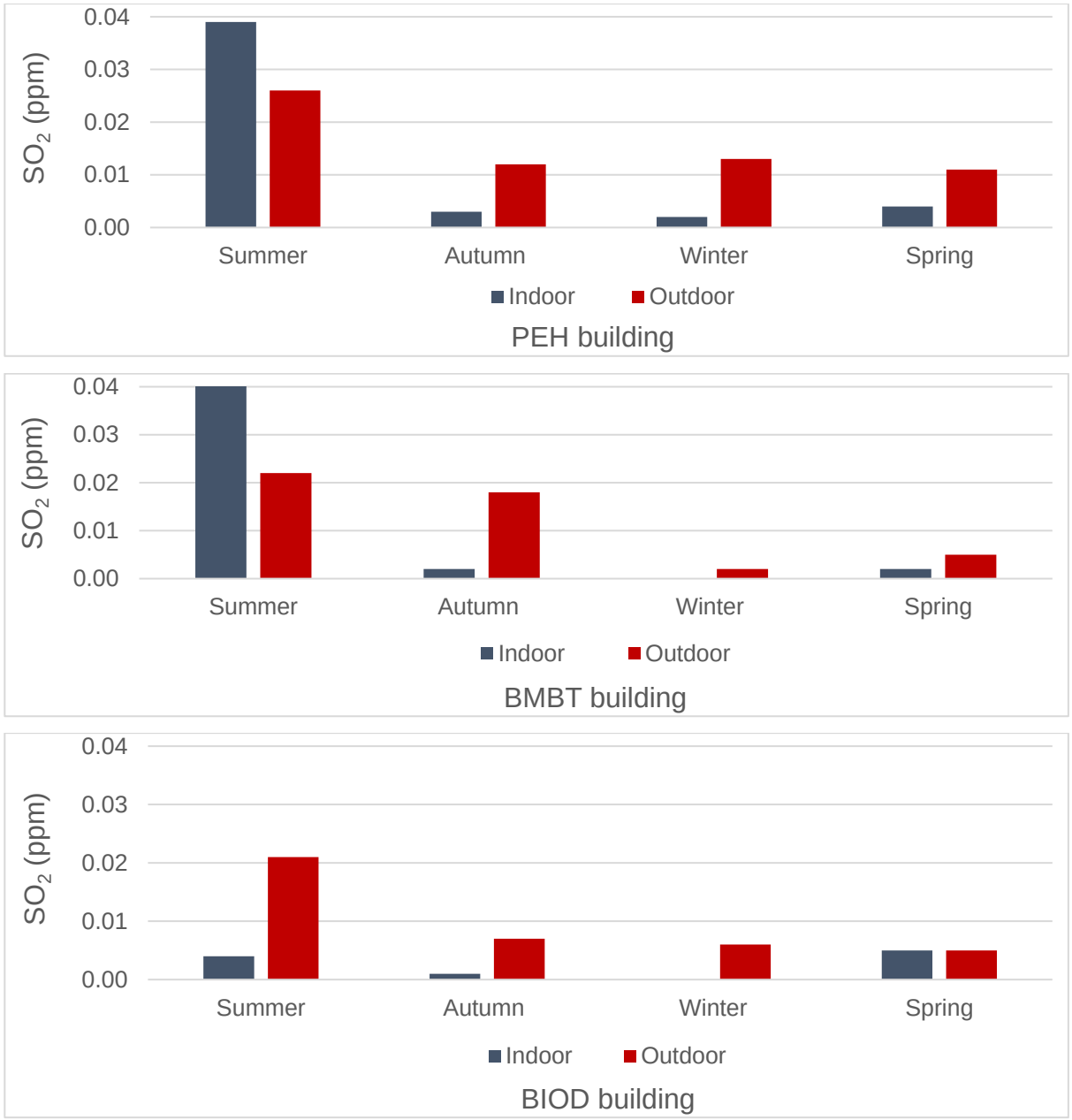

Figure 5.10: Seasonal variations of sulphur dioxide for all three buildings.

Outdoor SO_2 concentrations in the PEH building were found to be significantly higher in summer than in autumn ($p = 0.005$). Seasonal outdoor SO_2 concentrations between summer and winter, autumn and winter and autumn and spring seasons across the BMBT building were also significantly different ($p < 0.008$), whereas insignificant seasonal differences were found for the outdoor BIOD building's SO_2 concentrations ($p > 0.008$). Except for one instance around the BIOD building during summer where values for outdoor SO_2 were unusually high (attributed to vehicular and generator emissions in close proximity to the sampling point), the concentrations of SO_2 were generally within the standards proposed by the DEA (Figure 5.11). Summer and autumn highs were seen with winter and spring lows. The summer and autumn highs may be due to the generator emissions, due to them being primarily used during these months and not in winter or spring. Upon introspection and evaluation of Figure 5.12, the diurnal variation of SO_2 concentrations across the entire period of sampling were also found to be compliant to the DEA standard (2009) of 0.134 ppm. The peaks encountered in the early morning around 7h00 and 15h00 can be attributed to vehicular emissions from university personnel entering and leaving the parking bays during those specific times of the day. Additionally, those times of the day were characterised by miniscule wind speeds, which would lead to higher SO_2 concentrations at the respective sampling sites.

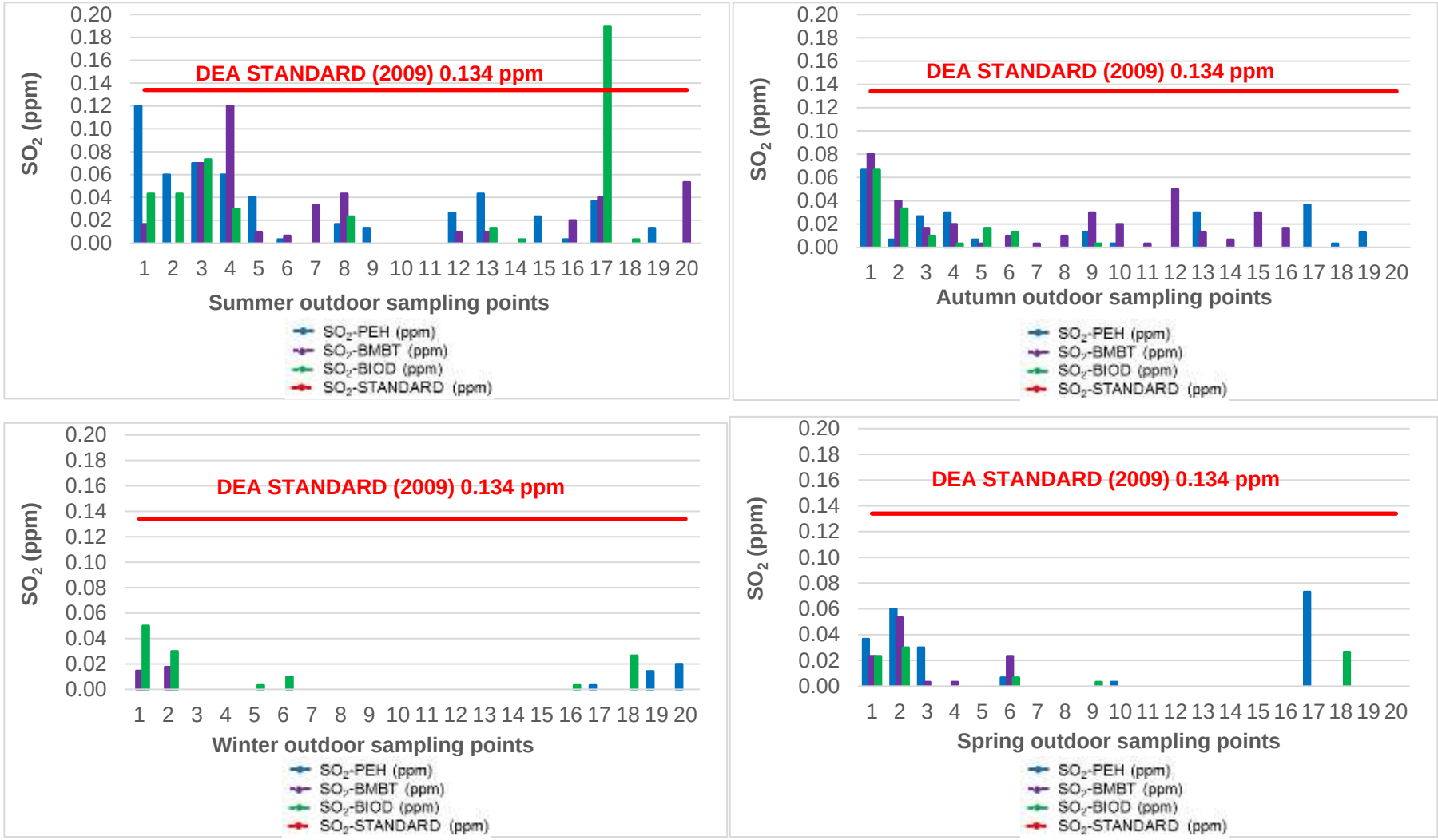

Figure 5.11: Outdoor sulphur dioxide concentrations for all seasons across the three departments in comparison to the DEA standard of 2009.

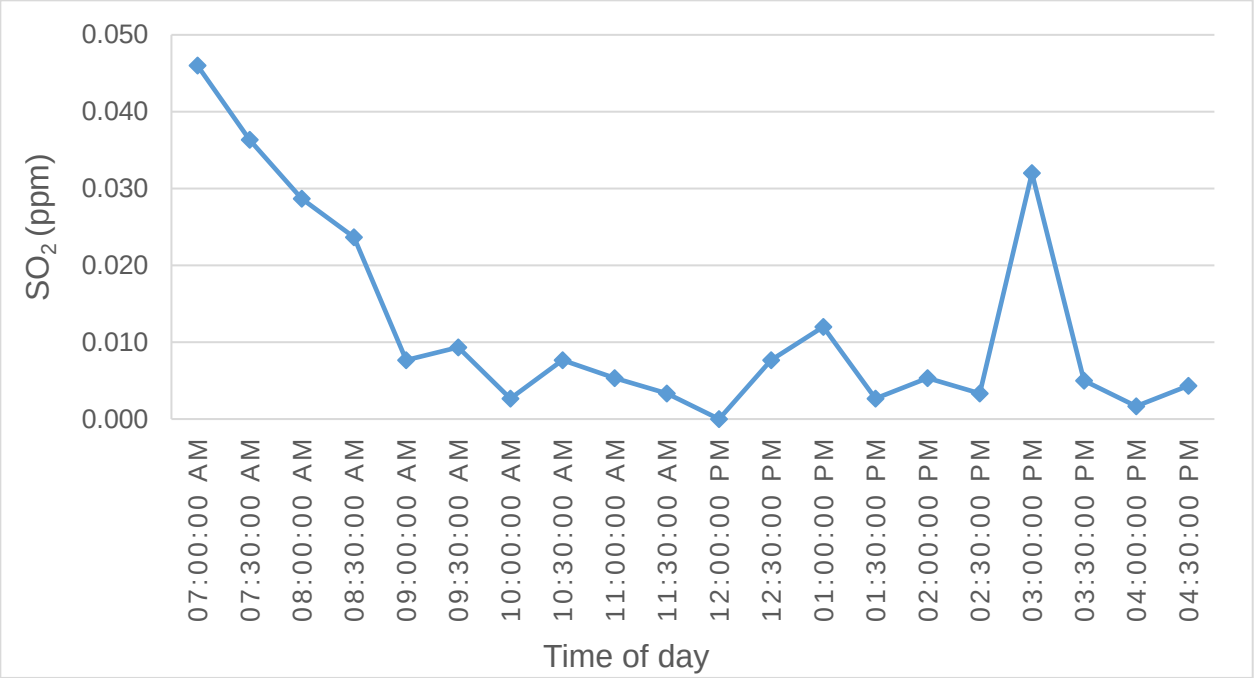

Figure 5.12: Diurnal sulphur dioxide concentration variations during the entire sampling period.

Literature points to outdoor SO_2 concentration peaks in winter (Poberžnik and Štrumbelj, 2016; Zhao *et al*., 2016; Wang *et al*., 2014; Zhao *et al*., 2013) and dips in summer (Chen *et al*., 2015; Wang *et al*., 2014; Zhao *et al*., 2013) due to fossil fuel burning initiatives for purposes of heating (Masiol *et al*., 2017; Poberžnik and Štrumbelj, 2016; Zhao *et al*., 2013; Lourens *et al*., 2011).

Further to this, outdoor SO_2 concentrations are also affected by meteorological conditions (Beard, 2013; Davis and Masten, 2004; De Santis *et al*., 1997; Smith and Jeffrey, 1967). In the current study, winter SO_2 peaks as per literature were not found. Sulphur is present in the atmosphere as a result of sulphur-based fossil fuel usage. It readily combines with O_2 in the air to produce SO_2 in the presence of intense sunlight, elevated temperatures and low RH levels (especially encountered during summer) (Beard, 2013; Davis and Masten, 2004). Wind direction also affects SO_2 concentrations in an outdoor environment (Mohtar *et al*., 2018; Chen *et al*., 2015). The higher SO_2 concentrations observed during summer may have been brought about by the long-range transport of SO_2 from neighbouring areas (coal-powered power plants) due to the prevalent wind direction. The summer wind direction reported by the SAWS (2019a) validates the long-range transport argument since the functional generators and the coal-powered power plants were both situated on the south-easterly side of the sampling sites. There were zero SO_2

concentration readings encountered throughout all four seasons; however, these values are predominantly seen during the winter and spring seasons. All these zero values may have come about due to the miniscule sunlight in these months and lower temperatures, compared to summer months which negate the formation of SO_2 (Beard, 2013; Davis and Masten, 2004).

5.3.3. Nitrogen dioxide

Seasonal patterns for NO_2 varied among the three buildings studied. In the PEH and BMBT buildings, indoor NO_2 concentrations displayed highest concentrations in spring and lowest concentrations in autumn and winter. In contrast, the BIOD building had spring highs and summer lows (Figure 5.13). These seasonal patterns for the PEH and BMBT buildings are similar to the results of Masiol *et al*. (2017), Kalimeri *et al*. (2016), Poberžnik and Štrumbelj, (2016) and Zhao *et al*. (2016). However, Ni *et al*. (2016) found indoor NO_2 concentrations to be elevated during summer and at minuscule concentrations during winter. Differences in indoor NO_2 concentrations between the autumn and spring seasons in the PEH building were significant ($p = 0.003$), whereas in the BMBT building, there were no significant differences ($p > 0.008$).

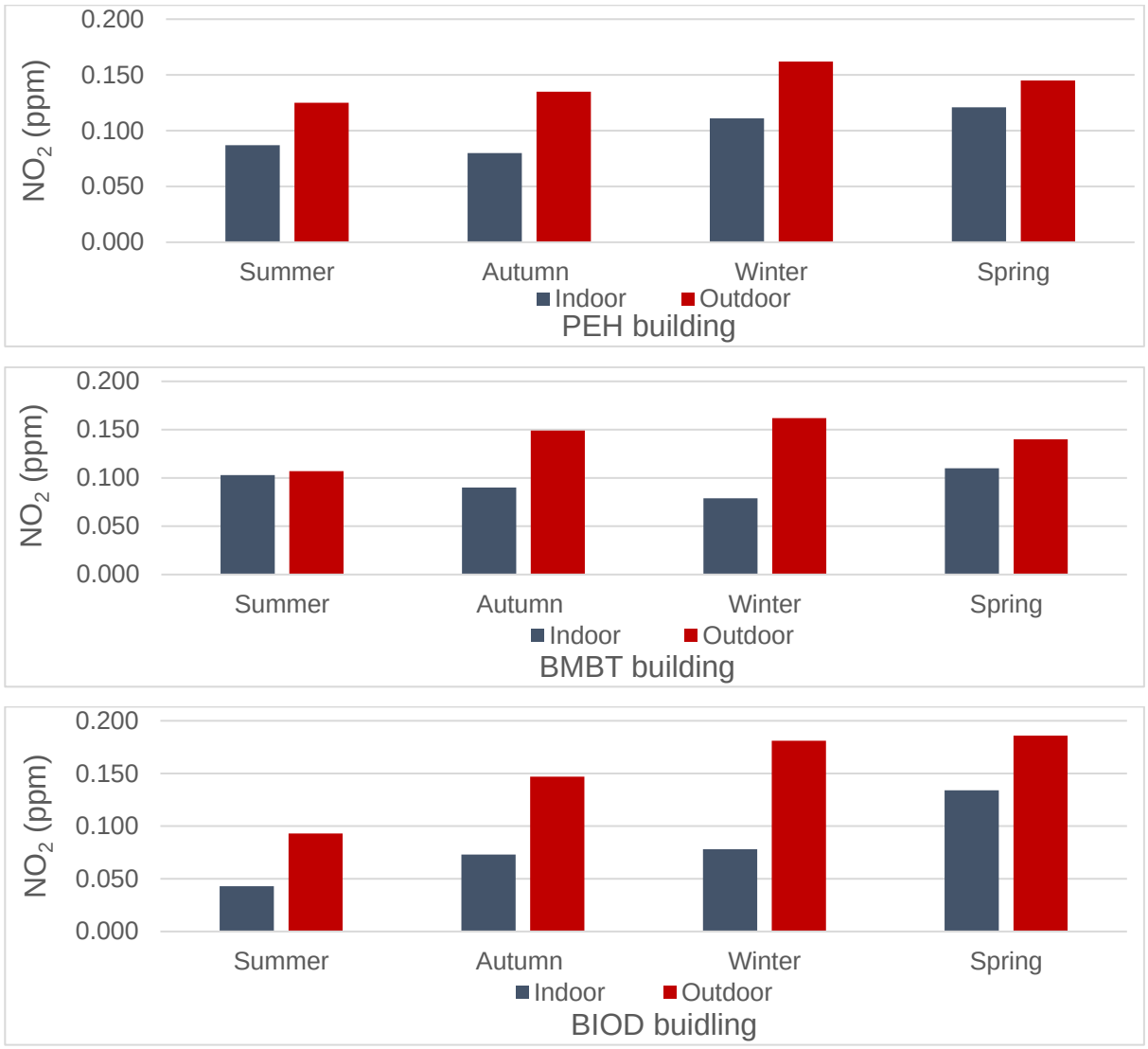

Figure 5.13: Seasonal variations of nitrogen dioxide for all three buildings.

In the BIOD building, values for indoor NO_2 concentrations during summer were lower than what was obtained during the other seasons ($p < 0.008$). Values obtained for NO_2 in spring were also significantly higher than those obtained in autumn and spring ($p < 0.008$). Across the outdoor environment, both the PEH and BMBT buildings exhibited winter highs and summer lows, whilst across the BIOD building, spring highs and summer lows were found for NO_2 concentrations. Outdoor NO_2 concentrations in the BMBT building during summer were significantly lower than what was obtained in winter ($p = 0.001$). In the BIOD building, outdoor NO_2 concentrations values obtained in summer were lower than NO_2 concentrations obtained during the other seasons ($p < 0.008$). Results from studies carried out by Lourens *et al*. (2011), Pearce *et al*. (2011), and Khoder (2009) indicated that outdoor NO_2 summer minima occurred as a result of rain scavenging and unstable meteorological conditions. They found winter maxima to be associated with surface

inversion layers increasing primary pollutant concentrations at the ground surface, in addition to combustion initiatives used for heating during winter.

The high outdoor NO_2 concentrations attained in winter across all three buildings could be as a result of stagnant meteorological conditions, characterised by slow or weak winds as described in section 5.2.1. Additionally, shallow mixing layers that are prevalent during winter may trap NO_2 near the ground surface leading to higher concentrations (Zhao *et al*., 2016; Wang *et al*., 2014) as described in section 5.2.2. Nitrogen and O_2 coexist in the atmosphere without any reaction. However, at extremely high temperatures and pressure, nitrogen and O_2 react to form NO which reacts with O_3 or O_2 to form NO_2 (Davis and Masten, 2004). These reactions may explain the concentrations of NO_2 encountered in this study during the summer and spring seasons. Further to this, weak to moderate winds would have led to increased NO_2 concentrations at the ground surface. The persistent inversion layers ensured that NO_2 is trapped at the ground surface and not dissipated higher into the atmosphere. The prevalent wind speeds for the respective seasons across the three buildings were 0.505 m/s in summer, 0.771 m/s in autumn, 0.635 m/s in winter and 0.707 m/s in spring, thus indicative of weak to moderate winds that would accentuate the elevated ground-level NO_2 concentrations throughout the four seasons.

Non-compliance with outdoor NO_2 concentrations with the DEA standard (2009) were observed for several sampling points across all four seasons in the different buildings (Figure 5.14). The dips seen during summer may have come about due to precipitation having washed out NO_2. A similar trend is also seen with the diurnal variations of NO_2 concentrations during the entire sampling period, with the majority of values being non-compliant to the DEA standard (2009) of 0.106 ppm (Figure 5.15). Early morning peaks are seen, which may be attributed to the arrival of university personnel at work with their respective vehicles. Throughout the rest of the day, NO_2 concentrations tended to meander close to one another, in addition to being non-compliant to the DEA standard (2009); which indicated the presence of inherent NO_2 sources in the study area. These exceedances may have been brought about by electricity generating areas in the outskirts of the university premises, laboratories and welding workshops harbouring nitrogenous chemicals and nitric acid within the university, biogenic emissions and motor vehicle and generator emissions as established in section 4.4.3.

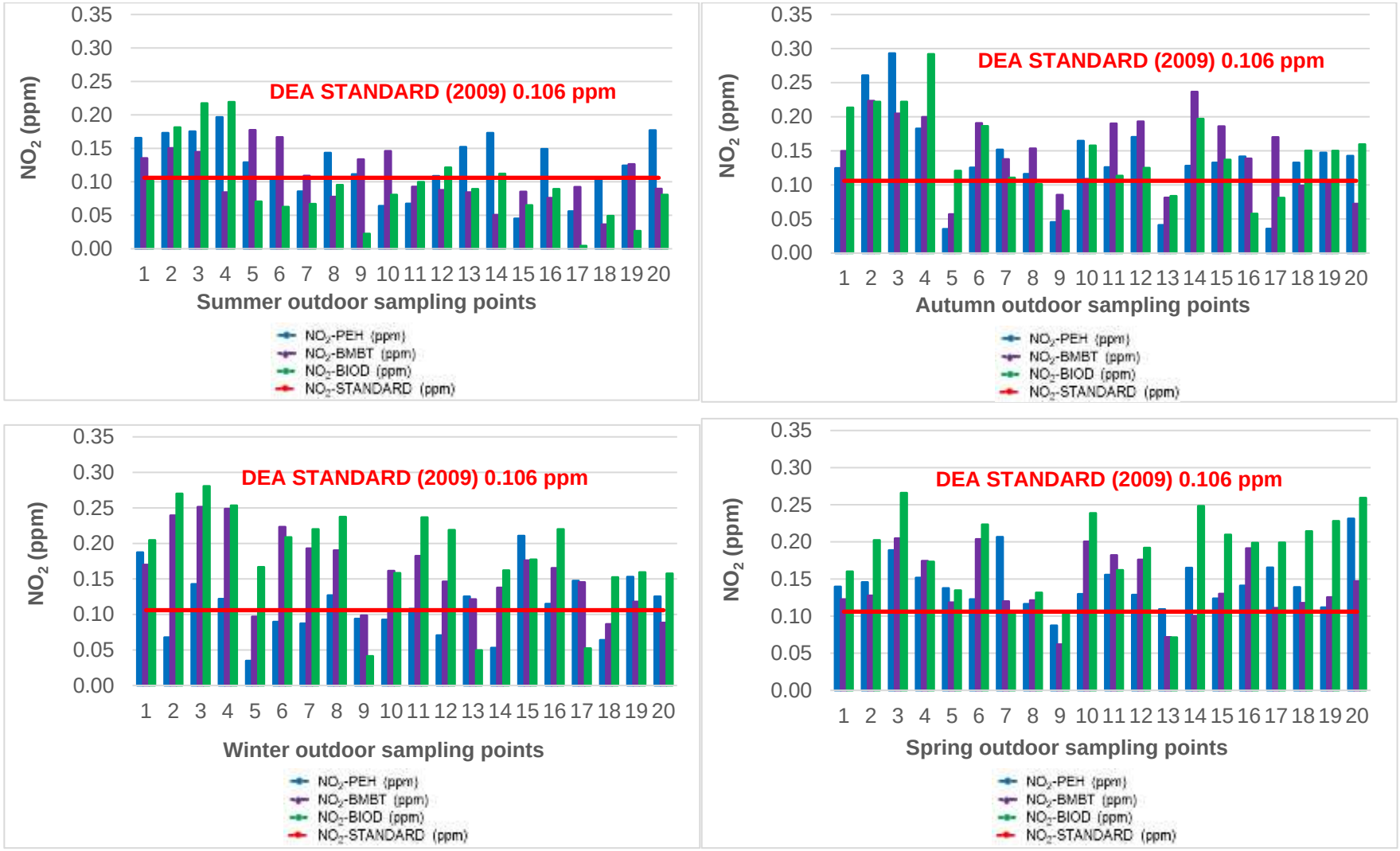

Figure 5.14: Outdoor nitrogen dioxide concentrations for all seasons across the three departments in comparison to the DEA standard of 2009.

