

# **RE-ENGINEERING OF EXISTING WASTEWATER STABILISATION PONDS FOR IMPROVED CONTAMINANT REMOVAL**

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Bloemfontein, South Africa  
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- (a) *Winner (July 2023)*: Institutional (CUT) Southern-African Nordic Centre (SANORD) Three Minute Thesis (3MT) Competition, Bloemfontein, RSA. [Weblink](#)
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- plants: A case study of the Free State Province, South Africa. *South Africa-Sweden University Forum (SASUF) Sustainability Forum*, 15–17 May 2024. Swedish University of Agricultural Sciences, Malmö and Lund Universities, Sweden.
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## ABSTRACT

Wastewater treatment plants (WWTPs) are critical in resource recovery, recharging surface waters, and recycling essential nutrients. WWTPs also significantly reduce pollutant loads, which are sometimes marred by operational inefficiencies leading to inadequately treated wastewater effluent. Therefore, this research attempts to solve some of the above-mentioned problems by means of the re-engineering of wastewater stabilisation ponds (WSPs) for improved contaminant removal. Firstly, the wastewater influent and effluent samples from the Bainsvlei, Botshabelo, North-East, and Sterkwater WWTPs located in the Mangaung Metropolitan Municipality, Free State province, South Africa were obtained and analysed for a 3-year period. Thereafter, in-situ sampling at effluent discharge points, i.e., Bloem Spruit, Klein Modder River, and Renoster Spruit, was conducted to establish the associated impact on the environment and public health. Finally, the re-engineered WSPs were modelled and simulated for four distinctive design configurations using the Hydromantis extended simulation software tool (GPS-X 8.5). The monthly wastewater influent and effluent samples at all the WWTPs failed to comply with the South African National Standards (SANS) pertained to the Chemical Oxygen Demand ( $\text{COD} \leq 75 \text{ mg}/\ell$ ). For the period under consideration, the average COD concentrations at the four WWTPs varied between  $107.6 \text{ mg}/\ell$  and  $141.8 \text{ mg}/\ell$ . Further investigations revealed that the ammonia ( $\text{NH}_3\text{-N}$ ) levels for all the WWTPs are also higher than the specified national standard which resulted in lower levels of Dissolved Oxygen (DO) at the three in-situ sampling/ effluent discharge points. A DO deficiency typically impacts negatively on the survival of aquatic species, while aggravated (lower) levels will eventually lead to eutrophic waters and increasing treatment costs. Subsequently, the four re-engineered WSP design configurations were evaluated in terms of the Volatile Suspended Solids (VSS), Total Suspended Solids (TSS), and organics and nutrient (ammonia, nitrogen, and phosphate) removal efficiencies. The results demonstrated that Scenario 3 produced higher TSS values compared to the other three design scenarios to subsequently result in the highest VSS/TSS ratio  $\geq 0.8$ , which reflects an efficient removal rate of organics and nutrients. Hence, WSPs can be successfully re-engineered by changing design parameters (e.g., flow rate, number of baffles, the arrangement of ponds, and the type of ponds) to meet the SANS criteria for wastewater effluent.

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## LIST OF ABBREVIATIONS

ARB	Antibiotic-Resistant Bacteria
ARG	Antibiotic-Resistant Genes
ARV	Antiretroviral
AST	Activated Sludge Treatment
BOD	Biochemical Oxygen Demand
CEC	Contaminants of Emerging Concern
CFD	Computational Fluid Dynamics
COD	Chemical Oxygen Demand
CRR	Critical Risk Rating
CSTR	Continuous Stirred Tank Reactor
DO	Dissolved Oxygen
DWS	Department of Water and Sanitation
<i>E. coli</i>	<i>Escherichia coli</i>
ECs	Emerging Contaminants
FS	Free State
GDS	Green Drop Score
GHG	Greenhouse Gas
HESS	Hydromantis Environmental Software Solutions
HRT	Hydraulic Retention Time
MAR	Mean Annual Runoff
MLSS	Mixed Liquor Suspended Solids
MMM	Mangaung Metropolitan Municipality
OA	Oxygen Absorbed
PAR	Photosynthetic Active Radiation
PPD	Percentage Population Distribution
SANS	South African National Standards
SDG	Sustainable Development Goal
SS	Suspended Solids
SSV	Settleable Solids
SVI	Sludge Volume Index
SW	Surface Water
TDS	Total Dissolved Solids

TSS	Total Suspended Solids
UCT	University of Cape Town
UN SDGs	United Nations Sustainable Development Goals
USA	United States of America
<i>V. radiata</i>	<i>Vigna radiata</i>
VSS	Volatile Suspended Solids
WHO	World Health Organisation
WMA	Water Management Area
WSP	Wastewater Stabilisation Pond
WWTT	Wastewater Treatment Technology
WWTP	Wastewater Treatment Plant

## CHAPTER 1 : INTRODUCTION

This chapter provides some background information on the problems associated with water scarcity and pollution by focusing on the prevention and reduction of the widespread pollution of this valuable resource by means of conventional and advanced wastewater treatment technologies (WWTTs).

The rationale, problem statement, research aim, objectives, questions, assumptions, scope and limitations, and an outline of the thesis structure are presented as follows.

### 1.1 Rationale

Freshwater is an essential compound required for sustaining all life on Earth. Although water scarcity is a problem, its severity has been increasing in South Africa. Miller et al. (2020) reported that water shortages experienced in some parts of South Africa in the spring of 2019 were severe. This was regarded as one of the driest seasons in over ten years. Water scarcity and its impact on the region were also reported by Apraku et al. (2025). This has initiated debates on water security, as reported by Esan et al. (2024). Donnenfeld et al. (2018) argued that water security policies have not been achieved primarily owing to water use practices and the increasing population and associated pollution of this valuable natural resource. Hence, the increasing population of South Africa and the need for improved public health care demand an innovative approach towards access to water. According to Gruchlik et al. (2018), treating wastewater using conventional or advanced wastewater treatment plants (WWTPs) or both in many parts of the world is an effective water management strategy, while wastewater stabilisation ponds (WSPs) are considered a more cost-effective alternative.

The global human population is estimated to be over 7 billion, while recent studies indicated it could reach 9.7 billion and beyond by 2050 (UN DESA, 2019). The teeming global population would still rely on the dwindling and deteriorating surface waters for sustenance. Climate change and associated environmental distress syndromes contribute to the further deterioration of available freshwater resources, while the advent of novel chemicals used in emerging technologies has

exacerbated water quality issues. Owing to the projected high population growth and deteriorating freshwater resources, ensuring the global population has quality freshwater is vital (Lipponen and Nikiforova, 2017).

Some countries are highly water-stressed, particularly the Middle East, Australia, and Southern Africa, among others. Estimates showed that South Africa's population will have doubled by the year 2050; hence, improved WWTTs will be required to contribute towards sustainability, while WSPs will become prevalent techniques employed to remove pathogens and noxious agents from wastewater for non-potable use (Ouedraogo et al., 2016; Passos et al., 2019; Ragush et al., 2015).

## 1.2 Problem Statement

The current WSP technologies and infrastructure in South Africa and internationally are either outdated or old (Jack et al., 2006; Vagheei, 2021). WSPs are also normally associated with more frequent environmental and health concerns, e.g., greenhouse gas (GHG) emissions and poor residual antibiotics removal efficiencies (Ho et al., 2017). However, the GHG emissions data from WSPs are not readily available and normally limited to site-specific observations. In addition, processes such as the hydraulic flow patterns; chemical, biochemical, and mechanical processes; and sludge dynamics for a full-scale WSP system are not well understood or managed. Typically, the management of WSP sludge is regarded as problematic; ranging from the incineration of bio-solids and landfilling to open sea dumping which is now banned worldwide (Dentel and Qi, 2014).

In terms of WSP hydraulics, information on how baffles can be utilised to improve pond design is lacking (Shilton and Mara, 2005). The computational fluid dynamics (CFD) modelling widely used in WSP simulation is usually limited to a first-order decay kinetics model. The major flaw of first-order decay kinetics is embedded in the oversimplification and assumptions of uniform volume for a WSP. In addition, information on the interplay between the hydraulic flow patterns, ecology, and sludge dynamics is not readily available or well established for a full-scale WSP

(Nameche and Vasel, 1998; Abbas et al., 2006; Glaz et al., 2016; Coggins et al., 2019).

Given that WSPs are inexpensive alternative methods for treating wastewater globally, understanding the problems mentioned above, which are associated with existing WSPs, is crucial. Hence, the re-engineering of WSPs to cater for these factors is of critical importance to maximise the efficiency of WSPs towards meeting present and emerging WWTT challenges.

### 1.3 Research Aim and Objectives

The research aim is to re-engineer existing WSPs for improved contaminant removal efficiency by providing a detailed solution for the interplay among the hydraulic, chemical, biochemical, and mechanical processes involved. Subsequently, the latter solution will provide: (i) a better understanding of the hydrodynamics and biochemical processes of a WSP, (ii) guidelines for the efficient removal of contaminants using the re-engineered WSP configurations to ensure a prolonged lifespan, and (iii) clarity on the sludge distribution as a function of the inlet/outlet configurations, flow rates, wind effects, and pond geometry.

The specific objectives of the research are to:

- (a) establish the influence of influent and effluent physicochemical parameters in diagnosing and addressing operational challenges at WWTPs;
- (b) develop a cost-effective approach to sample and determine the various physicochemical parameters in (a) by considering the United Nations Sustainable Developmental Goals (UN SDGs), including SDG 3 (good health and well-being), SDG 6 (clean water and sanitation), and SDG 14 (life below water);
- (c) investigate and compare the effluent quality compliance at WWTPs with the South African National Standards (SANS) criteria for wastewater effluent;
- (d) investigate the potential environmental and health impacts of the wastewater effluent discharged into surface water bodies within the

Mangaung Metropolitan Municipality (MMM) in the Free State (FS) province, South Africa;

- (e) explore sludge distribution in a WSP system by considering different baffle configurations, flow rates, and/or pond geometry; and
- (f) explore the design and simulation of a re-engineered WSP for improved contaminant removal by comparing results with effluent originating from conventional WWTPs.

#### **1.4 Research Questions**

The following research questions were deemed necessary:

- (a) How quickly do changes in influent characteristics translate into changes in effluent quality, and how can this information be used to implement more responsive control strategies?
- (b) Are there specific periods or seasons when effluent non-compliance is more prevalent?
- (c) How does the wastewater effluent discharge affect the dissolved oxygen (DO), pH, turbidity, and temperature levels in the receiving surface water body?
- (d) How does the surface-area-to-volume ratio of a WSP influence the wastewater treatment efficiency?
- (e) How do the simulated contaminant removal efficiencies of the re-engineered WSPs compare to the measured effluent quality discharged from the existing WWTPs?

#### **1.5 Research Assumptions**

During the design and simulation of the re-engineered WSPs, it was assumed that meteorological parameters do not impact the ability of WSPs to remove contaminants effectively. Furthermore, it was also envisaged that the research findings will enable WWTPs to treat wastewater more efficiently, while reducing power consumption as the equipment used in conventional WWTPs is power intensive.

## 1.6 Scope and Limitations

The scope of this research is limited to exploring the possibilities of integrating re-engineered WSPs into existing WWTPs in the FS province, South Africa. Influent and effluent from four WWTPs in the MMM were sampled and downstream in-situ surface water tests were conducted at suitable river sites. To date, no guidelines are available in scientific literature which clearly specify the maximum and/or minimum number of WWTPs to be sampled in any study. For example, Masindi et al (2022) sampled two WTTTPs, while Mugudamani et al. (2024), sampled four WWTPs. Therefore, the decision to sample four WWTPs in this study is deemed sufficient. The Hydromantis extended simulation software tool (GPS-X 8.5) was used to design and simulate various WSP configurations based on the effluent data from the four WWTPs. A steady state configuration was adopted, and the influence of meteorological parameters and environmental conditions was assumed to be minimal. The primary limitations of this research are ascribed to limited funding, and the unavailability of information and observed data regarding the interplay among the hydraulic flow patterns, ecology, and sludge dynamics for a full-scale WSP.

## 1.7 Outline of Thesis Structure

Chapter 2 presents the literature review focusing on different WWTTs, wastewater research in South Africa, the status quo of WWTPs in the FS province, South Africa, as well as the environmental and health impacts associated with poor wastewater management strategies. Chapter 3 provides an overview and general description of the study area, relevant research materials, and the methodology adopted. The results pertaining to the monitoring and analyses of influent and effluent parameters at conventional WWTPs, the impact of effluent non-compliance, and the design and simulation of re-engineered WSPs are included and discussed in Chapters 4 to 6. Chapter 7 presents a synthesis of all the information as discussed in Chapters 1 to 6, as well as some final conclusions and recommendations. Chapter 8 contains the detailed reference list.

## CHAPTER 2 : LITERATURE REVIEW

Water is an essential element for life on Earth. All biota depend on its unique physicochemical characteristics for biochemical metabolism. Although some biota thrive in salty waters, humans rely on the scarce freshwater (surface and groundwater) for survival. According to Qadri and Bhat (2020), the available global freshwater is less than one per cent (<1%) of the total water bodies (75%) on Earth. Freshwater is not equally distributed across the globe as scarcity has been noted in various parts of Africa, China, Europe, and South America (Mekonnen and Hoekstra, 2016; Duan and Chen, 2017; Donnenfeld et al., 2018; Pulvento et al., 2021; Salehi, 2022). It should be noted that water scarcity does not only affect humans, as aquatic species are also lost in its absence (Liu et al., 2018).

Over the years, South Africa has witnessed severe water shortages. The water crisis in South Africa is attributed to drought due to climate change and various anthropogenic activities. The report by the Department of Water and Sanitation (DWS) (2018) predicted a water deficit of 17% by 2030 if decisive measures are not taken. Another report by Miller et al. (2020) projects an increase in temperature in the Northern and Southern sub-regions, exacerbated by anthropogenic activities. Furthermore, Miller et al. (2020) contend that these activities led to drought in the region between 2015 and 2016. Some steps utilised in tackling water shortages are reusing treated wastewater, harvesting rainwater, and drilling. Interestingly, in South Africa, water scarcity is due to ageing infrastructure, with 37 to 50% of unaccounted water losses being due to leakages (Tshona et al., 2025).

Unfortunately, various sources contribute to the pollution of surface and groundwaters. For example, the agricultural sector contributes to surface water pollution when excess fertilisers not absorbed by the crops are leached (Lipponen and Nikiforova, 2017). The leaching of fertiliser into surface water leads to the depletion of oxygen, thereby promoting algal growth (Yi, Hur et al., 2009; Yi, Kim et al., 2009; Amengual-Morro et al., 2012; Ragush et al., 2015; Bassin et al., 2021). Additionally, poor-quality effluent from WWTPs was

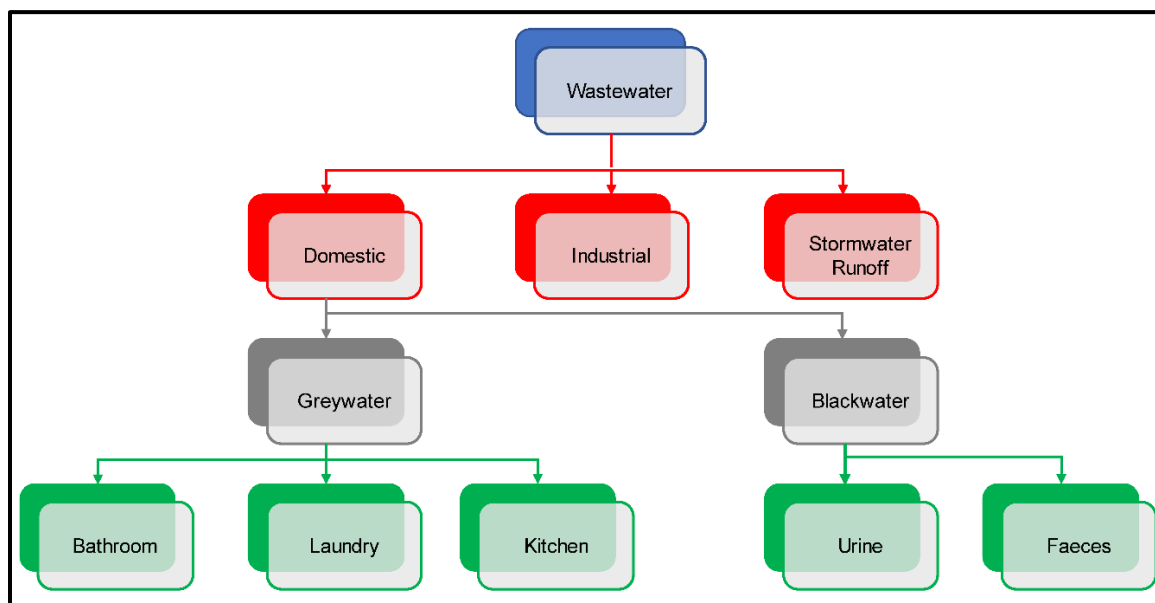
highlighted as a contributor to contaminating surface waters globally (Lu et al., 2022; MacEdo et al., 2022; Trommetter et al., 2022; Gwenzi et al., 2024).

Al-Jasser (2011) analysed samples from six WWTPs in Saudi Arabia over 10 months, using physicochemical and biological methods. The findings confirmed that some WWTPs do not meet the recommended effluent standard. Zhou et al. (2022) provided essential information for improving treatment procedures in WWTPs in China. Gwenzi et al. (2024) showed that wastewater effluent contaminates water bodies with environmental and public health concerns, e.g., infection, illness, and death. Contamination of surface waters in South Africa was reported by Masindi et al. (2022), all of which was primarily attributed to non-compliance with the SANS criteria for wastewater effluent. In the same light, the DWS (2022a) warned that most WWTPs in South Africa are in a dire situation. The study by Edokpayi et al. (2017b) showed that deficiencies in design, the absence of technical expertise, instances of corruption, hydraulically overloaded treatment plants, and inadequate funding plague most WWTPs in South Africa. Owing to insufficient influent treatment, poor-quality effluents have been discharged into waterbodies in South Africa, resulting in various waterborne diseases such as cholera, dysentery, and typhoid (Sibanda et al., 2015).

Hence, this chapter aims to review the potential impact of WWTPs on the contamination of surface and groundwater sources to highlight the degree of pollution across multiple water matrices in various continents, narrowed down focusing specifically on Africa, Southern Africa, South Africa and the case study area (FS province). Furthermore, a risk assessment was conducted to establish the environmental and health implications of water pollution to enable future monitoring programmes to account for pollutants in unexpectedly higher quantities from WWTP effluents. The review period under consideration is from 2013 to 2024 and coincides with the period since the last national report on the state of WWTPs.

## 2.1 Wastewater Characteristics and Sources

Wastewater contains organic and inorganic constituents (Rout et al., 2021; Saleh et al., 2022). Wastewater is either grey or black water originating from domestic, industrial, and other sources, including stormwater and runoff from agricultural lands. A graphical description of the constituents and origin of wastewater is presented in Figure 2.1.



**Figure 2.1:** Sources and constituents of wastewater (Edokpayi et al., 2017b)

The depiction of wastewater in Figure 2.1 is debatable, given that emerging contaminants (ECs) have recently been detected in wastewater streams globally (Rimayi et al., 2018; Rout et al., 2021; MacEdo et al., 2022). ECs are categorised as endocrine-disrupting compounds, pharmaceuticals, illicit drugs, non-controlled sweeteners, personal care products, and pesticide compounds. ECs were found in wastewater effluents sampled across North America, Europe, and Asia (Shah et al., 2020). Shah et al. (2020) also noted that the inability of the current WWTPs to treat these contaminants poses a severe challenge as the effluents are discharged into water bodies.

In South Africa, the University of Cape Town (UCT) developed an effective wastewater treatment process to reduce biological nutrients such as nitrogen and phosphorus (Motlagh and Goel, 2014); however, it does not cater for the removal

of ECs. In the same context, most WWTPs in South Africa experience problems with ECs as reported in Asia, Europe, and North America. Rimayi et al. (2018) found traces of antiretroviral (ARV) drugs in potable water and groundwater in the Gauteng province, South Africa, although, their impact on the populace is unknown. Despite South Africa being the most developed country on the continent, Gani et al. (2021) found that the incidence and distribution of ECs in the aquatic environment are limited in South Africa. Although South Africa has strict environmental regulations in place, environmental compliance, enforcement, and monitoring programmes are still regarded as insufficient (DEA, 2012; Rimayi et al., 2018). Reported illnesses related to the use of contaminated water and wastewater include the following:

- (a) Gastroenteritis (Sabar et al., 2022);
- (b) Amoebiasis (Archer et al., 2017; Edokpayi et al., 2017b);
- (c) Salmonellosis (Edokpayi et al., 2017a); and
- (d) Dysentery, cholera, typhoid fever, hepatitis A, and diarrhoea (Cowan et al., 2016; Verlicchi and Grillini, 2020; Bhat et al., 2022; Salehi, 2022).

The most recent occurrence is the pollution of the famous Durban coastline by wastewater, costing the Kwazulu-Natal province and the tourism sector colossal revenue losses. Hence, the distinct types of contaminants from WWTP effluents identified in surface waters in South Africa are presented in Table 2.1.

**Table 2.1:** Summary of WWTP ECs identified in South African waterbodies

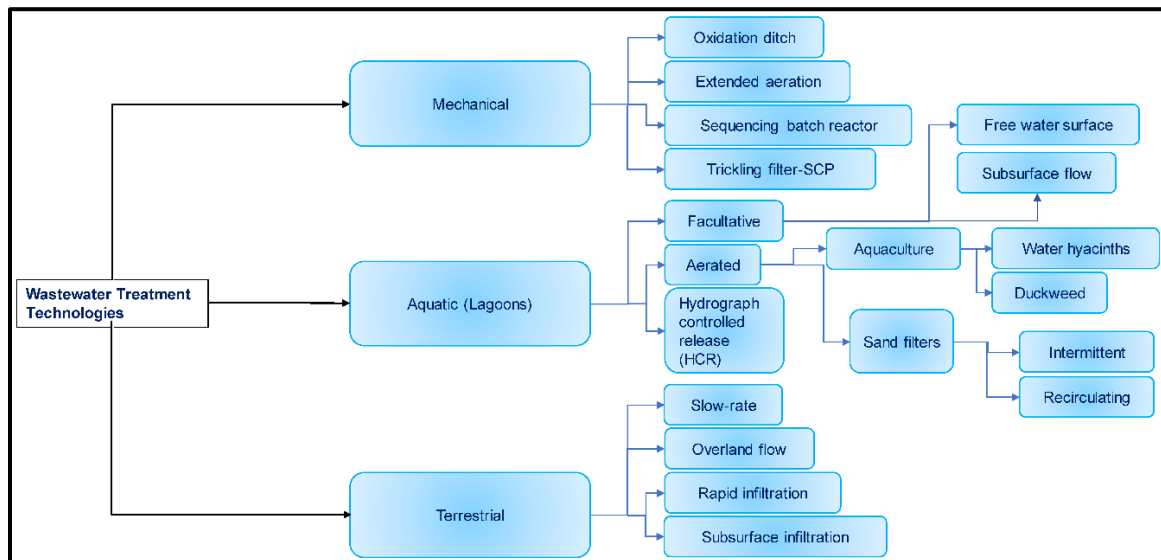
Contaminants	Pollution Source	Impact	Location	References
Ibuprofen	Surface water	Human health and wildlife ecosystem	Free State and Gauteng provinces	Archer et al. (2017) and Mhuka et al. (2020)
<i>Escherichia coli</i> ( <i>E. coli</i> )	Surface and wastewater	Intestinal and epidemiological diseases, and urinary tract infection	South Africa	Lépesová et al. (2020), Lenart-Boroń et al. (2020), and Mbanga et al. (2021)
Agricultural activities	Surface water	Nutrient loading, salinity, and turbidity	Eastern Cape province	Barasarathi et al. (2022)
WWTP leakages, and industrial and farming activities	Surface water	Turbidity, and high levels of nitrates and phosphates	Free State province	Belle et al. (2020)
Pyrolyzate products	Surface water	Health concerns	Gauteng province	Vilakati et al. (2021)
Micro and macro pollutants, and inorganic chemicals	Surface and groundwater	Health concerns	South Africa and Mozambique	Verlicchi and Grillini (2020), and Archer et al. (2021)
High levels of phosphates	Surface water	Eutrophication	Mpumalanga province	Negwamba and Dinka (2019)
Heavy metals	Surface water	Health concerns	South Africa	Edokpayi et al. (2017b)

Discharging industrial or domestic wastewater into surface water (rivers) is a global practice (Divahar et al., 2019; Kumar et al., 2022). However, the effluent must meet specific criteria before being discharged. The World Health Organisation (WHO, 2017) provided guidelines on treatment procedures and the accepted physicochemical allowable limit. Although countries have different permissible limits for effluent discharged into water bodies, the WHO (2017) standard is often used as a reference. Adhering to the specified guidelines ensures a thriving habitat and limits the spread of diseases such as cholera. Furthermore, the cost of treating surface water for drinking purposes is also reduced.

## 2.2 Types of Wastewater Treatment Technologies

Wastewater treatment technologies (WWTTs) fall into three categories: mechanical, lagoons, and terrestrial (Pham et al., 2014; Rout et al., 2021). WWTPs are classified according to their purpose for treating wastewater. The WHO (2017) states that WWTPs are designed for primary, secondary, or tertiary wastewater treatment. Primary treatment is a mechanical, physical, or chemical technique that settles suspended solids (SS) or any other process that reduces the biochemical oxygen demand (BOD) and total suspended solids (TSS) of the incoming water by at least 50% and 20%, respectively, before discharging into surface water (WHO, 2017; Lu et al., 2022). After undergoing primary treatment, water is subjected to secondary treatment, which often entails biological or other treatments to remove at least 70% of the BOD and at least 75% of the chemical oxygen demand (COD) (WHO, 2017; MacEdo et al., 2022).

Tertiary treatment eliminates nitrogen, phosphorus, and any other contaminants, e.g., microbiological pollution, odour, or colour, which impact the quality or require a specialised application of water. Wastewater can be reused for various purposes such as agriculture, domestic and industrial use if properly treated via wastewater treatment technologies. The three types of WTTs fall into other sub-groups and are presented in Figure 2.2.



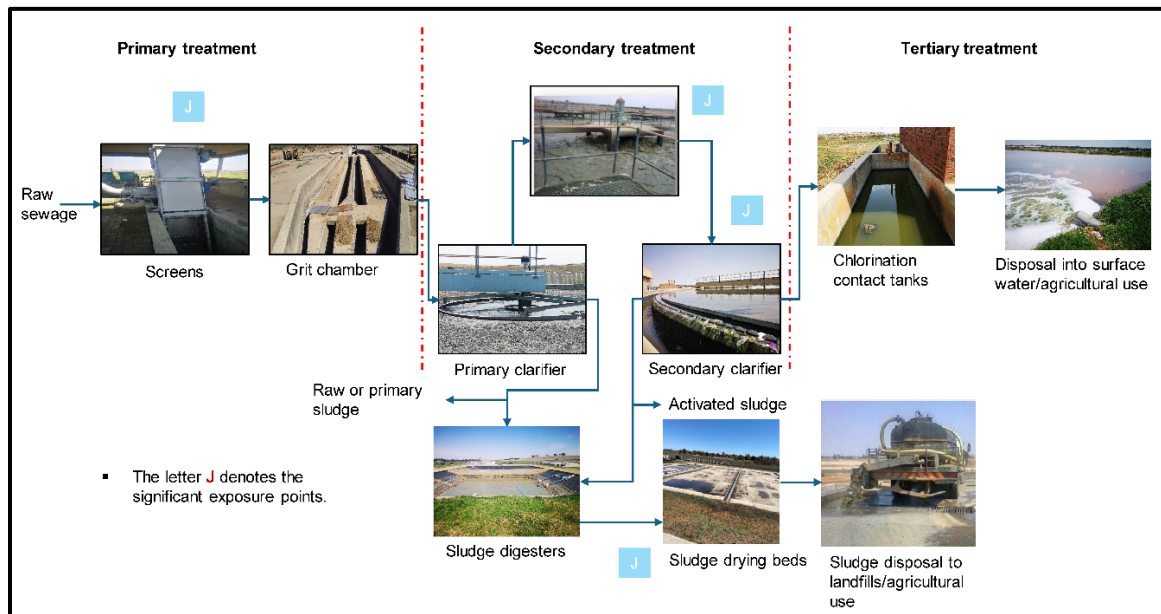
**Figure 2.2:** Different types of WWTs (UNEP, 1997)

The mechanical method of treating wastewater in Figure 2.2 is often used in places with limited land mass. The subtypes of mechanical WWTs are trickling filter single-cell protein, extended aeration, oxidation ditch, and sequencing batch reactors. These systems use chemical, biological, and physical processes to treat wastewater. Lagoons are used where there is sufficient land area and are either facultative ponds, hydrograph-controlled releases, or aerated ponds. These systems are cheaper than all other WWTPs and require lower costs for their operation and maintenance (Pham et al., 2014). The terrestrial approach to treating wastewater requires low levels of technical expertise and is ideal for households and rural areas. A significant advantage of this system is its ability to recharge groundwater and provide water for agriculture and reforestation (Akhoundi and Nazif, 2020). The types of terrestrial systems are subsurface infiltration, slow rate, rapid infiltration, and overland flow.

### 2.2.1 Activated sludge system

Activated sludge treatment (AST) converts the organic content in wastewater into carbon dioxide (CO<sub>2</sub>), bacterial cells, and water. According to Tran et al. (2022), AST is the most preferred method for wastewater treatment in urban areas. Moreover, this treatment process is also used in the treatment of industrial wastewater. The AST uses various techniques to ensure the effluent meets the

standard requirement before being reused or discharged to water bodies. A schematic of the multiple stages in the AST is presented in Figure 2.3.



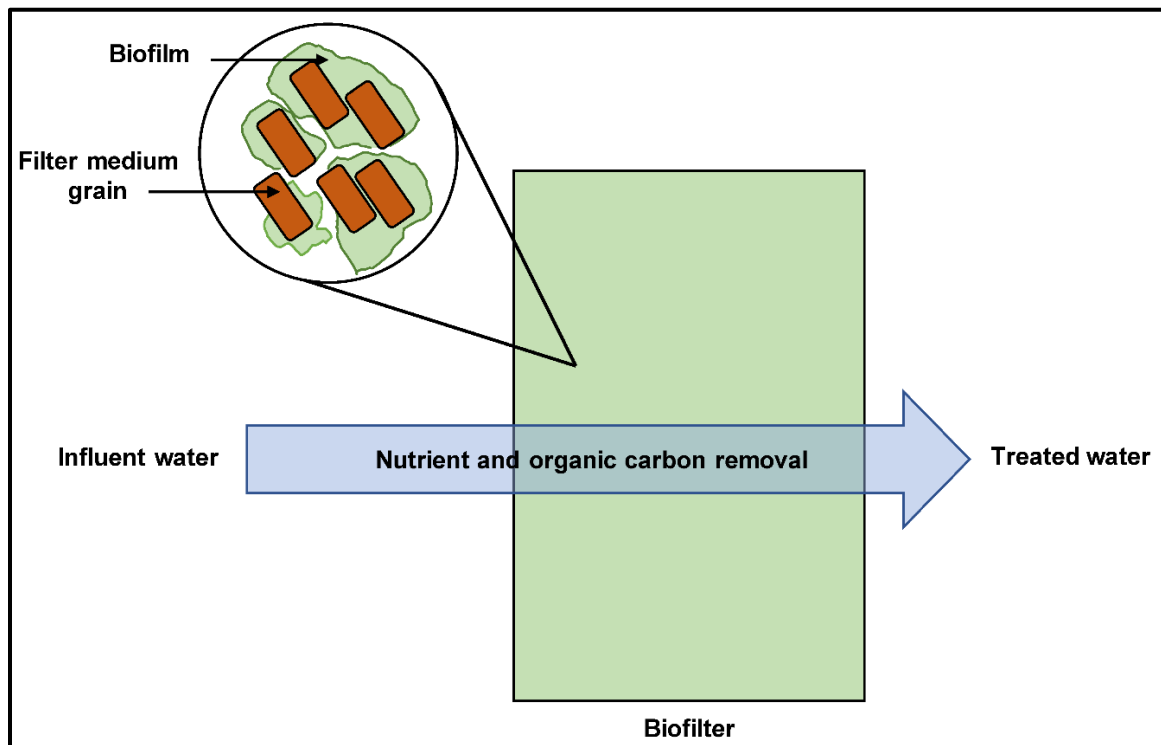
**Figure 2.3:** Schematic AST system layout (Author's image)

As shown in Figure 2.3, municipal wastewater treatment based on an AST system involves three stages, i.e., primary, secondary, and tertiary, with the letter 'J' denoting the significant exposure points. A detailed explanation of the three stages can be found in Scholz (2016) and Amoah et al. (2022). Over the years, AST has undergone numerous changes and modifications. These modifications have occasionally been the outcome of fundamental investigations into the underlying principles of the process. In recent times, the modifications have resulted from practical responses to specific issues with plant functioning (Wiśniowska, 2019). Although AST offers unique solutions in WWTPs, Thakur and Masakapalli (2022) argued that the system produces excessive sludge that could be reused if treated by anaerobic digesters rather than the disposal thereof.

## 2.2.2 Biofilters

Biofilters depend on living organisms to break down and remove organic and inorganic pollutants from wastewater (Jha et al., 2021). Biofilters are increasingly being used as an alternative to traditional WWTTs given that biofilters are cost effective, energy efficient, and environmentally sustainable (Jha et al., 2021;

Sarkar et al., 2021; Tripathi and Hussain, 2021; Yogalakshmi et al., 2021). The biofilms shown in Figure 2.4 provide a porous medium for the growth of microorganisms, which degrade and remove contaminants in the wastewater (Parmar et al., 2021). The porous medium can be made of various materials, including sand, gravel, peat moss, and/or activated carbon. The microorganisms used in biofilters include bacteria, fungi, and algae.



**Figure 2.4:** Treating wastewater via biofilters (Sehar and Naz, 2016)

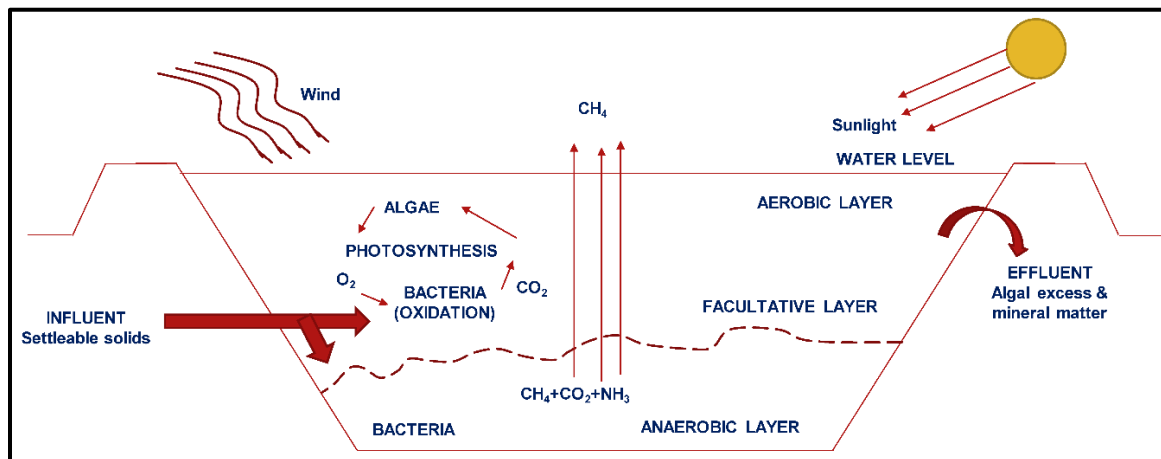
The distinct types of WWTs were introduced above, while the latter part of the literature review in this chapter will focus on lagoons, i.e., wastewater stabilisation ponds (WSPs).

## 2.3 Wastewater Stabilisation Ponds

### 2.3.1 Design considerations for wastewater stabilisation ponds

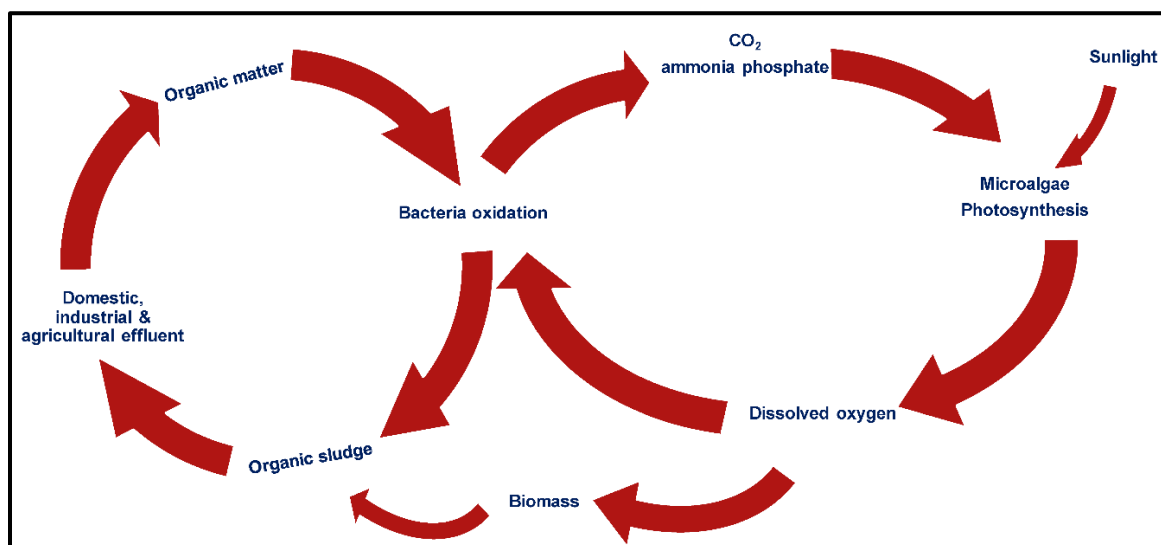
Wastewater stabilisation ponds (WSPs) are widely used where land is abundant for treating wastewater. According to Passos et al. (2019), these earth structures are robust, cheap, and require low maintenance. Treating wastewater via WSP is a natural process (Nuhu et al., 2020). The wastewater in the pond is exposed to

the atmosphere, where sunlight and ambient temperature influence the biotreatment processes. Studies have shown that industrialisation has significantly increased the quantity and nature of pollutants (Kohn et al., 2016; Gruchlik et al., 2018; Nuhu et al., 2020; Shah et al., 2020). These pollutants pose an imminent danger to humans, animals, and the environment if they are not adequately treated. A typical WSP design is presented in Figure 2.5.



**Figure 2.5:** Simplified schematic drawing of a WSP (Frédéric, 1999)

To improve these systems, some design considerations for WSPs are (i) to incorporate linings to control seepage to groundwater and excessive plant growth, (ii) improve the design and location of inlets and/or outlets, and (iii) employ hydraulic floating dividers, controls, and baffles. A schematic diagram of the treatment processes that occur in a typical WSP is shown in Figure 2.6.



**Figure 2.6:** Treatment processes for a typical WSP (Cowan et al., 2016)

The aerobic microorganisms in Figures 2.5 and 2.6 use dissolved oxygen (DO) in the pond to break down organic compounds to generate CO<sub>2</sub>. The algae subsequently use the CO<sub>2</sub> to produce oxygen (O<sub>2</sub>), after which the cycle repeats itself. The turbidity within the system leads to stratification, which encourages anaerobic activities at the bottom of the pond (Cowan et al., 2016). The digestion of the sludge at the bottom leads to the production of CH<sub>4</sub>, NH<sub>3</sub>, and CO<sub>2</sub>.

### 2.3.2 Pond design and performance

The conventional design of ponds is often rectangular; it would be interesting to investigate other shapes, especially in most urban areas where suitable land is scarce (Agunwamba, 2001). Land for WSP construction might be available only in specific shapes that may not allow optimal land utilisation if rectangular ponds are to be constructed. Hence, it is essential to consider whether other configurations that fit a given land site will yield similar or higher-quality effluent (Agunwamba, 2001). Research by Achag et al. (2021; 2023) on WSPs in the Sahara Desert, specifically in Morocco, examined the efficiency of three ponds. The research emphasised the impact of biogeochemical and operational variables on the treatment efficacy, underscoring the need to customise design approaches according to varying climatic conditions.

Another study by Olukanni and Ducoste (2011) in Nigeria utilised CFD simulations to improve the WSP design. According to Sah et al. (2012) and Coggins et al. (2018), the latter CFD simulations should have included meteorological data which could have impacted the system's performance. Another study in Turkey by Ali and Üçüncü (2023) also modelled and optimised WSPs without incorporating meteorological data, while no real-time data were used for comparison purposes.

### 2.3.3 Models and parameters

WSP models attempt to explain the system, analyse the impacts of various elements, and formulate hypotheses regarding the system's performance. Utilising WSP models saves time and cost and allows for necessary modifications before constructing a physical WSP. Various authors have developed and adopted

multiple models to predict performance regarding temperature effects on the rate of biological processes, oxygen balance in a pond, COD removal, hydraulic retention time, nitrogen transformation and removal, and pollutant removal. Hydrodynamic and transport models are often used to study a pond's flow patterns and pollutant transport. Therefore, the above design parameters can be expressed using the various mathematical models outlined below:

- (a) The temperature dependency model in Equation 2.1 determines the effect of temperature on the rate of biological processes in WSPs and is expressed as:

$$k_T = k_{20}\theta^{(T-20)} \quad (2.1)$$

Where:

- $k_T$  = rate constant at temperature  $T$ ,  
 $k_{20}$  = rate constant at 20°C,  
 $T$  = temperature (°C), and  
 $\theta$  = temperature coefficient.

- (b) Oxygen production and consumption models deal with the balance between oxygen produced in WSPs via photosynthesis and the oxygen produced through respiration and BOD degradation. The model is presented in Equation 2.2 as follows:

$$O_{net} = O_{prod} - O_{cons} \quad (2.2)$$

Where:

- $O_{cons}$  = oxygen consumption rate (mg/l/day),  
 $O_{net}$  = net oxygen balance (mg/l/day), and  
 $O_{prod}$  = oxygen production rate (mg/l/day).

But  $O_{prod}$  is expressed in Equation 2.3 as follows:

$$O_{prod} = P_{max}PARf(T) \quad (2.3)$$

Where:

- $f(T)$  = temperature correction factor,  
 $PAR$  = photosynthetic active radiation, and  
 $P_{max}$  = maximum photosynthetic oxygen production rate (mg/ l /day).

Also,  $PAR$  is expressed in Equation 2.4 as follows:

$$PAR = I_0 e^{-k_d d} \quad (2.4)$$

Where:

$d$  = depth of the water column (m),

$I_0$  = incident light intensity at the water surface ( $W/m^2$ ), and

$k_d$  = light attenuation coefficient.

And  $O_{cons}$  in Equation 2.2 is expressed in Equation 2.5 as follows:

$$O_{cons} = R_{BOD} + R_{resp} \quad (2.5)$$

Where:

$R_{BOD}$  = oxygen consumption rate due to BOD degradation ( $mg\ O_2/l.day$ ),  
and

$R_{resp}$  = oxygen consumption rate due to respiration of algae and other  
microorganisms ( $mg\ O_2/l.day$ ).

(c) Pathogen removal in WSPs is of utmost importance. Hence, models that predict the removal of pathogens take into consideration operational and environmental factors as presented in Equation 2.6:

$$P_f = P_o e^{-k_p HRT} \quad (2.6)$$

Where:

$P_f$  = final pathogen concentration (CFU/ml),

$HRT$  = hydraulic retention time (days),

$k_p$  = first-order rate constant for pathogen removal (per day), and

$P_o$  = initial pathogen concentration (CFU/ml).

HRT, denoted in Equation 2.6, signifies the mean duration that wastewater is retained within WSPs to guarantee that contaminants present in a pond have adequate temporal opportunity for treatment. Consequently, HRT is defined in Equation 2.7 as follows:

$$HRT = \frac{V}{Q} \quad (2.7)$$

Where:

$Q$  = flow rate of the influent wastewater ( $m^3/day$ ), and

$V$  = volume of the pond ( $m^3$ ).

(d) The COD removal model gives insight towards understanding how several factors affect the treatment of wastewater and the reduction of influent COD and various contaminants. To achieve the removal of pollutants and reduction of COD to comply with stipulated standards, the first-order kinetic model is used to predict WSP performance and provides insight for improving efficiency. The first-order kinetic model is expressed in Equation 2.8:

$$COD_{out} = COD_{in} e^{-k_{COD} HRT} \quad (2.8)$$

Where:

$COD_{in}$  = influent COD concentration (mg/l),

$COD_{out}$  = effluent COD concentration (mg/l),

$HRT$  = hydraulic retention time (days), and

$k_{COD}$  = first-order rate constant for COD removal (per day).

#### 2.3.4 Sludge distribution

The build-up of sludge in WSPs is an essential factor to consider as its accumulation is not evenly distributed, i.e., bathymetric heterogeneity. The traditional techniques for sludge measurement are labour intensive with a low spatial resolution (Coggins et al., 2020). For example, Coggins et al. (2017) highlighted that sludge surveys in WSPs can be used to create CFD models through bathymetric data with different degrees of success. In recent years, more advanced and reliable CFD models have shown the possibility of modelling the hydrodynamics and the removal of pathogens and nutrients in WSPs more effectively (Sah et al., 2012). The use of low-resolution bathymetric data makes it challenging to develop a hydrodynamic model for WSPs which can be validated against field measurements. Coggins et al. (2017) showed that increased bathymetry resolution in conjunction with a tracer could significantly increase the accuracy of the CFD modelling. Enhancing the hydraulic performance of a WSP is regarded as a critical management strategy, ensuring the protection of public health and the environment, while maximising the possible reuse of the treated effluent (Ouedraogo et al., 2016).

The hydrodynamic and transport models are used to understand and predict the behaviour of wastewater in WSPs. Both models support the simulation of wastewater movement, distribution, and contaminant removal. Furthermore, it allows for the improved design and optimisation of WSPs. These models, i.e., the hydrodynamic model (continuity and Navier-Stokes equation) and the transport model (advection-dispersion and mass balance equation), are presented in Equations 2.9 – 2.12:

The continuity equation is expressed as:

$$\frac{\delta C}{\delta t} + \bar{v}(vC) = D \nabla^2 C + S \quad (2.9)$$

Where:

- $C$  = concentration of the pollutant (mg/l),
- $D$  = diffusion coefficient (m<sup>2</sup>/s),
- $S$  = source or sink term representing production or removal of the pollutant (mg/l/s),
- $t$  = time (s), and
- $v$  = velocity vector of the flow (m/s).

The Navier-Stokes equation is expressed as:

$$\rho \left( \frac{\delta u}{\delta t} + (u\bar{v})u \right) = \bar{V}_p + \mu \bar{v}^2 u + f \quad (2.10)$$

Where:

- $f$  = body force per unit volume (N/m<sup>3</sup>)
- $u$  = velocity vector of the fluid (m/s)
- $\bar{V}_p$  = pressure gradient (Pa/m)
- $u\bar{v}u$  = convective acceleration (m/s<sup>2</sup>)
- $\bar{v}^2 u$  = body forces (s<sup>-1</sup>)
- $\rho$  = density of the fluid (kg/m<sup>3</sup>)
- $\mu$  = dynamic viscosity (N·s/m<sup>2</sup>), and
- $\frac{\delta u}{\delta t}$  = time derivative of velocity (m/s<sup>2</sup>).

The advection-dispersion equation is expressed as:

$$\frac{\delta C}{\delta t} + v\bar{V}C = D\bar{V}^2C + S \quad (2.11)$$

Where:

$D\bar{V}^2C$  = dispersion term ( $m^2/s \cdot mg/l$ )

$v\bar{V}C$  = advection term ( $m/s \cdot mg/l$ ), and

$\bar{V}^2$  = Laplacian operator of the second spatial derivatives ( $m^{-2}$ ).

The mass balance equation for a WSP can be expressed as:

$$\frac{\Delta M}{\Delta t} = I_R - O_R + G_R - C_R \quad (2.12)$$

Where:

$C_R$  = rate of consumption or degradation of the organics within the WSP ( $mg/s$ ),

$G_R$  = rate of generation of the substance within the WSP ( $mg/s$ ),

$I_R$  = rate at which the substance enters the WSP ( $mg/s$ ),

$M$  = mass of the substance within the control volume ( $mg$ ),

$O_R$  = rate at which the substance leaves the WSP ( $mg/s$ ), and

$t$  = time interval (seconds).

### 2.3.5 Ecology

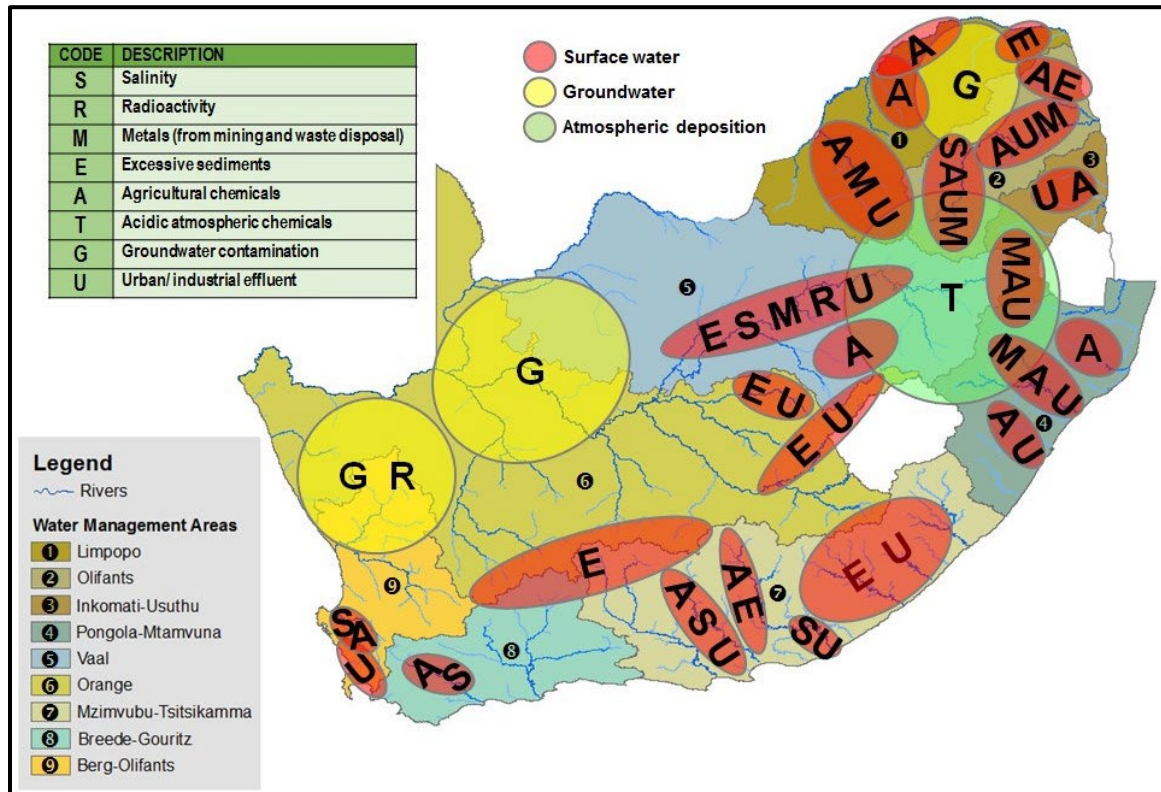
Sewage habitats, WWTPs, and wetlands have long been recognised as valuable breeding, foraging, and stop-over areas for many bird species in many regions of the world, including the southern United States of America (USA), Australia, and Africa (Orłowski, 2013). The droppings from birds also increase the existing microbial community. The ecology of a WSP plays a vital role in the treatment process given that bacteria and algae are essential in both the experimental design and operation of WSPs (Coggins et al., 2019). It is challenging to optimise the ecology of WSPs' growing interest, and a more in-depth understanding of the microbial processes would help in the productive and resilient treatment performance (Ferguson et al., 2018). The first-order kinetic model is used for several reasons, such as its ability to predict pollutant removal, expressed in Equation 2.8.

### **2.3.6 Re-engineered WSP response in a changing climate**

The predictive performance models of earlier WSPs were reliant on average temperature and hydraulic data (Mara, 2009). These models were unable to accurately predict WSP performance under extreme weather events, e.g., rising temperatures and intense rainfall, which are expected to increase in frequency due to climate change (Ghalhari et al., 2021). To date, studies on the subject are limited and have not adequately captured climate change in their models. A second challenge that needs to be addressed is the ability of WSPs to treat contaminants of emerging concern (CEC), such as pharmaceuticals. In addition, advanced WWTPs struggle to adequately remove CECs from treated wastewater (Rimayi et al., 2018; Rout et al., 2021; MacEdo et al., 2022). It should also be noted that this factor could potentially have an impact on the performance of re-engineering and is therefore regarded as beyond the scope of this study.

## **2.4 Wastewater Research in South Africa**

Various sources can contaminate water bodies. However, policies help in reducing the occurrence to the bare minimum. The lack of implementation of policies and maintenance of WWTPs seem to be the primary contributors to the contamination of water bodies in South Africa and the rest of the world (Edokpayi et al., 2017b; Verlicchi and Grillini, 2020). As a region continues to witness growth in population and industrialisation, the existing WWTPs need to match the upcoming challenges. These challenges have been highlighted as ECs, new compounds, pesticides, and pharmaceuticals (Rimayi et al., 2018; Gani et al., 2021; Kasonga et al., 2021; Rout et al., 2021). However, limited funding, technical expertise, and institutional capacity were highlighted as the main reasons why WWTPs continuously contaminate surface and groundwater (DWS, 2018). The state of surface and groundwater in South Africa is shown in Figure 2.7.



**Figure 2.7:** Surface and groundwater contamination in the provinces across South Africa (DWS, 2018)

The water body contaminated in the study area, as shown in Figure 2.7, consists of the surface water bodies (e.g., Vaal, Sand, and Vet Rivers), while the pollution sources range from wastewater effluent to heavy metals. More specifically, local activities, e.g., mining, agriculture, and failing infrastructure, are regarded as the main contributors to the contamination of water bodies in the FS province (Oke and Fourie, 2017; Marara and Palamuleni, 2019; Horn et al., 2022).

### 2.4.1 Urban and industrial wastewater discharge

The Gauteng province is home to densely populated cities, e.g., Johannesburg and Pretoria, which generate substantial domestic and industrial wastewater volumes (Teklehaimanot et al., 2015). Discharging untreated or inadequately treated wastewater from urban areas and industrial facilities into the Vaal River system contributes significantly to pollution. The high population density, inadequate wastewater treatment infrastructure, and operational challenges exacerbate the impact on water quality.

### **2.4.2 Mining activities**

The Gauteng province has a rich mining history, with significant mining operations focused on gold and other minerals. Mining activities, including extraction, processing, and mine drainage, can introduce pollutants such as heavy metals, metalloids, and acid mine drainage into the Vaal River System (DWS, 2019). These contaminants severely threaten water quality and ecosystems, affecting downstream areas, including the FS province.

### **2.4.3 Agricultural practices**

Agricultural activities within and surrounding the Gauteng province can indirectly impact the water quality of the Vaal River System (DWS, 2019). Using fertilisers, pesticides, and herbicides in agricultural practices can lead to the runoff of nutrients and chemicals into the river, potentially causing eutrophication and degradation of water quality.

It is alarming to note that the quality of surface water in the study area did not show any significant improvement during the past five years since the DWS publication on the deplorable state of surface water (DWS, 2022b). Children under the age of five represent 1.8 million of the 6 to 8 million people who pass away annually at a global scale owing to waterborne illnesses (Lipponen and Nikiforova, 2017; Osiemo et al., 2019). Information on waterborne disease mortality rates in the study area is limited. Although waterborne diseases pose a severe threat to humans, ECs were not mentioned in Figure 2.7, despite their prevalence in water bodies across the country. Archer et al. (2017) provided a comprehensive list of ECs present in South Africa's treated wastewater and surface waters.

## **2.5 Status Quo of WWTPs in the FS Province**

Limited information is documented on the quality of WWTP effluents discharged into water bodies for the study area and South Africa. The Green Drop Report (DWS, 2022a) examines the state of municipal wastewater treatment and the national wastewater infrastructure and management conditions. An example of a WSP in Bothaville, Nala Municipality, FS is shown in Figure 2.8.



**Figure 2.8:** Dilapidated state of the WSP in the Nala (Bothaville) Municipality, FS province (Jack et al., 2006)

Both the dilapidated state of the WSP and the unwanted presence of debris and dead animals are evident in Figure 2.8. The Green Drop Report for the latter municipality declined from 20% in 2011 to 6% in 2021 (DWS, 2022b). Further revelations on the operational state of WWTPs in the Dihlabeng and Maluti-A-Phofung municipalities in the FS province are highlighted in Figure 2.9.



**Figure 2.9:** Infrastructural failure of WWTPs in the FS province (DWS, 2022b)

At the WSPs shown in Figures 2.8 and 2.9, the major problems present are: (i) no final effluent disinfection, (ii) lack of activated sludge biomass, (iii) dilapidated aeration racetracks, (iv) faulty recycling pumps, and (v) a lack of operations, water quality information, and flow meter readings (DWS, 2022b).

A comprehensive list of the 96 WWTPs and their state as reported by DWS (2022b) in the FS province is presented in Table 2.2, detailing the impact of various activities and the contributions by WWTPs on the respective water bodies. The WWTPs Green Drop Score (GDS) classified as: Excellent (90 – 100%), Good (80 – 89%), Average (50 – 79%), Very Poor (30 – 49%), and Critical (< 30%), and Critical Risk Rating (CRR%) classified as: Critical (90 – 100%), High (70 – 89%), Medium (50 – 69%), and Low (< 50%), are respectively used to highlight the overall status of the WWTPs under consideration.

**Table 2.2:** State of WWTPs in the FS province (after DWS, 2022b)

Municipality	City/ Town	Discharge Point	GDS	WWTP CRR%
Dihlabeng	Bethlehem	Jordan River	49%	66.7%
	Clarens	Little Caledon River		47.1%
	Mashaeng	Meiringspoort Spruit		70.6%
	Mautse	Meul Spruit		70.6%
	Paul Roux	Sand River		70.6%
Kopanong	Bethulie	Orange River	26%	47.1%
	Edenburg	Riet River		47.1%
	Fauriesmith	Riet River		94.1%
	Gariep Dam	Natural Pan		94.1%
	Jagersfontein	Re-use		94.1%
	Philippolis	Otterspoort Spruit		47.1%
	Reddersburg	Fourie Spruit		94.1%
	Springfontein	Bossie Spruit		47.1%
Trompsburg	Van Zyl Spruit	47.1%		
Letsemeng	Jacobsdal	Riet River	40%	88.2%
	Koffiefontein	Riet River		94.1%
	Luckhoff	Riet River		47.1%
	Oppermansgronde	Evaporation ponds		94.1%
	Petrusburg	Evaporation ponds		47.1%
Mafube	Cornelia	Unnamed stream	0%	100%
	Frankfort	Wilge River		100%
	Namahadi	Wilge River		100%
	Tweeling	Liebensberg Vlei		100%
	Villiers	Vaal River		100%
Maluti-A-Phofung	Elands	Elands River	18%	88.2%
	Kestell	Sand Spruit		88.2%
	Makwane	Namahadi River		88.2%
	Moeding	Namahadi River		94.1%
	Phuthaditjhaba	Namahadi River		90.9%
	Tshiame	Wilge River		88.2%
	Harrismith	Nuwejaar Spruit		95.5%
Mangaung	Bainsvlei	Unknown stream	33%	68.2%
	Bloemindustria	Renoster Spruit		82.4%
	Bloemspruit	Bloem Spruit		84.4%
	Botshabelo	Klein Modder River		77.3%
	Dewetsdorp	NI		94.1%
	North-Eastern Works	Irrigation		77.3%
	Northern Works	Bree River		68.2%
	Soutpan	Klein Modder River		94.1%
Sterkwater	Renoster Spruit	86.4%		

Municipality	City/ Town	Discharge Point	GDS	WWTP CRR%
	Thaba N'chu	Korana Spruit		77.3%
	Van Stadensrus	Unknown		94.1%
	Welvaart	Kaal Spruit		77.3%
	Wepener	Caledon River		94.1%
Mantsopa	Excelsior	Lilana Spruit	29%	82.4%
	Hobhouse	Non-discharge		82.4%
	Ladybrand	Cathcartdrift Dam		72.7%
	Thaba Patchoa	Private land		82.4%
	Tweespruit	No discharge		82.4%
Masilonyana	Brandfort	Keerom Spruit	16%	100%
	Theunissen	Klein Vet River		88.2%
	Verkeerdevlei	No discharge		52.9%
	Winburg	Rietfontein River		100%
Matjhabeng	Allanridge	Voëlpan (Evaporation)	26%	76.5%
	Henneman	Riet Spruit		94.1%
	Kutlwangong	Sand Spruit		95.5%
	Mmamahabane	Erasmus Spruit		100%
	Odendaalsrus	Sand Spruit		95.5%
	Phomolong	Sloot Spruit		94.1%
	Thabong	Mosterd Canal - Sand River		100%
	Theronia	Flamingo Pan		90.9%
	Ventersburg	Erasmus Spruit		94.1%
	Virginia	Sand River		63%
	Witpan	Mostert Canal - Sand River		95.5%
Metsimaholo	Deneysville- Refengkgotso	Vaal Dam	11%	76.5%
	Oranjeville	Vaal Dam		76.5%
Mohokare	Rouxville	Caledon River	21%	94.1%
	Smithfield	Caledon River		82.4%
	Zastron	Montagu Dam		94.1%
Moqhaka	Kroonstad	Vals River	10%	81.8%
	Steynsrus	Jas Spruit/ Blom Spruit		70.6%
	Viljoenskroon	Olifant Vlei		70.6%
Nala	Bothaville	Vals River	6%	95.5%
	Monyakeng	Irrigation Dam		94.1%
	Wesselsbron	Irrigation Dam		94.1%
Ngwathe	Edenville	Rooikraal Spruit	10%	94.1%
	Heilbron	Eland Spruit		90.9%
	Koppies	Renoster River		94.1%
	Parys	Vaal River		95.5%
	Vredefort	Vaal River		90.9%
Nketoana	Arlington	NI	34%	64.7%
	Lindley	Vals River		88.2%
	Petrus Steyn	NI		82.4%
	Reitz	Langspruit		76.5%
Phumelela	Memel	Klip River/ Pampoen Spruit	4%	100%
	Vrede	Klip River		100%
	Warden	Cornelius River		94.1%
Setsoto	Clocolan	Mopedi River	19%	88.2%
	Ficksburg	Caledon River		95.5%
	Marquad	Laai Spruit		70.6%
	Senekal	Sand River		64.7%
Tokologo	Boshof	No discharge	39%	29.4%
	Dealesville	No discharge		47.1%
	Hertzogville	No discharge		41.2%
Tswelopele	Bultfontein	Natural Pan	40%	64.7%

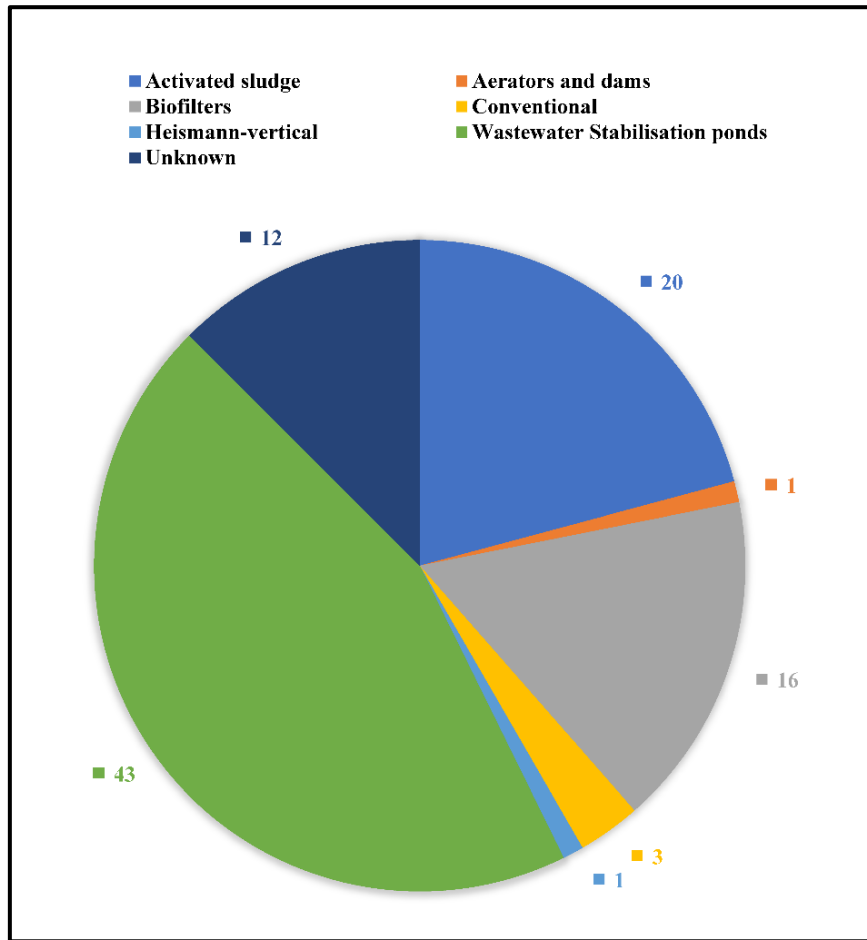
It is important to note that Gauteng, a highly urbanised and industrialised province, contributes significantly to the pollution load discharged into the atmosphere and river systems in the FS province (Madzunya et al., 2020). Reported contamination regarding waterbodies in the FS province are ascribed activities in the Gauteng province. The reports mostly focus on the contributions from mining, pharmaceuticals and agricultural activities (Durand, 2012; Madzunya et al., 2020; Mhlongo, 2023; Thomas et al., 2023). This is a cause for concern, given the

evidence on the state of the FS WWTPs listed in Table 2.2. The Green Drop Report (DWS, 2022a) shows that the situation is no different for most WWTPs in the province.

## **2.6 Wastewater Treatment Processes in the FS Province**

The wastewater treatment responsibilities in the study area fall under the Middle Vaal, Upper Orange and Upper Vaal Water Management Areas (WMAs) (Du Plessis, 2017). The WWTT associated with each of the 96 WWTPs in the FS province as listed in Table 2.2, are shown in Figure 2.10. It should be noted that despite advances made internationally in the area of wastewater treatment (Dong et al., 2022; Kaviya, 2022; Rout et al., 2022; Saidulu et al., 2022), there are only 20 AST and 16 biofiltration plants in the FS province.

In Figure 2.10, twelve (12) of the treatment methods are unknown. This raises serious concerns because the WMA in charge of managing wastewater should have a detailed inventory of the methodologies employed in treating wastewater. This is a step towards proper record-keeping and will assist in identifying which of the treatment processes and effluent disposal methods are potential sources of pollution.



**Figure 2.10:** WWTs at WWTPs in the FS province (after DWS, 2022b)

Table 2.3 shows the treatment processes and effluent disposal for the WMAs in the FS province.

**Table 2.3:** WWTs and effluent disposal/utilisation in the FS province (Bailey, 2013)

WMA	WWTs	Effluent disposal/utilisation
Middle Vaal	Biofilters and Biological seep beds AST Dasveer System WSPs Unknown	All effluents reused by mines Discharge to surface water Evaporation ponds Irrigation No discharge Swamp Unknown
Upper Vaal	AST Biofilters and biological seep beds Biofilters Unknown	Discharge to surface water Unknown
Upper Orange	AST Aerators and dams Biofilters Conventional Heismann-vertical WSPs Unknown	Discharge to surface water Evaporation Irrigation No returns Unknown

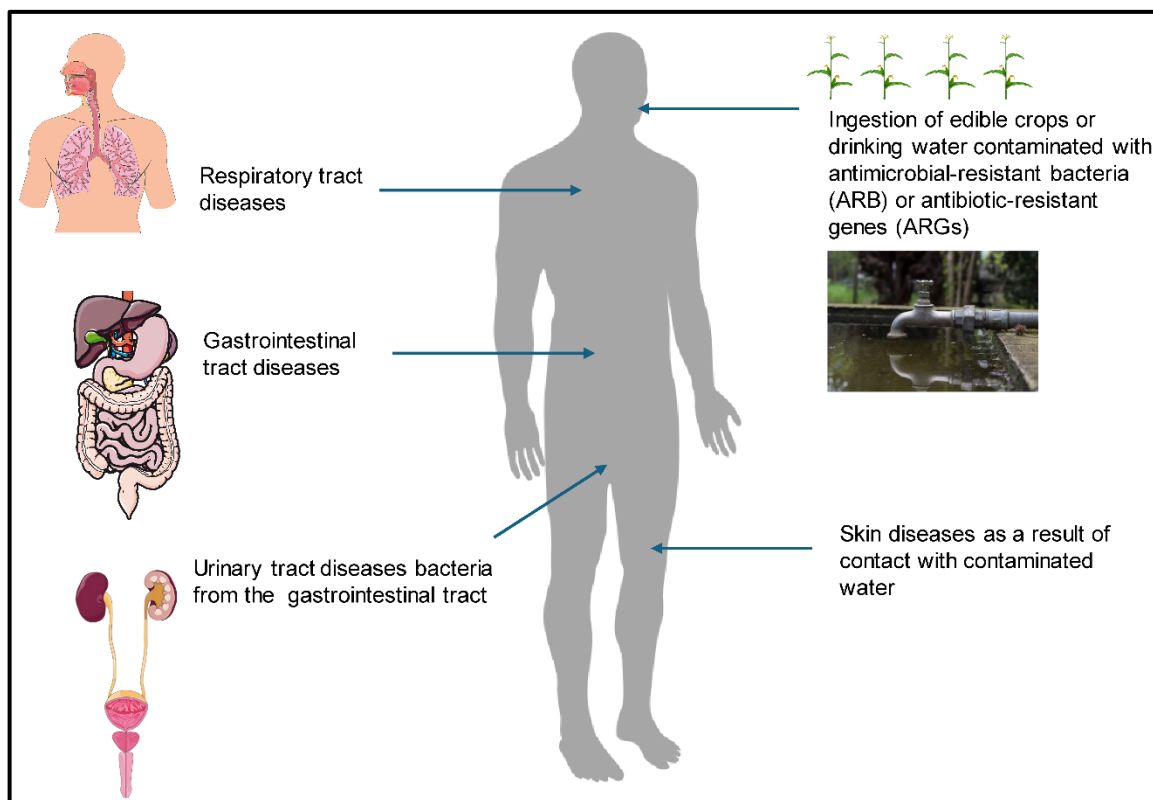
## 2.7 Environmental and Health Impacts

Scientific knowledge has grown dramatically owing to the recent increase in research conducted on wastewater characteristics for analysing specific constituents and their effect on the environment and human health (Campos-Mañas et al., 2018; 2019; Rimayi et al., 2018; Bijay-Singh and Craswell, 2021; Kasonga et al., 2021; Revilla Pacheco et al., 2021; Pu et al., 2022; Saidulu et al., 2022) and improved methods (Hlophe and Hillie, 2014; Al-Qodah et al., 2020; López-Vinent et al., 2021; Mateus et al., 2021; Nair et al., 2021; Pandey et al., 2021; Rout et al., 2021; Julian et al., 2022; Lescano et al., 2022). In addition, when wastewater which does not meet the required quality standards is discharged into water bodies serving as a source of water supply, the cost implication for water treatment for potable use can become significant.

The associated environmental health impact of wastewater reuse was addressed by Li et al. (2022), who investigated the environmental contamination from WWTPs due to antibiotic-resistant bacteria (ARB), antibiotic-resistant genes (ARG), and their impact on human health. Perez-Mercado et al. (2022) found that lettuce primarily irrigated with wastewater did not significantly indicate a decrease in helminth eggs and *E. coli* concentrations. In another finding, food safety and public health hazards occur globally owing to watering food crops with effluent containing chromium from the tannery sector (Younas et al., 2022). Toxic effects might be immediate or build up over time (Edokpayi et al., 2017a). Hence, the investigation by Li et al. (2022) on the possible effects of contaminated water and/or agricultural produce irrigated with wastewater effluent on human health is presented in Figure 2.11.

It is evident from Figure 2.11 that using wastewater effluent containing unacceptable concentrations of ARBs and ARGs can lead to conditions ranging from gastrointestinal tract to respiratory tract diseases. Various researchers, e.g., Leonel and Tonetti (2021) and Onalenna and Rahube (2022), demonstrated that reusing wastewater poses significant health risks as these waters contain harmful bacteria. These findings should further strengthen the decision of respective

municipalities and waterboards that wastewater effluent, irrespective of its use, should meet the required standards before discharge or reuse.



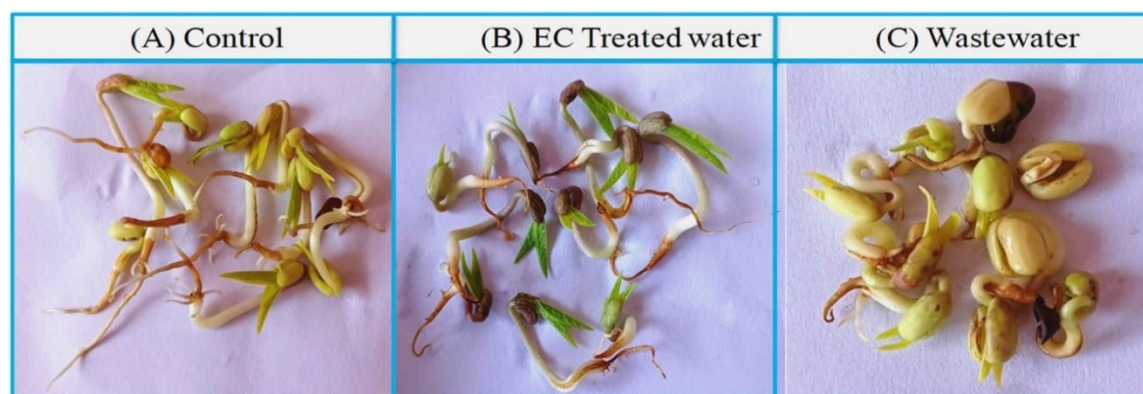
**Figure 2.11:** Impact of contaminated drinking water or agricultural produce from wastewater effluent containing ARBs and ARGs (after Li et al., 2022)

Studies by Vymazal et al. (2021) found that agricultural and horticultural crops irrigated with domestic wastewater effluent resulted in significant savings in freshwater and fertilisers. According to Shannag et al. (2021), agricultural irrigation constitutes the most considerable demand for wastewater reuse, namely about 73% of the global water usage (De Sanctis et al., 2017). Some researchers argue that reusing wastewater from WWTPs for agriculture effectively addresses the water shortage (Vymazal et al., 2021; Li et al., 2022). For example, wastewater has been reused for agricultural purposes in India (Minhas et al., 2022), Saudi Arabia (Al-Jasser, 2011), Israel (Reznik et al., 2017), Canada (Dhiman et al., 2020), and South Africa (Gemmell and Schmidt, 2012; Mendoza-Espinosa et al., 2019; Ben Hassena et al., 2022).

Although using wastewater has become widespread, there have been associated health and environmental concerns. Subsequently, careful screening and

monitoring of wastewater effluent intended for reuse in agriculture is required to prevent complications (Alygizakis et al., 2020). However, in the FS province, the WWTTPs in the towns of Botshabelo, Mafube, and Thaba N'chu provide wastewater effluent for irrigation purposes without the appropriate monitoring of the effluent quality, while the relevant end-users are also not informed or aware of the possible impact on their agricultural produce and health.

According to Alderson et al. (2015), wastewater reuse in agriculture is evaluated via risk assessment rather than reliability analyses. In the latter study, it was noted that the need for effective management and operation is imperative. Furthermore, the performance statistics revealed significant variations between similar operational WWTTPs. Kumar and Sharma (2022) investigated the effect of three different water sources, e.g., potable (tap) water, electrocoagulation (EC) treated water, and industrial wastewater on the mung bean (*V. radiata*), presented in Figure 2.12.



**Figure 2.12:** Impact of different water sources and quality on the sprouting of mung beans (Kumar and Sharma, 2022)

As expected, the germination test with tap water (control) had no defects, and the EC-treated water was appropriate for irrigation. In addition, Singh (2021) showed that wastewater effluent used for irrigation has several drawbacks compared to its benefits. The primary concern with reusing treated wastewater for agricultural purposes are the health and environmental impacts (Verbyla et al., 2016; Biswas et al., 2021; Mateus et al., 2021; Perez-Mercado et al., 2022).

## 2.8 Conclusions

The review considered the potential surface and groundwater pollution sources in the FS province, South Africa. The removal efficiency discussed refers to the overall removal (primary, secondary, and tertiary) of pollutants by WWTPs which are regarded as having the potential to improve significantly on the quality of surface and groundwaters. Unfortunately, literature on surface water contamination from discharged wastewater effluent in the FS province is scarce. For example, available literature focuses on pollution from agricultural, industry, and mining activities, despite the WWTP shortcomings documented in the Green Drop Report (DWS, 2022a). It is evident that the overall and effective operation and management of WWTPs are impacted by a lack of proper infrastructure, maintenance, and resources, which seriously threaten the environment and public health owing to the wastewater effluents not meeting the required standards.

To mitigate the identified problem, it is essential that WWTPs are adequately designed, constructed, and maintained to minimise the discharge of harmful pollutants into the environment. Regular monitoring and testing of water quality are crucial to identify and address contamination issues promptly. In addition, the government should provide adequate resources and funding to improve the wastewater treatment infrastructure, promote public water conservation and management education, and implement strict regulations to ensure compliance with environmental standards.

Given the potential environmental and health impacts, it is crucial that the protection of water resources and the environment is prioritised in South Africa, with specific reference to the study area (FS province) considered in this research. Hopefully, this review will draw the urgent attention of government and the respective stakeholders to take the necessary measures to mitigate contamination from WWTPs, preserve our natural resources, safeguard public health, and promote sustainable regional development. Given the lack of information available for the FS province, further research initiatives are already underway to study the impact of wastewater effluent on surface and groundwater in the province. One of the approaches is to examine the physicochemical properties of discharged

wastewater effluent and compare it to recommended national standards, which will be elaborated on in the next chapter(s).

The location and characteristics of the study area are presented in the next chapter. Thereafter, the methodologies adopted to investigate the research aim and achieve the specific study objectives, are outlined.

## CHAPTER 3 : MATERIALS AND METHODS

The location and characteristics of the study area are presented in this chapter. Thereafter, the methodologies adopted to investigate the research aim and achieve the specific research objectives are outlined. Moreover, to highlight the relevance of the literature review conducted in Chapter 2, an overview and justification of the adopted methodology are presented as follows.

### 3.1 Overview and Justification of the Methodology

Distinct options were considered in attempt to address the surface water quality issues experienced in the MMM located in the FS province, South Africa. The current water quality issues experienced by the MMM are due to below standard wastewater effluent being discharged into the receiving surface waters.

In accordance with the specific research objectives, the methodology is divided into three parts: (i) sampling and determination of various influent and effluent physicochemical wastewater parameters at selected WWTPs, (ii) determination of the receiving surface water bodies' water quality characteristics and compliance in relation to the SANS criteria for wastewater effluent, and (iii) the design and simulation of re-engineered WSPs to ensure efficient contaminant removal.

In Part (i), a cost-effective approach was adopted to sample and determine the various physicochemical parameters, while aligning it to the UN SDGs 3, 6, and 14. The cost-effective approach entailed the monthly sampling of influent and effluent at four WWTPs over a three-year period, while only COD was used as an indicator of microorganisms (*E. coli* count) in the sampled wastewater. Given that biological tests to determine the *E. coli* count are normally time-consuming and costly, considering only COD as an indicator has been widely adopted (Mbanga et al., 2020; Gumede et al., 2021).

In Part (ii), influent and effluent from four WWTPs were sampled, and downstream in-situ surface water tests were conducted at suitable river sites. Lastly, in Part (iii), the Hydromantis extended simulation software tool (GPS-X 8.5) was used to design and simulate different WSP configurations based on the effluent data from

the four WWTPs. A steady state configuration was adopted, and it was assumed that meteorological parameters do not impact the ability of WSPs to remove contaminants effectively.

### 3.2 Description of the Study Area

According to Du Plessis (2017), the Vaal River shares a boundary with the FS province, and it is one of the most important rivers in South Africa. In addition to the Vaal River, the Modder River is also regarded as an important river within the FS province, especially in the study area, i.e., the MMM which comprises Bloemfontein, Botshabelo, Thaba N'chu, Sout Pan, and Dewetsdorp (Khoza and Maramura, 2024). Both these river systems are prone to contamination from human activities (Koning et al., 2000; Mollo and Nomngongo, 2022). Despite the suspected contamination by human activities, the FS province has the second lowest percentage population distribution (PPD) in South Africa, with the PPD (2013 – 2022) showing a steady decline from 5.3% to 4.9% (Statistics South Africa, 2022). The source of the Modder River lies in the eastern hills of the quaternary catchment C52A, which is situated in the central region of the FS province (Griesel and Jagals, 2002). It is a comparatively small river with a contributing catchment area of 7 960 km<sup>2</sup>, while the mean annual runoff (MAR) equals 184 x 10<sup>6</sup> m<sup>3</sup> (Gwate et al., 2015). The water originating from the Modder River is stored in various dams, including the Rustfontein, Mockes, Mazelspoort, and Krugersdrift Dams. The location details of the surface water (SW) sampling sites associated with the four WWTPs under consideration are summarised in Table 3.1 and shown in Figure 3.1. Google Earth satellite images of each of the WWTPs are shown in Figures 3.2 to 3.5.

**Table 3.1:** Location of the WWTPs and SW sampling sites in the MMM

Identification (ID)	Name	Process (Capacity)/Source	Latitude	Longitude
WWTPBA01	Bainsvlei WWTP, Bloemfontein	AST (5 Ml/day)	-29.101458	26.114807
WWTPBO02	Botshabelo WWTP, Mangaung	Biofilter and WSP (20 Ml/day)	-29.239139	26.685472
WWTPNE03	North-Eastern WWTP, Bloemfontein	AST (20 Ml/day)	-29.090945	26.321749
WWTPSW04	Sterkwater WWTP, Bloemfontein	AST (20 Ml/day)	-29.188384	26.308462
SWRS01	Renoster Spruit	WWTPNE03 & WWTPSW04	-29.026528	26.326444
SWBS02	Bloem Spruit	Bloem Spruit WWTP	-29.120833	26.281306
SWKLM03	Klein Modder River	WWTPBO02	-29.242861	26.673944

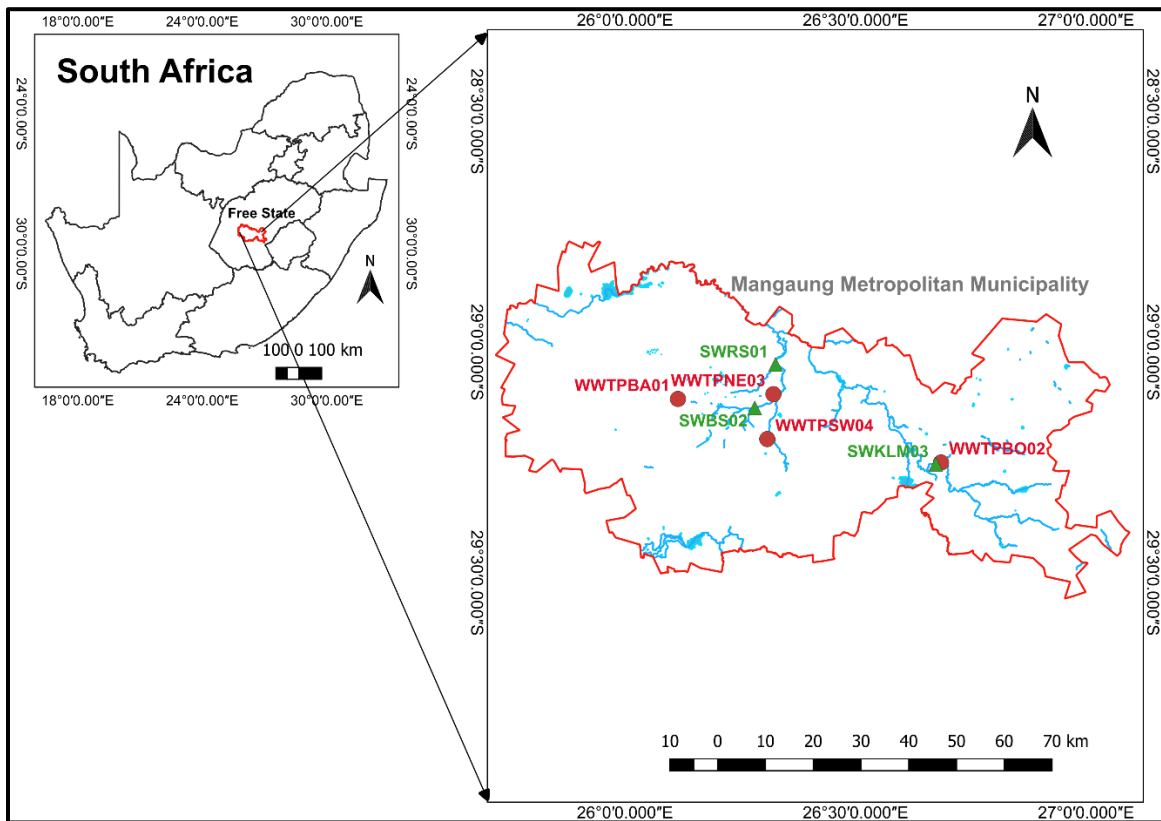


Figure 3.1: Location of the WWTPs and SW sampling sites within the MMM



Figure 3.2: Bainsvlei WWTP



Figure 3.3: Botshabelo WWTP



Figure 3.4: North-Eastern WWTP



**Figure 3.5:** Sterkwater WWTP

### 3.3 Materials

The materials used to determine the physicochemical parameters at the four WWTPs and SW sampling sites include the following:

- (a) **Hardware:** This includes an HQ40d multimeter probe, a thermoreactor, Büchner funnels, a spectrophotometer, cuvette cells, an oven, filter papers, test tubes, flasks, a mass balance, a measuring cylinder, and probes, e.g., HANNA's HI 98195 and HI 98193.
- (b) **Chemicals, reagents, and indicators:** These include concentrated sulphuric acid ( $\text{H}_2\text{SO}_4$ ), potassium dichromate ( $\text{K}_2\text{Cr}_2\text{O}_7$ ), ferroin indicator, ferrous ammonium sulphate ( $\text{Fe}(\text{NH}_4)_2(\text{SO}_4)_2 \cdot 6\text{H}_2\text{O}$ ), sodium thiosulphate ( $\text{Na}_2\text{S}_2\text{O}_3$ ), potassium permanganate ( $\text{KMnO}_4$ ), starch indicator, boric acid ( $\text{H}_3\text{BO}_3$ ), phosphate solution and powder ( $\text{KH}_2\text{PO}_4$ ), nitrate I ( $\text{KNO}_3$ ), and nitrate II ( $\text{NaNO}_3$ ) solutions.
- (c) **Software:** The Hydromantis extended simulation software tool (GPS-X 8.5) was used to simulate the various WSP design configurations and operation.

Data analyses and visualisation were done using OriginPro 9.0, Microsoft Excel, and IBM SPSS 27.

### 3.4 Methodology

As highlighted in Section 3.1, the methodology as described in this section is divided into three parts, i.e., influent and effluent physicochemical wastewater parameters at WWTPs, water quality characteristics at SW sampling sites, and the design and simulation of re-engineered WSPs.

#### 3.4.1 Influent and effluent physicochemical parameters at WWTPs

**Conductivity, temperature, pH, and total dissolved solids:** An HQ40d multimeter probe as shown in Figure 3.6 was used to determine the conductivity temperature, pH, and total dissolved solids (TDS). Typically, the probe consists of a potentiometer, glass electrode, reference electrode, and a temperature compensating device and beakers. The electrometric method was applied to determine the activity of hydrogen ions reflected by the potentiometric measurement using standard hydrogen and reference electrodes.



**Figure 3.6:** HQ40d probe for determining pH, conductivity, and temperature

The materials used include solutions, standard pH buffer 4, 7, and 10 solutions, and conductivity standards. To ensure the accuracy of any probe reading, the HQ40d probe was calibrated against the buffer pH values and conductivity standards. Thereafter, the electrodes were rinsed with distilled water. A grab sample of the influent and effluent wastewater was transferred into a beaker, with the HQ40d multimeter probes being immersed. Subsequently, readings associated with the three parameters were recorded, while the TDS was determined by using Equation 3.1:

$$TDS = 6.67EC \quad (3.1)$$

Where:

$TDS$  = total dissolved solids (mg/l), and

$EC$  = electrical conductivity ( $\mu\text{S}/\text{cm}$ ).

**Chemical oxygen demand:** The COD of the influent and effluent wastewater was determined using a thermoreactor as shown in Figure 3.7 by deploying a closed reflux titrimetric method (Negwamba and Dinka, 2019; Kermet-Said et al., 2024; Lamssali et al., 2024). In applying the closed reflux titrimetric method, most of the organic matter is oxidised by a boiling mixture of chromic and sulphuric acids. Typically, a sample is refluxed in a strong acidic solution with a known excess of potassium dichromate. After digestion, the remaining unreduced potassium dichromate is titrated with ferrous ammonium sulphate to determine the amount of potassium dichromate consumed.

The apparatus used includes a thermoreactor, test tubes with lids, Erlenmeyer flasks, and a test tube rack. Chemicals and reagents used include concentrated sulphuric acid, potassium dichromate, ferroin indicator, and 0.1 M ferrous ammonium sulphate. The steps followed to determine the COD of the influent and effluent wastewater samples are summarised below:

- (a) Pipette 3.5 ml concentrated sulphuric acid, 1.5 ml potassium dichromate, and 2.5 ml distilled water into a test tube (control), close tightly with a lid and shake.

- (b) Pipette 3.5 ml concentrated sulphuric acid, 1.5 ml potassium dichromate, and 2.5 ml sample (influent/effluent) into a test tube, close tightly with a lid and shake.
- (c) Place the test tubes into the thermoreactor preheated to 140°C and reflux for two (2) hours.
- (d) Cool to room temperature and place in a test tube rack.
- (e) Transfer the solution into an Erlenmeyer flask, rinse a test tube with distilled water three times, and add it to the flask.
- (f) Add 2 to 3 drops of ferroin indicator and shake vigorously (the solution will turn blue-green).
- (g) Titrate with 0.1 M ferrous ammonium sulphate (endpoint should be reddish brown).



**Figure 3.7:** Example of the thermoreactor used to determine the COD

**Oxygen absorbed:** The azide modification method was used as a standard test to determine the amount of dissolved oxygen (DO) or oxygen absorbed (OA) or both. In the analyses, the manganous ion reacts with the DO in the alkaline solution as

shown in Figure 3.8. To determine the OA, the following apparatus, chemicals, reagents, and indicator were used: OA flasks with caps, 100 ml measuring cylinder, 25% sulphuric acid, 0.025 M sodium thiosulphate, 0.0025 M potassium permanganate, and a starch indicator.



**Figure 3.8:** Endpoint results associated with the DO/OA experiment

The steps followed to determine the OA of the influent and effluent wastewater samples are summarised below:

- (a) Add 100 ml of distilled water to a measuring cylinder to be transferred into an OA flask (control).
- (b) Add 20 ml of a sample to a 100 ml measuring cylinder, fill up to the mark with distilled water, and transfer it to a flask.
- (c) Add 10 ml of 25% sulphuric acid and 25 ml of 0.0025 potassium permanganate to the flask in (b), close tightly with lids and store in a dark room for four (4) hours.
- (d) Titrate with 0.025 M sodium thiosulphate until a faint yellow colour appears.
- (e) Add a few drops of starch until the solution turns deep blue.
- (f) Titrate with 0.025 M sodium thiosulphate to result in the end point (colourless).

**Ammonia:** The distillation and titrimetric methods were used to determine the quantity of ammonia present in the wastewater samples. The samples were buffered with a borate buffer to decrease hydrolysis of cyanates and organic nitrogen compounds while being distilled into a solution of boric acid. Thereafter, the ammonia in the distillate was determined trimetrically with standard sulphuric acid and a mixed indicator to result in the endpoint shown in Figure 3.9.



**Figure 3.9:** Endpoint results associated with the ammonia experiment

The following apparatus, chemicals, reagents, and indicator were used: Distillation apparatus, round bottom flask, heating mantle, Erlenmeyer flasks, 500 ml measuring cylinder, boric acid, borate buffer, 0.02 M sulphuric acid, and a mixed indicator. The steps followed to determine the ammonia concentrations in the influent and effluent wastewater samples are summarised below:

- (a) Add 140 ml of the sample in a 500 ml measuring cylinder, fill it up to 280 ml with distilled water and transfer it into a round bottom flask.
- (b) Add 5 ml of borate buffer to the solution.
- (c) Add 25 ml of boric acid to an Erlenmeyer flask and 3 to 4 drops of a mixed indicator.
- (d) Put the solution in a round bottom flask on a heating mantle preheated to 100°C.
- (e) Distillate against the boric acid solution and collect approximately 150 ml of the distillate.
- (f) Titrate with 0.02 M sulphuric acid (from blue to pale pink).