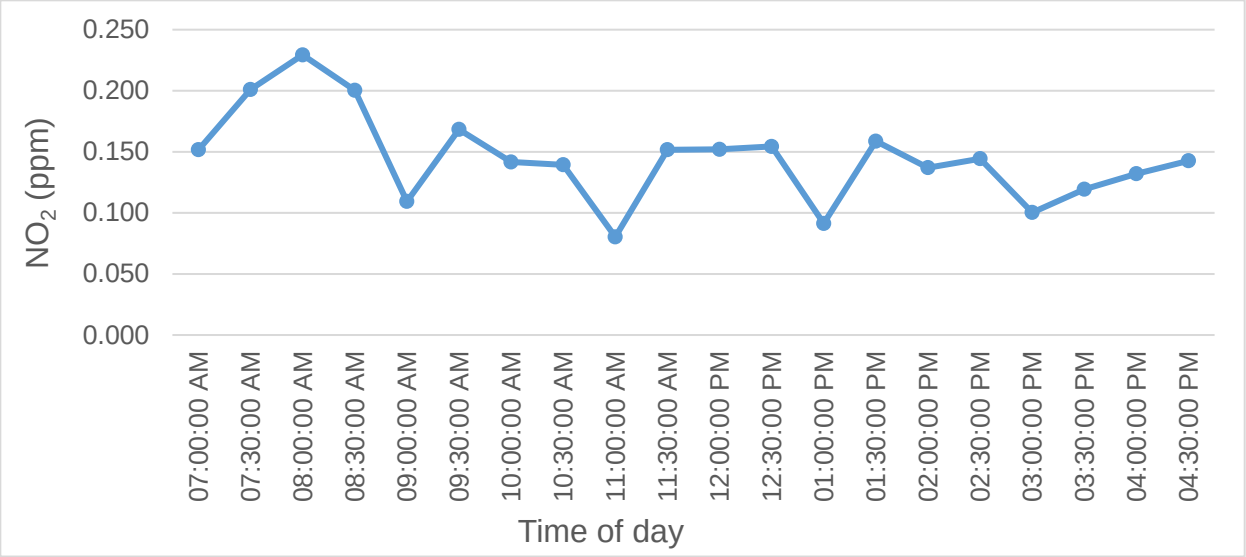

Figure 5.15: Diurnal nitrogen dioxide concentration variations during the entire sampling period.

5.3.4. Carbon monoxide

In the PEH building, the highest indoor CO concentrations were recorded during spring and the lowest in autumn, whereas outdoor CO concentrations were the highest in spring and the lowest in winter (Figure 5.16). The seasonal patterns for indoor CO concentrations across BMBT and BIOD buildings were similar, with both peaking during autumn and declining to a minimum in summer (Figure 5.16). Insignificant differences were found in indoor seasonal CO concentrations in the PEH building ($p = 0.151$).

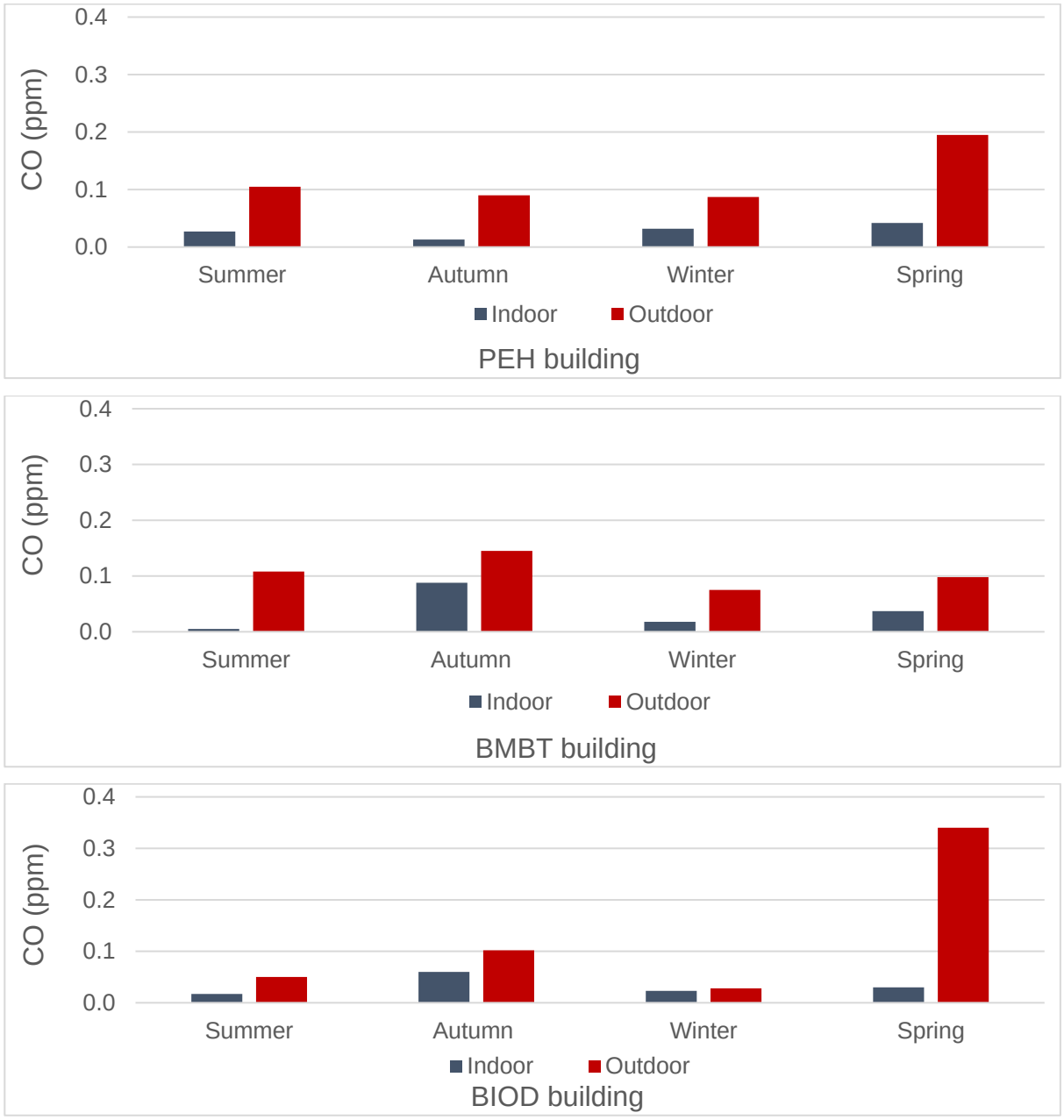

Figure 5.16: Seasonal variations of carbon monoxide for all three buildings.

In the BMBT building, indoor CO concentrations were found to be significantly different between the summer and the autumn and spring seasons ($p < 0.008$). Similarly, autumn indoor CO concentrations were significantly lower than concentrations in spring and winter across the BMBT building ($p < 0.008$). Indoor zero values for CO concentrations may be attributed to no inherent source of CO being present in all three buildings, especially during times of insufficient natural ventilation provisioning; hence rendering values below the detection limit of the instrument (Figure 5.17). Insignificant seasonal differences in indoor CO were found for the BIOD building ($p > 0.008$). The outdoor seasonal patterns

for CO concentrations at the BMBT building reflected autumn highs and winter lows, whereas in both the PEH and BIOD buildings, outdoor CO concentrations were highest during spring and lowest during winter (Figure 5.16). Outdoor CO concentrations in both the PEH and BMBT buildings had insignificant differences between seasons ($p = 0.254$ and $p = 0.072$). Outdoor CO concentrations of the BIOD building in summer, autumn and winter were significantly different from those for the spring season ($p < 0.008$). Both the indoor and outdoor CO concentrations during all four seasons conformed to the ASHRAE guideline (2010) of 9.00 ppm and the outdoor DEA standard (2009) of 8.70 ppm (Figures 5.17 and 5.18 respectively). Similarly, the diurnal CO variations across the entire sampling period were also found to be compliant to the DEA standard (2009) of 8.70 ppm (Figure 5.19). In this study, outdoor CO concentrations fluctuated throughout the day, due to several inherent sources of CO being present across the sampling sites of the study area at different times of the day (Figure 5.19). The peaks may be attributed to periods of low LAV, high temperature, vehicular and generator emissions in the immediate surrounds of the sampling areas and the lows may be attributed to times characterised by high LAV, low temperature and no inherent sources of CO in the immediate sampling surrounds (Figure 5.19).

Most studies conducted (Masiol *et al*., 2017; Zhao *et al*., 2016; Chen *et al*., 2015) support the notion that outdoor CO concentrations tend to be elevated during winter which is contradictory to the findings of this study. High winter CO concentrations encountered in literature were principally attributed to the burning of fossil fuels for heating purposes, biomass burning initiatives, higher traffic volumes (winter favours the use of vehicles), and the lower combustion efficiencies of engines, gasoline and other fuels (Masiol *et al*., 2017; Zhao *et al*., 2016; Chen *et al*., 2015). The high outdoor CO concentration in spring encountered in this study is therefore not a common occurrence. However, Lourens *et al*. (2011) had found CO concentrations to peak during spring across the Highveld in South Africa and they attributed the peaks to the regional increase in biomass burning (veld fires) events during late winter and the onset of spring. Incidences of biomass burning in the Mankweng area could have affected the outdoor CO concentrations resulting in the higher concentrations observed in spring. The low and in some cases zero outdoor CO concentrations measured during summer and other seasons in this study could be due to the photochemical production of O_3 from CO which occurs during periods of intense solar

radiation and elevated temperatures as depicted in equations 5.6–5.10 below (Kondratyev and Varotsos, 2001, Lu and Khalil, 1993; NRC, 1991):

$CO + OH^- \rightarrow CO_2 + H^+$ (Equation 5.6);

$H^+ + O2 \rightarrow HO_2$ (Equation 5.7);

$HO_2 + NO \rightarrow OH + NO_2$ (Equation 5.8);

$NO_2 + \hbar\upsilon^* \rightarrow NO + O$ (Equation 5.9);

$O + O_2 \rightarrow O_3$ (Equation 5.10)

* $\hbar\upsilon$: photon of light energy.

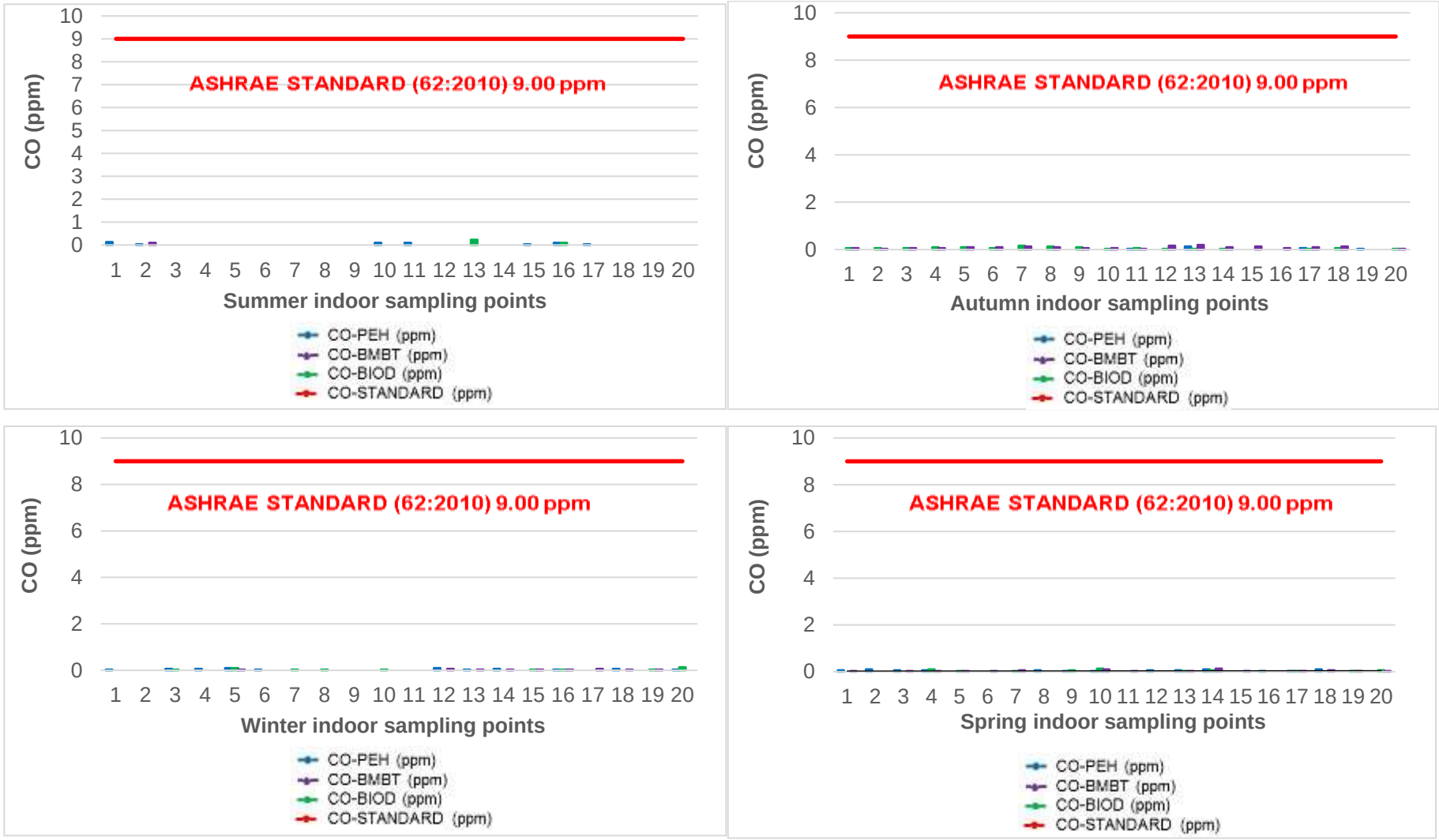

Figure 5.17: Indoor carbon monoxide concentrations for all seasons across the three departments in comparison to the ASHRAE standard of 2010.

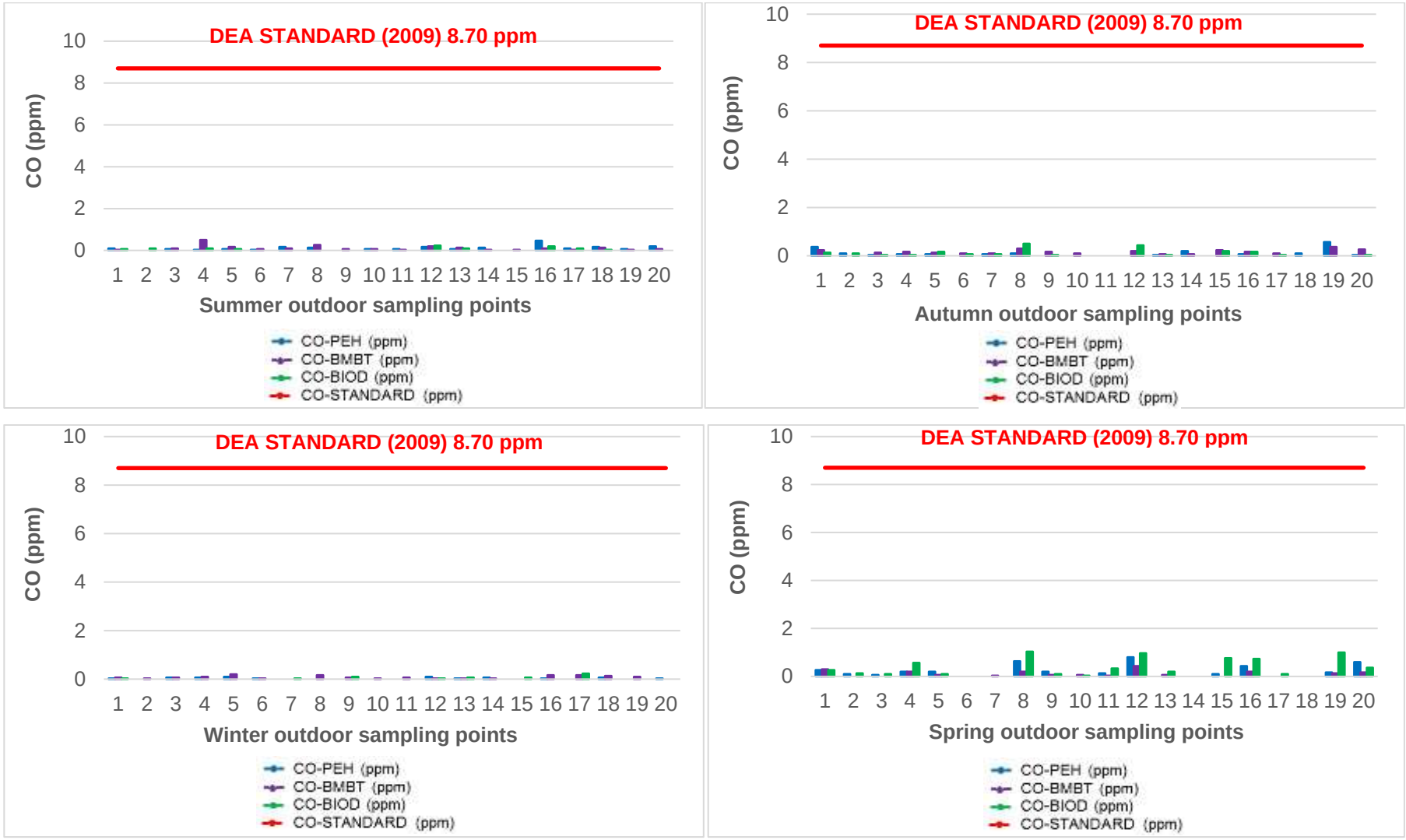

Figure 5.18: Outdoor carbon monoxide concentrations for all seasons across the three departments in comparison to the DEA standard of 2009.

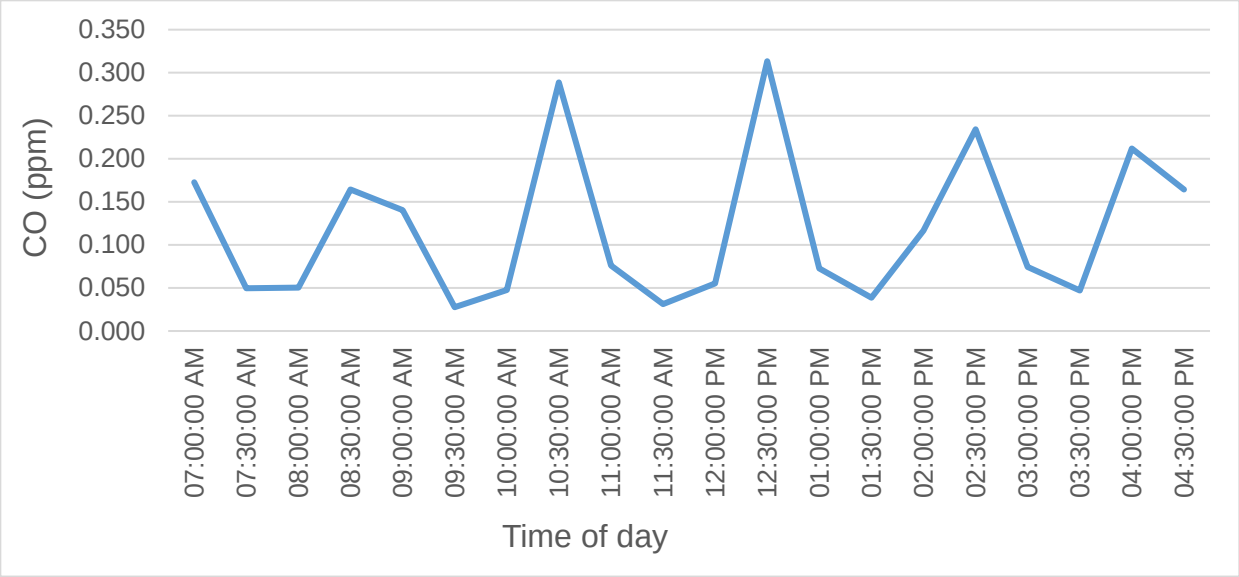

Figure 5.19: Diurnal carbon monoxide concentration variations during the entire sampling period.

5.3.5. Carbon dioxide

In the PEH building, the highest indoor CO_2 concentrations were recorded during summer (702.800 ± 37.408 ppm) and the lowest in spring (539.600 ± 13.054 ppm). Indoor CO_2 concentrations in the BMBT and BIOD buildings were found to peak during autumn and decline to minimum concentrations in spring (Figure 5.20). A different seasonal trend was observed by Gao *et al*. (2014) and Mentese *et al*. (2012) who both found winter maxima and summer minima for CO_2 concentrations within schools and urban environments. Differences in indoor CO_2 concentrations between the summer and spring, autumn and spring and winter and spring seasons in the PEH and BIOD buildings were significant ($p < 0.008$). For the indoor CO_2 concentrations within the BMBT building, significant differences were found between the autumn and winter and the autumn and spring seasons ($p < 0.008$). During autumn and winter which are the coldest months, occupants tend to switch on heaters and set air conditioners at higher temperatures, in addition to primarily staying indoors. Furthermore, in autumn and winter, natural ventilation usage is minimal across the three buildings due to colder outdoor temperatures. Increases in LAV during summer months tend to reduce pollutant concentration through dispersion, dilution and long-range transport of the contaminant. The higher CO_2 concentrations encountered in the winter months compared to the summer months may be attributed to the burning of different types of fossil fuels and poor ventilation rates commonly seen across colder months. It could be that the prevalent occupant densities within the indoor setting had a

confounding effect on the indoor CO_2 concentrations, rather than any seasonal differences. Outdoor CO_2 concentrations for both the PEH and BMBT buildings were highest in winter and lowest in summer, whereas in the BIOD building, the highest outdoor CO_2 concentrations were also recorded during winter (423.500 ± 1.585 ppm) but the lowest concentrations occurred during spring (393.800 ± 7.738 ppm) (Figure 5.20).

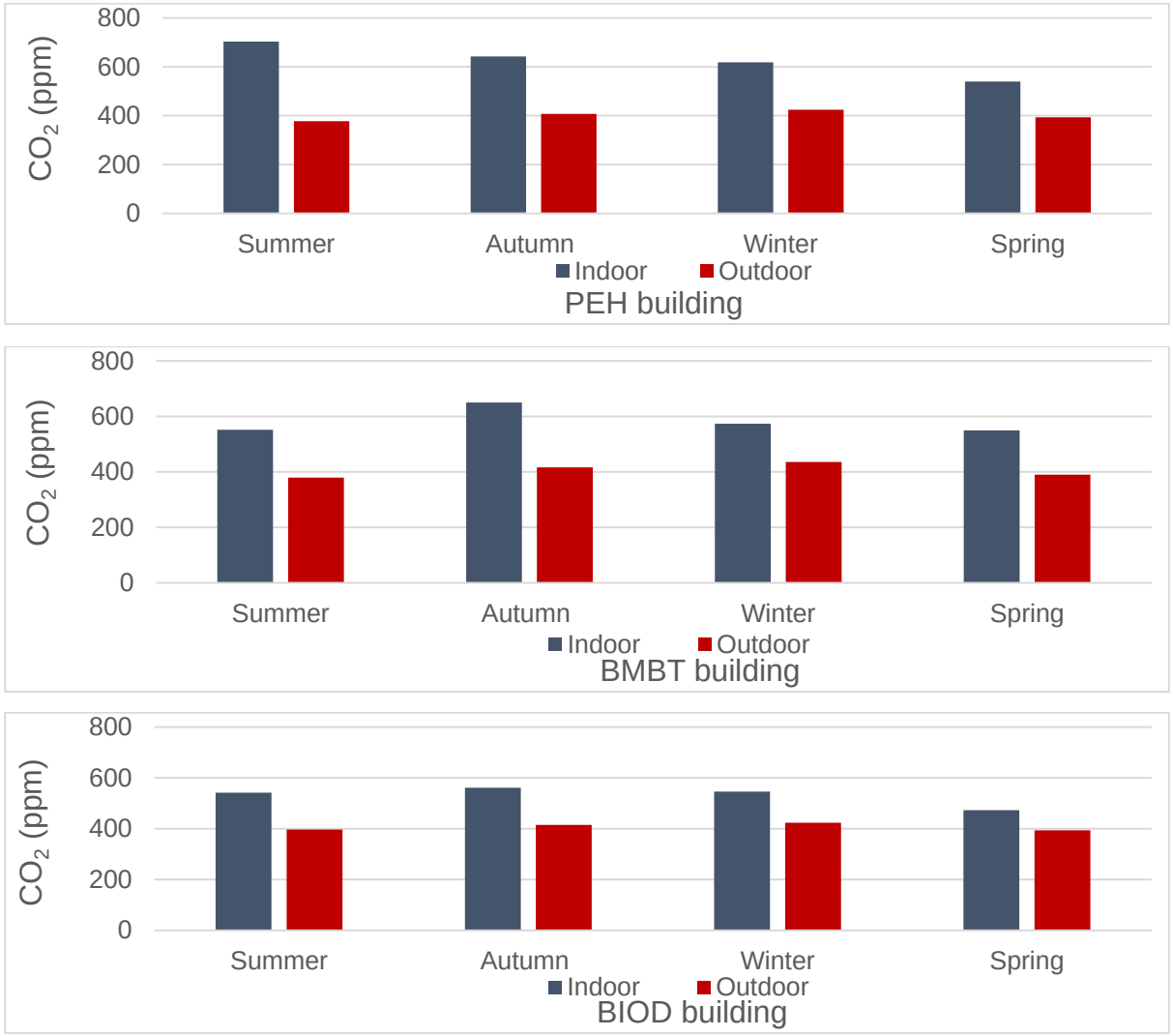

Figure: 5.20: Seasonal variations of carbon dioxide for all three buildings.

Concerning the outdoor PEH building CO_2 concentrations, significant differences were found between summer and autumn and winter seasons, in addition to the winter and the autumn and spring seasons ($p < 0.008$). Outdoor BMBT building CO_2 concentrations between the summer and autumn, summer and winter, autumn and winter, autumn and spring and the winter and spring seasons were also significantly different ($p < 0.008$). In the BIOD building outdoor CO_2 concentrations significantly differed between the summer and winter, autumn and spring and the winter and spring seasons ($p < 0.008$). Changes

in temperature, LAV and RH were not found to influence the outdoor CO_2 concentrations due to the inherent outdoor CO_2 sources present. Indoor CO_2 concentrations across all four seasons and all three buildings were within the limits set by the ASHRAE guideline of 2010 (Figure 5.21). However, elevated indoor CO_2 concentrations were encountered occasionally across all four seasons. These peaks were as a result of areas characterised by high occupant densities, in addition to areas characterised by diminished and absent airflow.