**Phosphate:** The level of phosphate in the samples was determined via the spectroscopic method. This is a physical method that separates radiation according to certain properties, such as wavelength, and assists in identifying the chemical element composition. The spectrophotometer is shown in Figure 3.10, while cuvette cells, test tubes, phosphate solution, and phosphate powder were used to determine the amount of phosphate in the influent and effluent wastewater samples.



**Figure 3.10:** NOVA 60 Spectroquant® to determine the phosphate concentrations

The following steps were deployed to determine the phosphate concentrations:

- (a) Pipette 5.0 ml of a sample into a test tube.
- (b) Add five drops of phosphate solution and mix.
- (c) Add one level of blue micro spoon of phosphate powder and shake vigorously to dissolve the solid substance (solution turns blue). The reaction time is five minutes.
- (d) Transfer the solution into a cuvette cell.
- (e) Select the relevant method using the auto selector.
- (f) Place the cuvette cell into the cell compartment.

**Nitrate:** The spectroscopic method (*cf.* Figure 3.10) was also used to determine the levels of nitrate and nitrite levels in the influent and effluent wastewater samples. The same apparatus as used for phosphate testing were used. However, the reagents were different, i.e., Nitrate I ( $C_6H_7NO_3S$ ) and Nitrate II ( $C_{12}H_{14}N_2$ ).

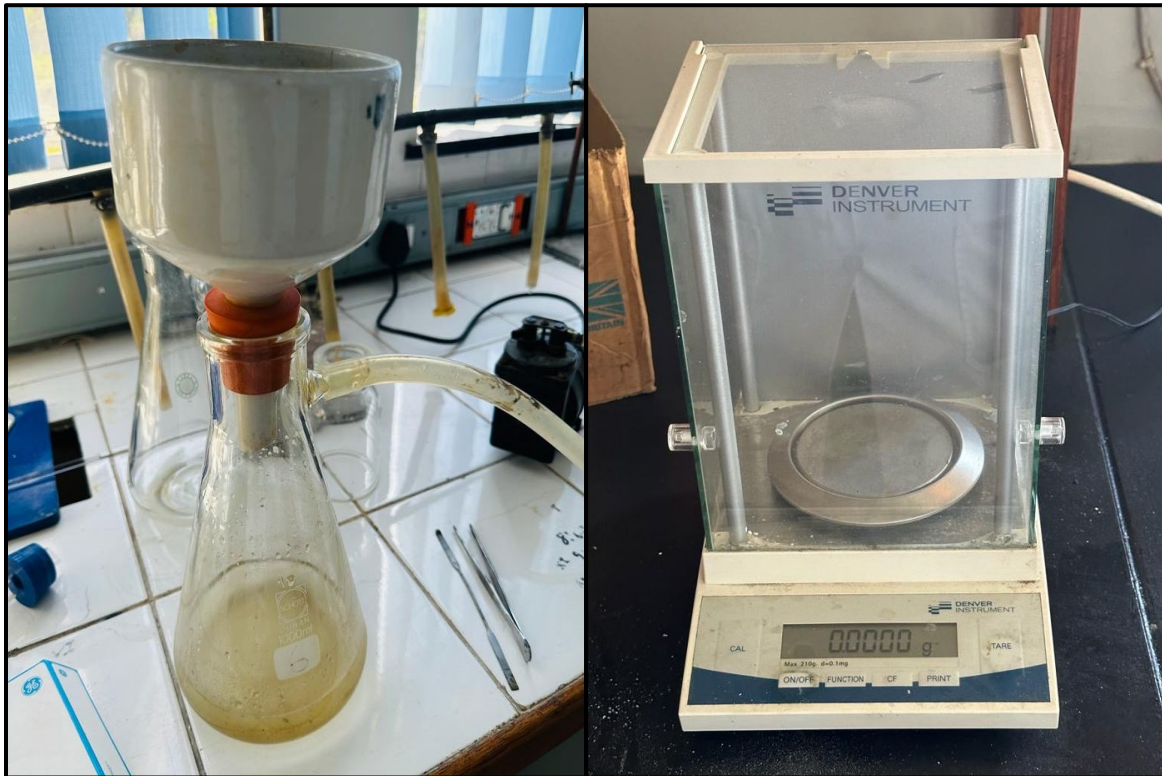
The steps followed are summarised below:

- (a) Pipette 4 ml of Nitrate I into a test tube. Thereafter, add 0.5 ml of sample (do not mix).
- (b) Add 0.5 ml Nitrate II, close the lid and mix. Exercise caution, given that the cell becomes hot. The reaction time is 10 minutes.
- (c) Transfer the solution to a cuvette cell and set the mode on the spectrophotometer to auto-selector.
- (d) Place the cuvette cell in the compartment and take the readings displayed for the  $NO_3^-$  and  $NO_2^-$  levels.

**Suspended solids:** The gravimetric method was used to determine the analyte, i.e., suspended solids (SS), quantitatively based on its mass in the effluent samples. No reagents were required or used. The apparatus used included: a Buchner funnel (*cf.* Figure 3.11), filter paper (5 $\mu$ m MCE membrane), oven, and balancing scale.

The steps followed to determine the SS concentrations are summarised below:

- (a) Weigh the mass of a dry filter paper on the balancing scale.
- (b) Filter 100 ml of the sample on the Buchner funnel.
- (c) Dry the filter paper in an oven preheated to 100°C for about 1 to 2 hours.
- (d) Allow the filter paper to cool down to room temperature and weigh again.  
Subtract the mass of the dry filter paper from the current weight to ascertain the actual mass of the SS (mg/l) contained in the effluent sample.



**Figure 3.11:** SS experimental setup

**Mixed liquor suspended solids:** Mixed liquor suspended solids (MLSS) assist WWTP operators to determine whether there are enough microorganisms present to treat the influent effectively. The experimental setup and procedure used to determine the SS concentrations are the same for MLSS, except for the filter paper size. Filter paper (5 $\mu$ m MCE membrane) was used to determine the MLSS in the effluent.

**Settleable solids and sludge volume index:** The volumetric method provides a way to determine the settleable solids accurately by carefully measuring the concentration thereof in the solution shown in Figure 3.12, while the sludge volume index (SVI) is determined mathematically using Equation 3.2:

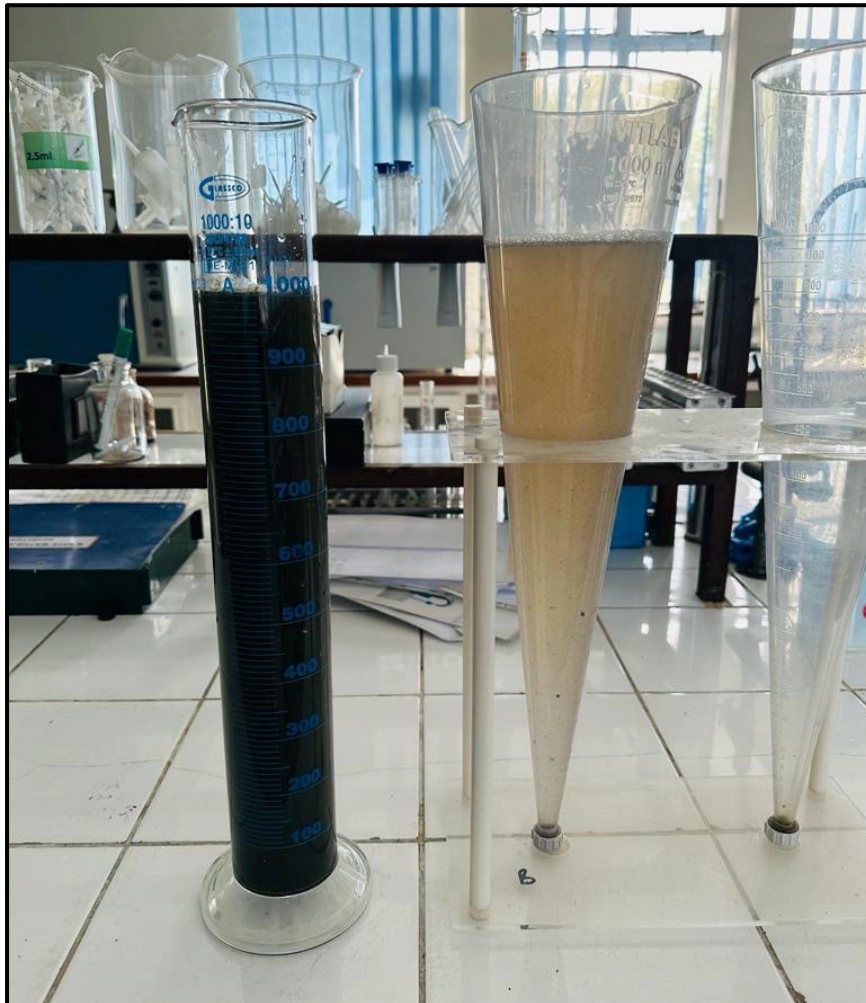
$$SVI = \frac{1000SSV}{MLSS} \quad (3.2)$$

Where:

$SVI$  = sludge volume index (m $\ell$ /g),

$MLSS$  = mixed liquor suspended solids (g/ $\ell$ ), and

$SSV$  = settleable sludge volume (m $\ell$ / $\ell$ ).



**Figure 3.12:** Settleable solids experimental setup

A 1000 ml measuring cylinder, as shown in Figure 3.12, was used with no reagents to determine the settleable solids. The effluent was mixed or shaken and transferred into a 1000 ml measuring cylinder. The sample was allowed to settle for 30 minutes, with the results subsequently recorded. After allowing settlement for another 30 minutes, the results are recorded again.

### **3.4.2 Water quality characteristics at SW sampling sites**

In MMM, the practice is to discharge effluent from the WWTPs into the tributaries of the Modder River. Hence, a seasonal (from spring to winter) sampling schedule was implemented. The sampling was in situ, with the logged data retrieved onsite. The sampling and data collection processes were facilitated using the HANNA instruments HI 98195 and HI 98193 probes, respectively as shown in Figure 3.13.



**Figure 3.13:** Example of HANNA probes used for the in-situ seasonal SW sampling: (a) HI 98195, and (b) HI 98193

The parameters measured include pH, EC, TDS, temperature, and DO. A physicochemical approach was utilised, given that Tchobanoglous et al. (2003) highlighted that the COD level indicates the presence of a high count of microorganisms in the wastewater, while this approach is also further supported by Gökçekuş et al. (2023). Furthermore, monitoring of the EC and TDS levels was necessary as these are essential parameters in assessing overall water quality.

### 3.4.3 Design and simulation of re-engineered WSPs

**Selection of suitable simulation software:** In general, simulation software enables and contributes towards: (i) building institutional knowledge via operator training, (ii) reducing plant life cycle costs and promoting sustainability, (iii) improving focus on resource recovery, (iv) reducing cost, and (v) encouraging compliance to stricter treatment standards. All the above criteria can be met using the Hydromantis extended simulation software tool (GPS-X 8.5). The application was selected given that it pioneered WWTP simulation and has the most extensive wastewater treatment unit process library. The tool allows for both steady-state and dynamic simulations, while various functionalities, depending on the licence, are available, e.g., analyser, optimiser, influent advisor, Sankey, mass balance, and Monte Carlo analyses.

**Design and process considerations:** The WSP as shown in Figure 3.14 function as a biological reactor, usually not fully mixed and plug flow in nature. Therefore, complex and dynamic flow paths are typical through the inlet and outlet. In general, not much data about the flow patterns or behaviour or both within the system are collected or available. Subsequently, various assumptions are normally made in this regard.



**Figure 3.14:** WSP (oxidation pond) at the Botshabelo WWTP

Amongst the four WWTPs under consideration, only the Botshabelo WWTP, as shown in Figure 3.14, has a WSP with an asymmetrical shape. The WSP also has no aerators or mixers. In the absence of aerators, only surface transfers will be present. Given that the volume and dimensions of the Botshabelo WSP shown in Figure 3.14 were not readily available, these were obtained using measurement tools within the QGIS environment. Based on these measurements, the pond's area is approximately 1 340 m<sup>2</sup>. Several iterations were incorporated into the model calibration to determine the correct (required) pond size for the simulation.

Another design consideration is the incorporation of solids removal. To remove sludge, a steady-state approximation of periodic dredging is required to account for the removal of solids. Therefore, it is essential to note that the application of modelling techniques is site specific and relies primarily on the pond's purpose: Is it for storage with no end-use, polishing, or solids removal? Therefore, it is vital to communicate with the plant operators for an enhanced understanding of the pond's purpose(s) and the observed flow pattern.

To simulate the current Botshabelo WSP, the pond as shown in Figure 3.15 was divided into a maximum of eight aerobic and anaerobic zones. These zones are regarded as a combination of ponds in series and parallel, with the length of the arrows being proportionate to the flow. Based on observation, the flow within the WSP was assumed to be clockwise.



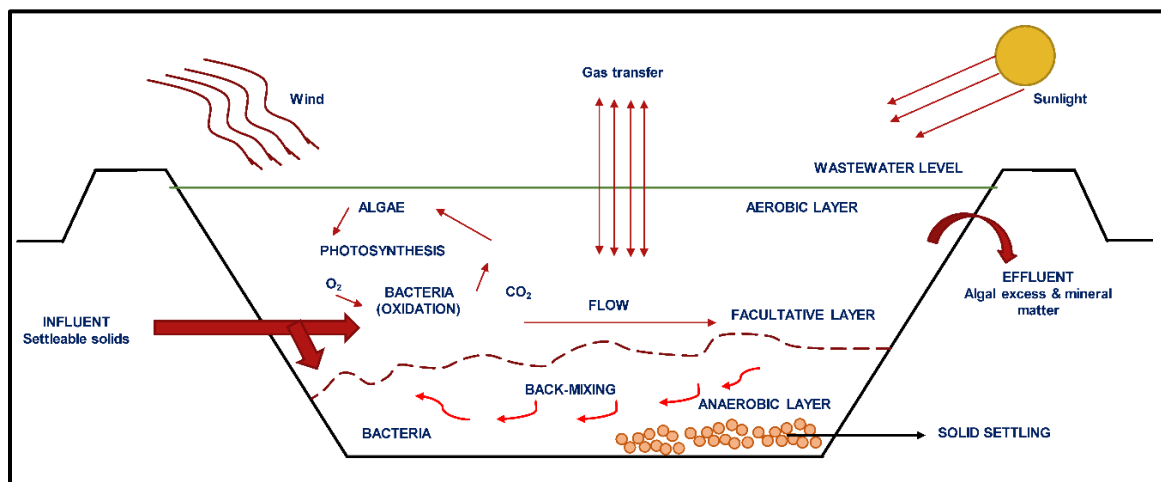
**Figure 3.15:** Re-engineered Botshabelo WSP

To re-engineer a WSP successfully, as in the case of Figure 3.15, the design considerations shown in Figure 3.16 and listed below, should be considered:

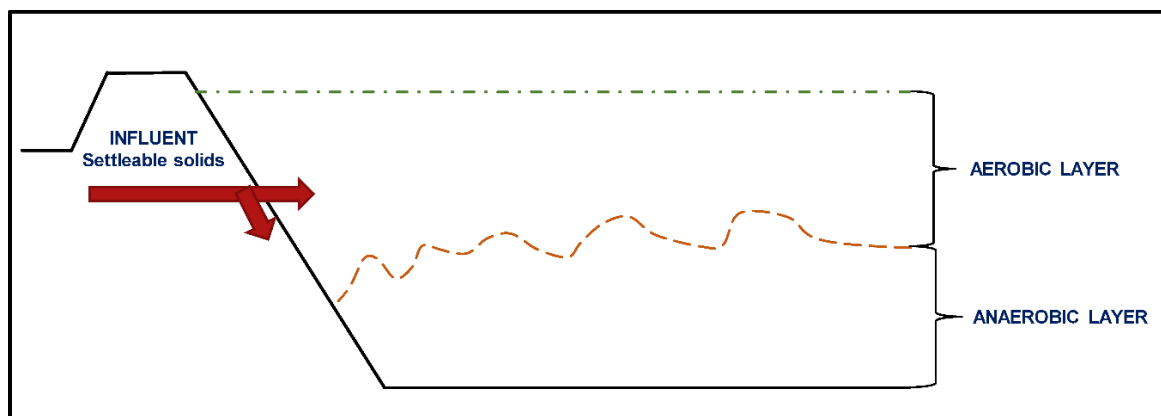
- (a) Are there any back mixings/recycles for the re-engineered WSP?

- (b) Will the effect of wind and other meteorological parameters be considered in the simulation?
- (c) How will the WSP be divided, and how will the oxygen transfer within the pond be adjusted?

Based on the design considerations listed above and shown in Figure 3.16, the identification of possible aerobic or anaerobic zones, or both, and their distribution within the system are critical; therefore, such design considerations as shown in Figure 3.17 were also implemented.



**Figure 3.16:** Schematic of the design considerations for a re-engineered WSP (after Frédéric, 1999)

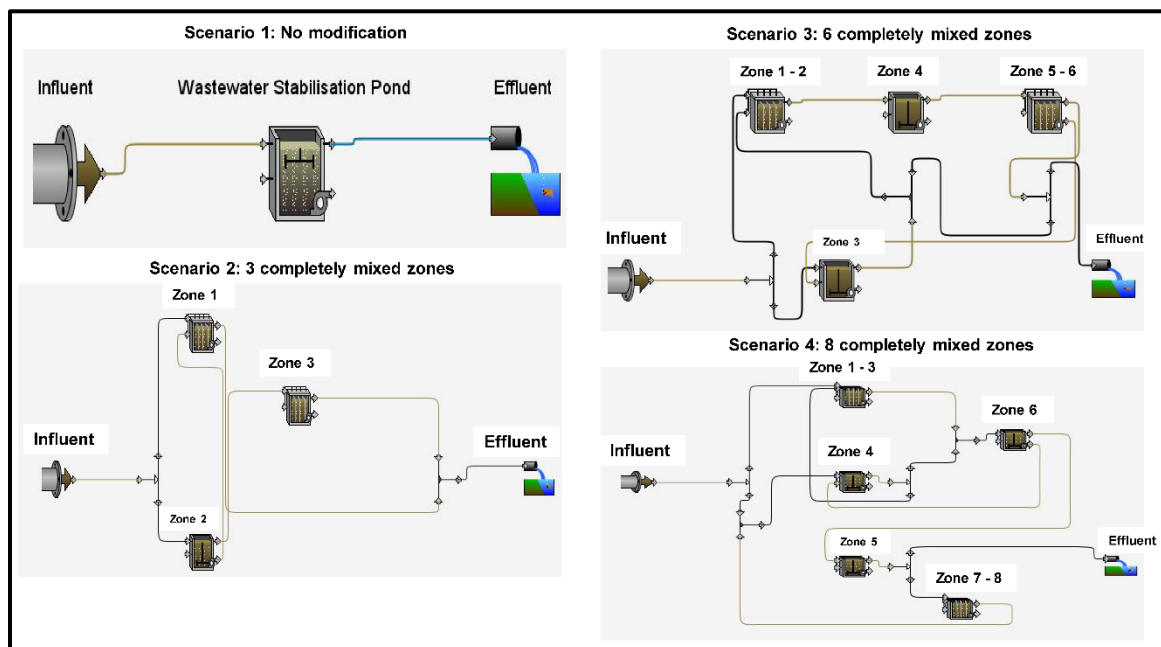


**Figure 3.17:** Schematic showing the aerobic and anaerobic zones of a re-engineered WSP

In Figure 3.17, the effect of wind is assumed to be minimal. Hence, the pond design is mostly anaerobic. In this research, four different scenarios were

considered (*cf.* Figure 3.18), given that this ‘optimum’ number of scenarios are suggested in literature, e.g., Shilton and Mara (2005), Abbas et al. (2006), and Kilian et al. (2024).

Although the simulation scenarios shown in Figure 3.18 are illustrative of the Botshabelo WSP, a similar approach was adopted at all the other WWTPs under consideration. Typically, influent data from the four WWTPs are considered when the simulation focus is on the pre-treatment of wastewater as influent. However, if the re-engineered WSPs are to be used for post-treatment, the effluent data from the four WWTPs will be used for the simulation. In either case, four design scenarios were deemed necessary. However, the WSP was not modified in Scenario 1 for both the pre- and post-treatment. Scenario 2 had the WSP divided into three zones with continuous stirred-tank reactors (CSTRs), thereby ensuring that some areas within the pond are anaerobic. The same process was followed for Scenarios 3 and 4, with the difference being the division of the pond into six and eight zones, respectively. However, before simulating the re-engineered WSP, the model was calibrated to match the input data.



**Figure 3.18:** Example of the four simulation scenarios at the Botshabelo WSP (after Shilton and Mara, 2005; Abbas et al., 2006; Kilian et al., 2024)

**Model calibration and assumptions:** The default parameters for typical municipal wastewater influent for the model were calibrated to match the influent

and effluent data from the four WWTPs before simulating the re-engineered WSP. The base parameters included, but were not limited to, pH, COD, alpha factor, and a reduction factor for denitrification on nitrate-N. The following assumptions were made during the simulation of the re-engineered WSPs:

- (a) The surface area of each zone within a WSP was estimated using the QGIS measurement tool.
- (b) The depth of the pond was 4m.
- (c) The back mixing ratio was 1:1.
- (d) The location of the aerators remained fixed during the calibration period.

**Data analyses:** The experimental results obtained from the influent and effluent, surface water, and WSP simulations were analysed using Microsoft Excel, OriginPro 9.0, and IBM SPSS Statistics 27. The experimental results were compared to those obtained from simulating the re-engineered WSPs. Data visualisation was done with the aid of OriginPro 9.0 and GPS-X 8.5. In general, all the data obtained from the four WWTPs were statistically analysed to check for variability within the WWTPs and to identify the factors influencing poor effluent quality.

The results pertained to the monitoring and analyses of influent and effluent parameters at conventional WWTPs, the impact of effluent non-compliance on SW sampling sites, and the design and simulation of re-engineered WSPs are included and discussed in the following three chapters, i.e., Chapters 4 to 6.

## CHAPTER 4 : PHYSICOCHEMICAL WASTEWATER PARAMETERS

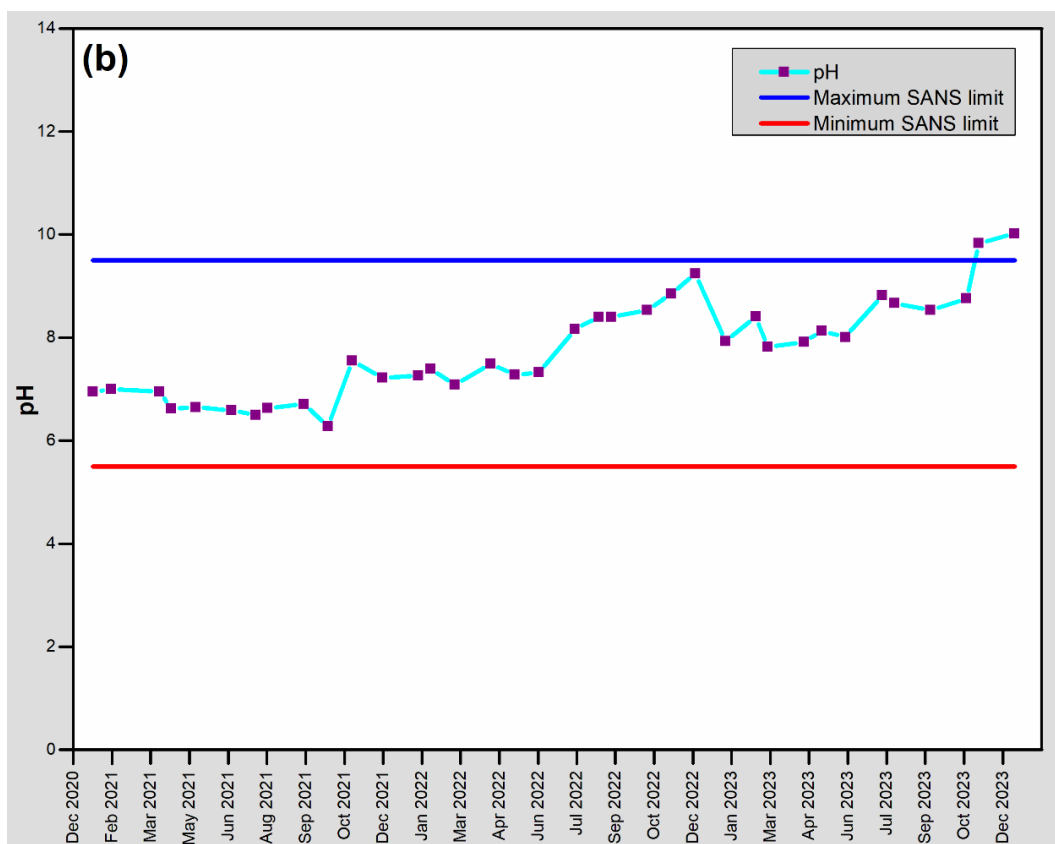
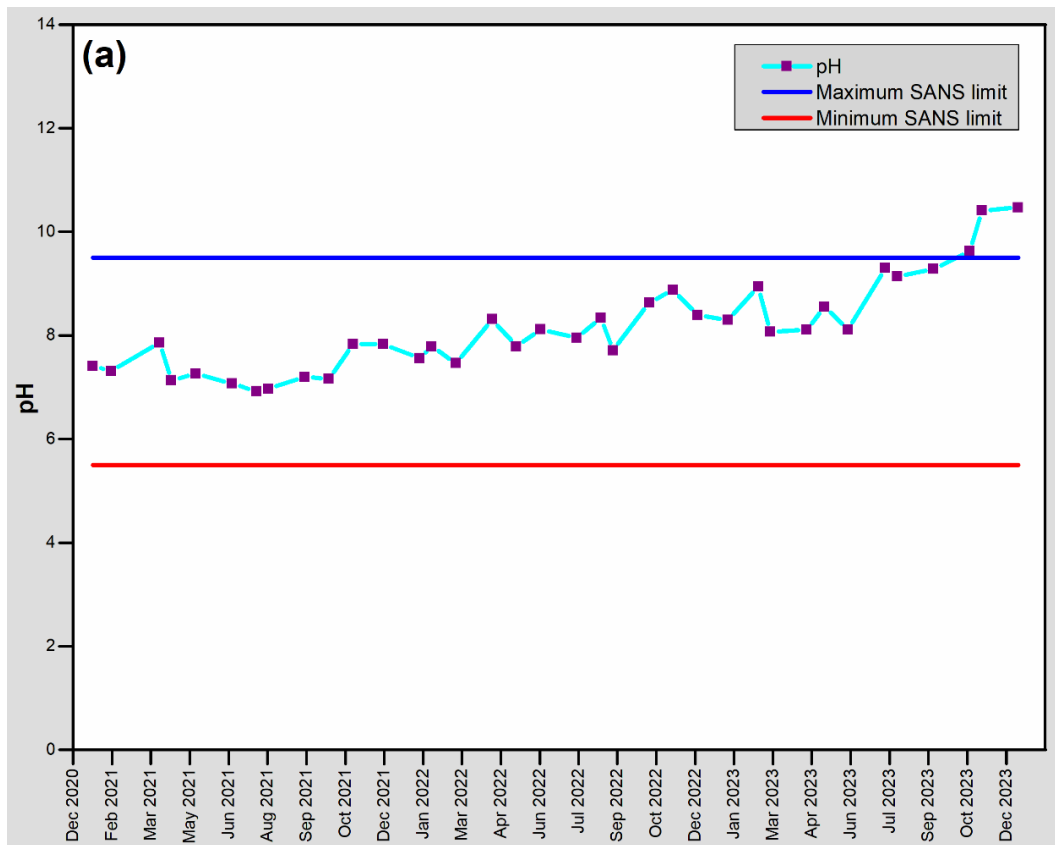
This chapter presents the analyses and discussion of the results obtained from the four WWTPs towards achieving the following specific objectives as outlined in Chapter 1:

- (a) Establish the influence of influent and effluent physicochemical parameters in diagnosing and addressing operational challenges at WWTPs; and
- (b) Develop a cost-effective approach to sample and determine the various physicochemical parameters by considering the UN SDGs 3, 6, and 14.

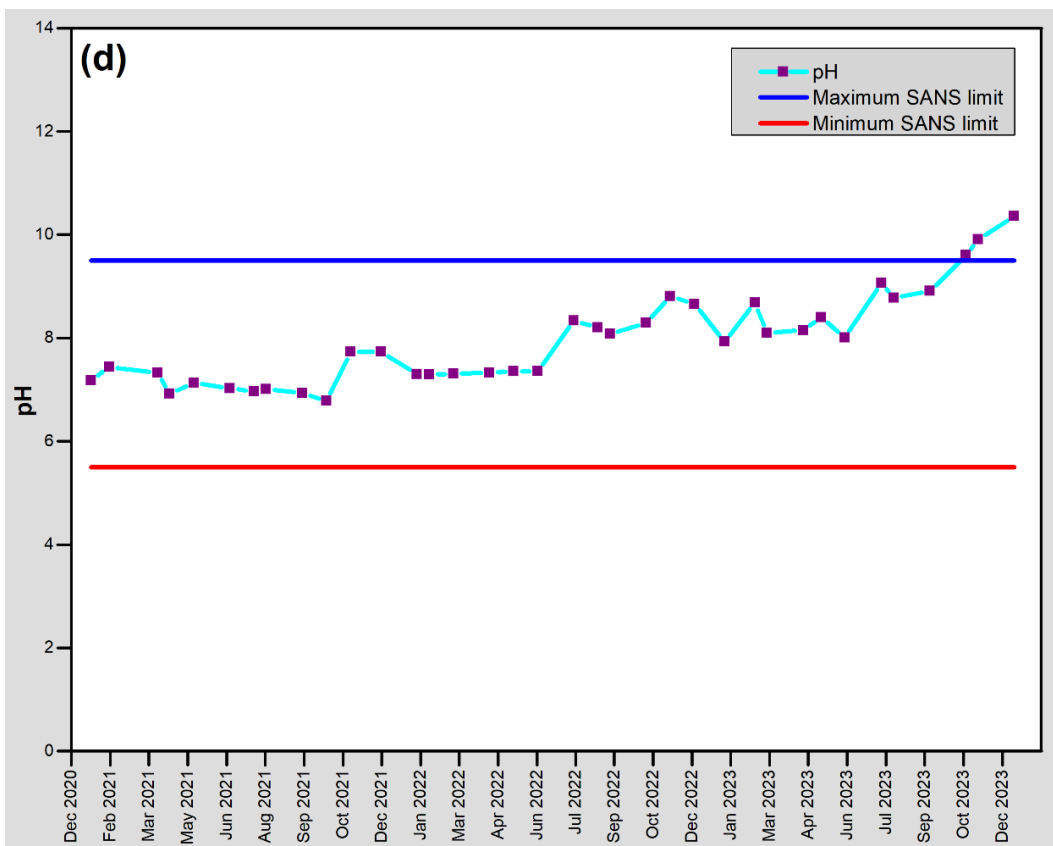
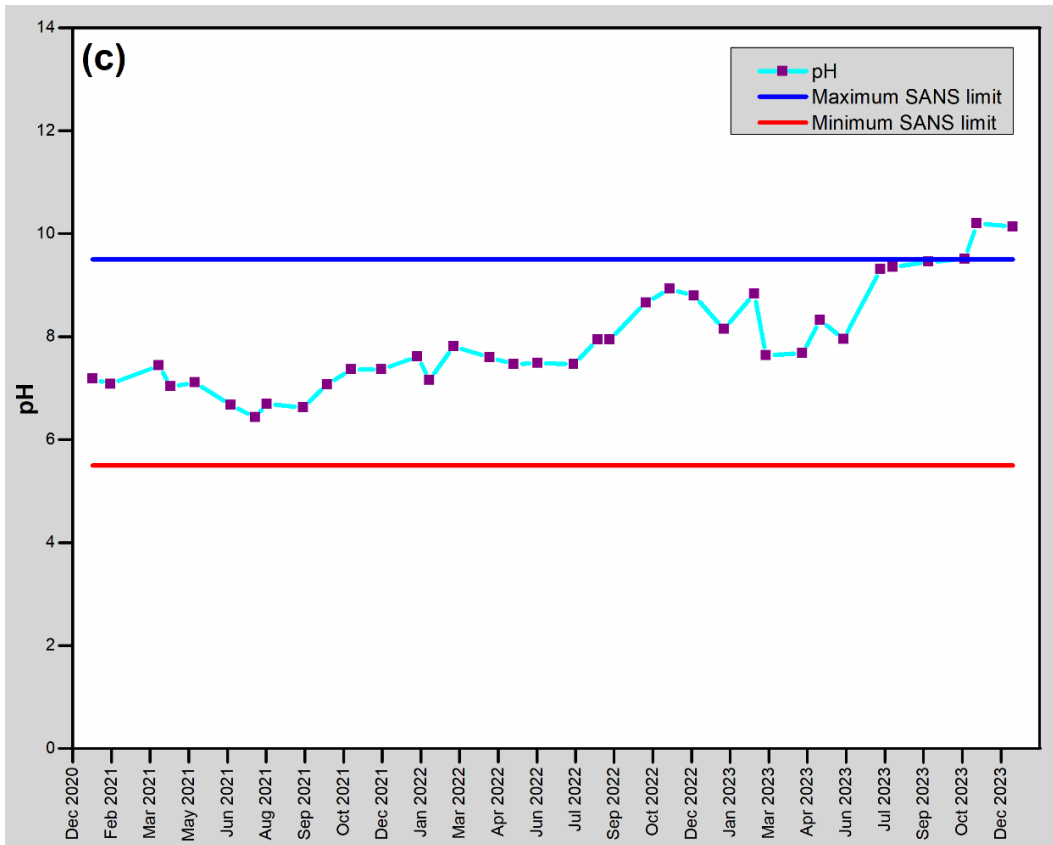
The physicochemical parameters are presented and discussed to assist operators and managers of WWTPs in addressing operational challenges and the potential environmental and health effects. Secondly, statistical analyses of the wastewater effluent were conducted to determine which physicochemical parameter(s) influence the dependent (indicator) variable, i.e., COD.

### 4.1 Monthly Performance of WWTPs

The monthly performance of the Bainsvlei, Botshabelo, North-East, and Sterkwater WWTPs was evaluated over a three-year period in terms of pH, EC, TDS, temperature, COD, OA, ammonia, nitrate-nitrite, phosphate, SS, MLSS, settleable solids, and SVI. It is imperative to note that physicochemical parameters, especially pH, can alter the removal of viruses present in wastewater (Saba et al., 2021). Consequently, the pH of the wastewater effluent at the four WWTPs was measured and is presented in Figures 4.1 (a – d).



**Figure 4.1:** Monthly observed pH values at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs



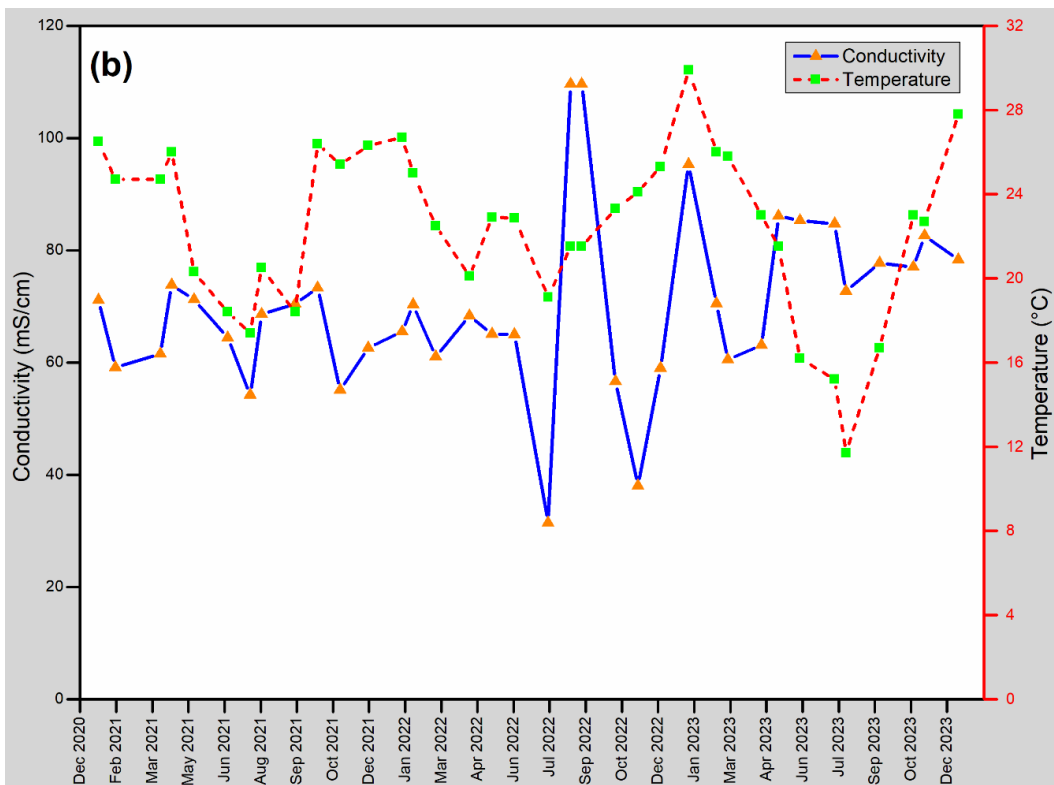
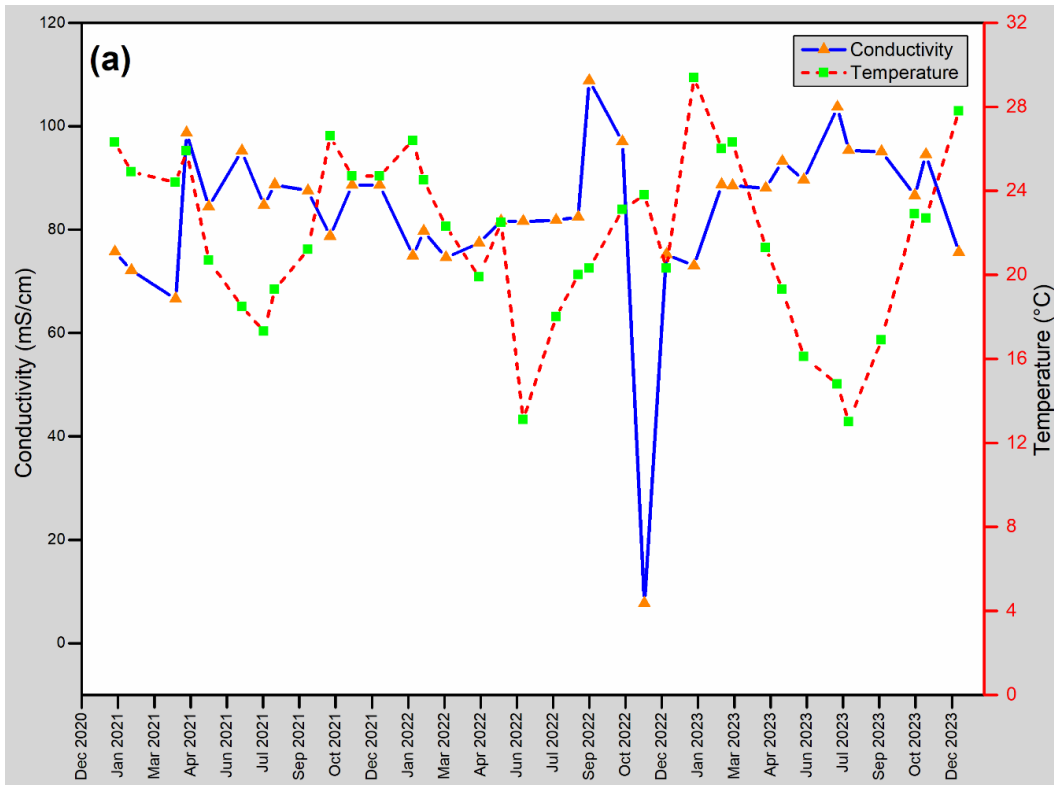
**Figure 4.1:** Monthly observed pH values at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

The influent and effluent pH levels for the Bainsvlei WWTP in Figure 4.1 (a) fluctuated between 6.5 to 10 and 6.9 to 10.5, respectively. At the Botshabelo WWTP in Figure 4.1 (b), the pH values fluctuated between 6.6 to 10.1 (influent) and 6.3 to 10 (effluent). At the North-East WWTP depicted in Figure 4.1 (c), the pH values fluctuated between 6.3 to 10.1 (influent) and 6.4 to 10.2 (effluent), while at the Sterkwater WWTP presented in Figure 4.1 (d), the pH values fluctuated between 6.6 to 10.2 (influent) and 6.8 to 10.4 (effluent). In general, it is evident from Figure 4.1 that the pH values tend to increase from spring to a maximum in summer at all the WWTPs, a trend which concurs with the findings of Hridoy et al. (2025). Hence, it is confirmed that pH levels tend to increase with an increasing water temperature, while Saba et al. (2021) also highlighted that such fluctuating temperature and pH levels can impact the viral removal rate in wastewater effluent.

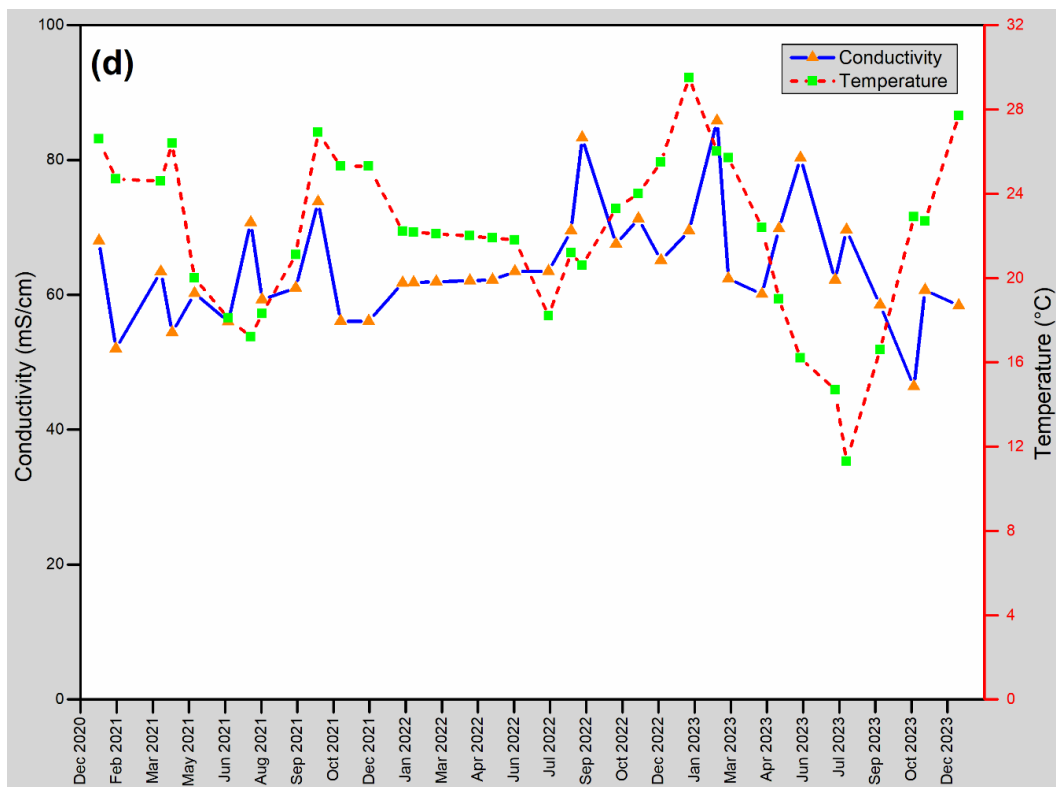
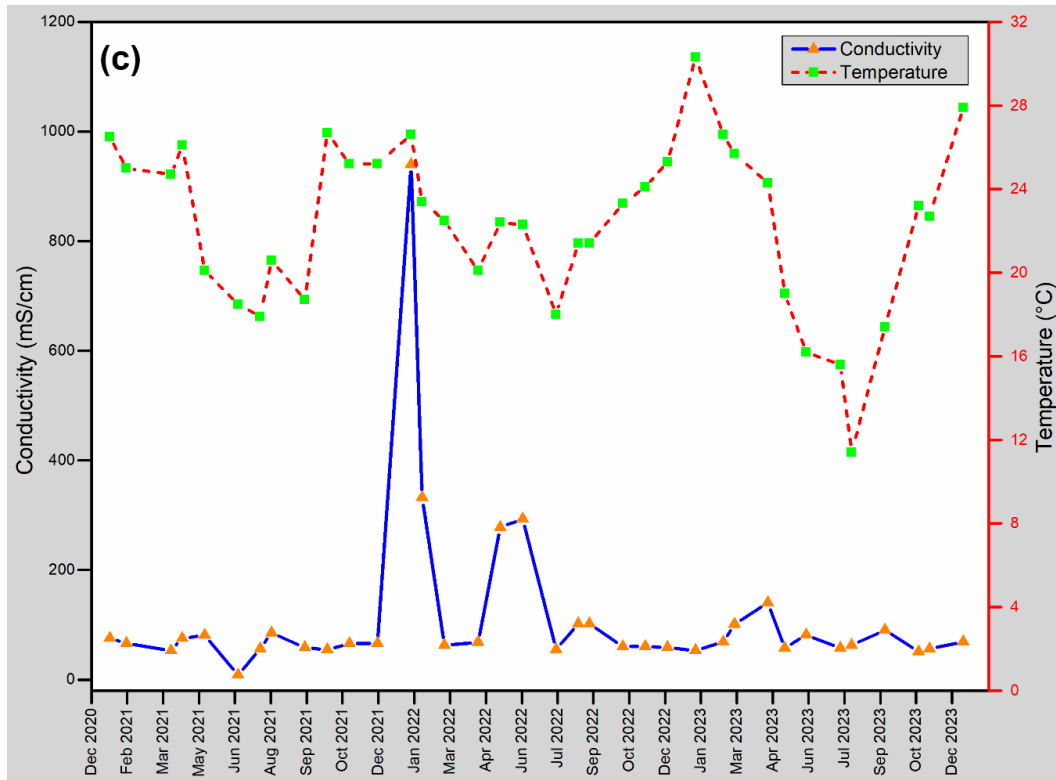
The EC of wastewater effluent defines the ability of the effluent to conduct electricity (Kumar et al., 2023). This often depends on the concentration and mobility of dissolved ions in the effluent. According to Van der Hoven et al. (2017), higher evaporation rates lead to higher EC values. This phenomenon indicates a temperature dependency for EC. However, the relationship between temperature and EC is not directly proportional; an increase in temperature would normally lead to an increase in EC as the ions in the effluent become mobile. Therefore, the temperature of the influent wastewater and the corresponding EC values of the four WWTPs are presented in Figures 4.2 (a – d).

In Figure 4.2, it is evident that during certain months, an increase in temperature did not translate into EC increases, while in some months, EC increased despite no substantial increases in temperature.