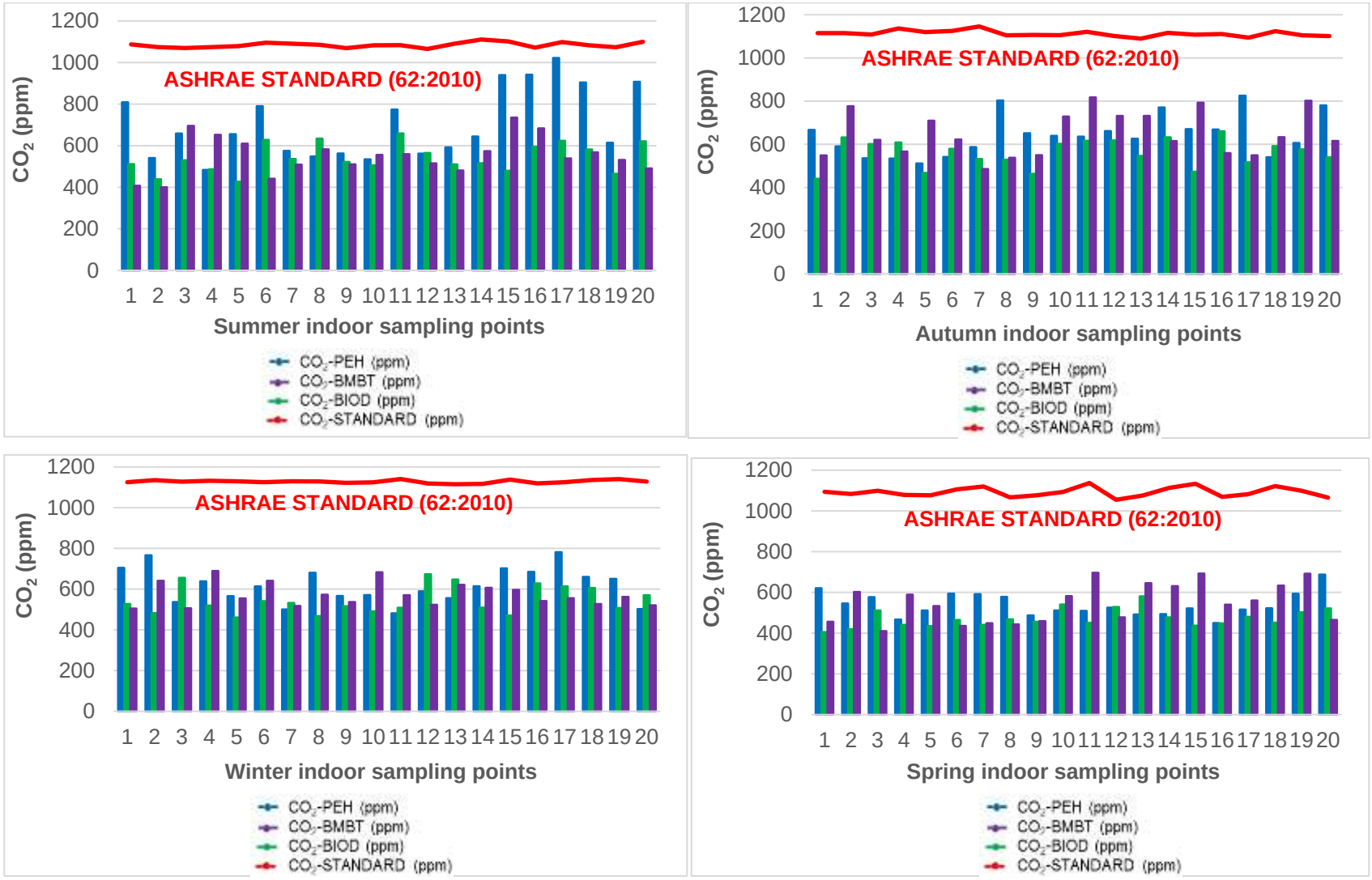

Figure 5.21: Indoor carbon dioxide concentrations for all seasons across the three departments in comparison to the ASHRAE standard of 2010.

5.3.6. Hydrogen sulphide

Indoor and outdoor H_2S concentrations across all three buildings fluctuated during the various seasons (Figure 5.22). Within both the PEH and BMBT buildings, the highest indoor H_2S concentrations were recorded during summer and the lowest concentrations during winter and spring (Figure 5.22). Indoor H_2S concentrations in the BIOD building were highest during spring (0.019 ± 0.005 ppm) and lowest during autumn (0.013 ± 0.003 ppm). The PEH building's summer indoor H_2S concentrations were significantly higher compared to the other three seasons ($p < 0.008$). Summer indoor H_2S concentrations across the BMBT building were also higher than winter concentrations ($p < 0.008$). Differences in seasonal indoor H_2S concentrations in the BIOD building were insignificant ($p = 0.345$).

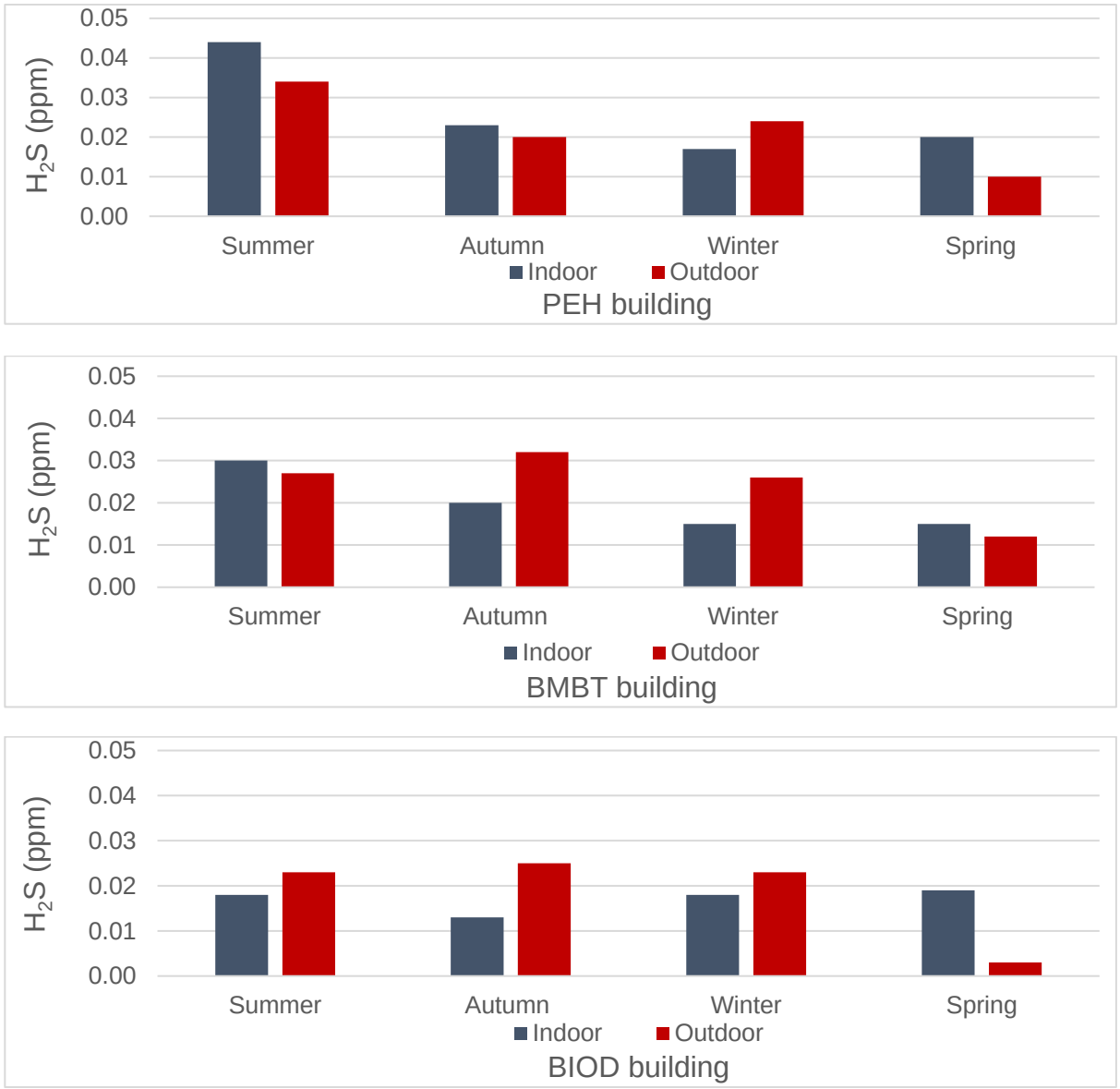

Figure 5.22: Seasonal variations of hydrogen sulphide for all three buildings.

A similar outdoor seasonal pattern of H_2S concentration was observed in the PEH building where the highest values were recorded in summer (0.034 ± 0.005 ppm) and the lowest in spring (0.010 ± 0.004 ppm). The BMBT and BIOD buildings, however, showed a different trend with outdoor autumn highs and spring lows (Figure 5.22). Significant differences in outdoor H_2S concentrations were found between the summer and spring and between the winter and spring seasons around the PEH building ($p < 0.008$). Differences in outdoor H_2S concentrations between the autumn and spring and the winter and spring seasons in the BMBT building were also significant ($p < 0.008$). In the BIOD building, outdoor H_2S concentrations in spring were significantly lower than what was obtained in the other three seasons ($p < 0.008$). Generally, rainfall leads to the washout of pollutants from the atmosphere and in the case of H_2S, the formation of acid rain or sulphuric acid could also occur (Cheptum, 2015; Pineda, 2007). The summer highs indicated across the PEH building may be as a result of an internal source of H_2S being present. A possible explanation for the zero H_2S values may be due to washout encountered during periods of rainfall, throughout all four seasons. Moreover, the prevalent meteorological conditions such as high LAV values may have led to the long-range transport of H_2S to other sites; consequently, rendering zero H_2S values across the areas of interest. Thus, readings below the detection limit of the instrument were commonly encountered. Hydrogen sulphide concentrations across all four seasons and all three buildings were within the WHO standard of 2000 (Figure 5.23). In contrast, when comparing the diurnal H_2S concentration variations across the entire sampling period with the WHO standard (2000) of 0.100 ppm, non-compliance was seen in the early morning hours between 7h30 and 8h30 (Figure 5.24). These elevated concentrations of H_2S are validated due to the early morning surface inversion layers and the corresponding low LAV readings during these specific time periods of sampling; subsequently, leading to pollutant accumulation (Figure 5.24). Furthermore, H_2S could be the by-product of certain organic reactions in the laboratories during pre-practical preparation sessions which took place between 7h30 and 8h30 across all laboratories surrounding the sampling areas (Figure 5.24).

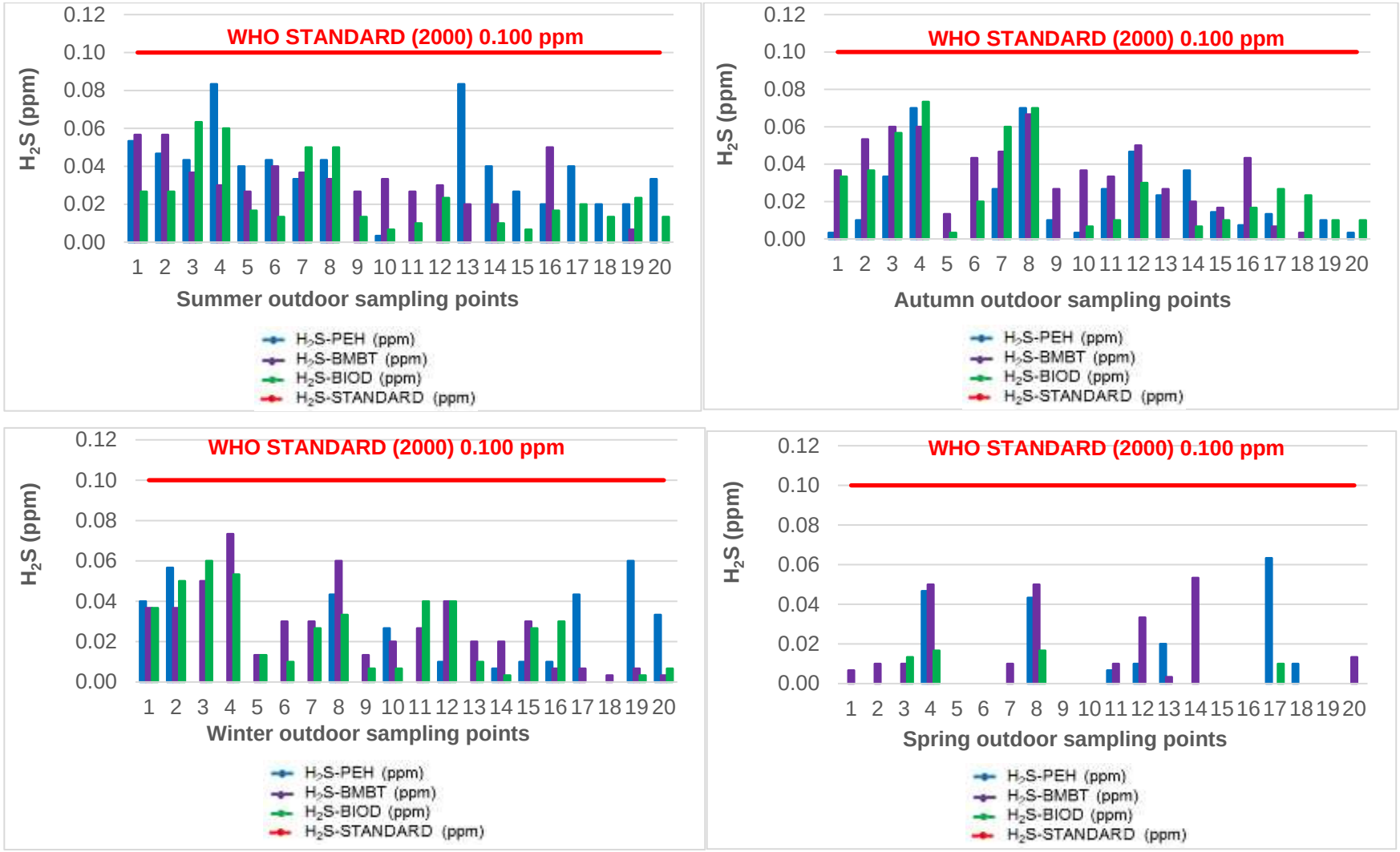

Figure 5.23: Outdoor hydrogen sulphide concentrations for all seasons across the three departments across the 20 sampling points in comparison to the WHO standard of 2000.

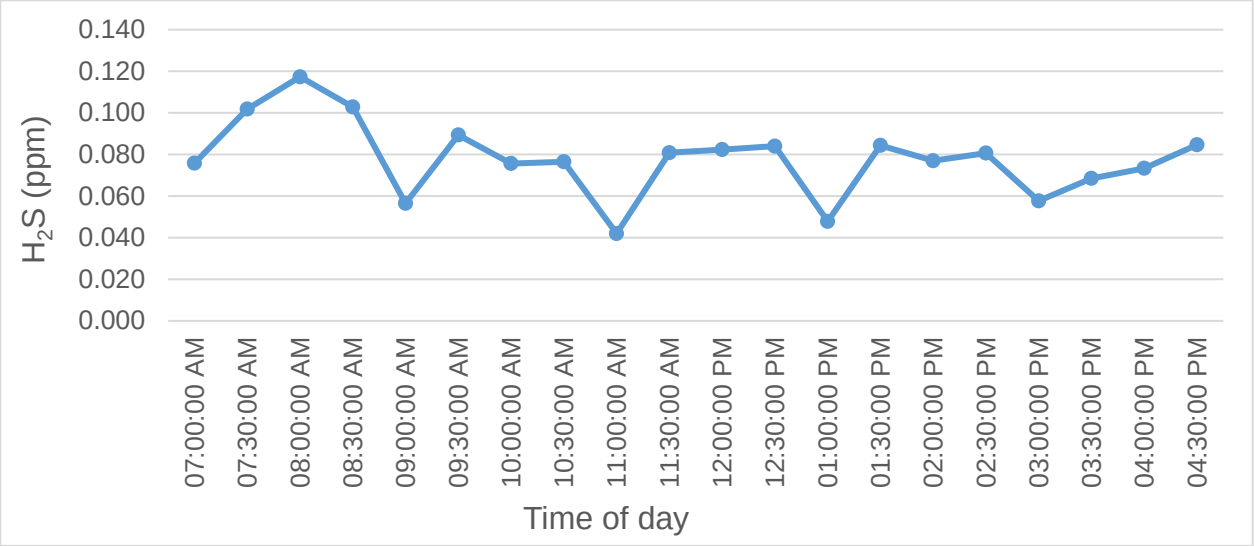

Figure 5.24: Diurnal hydrogen sulphide concentration variations during the entire sampling period.

5.3.7. Non-methane hydrocarbons

Indoor NMHC concentrations across all three buildings were higher than outdoor concentrations during all but the autumn season in the PEH building (Figure 5.25). In the PEH building, the highest indoor NMHC concentrations were recorded during spring (0.410 ± 0.034 ppm) and the lowest in summer (0.070 ± 0.007 ppm). The seasonal pattern for indoor NMHC concentrations in the BMBT and BIOD buildings were found to peak in spring and decline in winter (Figure 5.25). Except for differences in indoor NMHC concentrations between summer and all three seasons in the BMBT building and summer and autumn in the BIOD building ($p > 0.008$), all other seasonal differences were significant ($p < 0.008$). Outdoor NMHC concentrations in the PEH and BMBT buildings had similar trends with peak concentrations encountered in spring and the lowermost concentrations seen during summer. Outdoor seasonal NMHC concentrations in the BIOD building were highest during spring (0.680 ± 0.099 ppm) and lowest during winter (0.090 ± 0.015 ppm) (Figure 5.25). Significant seasonal differences in outdoor NMHC concentrations were observed between all seasons across buildings ($p < 0.008$). Exceptions in outdoor NMHC concentrations were encountered between the summer and autumn, summer and winter and autumn and winter seasons in the BMBT building and between the summer and autumn seasons in the BIOD building ($p > 0.008$).

An important characteristic in determining the seasonal variation of indoor NMHC concentrations is source emission strength variations (Li and Wang, 2012). The summer

months coincide with the end of the academic year when all classes and practicals cease. All the chemicals and reagents used for practicals purposes have been used up and in most cases, completely depleted. From the period of around March onwards, chemicals and reagents are replenished and practicals are on-going through to November, thereby rendering elevated concentrations as depicted in Figure 5.25 below. Normally, outdoor NMHC concentrations peak during autumn and winter and are found at minimal concentrations during summer and spring because of seasonal wind changes and differences in meteorological conditions (Kumar *et al*., 2017; Jaimes-Palomera *et al*., 2016; Kramer *et al*., 2015; Li and Wang, 2012). In the current study, a contradictory finding is seen compared to those documented in the literature. Outdoor NMHC concentrations peak during spring across all three buildings. The seasonal variations of NMHC concentrations are generally influenced by the prevalent meteorological conditions (Li and Wang, 2012) and as such, the seasonal variations are expected. Standards usually exist for individual NMHCs and not for them collectively. Currently, there are no indoor or outdoor standards for NMHCs and as such, no comparisons with standards have been made for NMHC concentrations for the four seasons.

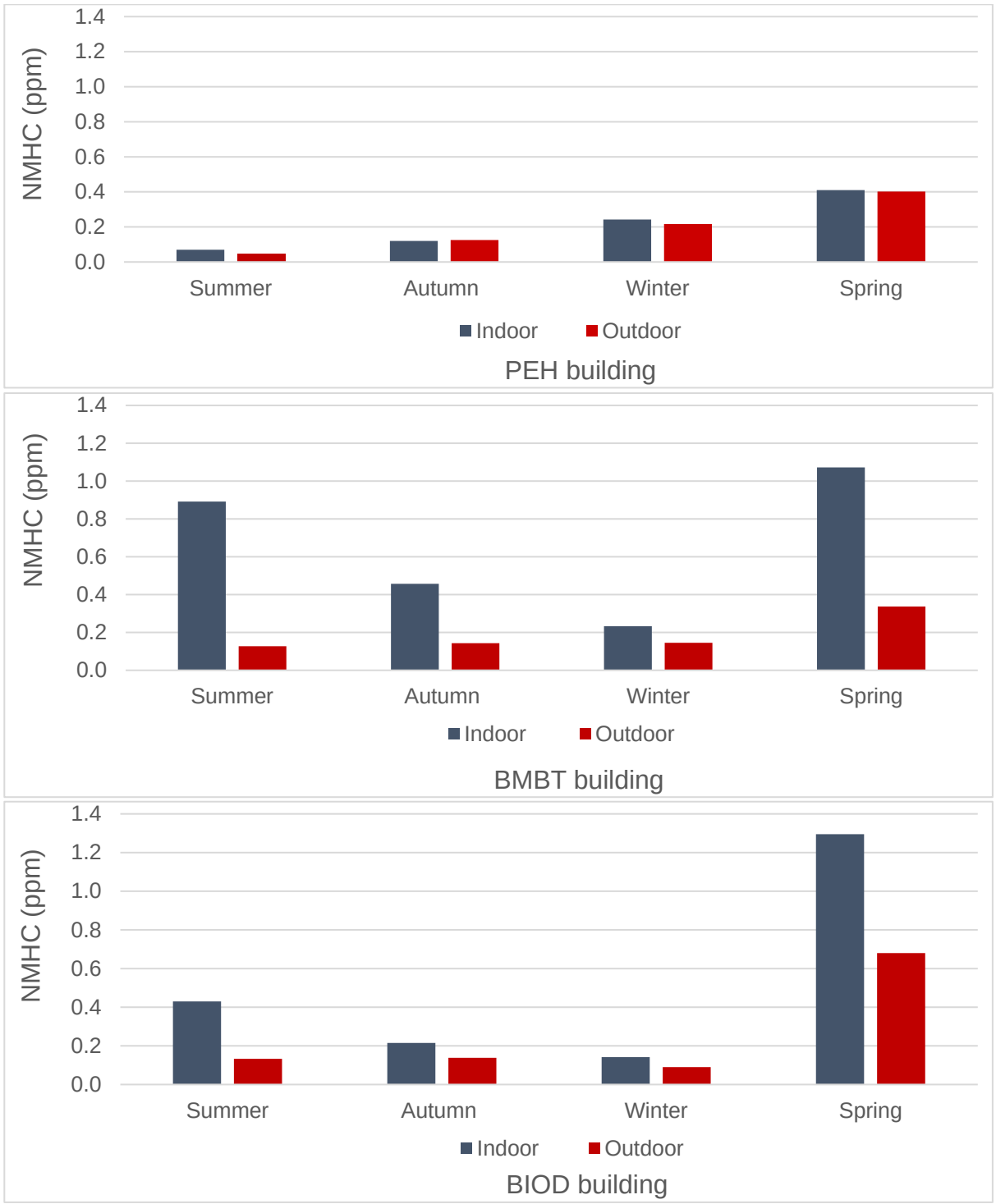

Figure 5.25: Seasonal variations of non-methane hydrocarbons for all three buildings.

5.3.8. Volatile organic compounds

Indoor VOC concentrations within all three buildings were highest during summer and lowest in winter (Figure 5.26). Contrastingly, outdoor VOC concentrations in the PEH building were highest during both summer and autumn seasons and lowest during winter.

The outdoor seasonal patterns for VOC concentrations in the BMBT building reflected a similar inclination for summer peaks and winter lows. Outdoor seasonal VOC concentrations in the BIOD building were highest during autumn (0.025 ± 0.004 ppm) and lowest during winter.

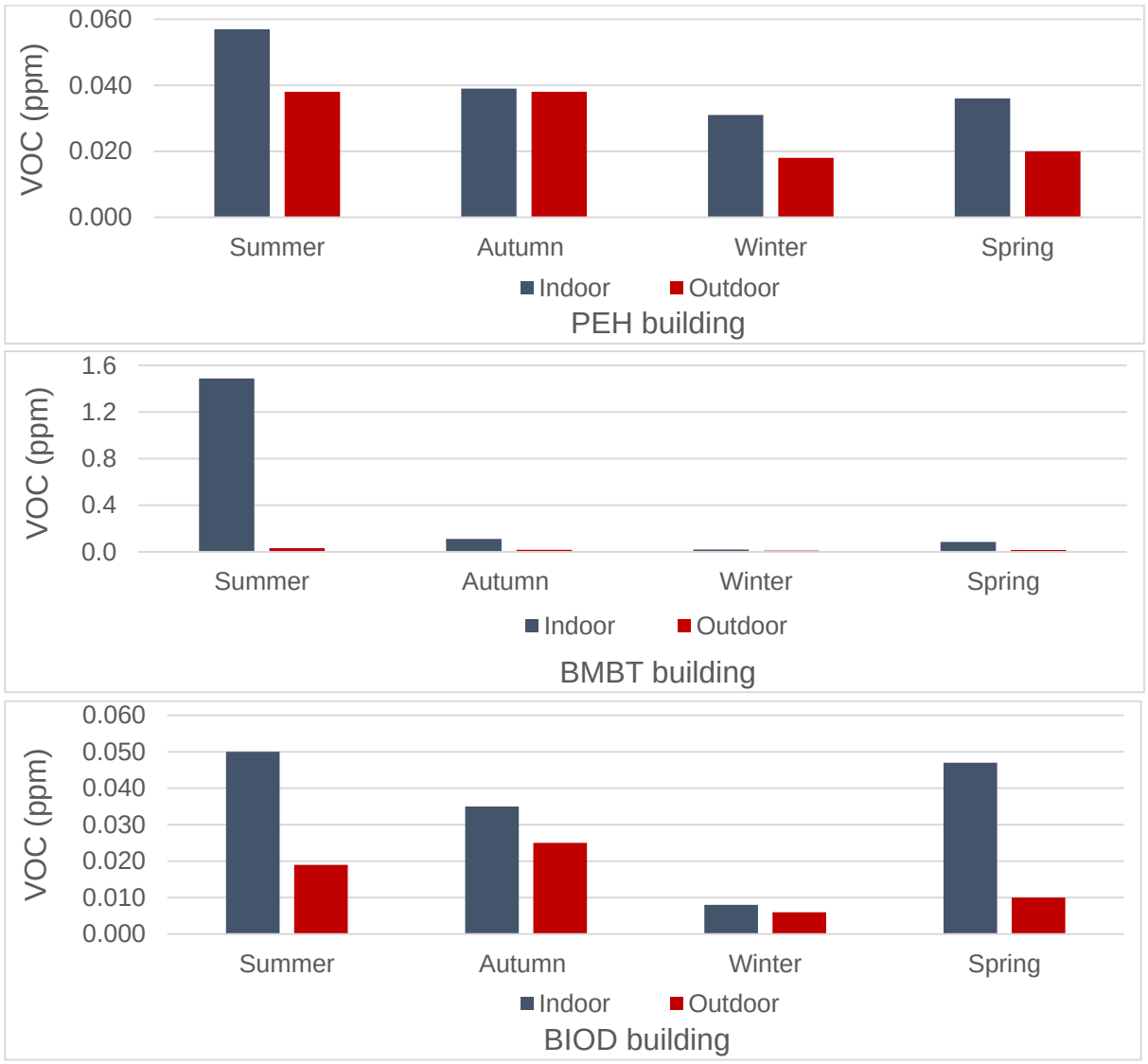

Figure 5.26: Seasonal variations of volatile organic compounds for all three buildings.