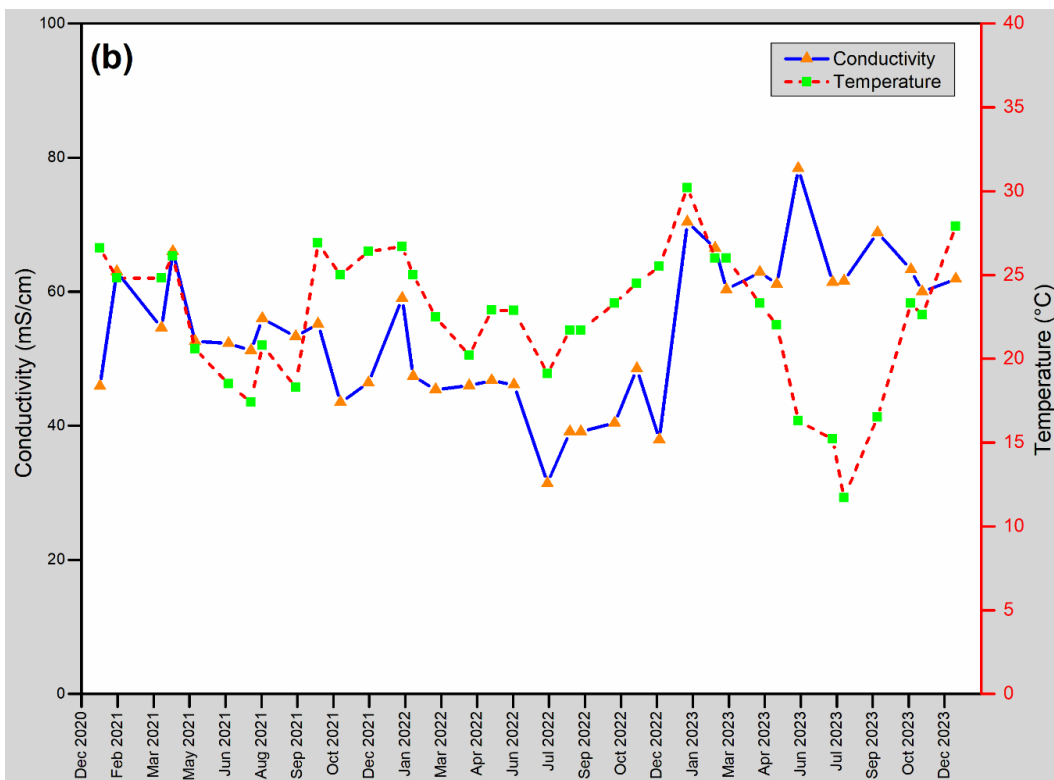
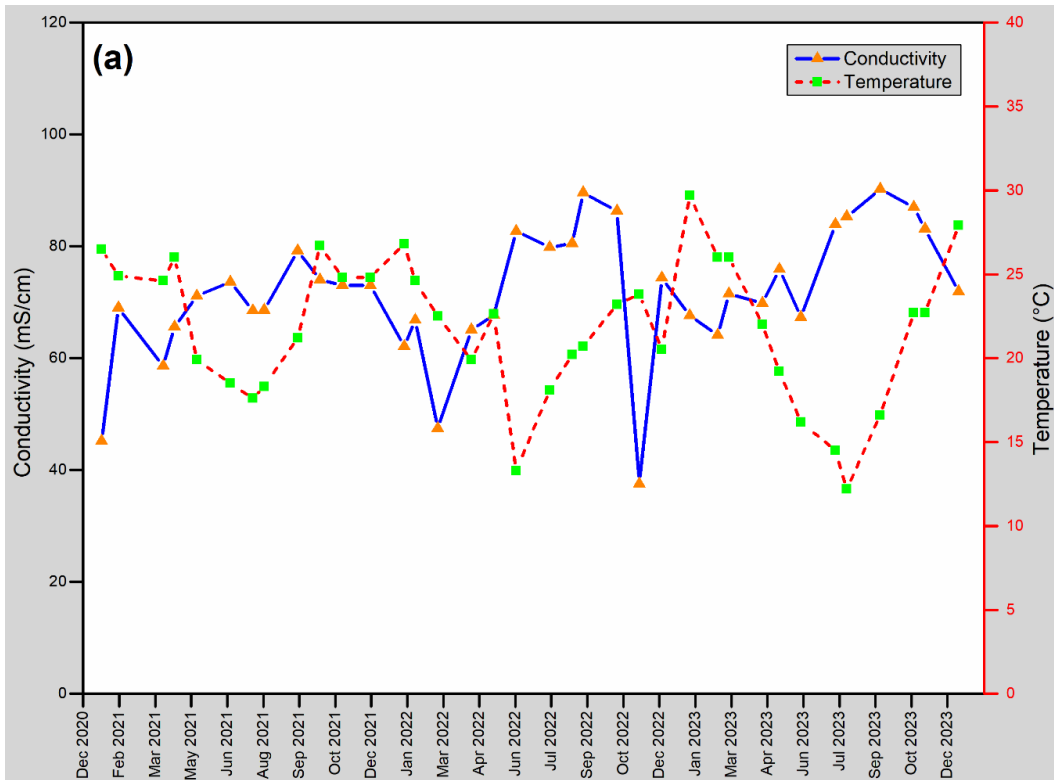
After treating the influent, the effluent results (temperature and EC) at the four WWTPs are shown in Figures 4.3 (a – d).



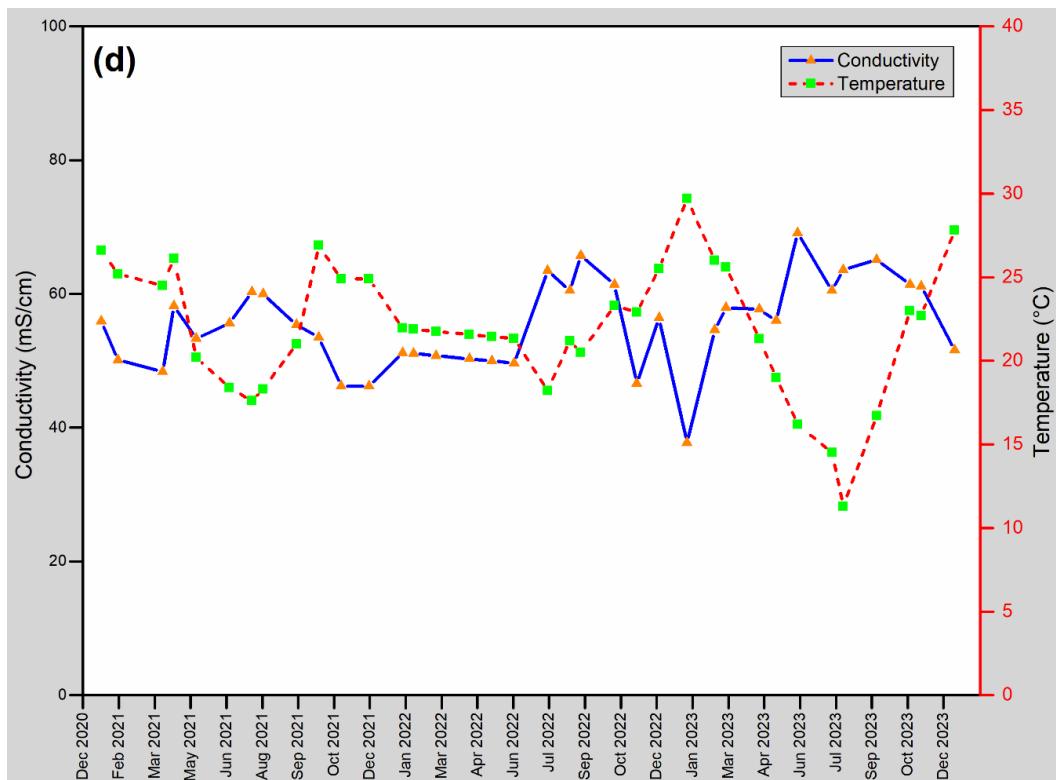
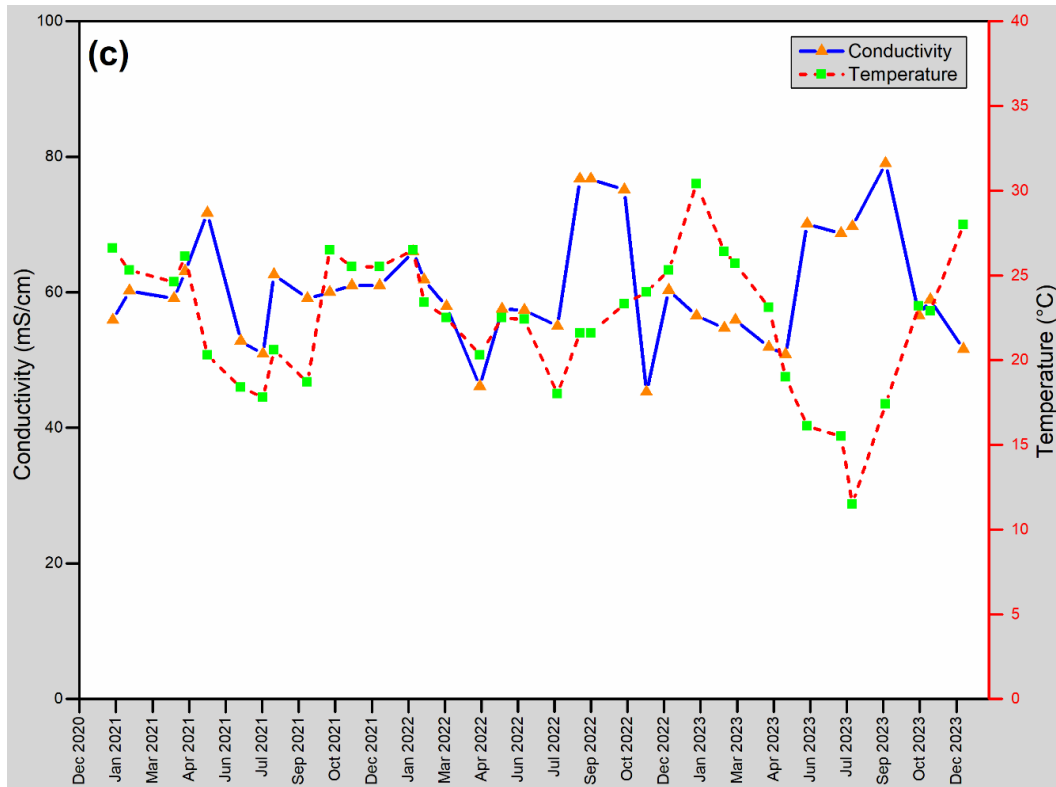
**Figure 4.2:** Wastewater influent temperature and EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTTPs



**Figure 4.2:** Wastewater influent temperature and EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs



**Figure 4.3:** Wastewater effluent temperature and EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTWs



**Figure 4.3:** Wastewater effluent temperature and EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

It is evident from Figure 4.3 (a) that the minimum and maximum temperatures and EC values for the Bainsvlei WWTP effluent varied between 12.2 and 29.7°C and 37.6 and 90.2 mS/cm, respectively. At the Botshabelo WWTP in Figure 4.3 (b), a minimum temperature and EC value of 11.7°C and 31.4 mS/cm, respectively, are recorded., while the maximum temperature and EC values are 30.2°C and 78.4 mS/cm, respectively. Similarly, the temperature and EC values for the North-East and Sterkwater WWTPs varied between 11.3 and 29.7°C, and 37.7 and 79 mS/cm, respectively. Based on the above results, it is evident that a direct comparison between temperature and EC could not be established. This is also in agreement with the findings made by Shrestha et al. (2023).

Subsequently, the EC was adjusted via temperature compensation at 25°C using Equation 4.1 (Barron and Ashton, 2005) to aid the comparison between temperature and EC as shown in Figures 4.4 (a – d).

$$\sigma_{25} = \sigma_T \left[ \frac{1}{1 + \alpha(T - 25)} \right] \quad (4.1)$$

Where:

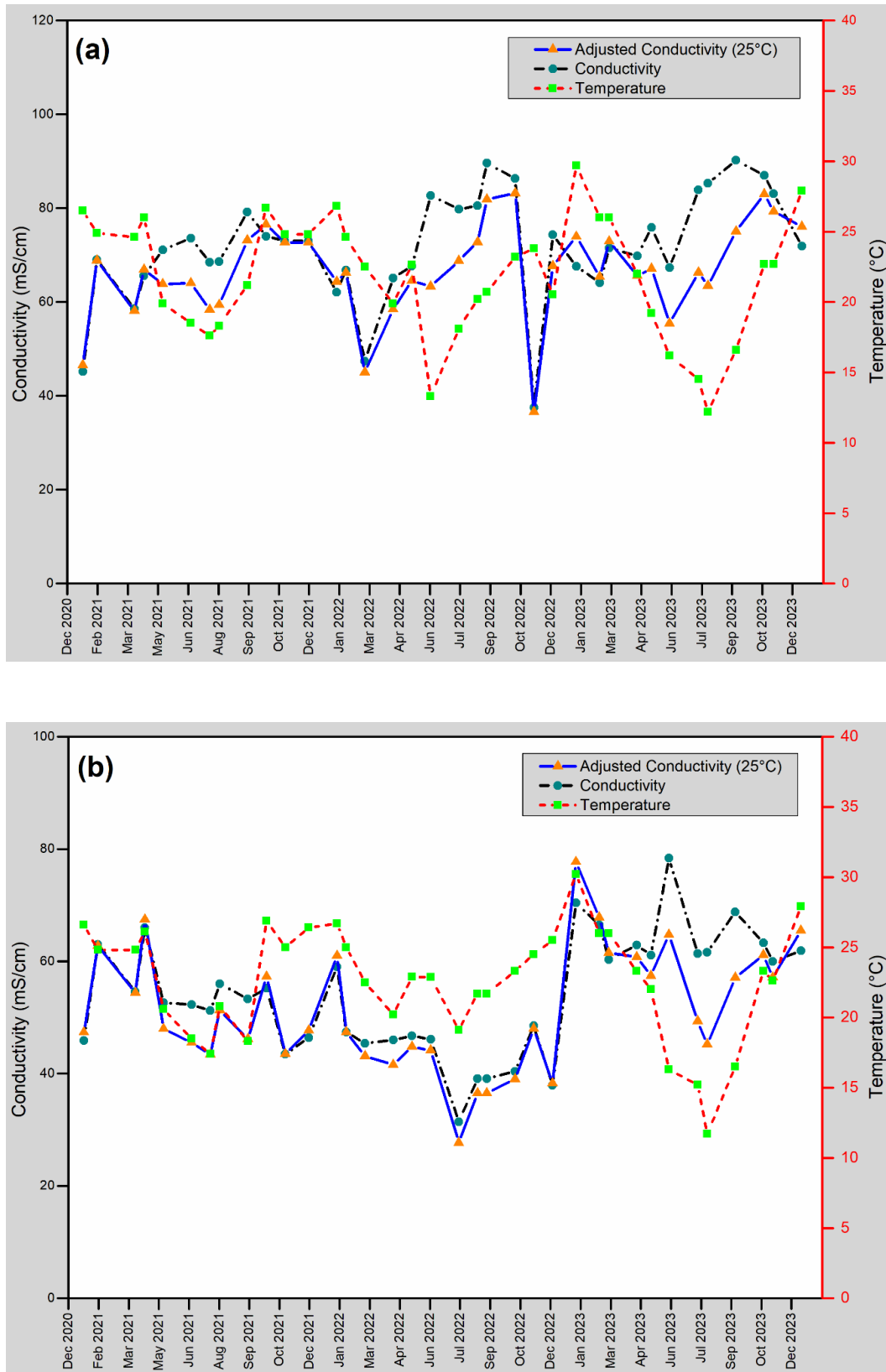
$\sigma_{25}$  = adjusted conductivity at 25°C (mS/cm),

$\alpha$  = temperature coefficient (0.02°C<sup>-1</sup>),

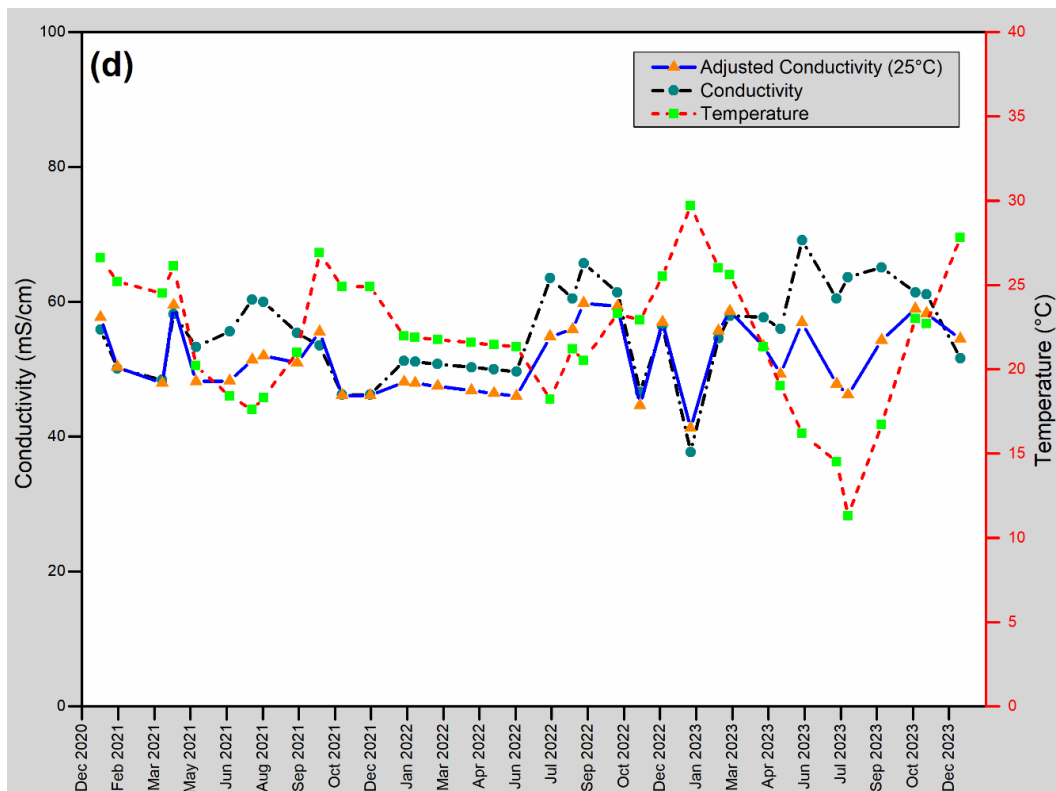
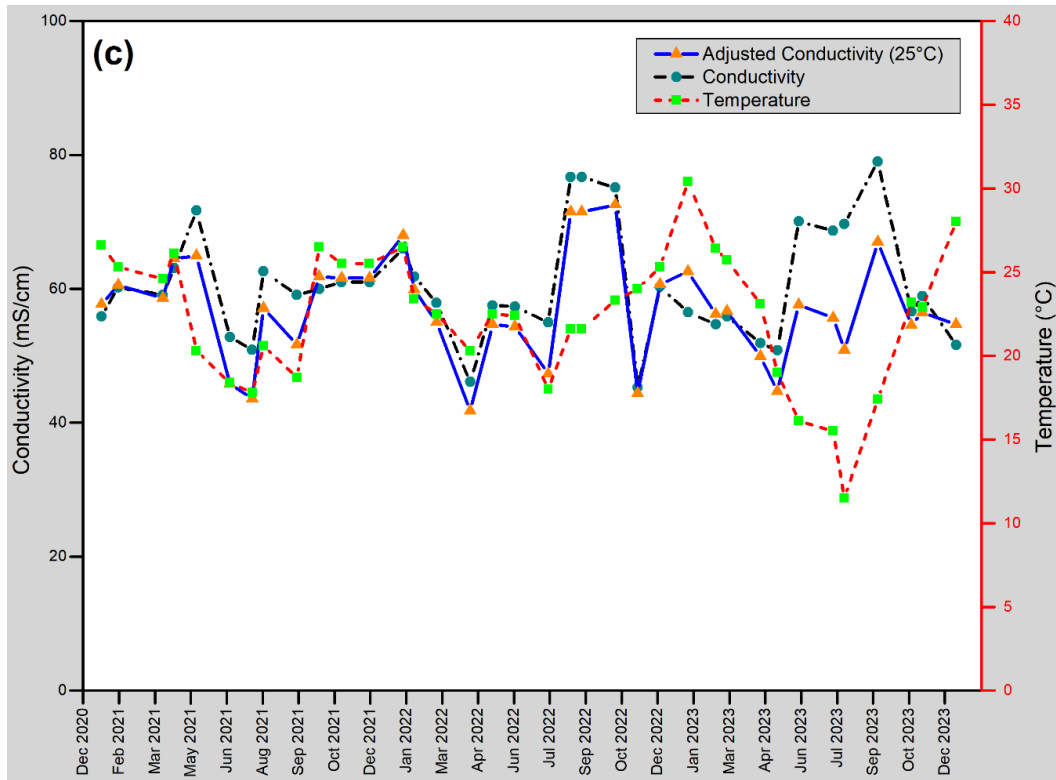
$\sigma_T$  = measured conductivity at temperature  $T$  (mS/cm), and

$T$  = observed effluent temperature (°C).

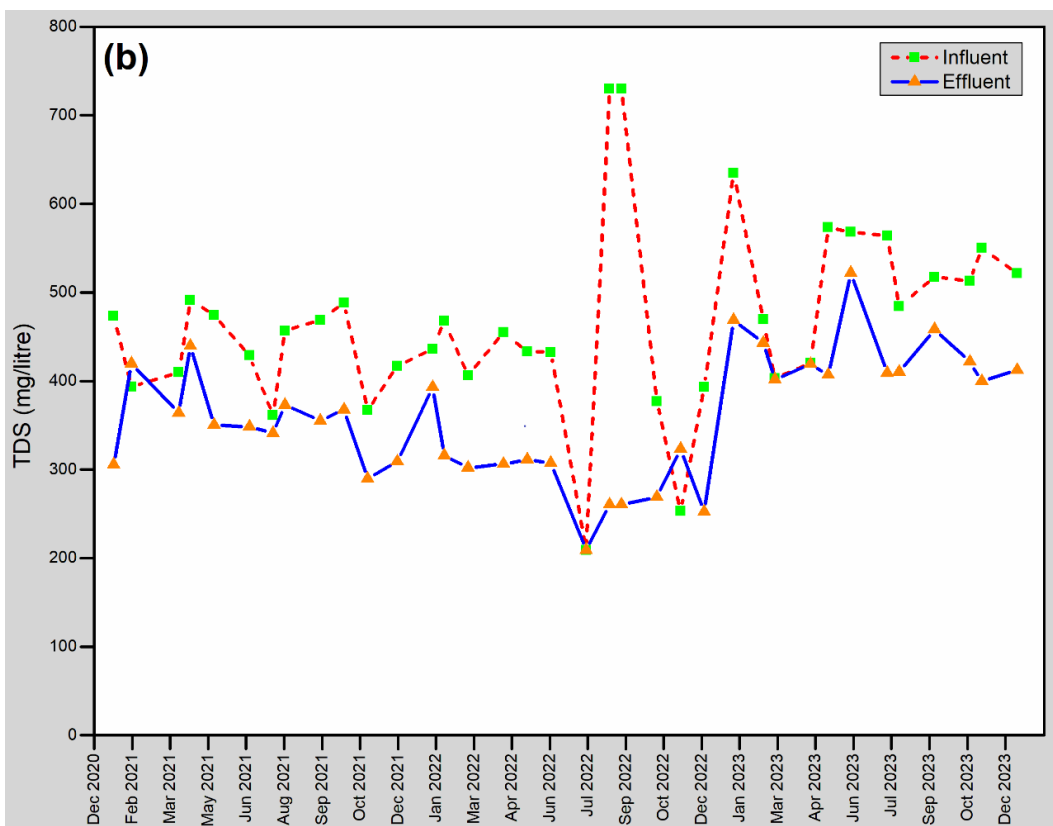
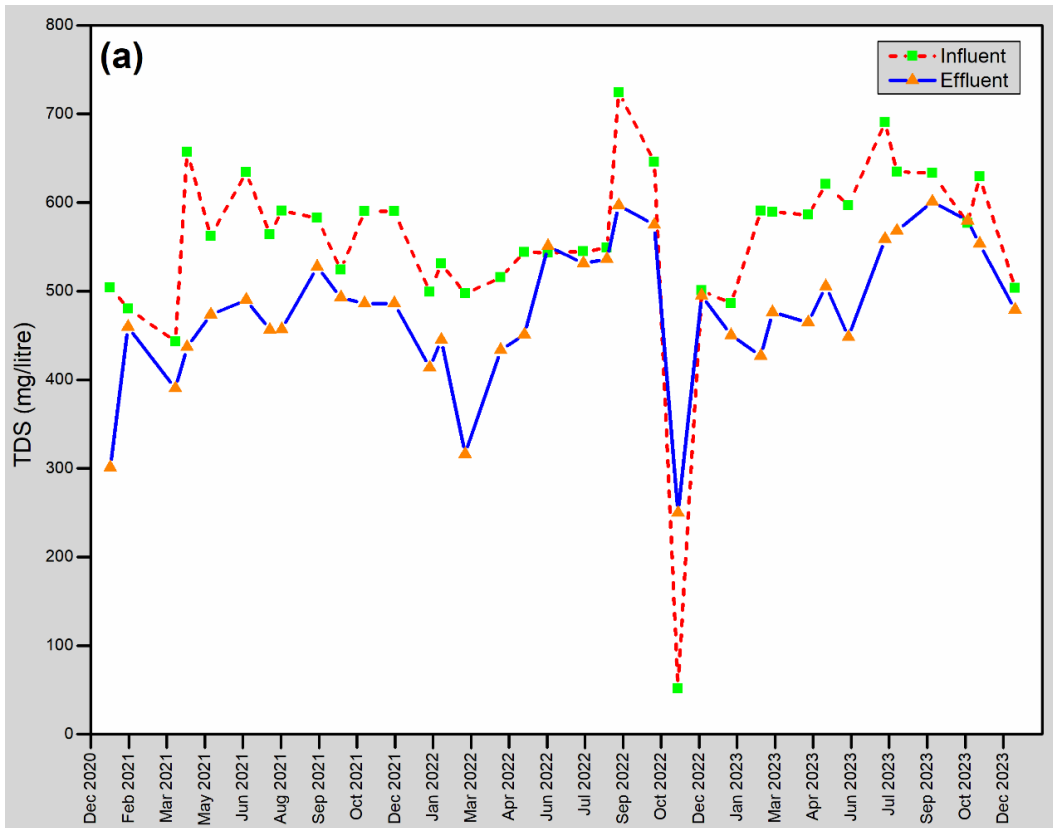
In applying Equation 4.1, a direct comparison can be made between temperature and the adjusted EC of the effluent wastewater. In general, in Figure 4.4, it is evident that an increase in temperature is associated with an increase in EC. However, Kumar et al. (2023) and Corbett et al. (2025) demonstrated that TDS also has an impact on the EC of effluent wastewater. Subsequently, the TDS (using Eq. 3.1) for the influent and effluent wastewater are also presented in Figures 4.5 (a – d). Typically, effluent TDS consists of a combination of dissolved organic and inorganic materials and is always expected to be less than influent TDS.



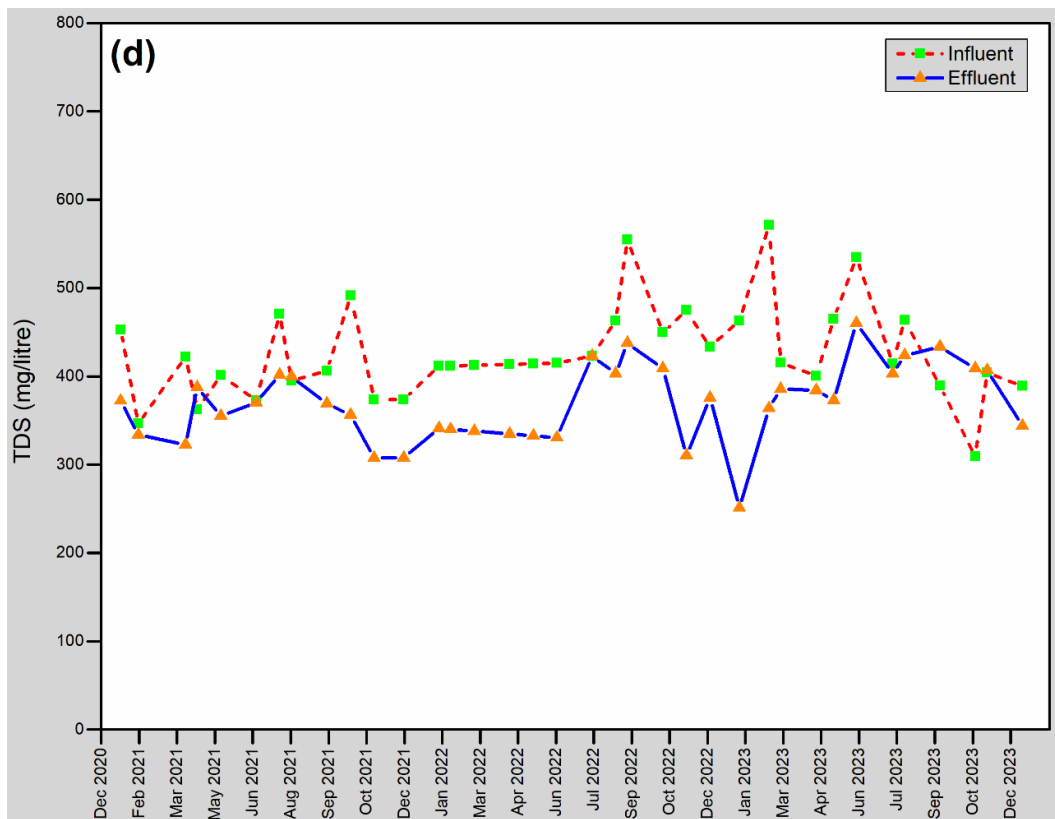
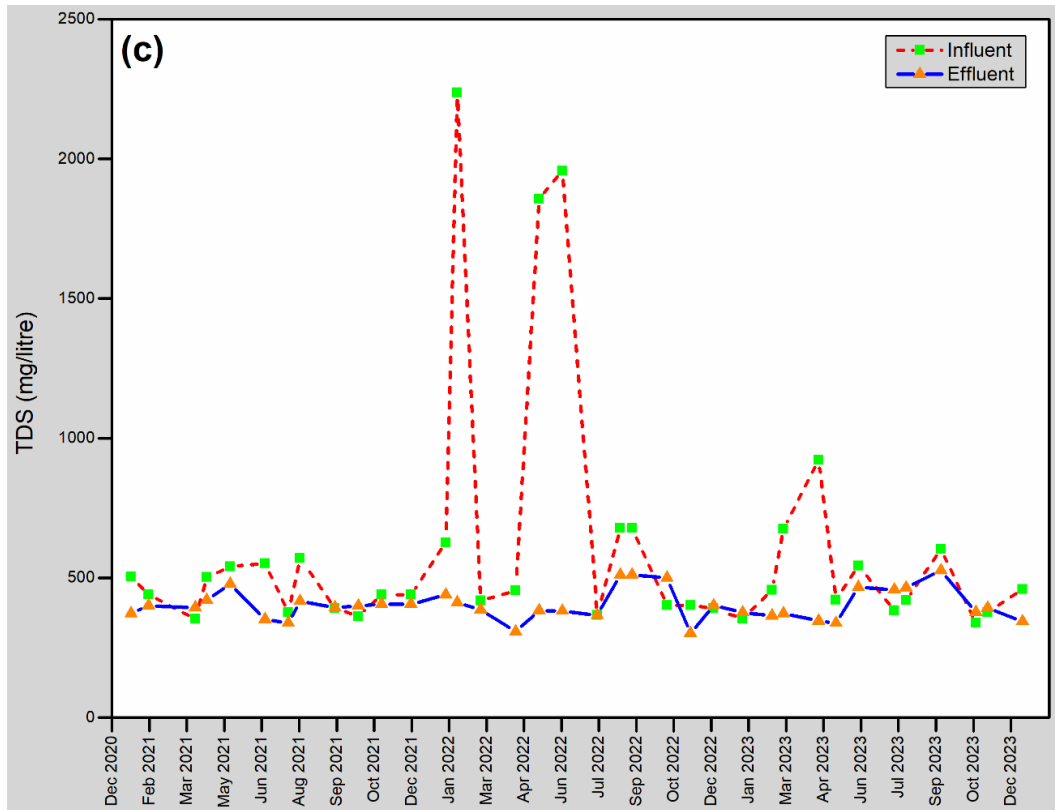
**Figure 4.4:** Wastewater effluent temperature and adjusted EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTs



**Figure 4.4:** Wastewater effluent temperature and adjusted EC at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs



**Figure 4.5:** Influent and effluent wastewater TDS at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs



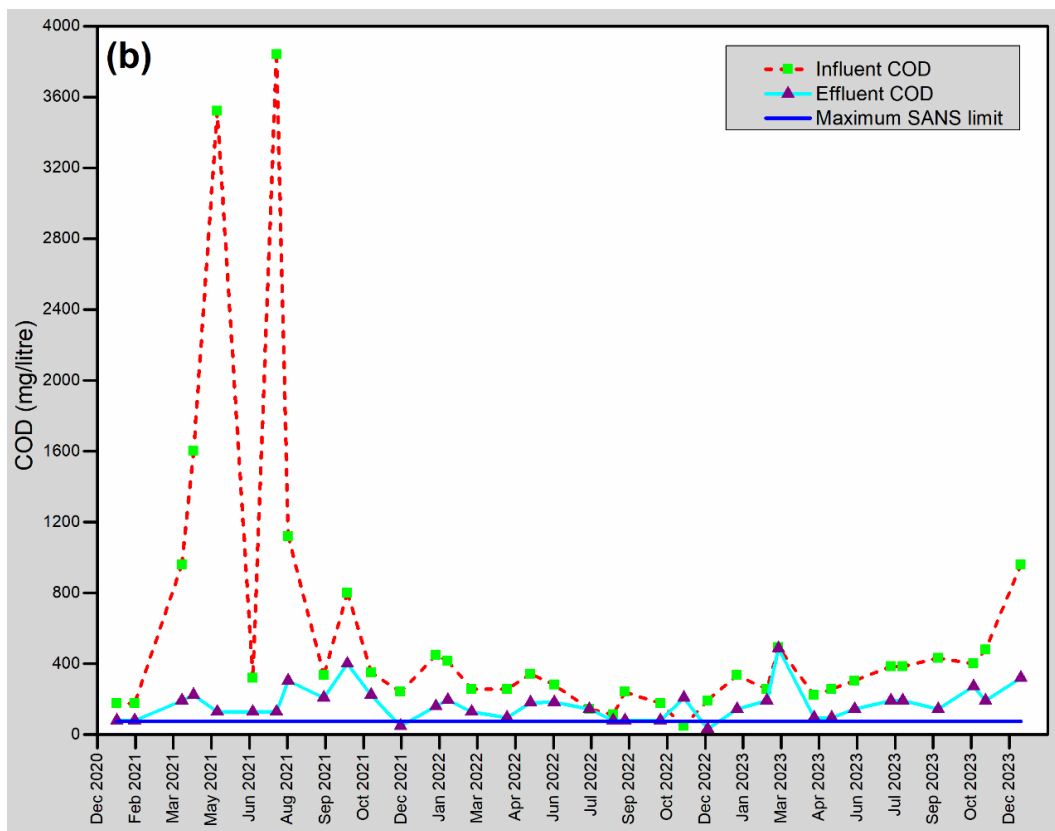
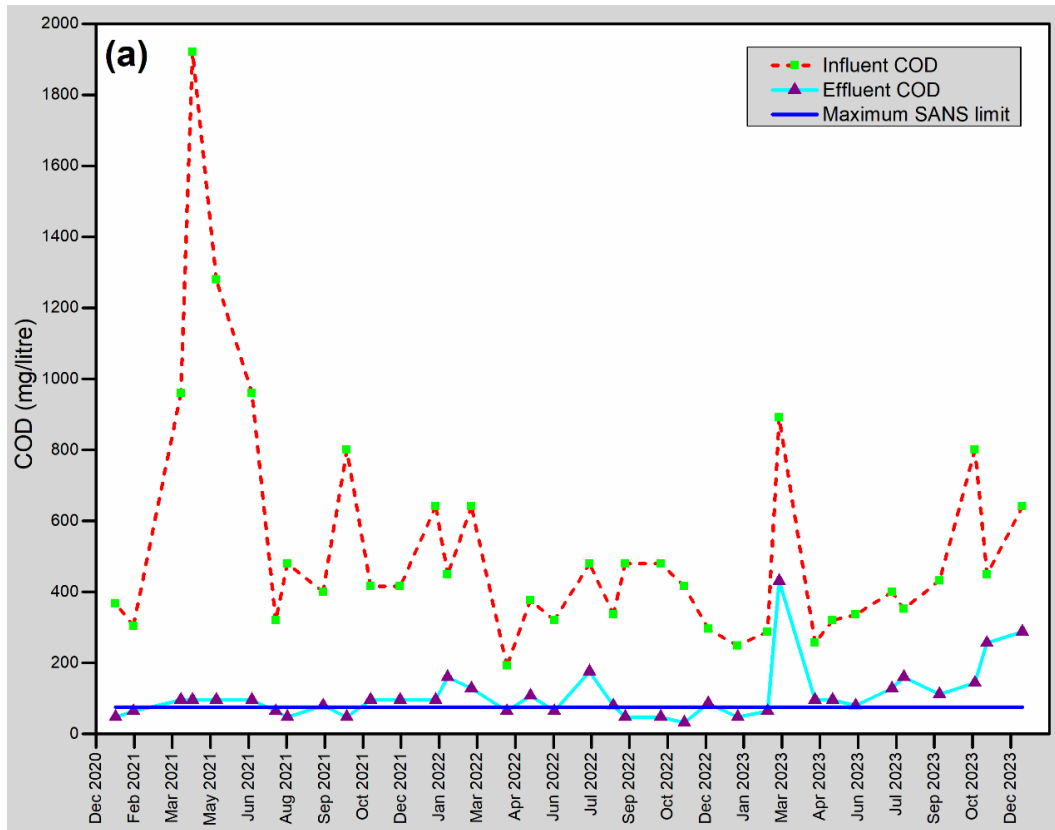
**Figure 4.5:** Influent and effluent wastewater TDS at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

After treatment, the TDS levels were reduced, but they fluctuated monthly. The lowest effluent TDS at the Bainsvlei WWTP was 249.8 mg/l, while the maximum was 600.7 mg/l. The Botshabelo WWTP had a minimum effluent TDS of 209.1 mg/l, while the maximum was 522.1 mg/l. Similarly, the effluent TDS at the North-East WWTP varied between 301.7 mg/l and 526.1 mg/l, while at the Sterkwater WWTP, the effluent TDS varied between 251.1 mg/l and 460.2 mg/l. It should be noted that the maximum TDS levels at all the WWTPs exceed 500 mg/l, except at Sterkwater. However, the maximum TDS reported in this research falls within the chronic threshold recommended by Brent et al. (2022). According to Brent et al. (2022), the chronic TDS threshold is 463 mg/l, while the acute TDS threshold is 938 mg/l. Subsequently, there is an imminent threat to aquatic life when considering the maximum TDS thresholds of discharged effluents into the surface waters present in the MMM.

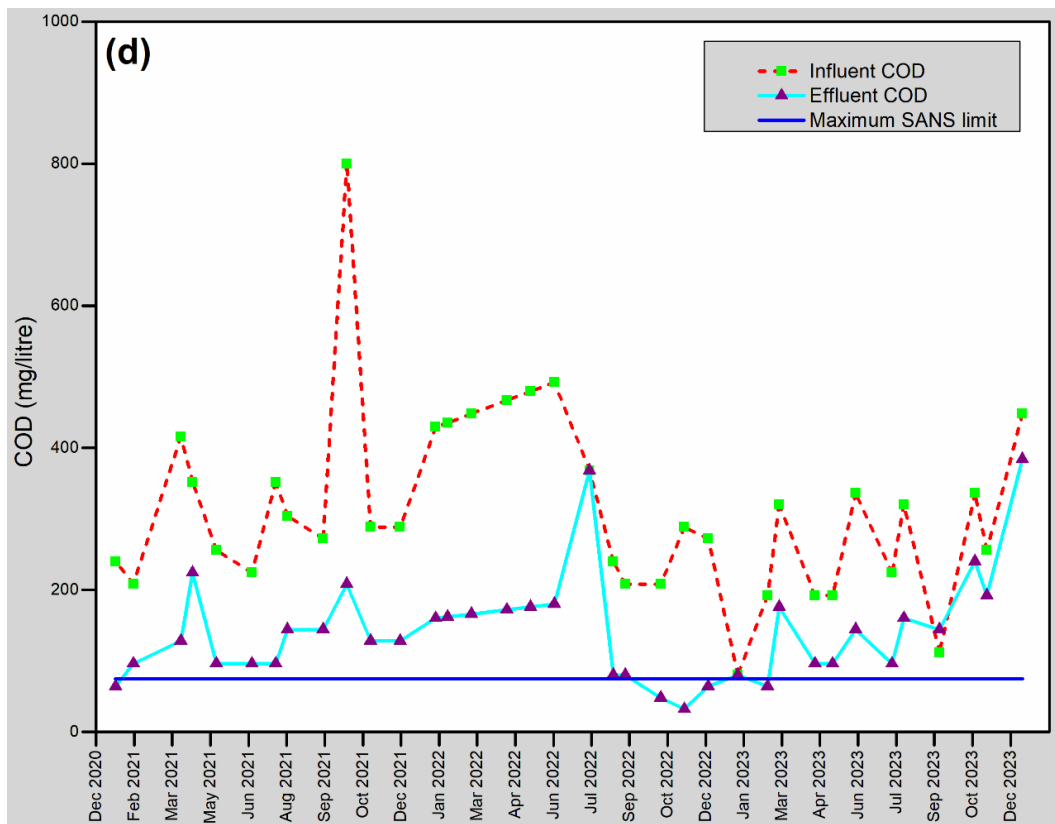
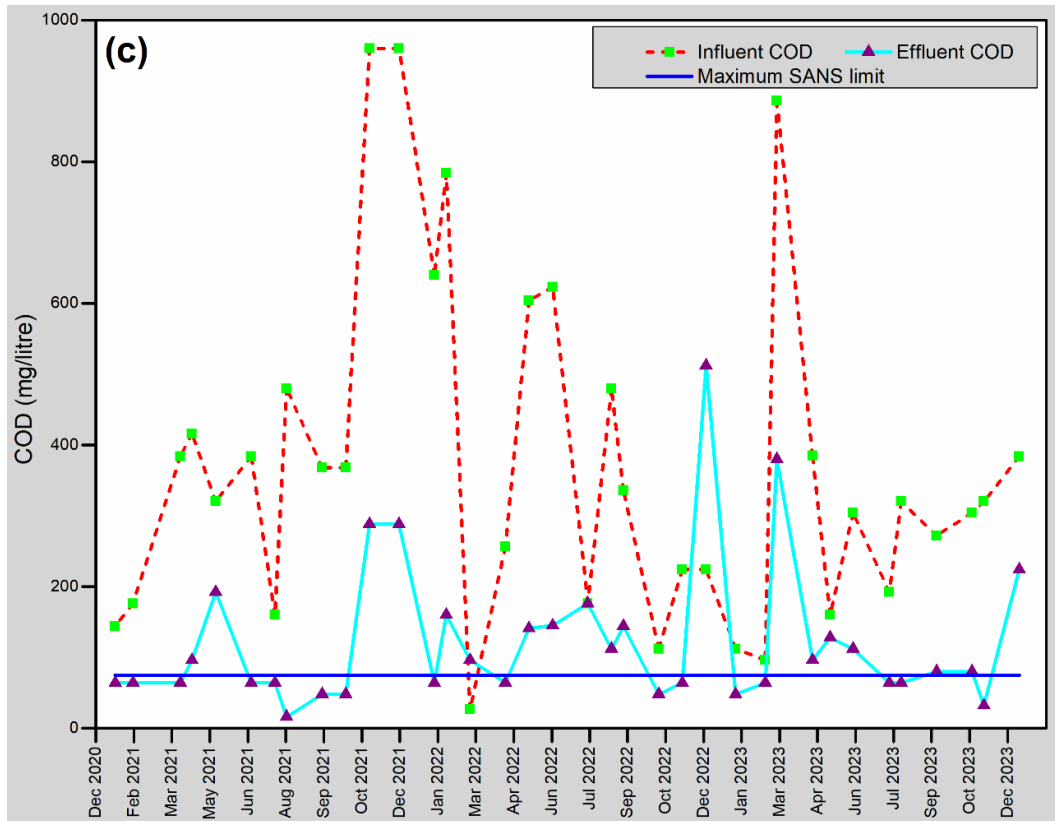
#### 4.2 COD Analyses of Influent and Effluent from WWTPs

The COD is the most critical physicochemical parameter for the sampled WWTPs in the MMM. The justification for selecting COD as dependent (indicator) parameter was already included in Chapters 1 and 3.

In Figure 4.6, the maximum influent COD for all the WWTPs during the sampling period ranged between 800 mg/l and 3 840 mg/l. After treatment, the lowest effluent COD for all the WWTPs ranged between 16 mg/l in winter and 32 mg/l in spring. Despite the acceptable COD levels after treatment, it is important to note that only a few months had COD levels conforming to the SANS criterion, i.e., 75 mg/l. For example, at the Bainsvlei WWTP, the COD ranged between 32 mg/l (November) and 431 mg/l (July), with the lowest and highest values been recorded in the second and third year, respectively. The highest COD value exceeded the SANS criterion by 140.7%. At the Botshabelo WWTP, the COD ranged between 32 mg/l (December) and 486 mg/l (July); hence, exceeding the SANS COD of 75 mg/l with 146.5%. Contrasting seasonal trends were evident at the North-East and Sterkwater WWTPs, with the COD ranging between 16 and 32 mg/l (August), and 512 and 384 mg/l (December).

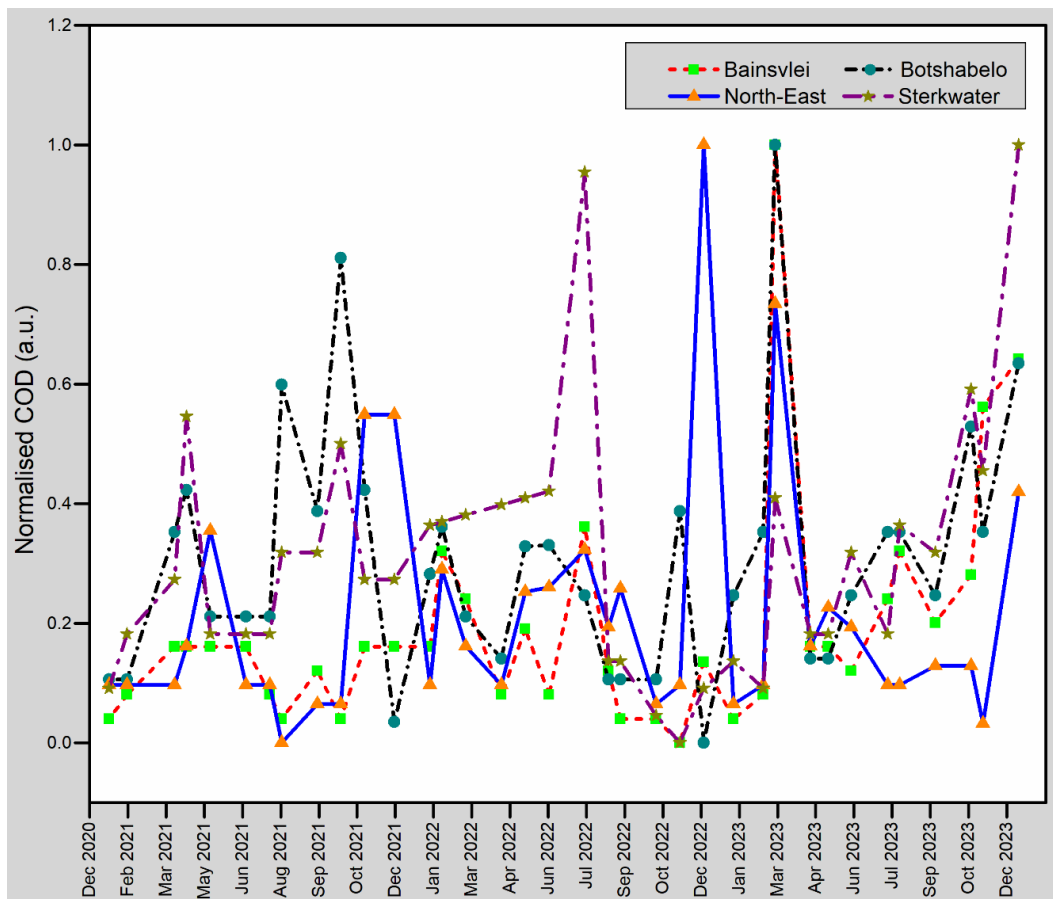


**Figure 4.6:** Influent and effluent wastewater COD at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTWs



**Figure 4.6:** Influent and effluent wastewater COD at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

The high percentage differences between the effluent COD values and the SANS criteria are regarded as problematic. However, it is not uncommon to have high effluent COD levels, as reported by Mateus et al. (2021) and Negwamba and Dinka (2019), with final effluent COD levels twice that of the recommended level. Furthermore, Han et al. (2022) also reported final effluent COD levels exceeding 600 mg/l. Ultimately, such high final effluent COD levels at WWTPs are indicative of a system marred by dilapidating infrastructure, poor maintenance culture, and a lack of funding and trained personnel, which is the case with most WWTPs in South Africa (Jack et al., 2006; Edokpayi et al., 2020). Given that the Bainsvlei WWTP has a hydraulic capacity four times less than the other three WWTPs (*cf.* Table 3.1), the effluent COD values for all the WWTPs were normalised for comparison purposes as shown in Figure 4.7.



**Figure 4.7:** Normalised effluent COD at the four WWTPs

It is evident from the normalised COD values in Figure 4.7 that the hydraulic capacity of a WWTP does not necessarily have an impact on the treatment

efficiency. In this case, the Bainsvlei WWTP had the best-treated effluent compared to the other WWTPs; however, none of the four WWTPs met the recommended SANS COD criterion of 75 mg/l throughout the year, which signifies an increased presence of organic substances. This scenario could potentially lead to reduced oxygen levels and the proliferation of microorganisms when the effluent is discharged into surface waters. Under uncontrolled conditions, this could have significant consequences for aquatic life, the environment, and human health. Typically, as reported by the Department of Health (2023), such contaminated surface waters resulted in cholera outbreaks and deaths in the FS province and four other provinces. Hence, in addition to COD, the biological oxygen demand (BOD) of the influent and effluent needs to be established.

### 4.3 BOD Analyses of WWTPs

The BOD considers the amount of dissolved oxygen aerobic microorganisms use to break down the organic matter in wastewater, typically within five days, often referred to as BOD<sub>5</sub>. Since the BOD is associated with the COD, it was not determined experimentally owing to a limited budget and equipment. The BOD was mathematically determined using Equation 4.2 to highlight temporal trends within the WWTPs and not necessarily the precise, absolute BOD levels. Although BOD is crucial in wastewater analyses, it is not considered in the SANS code.

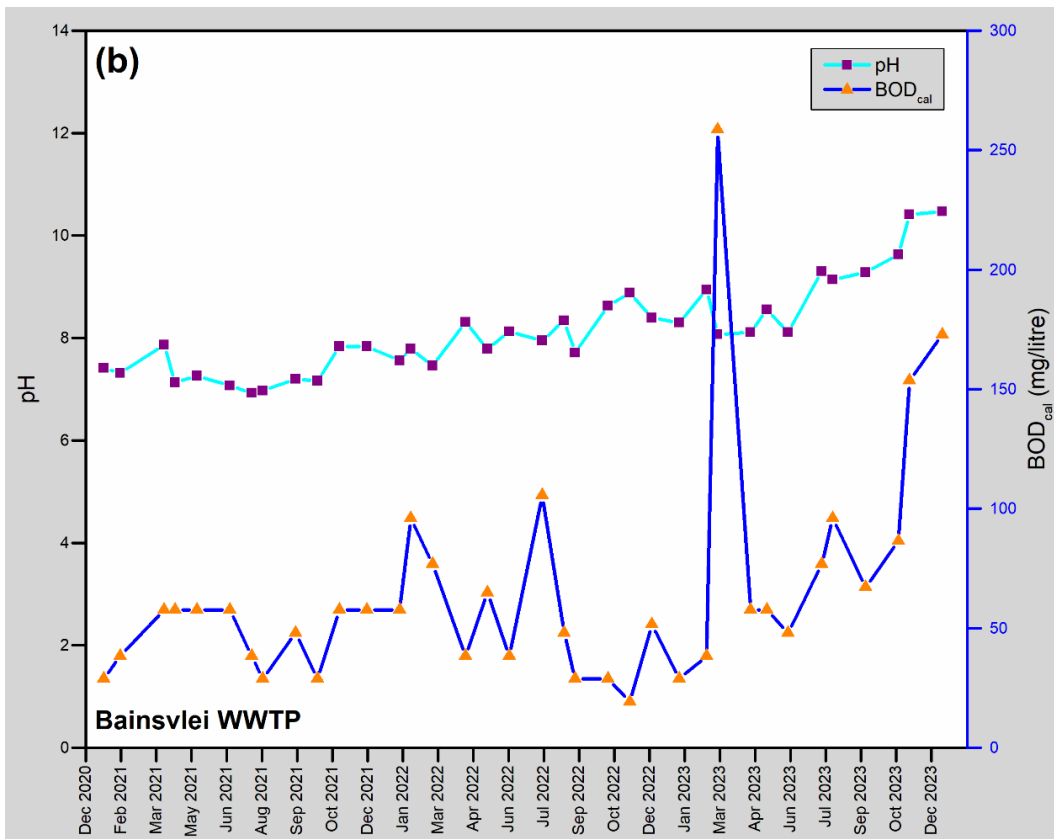
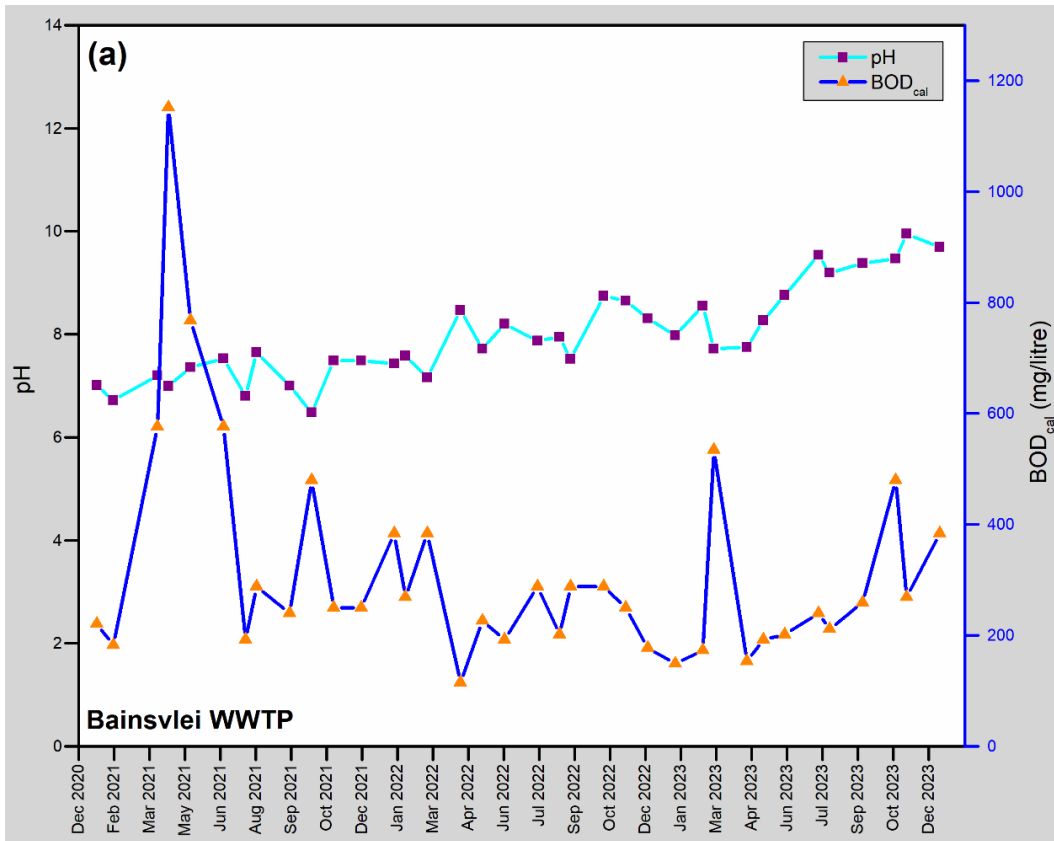
$$BOD_5 = 0.66COD \quad (4.2)$$

Where:

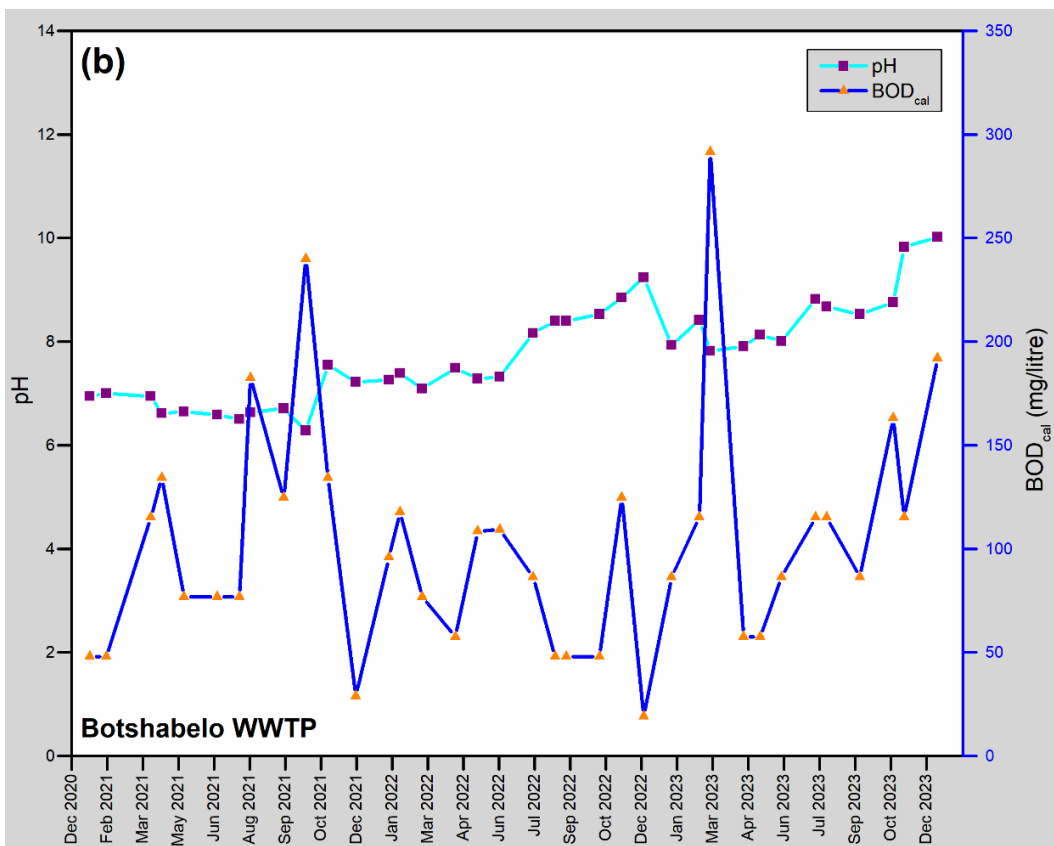
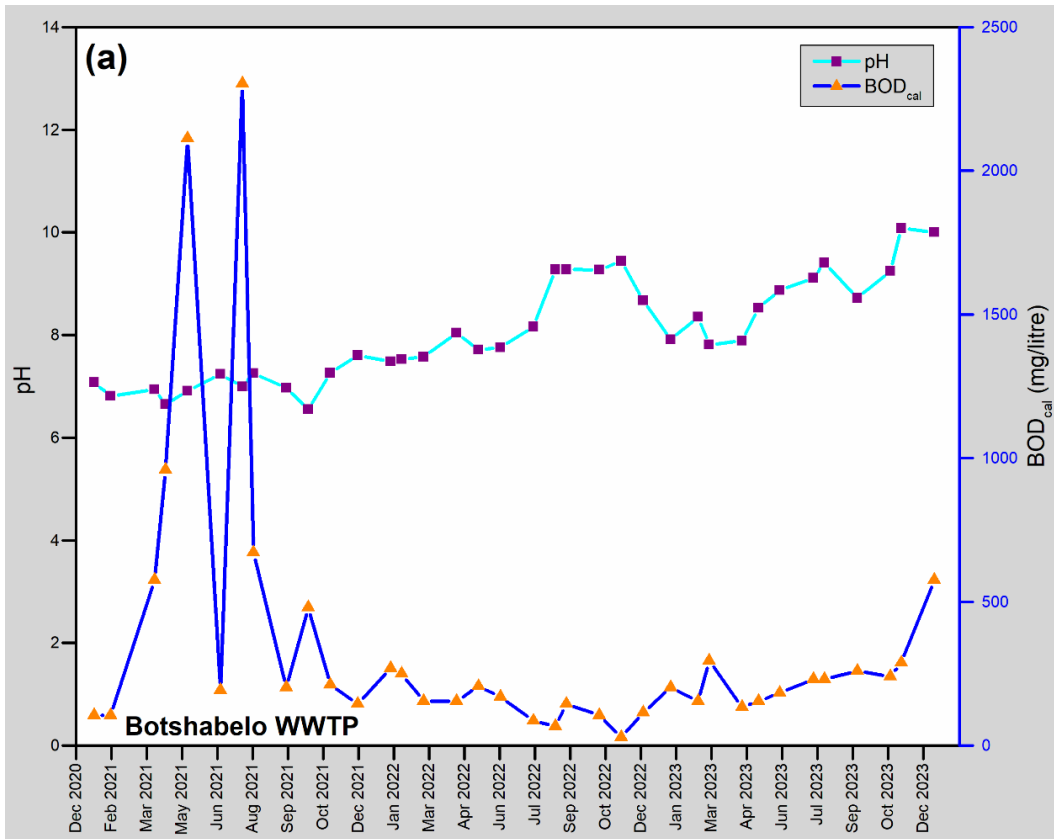
$BOD_5$  = biological oxygen demand after five days (mg/l), and

$COD$  = chemical oxygen demand (mg/l).

Establishing which parameter influences or has a strong linear correlation with the calculated BOD (BOD<sub>cal</sub>) is essential. Once this correlation has been established, it further enforces using Equation 4.2 rather than experimentally determining the BOD, which also render the process more cost effective. The parameters assumed to influence the influent and effluent BOD<sub>cal</sub> are pH, temperature, OA, and COD. The relationship between pH and BOD<sub>cal</sub> at the four WWTPs is shown in Figure 4.8.



**Figure 4.8:** pH and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs



**Figure 4.8:** pH and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

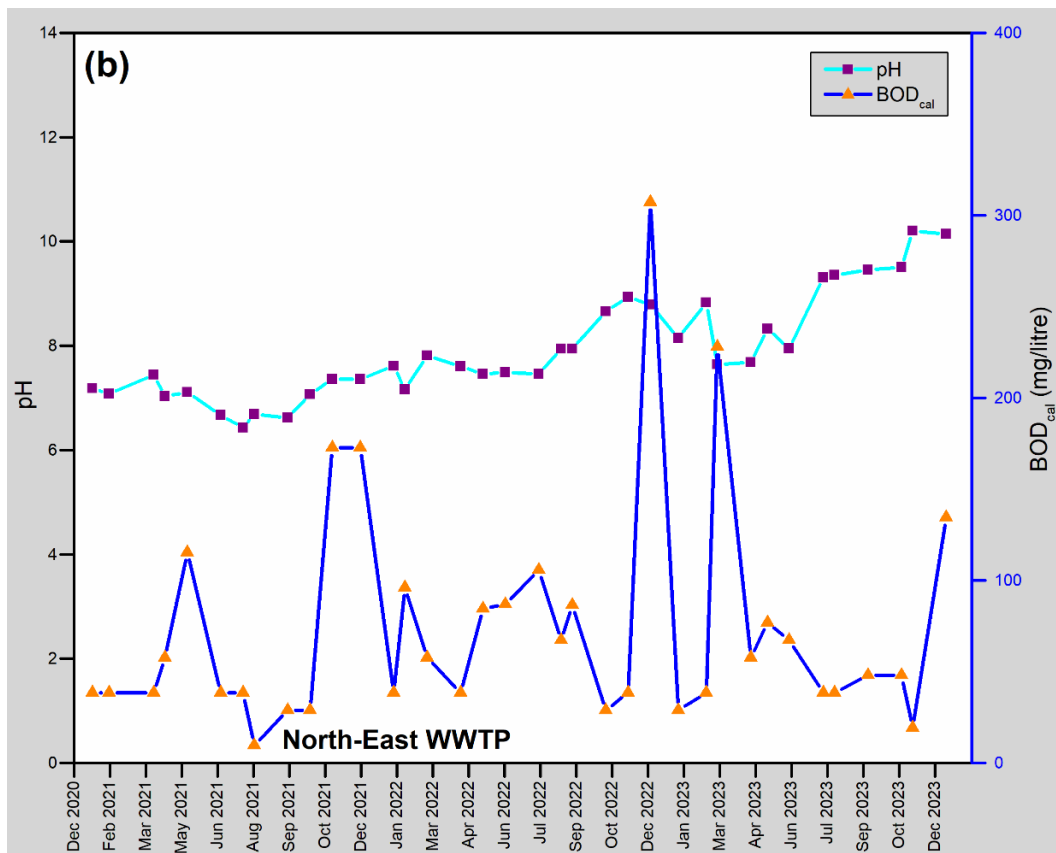
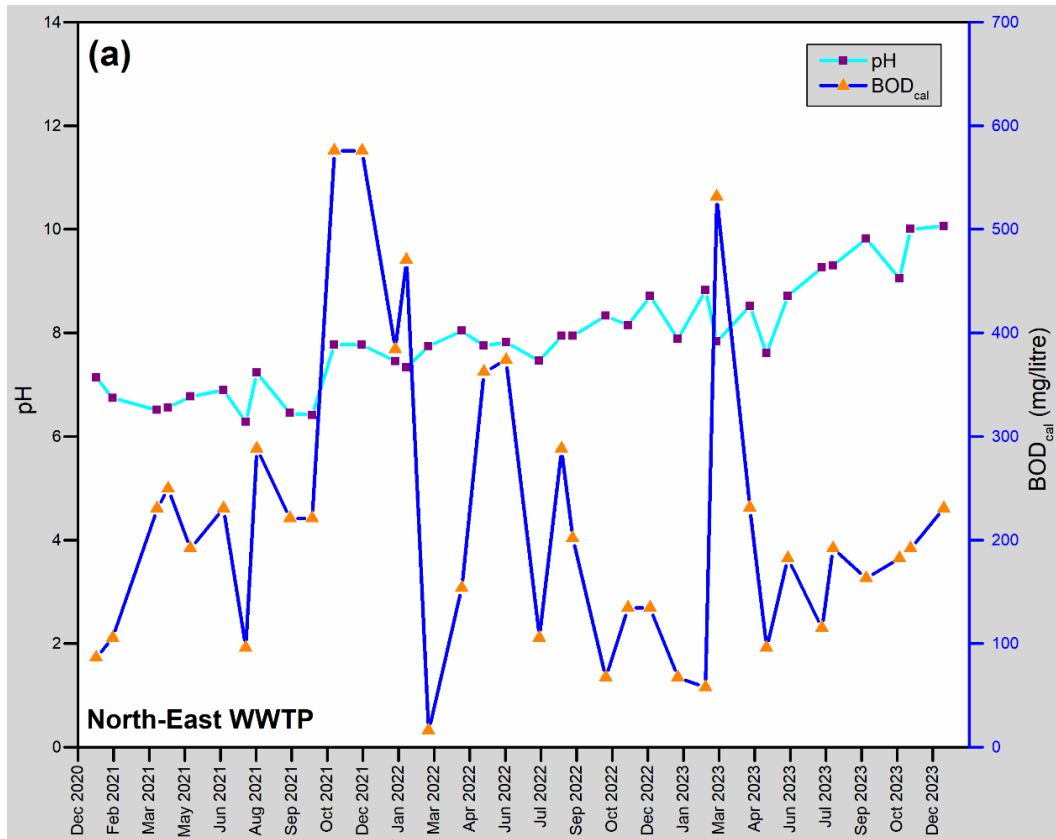
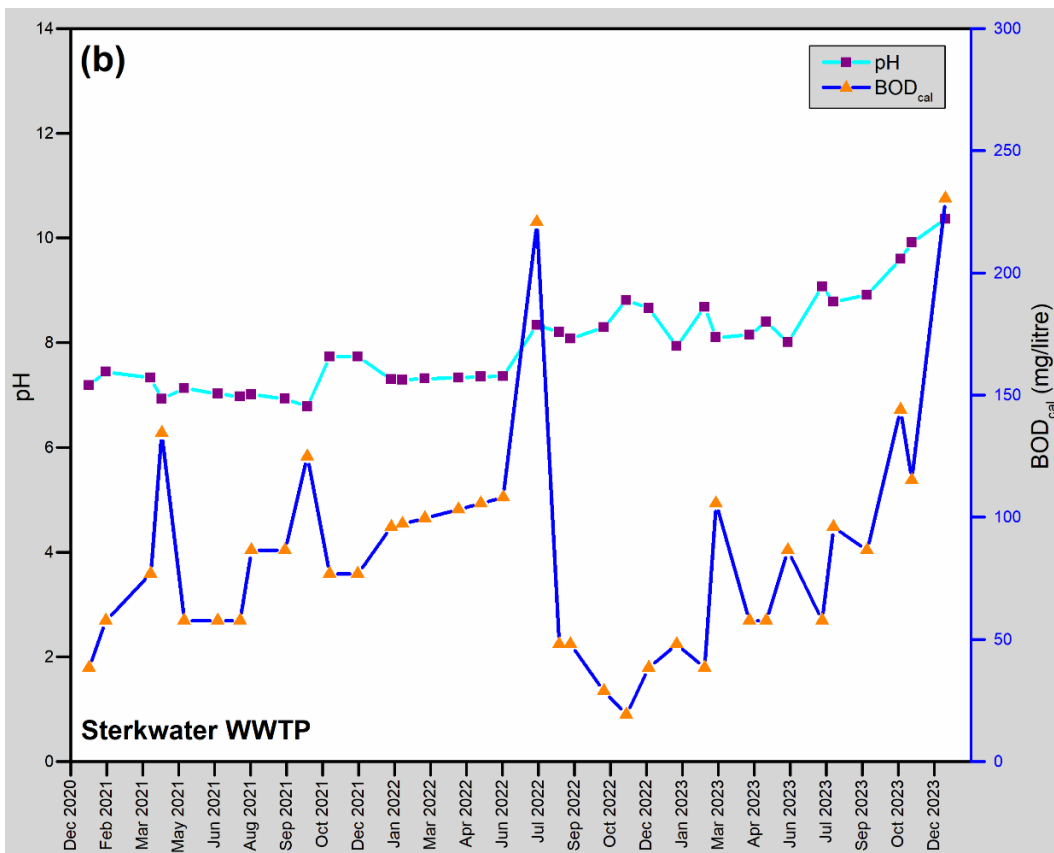
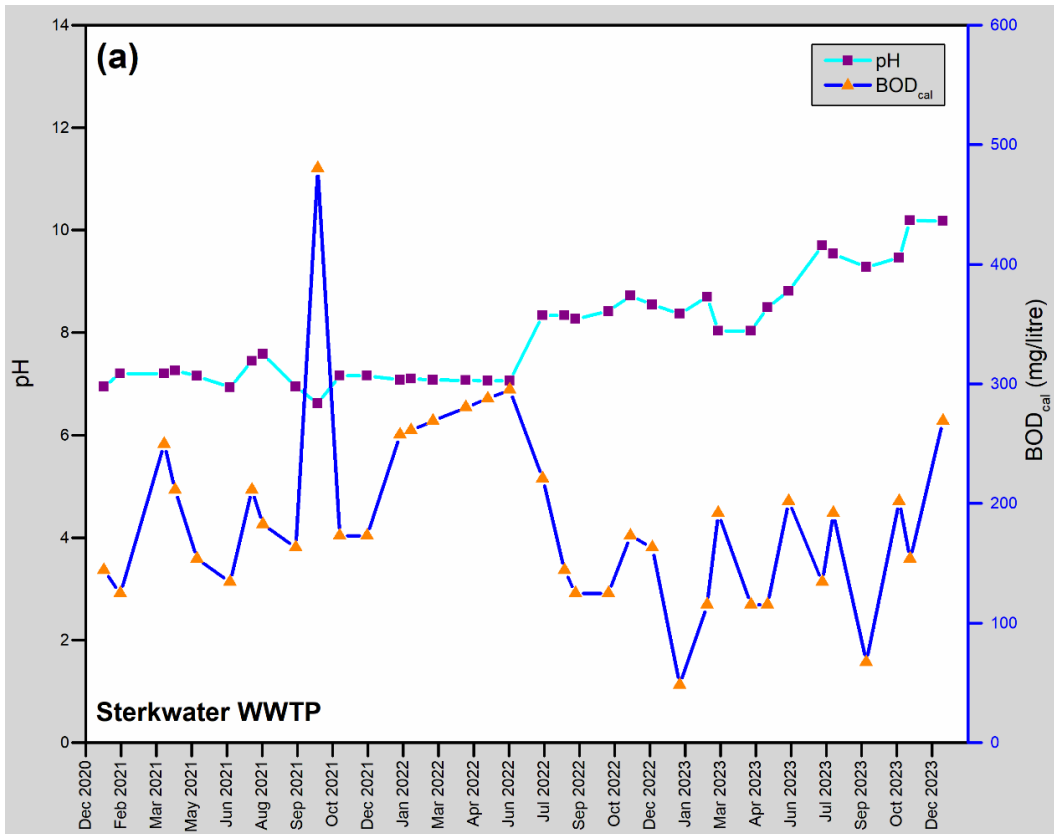


Figure 4.8: pH and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

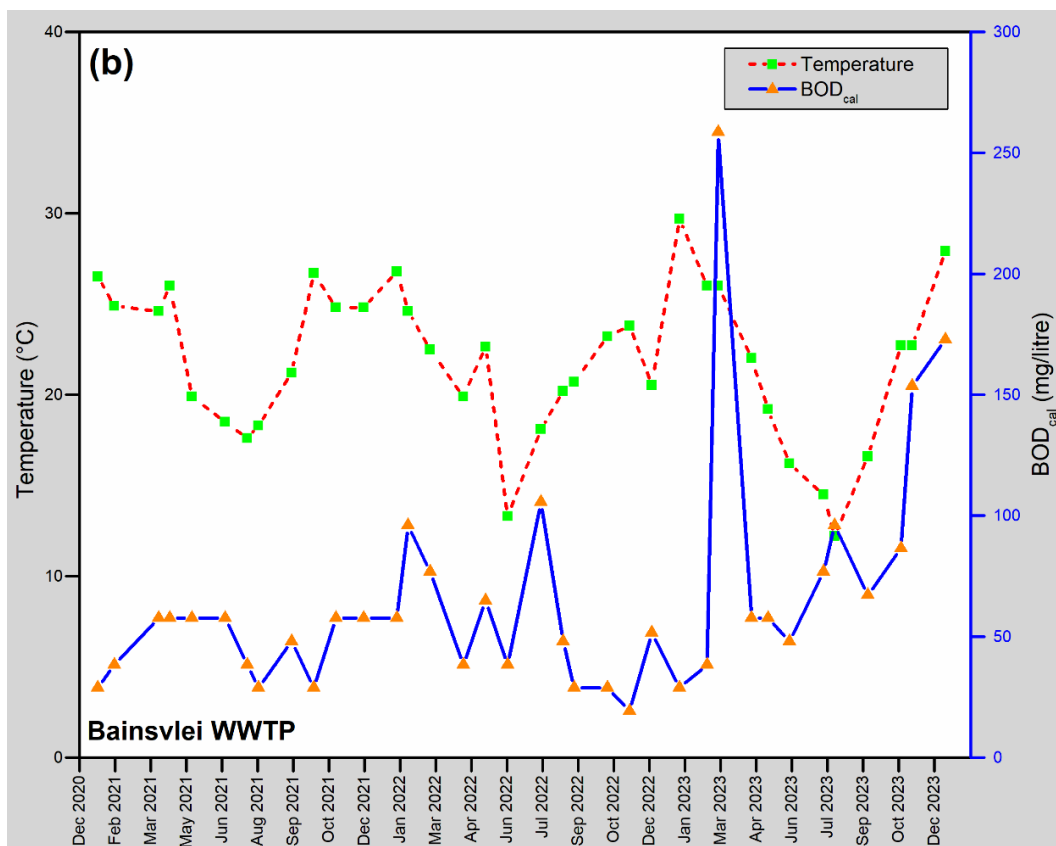
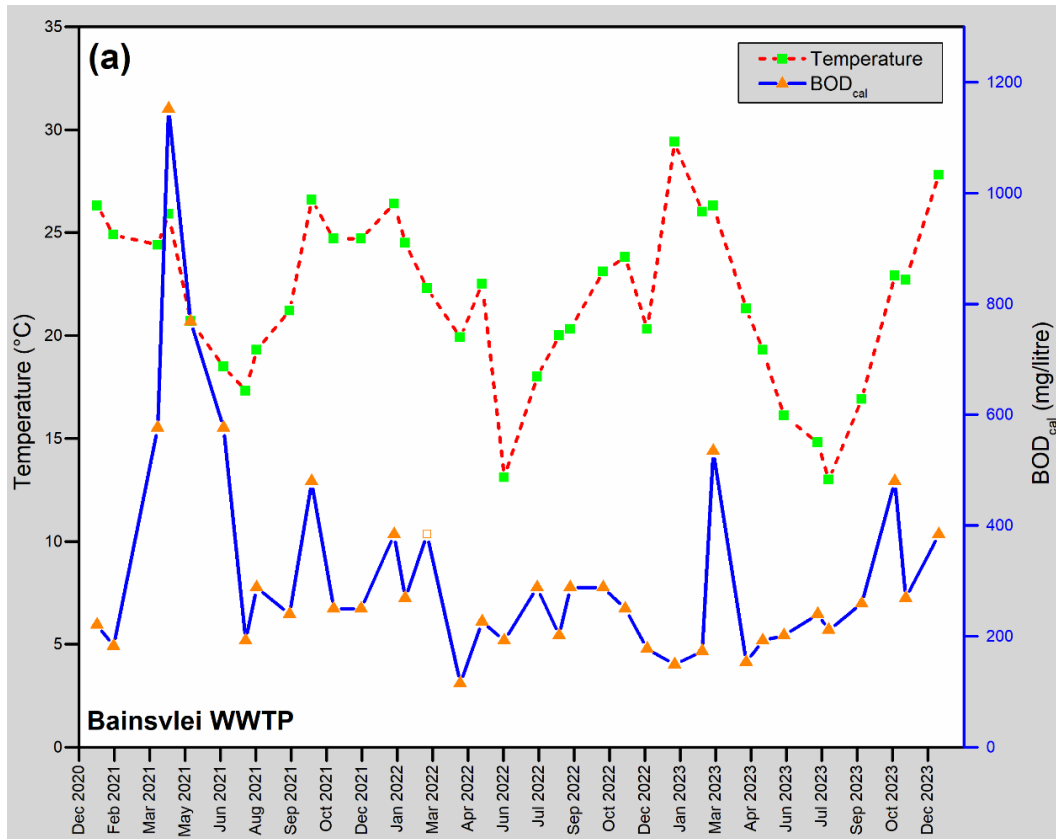


**Figure 4.8:** pH and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

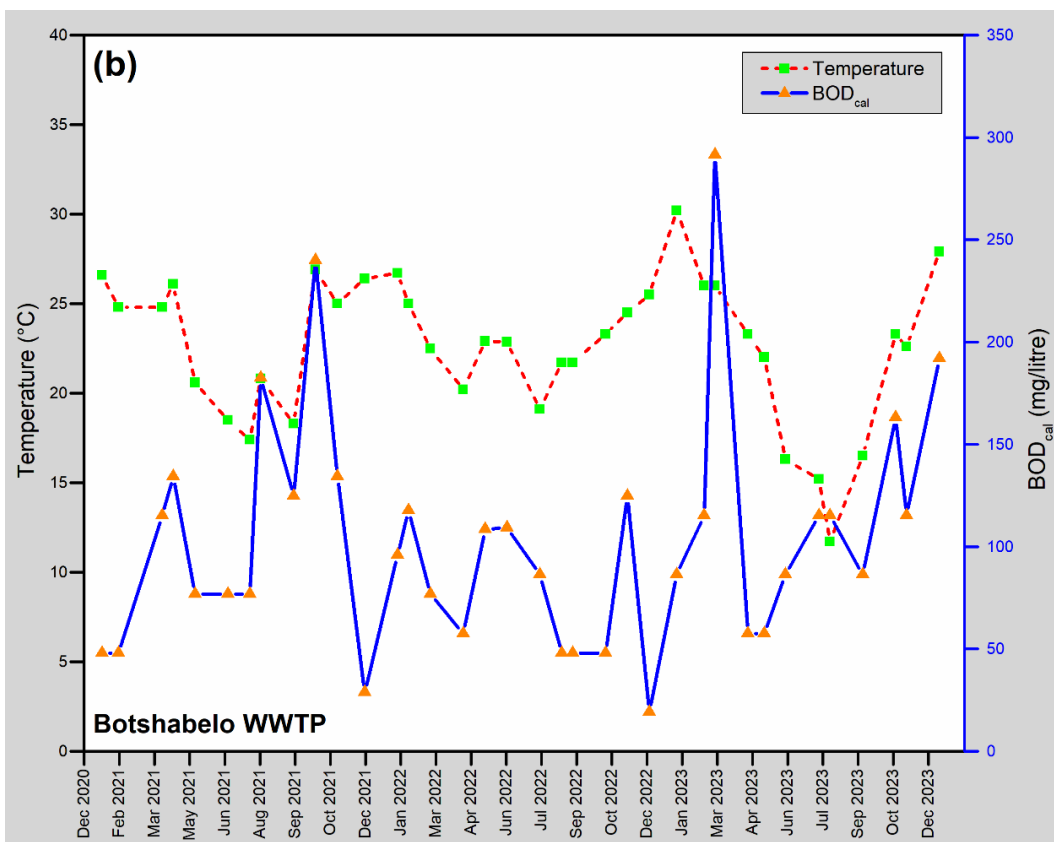
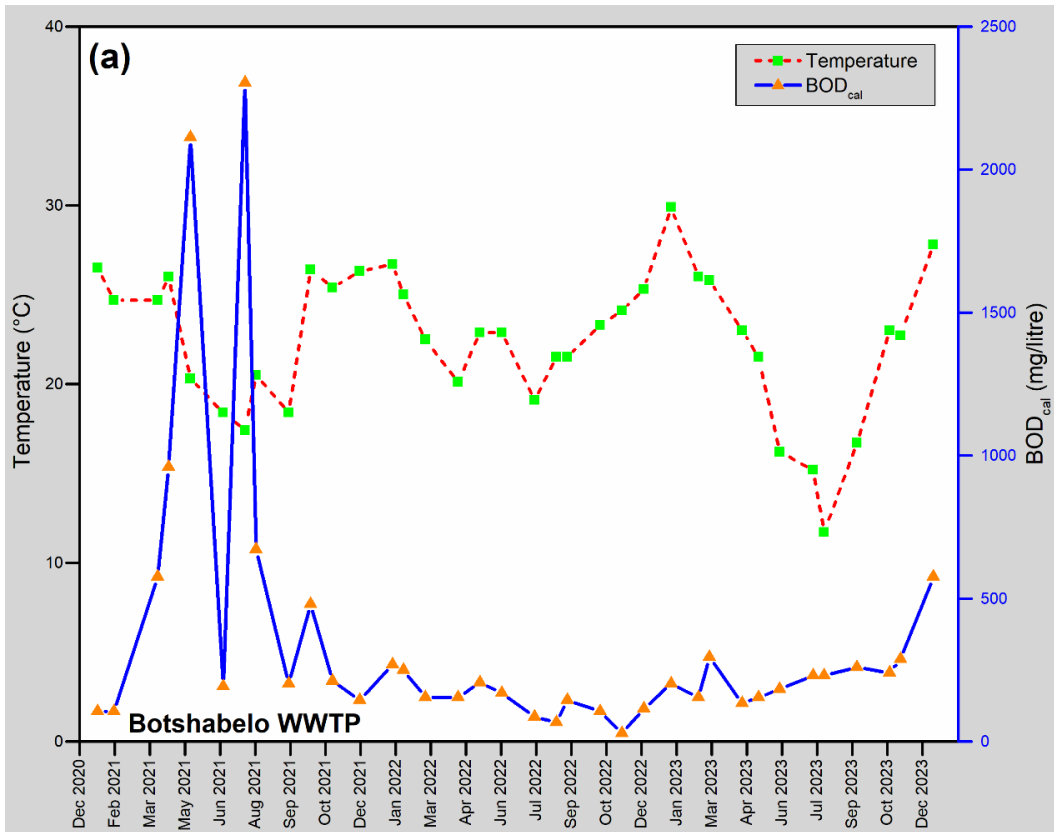
In Figure 4.8 (a), the influent pH for the Bainsvlei WWTP fluctuated between 6.48 and 9.95, while the influent BOD<sub>cal</sub> ranged between 115.2 mg/l and 1 152 mg/l. There are no noticeable trends present between the influent and effluent pH and BOD<sub>cal</sub> values. The effluent BOD<sub>cal</sub> fluctuated between 19.2 mg/l and 258.6 mg/l. Similar trends were evident at the other three WWTPs, thereby confirming that pH had no direct impact on the BOD<sub>cal</sub>.

Subsequently, the effect of temperature on the BOD<sub>cal</sub> values at the four WWTPs was determined as shown in Figure 4.9. It is evident from Figure 4.9 that during some months, a decrease/increase in temperature led to decreasing/increasing BOD<sub>cal</sub> values at all the WWTPs, while during other months, no trend is noticeable.

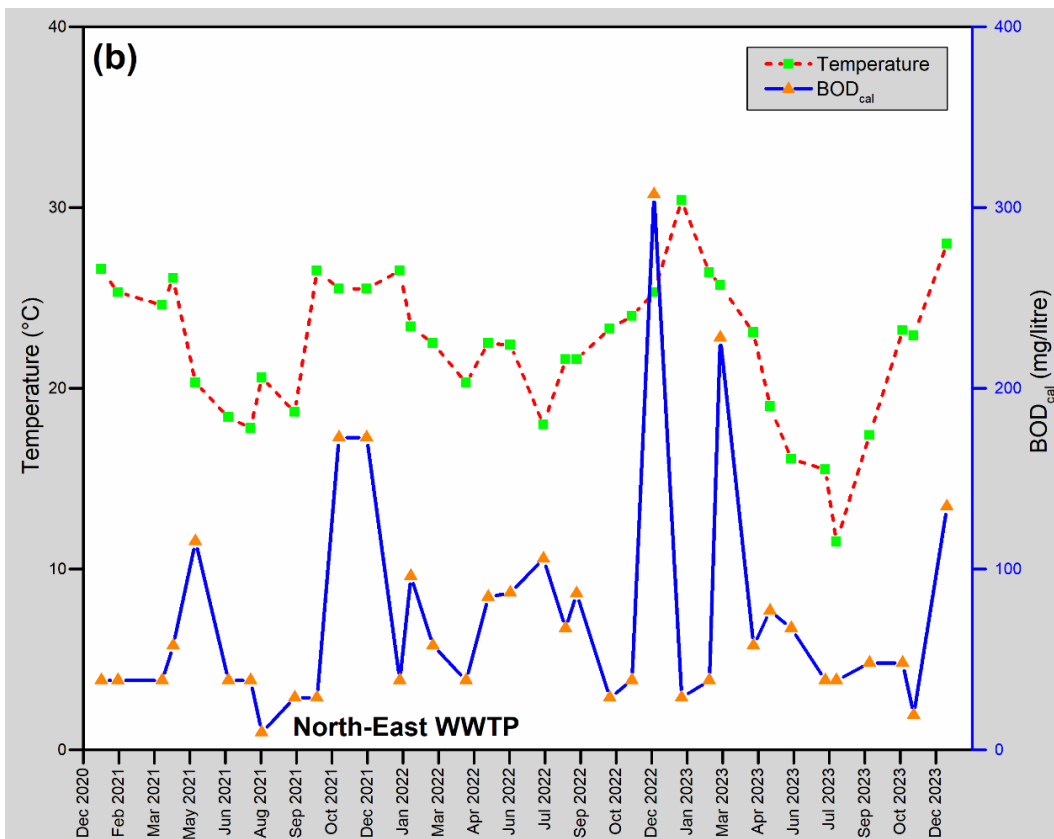
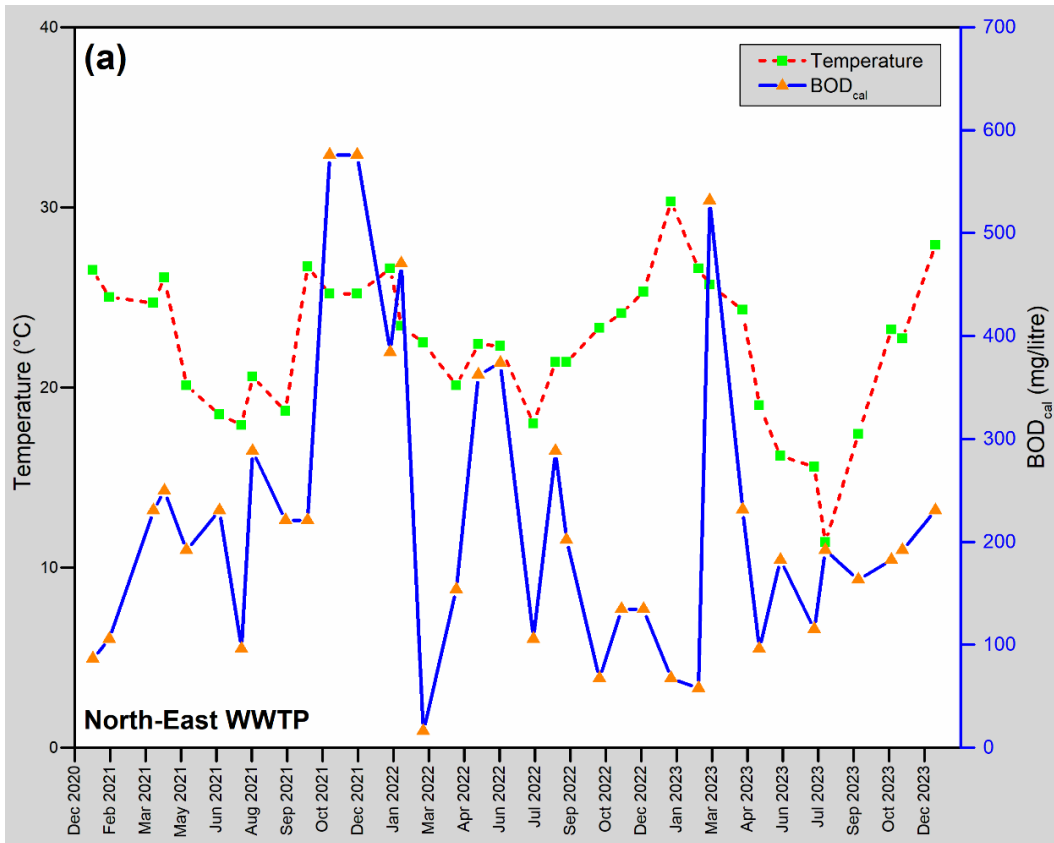
Typically, it is evident from Figure 4.9 that the temperature of the Bainsvlei WWTP influent fluctuated between 13°C and 29.4°C, while the effluent fluctuated between 12.2°C and 29.7°C. As expected, the lowest temperature was recorded in winter, while the highest temperature was recorded in summer. Variable and contradictory trends were observed, e.g., an increasing temperature resulted in decreasing BOD<sub>cal</sub> values, and/or a decreasing temperature resulted in increasing BOD<sub>cal</sub> values at the Botshabelo, North-East, and Sterkwater WWTPs. Their influent temperature fluctuated between 11.3°C and 30.3°C, while the effluent temperatures fluctuated between 11.3°C and 30.4°C. The BOD<sub>cal</sub> values of the influent and effluent at these three WWTPs fluctuated between 15.9 mg/l and 2 304 mg/l, and 9.6 mg/l and 307.2 mg/l, respectively.