Jiang *et al*. (2013) found total VOC concentrations to peak in summer months, followed by winter then spring months. In disagreement with the current study findings, Pekey and Arslanbaș (2008) and Schlink *et al*. (2004) both found winter maxima and summer minima VOC concentrations. Sexton *et al*. (2004) on the other hand found no seasonal differences in VOC concentrations. Hence, the findings of literature tend to illustrate that the seasonal variations of VOCs are not clear-cut. Seasonal differences of VOC concentrations across

the indoor environment of the PEH building were significant between the summer and autumn, and winter and spring seasons ($p < 0.008$). Across both BMBT and BIOD buildings, indoor seasonal differences were significant between the winter and summer and autumn and spring seasons ($p < 0.008$). The summer maxima encountered within the BMBT building are due to the high temperatures prevalent during the summer months. Because these compounds are highly volatile, high temperatures tend to increase their concentrations, especially in a closed environment. In addition, indoor VOC concentrations can accumulate as a result of low ventilation and air exchange rates when doors and windows are closed (Jia *et al*., 2008a; Pekey and Arslanbaș, 2008). On the other hand, the zero readings encountered across the outside environment of BMBT and both the inside and outside environment of BIOD during winter may be attributed to the lower temperatures encountered during this season (Figure 5.26). Volatile organic compounds tend to increase in concentration with corresponding increases in temperature (Jiang *et al*., 2013).

Regarding outdoor VOC concentrations in the PEH, BMBT, and BIOD buildings, significant differences were found between the summer and winter, summer and spring, autumn and winter and the autumn and spring seasons ($p < 0.008$), in addition to the BMBT building, also having significant differences between the summer and autumn seasons ($p < 0.008$). The summer periods were characterised by elevated VOC concentrations in the outdoor environment, along with the release of fumes emanating from the chemicals within the laboratories towards the outside environment. As was the case with NMHCs, VOCs also do not have any indoor or outdoor standards (standards exist for individual VOCs and total VOCs). Hence, no comparisons with standards have been made for VOC concentrations over the four seasons.

5.4. Summary

With reference to the indoor comfort parameters, seasonal differences across all three buildings can be attributed to innate differences between individuals and their thermal comfort preferences. Parameters such as RH and T_{db} are very difficult to control in an indoor environment (Kalimeri *et al*., 2016). The outdoor comfort parameters were mainly affected by the prevailing meteorological conditions per season, consistent with the findings of Chaudhuri and Dutta (2014).

Significant differences in AQ parameters are seen in both the indoor and outdoor environments between seasons in all three buildings. Across the indoor environment, the winter season was found to be the seasons with the best AQ since all of the pollutants were found at minimum concentrations. The best AQ across the outdoor environment occurred during the autumn season due to all of the air pollutants being present at minimal concentrations. The prevalent meteorological conditions, burning of fossil fuel in nearby environments for purposes of heating, biomass burning initiatives, occupant densities, provisioning of natural ventilation, and chemicals and reagents used during practicals are some of the factors which influenced the seasonal differences observed.

CHAPTER 6
EFFECT OF ENVIRONMENT, BUILDING AND SEASON ON AIR QUALITY

6.1. Introduction

This chapter presents a discussion on the best predictors of AQ in the university environment. To determine how each of the AQ parameters affected AQ, a multiple regression model was run using independent (predictor) variables against a dependent variable. The dependent variables were each AQ parameter's annual average whereas the independent variables for each of the parameters were the environments (indoor and outdoor), the three buildings (PEH, BMBT, and BIOD) and lastly the four seasons (summer, autumn, winter, and spring). The results of the regression model are discussed below. This chapter proceeds to formulate an AQI within the study area for the selected pollutants.

6.2. Multiple regression model inclusion and exclusion criteria

The multiple regression analysis was run using the annual means of each of the different AQ parameters determined in this study (Appendix A2, Tables A2.5 to 2.7). From the model summary, it was evident that the selected independent variables (indoor and outdoor environments, the PEH, BMBT and BIOD buildings and the summer, autumn, winter and spring seasons) had extremely strong positive correlations with the dependent variable (average AQ per variable, i.e. LAV, T_{db}, RH, O_3, SO_2, NO_2, CO, CO_2, H_2S, and NMHCs). However, some independent variables had only partial correlations with the dependent variables. All predictor variables that had an R^2 value of less than 0.996 with the dependent variable were deemed to be variables that have partial correlations. Subsequently, those predictor variables with partial correlation coefficients were excluded from the ANOVA and multiple regression analysis. It is critical to exclude these variables with partial correlations, to exclude the confounding effect of these variables on the dependent variables. This process of exclusion increases the validity and precision of the predictor variables. The following variables were excluded from the multiple regression model:

- In the computation of the influence of environment (indoor and outdoor) on the AQ of the study area, the following parameters were excluded: outdoor LAV (R^2 = 0.029), outdoor RH (R^2 = 0.555), outdoor T_{db} (R^2 = 0.079), outdoor NO_2 (R^2 = 0.483), outdoor

CO (R^2 = 0.679), indoor CO_2 (R^2 = 0.777), indoor NMHC (R^2 = 0.397) and indoor VOC (R^2 = 0.367).

- To determine the role that buildings played in the AQ of the study area, the following parameters were excluded: LAV in the BIOD building (R^2 = 0.099); T_{db} (R^2 = 0.435), NO_2 (R^2 = 0.053) and NMHCs (R^2 = 0.236) for the BMBT building; CO in the BIOD building (R^2 = 0.165) and CO_2 in the PEH building (R^2 = 0.421).
- The only variable excluded from the model for seasonal influence was summer VOCs (R^2 = 0.225).

6.3. Effect of environment

The indoor and outdoor environments were found not to affect the variables as depicted in Table 6.1. Sulphur dioxide was found to be affected by the indoor and outdoor environment, with approximately 99.9 % of SO_2 concentration in the study area explained by the environment ($p < 0.001$, R^2 = 0.999) (see Table 6.2). Indoor environment laboratories, vehicular and generator emissions across the outdoor environment seemed to have the greatest influence on SO_2 contributions in the university environments. Neither building structure nor function, nor seasonal conditions influenced the annual SO_2 averages.

Table 6.1: P-values of variables that were not affected by the environment variable.

Environment	P-values								
	LAV	T_{db}	RH	O_3	NO_2	CO	CO_2	H_2S	NMHCs
Indoor	0.698	0.956	0.543	0.630	0.955	0.068	-	0.701	-
Outdoor	-	-	-	0.363	-	-	0.325	0.698	0.544

Table 6.2: Percentage variation of air quality parameter explained by the environment.

Air quality parameter	R^2	p-value	Percentage variation accounted for by environment
Sulphur dioxide	0.999	0.001	99.9 %
Carbon monoxide	0.679	0.011	67.9 %
Carbon dioxide	0.777	0.002	77.7 %

Although the outdoor environment for CO was initially excluded from the model on the basis of moderate to strong positive correlation ($R^2 = 0.679$; $p < 0.05$), it was found to be a good predictor of the CO concentrations in the study area (Table 6.2). Approximately 67.9 % of the CO concentration in the campus was explained by the environment, especially the outdoor environment. Similar to SO_2, CO precursors such as emissions from motor vehicles, functional generators and surrounding industries all contributed towards the values of CO obtained in this study environment. The indoor environment for CO_2 was also excluded from the model on the basis of a moderate to strong positive correlation ($R^2 = 0.777$; $p < 0.05$). However, it was also found to be a good predictor of the CO_2 annual average (Table 6.2). Approximately 77.7 % of the CO_2 concentration in the campus was explained by the indoor environment. Occupant densities would have played the greatest role in the CO_2 concentrations in the university environment. Details of the regression analyses can be found in Appendix A2, Tables A2.5 to 2.7. Indoor and outdoor sources of SO_2, CO and CO_2 therefore play a significant role in the amount of these air pollutants contained in the university environment. From the analyses and literature available, vehicular and generator emissions, occupant densities and laboratories may be the major sources of these gases in a university environment.

6.4. Effect of building

The building was found not to have any effect on the variables presented in Table 6.3. The remaining three AQ parameters studied namely RH ($R^2 = 1.00$), H_2S ($R^2 = 1.00$) and VOCs ($R^2 = 1.00$) were found to be affected by the buildings (Table 6.4). Anthropogenic activities play a significant role in the introduction of H_2S and VOCs in the environment. Though RH is influenced by prevailing rainfall patterns, its context in an environment can be manipulated by man, based on personal preferences. This may explain why building functions and structures play a significant role in their presence in the university environment.

Table 6.3: P-values of variables which were not affected by building variable.

Building	p-values							
	LAV	T_{db}	O_3	SO_2	NO_2	CO	CO_2	NMHCs
PEH	0.801	0.767	0.314	0.628	0.941	0.269	-	0.906
BMBT	0.586	-	0.137	0.671	-	0.921	0.079	-
BIOD	-	0.548	0.178	0.523	0.980	-	0.061	0.619

Table 6.4: Percentage variation of air quality parameter explained by the building.

Air quality parameter	R^2	p-value	Percentage variation accounted for by environment
Relative humidity	1.000	0.001	100 %
Hydrogen sulphide	1.000	0.001	100
Volatile organic compounds	1.000	0.001	100 %

Potential sources of RH in various buildings include occupant densities, usage of kettles and water baths indoors, renovation activities and walling material. The differences in building function and structure also accounted for 100 % of the variation observed in H_2S concentration in this study ($p < 0.05$). With regards to H_2S, buildings used for laboratory purposes are more likely to have higher concentrations of H_2S. Volatile organic compounds were also significantly influenced by building type with 100% of the variation observed in VOC concentration explained by the building type ($p < 0.001$). Buildings harbouring laboratories in addition to those with offices and stores where solvents and reagents are kept are most likely to have higher concentrations of VOCs. Lack of ventilation in buildings could also result in elevated concentrations of VOCs as observed in the PEH building. This highlights the role of building function in the distribution of pollutants in a university environment. Air quality in a university environment is rarely monitored when the university is in operation. The results of this study has highlighted the potential of health threats associated with poor AQ in a university environment, especially in buildings where laboratories are located.

6.5. Effect of seasons

Seasonal changes were found to be have little to no effect on the AQ parameters in Table 6.5 below.

Table 6.5: P-values of variables that were not affected by season variable.

Season	p-values				
	RH	O_3	SO_2	H_2S	VOCs
Summer	0.567	0.466	0.471	0.052	-
Autumn	0.614	0.275	0.445	0.063	0.515
Winter	0.618	0.424	0.590	0.068	0.840
Spring	0.559	0.306	0.274	0.061	0.498

Linear air velocity (R^2 = 1.00), T_{db} (R^2 = 1.00), NO_2 (R^2 = 1.00), CO (R^2 = 1.00), CO_2 (R^2 = 1.00) and NMHCs (R^2 = 1.00) were all affected by seasons (Table 6.6). Appendix A2, Tables A2.5 to 2.7 contain the detailed results for the regression analyses.

Table 6.6: Percentage variation of air quality parameter explained by the seasons.

Air quality parameter	**R^2**	**p-value**	**Percentage variation accounted for by environment**
Linear air velocity	1.000	0.001	100 %
Dry-bulb temperature	1.000	0.001	100 %
Nitrogen dioxide	1.000	0.001	100 %
Carbon monoxide	1.000	0.001	100 %
Carbon dioxide	1.000	0.001	100 %
Non-methane hydrocarbons	1.000	0.001	100 %

Meteorological conditions such as the prevalent wind speed and direction and amount of solar radiation play an important role in the LAV and temperature fluctuations, as air movements and outdoor temperatures vary per season. Consequently, these AQ parameters were influenced mostly by the prevalent season itself ($p < 0.001$) and not by environmental and building influences. The summer, autumn, winter, and spring season were significant to the predictors of the annual NO_2, CO, CO_2 and NMHCs averages in this study ($p < 0.001$). Weather patterns influence the movement and dispersion of pollutants in the atmosphere through the effect of winds, vertical mixing, and precipitation (Grinn-Gofroń *et al*., 2011). Nitrogen dioxide, CO, CO_2 and NMHCs are specifically affected by seasonal influences, due to these pollutants being sensitive to temperature (solar radiation) and RH (precipitation) changes. Generally increase in temperature favour the production of these pollutants with increases in RH leading to lower pollutant concentrations, due to washout and dissolution. Furthermore, seasonal fluctuations in prevalent wind speed and directions would lead to the dispersion and long-range transport of these pollutants.

Further to these, most of these pollutants are gaseous in nature. The behaviour of gases under different meteorological conditions is highly variable as explained by the various gas laws: Boyle's law, Charles Law, Gay-Lussa's Law, and Avogadro's Law. These laws explain the influence of and the relationships of pressure, volume, temperature and moles of gas on the behaviour of gases. Boyle's law relates to gases at a constant temperature,

whereby pressure is inversely proportional to the volume of a gas (Chandan and Cascella, 2019). As altitude increases, pressure decreases and according to Boyle's law, the volumes of these gases would expand in enclosed areas such as within buildings (Chandan and Cascella, 2019). The Charles law relates to a gas that is at constant pressure; the volume of the gas is directly proportional to the absolute temperature of the gas (Chandan and Cascella, 2019). With all the above-mentioned pollutants, an increase in their volume is seen with corresponding increases in temperature. Gay-Lussac's law refers to the pressure of gas being directly proportional to the absolute temperature, provided that the volume of the gas is constant (Chandan and Cascella, 2019). Lastly, Avogadro's law relates to the volume of gas being proportional to the number of moles and molar volume at the same temperature and pressure (Chandan and Cascella, 2019). The gaseous nature of these pollutants and the reactivity under different meteorological conditions play a significant role in their presence in any environment, including a university environment. Specific attention needs to be given to the prevalent environmental conditions within laboratories. The temperature, pressure, and airflow within these environments may influence their concentrations. Low LAV and elevated T_{db} values exacerbate the pollutant concentrations within a given environment and upon exposure to these pollutants for prolonged periods, respiratory irritations, pulmonary and nervous effects may result (Kumar *et al*., 2017; ATSDR, 2000).

6.6. Pollutant without predictor variables

It is evident that the environment, building and seasons do not individually affect O_3 concentrations at all ($p > 0.05$) but the environments, buildings and seasons in unison, however, seemed to have an effect on O_3 concentrations ($R^2 = 0.995$). Ozone is a naturally occurring unstable gas with a half-life that fluctuates in accordance to the current atmospheric conditions (LAV, T_{db} and RH). The presence of other air pollutants such as nitrogenous and sulphur compounds also influence the concentrations of O_3 in the atmosphere because of its highly oxidative nature. These atmospheric conditions, along with other pollutant concentrations in the atmosphere vary with environment, building and seasons, which may explain why these three variables interact to determine the level of O_3 concentration in any environment.

6.7. Air quality index of the University of Limpopo environment

An AQI or API is universally used to quantify the extent of air pollution in a given area. In South Africa, an API is commonly used (Cairncross *et al*., 2007). The South African API is centred around the relative risk of well-established daily mortality rates linked with acute exposure to contaminants such as O_3, SO_2, NO_2, CO and PM (Gorai and Goyal, 2015; Cairncross *et al*., 2007). The relative risk is a central estimate based on impact assessments of health due to exposure to air pollution (Cairncross *et al*., 2007). Since relative risks are not established in the current study area, it could not be used to determine the relative risk of all the parameters of interest in this study. Subsequently, the more recent USEPA (2015b) method for computing AQI was used since this method calculates AQI for each pollutant in connection with their preceding maximum concentration per averaging time (USEPA, 2015b). According to the USEPA (2015b) method for computing an AQI, there are five AQI categories, with each category having an index range as depicted in Table 6.7 below.

Table 6.7: Classification of air quality index categories with corresponding air quality index ranges (USEPA, 2015b).

Category	Index range
Very good air quality	0–33
Good air quality	34–66
Fair air quality	67–99
Poor air quality	100–149
Very poor air quality	≥ 150

For all the pollutants of interest in this study for which a standard existed in the outdoor environment, a corresponding API was calculated. Of the parameters of interest in this study, only O_3, SO_2, NO_2, CO and H_2S had standards in the outdoor environment. Both the maximum and the mean outdoor pollutant concentrations across the entire duration of the study were used to compute the API for each pollutant. The API for each pollutant is presented in Table 6.8. The current methodology in the calculation of the API as suggested by the USEPA (2015b) uses the maximum concentration which could be a value that occurs just once a year, thus the mean average concentrations of these pollutants gives one a clearer picture of the AQ in the study area and was also used in this study.

An API range of between 4.31 (CO) to 250 (NO_2) was obtained for the outdoor pollutants with standards, when the API was determined using the maximum concentrations. On the other hand, an API range of 0.837 (CO) to 111.321 (NO_2) was established when using mean concentrations for the calculation of APIs (Table 6.8). When using the maximum pollutant concentrations, the BIOD building had the highest API for SO_2 and NO_2, whilst the PEH building had the highest API for O_3, CO, and H_2S. The API lows for O_3, SO_2, NO_2, and CO were all found across BMBT, whilst the lowest API for H_2S was found in BIOD when using the maximum pollutant concentrations (Table 6.8). Ozone's API index when utilising the maximum concentrations for PEH fell within the fair AQ tier (Table 6.8) The BMBT building API corresponded to the very good AQ tier whereas the BIOD building fell within the good AQ tier. When using the mean concentrations for computing the APIs, the PEH building had the lowest NO_2 values, the BMBT building had the lowest O_3 and CO values, and the BIOD had the lowest SO_2 and H_2S values.

The highest API values obtained utilising both the maximum and mean concentrations in the determination of API were recorded for NO_2. Upon introspection of the NO_2 API indices determined by making use of the maximum concentrations for all three buildings, the indices surpassed the range of ≥ 150, consequently falling within the tier of very poor AQ. The APIs which made use of the mean concentrations reflected poor AQ across all three buildings as well. This is not unexpected given that NO_2 concentrations in this study frequently exceeded the DEA standard (2009) across all four seasons and all three buildings. Some of the health outcomes that arise from excessive NO_2 exposure are respiratory irritation, coughing, worsening of existing respiratory illnesses, nausea, bronchial hyper-responsiveness and decreased lung function in asthmatics and in extreme cases spasms, upper respiratory tract swelling, below par oxygenation in body tissues, pulmonary oedema and eventual death (Pénard-Morand and Annesi-Maesano, 2004; ATSDR, 2002).

With several exceedances of O_3 standard concentrations, the acute health effects that tend to arise include eye irritation, irritation of the respiratory tract, coughing, lung irritation and induced shortness of breath. Health complications such as bronchitis and pneumonia and an increase in asthma attacks are seen upon chronic exposure (Pénard-Morand and Annesi-Maesano, 2004).

Table 6.8: Pollutant concentrations, standards, and air pollution indices in the study area.

Pollutant	Maximum concentration (ppm)	Mean concentration (ppm)	Pollutant standard concentration (ppm)	API (using maximum concentrations)	API (using mean concentrations)
O_3 PEH	0.043	0.006	0.061 (DEA, 2009)	70.491	9.836
O_3 BMBT	0.020	0.004		32.686	6.557
O_3 BIOD	0.028	0.007		45.902	11.475
SO_2 PEH	0.069	0.014	0.134 (DEA, 2009)	58.955	10.448
SO_2 BMBT	0.068	0.011		50.646	8.209
SO_2 BIOD	0.084	0.006		62.686	4.478
NO_2 PEH	0.250	0.122	0.106 (DEA, 2009)	235.849	105.660
NO_2 BMBT	0.218	0.118		205.660	111.321
NO_2 BIOD	0.265	0.117		250.000	110.377
CO PEH	0.566	0.074	8.60 (DEA, 2009)	6.516	0.860
CO BMBT	0.365	0.072		4.310	0.837
CO BIOD	0.500	0.081		5.646	0.942
H_2S PEH	0.066	0.024	0.100 (WHO, 2000a)	66.000	24.000
H_2S BMBT	0.063	0.022		63.000	22.000
H_2S BIOD	0.053	0.018		53.000	18.000

With reference to O_3 concentrations, the exceedance of the DEA standard (2009) was seen just once during the entire period of sampling (during the summer season at the PEH building) and the building occupants did not manifest any of the above-mentioned health effects. However, regular O_3 monitoring initiatives should be undertaken to ensure O_3 concentrations fall within the very good AQ tier. The O_3 API index when using the mean concentrations all fell within the very good AQ tier across all three buildings.

In connection with the SO_2 API values when using the maximum pollutant concentrations, all three buildings fell within the tier of good AQ, whereas the mean concentrations yielded APIs which fell within the very good AQ tier. When the standard concentration of SO_2 is surpassed, the prominent acute health challenges that come to the fore are respiratory irritation (burning sensation of nose and throat) and coughing (Pénard-Morand and Annesi-Maesano, 2004; ATSDR, 1998). Chronic health effects include asthma attacks in asthmatics, airway obstructions and altered lung functioning in children (Pénard-Morand and Annesi-Maesano, 2004; ATSDR, 1998). As was the case with O_3, SO_2 concentrations throughout the sampling period had a single exceedance of the DEA standard (2009) (summer season in BIOD) and as such would not lead to any of the health effects described above.

With reference to the CO API indices, all three building indices fell within the tier of very good AQ when using both the maximum and mean concentrations of the pollutants. Upon exposure to elevated CO concentrations, CO tends to bind to the oxygen-carrying site on haemoglobin, thus reducing O_2 transport in the body (ATSDR, 2012). Consequently, health effects such as headaches, dizziness, weakness, nausea, vomiting, a reduced ability to think, confusion, disorientation, visual disturbances and in worst cases, coma, convulsions, cardiorespiratory arrest, and eventual death may occur (ATSDR, 2012; Pénard-Morand and Annesi-Maesano, 2004; Chaloulakou *et al*., 2003; Raub *et al*., 2000). All the way through the sampling periods, CO concentrations never exceeded the outdoor DEA standard (2009) and as such, none of the above-mentioned health effects would be expected to arise in the occupants across the study area.

The health effects that are associated with excess H_2S exposure are eye, nose and throat irritations, poor memory, balance problems, tiredness, breathing difficulties amongst asthmatics and in extreme concentrations for prolonged periods, loss of consciousness,

respiratory distress or arrest may occur (ATSDR, 2016). The H_2S API index while using maximum concentrations for PEH fell within the fair AQ tier whereas for both the BMBT and BIOD buildings they fell within the good AQ tier. In contrast, upon computation of the API using the pollutant's mean concentrations, very good AQ was the result across all three buildings. Hydrogen sulphide concentrations never exceeded the WHO standard (2000) across the entire sampling period and as such the above-mentioned health effects are not of concern to the current study population.

Although the other pollutants studied such as CO_2, NMHCs and VOCs do not have outdoor standards that can be used in the calculation of an API, there are health-related implications for their exposure at various concentrations across the indoor environment. In relation to CO_2, exposure to concentration ranges around 600 ppm tend to lead to symptoms such as headache, fatigue, eye, nose and throat-related symptoms (Erdmann *et al*., 2002). Concentrations of 1000 ppm and higher tend to lead to effects on respiratory health in children, cognitive performance effects, including decision making and problem resolution (Azuma *et al*., 2018). In the current study, only the PEH building had indoor concentrations in exceedance of 600 ppm (i.e. 625.6 ppm). Consequently, the PEH building occupants may have the possibility of manifesting the above-mentioned symptoms.

Acute exposure to NMHCs such as xylene may bring about increased prevalence of asthma, eye, nose, and throat irritation, breathing problems and effects on the pulmonary and nervous systems (Kumar *et al*., 2017). Chronic exposure to NMHCs like benzene presents a risk of anaemia, leukaemia, multiple myeloma, and other blood disorders. In addition, adverse effects on the immune system may occur (Kumar *et al*., 2017). Chronic exposure to NMHCs such as propane and butane may even lead to death (Kumar *et al*., 2017). As elucidated earlier, numerous solvents, compounds and reagents all contributed to the NMHC concentrations measured. It was impossible to ascertain the respective percentages of each compound to the total NMHC concentrations in the study area, subsequently, individual NMHC health effects have not been discussed.

Some acute health endpoints to VOC exposure include headaches, eye, nose and throat irritation, fatigue, nausea, loss of coordination, dermatitis, dyspnoea, memory impairment and dizziness (USEPA, 2016b). Chronic exposure to VOCs may bring about liver-related

health endpoints, the central nervous system and kidney effects (USEPA, 2016b). Furthermore, chronic VOC exposure to solvents such as benzene, carbon tetrachloride, chloroform, *p*-dichlorobenzene, methylene chloride, and trichloroethylene may lead to cancer (Adgate *et al*., 2004). As was the case for NMHCs, the respective percentages of each compound described as a VOC was not determined. However, the DEA (2010) has numerous standards across various working environments with a minimum of 5 ppm. In the current study, the DEA (2010) standard of 5 ppm was never exceeded in the study area in either the indoor nor outdoor environments, and as such one would not expect any health-related endpoints or other symptoms to arise.

6.8. Summary

The best predictor variables of LAV, T_{db}, CO, CO_2, NO_2, and NMHCs were seasons. For the parameters RH, H_2S, and VOCs, the best predictor variable was the building type. The indoor and outdoor environment variables were the best predictors for SO_2. Ozone had no single predictor variable that was found to significantly influence its concentration in the university environment.

With reference to an API, when employing the maximum concentrations of the pollutants in the calculation of the APIs, O_3 APIs for PEH, BMBT and BIOD were found to fall under the fair, very good and good AQ tiers respectively. On the other hand, SO_2 and CO APIs were determined to be good and very good respectively, across all three buildings. In contrast, NO_2 APIs for all three buildings were deemed to be very poor. Lastly, H_2S APIs for PEH was fair; contrastingly, both BMBT and BIOD APIs were found to be within the good AQ tier. Upon usage of the mean concentrations for calculation of the pollutant's APIs, all pollutants were found to be within very good concentration ranges, except NO_2, which was established to be within the poor AQ level.

CHAPTER 7
CONCLUSION

7.1. Introduction

This chapter shows an overview of the research's central findings, and goes on to present the study's contribution to new and scientific knowledge including conclusions and key recommendations in connection with future studies and AQ management in a university setting.

7.2. Summary

This study was designed to decipher differences in IAQ and OAQ, in addition to, seasonal variations within UL, situated in Mankweng, Polokwane, South Africa. The study was carried out across the following buildings in the Faculty of Science and Agriculture (FSA), School of Molecular and Life Sciences (SMLS): Department of Physiology and Environmental Health (PEH), Department of Biochemistry, Microbiology, and Biotechnology (BMBT) and the Department of Biodiversity (BIOD). Real-time measurements of AQ parameters were carried out across both the indoor and outdoor environments within these buildings with the aid of a hot-wire anemometer (LAV), Q-Trak IAQ monitor (T_{db}, RH, CO, and CO_2) and Aeroqual AQ monitors (O_3, SO_2, NO_2, H_2S, NMHCs, and VOCs). The measurements were taken over three consecutive days across all four seasons. The mean concentrations were computed for all AQ parameters and used for all the statistical outputs. Statistically significant differences ($p < 0.05$) were found between the indoor and outdoor environment for LAV (all three buildings), T_{db} (PEH and BMBT), RH (BIOD), O_3 (all three buildings), NO_2 (all three buildings), CO (all three buildings), CO_2 (all three buildings), NMHCs (BMBT and BIOD), and VOCs (all three buildings). Non-conformity was the norm for the comfort parameters T_{db} and RH in comparison to the ASHRAE guidelines, whilst LAV values on average conformed to the ASHRAE guideline with an exception seen across the spring season. Outdoor O_3, SO_2, CO and H_2S concentrations all complied with their respective standards. Similarly, indoor CO and CO_2 were conformant to the ASHRAE guidelines. In stark contrast, outdoor NO_2 concentrations were non-compliant to their respective standards. Non-methane hydrocarbons and VOCs had no comparative standards. Seasonal differences between buildings were also identified between IAQ and OAQ among the PEH, BMBT and BIOD buildings ($p < 0.008$).

7.3. Conclusions

In line with the objectives of this study, and the data collection and analyses carried out, the following conclusions can be made from the study:

- Significant differences exist in AQ between the indoor and outdoor environments. From literature, the indoor environment is an area whereby more pollutant exposure occurs because people spend most of their time in the indoor setting. However, the current study findings, reveal that air pollution exposure varies with environments and is dependent on the pollutant type.
- In this study, the factors which affected IAQ in the university were: the innate behaviour and thermal comfort of occupants, their use of HVAC systems, natural ventilation provisioning (opening of doors and windows), occupant densities, building structure and function, laboratory emissions, consumer products, renovation and construction initiatives.
- The factors affecting OAQ were: the prevalent meteorological conditions (wind speed and direction, T_{db}, RH and rainfall), tunnelling effects, proximity of buildings to one another, vehicular, generator and laboratory exhaust emissions, soil and vegetative emissions and municipal garbage skips.
- Air quality in the university environment varied with building, with the best AQ across both environments being within the BIOD building, whilst the worst was encountered in the PEH building.
- Linear air velocity, O_3, SO_2, CO, CO_2, and H_2S values/concentrations across the indoor/outdoor environments were within the ASHRAE/DEA/WHO guidelines/standards whereas T_{db}, RH and NO_2 values/concentrations were not.
- The winter season was found to be the season with the best AQ, with most pollutants being at minimal concentrations, whilst the summer season had the worst AQ, based on the fact that most pollutants had maximum concentrations during this season.
- The best predictor of LAV, T_{db}, CO, CO_2, NO_2, and NMHCs were seasons. For the parameters RH, H_2S, and VOCs, the best predictor was the building type. The indoor and outdoor environment were the best predictors for SO_2. Ozone had no single predictor variable that was found to significantly influence its concentration in the university environment.

- Air pollution index values for all pollutants fell within the fair, good to very good range when using mean and maxima concentrations, whereas, corresponding NO_2 concentrations throughout the study fell within the poor to very poor range.
- The PEH building was deemed to have the worst API whereas the BMBT building had the best.

7.4. Rejection of the null hypotheses

On completion of the study and all the statistical analyses, it is evident that there are both differences between indoor and outdoor environments and seasons. Thereby, we can reject both null hypotheses:

H_{01}: There are no statistically significant differences between indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa;

H_{02}: There are no statistically significant seasonal variations in indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa.

Consequently, both the alternative hypotheses can be accepted:

H_{A1}: There are statistically significant differences between indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa;

H_{A2}: There are statistically significant seasonal variations in indoor and outdoor air quality within selected buildings at the University of Limpopo, Mankweng, South Africa.

7.5. Contributions of the study to the body of knowledge

- Very few AQ studies look both into the indoor and outdoor environment in tandem, this study evaluated both environments.
- The current AQ status was evaluated based on 11 AQ parameters, which has rarely been done previously.
- The AQ across both environments in the current study has been carried out in a university setting, which is rarely used as a site to ascertain the status quo of AQ. This study has revealed how environment, building and season interplay in the determination of AQ in a university environment and can be used as a baseline for other related studies.
- This study assessed the AQ over four seasons, i.e. summer, autumn, winter, and spring. Most AQ studies done across the globe focus on the summer (non-heating

period) and winter (heating period) seasons or the monsoon and non-monsoon periods and not all four seasons altogether.

- Monitoring and evaluation is a critical component of AQ and in a South African context, an AQI could be a tool that could be looked into when formulating transition-type guidelines so as to validate the need for action to be taken. An AQI for a university setting in South Africa has been established through this study, which has not been done before in South Africa. These indices can be used as reference values for future AQ-related studies.
- The best predictors of the concentrations of various air pollutants in a university environment among environment, building and season were identified which has also not been carried out anywhere in the country.
- Major origin points of air pollution in a university environment have also been revealed.
- The role of load shedding in AQ has also been highlighted as the use of generators during load shedding has been identified as a potential source of air pollutants in environments where they are used.
- The results and findings of this study can lead to the conceptualisation of an AQ management plan for UL.

7.6. Recommendations

7.6.1. Future studies

- Future studies of this nature could be designed as a longitudinal follow-up study, to determine causality and not just associations. Such studies can keep track of the respective study populations or study sites for several years.
- Questionnaires could also form part of future studies to get an idea from the occupants as to their perception of AQ within a given environment. A combination of both a qualitative and quantitative study design may add value to future studies.
- A critical component of all studies is to attain a true and representative sample population; hence, increasing the sample size of all future studies will increase the validity and precision of future studies. The results attained can then be extrapolated and serve as comparatives to other universities worldwide.
- Additional pollutants such as PM ($PM_{2.5}$ and PM_{10}), CH_4, specific VOCs and NMHCs could be looked into for future studies similar in nature to this, thereby being able to pinpoint and decipher the exact pollutants contributing to a contaminated environment or an environment with poor AQ.

7.6.2. University air quality management

- There should be designated parking areas and bays that should be constructed and used by all personnel to avoid the parking of vehicles close to buildings as this study has revealed that the proximity of a building to car parks is a source of air pollutants.
- The issue of over-crowdedness in certain lecture halls and practical venues need to be looked into so as to reduce indoor RH levels and CO_2 concentrations within buildings which in this study were found to be influenced by occupant densities.
- Source monitoring and evaluation of pollutants should be done regularly in order to ascertain the concentrations in environments within the university that have high occupant densities.
- Furthermore, municipal waste bins need to also be located in a designated area far from university buildings and not in the immediate surroundings of buildings as is the case currently.
- All buildings need to have HVAC systems that are functional and well maintained in order to ensure good ventilation and AQ within indoor settings across UL.
- Appropriate storage facilities and management of chemicals and reagents should be taken into consideration in the planning of laboratory spaces within universities to ensure that reagents are not stored within office spaces and that emissions from these storage facilities are well taken care of.
- Staff should be discouraged from making use of kettles within office spaces, which tend to increase indoor RH levels which have been documented as contributors to fungal growth within the indoor environment.
- The university management might have to consider alternatives to dry walls being used as dividers between subsequent office spaces, so as to reduce RH levels and H_2S concentrations within buildings.

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APPENDIX

APPENDIX A1: RAW DATA MEASUREMENTS

Table A1.1: Typical example of the data collection sheet used to record measurements for all three departments across all four seasons of sampling.

Date: 29/01/2018 **Department:** Physiology and Environmental Health (PEH) **Day and Season:** Day 1: Summer

Time	**Location**	**LAV (m/s)**	**CO_2 (ppm)**	**CO (ppm)**	**RH (%)**	**T_{db} (°C)**	**SO_2 (ppm)**	**O_3 (ppm)**	**NO_2 (ppm)**	**NMHCs (ppm)**	**VOCs (ppm)**	**H_2S (ppm)**	**Observations**
07:30	Outdoor (South)												
07:43	Outdoor (East)												
07:56	Outdoor (North)												
08:09	Outdoor (West)												
Time	**Location**	**LAV (m/s)**	**CO_2 (ppm)**	**CO (ppm)**	**RH (%)**	**T_{db} (°C)**	**SO_2 (ppm)**	**O_3 (ppm)**	**NO_2 (ppm)**	**NMHCs (ppm)**	**VOCs (ppm)**	**H_2S (ppm)**	**Observations**
08:22	Indoor (1047)												
08:35	Indoor (1043a)												
08:48	Indoor (1043)												
09:01	Indoor (1037)												

The above data collection sheet was as repeated for the three days during which data was collected in all four seasons

Table A1.2: Mean indoor concentrations of the different air quality parameters collected during the period of study

Building	Sampling site	LAV (m/s)	CO_2 (ppm)	CO (ppm)	RH (%)	T_{db} (°C)	SO_2 (ppm)	O_3 (ppm)	NO_2 (ppm)	NMHCs (ppm)	VOCs (ppm)	H_2S (ppm)
PEH Building	1	0.010	700.250	0.058	55.425	20.075	0.000	0.000	0.118	0.375	0.075	0.058
	2	0.038	610.500	0.033	51.750	20.375	0.000	0.000	0.060	0.360	0.068	0.025
	3	0.025	577.000	0.035	49.025	20.950	0.000	0.000	0.093	0.300	0.045	0.008
	4	0.010	530.500	0.035	47.825	21.300	0.005	0.000	0.093	0.255	0.045	0.010
	5	0.010	561.000	0.025	47.450	21.375	0.005	0.000	0.075	0.265	0.038	0.008
	6	0.000	634.500	0.008	47.075	21.700	0.005	0.000	0.063	0.275	0.053	0.013
	7	0.063	563.500	0.000	46.525	21.750	0.005	0.000	0.133	0.268	0.033	0.013
	8	0.013	652.250	0.018	49.200	21.175	0.005	0.000	0.100	0.160	0.033	0.033
	9	0.020	566.750	0.008	45.925	21.650	0.013	0.003	0.083	0.150	0.033	0.015
	10	0.020	563.750	0.033	44.875	21.900	0.028	0.000	0.088	0.165	0.033	0.028
	11	0.008	599.750	0.033	43.275	22.225	0.013	0.000	0.103	0.178	0.028	0.025
	12	0.060	584.250	0.043	43.200	22.550	0.005	0.000	0.095	0.175	0.030	0.013
	13	0.100	566.250	0.058	43.850	21.375	0.008	0.003	0.118	0.168	0.025	0.018
	14	0.000	630.500	0.043	47.500	22.375	0.008	0.025	0.075	0.158	0.058	0.025
	15	0.018	708.500	0.008	45.350	23.000	0.013	0.000	0.175	0.150	0.030	0.033
	16	0.003	685.750	0.040	45.575	22.900	0.020	0.000	0.088	0.158	0.040	0.033
	17	0.003	786.000	0.033	45.800	22.975	0.023	0.000	0.125	0.158	0.030	0.040
	18	0.003	656.500	0.043	43.075	23.675	0.020	0.000	0.135	0.193	0.050	0.020
	19	0.005	616.000	0.008	46.425	22.575	0.033	0.000	0.100	0.160	0.030	0.060
	20	0.020	719.250	0.008	45.225	23.000	0.033	0.000	0.098	0.143	0.030	0.048
BMBT building	21	0.008	479.000	0.025	59.150	18.225	0.010	0.000	0.220	0.318	0.058	0.043
	22	0.033	605.000	0.033	60.825	18.550	0.010	0.000	0.128	0.390	0.068	0.065
	23	0.020	558.000	0.025	51.375	19.775	0.003	0.000	0.078	0.400	0.075	0.003
	24	0.015	624.250	0.025	53.875	20.175	0.008	0.000	0.185	0.393	0.095	0.013
	25	0.008	601.500	0.040	52.225	20.450	0.013	0.000	0.103	0.400	0.103	0.010

	26	0.003	535.000	0.033	50.575	20.700	0.005	0.000	0.073	0.318	0.045	0.028
	27	0.018	490.500	0.050	49.775	20.675	0.013	0.000	0.105	0.308	0.030	0.020
	28	0.008	534.000	0.025	50.325	20.500	0.008	0.000	0.058	0.415	0.113	0.013
	29	0.015	514.000	0.018	49.725	20.525	0.008	0.000	0.095	0.435	0.088	0.008
	30	0.028	637.250	0.043	51.625	20.925	0.015	0.000	0.105	0.933	0.480	0.025
	31	0.093	661.000	0.015	45.075	22.875	0.015	0.000	0.095	0.808	0.318	0.023
	32	0.005	562.000	0.060	45.825	22.475	0.013	0.000	0.045	0.635	0.178	0.013
	33	0.035	620.000	0.065	46.000	22.850	0.020	0.000	0.080	0.758	0.165	0.018
	34	0.058	606.750	0.065	46.625	22.375	0.010	0.000	0.060	2.028	5.073	0.018
	35	0.005	704.000	0.048	47.025	22.400	0.028	0.000	0.098	1.345	1.118	0.030
	36	0.003	580.750	0.025	44.900	22.175	0.018	0.000	0.103	0.815	0.115	0.023
	37	0.003	550.750	0.050	44.475	22.625	0.015	0.000	0.068	0.685	0.065	0.023
	38	0.043	590.500	0.058	44.025	22.450	0.010	0.000	0.068	0.710	0.138	0.003
	39	0.005	646.750	0.015	45.225	22.825	0.010	0.000	0.083	0.808	0.148	0.013
	40	0.005	523.250	0.015	47.025	21.425	0.003	0.000	0.075	0.375	0.065	0.010
BIOD building	41	0.058	471.250	0.018	53.025	18.875	0.003	0.000	0.100	0.408	0.023	0.018
	42	0.013	493.250	0.018	50.300	19.375	0.000	0.003	0.090	0.360	0.018	0.023
	43	0.085	574.750	0.025	52.050	20.700	0.003	0.003	0.093	0.465	0.030	0.023
	44	0.010	513.750	0.050	49.625	20.325	0.005	0.000	0.073	0.433	0.025	0.015
	45	0.010	447.250	0.058	48.075	20.250	0.005	0.000	0.068	0.375	0.023	0.018
	46	0.018	553.250	0.018	47.725	20.825	0.000	0.000	0.080	0.433	0.030	0.005
	47	0.013	510.250	0.058	46.050	20.825	0.000	0.000	0.055	0.428	0.025	0.008
	48	0.008	524.750	0.040	48.225	20.550	0.000	0.000	0.085	0.383	0.023	0.018
	49	0.003	489.500	0.043	48.275	20.550	0.000	0.003	0.085	0.408	0.040	0.010
	50	0.018	535.000	0.048	47.475	20.850	0.003	0.000	0.090	0.368	0.043	0.010
	51	0.003	558.500	0.018	46.700	20.900	0.003	0.000	0.090	0.318	0.033	0.008
	52	0.023	596.000	0.008	46.550	21.150	0.000	0.000	0.078	0.635	0.078	0.020
	53	0.005	570.750	0.073	45.250	21.200	0.000	0.000	0.083	1.658	0.125	0.015
	54	0.005	533.750	0.025	44.775	21.800	0.000	0.000	0.105	0.675	0.025	0.013
	55	0.005	464.750	0.008	47.600	20.575	0.000	0.000	0.063	0.375	0.020	0.015

	56	0.003	583.000	0.033	44.800	21.825	0.003	0.000	0.068	0.708	0.033	0.018
	57	0.013	558.500	0.015	41.725	22.250	0.008	0.000	0.080	0.515	0.030	0.028
	58	0.013	558.000	0.018	41.350	22.375	0.003	0.000	0.108	0.468	0.028	0.023
	59	0.050	512.500	0.015	37.925	22.775	0.005	0.003	0.065	0.530	0.015	0.028
	60	0.010	563.000	0.058	38.025	23.350	0.005	0.005	0.100	0.470	0.033	0.023

Table A1.3: Mean Outdoor concentrations of the different air quality parameters collected during the period of study

Building	Sampling site	LAV (m/s)	CO_2 (ppm)	CO (ppm)	RH (%)	T_{db} (°C)	SO_2 (ppm)	O_3 (ppm)	NO_2 (ppm)	NMHCs (ppm)	VOCs (ppm)	H_2S (ppm)
PEH Building	1	0.438	404.750	0.235	56.775	16.875	0.058	0.000	0.140	0.383	0.055	0.020
	2	0.548	400.000	0.058	60.075	16.275	0.048	0.000	0.200	0.433	0.053	0.020
	3	0.613	394.250	0.043	58.375	17.025	0.043	0.005	0.235	0.465	0.063	0.023
	4	0.603	401.750	0.075	57.250	17.000	0.028	0.003	0.193	0.585	0.083	0.063
	5	0.530	402.250	0.193	51.550	18.675	0.015	0.000	0.088	0.143	0.023	0.013
	6	0.503	397.000	0.015	51.850	19.000	0.010	0.010	0.140	0.140	0.020	0.013
	7	0.770	408.250	0.060	49.325	20.050	0.000	0.008	0.160	0.140	0.025	0.020
	8	0.415	393.750	0.248	47.450	19.800	0.005	0.020	0.140	0.133	0.033	0.048
	9	0.560	388.750	0.075	48.025	20.775	0.008	0.003	0.088	0.125	0.018	0.008
	10	0.863	401.000	0.018	46.600	21.500	0.005	0.010	0.140	0.125	0.020	0.005
	11	0.833	413.000	0.050	43.575	22.300	0.000	0.010	0.143	0.108	0.018	0.015
	12	0.690	378.750	0.310	45.400	21.175	0.008	0.018	0.145	0.108	0.018	0.023
	13	0.553	399.000	0.043	46.875	21.400	0.028	0.000	0.113	0.148	0.015	0.040
	14	0.735	398.000	0.083	46.375	21.600	0.005	0.010	0.163	0.148	0.023	0.028
	15	1.213	426.500	0.025	43.950	22.625	0.008	0.018	0.113	0.148	0.020	0.013
	16	0.908	392.500	0.268	43.625	22.700	0.000	0.018	0.145	0.148	0.025	0.010
	17	0.598	395.250	0.033	43.925	22.250	0.038	0.010	0.088	0.118	0.013	0.035
	18	0.935	411.750	0.068	42.500	23.900	0.000	0.013	0.130	0.108	0.013	0.020
	19	1.305	414.000	0.228	39.725	24.150	0.005	0.013	0.133	0.108	0.015	0.015
	20	0.955	388.250	0.265	38.375	24.375	0.000	0.035	0.163	0.115	0.023	0.008
BMBT building	21	0.285	407.000	0.158	60.275	16.175	0.035	0.000	0.145	0.325	0.038	0.038
	22	0.408	403.250	0.008	63.650	15.700	0.028	0.003	0.185	0.325	0.043	0.040
	23	0.458	400.250	0.075	62.975	16.150	0.023	0.003	0.205	0.340	0.043	0.040
	24	0.410	397.250	0.243	60.850	16.175	0.035	0.005	0.175	0.383	0.053	0.053
	25	0.400	401.000	0.143	54.900	17.550	0.003	0.003	0.115	0.175	0.023	0.013

	26	0.335	409.750	0.050	56.950	18.250	0.010	0.008	0.195	0.165	0.018	0.028
	27	0.425	411.750	0.058	54.700	18.625	0.008	0.008	0.140	0.158	0.018	0.033
	28	0.295	401.500	0.235	54.450	18.325	0.013	0.008	0.135	0.165	0.020	0.053
	29	0.363	391.750	0.095	48.650	19.675	0.008	0.000	0.095	0.140	0.010	0.018
	30	0.408	394.250	0.068	51.375	19.300	0.005	0.005	0.155	0.150	0.025	0.023
	31	0.500	418.750	0.033	48.425	21.000	0.000	0.008	0.160	0.143	0.013	0.025
	32	0.253	393.250	0.215	47.700	20.325	0.015	0.005	0.153	0.165	0.010	0.038
	33	0.713	388.250	0.075	46.325	20.225	0.005	0.010	0.088	0.150	0.008	0.018
	34	0.468	401.000	0.033	46.575	20.475	0.003	0.010	0.133	0.133	0.008	0.028
	35	0.408	430.750	0.065	43.325	22.625	0.008	0.013	0.148	0.140	0.015	0.013
	36	0.465	394.500	0.160	43.875	22.075	0.010	0.013	0.145	0.150	0.008	0.025
	37	0.688	422.500	0.075	40.175	22.925	0.010	0.015	0.130	0.140	0.013	0.005
	38	0.520	421.500	0.065	38.400	24.225	0.000	0.015	0.088	0.133	0.008	0.000
	39	0.650	416.750	0.158	36.900	24.425	0.000	0.013	0.120	0.133	0.015	0.005
	40	0.465	400.750	0.128	40.050	23.400	0.013	0.020	0.100	0.133	0.005	0.003
BIOD building	41	0.865	404.500	0.125	56.700	16.975	0.045	0.000	0.170	0.518	0.025	0.025
	42	0.765	402.750	0.083	57.375	17.400	0.033	0.005	0.218	0.535	0.033	0.030
	43	0.585	409.500	0.033	56.750	17.825	0.020	0.008	0.248	0.565	0.035	0.048
	44	0.265	417.250	0.175	56.475	17.550	0.008	0.008	0.233	0.625	0.043	0.050
	45	1.213	400.750	0.085	50.725	18.850	0.005	0.008	0.125	0.218	0.013	0.008
	46	0.425	432.250	0.018	48.150	20.800	0.008	0.013	0.170	0.200	0.010	0.010
	47	1.078	444.500	0.025	44.575	21.275	0.000	0.013	0.125	0.208	0.008	0.035
	48	0.360	394.250	0.383	45.675	20.400	0.005	0.013	0.143	0.208	0.010	0.043
	49	1.080	401.250	0.058	41.175	20.750	0.000	0.008	0.058	0.193	0.010	0.005
	50	0.370	410.250	0.008	39.600	22.525	0.000	0.015	0.160	0.173	0.015	0.008
	51	1.645	430.500	0.083	35.850	23.075	0.000	0.020	0.153	0.175	0.005	0.015
	52	0.388	383.250	0.415	38.175	21.575	0.000	0.018	0.165	0.175	0.010	0.023
	53	0.803	390.250	0.100	36.300	21.800	0.003	0.003	0.073	0.200	0.010	0.003
	54	0.555	443.250	0.000	33.975	24.375	0.000	0.010	0.180	0.185	0.010	0.005
	55	1.070	403.250	0.260	32.400	23.375	0.000	0.020	0.150	0.185	0.010	0.013

	56	0.665	390.750	0.275	33.400	23.325	0.000	0.020	0.143	0.185	0.005	0.018
	57	0.985	381.750	0.115	32.625	23.525	0.048	0.020	0.083	0.183	0.020	0.015
	58	1.075	415.500	0.008	31.850	24.925	0.015	0.025	0.140	0.175	0.008	0.008
	59	1.225	382.750	0.250	30.800	23.725	0.000	0.018	0.143	0.165	0.008	0.008
	60	0.360	407.000	0.100	29.025	25.900	0.000	0.025	0.165	0.143	0.013	0.008

Table A1.4: Indoor:outdoor ratios of the different air quality parameters collected during the period of study

Building	Sampling site	LAV	CO_2	CO	RH	T_{db}	SO_2	O_3	NO_2	NMHCs	VOCs	H_2S
PEH Building	1	0.023	1.730	0.247	0.976	1.190	0.000	-	0.843	0.979	1.364	2.900
	2	0.069	1.526	0.569	0.861	1.252	0.000	-	0.300	0.831	1.283	1.250
	3	0.041	1.464	0.814	0.840	1.231	0.000	0.000	0.396	0.645	0.714	0.348
	4	0.017	1.320	0.467	0.835	1.253	0.179	0.000	0.482	0.436	0.542	0.159
	5	0.019	1.395	0.130	0.920	1.145	0.333	-	0.852	1.853	1.652	0.615
	6	0.000	1.598	0.533	0.908	1.142	0.500	0.000	0.450	1.964	2.650	1.000
	7	0.082	1.380	0.000	0.943	1.085	-	0.000	0.831	1.914	1.320	0.650
	8	0.031	1.657	0.073	1.037	1.069	1.000	0.000	0.714	1.203	1.000	0.688
	9	0.036	1.458	0.107	0.956	1.042	1.625	1.000	0.943	1.200	1.833	1.875
	10	0.023	1.406	1.833	0.963	1.019	5.600	0.000	0.629	1.320	1.650	5.600
	11	0.010	1.452	0.660	0.993	0.997	-	0.000	0.720	1.648	1.556	1.667
	12	0.087	1.543	0.139	0.952	1.065	0.625	0.000	0.655	1.620	1.667	0.565
	13	0.181	1.419	1.349	0.935	0.999	0.286	-	1.044	1.135	1.667	0.450
	14	0.000	1.584	0.518	1.024	1.036	1.600	2.500	0.460	1.068	2.522	0.893
	15	0.015	1.661	0.320	1.032	1.017	1.625	0.000	1.549	1.014	1.500	2.538
	16	0.003	1.747	0.149	1.045	1.009	-	0.000	0.607	1.068	1.600	3.300
	17	0.005	1.989	1.000	1.043	1.033	0.605	0.000	1.420	1.339	2.308	1.143
	18	0.003	1.594	0.632	1.014	0.991	-	0.000	1.038	1.787	3.846	1.000
	19	0.004	1.488	0.035	1.169	0.935	6.600	0.000	0.752	1.481	2.000	4.000
	20	0.021	1.853	0.030	1.179	0.944	-	0.000	0.601	1.243	1.304	6.000
BMBT building	21	0.028	1.177	0.158	0.981	1.127	0.286	-	1.517	0.978	1.526	1.132
	22	0.081	1.500	4.125	0.956	1.182	0.357	0.000	0.692	1.200	1.581	1.625
	23	0.044	1.394	0.333	0.816	1.224	0.130	0.000	0.380	1.176	1.744	0.075