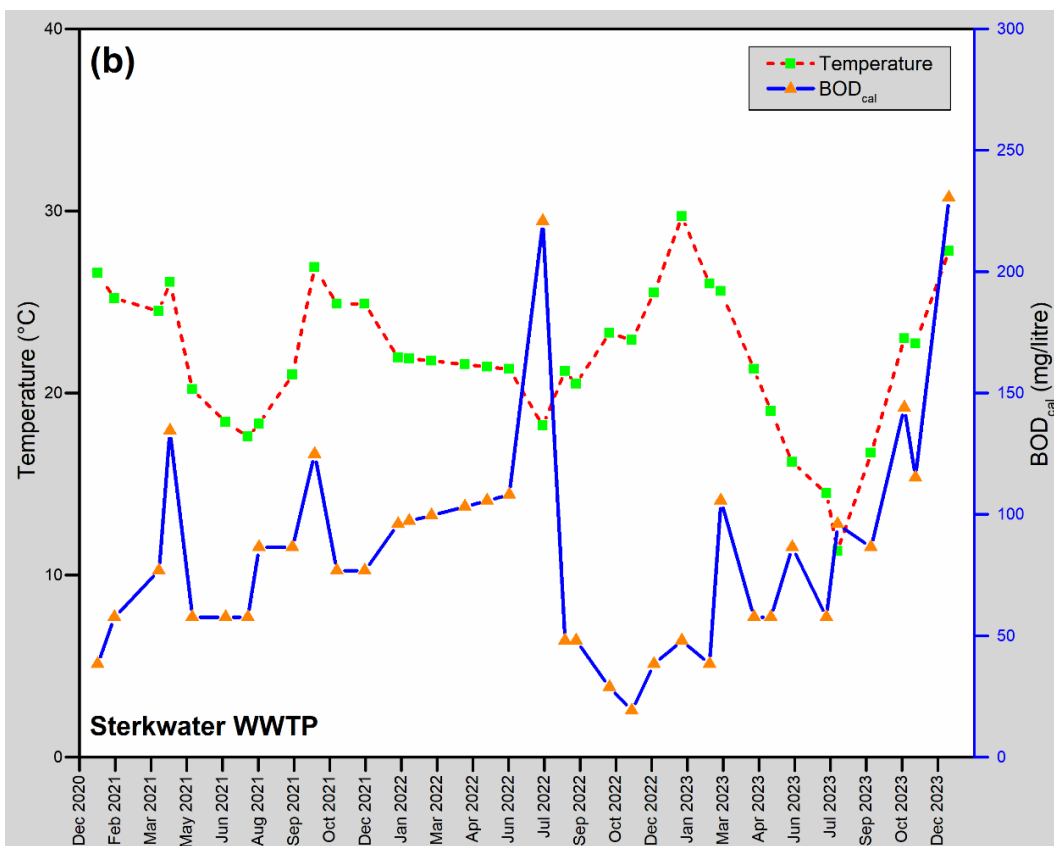
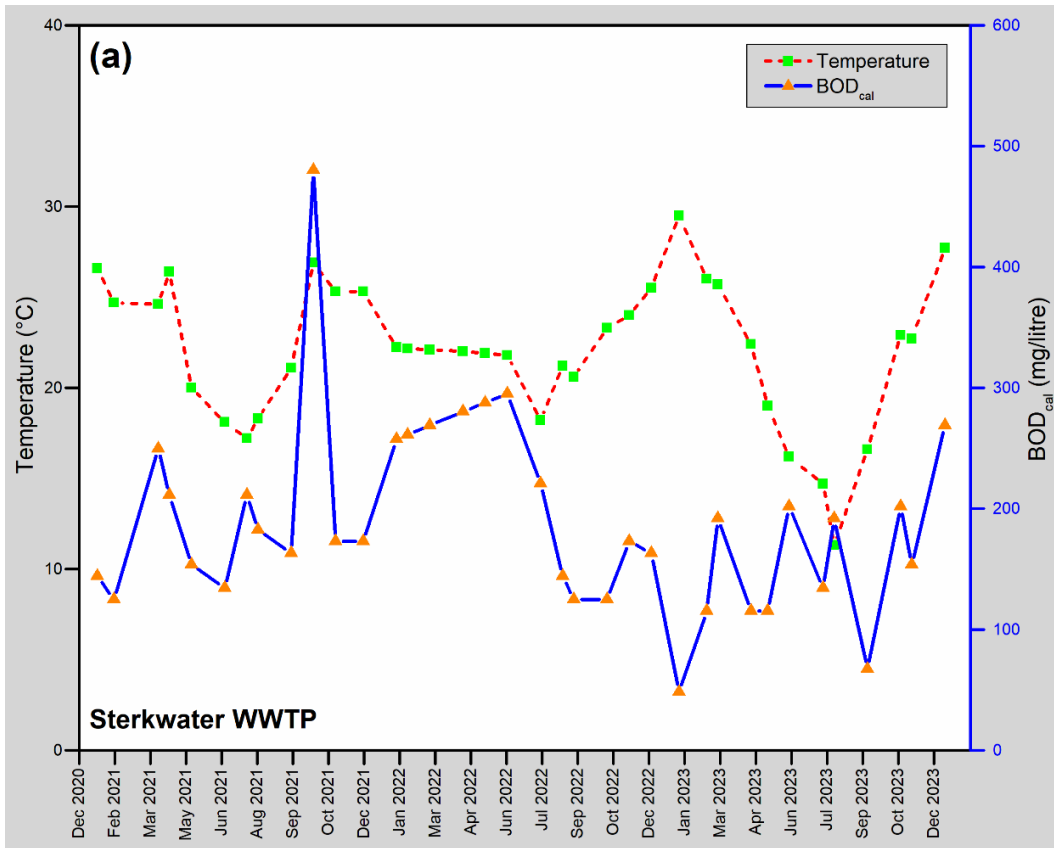
**Figure 4.9:** Temperature and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs



**Figure 4.9:** Temperature and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs



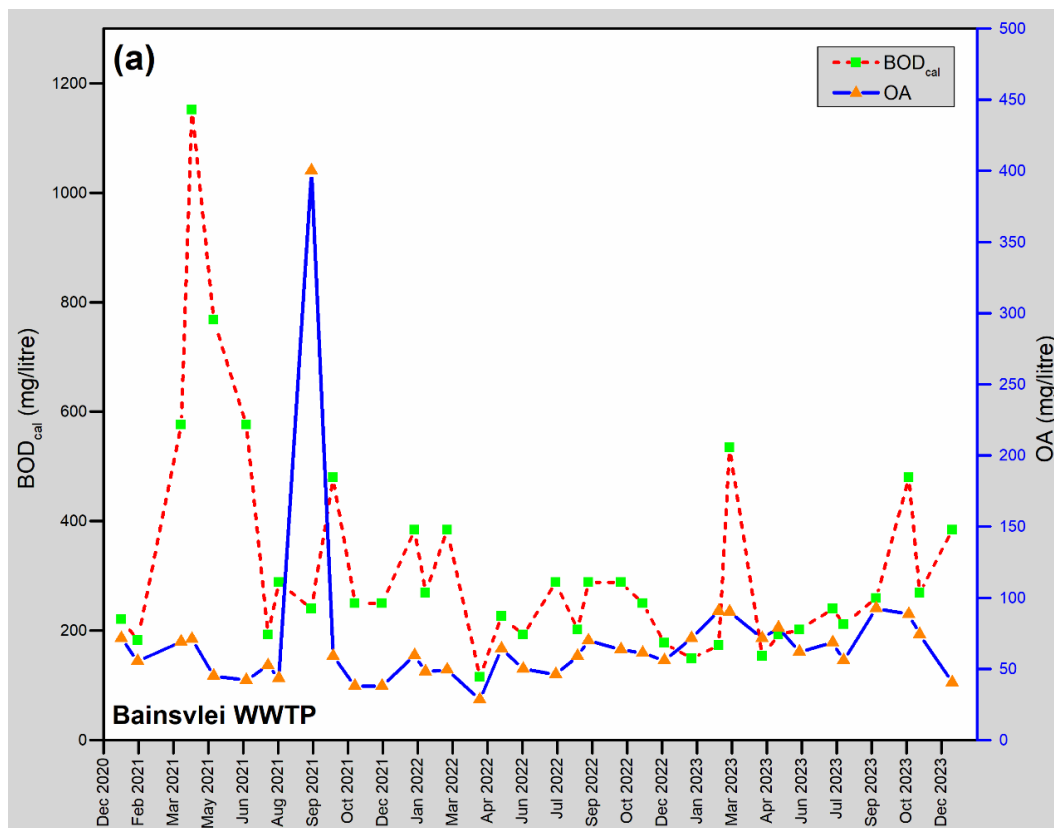
**Figure 4.9:** Temperature and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs



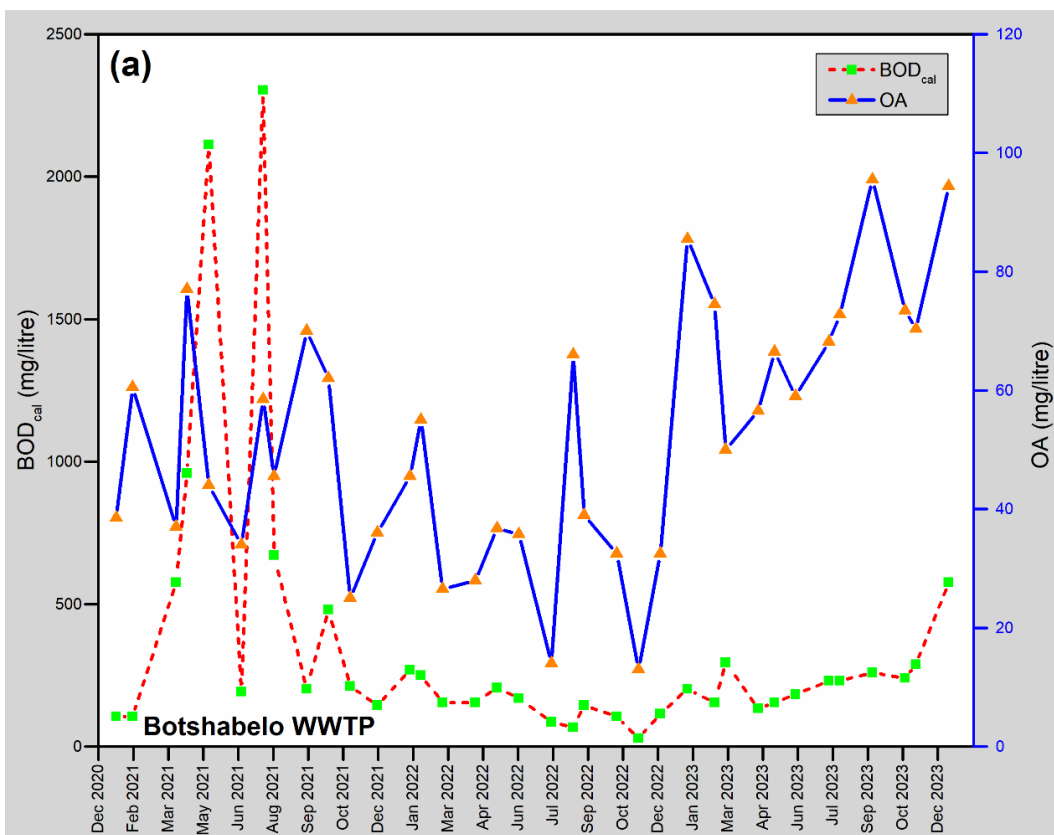
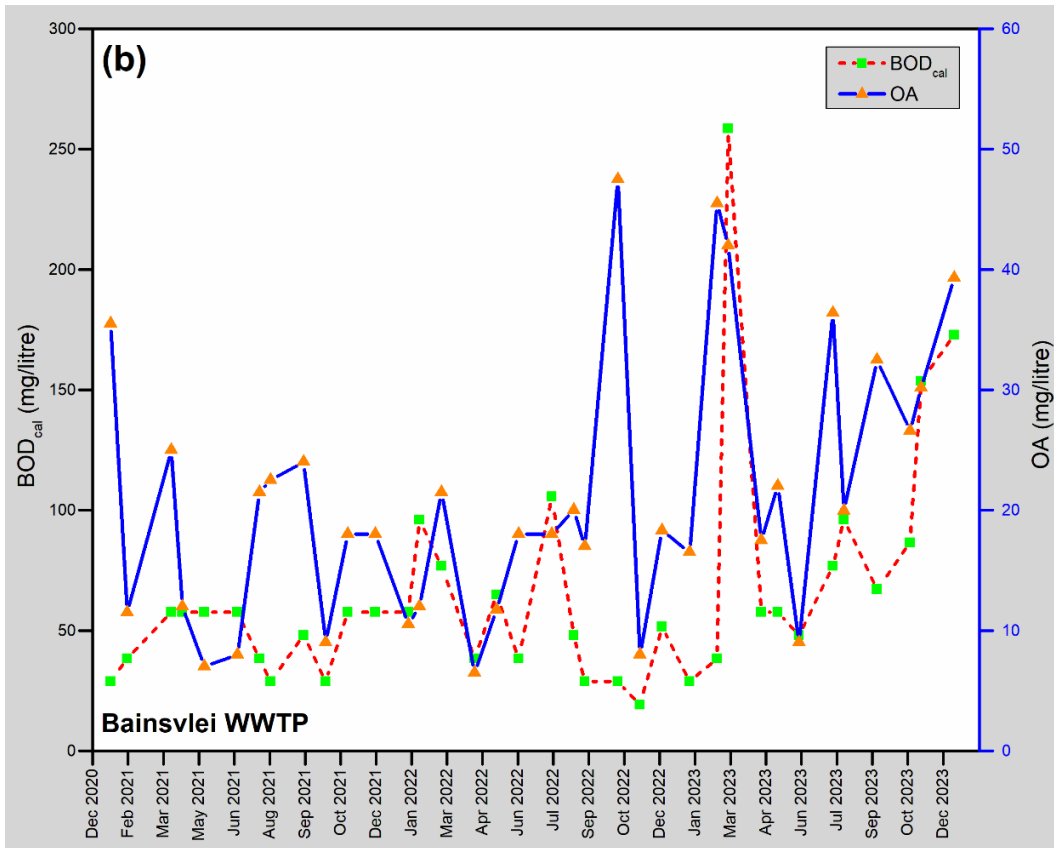
**Figure 4.9:** Temperature and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

In considering the above results, it is evident that seasonal variations are present in wastewater effluent, which was also noted by Mugudamani et al. (2023) and Masindi et al. (2022). Given that BOD is not subjected to any SANS effluent standards (Department of Water and Environmental Affairs, 2013; Department of Environmental Affairs, 2014), a statistical analysis was deemed necessary to investigate the possible cause(s) of such seasonal variations. The statistical analyses are presented in Chapter 5.

Subsequently, the potential impact of OA on  $BOD_{cal}$  values at the four WWTPs were further investigated as shown in Figure 4.10, given that Yang et al. (2025) identified the relationship between OA and BOD as a vital indicator of organic pollution. It is evident from Figure 4.10 that changes in the influent OA levels translate to slight changes in the  $BOD_{cal}$  values. However, the latter trend is significantly more evident in the wastewater effluent. In addition, the effluent  $BOD_{cal}/COD$  ratios at each WWTP were determined as suggested by Samudro and Mangkoedihardjo (2010) and presented in Figures 4.11 (a – d).



**Figure 4.10:** OA and  $BOD_{cal}$  values of the (a) influent and (b) effluent at the four WWTPs



**Figure 4.11:** OA and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

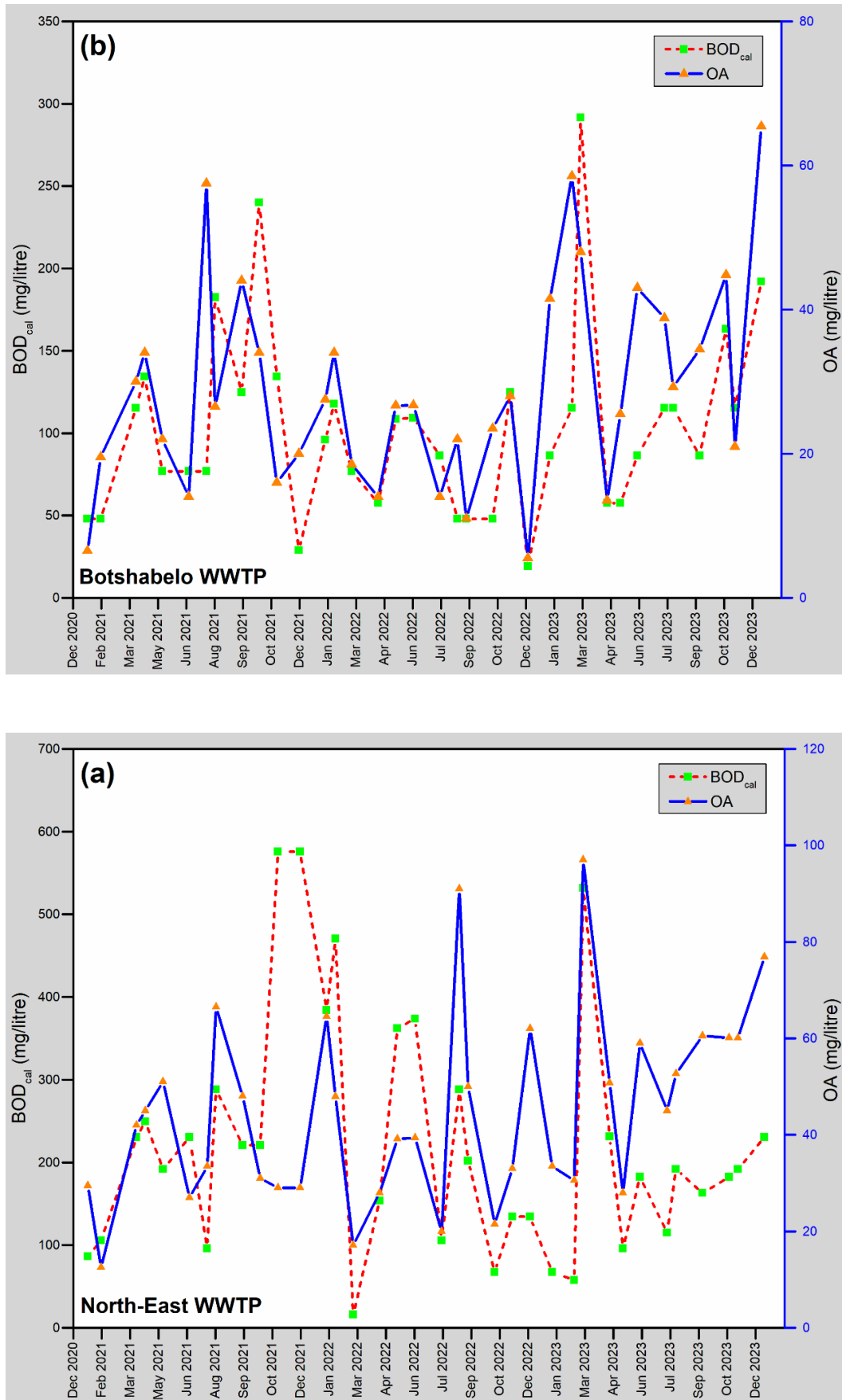


Figure 4.12: OA and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs

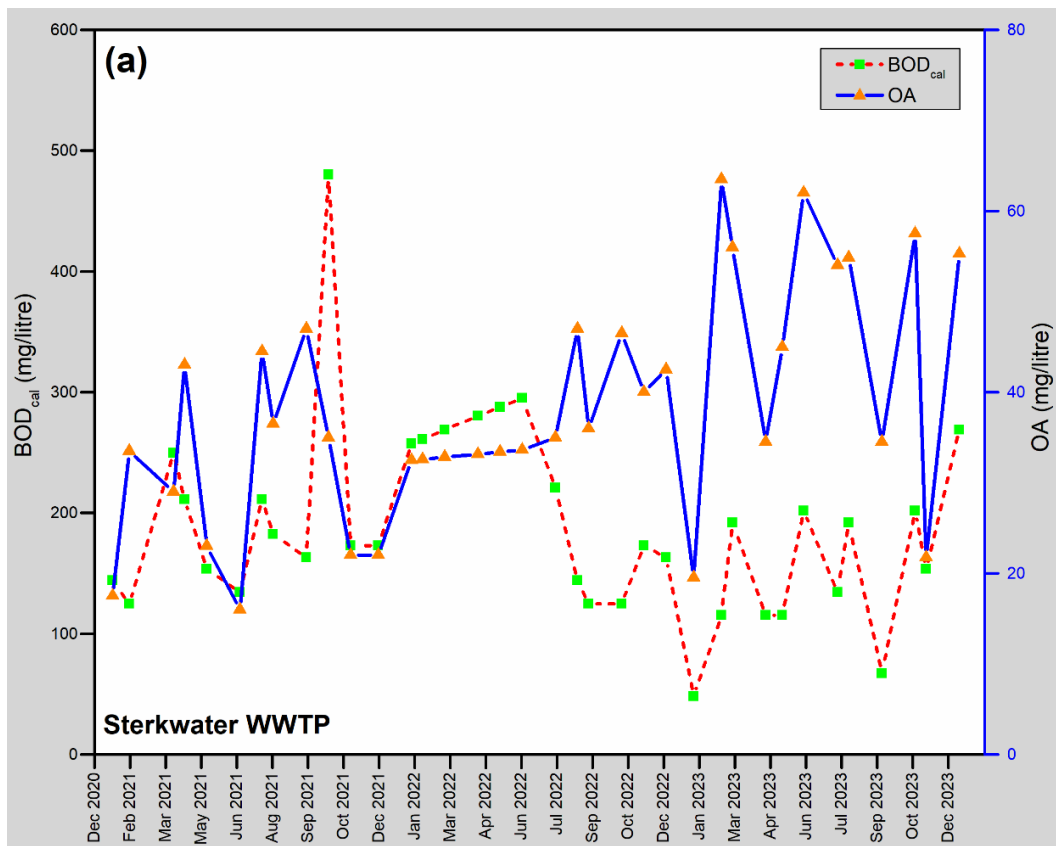
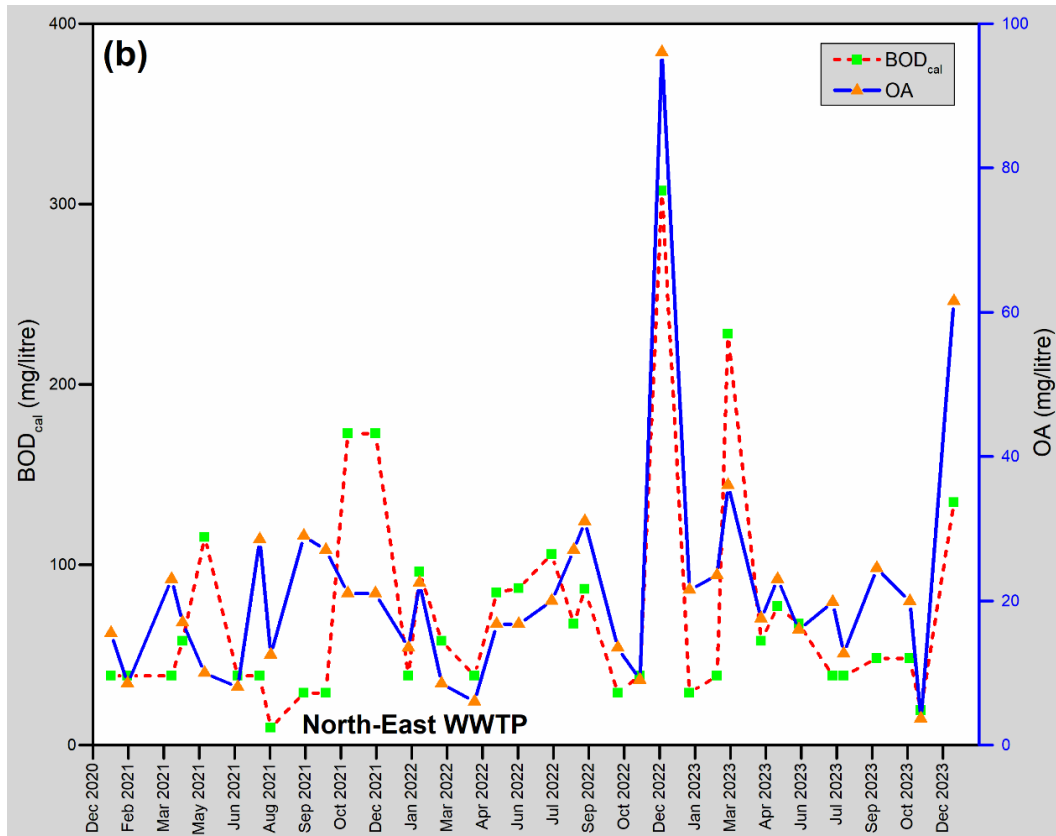
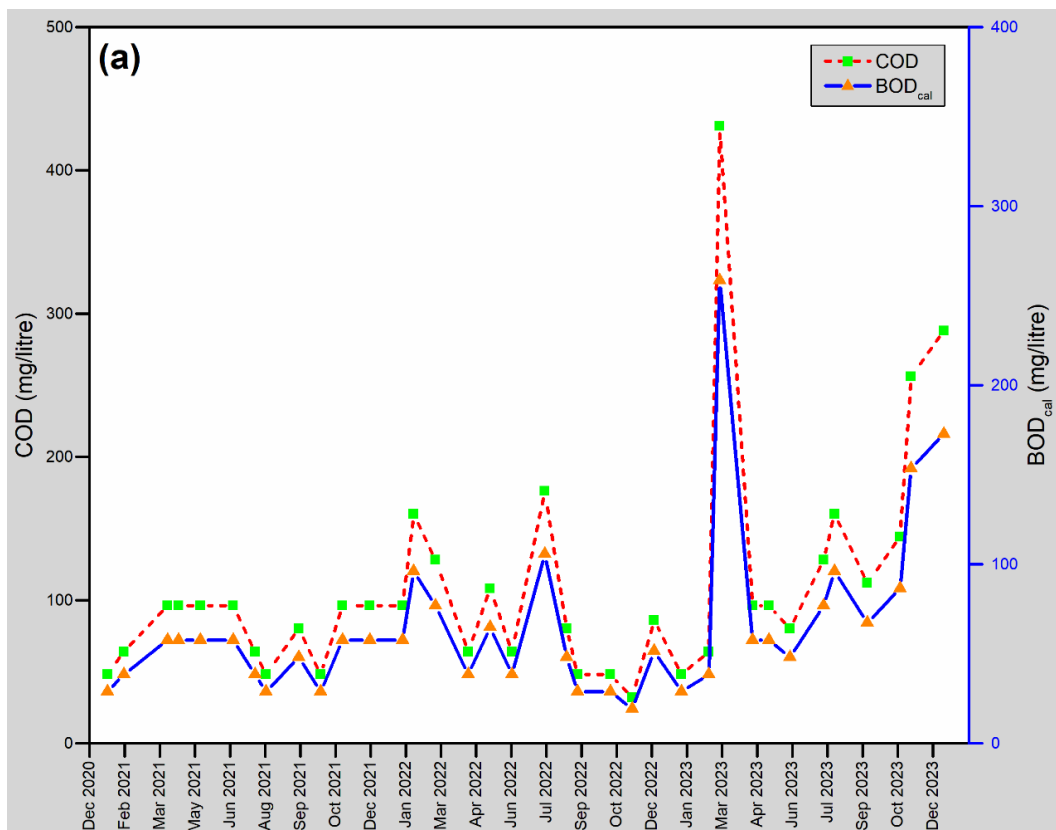
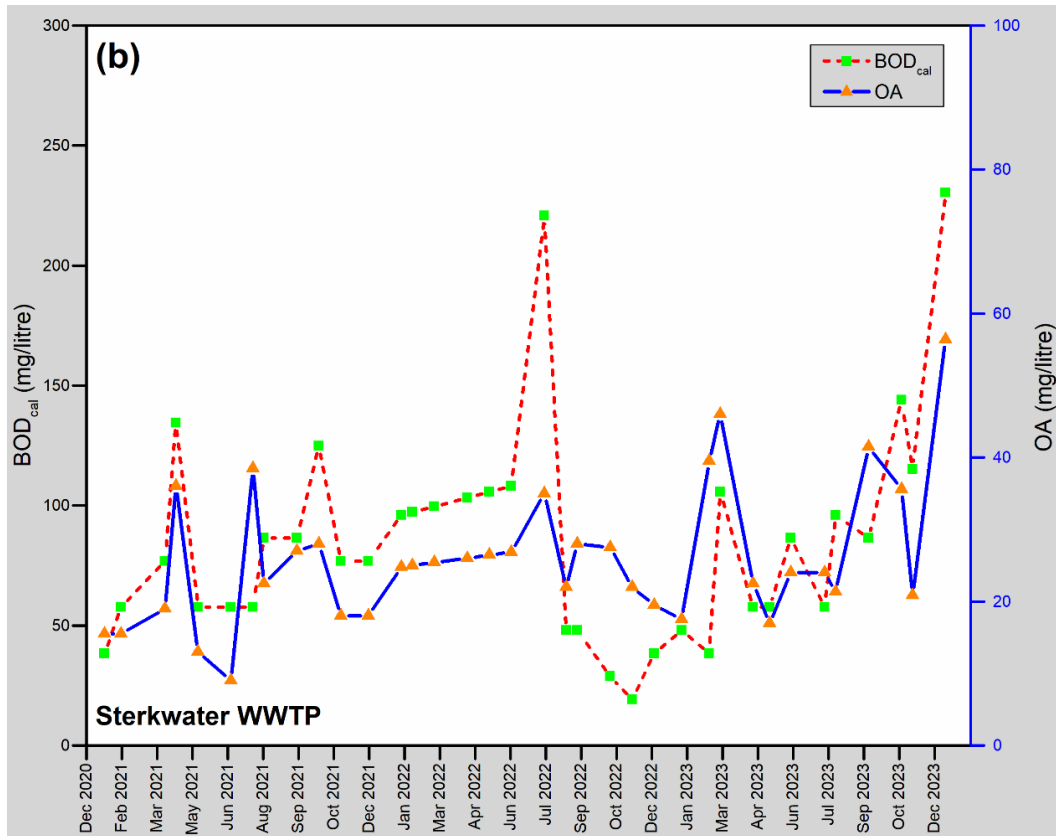
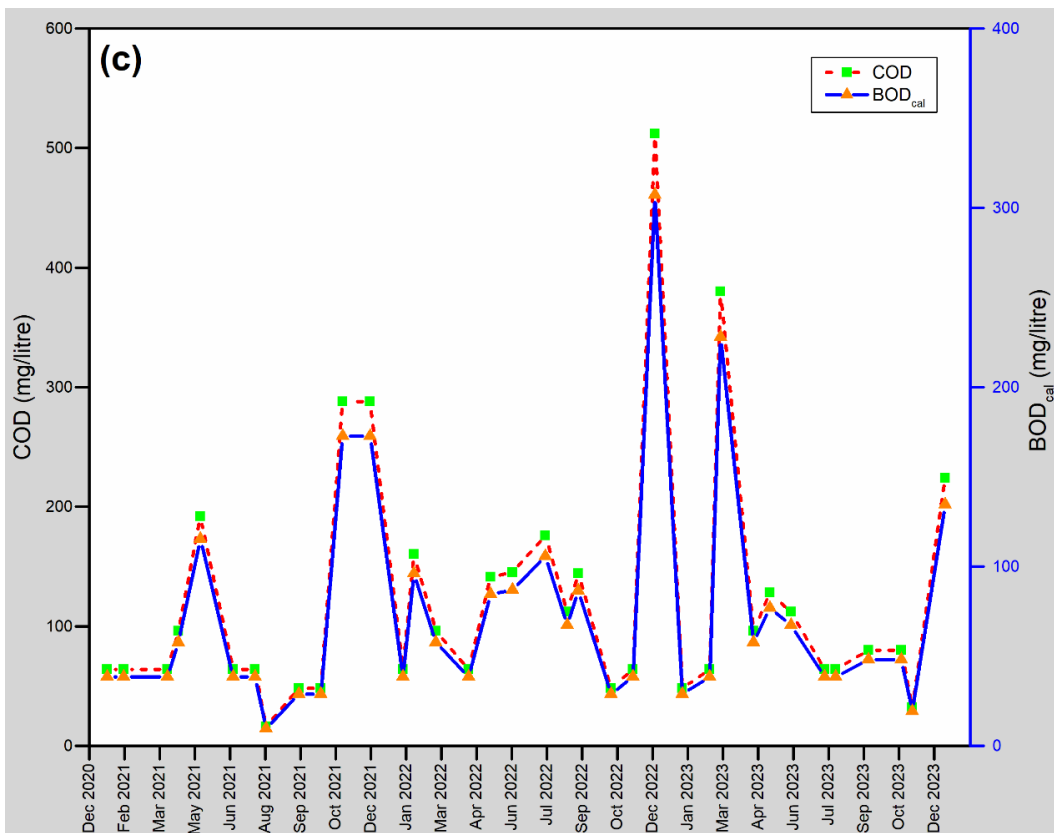
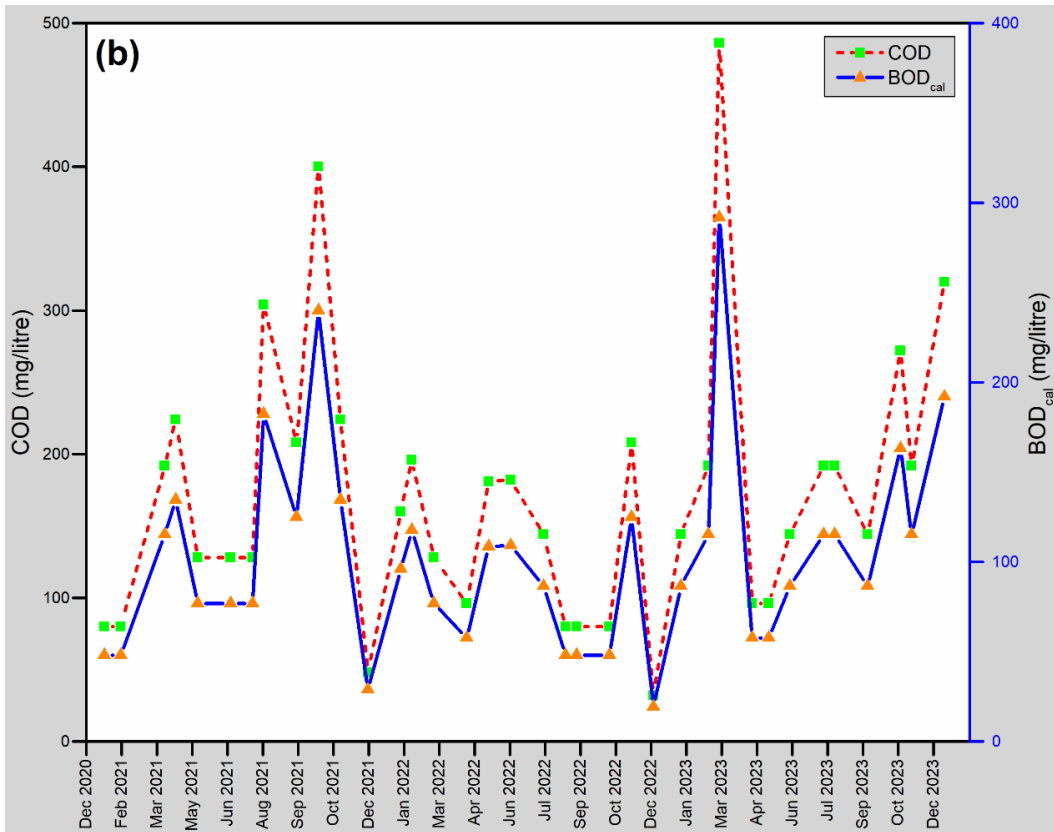


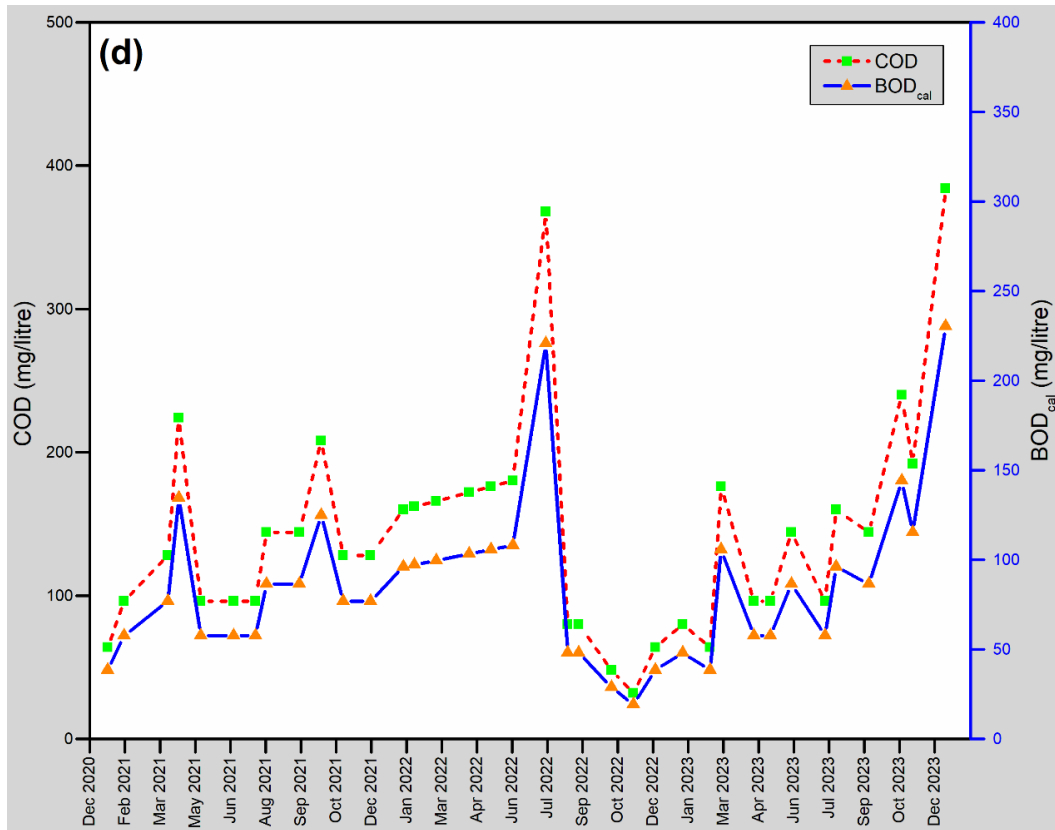
Figure 4.13: OA and  $BOD_{cal}$  values of the (a) influent and (b) effluent at the four WWTPs



**Figure 4.14:** OA and BOD<sub>cal</sub> values of the (a) influent and (b) effluent at the four WWTPs



**Figure 4.15:** Effluent BOD<sub>cal</sub>/COD ratios at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs



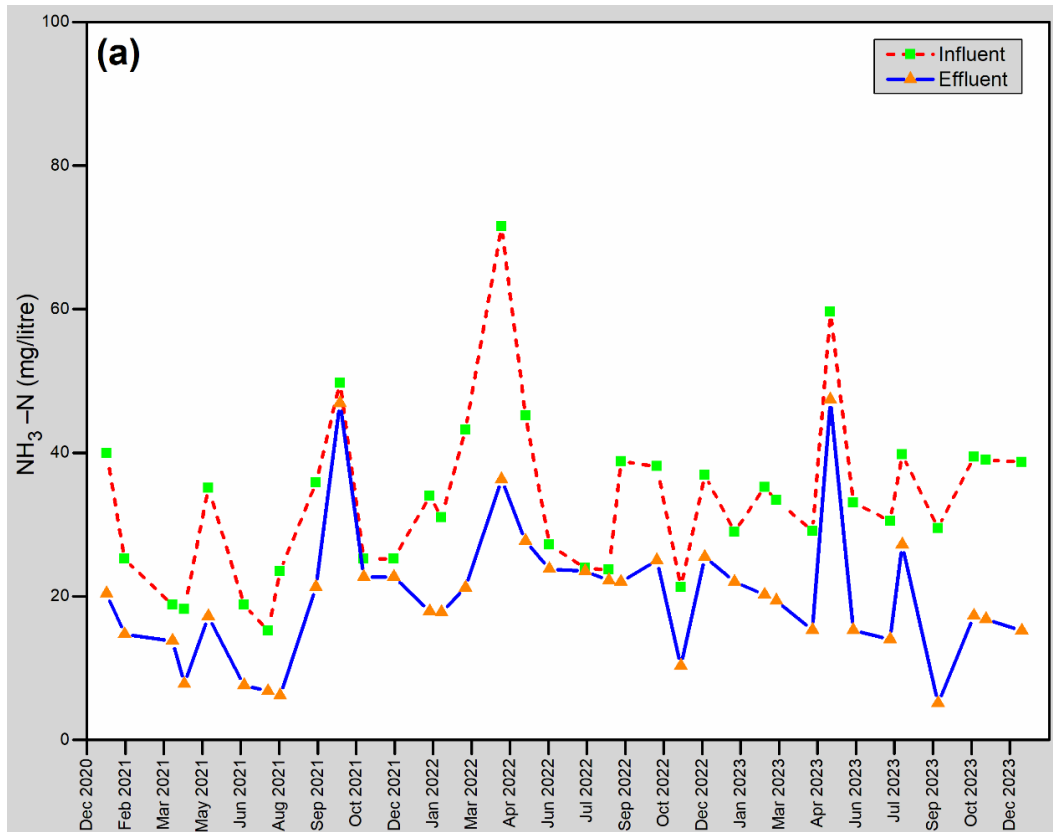
**Figure 4.16:** Effluent BOD<sub>cal</sub>/COD ratios at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

In considering the BOD<sub>cal</sub>/COD ratios for all the WWTPs in Figures 4.11 (a – d), it became evident that the effluent COD is approximately 1.67 to 2-times larger than the BOD<sub>cal</sub>. This ratio corresponds with the recommendations made by Gaur (2008), who confirmed that the BOD<sub>cal</sub>/COD ratio typically varies between 2 and 2.5. Subsequently, the latter agreement in ratios confirmed the suitability of Equation 4.2 in determining the BOD.

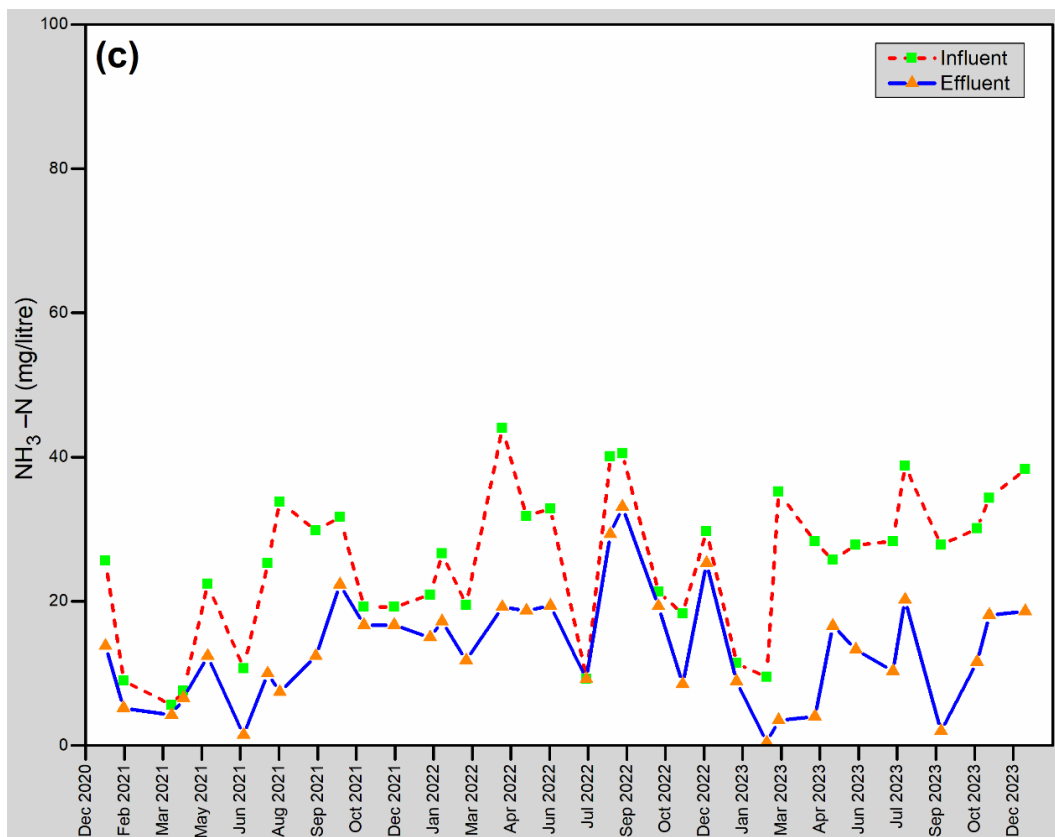
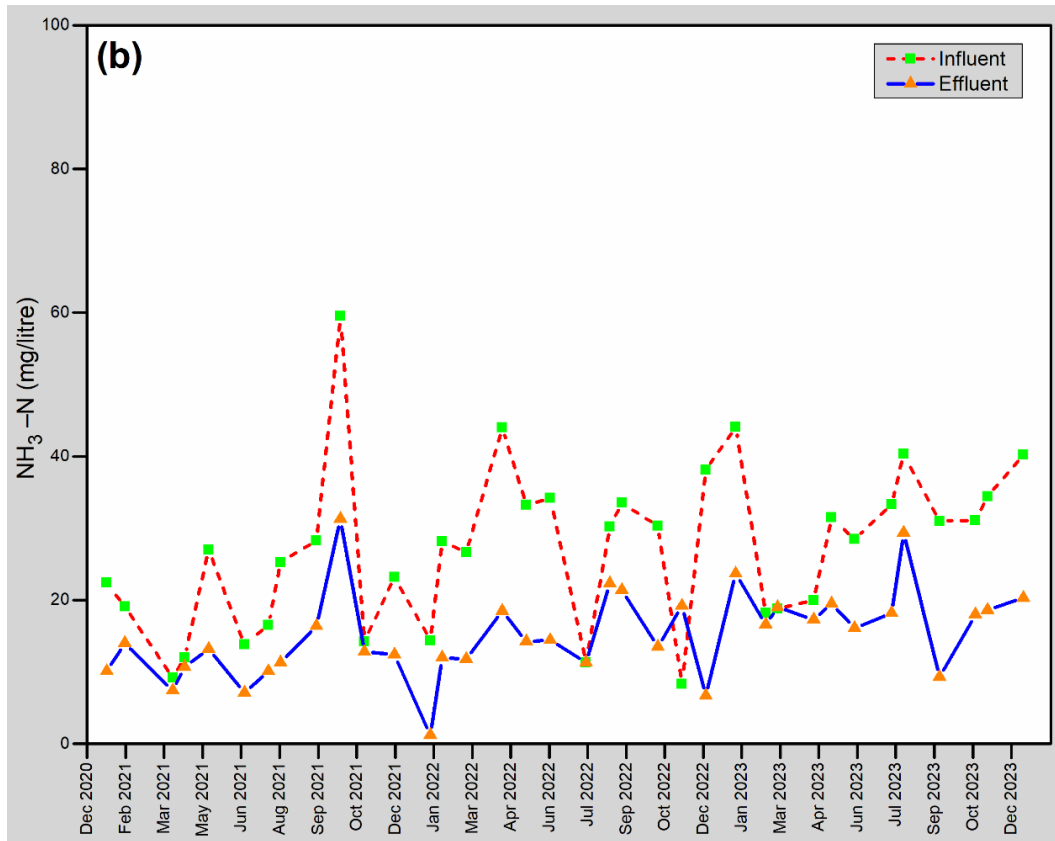
#### 4.4 Analyses of Ammonia, Nitrogen, and Phosphate Concentrations at WWTPs

The presence of excess ammonia in poorly treated wastewater effluent has been widely studied, with researchers having varying opinions on the subject. Figures 4.17 (a – d) show that the influent ammonia (NH<sub>3</sub>-N) concentrations at the four WWTPs have minimum values of 15.2 mg/l, 8.3 mg/l, 5.6 mg/l, and 6.4 mg/l, respectively. The maximum NH<sub>3</sub>-N values for the influent were 71.5 mg/l, 59.5 mg/l, 44 mg/l, and 55.3 mg/l, respectively. In the case of the effluent from these WWTPs, the minimum NH<sub>3</sub>-N levels are 5.1 mg/l, 1.2 mg/l, 0.4 mg/l, and

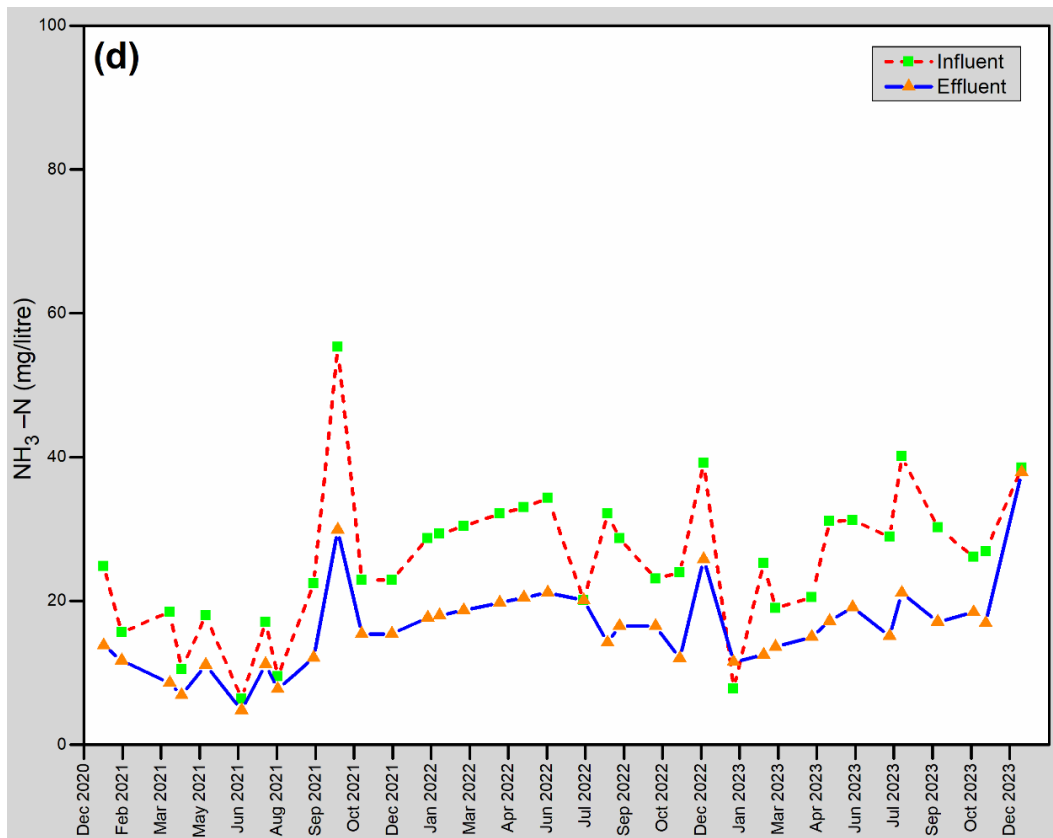
4.8 mg/l, respectively. The maximum effluent  $\text{NH}_3\text{-N}$  concentrations were 47.4 mg/l, 31.3 mg/l, 33.1 mg/l, and 37.9 mg/l, respectively. In considering the above influent and effluent  $\text{NH}_3\text{-N}$  concentrations, the removal efficiencies typically ranged between 25% and 92.9% (minimum values), and 24.8% and 47.4% (maximum values). However, in considering the average removal efficiency over the three-year period, the WWTPs are underperforming in this regard.