	24	0.037	1.571	0.103	0.885	1.247	0.229	0.000	1.057	1.026	1.792	0.245
	25	0.020	1.500	0.280	0.951	1.165	4.333	0.000	0.896	2.286	4.478	0.769
	26	0.009	1.306	0.660	0.888	1.134	0.500	0.000	0.374	1.927	2.500	1.000
	27	0.042	1.191	0.862	0.910	1.110	1.625	0.000	0.750	1.949	1.667	0.606
	28	0.027	1.330	0.106	0.924	1.119	0.615	0.000	0.430	2.515	5.650	0.245
	29	0.041	1.312	0.189	1.022	1.043	1.000	-	1.000	3.107	8.800	0.444
	30	0.069	1.616	0.632	1.005	1.084	3.000	0.000	0.677	6.220	19.200	1.087
	31	0.186	1.579	0.455	0.931	1.089	-	0.000	0.594	5.650	24.462	0.920
	32	0.020	1.429	0.279	0.961	1.106	0.867	0.000	0.294	3.848	17.800	0.342
	33	0.049	1.597	0.867	0.993	1.130	4.000	0.000	0.909	5.053	20.625	1.000
	34	0.124	1.513	1.970	1.001	1.093	3.333	0.000	0.451	15.248	634.125	0.643
	35	0.012	1.634	0.738	1.085	0.990	3.500	0.000	0.662	9.607	74.533	2.308
	36	0.006	1.472	0.156	1.023	1.005	1.800	0.000	0.710	5.433	14.375	0.920
	37	0.004	1.304	0.667	1.107	0.987	1.500	0.000	0.523	4.893	5.000	4.600
	38	0.083	1.401	0.892	1.146	0.927	-	0.000	0.773	5.338	17.250	-
	39	0.008	1.552	0.095	1.226	0.934	-	0.000	0.692	6.075	9.867	2.600
	40	0.011	1.306	0.117	1.174	0.916	0.231	0.000	0.750	2.820	13.000	3.333
BIOD building	41	0.067	1.165	0.144	0.935	1.112	0.067	-	0.588	0.788	0.920	0.720
	42	0.017	1.225	0.217	0.877	1.114	0.000	0.600	0.413	0.673	0.545	0.767
	43	0.145	1.404	0.758	0.917	1.161	0.150	0.375	0.375	0.823	0.857	0.479
	44	0.038	1.231	0.286	0.879	1.158	0.625	0.000	0.313	0.693	0.581	0.300
	45	0.008	1.116	0.682	0.948	1.074	1.000	0.000	0.544	1.720	1.769	2.250
	46	0.042	1.280	1.000	0.991	1.001	0.000	0.000	0.471	2.165	3.000	0.500
	47	0.012	1.148	2.320	1.033	0.979	-	0.000	0.440	2.058	3.125	0.229
	48	0.022	1.331	0.104	1.056	1.007	0.000	0.000	0.594	1.841	2.300	0.419
	49	0.003	1.220	0.741	1.172	0.990	-	0.375	1.466	2.114	4.000	2.000
	50	0.049	1.304	6.000	1.199	0.926	-	0.000	0.563	2.127	2.867	1.250

	51	0.002	1.297	0.217	1.303	0.906	-	0.000	0.588	1.817	6.600	0.533
	52	0.059	1.555	0.019	1.219	0.980	-	0.000	0.473	3.629	7.800	0.870
	53	0.006	1.463	0.730	1.247	0.972	0.000	0.000	1.137	8.290	12.500	5.000
	54	0.009	1.204	-	1.318	0.894	-	0.000	0.583	3.649	2.500	2.600
	55	0.005	1.153	0.031	1.469	0.880	-	0.000	0.420	2.027	2.000	1.154
	56	0.005	1.492	0.120	1.341	0.936	-	0.000	0.476	3.827	6.600	1.000
	57	0.013	1.463	0.130	1.279	0.946	0.167	0.000	0.964	2.814	1.500	1.867
	58	0.012	1.343	2.250	1.298	0.898	0.200	0.000	0.771	2.674	3.500	2.875
	59	0.041	1.339	0.060	1.231	0.960	-	0.167	0.455	3.212	1.875	3.500
	60	0.028	1.383	0.580	1.310	0.902	-	0.200	0.606	3.287	2.538	2.875

APPENDIX A2: STATISTICAL OUTPUTS

Table A2.1: Friedman test to identify building differences in the indoor and outdoor environment.

Parameter	N (df)	Chi-square (Friedman's Q statistic)	p-value
LAV_PEH_IAQ LAV_BMBT_IAQ LAV_BIOD_IAQ*	20 (2)	1.371	0.504
LAV_PEH_OAQ LAV_BMBT_OAQ LAV_BIOD_OAQ**	20 (2)	17.100	0.000
RH_PEH_IAQ RH_BMBT_IAQ RH_BIOD_IAQ	20 (2)	10.800	0.005
RH_PEH_OAQ RH_BMBT_OAQ RH_BIOD_OAQ	20 (2)	32.500	0.000
T_{db}_PEH_IAQ T_{db} _BMBT_IAQ T_{db} _BIOD_IAQ	20 (2)	16.278	0.000
T_{db}_PEH_OAQ T_{db} _BMBT_OAQ T_{db} _BIOD_OAQ	20 (2)	26.911	0.000
O_3_PEH_IAQ O_3_BMBT_IAQ O_3_BIOD_IAQ	20 (2)	5.429	0.066
O_3_PEH_OAQ O_3_BMBT_OAQ O_3_BIOD_OAQ	20 (2)	19.159	0.000
SO_2_PEH_IAQ SO_2_BMBT_IAQ SO_2_BIOD_IAQ	20 (2)	19.307	0.000
SO_2_PEH_OAQ SO_2_BMBT_OAQ SO_2_BIOD_OAQ	20 (2)	8.576	0.014
NO_2_PEH_IAQ NO_2_BMBT_IAQ NO_2_BIOD_IAQ	20 (2)	2.608	0.271
NO_2_PEH_OAQ NO_2_BMBT_OAQ NO_2_BIOD_OAQ	20 (2)	4.886	0.087
CO_PEH_IAQ CO_BMBT_IAQ CO_BIOD_IAQ	20 (2)	2.079	0.354
CO_PEH_OAQ CO_BMBT_OAQ CO_BIOD_OAQ	20 (2)	0.100	0.951
CO_2_PEH_IAQ CO_2_BMBT_IAQ CO_2_BIOD_IAQ	20 (2)	13.300	0.001
CO_2_PEH_OAQ CO_2_BMBT_OAQ CO_2_BIOD_OAQ	20 (2)	6.300	0.043
H_2S_PEH_IAQ H_2S_BMBT_IAQ H_2S_BIOD_IAQ	20 (2)	6.256	0.044
H_2S_PEH_OAQ H_2S_BMBT_OAQ H_2S_BIOD_OAQ	20 (2)	3.746	0.154
NMHCs_PEH_IAQ NMHCs_BMBT_IAQ NMHCs_BIOD_IAQ	20 (2)	26.846	0.000

NMHCs_PEH_OAQ NMHCs_BMBT_OAQ NMHCs_BIOD_OAQ	20 (2)	31.600	0.000
VOCs_PEH_IAQ VOCs_BMBT_IAQ VOCs_BIOD_IAQ	20 (2)	23.769	0.000
VOCs_PEH_OAQ VOCs_BMBT_OAQ VOCs_BIOD_OAQ	20 (2)	25.054	0.000

* Linear air velocity_Department of Physiology and Environmental Health_Indoor air quality-Linear air velocity_Department of Biochemsitry, Microbiology and Biotechnology_Indoor air quality- Linear air velocity_Department of Biodiversity_Indoor air quality;
** * Linear air velocity_Department of Physiology and Environmental Health_Outdoor air quality-Linear air velocity_Department of Biochemsitry, Microbiology and Biotechnology_Outdoor air quality- Linear air velocity_Department of Biodiversity_Outdoor air quality.

Table A2.2: Wilcoxon signed ranks test to identify individual building differences in indoor and outdoor air quality.

Parameter	Z	p-value
LAV_PEH_OAQ – LAV_BMBT_OAQ	-3.621[b]	0.000
LAV_PEH_OAQ – LAV_BIOD_OAQ	-0.560[a]	0.575
LAV_BMBT_OAQ – LAV_BIOD_OAQ	-3.323[a]	0.001
RH_PEH_IAQ – RH_BMBT_IAQ	-3.398[a]	0.001
RH_PEH_IAQ – RH_BIOD_IAQ	-0.112[b]	0.911
RH_BMBT_IAQ – RH_BIOD_IAQ	-3.323[b]	0.001
RH_PEH_OAQ – RH_BMBT_OAQ	-2.333[a]	0.020
RH_PEH_OAQ – RH_BIOD_OAQ	-3.920[b]	0.000
RH_BMBT_OAQ – RH_BIOD_OAQ	-3.920[b]	0.000
T_{db}_PEH_IAQ – T_{db}_BMBT_IAQ	-2.898[b]	0.004
T_{db}_PEH_IAQ – T_{db}_BIOD_IAQ	-3.696[b]	0.000
T_{db}_BMBT_IAQ – T_{db}_BIOD_IAQ	-0.728[b]	0.467
T_{db}_PEH_OAQ – T_{db}_BMBT_OAQ	-3.501[a]	0.000
T_{db}_PEH_OAQ – T_{db}_BIOD_OAQ	-3.659[b]	0.000
T_{db}_BMBT_OAQ – T_{db}_BIOD_OAQ	-3.828[b]	0.000
O_3_PEH_OAQ – O_3_BMBT_OAQ	-1.274[b]	0.203
O_3_PEH_OAQ – O_3_BIOD_OAQ	-2.358[a]	0.018
O_3_BMBT_OAQ – O_3_BIOD_OAQ	-3.235[a]	0.001
SO_2_PEH_IAQ – SO_2_BMBT_IAQ	-0.423[a]	0.672
SO_2_PEH_IAQ – SO_2_BIOD_IAQ	-3.487[b]	0.000
SO_2_BMBT_IAQ – SO_2_BIOD_IAQ	-3.787[b]	0.000
SO_2_PEH_OAQ – SO_2_BMBT_OAQ	-1.163[b]	0.245
SO_2_PEH_OAQ – SO_2_BIOD_OAQ	-2.276[b]	0.023
SO_2_BMBT_OAQ – SO_2_BIOD_OAQ	-1.353[b]	0.176
CO_2_PEH_IAQ – CO_2_BMBT_IAQ	-1.717[b]	0.086
CO_2_PEH_IAQ – CO_2_BIOD_IAQ	-3.733[b]	0.000

CO_2_BMBT_IAQ – CO_2_BIOD_IAQ	-2.333[b]	0.020
CO_2_PEH_OAQ – CO_2_BMBT_OAQ	-2.427[a]	0.015
CO_2_PEH_OAQ – CO_2_BIOD_OAQ	-1.680[a]	0.093
CO_2_BMBT_OAQ – CO_2_BIOD_OAQ	-0.560[a]	0.575
H_2S_PEH_IAQ – H_2S_BMBT_IAQ	-1.897[b]	0.058
H_2S_PEH_IAQ – H_2S_BIOD_IAQ	-2.560[b]	0.010
H_2S_BMBT_IAQ – H_2S_BIOD_IAQ	-0.617[b]	0.537
NMHCS_PEH_IAQ – NMHCS_BMBT_IAQ	-3.771[a]	0.000
NMHCS_PEH_IAQ – NMHCS_BIOD_IAQ	-3.823[a]	0.000
NMHCS_BMBT_IAQ – NMHCS_BIOD_IAQ	-1.248[b]	0.212
NMHCS_PEH_OAQ – NMHCS_BMBT_OAQ	-0.878[b]	0.380
NMHCS_PEH_OAQ – NMHCS_BIOD_OAQ	-3.923[b]	0.000
NMHCS_BMBT_OAQ – NMHCS_BIOD_OAQ	-3.923[b]	0.000
VOCS_PEH_IAQ – VOCS_BMBT_IAQ	-3.582[a]	0.000
VOCS_PEH_IAQ – VOCS_BIOD_IAQ	-1.873[b]	0.061
VOCS_BMBT_IAQ – VOCS_BIOD_IAQ	-3.920[b]	0.000
VOCS_PEH_OAQ – VOCS_BMBT_OAQ	-3.462[b]	0.001
VOCS_PEH_OAQ – VOCS_BIOD_OAQ	-3.758[b]	0.000
VOCS_BMBT_OAQ – VOCS_BIOD_OAQ	-2.835[b]	0.005

a. Based on negative ranks.
b. Based on positive ranks.

Table A2.3: Friedman test to identify seasonal differences in the Department of Physiology and Environmental Health.

Parameter	N (df)	Chi-square (Friedman's Q statistic)	p-value
LAV_PEH_SUM_IAQ LAV_PEH_AUT_IAQ LAV_PEH_WIN_IAQ LAV_PEH_SPR_IAQ	20 (3)	4.039	0.257
LAV_PEH_SUM_OAQ LAV_PEH_AUT_OAQ LAV_PEH_WIN_OAQ LAV_PEH_SPR_OAQ	20 (3)	24.300	0.000
RH_PEH_SUM_IAQ RH_PEH_AUT_IAQ RH_PEH_WIN_IAQ RH_PEH_SPR_IAQ	20 (3)	57.840	0.000
RH_PEH_SUM_OAQ RH_PEH_AUT_OAQ RH_PEH_WIN_OAQ RH_PEH_SPR_OAQ	20 (3)	54.540	0.000
T_{db}_PEH_SUM_IAQ T_{db}_PEH_AUT_IAQ T_{db}_PEH_WIN_IAQ T_{db}_PEH_SPR_IAQ	20 (3)	54.545	0.000
T_{db}_PEH_SUM_OAQ T_{db}_PEH_AUT_OAQ T_{db}_PEH_WIN_OAQ T_{db}_PEH_SPR_OAQ	20 (3)	54.000	0.000
O_3_PEH_SUM_IAQ O_3_PEH_AUT_IAQ O_3_PEH_WIN_IAQ O_3_PEH_SPR_IAQ	20 (3)	3.444	0.328
O_3_PEH_SUM_OAQ O_3_PEH_AUT_OAQ O_3_PEH_WIN_OAQ O_3_PEH_SPR_OAQ	20 (3)	34.803	0.000
SO_2_PEH_SUM_IAQ SO_2_PEH_AUT_IAQ SO_2_PEH_WIN_IAQ SO_2_PEH_SPR_IAQ	20 (3)	43.667	0.000
SO_2_PEH_SUM_OAQ SO_2_PEH_AUT_OAQ SO_2_PEH_WIN_OAQ SO_2_PEH_SPR_OAQ	20 (3)	11.083	0.011
NO_2_PEH_SUM_IAQ NO_2_PEH_AUT_IAQ NO_2_PEH_WIN_IAQ NO_2_PEH_SPR_IAQ	20 (3)	19.386	0.000
NO_2_PEH_SUM_OAQ NO_2_PEH_AUT_OAQ NO_2_PEH_WIN_OAQ NO_2_PEH_SPR_OAQ	20 (3)	8.803	0.032
CO_PEH_SUM_IAQ CO_PEH_AUT_IAQ CO_PEH_WIN_IAQ CO_PEH_SPR_IAQ	20 (3)	5.300	0.151
CO_PEH_SUM_OAQ CO_PEH_AUT_OAQ CO_PEH_WIN_OAQ CO_PEH_SPR_OAQ	20 (3)	4.073	0.254
CO_2_PEH_SUM_IAQ CO_2_PEH_AUT_IAQ CO_2_PEH_WIN_IAQ CO_2_PEH_SPR_IAQ	20 (3)	16.296	0.001
CO_2_PEH_SUM_OAQ CO_2_PEH_AUT_OAQ CO_2_PEH_WIN_OAQ CO_2_PEH_SPR_OAQ	20 (3)	33.720	0.000
H_2S_PEH_SUM_IAQ H_2S_PEH_AUT_IAQ H_2S_PEH_WIN_IAQ H_2S_PEH_SPR_IAQ	20 (3)	24.422	0.000
H_2S_PEH_SUM_OAQ H_2S_PEH_AUT_OAQ H_2S_PEH_WIN_OAQ H_2S_PEH_SPR_OAQ	20 (3)	20.409	0.000
NMHCs_PEH_SUM_IAQ NMHCs_PEH_AUT_IAQ NMHCs_PEH_WIN_IAQ NMHCs_PEH_SPR_IAQ	20 (3)	58.846	0.000
NMHCs_PEH_SUM_OAQ NMHCs_PEH_AUT_OAQ NMHCs_PEH_WIN_OAQ NMHCs_PEH_SPR_OAQ	20 (3)	58.445	0.000
VOCs_PEH_SUM_IAQ VOCs_PEH_AUT_IAQ VOCs_PEH_WIN_IAQ VOCs_PEH_SPR_IAQ	20 (3)	19.406	0.000
VOCs_PEH_SUM_OAQ VOCs_PEH_AUT_OAQ VOCs_PEH_WIN_OAQ VOCs_PEH_SPR_OAQ	20 (3)	24.613	0.000

The same test was carried out to find seasonal differences in both the Department of Biochemistry, Microbiology and Biotechnology and the Department of Biodiversity.

Table A2.4: Wilcoxon signed ranks test to identify individual seasonal differences between indoor and outdoor air quality in the Department of Physiology and Environmental Health.

Parameter	Z	p-value
LAV_PEH_AUT_OAQ – LAV _PEH_SUM_OAQ	-3.771[b]	0.000
LAV_PEH_WIN_OAQ – LAV_PEH_SUM_OAQ	-2.315[b]	0.021
LAV_PEH_SPR_OAQ – LAV_PEH_SUM_OAQ	-2.949[b]	0.003
LAV_PEH_WIN_OAQ – LAV_PEH_AUT_OAQ	-3.547[a]	0.000
LAV_PEH_SPR_OAQ – LAV_PEH_AUT_OAQ	-2.501[a]	0.012
LAV_PEH_SPR_OAQ – LAV_PEH_WIN_OAQ	-1.736[b]	0.083
RH_PEH_AUT_IAQ – RH_PEH_SUM_IAQ	-3.772[b]	0.000
RH_PEH_WIN_IAQ – RH_PEH_SUM_IAQ	-3.922[a]	0.000
RH_PEH_SPR_IAQ – RH_PEH_SUM_IAQ	-3.920[a]	0.000
RH_PEH_WIN_IAQ – RH_PEH_AUT_IAQ	-3.921[a]	0.000
RH_PEH_SPR_IAQ – RH_PEH_AUT_IAQ	-3.920[a]	0.000
RH_PEH_SPR_IAQ – RH_PEH_WIN_IAQ	-3.920[a]	0.000
RH_PEH_AUT_OAQ – RH_PEH_SUM_OAQ	-2.315[b]	0.021
RH_PEH_WIN_OAQ – RH_PEH_SUM_OAQ	-3.921[a]	0.000
RH_PEH_SPR_OAQ – RH_PEH_SUM_OAQ	-3.920[a]	0.000
RH_PEH_WIN_OAQ – RH_PEH_AUT_OAQ	-3.921[a]	0.000
RH_PEH_SPR_OAQ – RH_PEH_AUT_OAQ	-3.921[a]	0.000
RH_PEH_SPR_OAQ – RH_PEH_WIN_OAQ	-3.920[a]	0.000
T_{db}_PEH_AUT_IAQ – T_{db}_PEH_SUM_IAQ	-3.923[a]	0.000
T_{db}_PEH_WIN_IAQ – T_{db}_PEH_SUM_IAQ	-3.921[a]	0.000
T_{db}_PEH_SPR_IAQ – T_{db}_PEH_SUM_IAQ	-3.921[a]	0.000
T_{db}_PEH_WIN_IAQ – T_{db}_PEH_AUT_IAQ	-3.922[a]	0.000
T_{db}_PEH_SPR_IAQ – T_{db}_PEH_AUT_IAQ	-0.349[b]	0.727
T_{db}_PEH_SPR_IAQ – T_{db}_PEH_WIN_IAQ	-3.922[b]	0.000
T_{db}_PEH_AUT_OAQ – T_{db}_PEH_SUM_OAQ	-3.921[a]	0.000

T_{db}_PEH_WIN_OAQ – T_{db}_PEH_SUM_OAQ	-3.920[a]	0.000
T_{db}_PEH_SPR_OAQ – T_{db}_PEH_SUM_OAQ	-1.251[b]	0.211
T_{db}_PEH_WIN_OAQ – T_{db}_PEH_AUT_OAQ	-3.923[a]	0.000
T_{db}_PEH_SPR_OAQ – T_{db}_PEH_AUT_OAQ	-3.920[b]	0.000
T_{db}_PEH_SPR_OAQ – T_{db}_PEH_WIN_OAQ	-3.921[b]	0.000
O_3_PEH_AUT_OAQ – O_3_PEH_SUM_OAQ	-2.148[a]	0.032
O_3_PEH_WIN_OAQ – O_3_PEH_SUM_OAQ	-2.244[b]	0.025
O_3_PEH_SPR_OAQ – O_3_PEH_SUM_OAQ	-3.101[b]	0.002
O_3_PEH_WIN_OAQ – O_3_PEH_AUT_OAQ	-2.691[b]	0.007
O_3_PEH_SPR_OAQ – O_3_PEH_AUT_OAQ	-3.383[b]	0.001
O_3_PEH_SPR_OAQ – O_3_PEH_WIN_OAQ	-1.855[b]	0.064
SO_2_PEH_AUT_IAQ – SO_2_PEH_SUM_IAQ	-3.725[a]	0.000
SO_2_PEH_WIN_IAQ – SO_2_PEH_SUM_IAQ	-3.727[a]	0.000
SO_2_PEH_SPR_IAQ – SO_2_PEH_SUM_IAQ	-3.279[a]	0.000
SO_2_PEH_WIN_IAQ – SO_2_PEH_AUT_IAQ	-0.843[a]	0.399
SO_2_PEH_SPR_IAQ – SO_2_PEH_AUT_IAQ	-0.170[a]	0.865
SO_2_PEH_SPR_IAQ – SO_2_PEH_WIN_IAQ	-1.289[b]	0.197
SO_2_PEH_AUT_OAQ – SO_2_PEH_SUM_OAQ	-2.840[a]	0.005
SO_2_PEH_WIN_OAQ – SO_2_PEH_SUM_OAQ	-2.063[a]	0.039
SO_2_PEH_SPR_OAQ – SO_2_PEH_SUM_OAQ	-2.450[a]	0.014
SO_2_PEH_WIN_OAQ – SO_2_PEH_AUT_OAQ	-0.589[b]	0.556
SO_2_PEH_SPR_OAQ – SO_2_PEH_AUT_OAQ	-0.625[a]	0.532
SO_2_PEH_SPR_OAQ – SO_2_PEH_WIN_OAQ	-1.400[a]	0.162
NO_2_PEH_AUT_IAQ – NO_2_PEH_SUM_IAQ	-0.075[b]	0.940
NO_2_PEH_WIN_IAQ – NO_2_PEH_SUM_IAQ	-1.851[b]	0.064
NO_2_PEH_SPR_IAQ – NO_2_PEH_SUM_IAQ	-2.427[b]	0.015
NO_2_PEH_WIN_IAQ – NO_2_PEH_AUT_IAQ	-2.110[b]	0.035
NO_2_PEH_SPR_IAQ – NO_2_PEH_AUT_IAQ	-2.987[b]	0.003
NO_2_PEH_SPR_IAQ – NO_2_PEH_WIN_IAQ	-1.438[b]	0.151

NO_2_PEH_AUT_OAQ – NO_2_PEH_SUM_OAQ	-0.597[b]	0.550
NO_2_PEH_WIN_OAQ – NO_2_PEH_SUM_OAQ	-2.352[b]	0.019
NO_2_PEH_SPR_OAQ – NO_2_PEH_SUM_OAQ	-1.232[b]	0.218
NO_2_PEH_WIN_OAQ – NO_2_PEH_AUT_OAQ	-2.475[b]	0.013
NO_2_PEH_SPR_OAQ – NO_2_PEH_AUT_OAQ	-0.724[b]	0.469
NO_2_PEH_SPR_OAQ – NO_2_PEH_WIN_OAQ	-1.419[a]	0.156
CO_2_PEH_AUT_IAQ – CO_2_PEH_SUM_IAQ	-1.605[a]	0.108
CO_2_PEH_WIN_IAQ – CO_2_PEH_SUM_IAQ	-2.016[a]	0.044
CO_2_PEH_SPR_IAQ – CO_2_PEH_SUM_IAQ	-3.547[a]	0.000
CO_2_PEH_WIN_IAQ – CO_2_PEH_AUT_IAQ	-0.877[a]	0.380
CO_2_PEH_SPR_IAQ – CO_2_PEH_AUT_IAQ	-3.340[a]	0.001
CO_2_PEH_SPR_IAQ – CO_2_PEH_WIN_IAQ	-2.688[a]	0.007
CO_2_PEH_AUT_OAQ – CO_2_PEH_SUM_OAQ	-3.211[b]	0.001
CO_2_PEH_WIN_OAQ – CO_2_PEH_SUM_OAQ	-3.753[b]	0.000
CO_2_PEH_SPR_OAQ – CO_2_PEH_SUM_OAQ	-2.016[b]	0.044
CO_2_PEH_WIN_OAQ – CO_2_PEH_AUT_OAQ	-3.753[b]	0.000
CO_2_PEH_SPR_OAQ – CO_2_PEH_AUT_OAQ	-1.999[a]	0.046
CO_2_PEH_SPR_OAQ – CO_2_PEH_WIN_OAQ	-3.734[a]	0.000
H_2S_PEH_AUT_IAQ – H_2S_PEH_SUM_IAQ	-3.302[a]	0.001
H_2S_PEH_WIN_IAQ – H_2S_PEH_SUM_IAQ	-3.215[a]	0.001
H_2S_PEH_SPR_IAQ – H_2S_PEH_SUM_IAQ	-2.766[a]	0.006
H_2S_PEH_WIN_IAQ – H_2S_PEH_AUT_IAQ	-1.223[a]	0.221
H_2S_PEH_SPR_IAQ – H_2S_PEH_AUT_IAQ	-1.090[a]	0.276
H_2S_PEH_SPR_IAQ – H_2S_PEH_WIN_IAQ	-0.830[b]	0.406
H_2S_PEH_AUT_OAQ – H_2S_PEH_SUM_OAQ	-2.095[a]	0.036
H_2S_PEH_WIN_OAQ – H_2S_PEH_SUM_OAQ	-1.682[a]	0.093
H_2S_PEH_SPR_OAQ – H_2S_PEH_SUM_OAQ	-3.181[a]	0.001
H_2S_PEH_WIN_OAQ – H_2S_PEH_AUT_OAQ	-0.804[b]	0.422

H_2S_PEH_SPR_OAQ – H_2S_PEH_AUT_OAQ	-2.620[a]	0.009
H_2S_PEH_SPR_OAQ – H_2S_PEH_WIN_OAQ	-2.944[a]	0.003
NMHCs_PEH_AUT_IAQ – NMHCs_PEH_SUM_IAQ	-3.436[b]	0.001
NMHCs_PEH_WIN_IAQ – NMHCs_PEH_SUM_IAQ	-3.932[b]	0.000
NMHCs_PEH_SPR_IAQ – NMHCs_PEH_SUM_IAQ	-3.925[b]	0.000
NMHCs_PEH_WIN_IAQ – NMHCs_PEH_AUT_IAQ	-3.943[b]	0.000
NMHCs_PEH_SPR_IAQ – NMHCs_PEH_AUT_IAQ	-3.934[b]	0.000
NMHCs_PEH_SPR_IAQ – NMHCs_PEH_WIN_IAQ	-3.928[b]	0.000
NMHCs_PEH_AUT_OAQ – NMHCs_PEH_SUM_OAQ	-3.969[b]	0.000
NMHCs_PEH_WIN_OAQ – NMHCs_PEH_SUM_OAQ	-3.998[b]	0.000
NMHCs_PEH_SPR_OAQ – NMHCs_PEH_SUM_OAQ	-3.928[b]	0.000
NMHCs_PEH_WIN_OAQ – NMHCs_PEH_AUT_OAQ	-3.019[b]	0.003
NMHCs_PEH_SPR_OAQ – NMHCs_PEH_AUT_OAQ	-3.938[b]	0.000
NMHCs_PEH_SPR_OAQ – NMHCs_PEH_WIN_OAQ	-3.953[b]	0.000
VOCs_PEH_AUT_IAQ – VOCs_PEH_SUM_IAQ	-3.263[a]	0.001
VOCs_PEH_WIN_IAQ – VOCs_PEH_SUM_IAQ	-3.529[a]	0.000
VOCs_PEH_SPR_IAQ – VOCs_PEH_SUM_IAQ	-2.971[a]	0.003
VOCs_PEH_WIN_IAQ – VOCs_PEH_AUT_IAQ	-2.086[a]	0.037
VOCs_PEH_SPR_IAQ – VOCs_PEH_AUT_IAQ	-0.786[a]	0.432
VOCs_PEH_SPR_IAQ – VOCs_PEH_WIN_IAQ	-1.199[b]	0.231
VOCs_PEH_AUT_OAQ – VOCs_PEH_SUM_OAQ	-0.104[a]	0.917
VOCs_PEH_WIN_OAQ – VOCs_PEH_SUM_OAQ	-3.418[a]	0.001
VOCs_PEH_SPR_OAQ – VOCs_PEH_SUM_OAQ	-3.382[a]	0.001
VOCs_PEH_WIN_OAQ – VOCs_PEH_AUT_OAQ	-3.343[a]	0.001
VOCs_PEH_SPR_OAQ – VOCs_PEH_AUT_OAQ	-2.963[a]	0.003
VOCs_PEH_SPR_OAQ – VOCs_PEH_WIN_OAQ	-1.067[b]	0.286

a. Based on negative ranks.
b. Based on positive ranks.

The same test was carried out to find seasonal differences in both the Department of Biochemistry, Microbiology and Biotechnology and the Department of Biodiversity.

Table A2.5: Model summary for identifying the best predictors (environment, building and season) of air quality per parameter.

Predictors (constant)	Excluded variables*	R	R^2	Adjusted R^2	Standard error of estimate
LAV_IAQ_AA, LAV_PEH_AA, LAV_BMBT_AA, LAV_SUM_AA-LAV_AUT_AA, LAV_WIN_AA, LAV_SPR_AA	LAV_OAQ_AA, LAV_BIOD_AA	1.000	1.000	1.000	0.00025
T_{db}_IAQ_AA, T_{db}_PEH_AA, T_{db}_BIOD_AA, T_{db}_SUM_AA, T_{db}_AUT_AA, T_{db}_WIN_AA, T_{db}_SPR_AA	T_{db}_OAQ_AA, T_{db}_BMBT_AA	1.000	1.000	1.000	0.00038
RH_IAQ_AA, RH_PEH_AA, RH_BMBT_AA, RH_BIOD_AA, RH_SUM_AA, RH_AUT_AA, RH_WIN_AA, RH_SPR_AA	RH_OAQ_AA	1.000	1.000	1.000	0.00028
O_3_IAQ_AA, O_3_OAQ_AA, O_3_PEH_AA, O_3_BMBT_AA, O_3_BIOD_AA, O_3_SUM_AA, O_3_AUT_AA, O_3_WIN_AA, O_3_SPR_AA	None	0.998	0.995	0.991	0.00032
SO_2_IAQ_AA, SO_2_OAQ_AA, SO_2_PEH_AA, SO_2_BMBT_AA, SO_2_BIOD_AA, SO_2_SUM_AA, SO_2_AUT_AA, SO_2_WIN_AA, SO_2_SPR_AA	None	0.999	0.999	0.998	0.00029
NO_2_IAQ_AA, NO_2_PEH_AA, NO_2_BIOD_AA, NO_2_SUM_AA, NO_2_AUT_AA, NO_2_WIN_AA, NO_2_SPR_AA	NO_2_OAQ_AA, NO_2_BMBT_AA	1.000	1.000	1.000	0.00031
CO_IAQ_AA, CO_PEH_AA, CO_BMBT_AA, CO_SUM_AA, CO_AUT_AA, CO_WIN_AA, CO_SPR_AA	CO_OAQ_AA, CO_BIOD_AA	1.000	1.000	1.000	0.00029
CO_2_OAQ_AA, CO_2_BMBT_AA, CO_2_BIOD_AA, CO_2_SUM_AA, CO_2_AUT_AA, CO_2_WIN_AA, CO_2_SPR_AA	CO_2_IAQ_AA, CO_2_PEH_AA	1.000	1.000	1.000	0.00019
H_2S_IAQ_AA, H_2S_OAQ_AA, H_2S_PEH_AA, H_2S_BMBT_AA, H_2S_BIOD_AA, H_2S_SUM_AA, H_2S_AUT_AA, H_2S_WIN_AA, H_2S_SPR_AA	None	1.000	1.000	0.999	0.00016

NMHCs_OAQ_AA, NMHCs_PEH_AA, NMHCs_BIOD_AA, NMHCs_SUM_AA, NMHCs_AUT_AA, NMHCs_WIN_AA, NMHCs_SPR_AA	NMHCs_IAQ_AA, NMHCs_BMBT_AA	1.000	1.000	1.000	0.00046
VOCs_OAQ_AA, VOCs_PEH_AA, VOCs_BMBT_AA, VOCs_BIOD_AA, VOCs_AUT_AA, VOCs_WIN_AA, VOCs_SPR_AA	VOCs_IAQ_AA, VOCs_SUM_AA	1.000	1.000	1.000	0.00036

* Variables were excluded from the model due to partial correlations found.

Table A2.6: Analysis of variance for identifying differences between predictor and outcome variables per parameter.

Dependent Variable	Predictors (constant)	Model	Sum of squares	df	Mean square	F-statistic	p-value
LAV_AA_AQ	LAV_IAQ_AA, LAV_PEH_AA, LAV_BMBT_AA, LAV_SUM_AA, LAV_AUT_AA, LAV_WIN_AA, LAV_SPR_AA	Regression	0.166	6	0.025	3.86 E+5	0.000
		Residual	0.000	12	0.000		
		Total	0.166	19			
RH_AA_AQ	RH_IAQ_AA, RH_PEH_AA, RH_BMBT_AA, RH_BIOD_AA, RH_SUM_AA, RH_AUT_AA, RH_WIN_AA, RH_SPR_AA	Regression	609.086	8	66.136	9.53 E+8	0.000
		Residual	0.000	11	0.000		
		Total	609.086	19			
T_{db}_AA_AQ	T_{db}_IAQ_AA, T_{db}_PEH_AA, T_{db}_BIOD_AA, T_{db}_SUM_AA, T_{db}_AUT_AA, T_{db}_WIN_AA, T_{db}_SPR_AA	Regression	62.653	6	8.965	6.16 E+6	0.000
		Residual	0.000	12	0.000		
		Total	62.653	19			
O_3_AA_AQ	O_3_IAQ_AA, O_3_OAQ_AA, O_3_PEH_AA, O_3_BMBT_AA, O_3_BIOD_AA, O_3_SUM_AA, O_3_AUT_AA, O_3_WIN_AA, O_3_SPR_AA	Regression	0.000	9	0.000	2.28 E+2	0.000
		Residual	0.000	10	0.000		
		Total	0.000	19			
SO_2_AA_AQ	SO_2_IAQ_AA, SO_2_OAQ_AA, SO_2_PEH_AA, SO_2_BMBT_AA, SO_2_BIOD_AA, SO_2_SUM_AA, SO_2_AUT_AA, SO_2_WIN_AA, SO_2_SPR_AA	Regression	0.001	9	0.000	9.39 E+2	0.000
		Residual	0.000	10	0.000		
		Total	0.001	19			
NO_2_AA_AQ	NO_2_IAQ_AA, NO_2_PEH_AA, NO_2_BIOD_AA, NO_2_SUM_AA, NO_2_AUT_AA, NO_2_WIN_AA, NO_2_SPR_AA	Regression	0.008	6	0.001	1.21 E+4	0.000
		Residual	0.000	12	0.000		
		Total	0.008	19			
CO_AA_AQ		Regression	0.036	6	0.005	6.32 E+4	0.000
		Residual	0.000	12	0.000		

	CO_IAQ_AA, CO_PEH_AA, CO_BMBT_AA, CO_SUM_AA, CO_AUT_AA, CO_WIN_AA, CO_SPR_AA	Total	0.036	19			
CO_2_AA_AQ	CO_2_OAQ_AA, CO_2_BMBT_AA, CO_2_BIOD_AA, CO_2_SUM_AA, CO_2_AUT_AA, CO_2_WIN_AA, CO_2_SPR_AA	Regression	5.3 E+3	6	654.946	2.20 E+10	0.000
		Residual	0.000	12	0.000		
		Total	5.3 E+3	19			
H_2S_AA_AQ	H_2S_IAQ_AA, H_2S_OAQ_AA, H_2S_PEH_AA, H_2S_BMBT_AA, H_2S_BIOD_AA, H_2S_SUM_AA, H_2S_AUT_AA, H_2S_WIN_AA, H_2S_SPR_AA	Regression	0.001	9	0.000	4.19 E+3	0.000
		Residual	0.000	10	0.000		
		Total	0.001	19			
NMHC_AA_AQ	NMHCs_OAQ_AA, NMHCs_PEH_AA, NMHCs_BIOD_AA, NMHCs_SUM_AA, NMHCs_AUT_AA, NMHCs_WIN_AA, NMHCs_SPR_AA	Regression	0.162	6	0.023	1.11 E+5	0.000
		Residual	0.000	12	0.000		
		Total	0.162	19			
VOC_AA_AQ	VOCs_OAQ_AA, VOCs_PEH_AA, VOCs_BMBT_AA, VOCs_BIOD_AA, VOCs_AUT_AA, VOCs_WIN_AA, VOCs_SPR_AA	Regression	0.656	6	0.094	6.41 E+5	0.000
		Residual	0.000	12	0.000		
		Total	0.656	19			

Table A2.7: Coefficients of determination for identifying the best predictors of the outcome variable per parameter.

Dependent (Outcome) Variable	Independent (Predictor) variables	Unstandardized B	Coefficients standard error	Standardized Coefficients Beta	T-statistic	p-value
LAV_AA_AQ	Constant	0.000	0.000		0.493	0.631
	LAV_IAQ_AA	-0.002	0.005	0.000	-0.396	0.698
	LAV_PEH_AA	0.000	0.001	0.000	-0.256	0.801
	LAV_BMBT_AA	0.001	0.002	0.001	0.560	0.586
	LAV_SUM_AA	0.251	0.001	0.241	334.998	0.000
	LAV_AUT_AA	0.249	0.001	0.358	209.664	0.000
	LAV_WIN_AA	0.249	0.001	0.339	338.268	0.000
	LAV_SPR_AA	0.251	0.001	0.411	243.803	0.000
RH_AA_AQ	Constant	0.001	0.003		0.519	0.614
	RH_IAQ_AA	5.39 E-5	0.000	0.000	0.628	0.543
	RH_PEH_AA	0.333	0.000	0.253	3.06 E+3	0.000
	RH_BMBT_AA	0.333	0.000	0.369	1.60 E+3	0.000
	RH_BIOD_AA	0.334	0.000	0.393	1.63 E+3	0.000
	RH_SUM_AA	-8.08 E-5	0.000	0.000	-0.590	0.566
	RH_AUT_AA	-6.65 E-5	0.000	0.000	-0.520	0.614
	RH_WIN_AA	-6.68 E-5	0.000	0.000	-0.513	0.618
	RH_SPR_AA	-9.34 E-5	0.000	0.000	-0.603	0.559
T_{db}_AA_AQ	Constant	0.000	0.005		0.083	0.935
	T_{db}_IAQ_AA	2.24 E-5	0.000	0.000	0.056	0.956
	T_{db}_PEH_AA	0.000	0.001	0.000	-0.304	0.666
	T_{db}_BIOD_AA	0.000	0.000	0.000	0.619	0.548
	T_{db}_SUM_AA	0.250	0.000	0.192	658.423	0.000
	T_{db}_AUT_AA	0.250	0.001	0.234	306.608	0.000
	T_{db}_WIN_AA	0.250	0.000	0.262	868.838	0.000

	T_{db}_SPR_AA	0.250	0.000	0.321	626.625	0.000
O_3_AA_AQ	Constant	0.000	0.000		1.396	0.193
	O_3_IAQ_AA	-0.111	0.224	-0.062	-0.496	0.630
	O_3_OAQ_AA	-0.223	0.234	-0.452	-0.953	0.363
	O_3_PEH_AA	0.220	0.206	0.346	1.060	0.314
	O_3_BMBT_AA	0.353	0.219	0.295	1.615	0.136
	O_3_BIOD_AA	0.264	0.182	0.306	1.449	0.168
	O_3_SUM_AA	0.125	0.165	0.180	0.658	0.466
	O_3_AUT_AA	0.180	0.156	0.112	1.155	0.265
	O_3_WIN_AA	0.109	0.131	0.134	0.834	0.424
	O_3_SPR_AA	0.166	0.163	0.336	1.080	0.306
SO_2_AA_AQ	Constant	-1.00 E-5	0.000		-0.032	0.965
	SO_2_IAQ_AA	0.406	0.148	0.296	2.654	0.020
	SO_2_OAQ_AA	0.408	0.141	0.883	2.899	0.016
	SO_2_PEH_AA	-0.096	0.191	-0.124	-0.500	0.628
	SO_2_BMBT_AA	-0.086	0.196	-0.064	-0.438	0.661
	SO_2_BIOD_AA	-0.120	0.182	-0.160	-0.661	0.523
	SO_2_SUM_AA	0.113	0.151	0.259	0.649	0.461
	SO_2_AUT_AA	0.119	0.150	0.151	0.696	0.445
	SO_2_WIN_AA	0.089	0.160	0.062	0.556	0.590
	SO_2_SPR_AA	0.166	0.153	0.192	1.156	0.264
NO_2_AA_AQ	Constant	0.000	0.001		0.312	0.660
	NO_2_IAQ_AA	0.001	0.009	0.000	0.058	0.955
	NO_2_PEH_AA	0.000	0.006	0.000	-0.066	0.941
	NO_2_BIOD_AA	0.000	0.009	0.000	-0.026	0.980
	NO_2_SUM_AA	0.250	0.006	0.260	42.160	0.000
	NO_2_AUT_AA	0.253	0.006	0.396	36.195	0.000
	NO_2_WIN_AA	0.248	0.005	0.391	54.630	0.000

	NO_2_SPR_AA	0.248	0.006	0.252	36.642	0.000
CO_AA_AQ*	Constant	-0.001	0.000		-2.652	0.021
	CO_IAQ_AA	0.016	0.008	0.004	2.006	0.068
	CO_PEH_AA	-0.004	0.003	-0.004	-1.159	0.269
	CO_BMBT_AA	-0.001	0.005	0.000	-0.101	0.921
	CO_SUM_AA	0.249	0.003	0.213	65.354	0.000
	CO_AUT_AA	0.253	0.003	0.253	86.048	0.000
	CO_WIN_AA	0.248	0.003	0.186	66.860	0.000
	CO_SPR_AA	0.251	0.002	0.606	145.034	0.000
CO_2_AA_AQ*	Constant	0.002	0.003		0.641	0.463
	CO_2_OAQ_AA	-6.02 E-6	0.000	0.000	-1.025	0.325
	CO_2_BMBT_AA	-4.66 E-6	0.000	0.000	-1.918	0.069
	CO_2_BIOD_AA	-5.00 E-6	0.000	0.000	-2.065	0.061
	CO_2_SUM_AA	0.250	0.000	0.629	2.20 E+5	0.000
	CO_2_AUT_AA	0.250	0.000	0.316	6.91 E+4	0.000
	CO_2_WIN_AA	0.250	0.000	0.264	9.52 E+4	0.000
	CO_2_SPR_AA	0.250	0.000	0.346	6.50 E+4	0.000
H_2S_AA_AQ	Constant	0.000	0.000		-2.466	0.033
	H_2S_IAQ_AA	-0.041	0.105	-0.048	-0.395	0.601
	H_2S_OAQ_AA	-0.043	0.106	-0.065	-0.400	0.698
	H_2S_PEH_AA	0.186	0.086	0.243	2.166	0.055
	H_2S_BMBT_AA	0.205	0.084	0.328	2.426	0.036
	H_2S_BIOD_AA	0.224	0.090	0.246	2.490	0.032
	H_2S_SUM_AA	0.125	0.056	0.131	2.206	0.052
	H_2S_AUT_AA	0.120	0.056	0.169	2.089	0.063
	H_2S_WIN_AA	0.119	0.058	0.162	2.050	0.068
	H_2S_SPR_AA	0.123	0.058	0.153	2.112	0.061
NMHC_AA_AQ	Constant	-0.001	0.002		-0.324	0.651

	NMHC_OAQ_AA	-0.005	0.008	-0.006	-0.625	0.544
	NMHC_PEH_AA	-0.001	0.011	-0.001	-0.120	0.906
	NMHC_BIOD_AA	-0.001	0.002	-0.002	-0.510	0.619
	NMHC_SUM_AA	0.250	0.001	0.645	413.156	0.000
	NMHC_AUT_AA	0.254	0.006	0.066	34.216	0.000
	NMHC_WIN_AA	0.260	0.016	0.226	16.149	0.000
	NMHC_SPR_AA	0.250	0.002	0.441	109.038	0.000
VOC_AA_AQ	Constant	0.000	0.000		0.614	0.489
	VOC_OAQ_AA	0.006	0.019	0.001	0.388	0.605
	VOC_PEH_AA	0.328	0.021	0.026	15.620	0.000
	VOC_BMBT_AA	0.333	0.000	1.002	1.54 E+3	0.000
	VOC_BIOD_AA	0.340	0.010	0.023	33.901	0.000
	VOC_AUT_AA	-0.004	0.006	0.000	-0.661	0.515
	VOC_WIN_AA	-0.008	0.036	0.000	-0.206	0.840
	VOC_SPR_AA	-0.008	0.011	0.000	-0.699	0.498

* Although variables were excluded due to partial correlation; CO_OAQ_AA ($p = 0.011$) and CO_2_IAQ_AA ($p = 0.002$) were also found to be good predictors of CO_AA_AQ and CO_2_AA_AQ respectively.

APPENDIX B1: ETHICS APPROVAL CERTIFICATES

UNISA university of south africa

UNISA CAES GENERAL ETHICS REVIEW COMMITTEE

Date: 08/09/2017

NHREC Registration # : REC-170616-051
ERC Reference # : 2017/CAES/132
Name : Mr AJ Mundackal
Student #: 61953687

Dear Mr Mundackal

Decision: Ethics Approval from 07/09/2017 to 30/09/2018

Researcher(s): Mr AJ Mundackal
61953687@mylife.unisa.ac.za

Supervisor (s): Prof VM Ngole-Jeme
ngoievm@unisa.ac.za; (011) 471-3878

Working title of research:

Indoor and outdoor air quality: A case of university buildings in the University of Limpopo

Qualification: PhD Environmental Science

Thank you for the application for research ethics clearance by the Unisa CAES General Ethics Review Committee for the above mentioned research. Ethics approval is granted for a one-year period. After one year the researcher is required to submit a progress report, upon which the ethics clearance may be renewed for another year.

Due date for progress report: 30 September 2018

*The **low risk application** was **reviewed** by the CAES General Ethics Review Committee on 07 September 2017 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

The proposed research may now commence with the provisions that:

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Preller Street, Muckleneuk Ridge, City of Tshwane
PO Box 392 UNISA 0003 South Africa
Telephone: +27 12 429 3111 Facsimile: +27 12 429 4150
www.unisa.ac.za

1. The researcher(s) will ensure that the research project adheres to the values and principles expressed in the UNISA Policy on Research Ethics.
2. Any adverse circumstance arising in the undertaking of the research project that is relevant to the ethicality of the study should be communicated in writing to the Committee.
3. The researcher(s) will conduct the study according to the methods and procedures set out in the approved application.
4. Any changes that can affect the study-related risks for the research participants, particularly in terms of assurances made with regards to the protection of participants' privacy and the confidentiality of the data, should be reported to the Committee in writing, accompanied by a progress report.
5. The researcher will ensure that the research project adheres to any applicable national legislation, professional codes of conduct, institutional guidelines and scientific standards relevant to the specific field of study. Adherence to the following South African legislation is important, if applicable: Protection of Personal Information Act, no 4 of 2013, Children's act no 38 of 2005 and the National Health Act, no 61 of 2003.
6. Only de-identified research data may be used for secondary research purposes in future on condition that the research objectives are similar to those of the original research. Secondary use of identifiable human research data require additional ethics clearance.
7. No field work activities may continue after the expiry date. Submission of a completed research ethics progress report will constitute an application for renewal of Ethics Research Committee approval.

Note:

*The reference number **2017/CAES/132** should be clearly indicated on all forms of communication with the intended research participants, as well as with the Committee.*

Yours sincerely,

Prof MA Antwi
Acting Chair of CAES General ERC
E-mail: antwima@unisa.ac.za
Tel: (011) 670-9391

Prof MJ Linington
Executive Dean : CAES
E-mail: lininmj@unisa.ac.za
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UNISA university of south africa

CAES HEALTH RESEARCH ETHICS COMMITTEE

Date: 26/09/2018

NHREC Registration # : REC-170616-051
REC Reference # : 2017/CAES/132
Name : Mr AJ Mundackal
Student #: 61953687

Dear Mr Mundackal

Decision: Ethics Approval Renewal after First review from 01/10/2018 to 30/09/2019

Researcher(s): Mr AJ Mundackal
61953687@mylife.unisa.ac.za

Supervisor (s): Prof VM Ngole-Jeme
ngolevm@unisa.ac.za; (011) 471-3878

Working title of research:

Indoor and outdoor air quality: A case of university buildings in the University of Limpopo

Qualification: PhD Environmental Science

Thank you for the submission of your progress report to the CAES Health Research Ethics Committee for the above mentioned research. Ethics approval is renewed for a one-year period. After one year the researcher is required to submit a progress report, upon which the ethics clearance may be renewed for another year.

Due date for progress report: 30 September 2019

*The **low risk application** was **reviewed** by the CAES Health Research Ethics Committee on 07 September 2017 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

The proposed research may now commence with the provisions that:

1. The researcher(s) will ensure that the research project adheres to the values and principles expressed in the UNISA Policy on Research Ethics.

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2. Any adverse circumstance arising in the undertaking of the research project that is relevant to the ethicality of the study should be communicated in writing to the Committee.
3. The researcher(s) will conduct the study according to the methods and procedures set out in the approved application.
4. Any changes that can affect the study-related risks for the research participants, particularly in terms of assurances made with regards to the protection of participants' privacy and the confidentiality of the data, should be reported to the Committee in writing, accompanied by a progress report.
5. The researcher will ensure that the research project adheres to any applicable national legislation, professional codes of conduct, institutional guidelines and scientific standards relevant to the specific field of study. Adherence to the following South African legislation is important, if applicable: Protection of Personal Information Act, no 4 of 2013; Children's act no 38 of 2005 and the National Health Act, no 61 of 2003.
6. Only de-identified research data may be used for secondary research purposes in future on condition that the research objectives are similar to those of the original research. Secondary use of identifiable human research data require additional ethics clearance.
7. No field work activities may continue after the expiry date. Submission of a completed research ethics progress report will constitute an application for renewal of Ethics Research Committee approval.