**Figure 4.16:**  $\text{NH}_3\text{-N}$  concentrations at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTs



**Figure 4.17:**  $\text{NH}_3\text{-N}$  concentrations at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

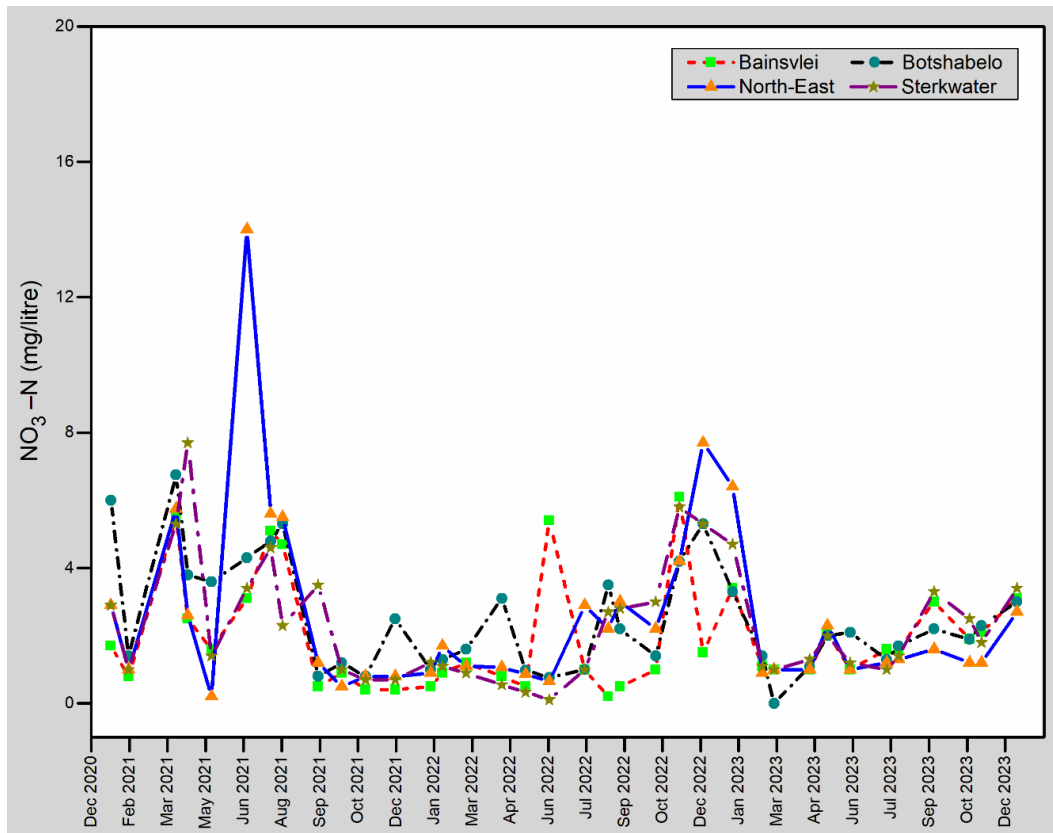


**Figure 4.17:**  $\text{NH}_3\text{-N}$  concentrations at the (a) Bainsvlei, (b) Botshabelo, (c) North-East, and (d) Sterkwater WWTPs

The discharge of ammonia-rich effluent into South African surface waters was identified by Phungela et al. (2022) as a potential environmental and health hazard. However, this was countered by the research conducted by Powders et al. (2025), who argue that ammonia from wastewater can be harnessed as a sustainable alternative to energy-intensive methods. The results from this latter study in terms of ammonia levels correlate with the findings of Masindi et al. (2022). Excess ammonia discharged into surface waters indicates that the complete oxidation of ammonia was not achieved by the WWTPs. Hence, if the situation is not addressed, it will lead to the death of marine species and associated health problems for users downstream (Karri et al., 2018).

The nitrogen levels at the four WWTPs were also investigated and are presented in Figure 4.18. The concentration of nitrate/nitrite as nitrogen ( $\text{NO}_3\text{-N}$ ) in Figure 4.18 shows that the North-East WWTP had the highest  $\text{NO}_3\text{-N}$  concentration of 14 mg/l, while all four WWTPs had  $\text{NO}_3\text{-N}$  concentrations less

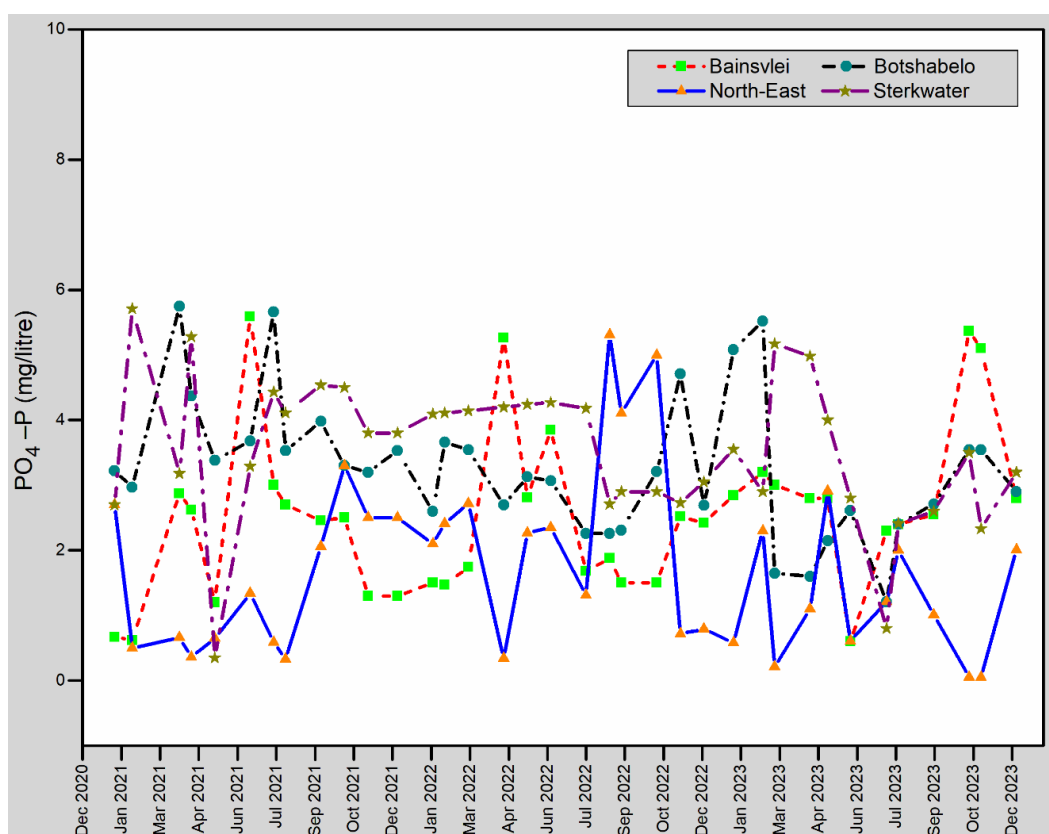
than the maximum allowable concentration of 15 mg/l throughout the sampling period (Department of Environmental Affairs, 2014).



**Figure 4.18:** Effluent NO<sub>3</sub>-N concentrations at the four WWTPs

The above findings were also in agreement with those of Masindi et al. (2022), which highlighted the implications when beneficial nitrogen required for the well-being of aquatic plants and species is lacking.

The last physicochemical parameter to consider in this section is the effluent orthophosphate (PO<sub>4</sub>-P) concentration, as shown in Figure 4.19. The minimum PO<sub>4</sub>-P concentrations at the Bainsvlei, Botshabelo, North-East, and Sterkwater WWTPs are 0.6 mg/l, 1.21 mg/l, 0.05 mg/l, and 0.35 mg/l, respectively. The maximum PO<sub>4</sub>-P concentrations at the four WWTPs are 5.59 mg/l, 5.75 mg/l, 5.31 mg/l, and 5.71 mg/l, respectively.



**Figure 4.19:** Effluent PO<sub>4</sub>-P concentrations at the four WWTPs

For all WWTPs under consideration, the PO<sub>4</sub>-P concentrations fluctuated between 0.05 mg/l and 5.75 mg/l during the three-year period. The low PO<sub>4</sub>-P concentrations confirm that the receiving water bodies will not necessarily be susceptible to eutrophication, a common phenomenon with effluent rich in nitrogen and phosphorus (Montwedi et al., 2021; Minhas et al., 2022; Zhou et al., 2022). Overall, the results correlated with the findings of Masindi et al. (2022), whose PO<sub>4</sub>-P concentrations ranged between 0.01 mg/l and 3.60 mg/l. Furthermore, statistical analyses of the variance (ANOVA) were conducted on the wastewater effluent discharged from the four WWTPs and are presented in Table 4.1.

**Table 4.1:** ANOVA testing of the sampled wastewater effluent from the four WWTPs

	Model	Sum of squares	df	Mean square	F	Significance
1	Regression	443027.4	1	443027.4	83.9	<.001 <sup>b</sup>
	Residual	749872.7	142	5280.8		
	Total	1192900.0	143			
2	Regression	463290.9	2	231645.5	44.8	<.001 <sup>c</sup>
	Residual	729609.1	141	5174.5		
	Total	1192900.0	143			

**Note:**

- a. Dependent variable: COD (mg/ℓ).
- b. Predictors: (Constant), and OA (mg/ℓ).
- c. Predictors: (Constant), OA (mg/ ℓ), and NO<sub>3</sub>-N (mg/ℓ).

The ANOVA test results listed in Table 4.1, and as recommended by Trommter et al. (2022), are characterised by a significant difference for OA and NO<sub>3</sub>-N at the 0.05 significance level. It was not clear which other parameters have a direct impact on OA and NO<sub>3</sub>-N. Hence, a Pearson correlation analysis was conducted to compare each parameter. The matrix results from the correlation analysis are presented in Table 4.2.

**Table 4.2:** Pearson correlation analysis of the physicochemical wastewater effluent parameters at the four WWTPs

Parameter	COD (mg/ℓ)	pH	EC (μS/cm)	TDS (mg/ℓ)	Temp. (°C)	OA (mg/ℓ)	NH <sub>3</sub> -N (mg/ℓ)	NO <sub>3</sub> -N (mg/ℓ)	PO <sub>4</sub> -P (mg/ℓ)	SS (mg/ℓ)
COD (mg/ℓ)	1	0.101	-0.013	-0.013	0.151	0.609**	0.169*	-0.081	0.168*	0.359**
pH	0.101	1	0.227**	0.227**	-0.055	0.238**	0.215**	-0.088	-0.094	0.273**
EC (μS/cm)	-0.013	0.227**	1	1.000**	-0.287	0.091	0.260**	-0.183*	-0.051	0.126
TDS (mg/ℓ)	-0.013	0.227**	1.000**	1	-0.287**	0.091	0.260**	-0.183*	-0.051	0.126
Temp. (°C)	0.151	-0.055	-0.287**	-0.287**	1	0.127	0.006	0.037	0.091	-0.172*
OA (mg/ℓ)	0.609**	0.238**	0.091	0.091	0.127	1	0.148	0.081	0.217**	0.394**
NH <sub>3</sub> -N (mg/ℓ)	0.169*	0.215**	0.260**	0.260**	0.006	0.148	1	-0.307**	0.143	0.218**
NO <sub>3</sub> -N (mg/ℓ)	-0.081	-0.088	-0.183*	-0.183*	0.037	0.081	-0.307**	1	0.069	-0.030
PO <sub>4</sub> -P (mg/ℓ)	0.168*	-0.094	-0.051	-0.051	0.091	0.217**	0.143	0.069	1	0.415**
SS (mg/ℓ)	0.359**	0.273**	0.126	0.126	-0.172**	0.394**	0.218**	-0.030	0.415**	1

**Note:**

\* Correlation is significant at the 0.05 level (2-tailed test).

\*\* Correlation is significant at the 0.01 level (2-tailed test).

Both two-tailed tests at 1% and 5% significance levels are listed in Table 4.2; however, the interpretation and discussion below focus on the 0.05 significance level. The Pearson correlation revealed a weak positive correlation between COD, NH<sub>3</sub>-N, and PO<sub>4</sub>-P at 0.169 and 0.168, respectively. However, a *p*-value of 0.43 and 0.44 suggests that this correlation is highly statistically different. There was no correlation between pH, COD, EC, TDS, temperature, OA, NH<sub>3</sub>-N, NO<sub>3</sub>-N, PO<sub>4</sub>-P, and SS. The correlation between EC and NO<sub>3</sub>-N is a weak negative linear relationship at -0.183 and a *p*-value of 0.028, highlighting significant differences at the 95% confidence interval. In addition, the correlation between TDS and NO<sub>3</sub>-N

was significant, i.e.,  $r(144) = -0.183$ , ( $p < 0.05$ ). Further investigations revealed that the correlation between temperature and SS is also significant,  $r(144) = -0.172$ , ( $p < 0.05$ ). For OA, pH, COD, EC, TDS, temperature,  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ ,  $\text{PO}_4\text{-P}$ , and SS, there was neither a significant correlation, nor a significant difference at the 95% confidence interval.

#### 4.5 Conclusions

In this chapter, a cost-effective approach was adopted to sample the influent and effluent at four WWTPs over a three-year period in the MMM. Only COD was used as an indicator of microorganisms (*E. coli* count) in the sampled wastewater. Given that biological tests to determine the *E. coli* count are normally time consuming and costly, considering COD as the primary indicator of pathogenic bacteria proved to be effective. The results also confirmed that all WWTPs in the MMM failed to meet the recommended SANS COD = 75 mg/l. Several factors were identified as contributors for not meeting the required standard, e.g., ranging from a lack of technical expertise to a lack of funding. Further investigations towards the properties of the sampled influent and effluent in terms  $\text{BOD}_{\text{cal}}$ , pH, temperature, and OA showed that OA significantly impacts the organic concentration in the wastewater effluent. Statistical analyses were deemed necessary given that no clear trend(s) were evident between  $\text{BOD}_{\text{cal}}$ , pH, and temperature. The BOD/COD ratios also confirmed that the COD is approximately twice the  $\text{BOD}_{\text{cal}}$  as per convention and recommended in literature.

The potential impact of ammonia, nitrogen and phosphate was also investigated. The results revealed that the ammonia levels were higher than the SANS effluent standards. However, the nitrogen and phosphate concentrations were within the national standards. Given that the levels of nitrogen and phosphorus were regarded as insufficiently low, concerns were raised about their impact on plants and aquatic species requiring a certain level of nutrients.

The elevated concentrations of COD found in the wastewater effluent due to non-conformance to the SANS criteria raised concerns. A correlation matrix inclusive of variance and the Pearson correlation between the various physicochemical

wastewater effluent parameters revealed high statistical differences associated with either positive or negative correlations, or both. The elevated pH levels, fluctuating EC, and TDS found in the sampled surface waters present potential hazards to the well-being of the general population. Although no direct link with the 2023 Cholera outbreak could be established, the results agreed with other literature and highlighted that the prolonged use of water from the Renoster Spruit, Bloem Spruit, and Klein Modder River could lead to kidney and cardiovascular complications owing to the high TDS concentrations.

The results pertaining to the impact of effluent non-compliance are included and discussed in the next chapter.

## CHAPTER 5 : POTENTIAL IMPACT OF EFFLUENT NON-COMPLIANCE ON SURFACE WATER

This chapter presents the analyses and discussion of the results obtained from the four WWTPs towards achieving the following specific objectives as outlined in Chapter 1:

- (a) Investigate and compare the effluent quality compliance at the four WWTPs with the SANS criteria for wastewater effluent; and
- (b) Investigate the potential environmental and health impacts of the wastewater effluent discharged into surface water bodies within the MMM.

Addressing the above-listed objectives will highlight how the quality of the wastewater effluent discharge will affect the DO, pH, turbidity, and temperature levels in the receiving surface water bodies. Ultimately, this will have a potentially negative impact on the environment and human health.

### 5.1 Investigation and Comparison of Effluent Quality Compliance at WWTPs

Given the poor wastewater effluent quality from the four WWTPs (Bainsvlei, Botshabelo, North-East, and Sterkwater), it was imperative to determine the percentage reduction in pollutant concentration from influent to effluent. The treatment efficiency at each of the four WWTPs was determined using Equation 5.1, with the results presented in Table 5.1.

$$COD_{\%} = 100 \left[ \frac{(COD_i - COD_e)}{COD_i} \right] \quad (5.1)$$

Where:

$COD_{\%}$  = COD removal efficiency (%),

$COD_e$  = effluent COD concentration (mg/l), and

$COD_i$  = influent COD concentration (mg/l).

**Table 5.1:** Average annual COD treatment efficiency at the four WWTPs

WWTP	Hydraulic Capacity (M <sup>3</sup> /day)	Year	Avg. COD removal efficiency (%)
Bainsvlei	5	2021	85.9
		2022	77.5
		2023	66.3
Botshabelo	20	2021	67.1
		2022	17.7
		2023	48.4
North-East	20	2021	72.8
		2022	22.5
		2023	60.7
Sterkwater	20	2021	59.2
		2022	62.4
		2023	34.6

The COD removal efficiencies (%) of the WWTPs in Table 5.1 show that the Bainsvlei WWTP outperformed the Botshabelo, North-East, and Sterkwater WWTPs, although, the Bainsvlei WWTP had a declining efficiency from the first year to the third year of sampling, with an efficiency reduction of 25.9%. This decline is unwarranted and could significantly impact the receiving surface water body. The Botshabelo and North-East WWTPs experienced a steep decline in the COD removal efficiency during the second year, with an efficiency reduction of over 100%. The situation improved slightly in the third year, from a yearly average of 17.7% to 48.4% in the case of the Botshabelo WWTP, while from the second to the third year at the North-East WWTP, the COD (%) increased from 22.5% to 60.7%. The COD removal efficiency at the Sterkwater WWTP declined less significant over the sampling period, i.e., 5.3%. It is alarming to witness such poor treatment efficiencies at WWTPs designed, built, and supposedly operated according to set standards. According to Masindi et al. (2022), treatment efficiencies might be impacted by a lack of proper maintenance, funding, and technical expertise. Improving the status of the WWTPs is crucial, as there have been reported cases of cholera outbreaks across South Africa, with 47 fatalities (Department of Health, 2023).

The other relevant physical and chemical wastewater effluent characteristics are presented in Table 5.2.

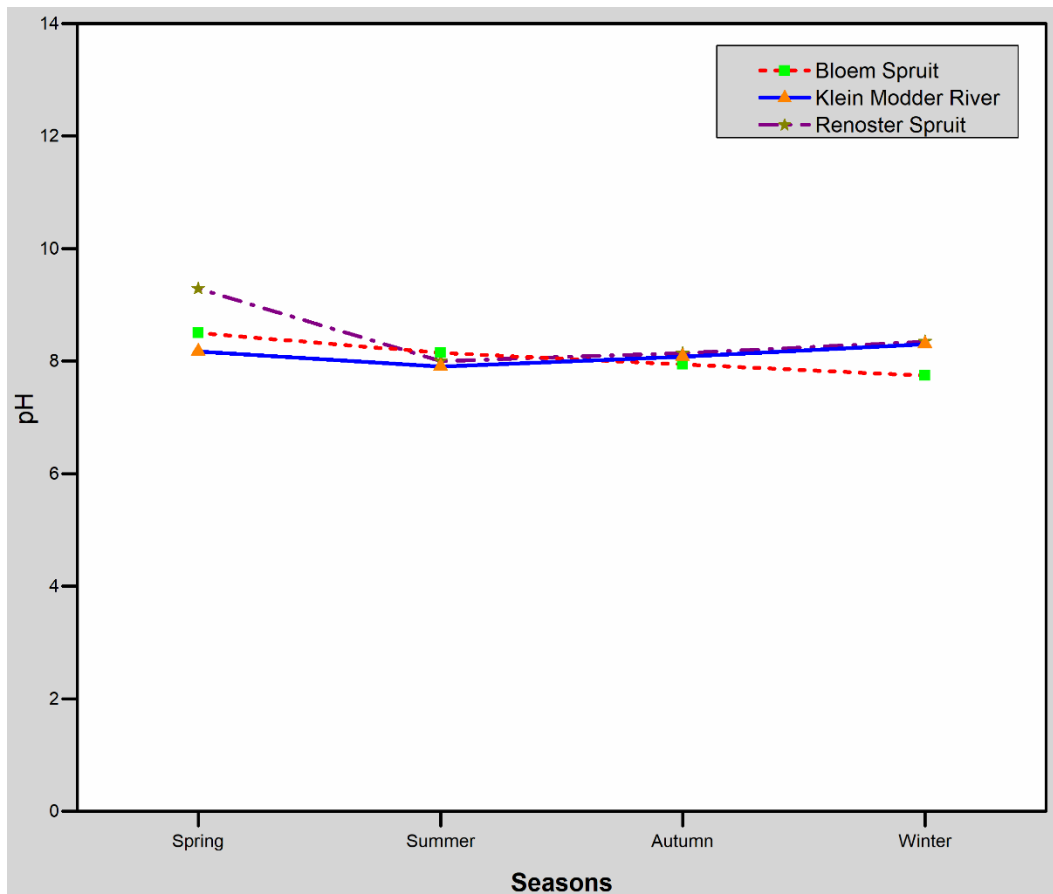
**Table 5.2:** Average annual physical and chemical wastewater effluent characteristics at the four WWTPs compared to the SANS criteria

WWTP	Year	pH	EC ( $\mu\text{S}/\text{cm}$ )	TDS ( $\text{mg}/\ell$ )	Temp. ( $^{\circ}\text{C}$ )	COD ( $\text{mg}/\ell$ )	OA ( $\text{mg}/\ell$ )	$\text{NH}_3\text{-N}$ ( $\text{mg}/\ell$ )	$\text{NO}_3\text{-N}$ ( $\text{mg}/\ell$ )	$\text{PO}_4\text{-P}$ ( $\text{mg}/\ell$ )	SS ( $\text{mg}/\ell$ )
Bainsvlei	2021	7.3	68.3	454.8	22.8	77.3	17.7	17.3	2.3	2.2	21.9
	2022	8.1	70.0	466.1	21.4	90.8	17.4	22.8	1.6	2.3	32.7
	2023	9.0	76.5	509.3	21.3	158.6	28.1	19.6	1.9	3.0	68.6
Botshabelo	2021	6.8	53.3	355.2	23.0	178.7	27.0	13.2	3.4	3.9	77.8
	2022	7.9	43.9	292.5	22.1	130.6	21.0	13.9	2.2	3.0	64.1
	2023	8.6	64.7	431.0	21.8	205.8	38.6	18.8	1.9	2.9	108.1
North-East	2021	7.0	59.8	398.2	23.0	108.0	18.4	10.8	3.4	1.5	24.4
	2022	7.9	61.3	408.4	22.6	143.8	23.4	18.8	3.7	2.5	51.8
	2023	8.9	60.4	402.0	21.6	114.3	23.3	10.6	1.8	1.2	39.9
Sterkwater	2021	7.2	53.6	356.9	22.9	129.3	21.7	12.4	2.9	3.8	40.6
	2022	7.9	54.8	364.6	21.8	140.7	25.7	18.4	2.1	3.6	64.8
	2023	8.8	58.0	386.5	21.2	156.0	30.5	18.0	2.1	3.2	70.3
<b>SANS Criteria</b>	<b>5.5 - 9.5</b>	<b>&lt; 150</b>	<b>&lt; 999</b>	<b>&lt; 35</b>	<b>&lt; 75</b>	<b>&lt; 10</b>	<b>&lt; 6</b>	<b>&lt; 15</b>	<b>&lt; 10</b>	<b>&lt; 25</b>	
											<b>&gt; SANS Criteria</b>

There are no temperature requirements for wastewater effluent discharged into surface waters per the SANS criteria. However, the SS from the four WWTPs were significantly higher than the recommended standards. Unfortunately, all four WWTPs had significant amounts of ammonia in terms of nitrogen ( $\text{NH}_3\text{-N}$ ), namely between 10.4 and 20.4  $\text{mg}/\ell$ . The high levels of  $\text{NH}_3\text{-N}$  at the Bainsvlei, North-East and Sterkwater WWTPs negatively impact available oxygen, as evident in Table 5.2. Furthermore, investigations by Zhang et al. (2018) showed that  $\text{NH}_3\text{-N}$  poses ecological risks and affects aquatic life. The EC,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  from the four WWTPs conform to the SANS criteria.

## 5.2 Potential Impact of Effluent Non-compliance on Surface Water

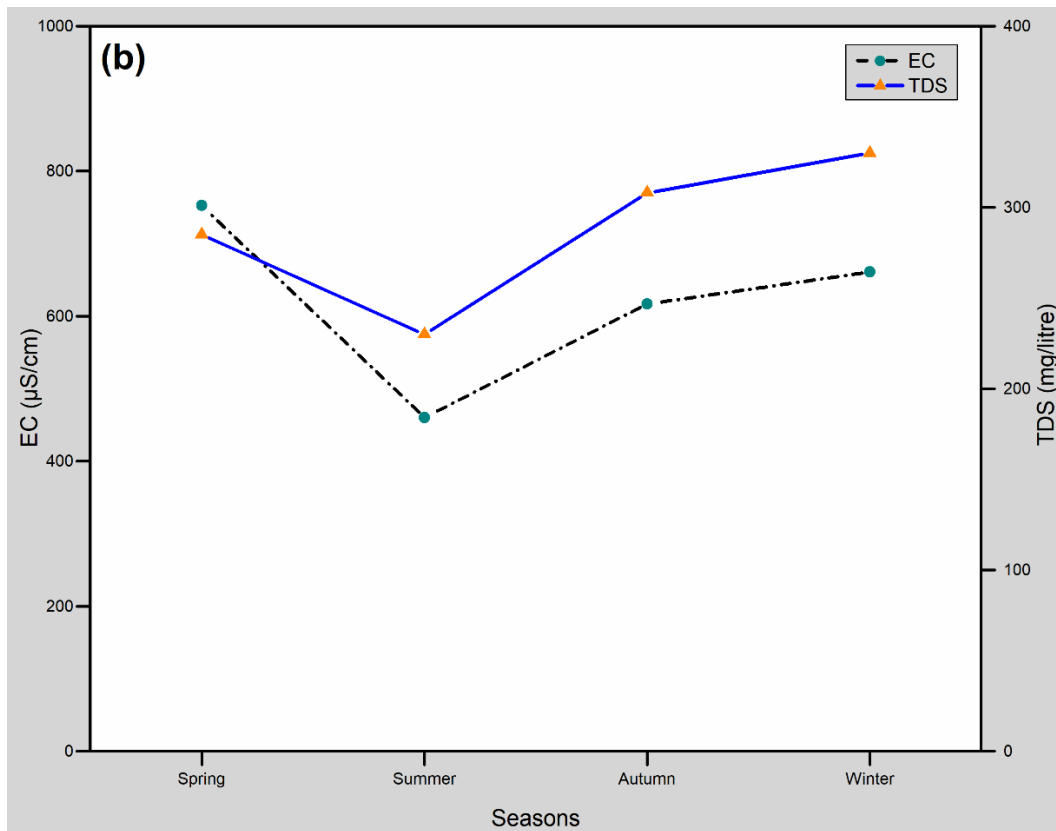
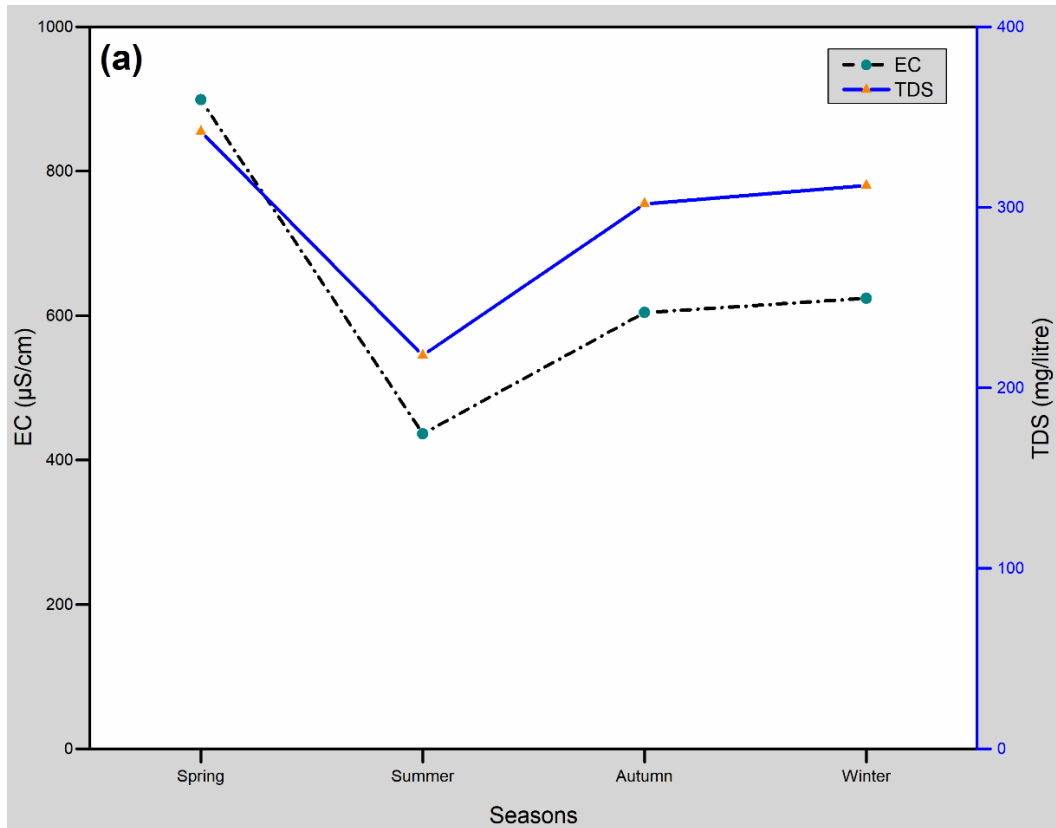
Although the Bloem Spruit WWTP was not sampled in this study, it still discharges wastewater effluent into the Bloem Spruit. Hence, sampling of the Bloem Spruit is crucial as it joins the Renoster Spruit (Belle et al., 2020). The pH values associated with the three sampled tributaries of the Modder River, i.e., Renoster Spruit, Bloem Spruit, and Klein Modder River, are shown in Figure 5.1.



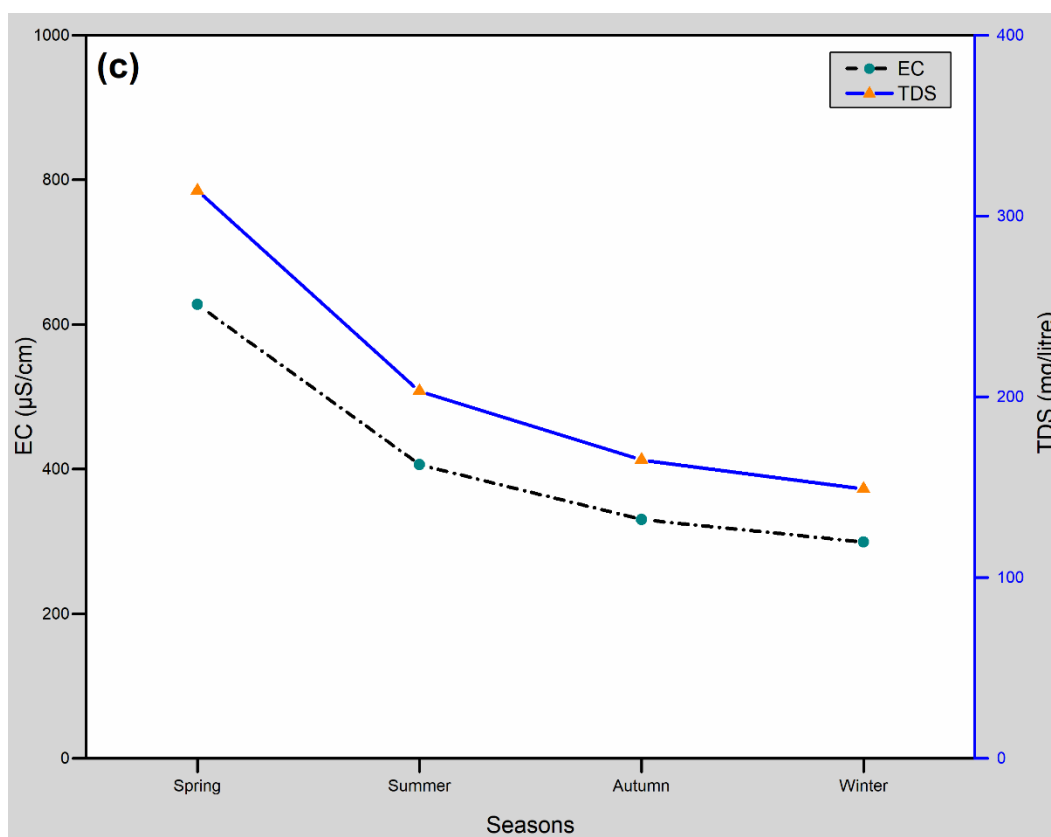
**Figure 5.1:** pH Values at the three sampled tributaries of the Modder River System

Results from Belle et al. (2020) showed that the Bloem Spruit is heavily polluted by anthropogenic activities, especially from the Bloem Spruit WWTP. The Renoster Spruit had the highest pH value of 9.3, which could be ascribed to the activities at the Bloem Spruit, as well as the wastewater effluent discharged into the Renoster Spruit (Genthe et al., 2004; Belle et al., 2020). According to the Department of Water Affairs and Forestry (DWAF) water quality guidelines (DWAF, 1996), a pH value > 9 poses the likelihood of adverse effects linked to deprotonated species, e.g., the deprotonation of ammonium to produce ammonia experiencing a notable surge, while water becomes bitter. Hence, regulating activities along the Bloem Spruit and wastewater effluent quality require urgent attention.

The EC and TDS from the three sampled sites are presented in Figures 5.2 (a–c).



**Figure 5.2:** Sampled EC and TDS at the surface water sites in the Modder River System: (a) Renoster Spruit, (b) Bloem Spruit, and (c) Klein Modder River



**Figure 5.2:** Sampled EC and TDS at the surface water sites in the Modder River System: (a) Renoster Spruit, (b) Bloem Spruit, and (c) Klein Modder River

EC and TDS are commonly utilised as general parameters to assess water's physical attributes and overall quality by offering insights into the salinity levels and concentration of dissolved substances. In Figure 5.2, EC and TDS also demonstrate seasonal fluctuations, with the highest EC values in spring. The measured EC values varied between 628  $\mu\text{S}/\text{cm}$  (Klein Modder River), 753  $\mu\text{S}/\text{cm}$  (Bloem Spruit), and 899  $\mu\text{S}/\text{cm}$  (Renoster Spruit). In terms of TDS, the highest recordings were during spring, e.g., Renoster Spruit (342  $\text{mg}/\ell$ ) and Klein Modder River (314  $\text{mg}/\ell$ ). In contrast at Bloem Spruit, the highest TDS value (330  $\text{mg}/\ell$ ) was recorded in winter, which is most likely attributed to the lower water volume, leading to fewer diluted dissolved solids and higher salinity (*cf.* Table 5.3). Subsequently, such elevated levels of TDS in surface water bodies will impact on human health with associated economic implications. Typically, high TDS concentrations indicate the presence of minerals such as magnesium, calcium, and sodium. Prolonged consumption of such contaminated water could lead to

kidney and cardiovascular complications. Furthermore, treating water to remove TDS is expensive and leads to corrosion and scaling in water pipes. Other physical in-situ parameters from the sampled surface waters, e.g., atmospheric pressure (ATM), oxidation reduction potential (ORP), resistivity, and salinity are presented in Table 5.3.

**Table 5.3:** In situ parameters from the seasonally sampled surface waters at the Renoster Spruit (SWRS01), Bloem Spruit (SWRS02), and Klein Modder River (SWRS03)

Parameter	Renoster Spruit (SWRS01)				Bloem Spruit (SWRS02)				Klein Modder River (SWKLM03)			
	A	B	C	D	A	B	C	D	A	B	C	D
Temp.(°C)	17.9	34.4	24.9	19.1	17.4	33.9	25.6	20.5	25.5	35.5	24.3	25.1
EC (µS/cm)	899	436	604	624	753	460	617	661	628	406	330	299
DO (mg/l)	3.9	2.6	3.3	4.4	3.3	3.2	3.9	4.5	5.0	2.3	2.7	3.2
ATM (mmHg)	639	652	654	660	637	653	651	658	650	650	649	655
pH	9.3	8.0	8.1	8.4	8.5	8.2	7.9	7.7	8.2	7.9	8.1	8.3
ORP (mV)	-141	-296	-502	-370	-470	-350	40	129	-180	-145	-614	-407
Resistivity (mΩ.cm)	0.0025	0.0023	0.0017	0.0016	0.0018	0.0022	0.0016	0.0015	0.0016	0.0025	0.003	0.003
TDS (ppm)	342	218	302	312	285	230	308	330	314	203	165	149
Salinity (PSU)	0.34	0.20	0.29	0.30	0.28	0.22	0.30	0.32	0.30	0.19	0.16	0.14

**Note:** Spring (A): September, October, and November; Summer (B): December, January, and February; Autumn (C): March, April, and May; and Winter (D): June, July, and August.

The parameters for the sampled surface water in Table 5.3 indicate elevated temperatures in summer, which could promote algal bloom, which is similar to previous reporting in this regard (Wang et al., 2022). Interestingly, the DWAF water quality guidelines report (DWAF, 1996) does not specify the temperature limit for surface waters; however, it indicates a correlation between TDS, EC, and temperature. Dissolved oxygen plays a vital role in water quality, indicating the amount of oxygen available for microorganisms to degrade organic matter. In this research, the DO concentrations ranged between 2.6 mg/l and 5 mg/l. These values are below the recommended criteria stipulated by the United States Environmental Protection Agency (EPA), i.e., 5 mg/l (Mohamed et al., 2014). However, SANS requires at least a 75% saturation (DWAF, 1991). Thus, meeting

this criterion would guarantee conformity with environmental regulations established to safeguard water bodies from the negative impacts of wastewater effluents. Unfortunately, the DO concentrations reported above were associated with saturation levels ranging between 28.7% and 55%, which confirm a DO deficiency having a direct impact on aquatic species. Hence, if the status quo is maintained, adverse effects will follow to impact the environmental sustainability negatively, leading to eutrophic waters. The major challenge in addressing poor effluent quality discharged into surface water is not related to policy or monitoring programmes given that the Department of Water and Sanitation (DWS) provides adequate monitoring programmes, as detailed in DWS (2019). However, policy enforcement and adherence by all stakeholders remain a huge challenge in South Africa. Hence, the government and all stakeholders must ensure that the aforementioned problems are addressed.

### **5.3 Conclusions**

In this chapter, the impact of wastewater effluent discharged into surface water bodies within the MMM was evaluated to understand the associated environmental and public health implications.

In general, the physical access to the WWTPs and SW sites and relevant data sets, along with the associated cumbersome administrative processes to grant such access, are regarded as problematic and have delayed the progress and completion of this research. Apart from the above concerns, it was also evident that sporadic power failures, financial constraints, a shortage of adequately skilled personnel, and inadequate maintenance protocols have a negative impact on the functionality, operation, and management of WWTPs in the MMM. Hence, these critical matters necessitate immediate consideration and resolution to protect the well-being and interests of the residents of the MMM in the FS province.

The results pertaining to the design and simulation of re-engineered WSPs for improved contaminant removal are included and discussed in the next chapter.

## CHAPTER 6 : WASTEWATER STABILISATION POND SIMULATION

This chapter presents the analyses and discussion of the results obtained from the four WWTPs towards achieving the following specific objectives as outlined in Chapter 1:

- (a) Explore sludge distribution in a WSP system by considering different baffle configurations, flow rates, and/or pond geometry; and
- (b) Explore the design and simulation of a re-engineered WSP for improved contaminant removal by comparing results with effluent originating from the four WWTPs.

Typically, addressing the above-listed objectives will highlight how the surface-area-to-volume ratio of a WSP influences the wastewater treatment efficiency. In addition, by comparing the simulated contaminant removal efficiencies of the re-engineered WSPs to the measured effluent quality discharged from the existing WWTPs, the suitability of the various design configurations suggested for the re-engineered WSPs will be confirmed or rejected.

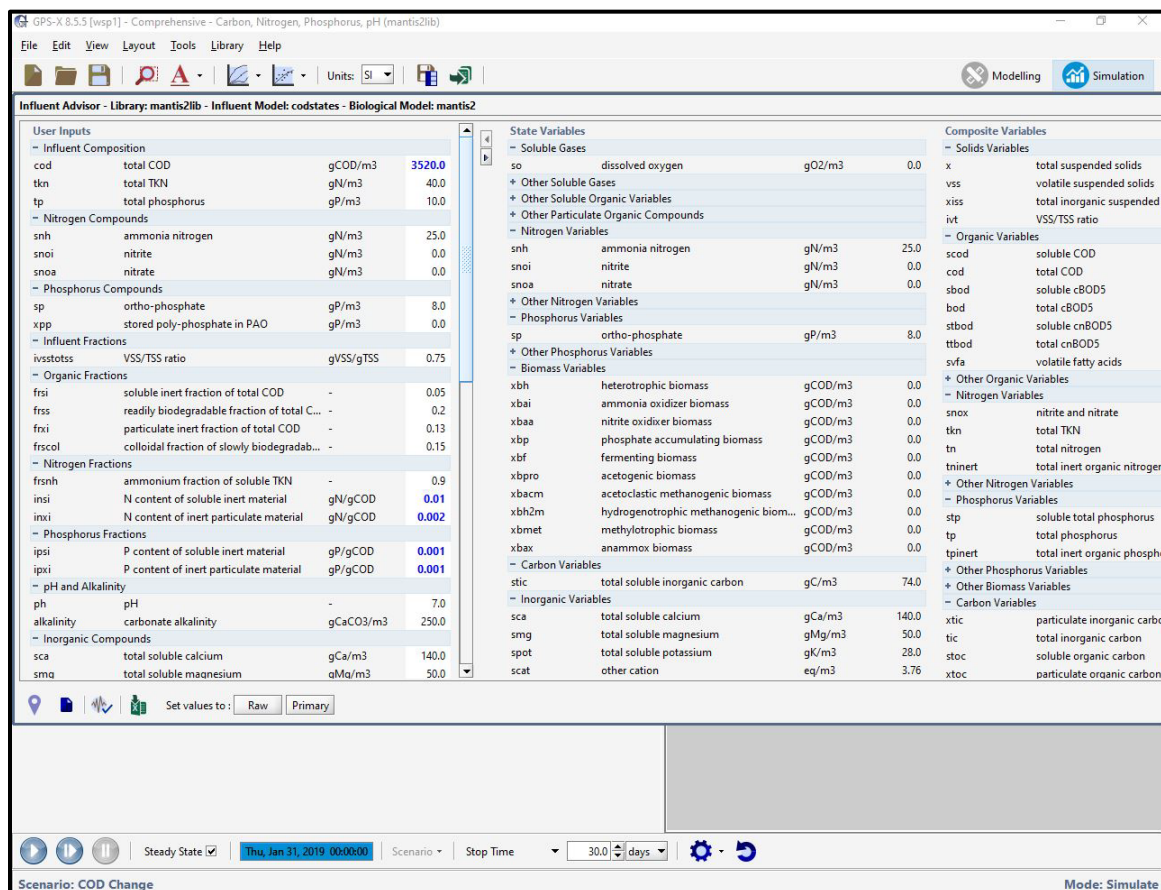
### 6.1 Sludge Distribution in a WSP

In general, modelling and simulation allow designers, operators, and WWTP managers to assess wastewater effluent and the various unit processes at a reduced cost. Furthermore, modelling also offers the opportunity to troubleshoot and optimise unit processes. Given the problems regarding poor wastewater effluent quality in the MMM, the Hydromantis GPS-X 8.5 simulation software was used to model and simulate various WSP design scenarios, as detailed in Chapter 3. To facilitate the simulation of the various WSP design configurations, the base WSP model had to be calibrated as shown in Table 6.1.