Note:
The reference number ***2017/CAES/132*** *should be clearly indicated on all forms of communication with the intended research participants, as well as with the Committee.*

Yours sincerely,

Prof EL Kempen
Chair of CAES Health REC
E-mail: kempeel@unisa.ac.za
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Prof MJ Lihington
Executive Dean : CAES
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APPENDIX B2: PARTICIPANT INFORMATION LEAFLET AND INFORMED CONSENT DOCUMENTS

PARTICIPANT INFORMATION SHEET

20/10/2017

Ethics clearance reference number: 2017/CAES/132

Research permission reference number: REC-170616-051

Dear Prospective Participant

I, Mr. A.J. Mundackal, am doing research with Prof. V.M. Ngole-Jeme, a Professor in the Department of Environmental Sciences and working towards a PhD (Environmental Sciences) at the University of South Africa. We are inviting you to participate in a study entitled: **An Assessment of Indoor and Outdoor Air Quality in a University Environment: A Case of University of Limpopo, South Africa.**

WHAT IS THE PURPOSE OF THE STUDY?

The purpose of this study is to determine the status quo of indoor air quality (IAQ) and outdoor air quality (OAQ) in the University of Limpopo, Mankweng, South Africa, which is presently not known through measurements of the following pollutant concentrations: carbon monoxide, carbon dioxide, ozone, sulphur dioxide, nitrogen dioxide, hydrogen sulphide, non-methyl hydrocarbons, volatile organic compounds, linear air velocity, relative humidity and dry-bulb temperature. The output of this research project will provide potentially useful information on the current IAQ and OAQ status within the University of Limpopo. The study will also evaluate statistical differences between indoor and outdoor AQ in tandem with seasonal differences between indoor and OAQ within the study setting. Additionally, the results will give insight into some of the air quality issues that arise in universities within South Africa and world-wide in similar settings.

WHY AM I BEING INVITED TO PARTICIPATE?

This study aims to ascertain the current AQ status in selected buildings within the University of Limpopo. Indoor and outdoor AQ concentrations levels will be ascertained by the measurement of numerous pollutant concentrations in offices and lecture halls within the School of Molecular and Life Sciences. Results are not going to be attained from you directly, however, the office or lecture hall that you work and lecture in will form

part of this study. I once again reiterate that you as an individual are not being studied upon, but the atmosphere in your offices is of interest in this study. Buildings within the following departments will constitute the study population: the Department of Biochemistry, Microbiology and Biotechnology, the Department of Physiology and Environmental Health and the Department of Biodiversity. All custodians of offices and lecture halls that give consent for measurements to be taken both in the indoor and outdoor environments of the buildings will be included in the study.

WHAT IS THE NATURE OF MY PARTICIPATION IN THIS STUDY?

As an individual, you will not be participating in the study but your office forms part of the indoor atmosphere which is being studied. Measurements of IAQ within your office and lecture halls, and OAQ (outside the building) will being taken. Measurements will be taken between 8am and 4pm, over three consecutive days per season. Consequently, your consistent availability in your office is crucial to the success of this study. The measurements to be taken in each venue should take approximately 15 minutes.

CAN I WITHDRAW FROM THIS STUDY EVEN AFTER HAVING AGREED TO PARTICIPATE?

The participation in this study is voluntary and that there is no penalty or loss of benefit for non-participation. Furthermore, participating in this study is voluntary and you are under no obligation to consent to participation. If you do decide to take part, you will be given this information sheet to keep and be asked to sign a written consent form. You are free to withdraw from this study before commencement of the measurements within the respective buildings, without giving a reason. However, it will be appreciated by the researcher if one does not disembark from the study, on inception of measurement taking in your respective office, since measurements need to be done over three consecutive days to be taken forward to the data analysis section, thereby enhancing the precision of the data obtained.

WHAT ARE THE POTENTIAL BENEFITS OF TAKING PART IN THIS STUDY?

There are no direct potential benefits that one can attain from participating in this study. However, indirectly, a person's well-being and comfort can be deciphered through the results of this study and furthermore, the quality of indoor and outdoor air in the surrounding environment can be ascertained.

ARE THEIR ANY NEGATIVE CONSEQUENCES FOR ME IF I PARTICIPATE IN THE RESEARCH PROJECT?

There are no foreseeable risks, harm or side-effects through participation in this study.

WILL THE INFORMATION THAT I CONVEY TO THE RESEARCHER AND MY IDENTITY BE KEPT CONFIDENTIAL?

Your name will not be recorded anywhere and no one will be able to connect your respective office measurements to you, since, unique identifier codes will be used to ensure the confidentiality and anonymity of the measurements attained. You will be referred to in this way in the data, any publications, or in conference proceedings.

HOW WILL THE RESEARCHER(S) PROTECT THE SECURITY OF DATA?

Hard copies of all the data sheets and observations will be stored by the researcher for a period of five years in a locked cupboard/filing cabinet within the University of Limpopo, Department of Physiology and Environmental Health, old Q-block, first floor, office number 1038. For future research or academic purposes; electronic information will be stored on a password protected computer. Future use of the stored data will be subject to further Research Ethics Review and approval if applicable.

WILL I RECEIVE PAYMENT OR ANY INCENTIVES FOR PARTICIPATING IN THIS STUDY?

There will not be any financial incentive or gain to you by participating in this study. Furthermore you will not incur an expense by participating in this study.

HAS THE STUDY RECEIVED ETHICS APPROVAL

This study has received written approval from the Research Ethics Review Committee of the *College of Agriculture and Environmental Sciences (CAES),* Unisa. A copy of the approval letter can be obtained from the researcher if you so wish.

HOW WILL I BE INFORMED OF THE FINDINGS/RESULTS OF THE RESEARCH?

If you would like to be informed of the final research findings and/or accessibility, please contact Mr. A.J. Mundackal on Tel: (015) 268 3993 or Cell: 082 558 9914 or Email: jino.mundackal@ul.ac.za.

Should you have concerns about the way in which the research has been conducted, you may contact Prof. V.M. Ngole-Jeme on Tel: (011) 471 3878 or Fax: (011) 471 2866 or Email: ngolevm@unisa.co.za

Thank you for taking time to read this information sheet and for participating in this study.

Antony Jino Mundackal

CONSENT TO PARTICIPATE IN THIS STUDY

I, _____ (participant name), confirm that the person asking my consent to take part in this research has told me about the nature, procedure, potential benefits and anticipated inconvenience of participation.

I have read (or had explained to me) and understood the study as explained in the information sheet.

I have had sufficient opportunity to ask questions and am prepared to participate in the study.

I understand that my participation is voluntary and that I am free to withdraw at any time without penalty (if applicable).

I am aware that the findings of this study will be processed into a research report, journal publications and/or conference proceedings, but that my participation will be kept confidential unless otherwise specified.

I agree to the measurements of air quality parameters within my office and/or lecture hall.

I have received a signed copy of the informed consent agreement.
Participant Name & Surname……. (please print)

Participant Signature……Date……

Researcher's Name & Surname: **Mr. Antony Jino Mundackal**

Researcher's signature  Date / /2018

APPENDIX B3: APPROVAL AND LETTERS OF SUPPORT FROM UNIVERSITY PERSONNEL

University of Limpopo
Office of the Registrar
Private Bag X1106, Sovenga, 0727, South Africa
Tel: (015) 268 2407, Fax: (015) 268 3048, Email: Office.Registrar@ul.ac.za

26 April 2018

Mr A.J. Mundackal
Department of Physiology
UNIVERSITY OF LIMPOPO

Email: *jino.mundackal@ul.ac.za*

Dear Mr Mundackal,

GATEKEEPER PERMISSION TO CONDUCT RESEARCH

TITLE	:	**AN ASSESSMENT OF INDOOR AND OUTDOOR AIR QUALITY: A CASE OF UNIVERSITY BUILDINGS IN THE UNIVERSITY OF LIMPOPO, WITHIN THE UNIVERSITY OF LIMPOPO PREMISES".**
SUPERVISOR	:	**Prof Ngole-geme V.M. (UNISA)**
SCHOOL	:	**School of Molecular and Life Sciences**
DEGREE	:	**Doctor of Philosophy**

Kindly be informed that Gatekeeper permission is granted to you to conduct research at the University of Limpopo entitled: "***An Assessment of Indoor and Outdoor Air Quality: A Case of University Buildings in the University of Limpopo, within the University of Limpopo Premises***".

Kind regards,

DR. JEFFREY MABELEBELE
UNIVERSITY REGISTRAR

Cc. Prof. RN Maḍadzhe, Deputy Vice-Chancellor: Teaching and Learning
Dr. T Mabila, Director: Research Development and Administration
Prof. TAB Mashego – Chairperson: Research and Ethics Committee
Ms. N Monene – Office Manager: Research Development and Administration

Finding solutions for Africa

Vice-Chancellor & Principal
Private Bag X1106, Sovenga, 0727, South Africa
Tel : (015) 268 2140/1, Fax : (015) 267 0142, e-Mail : frances.pratt@ul.ac.za
Website : www.ul.ac.za

31 July 2017

TO WHOM IT MAY CONCERN

PERMISSION LETTER TO CONDUCT PhD STUDY ON UNIVERSITY OF LIMPOPO PREMISES

I, Prof. N.M. Mokgalong, Vice-Chancellor & Principal of the University of Limpopo, hereby give permission for Mr. A.J. Mundackal to conduct his PhD study titled *An Assessment of Indoor and Outdoor Air quality : A case of University buildings in the University of Limpopo,* within the University of Limpopo premises.

He will be undertaking the study within the School of Molecular and Life Sciences (SMLS), in the following departments : Department of Biochemistry, Microbiology and Biotechnology (BMBT), Department of Physiology and Environmental Health (PEH) and the Department of Biodiversity (BIOD). The study will commence from December 2017 until June 2019.

Mokgalong

N.M. MOKGALONG (Prof)
VICE-CHANCELLOR & PRINCIPAL

Faculty of Science and Agriculture

UNIVERSITY OF LIMPOPO
TURFLOOP CAMPUS
OFFICE OF THE
DEPUTY VICE CHANCELLOR
TEACHING AND LEARNING
RECEIVED ON: 2017.07.18
FORWARD TO: Prof Siweya
ACTION:

OFFICE OF THE HOD - BIODIVERSITY

INTERNAL MEMO

To : **Prof HJ Siweya**
(Executive Dean, FSA)

From : **Dr MM Matla**
(Acting Director, SMLS)

Date : **12 July 2017**

Subject : **Investigations on air quality in SMLS buildings**

Dear Prof Siweya

1. I have received a letter of recommendation from Dr M van Staden that Mr AJ Mundackal be supported to measure air quality in SMLS offices, store rooms and laboratories as part of his PhD registered with UNISA.
2. I support and appreciate this endeavour to measure the indoor and outdoor air quality in the SMLS spaces.
3. This is to seek your approval for this activity to carry on.

Kind regards,

Dr MM Matla

Prof Madadzhe,
The submission is self-explanatory.
However, your approval is requested.
Thank you,
17/07/2017

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Faculty of Science and Agriculture

OFFICE OF THE HOD - BIODIVERSITY

INTERNAL MEMO

To : **Mr AJ Mundackal (Lecturer, PEH)**

From : **Dr MM Matla (Acting Director, SMLS)**

Date : **11 July 2017**

Subject : **Letter of support**

Dear Mr Mundackal

I support your endeavour to measure the indoor and outdoor air quality at the labs, storerooms and offices of the School of Molecular and Life Sciences. It is highly appreciated.

Kind regards,

Dr MM Matla

INTERNAL MEMO

To	:	**Prof LJ Mampuru:** **Director School of Molecular and Life Science**
From	:	**Dr M van Staden** **HOD Department of Physiology and Environmental Health**
Date	:	**7 July 2017**
Subject	:	**Request: Data collection in University of Limpopo buildings**

The above matter bears reference:

1. Mr Mundackal is a permanent Lecturer in PEH. He is currently registered for a PhD at UNISA. He is planning to measure indoor and outdoor air quality at some of the buildings on campus. At this stage the buildings where air quality will be measured are those that are being used by SMLS as offices, store rooms and some laboratories.
2. Mr Mundackal has already spoken to Mr Molaudzi the Occupational Health and Safety officer and he agreed that the results of these measurements will be valuable for him as well. Mr Mundackal has agreed to submit a report to Mr Molaudzi at the end of his study.
3. I support this endeavour and I will appreciate it if you can forward this request to the necessary officials so that Mr Mundackal can get written approval to do these measurements. He urgently needs this approval so that he can finalise his project proposal for ethical clearance.
4. Please contact me if you need more information.

Kind regards

Dr M van Staden

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INTERNAL MEMO

To: **Mr. AJ Mundackal**
Lecturer: Department of Physiology & Environmental Health

From: **Mr. SW Modubi**
Director: Facilities, Logistics and Transport

Date: **10 July 2017**

Subject: **Permission of surveying the air quality buildings in the School of Molecular & Life Sciences (offices, classrooms, storerooms etc.)**

Dear Mr. Mundackal,

As a stakeholder in the department of Facilities Management responsible with the development of buildings and maintenance, I support your project of measuring indoor and outdoor air quality or some of the buildings in particular to the buildings occupied by School of Molecular and Life Science.

Should you need any support, please contact myself at 2402 or Mr. Ramakadi at 2809.

Regards,

Mr. SW Modubi

UNIVERSITY OF LIMPOPO
APPROVED
TURFLOOP CAMPUS
2017 -07- 10
PRIVATE BAG X1106
SOVENGA 0727
OFFICE OF THE DIRECTOR: FACILITIES, LOGISTICS & TRANSPORT
TEL: 015 268 2404 / FAX: 015 268 2990

Finding solutions for Africa

University of Limpopo
Occupational Health and Safety
Private Bag X1106, Sovenga, 0727, South Africa
Tel: (015) 268 2855, Fax: (015) 3567, Email: tshepo.molaudzi@ul.ac.za

INTERNAL MEMORANDUM

TO : **Mr AJ Mundackal**
Lecturer: Department of Physiology and Environmental Health

FROM : **Mr TA Molaudzi**
Occupational Health and Safety

DATE : **11 July 2017**

SUBJECT : **Support Letter**

The purpose of this letter is to support the request by Mr AJ Mundackal to conduct an indoor and outdoor air quality study assessment at some of the buildings within the University of Limpopo.

I believe this study will provide baseline readings of indoor and outdoor air quality status. It will further assist in determining whether the air quality levels on the University of Limpopo campus meet the recommended levels according to law.

The office of the Occupational Health and Safety officer supports this study fully and commends the initiative.

Yours faithfully,

Tshepo Molaudzi
Occupational Health and Safety Officer
015 268 2855
082 937 2107

Finding solutions for Africa

APPENDIX C: CALIBRATION CERTIFICATES OF INSTRUMENTATION

Technology Solutions
Measurement Science Laboratory

Certificate of Calibration

The South African National Accreditation System (SANAS) is a member of the International Laboratory Accreditation Cooperation (ILAC) Mutual Recognition Agreement (MRA). This arrangement allows for the mutual recognition of technical test and calibration data by the member accreditation bodies worldwide. For more information on the arrangement please consult www.ilac.org. The accuracies of all measurements were traceable to the SI (International System of Units) through NIST, NMISA, PTB or International Measuring Standards, unless otherwise noted. The uncertainties of measurement were estimated for a coverage factor of k=2 which approximates to a 95% confidence level.

American Standard Calibration Laboratory
Measurement Science Laboratory

Certificate No	L63911
Manufacturer	TSI Alnor
Description	Velometer
Model No	AVM410
Serial No	AVM411146006
Plant No	313093
Calibrated for	University of Limpopo
	Private Bag X1106, Sovenga, 0727
Temperature	(17.6 to 17.8) °C ± 2 °C
Relative humidity	(48 to 48) % rh ± 5 % rh
Date of calibration	01 July 2016

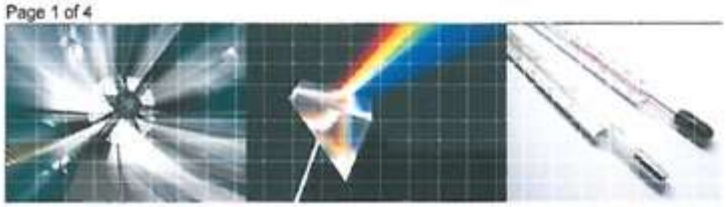
Issue Date 04 July 2016

Calibrated by *Ilze-Marie Jooste* Digitally signed by Ilze-Marie Jooste
Date: 2016.07.04 10:35:16 +02'00'

This certificate is issued without alteration, and in accordance with the conditions of accreditation granted by SANAS. Copyright of this certificate is owned by Technology Solutions & American Standard Calibration Laboratory and may not be reproduced other than in full, except with the prior written approval. It is a correct record of the measurements performed at the time of calibration.Subsequently the accuracy will depend on factors such as care exercised in handling the instrument and frequency of use. Recalibration should be performed after a period which has been chosen to ensure that, under normal circumstances, the instruments accuracy remains within the desired limits.The results relate to the device under calibration.

Technical Signatory *Ilze-Marie Jooste* Digitally signed by Ilze-Marie Jooste
Date: 2016.07.04 10:35:37 +02'00'

Page 1 of 4



Technology Solutions
Prospect Close, 311 Regency Drive,
R21 Corporate Park, Irene
Tel: +27 (0) 12 345 5358
Fax +27 (0) 12 345 3263
www.technologysolutions.co.za

CERTIFICATE OF CALIBRATION AND TESTING

TSI Instruments Ltd, Stirling Road, Cressex Business Park
High Wycombe Bucks HP12 3ST England
Tel: (Int +44) (UK 0) 1494 459200 Fax: (Int +44) (UK 0) 1494 459700 http://www.tsiinc.co.uk

ENVIRONMENT CONDITIONS		
TEMPERATURE	22.6	°C
RELATIVE HUMIDITY	47.04	%RH
BAROMETRIC PRESSURE	996.3	hPa

MODEL	7575-X
SERIAL NUMBER	7575X1318008

☒ AS LEFT ☐ AS FOUND
☒ IN TOLERANCE ☐ OUT OF TOLERANCE

– CALIBRATION VERIFICATION RESULTS –

THERMO COUPLE — SYSTEM PRESSURE02-02 — *Unit: °C*

#	STANDARD	MEASURED	ALLOWABLE RANGE	#	STANDARD	MEASURED	ALLOWABLE RANGE
1	20.9	20.9	19.8~22.0				

BAROMETRIC PRESSURE — SYSTEM PRESSURE02-02 — *Unit: hPa*

#	STANDARD	MEASURED	ALLOWABLE RANGE	#	STANDARD	MEASURED	ALLOWABLE RANGE
1	998.6	998.6	978.7~1018.6				

TSI does hereby certify that the above described instrument conforms to the original manufacturer's specification (not applicable to As Found data) and has been calibrated using standards whose accuracies are traceable to members of the European co-operation for Accreditation (EA) (for example: UKAS, SWEDAC, DAkkS) or has been verified with respect to instrumentation whose accuracy is traceable to some member of EA, or is derived from accepted values of physical constants. TSI's calibration system is registered to ISO-9001:2008.

Measurement Variable	System ID	Last Cal.	Cal. Due	Measurement Variable	System ID	Last Cal.	Cal. Due
Pressure	E006187	13-05-16	13-05-17	DC Volts	E006185	12-05-16	12-05-17
Temperature	E006134	03-02-16	03-02-17	Pressure	E006193	10-06-16	10-06-17

S Waterton
CALIBRATED

27 July, 2016
DATE

Doc. ID: CERT_GEN_V/CC

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 20580

Calibration Date: 20 Oct 2017 17:20

Model: Ozone Low 0-0.5 ppm

Serial No: OZLC1703200-235

Environmental Conditions

Temperature 27.6 °C

Relative Humidity 40.0 %

Measurements

Calibration Standard /ppm	0.001	0.108	0.467	0.000
AQL Sensor (Mean) /ppm	0.000	0.109	0.463	0.000
AQL Sensor (Std. Dev) /ppm	0.001	0.001	0.003	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

The Aeroqual ozone sensors are calibrated in a controlled environment against a NATA certified ozone UV photometer whose traceability is maintained with international standards organisations. *NATA is Australia's national body for accreditation of producers of certified reference materials.

QC Approval: Farid Yanes

Date: 20 Oct 2017

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 20240

Calibration Date: 14 Sep 2017 13:42

Model: Sulphur Dioxide 0-10 ppm

Serial No: ESO-1409174-007

Environmental Conditions

Temperature 23.5 °C

Relative Humidity 29.6 %

Measurements

Calibration Standard /ppm	0.00	6.19	0.00	0.00
AQL Sensor (Mean) /ppm	0.00	6.19	0.00	0.00
AQL Sensor (Std. Dev) /ppm	0.000	0.010	0.000	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

The Aeroqual sensor is calibrated against a certified UV fluorescence analyser.

QC Approval: Farid Yanes

Date: 14 Sep 2017

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 20342

Calibration Date: 21 Sep 2017 12:00

Model: Nitrogen dioxide 0-1 ppm

Serial No: ENW-0809171-016

Environmental Conditions

Temperature 25.5 °C

Relative Humidity 36.7 %

Measurements

Calibration Standard /ppm	0.000	0.715	0.000	0.000
AQL Sensor (Mean) /ppm	0.000	0.699	0.000	0.000
AQL Sensor (Std. Dev) /ppm	0.000	0.010	0.000	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

The Aeroqual sensor is calibrated against a chemiluminescence NOx analyser. The calibration of this analyser is traceable to primary standards.

QC Approval: **Farid Yanes**

Date: **21 Sep 2017**

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 19936

Calibration Date: 18 Aug 2017 13:49

Model: Hydrogen Sulphide 0-10 ppm

Serial No: EHS-1808174-003

Environmental Conditions

Temperature 23.5 °C

Relative Humidity 31.2 %

Measurements

Calibration Standard /ppm	0.00	6.74	0.00	0.00
AQL Sensor (Mean) /ppm	0.00	6.74	0.00	0.00
AQL Sensor (Std. Dev) /ppm	0.000	0.002	0.000	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

The Aeroqual sensor is calibrated against a certified UV fluorescence analyser.

QC Approval: Farid Yanes

Date: 18 Aug 2017

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 20255

Calibration Date: 15 Sep 2017 11:49

Model: NMHC 0-25 ppm

Serial No: VN--3006141-043

Environmental Conditions

Temperature 23.5 ℃

Relative Humidity 37.8 %

Measurements

Calibration Standard /ppm	0.0	10.0	20.0	0.0
AQL Sensor (Mean) /ppm	0.0	10.4	21.4	0.0
AQL Sensor (Std. Dev) /ppm	0.000	0.115	0.116	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

This sensor was calibrated against a certified mixture of isobutene in synthetic air diluted with zero air using mass flow controllers with calibrations traceable to the National Institute of Standards and Technology (NIST).

QC Approval: **Farid Yanes**

Date: **15 Sep 2017**

aeroqual

Aeroqual Limited
109 Valley Road, Mount Eden, Auckland, New Zealand
Phone: +64-9-623 3013 Fax: +64-9-623 3012
www.aeroqual.com

Calibration Certificate No. 19984

Calibration Date: 23 Aug 2017 16:24

Model: VOC PID 0-20 ppm

Serial No: PDL-1708171-008

Environmental Conditions

Temperature 24.5 °C

Relative Humidity 33.2 %

Measurements

Calibration Standard /ppm	0.00	20.00	0.00	0.00
AQL Sensor (Mean) /ppm	0.00	19.99	0.00	0.00
AQL Sensor (Std. Dev) /ppm	0.000	0.013	0.000	0.000

*The Mean and Standard Deviation are calculated from three consecutive readings.

Calibration Standard

This sensor was calibrated against a certified mixture of isobutene in synthetic air diluted with zero air using mass flow controllers with calibrations traceable to the National Institute of Standards and Technology (NIST).

QC Approval: Farid Yanes

Date: 23 Aug 2017

APPENDIX D: PUBLICATIONS

Mundackal A.J. and Ngole-Jeme V.M. 2020. Evaluation of indoor and outdoor air quality in university academic buildings and associated health risk. *International Journal of Environmental Health Research*, DOI: 10.1080/09603123.2020.1828304.

Makgoba L. 2018. Indoor air quality measurements across two seasons in the Department of Physiology and Environmental Health, University of Limpopo, South Africa. *Honours Mini-dissertation*. University of Limpopo, Turfloop Campus, Mankweng South Africa. (Supervisor: Mr. A.J. Mundackal; Co-Supervisor: Ms. K. Linde).

Mathabatha T.B. 2018. A survey of outdoor air quality at the old Q-bock building within the University of Limpopo. *Honours Mini-dissertation*. University of Limpopo, Turfloop Campus, Mankweng South Africa. (Supervisor: Mr. A.J. Mundackal; Co-Supervisor: Ms. K. Linde).

APPENDIX E: THESIS PROOFREADING AND EDITING CONFIRMATION LETTER

University of Limpopo, Faculty of Humanities
School of Languages and Communication Studies

Department of Linguistics, Translation, and Interpreting

Private Bag X1106, Sovenga, 0727, South Africa

Tel: (015) 268 2610, Email: Abram.mashatole@ul.ac.za

'Confirmation of thesis editing and proof-reading'

This serves to confirm that I have edited and proof-read the thesis titled 'An assessment of indoor and outdoor air quality in a university environment' by Mr. Mundackal, AJ. The thesis is ready for submission.

Yours Sincerely

Mr. M.A Mashatole, *BA, MA (UL), PhD Candidate (Wits)*

Lecturer (School of Languages and Communication Studies)

Department of Linguistics, Translation, and Interpreting

New K-Block, Office 7013

Abram.mashatole@ul.ac.za/ 0713030739

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