**Table 6.1:** Calibration parameters applicable to the base WSP model

Measured Parameter	Actual value	Model value	Units
Total COD	3 520	3 520	gCOD/m <sup>3</sup>
NH3-N	27	27	gN/m <sup>3</sup>
Total Kjeldahl Nitrogen (TKN)	40	40	gN/m <sup>3</sup>
Total Carbonaceous BOD <sub>5</sub>	1 816	1 817.8	gO <sub>2</sub> /m <sup>3</sup>
Soluble COD	671	672.3	gCOD/m <sup>3</sup>
Total Suspended Solids (TSS)	1 976	1 977.8	g/m <sup>3</sup>
Volatile Suspended Solids (VSS)	1 581	1 582.2	g/m <sup>3</sup>
Soluble TKN	31	30.7	gN/m <sup>3</sup>

Some of the wastewater influent fractions had to be adjusted to ensure the modelled values match the input parameters. If this were not done, the GPS-X simulation software will highlight the need for such adjustments. This makes it impossible to accept changes, nor would it allow the user to build the model for simulation. Hence, some of the nitrogen and phosphorus fractions were adjusted as shown in Figure 6.1.

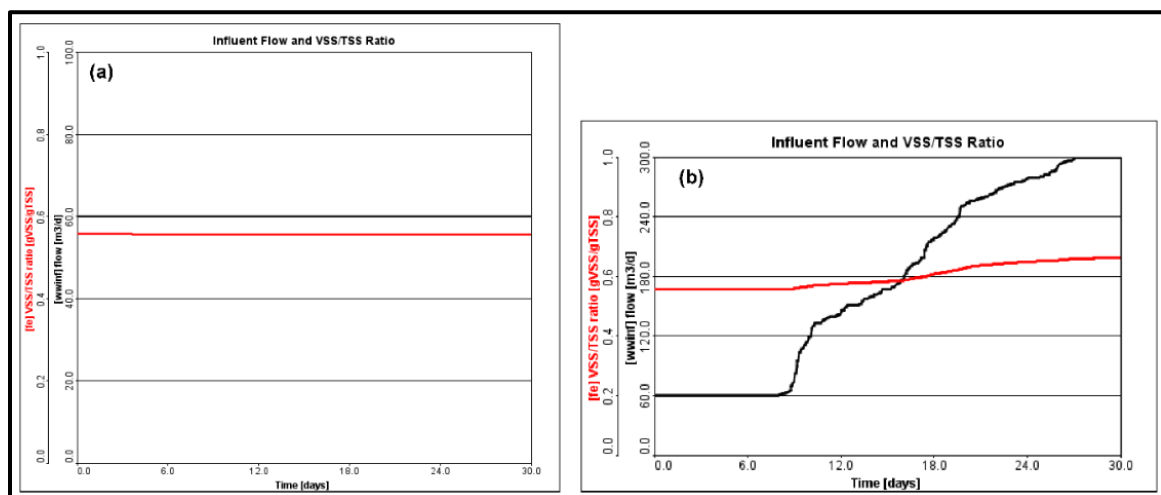


**Figure 6.1:** Adjusted nitrogen and phosphorus influent fractions for the simulated WSP model

As highlighted in the methodology (*cf.* Chapter 3), assessing the sludge distribution, design, and/or simulation of the base and re-engineered WSPs were based on the following scenarios (*cf.* Figure 3.14, Chapter 3):

- (a) **Scenario 1:** No modifications (base WSP model);
- (b) **Scenario 2:** Three zones with CSTRs;
- (c) **Scenario 3:** Six zones with CSTRs; and
- (d) **Scenario 4:** Eight zones with CSTRs.

**WSP Scenario 1:** Scenario 1 outlines the effect of sludge distribution and its interactions regarding baffles, flow rate, and/or pond geometry. The WSP was simulated for steady and dynamic state flow conditions as shown in Figures 6.2 (a) and (b), respectively.



**Figure 6.2:** WSP Scenario 1: Relationship between influent flow rates and (a) the effluent TSS under steady state, and (b) VSS/TSS ratio under dynamic state conditions

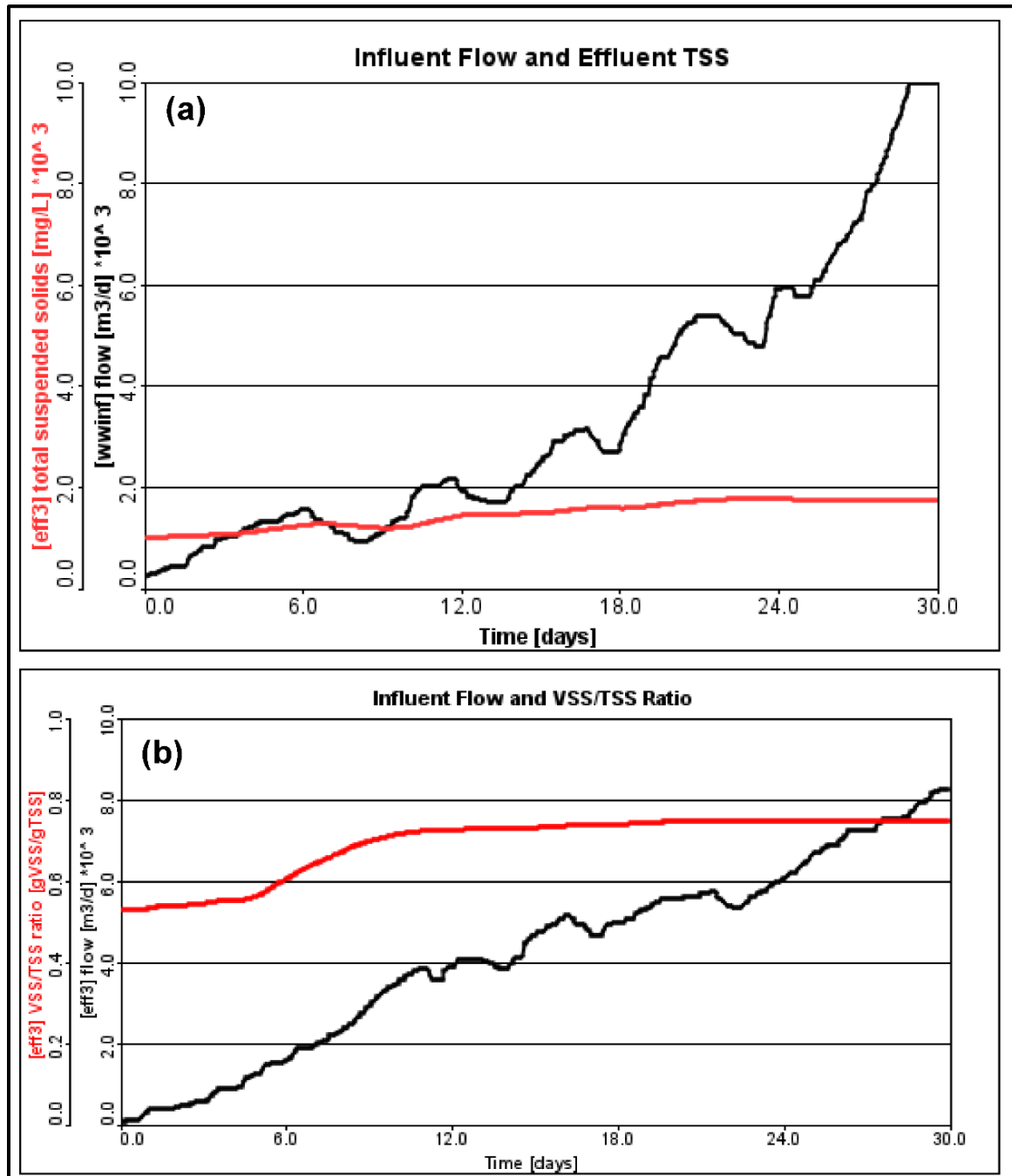
As shown in Figure 6.2 (a), the initial influent flow rate was 60 m<sup>3</sup>/day under steady state conditions, while a 30-day simulation period was carried out with varying influent flow rates to present the dynamic state conditions and their effect on the TSS as shown in Figure 6.2 (b). The sludge distribution was not catered for in the specific treatment process in the model. Given that a more expensive and complicated CFD tool would be required to model the sludge distribution accurately, the VSS and TSS were used as an alternative to evaluate the impact of

re-engineered WSPs on the sludge quality. Leal et al. (2020) and Lamssali et al. (2024) confirmed that VSS/TSS ratios serve as a practical metric for assessing the concentrations of organic pollutants and the efficacy of treatment processes in a WSP, thereby exerting a considerable effect on the management of effluent discharge and the subsequent environmental impacts thereof.

Under steady state flow conditions, the VSS/TSS ratio was typically less than 0.6, while for the dynamic state, the VSS/TSS ratios were  $\leq 0.6$ . The VSS/TSS ratios  $\approx 0.6$  are less than the values proposed by Menoni and Bertanza (2016), i.e., VSS/TSS ratio  $> 0.7$ . Hence, it was evident that the unmodified WSP requires further modifications.

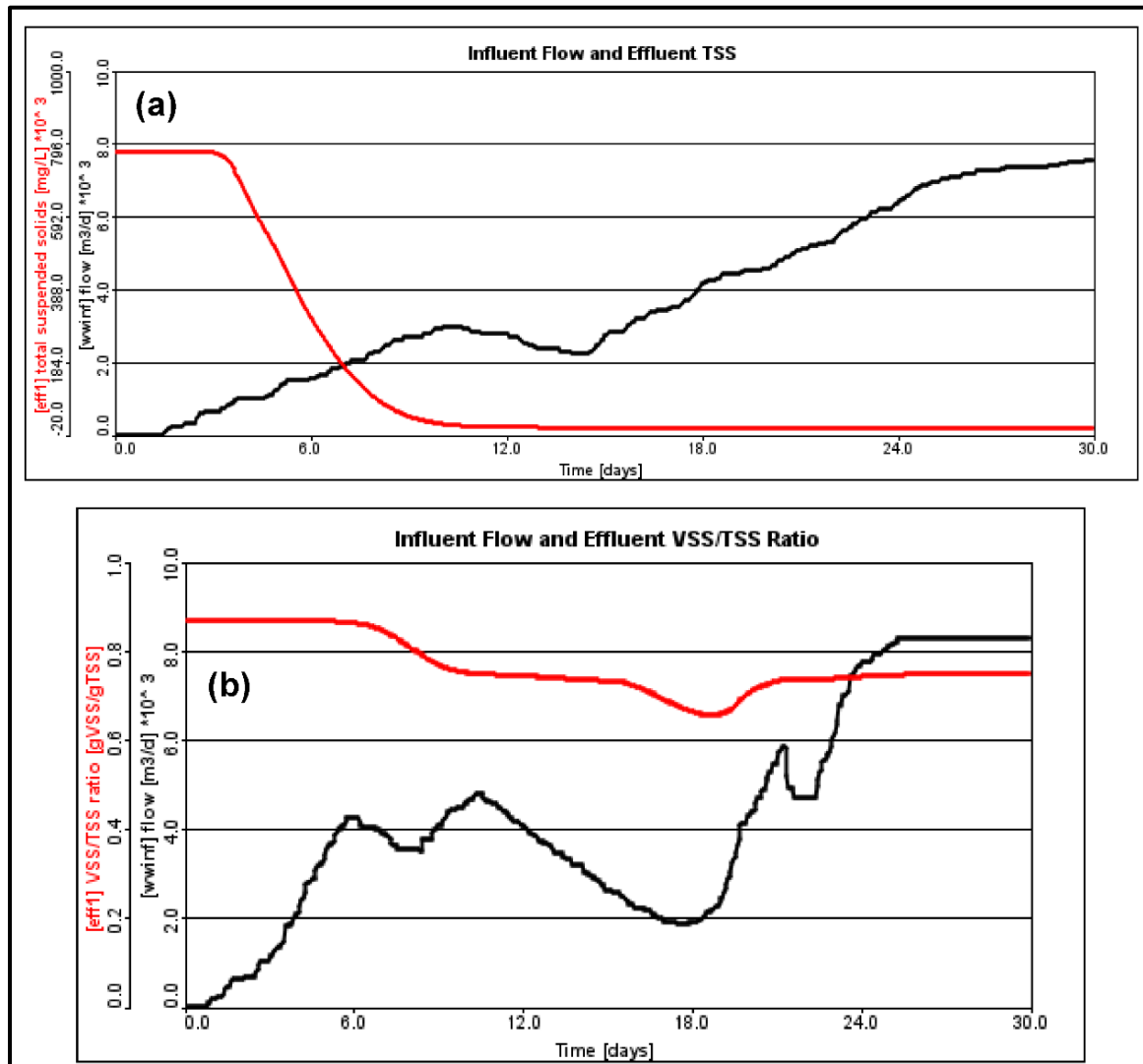
**WSP Scenario 2:** The re-engineered WSP for Scenario 2 is presented in Figures 6.3 (a) and (b).

Comparing the performance of the steady and dynamic state flow simulations, the results from the latter showcase real-world expectations and are comparable to the findings by Xue et al. (2025). Hence, dynamic state conditions were adopted for the remainder of the simulations, i.e., Scenarios 3 and 4. In considering the effect of flow rate on the TSS in Figure 6.3 (a), it is evident that TSS values  $< 2000$  mg/l are achieved, which are comparable to the TSS values achieved by Xue et al. (2025), i.e., TSS concentrations between 1 753 and 1 980 mg/l. In Figure 6.3 (b), the VSS/TSS ratio is lower than the typical values at the start of the simulation with no inflow. As the flow rate increases, the VSS/TSS ratio increases until the maximum VSS/TSS ratio ( $> 0.7$ ) is reached at approximately 10 days, after which, the ratios remained constant, thereby confirming that the WSP configuration efficiently removed the organic pollutants in the effluent.



**Figure 6.3:** WSP Scenario 2: Relationship between the influent flow rates and (a) the effluent TSS, and (b) the VSS/TSS ratio under dynamic state flow conditions

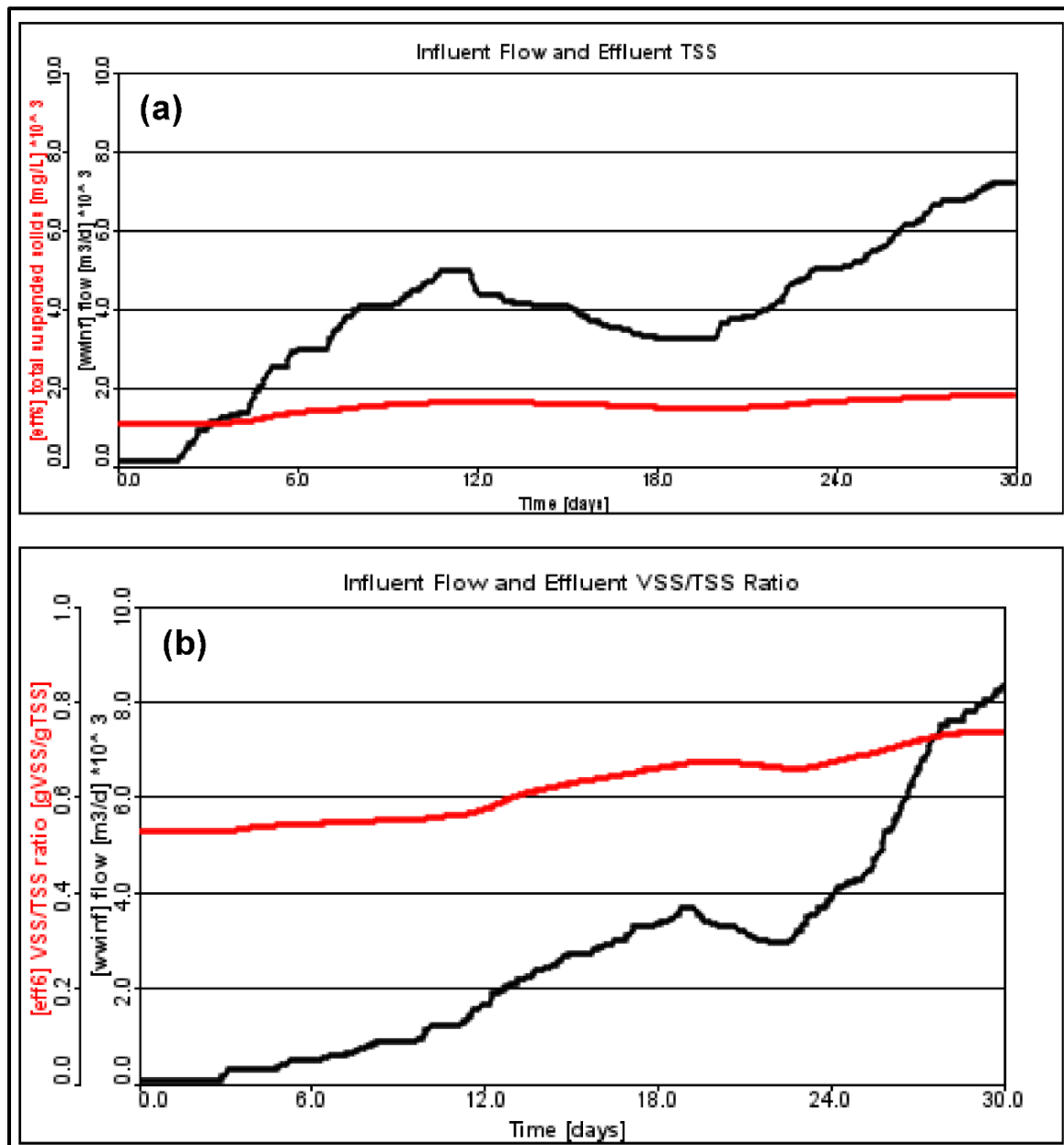
**WSP Scenario 3:** The re-engineered WSP for Scenario 3 is presented in Figures 6.4 (a) and (b).



**Figure 6.4:** WSP Scenario 3: Relationship between the influent flow rates and (a) the effluent TSS, and (b) the VSS/TSS ratio under dynamic state flow conditions

The effluent TSS values and VSS/TSS ratio in Figures 6.4 (a) and (b) showed remarkable improvements in comparison to Scenarios 1, 2 and 4, with the highest TSS values and VSS/TSS ratio ranging between 0.7 and 0.8, while adhering to the maximum limit as recommended by Xue et al. (2025).

**WSP Scenario 4:** The re-engineered WSP for Scenario 4 is presented in Figures 6.5 (a) and (b).



**Figure 6.5:** WSP Scenario 4: Relationship between the influent flow rates and (a) the effluent TSS, and (b) VSS/TSS ratio under dynamic state flow conditions

In Figure 6.5 (a), the increasing flow rate did not have much impact on effluent TSS. The TSS increased slightly at 1 500 m<sup>3</sup>/day during days 0 – 6, with some further increases until a flow rate of 1 900 m<sup>3</sup>/day was achieved. Thereafter, from Day 18, the TSS decreased slightly followed by constant values until the end of the simulation. At the start of the simulation, the VSS/TSS ratio values were lower than the recommended value of > 0.7. However, the VSS/TSS ratio improved as the flow rate increased and typically ranged between 0.56 and 0.76. The latter ratios are comparable to those obtained by Lamssali et al. (2024), i.e., VSS/TSS > 0.67.

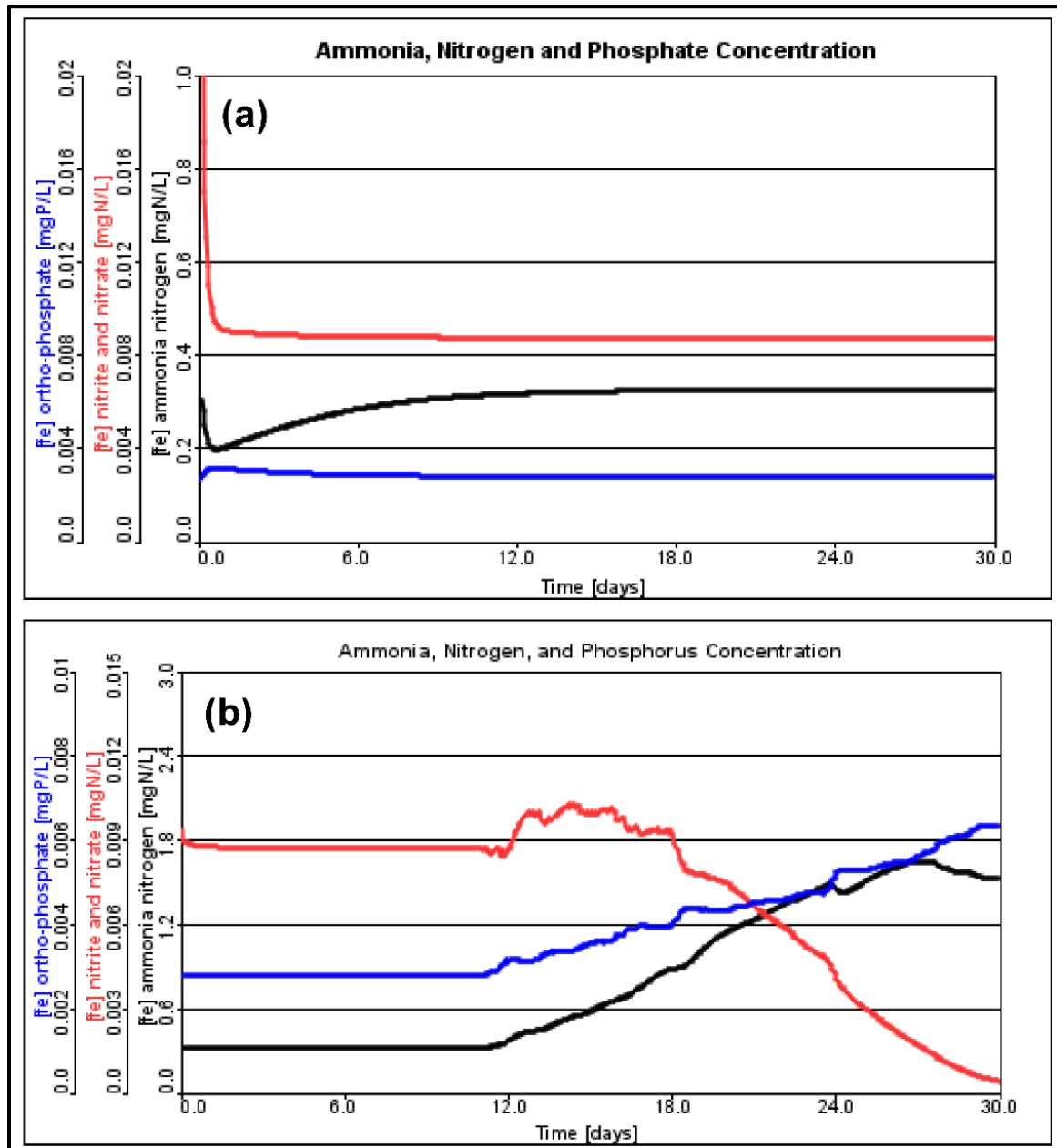
In terms of TSS values and VSS/TSS ratios achieved by considering the different WSP design configurations, Scenario 3 represented the best-performing WSP configuration.

## 6.2 Design and Simulation of the Re-engineered WSPs

This section explores the ability of the re-engineered WSPs to remove nutrients such as ammonia ( $\text{NH}_3\text{-N}$ ), nitrogen ( $\text{NO}_3\text{-N}$ ), and phosphate ( $\text{PO}_4\text{-P}$ ) in comparison to the wastewater effluent originating from the four WWTPs.

**WSP Scenario 1:** As in the case of Section 6.1, the impact of both steady (Figure 6.6 [a]) and dynamic state (Figure 6.6 [b]) flow conditions on nutrient removal were considered. In Figure 6.6 (a), no noticeable changes in nutrient removal are evident. However, the values obtained showed that the unmodified WSP under steady state flow conditions have low levels of ammonia and nitrogen in terms of nitrate/nitrite, and phosphate, thus meeting the SANS wastewater effluent criteria. Similarly, under dynamic state flow conditions in Figure 6.6 (b), all the  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  parameters conform to the SANS criteria.

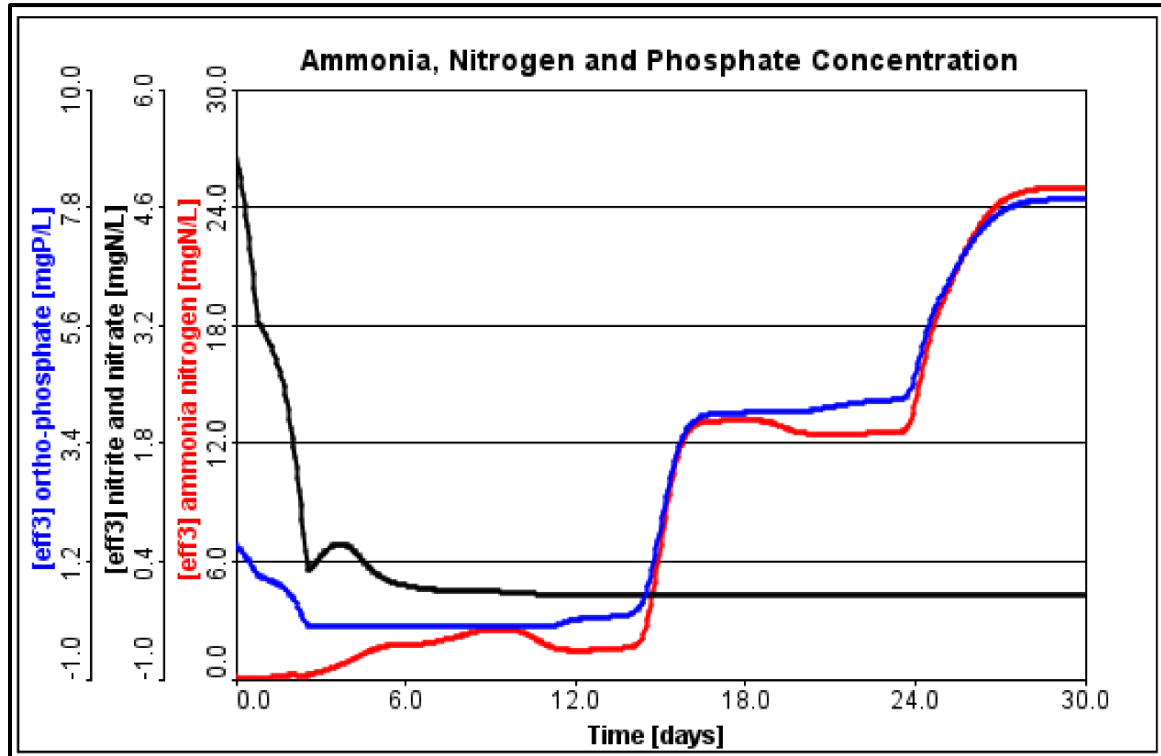
In comparing the steady and dynamic state results, the  $\text{NH}_3\text{-N}$  levels fluctuated between 0.3 mg/l and 0.38 mg/l (steady state), and 0.5 mg/l and 1.28 mg/l (dynamic state). The  $\text{NO}_3\text{-N}$  levels fluctuated between 0.0082 mg/l and 0.02 mg/l (steady state), and 0.0092 mg/l and 0.0012 mg/l (dynamic state). The  $\text{PO}_4\text{-P}$  levels ranged between 0.0038 mg/l and 0.0036 mg/l (steady state), and 0.0026 mg/l and 0.0062 mg/l (dynamic state). All the above nutrient levels are within acceptable levels and are comparable to those obtained by Navarro et al. (2024). The above  $\text{NH}_3\text{-N}$  levels achieved are better than those of Masindi et al. (2022), while the  $\text{PO}_4\text{-P}$  levels achieved are comparable with Masindi et al. (2022) and meet the SANS criteria for wastewater effluent.



**Figure 6.6:** WSP Scenario 1: Relationship between the influent flow rate and the effluent  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations for (a) steady state, and (b) dynamic state flow conditions

Given that the dynamic state flow conditions reflect the actual variable flow rate scenarios owing to fluctuations in pressure, only dynamic state simulations were conducted for Scenarios 2 to 4.

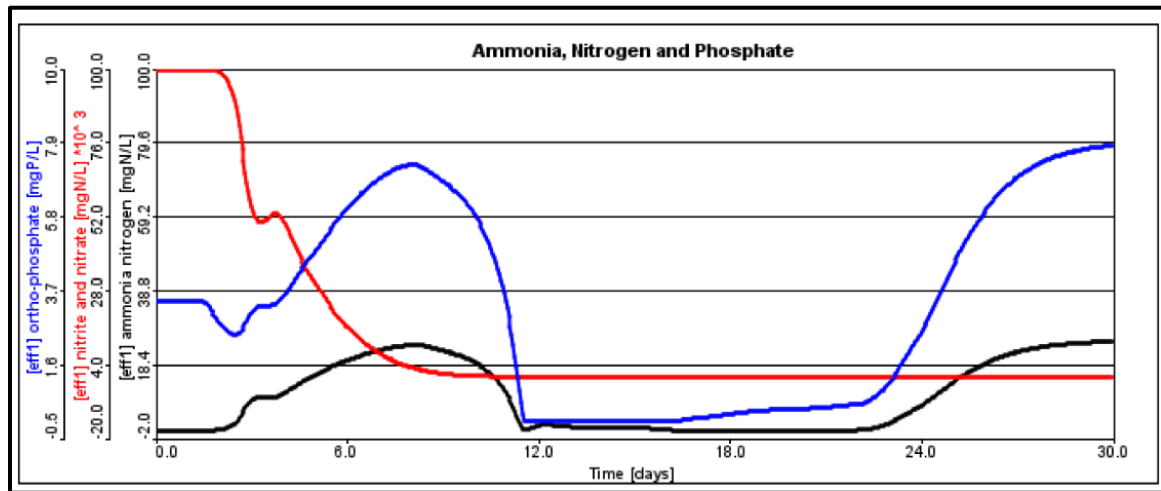
**WSP Scenario 2:** The relationships between the influent flow rate and the effluent  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$  and  $\text{PO}_4\text{-P}$  concentrations under dynamic state flow conditions are shown in Figure 6.7.



**Figure 6.7:** WSP Scenario 2: Relationship between the influent flow rate and the effluent  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations for dynamic state flow conditions

In Figure 6.7, the  $\text{NH}_3\text{-N}$  levels are higher than those in Scenario 1 and exceed the SANS criteria, with  $\text{NH}_3\text{-N}$  levels up to 24 mg/l, which is four times higher than the recommended SANS criteria. Apart from not meeting the required  $\text{NH}_3\text{-N}$  levels, low nitrogen and phosphate levels were maintained, which are comparable to those levels achieved by Lipponen and Nikiforova (2017) and Mamera et al. (2022). Given the identified shortcomings in Scenario 2, it was necessary to investigate the performance of Scenario 3 as shown in Figure 6.8.

**WSP Scenario 3:** In Figure 6.8, it is evident that the  $\text{NH}_3\text{-N}$  levels ranged between  $-6.68 \text{ mg/l}$  and  $19.3 \text{ mg/l}$ , while the  $\text{NO}_3\text{-N}$  levels fluctuated between  $3.8 \text{ mg/l}$  and  $100 \text{ mg/l}$ . The negative value for  $\text{NH}_3\text{-N}$  is ascribed to the levels being too low to detect at the onset of simulation. Nevertheless, the maximum levels of both  $\text{NH}_3\text{-N}$  and  $\text{NO}_3\text{-N}$  still exceed the recommended SANS limits. The  $\text{PO}_4\text{-P}$  levels ranged between  $3.5 \text{ mg/l}$  and  $7.9 \text{ mg/l}$  and meet the SANS criteria.

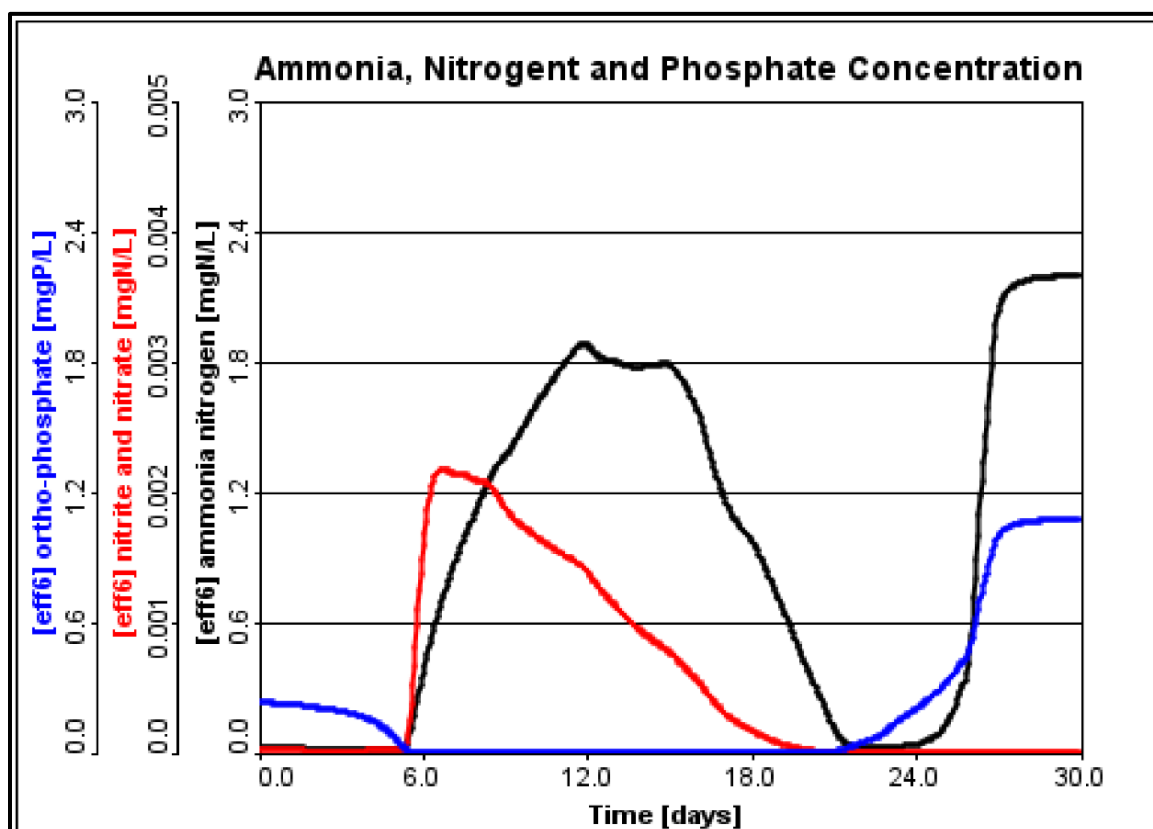


**Figure 6.8:** WSP Scenario 3: Relationship between the influent flow rate and the effluent  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations for dynamic state flow conditions

**WSP Scenario 4:** It is evident from Figure 6.9 that the  $\text{NH}_3\text{-N}$  ( $0 - 1.9 \text{ mg/l}$ ),  $\text{NO}_3\text{-N}$  ( $0 - 0.0021 \text{ mg/l}$ ), and  $\text{PO}_4\text{-P}$  ( $0 - 0.68 \text{ mg/l}$ ) levels meet the recommended SANS criteria and have outperformed Scenarios 1, 2, and 3 regarding nutrient removal. Therefore, the wastewater effluents originating from Scenario 4 were compared with the effluent results obtained from the four WWTPs for nutrient removal as presented in Table 6.2.

It is evident from Table 6.2 that the average concentrations for  $\text{NO}_3\text{-N}$  and  $\text{PO}_4\text{-P}$  at the four WWTPs are below the recommended SANS limits, while the  $\text{NH}_3\text{-N}$  concentrations are exceeded significantly in all cases. In comparing the wastewater effluent from the WWTPs with the WSP Scenario 4 ranges, it is evident that the  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations are significantly higher with a factor ranging from 7.2 to 10.5 ( $\text{NH}_3\text{-N}$ ), up to 1 191 ( $\text{NO}_3\text{-N}$ ), and between 2.1 and 5.3 ( $\text{PO}_4\text{-P}$ ), respectively. Hence, the re-engineering of a WSP is identified

as a progressive approach to tackle the inefficiencies in treating wastewater at the WWTPs in the MMM.



**Figure 6.9:** WSP Scenario 4: Relationship between the influent flow rate and the effluent  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations for dynamic state flow conditions

**Table 6.2:** Average  $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$  concentrations ( $\text{mg}/\ell$ ) at the four WWTPs (2021 – 2023) in comparison to the WSP Scenario 4 concentration ranges

WWTP	WWTP Avg. $\text{NH}_3\text{-N}$	WSP range $\text{NH}_3\text{-N}$	WWTP Avg. $\text{NO}_3\text{-N}$	WSP range $\text{NO}_3\text{-N}$	WWTP Avg. $\text{PO}_4\text{-N}$	WSP range $\text{PO}_4\text{-N}$
Bainsvlei	19.9		1.9		2.5	
Botshabelo	15.3	0 – 1.9	2.5	0 – 0.0021	3.3	0 – 0.68
North-East	13.6		2.5		1.4	
Sterkwater	16.2		2.3		3.6	
<b>SANS criteria</b>	<b>&lt; 6</b>		<b>&lt; 15</b>		<b>&lt; 10</b>	
			<b>Exceeding SANS Limits</b>			

Additionally, the significant differences between the WSP simulated and WWTP effluents highlight the inefficiency of conventional WWTPs to remove ammonia, a concern which was also noted by Ashkanani et al. (2019). Typically, such

ammonia levels could lead to the death of aquatic species, pose environmental risks, and raise public health concerns (Zhang et al., 2018; Powders et al., 2025). Powders et al. (2025) further highlighted that GHG emissions such as nitrous oxide emissions are linked to ammonia oxidation from wastewater effluent. Therefore, it is vital to incorporate the simulation findings into real-world solutions to reduce the ammonia concentration in effluents from WWTPs by intentionally including WSPs as secondary methods.

### 6.3 Conclusions

This chapter explored the design and simulation of four configurations of re-engineered WSPs and their ability to remove solids and nutrients. Both steady and dynamic state flow conditions were evaluated. Given the plausible results obtained using the dynamic flow simulations, various design configurations (Scenarios 2 to 4) were investigated to establish which scenario is the most effective in terms of solids (TSS and VSS/TSS ratio), and organics and nutrient ( $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$ ) removal. Scenario 3 proved to be the most effective in terms of solid/sludge removal, i.e., higher TSS values and VSS/TSS ratios  $\geq 0.8$  were achieved. In terms of organics and nutrient removal, Scenario 4 performed the best and was subsequently compared to the wastewater effluent discharged from the four conventional WWTPs in the MMM. Furthermore, the findings provide a site-specific calibration and modelling approach that sets a precedent for other South African municipalities. It also demonstrates the contextual applicability of GPS-X in semi-arid sub-tropical conditions. As there is no or limited literature relating to sub-Saharan Africa, this research addressed both a geographical and technical gap in the scientific and technical literature. However, the findings from this study will require validation by future studies.

A synthesis of all the information as discussed in Chapters 1 to 6, as well as some final conclusions and recommendations are included in the next chapter.

## CHAPTER 7 : CONCLUSIONS AND RECOMMENDATIONS

A synthesis of all the information as discussed in Chapters 1 to 6, as well as some final conclusions and recommendations is included in this chapter.

### 7.1 Research Aim

The research aim was to re-engineer existing WSPs for improved contaminant removal efficiency by providing a detailed solution for the interplay among the hydraulic, chemical, biochemical, and mechanical processes involved. Achieving the research aim enabled a better understanding of the hydrodynamics and biochemical processes of a WSP.

### 7.2 Specific Objectives and Major Findings

The specific objectives (SOs) were addressed successfully and ensured that the overall research aim was achieved. The details of the specific objectives and major findings associated with each SOs are listed and discussed below:

- (a) **SO 1:** Establish the influence of influent and effluent physicochemical parameters in diagnosing and addressing operational challenges at WWTPs; and
- (b) **SO 2:** Develop a cost-effective approach to sample and determine the various physicochemical parameters.

In Chapter 4, SO 1 and 2 were achieved by adopting a cost-effective approach to sample the influent and effluent at four WWTPs over a three-year period in the MMM. Only COD was used as an indicator of microorganisms (*E. coli* count) in the sampled wastewater. Given that biological tests to determine the *E. coli* count are normally time consuming and costly, considering COD as the primary indicator of pathogenic bacteria proved to be effective. The results also confirmed that all WWTPs in the MMM failed to meet the recommended SANS COD = 75 mg/ℓ.

Several factors were identified as contributors for not meeting the required standard, e.g., ranging from a lack of technical expertise to a lack of funding. Further investigations towards the properties of the sampled influent and effluent in

terms BOD<sub>cal</sub>, pH, temperature, and OA showed that OA significantly impacts the organic concentration in the wastewater effluent. Statistical analyses were deemed necessary given that no clear trend(s) were evident between BOD<sub>cal</sub>, pH, and temperature. The BOD/COD ratios also confirmed that the COD is approximately twice the BOD<sub>cal</sub> as per convention and recommended in literature.

In accordance with SO 1, the potential impact of ammonia, nitrogen, and phosphate was also investigated. The results revealed that the ammonia levels were higher than the SANS effluent standards. However, the nitrogen and phosphate concentrations were within the national standards. Given that the levels of nitrogen and phosphorus were regarded as insufficiently low, concerns were raised about their impact on plants and aquatic species requiring a certain level of nutrients.

- (c) **SO 3:** Investigate and compare the effluent quality compliance at the four WWTPs with the SANS criteria for wastewater effluent; and
- (d) **SO 4:** Investigate the potential environmental and health impacts of the wastewater effluent discharged into surface water bodies within the MMM.

In Chapter 5, both SO 3 and SO 4 were achieved. The elevated concentrations of COD found in the wastewater effluent raised concerns due to non-conforming to the SANS criteria. A correlation matrix inclusive of variance and Pearson correlation between the various physicochemical wastewater effluent parameters revealed high statistical differences associated with either positive or negative correlations, or both. The elevated pH levels, fluctuating EC, and TDS found in the sampled surface waters present potential hazards to the well-being of the general population. Although no direct link with the 2023 cholera outbreak could be established, the results agreed with the findings of other literature and highlighted that the prolonged use of water from the Renoster Spruit, Bloem Spruit, and Klein Modder River could lead to kidney and cardiovascular complications due to the high TDS concentrations.

In general, the physical access to the WWTPs and SW sampling sites and relevant data sets, along with the associated cumbersome administrative processes to

grant such access, is regarded as problematic and has delayed the progress and completion of this research. Apart from the above concerns, it was also evident that sporadic power failures, financial constraints, a shortage of adequately skilled personnel, and inadequate upkeep protocols have a negative impact on the functionality, operation, and management of WWTPs in the MMM. Hence, these critical matters necessitate immediate consideration and resolution to protect the well-being and interests of the residents of the MMM in the FS province.

- (e) **SO 5:** Explore sludge distribution in a WSP system by considering different baffle configurations, flow rates, and/or pond geometry; and
- (f) **SO 6:** Explore the design and simulation of a re-engineered WSP for improved contaminant removal by comparing results with effluent originating from the four WWTPs.

In Chapter 6, both SO 5 and SO 6 were achieved by exploring the design and simulation of four configurations of re-engineered WSPs and their ability to remove solids and nutrients. Both steady and dynamic state flow conditions were evaluated. Given the plausible results obtained using the dynamic flow simulations, various design configurations (Scenarios 2 to 4) were investigated to establish which scenario is the most effective in terms of solids (TSS and VSS/TSS ratio), and organics and nutrient ( $\text{NH}_3\text{-N}$ ,  $\text{NO}_3\text{-N}$ , and  $\text{PO}_4\text{-P}$ ) removal. Scenario 3 proved to be the most effective in terms of solid and sludge removal, i.e., higher TSS values and VSS/TSS ratios  $\geq 0.8$  were achieved. In terms of organics and nutrient removal, Scenario 4 performed the best and was subsequently compared to the wastewater effluent discharged from the four conventional WWTPs in the MMM.

### 7.3 Contributions to New Knowledge

The design and simulation of the re-engineered WSPs and the subsequent findings are regarded as contributions to new knowledge. Typically, the re-engineered WSPs were simulated and integrated as pre- or post-treatment, or both, at either conventional or advanced WWTPs.

It is envisaged that the research findings will advance the scientific and engineering understanding of WSP design and optimisation. The findings build directly upon two previously published articles (*cf.* Declaration: Publications). The research stems from these findings and details the downstream effects of underperforming WWTPs on riverine health and water quality, identifying urgent gaps in solids, nutrient, and organic load removal. However, while Chapters 4 and 5 focused on diagnosing performance shortcomings and highlighting ecological risks, the utilisation of the Hydromantis GPS-X transitions from impact assessment to proactive engineering optimisation, thus contributing significantly to new knowledge and practice in WSP management.

The newly developed simulation framework adopted an empirical modelling approach to incorporate the fundamental equations in Chapter 2. For example, Equation 2.1 (temperature dependency) specifically captures seasonal fluctuations in the microbiological and chemical WSP kinetics, while Equation 2.11 (advection-dispersion) measures the solute transport and flow dynamics throughout the pond system. Hence, the developed simulation framework enabled the translation of theoretical formulations into applied simulations by using the Hydromantis GPS-X, which is especially important in the understudied setting of sub-Saharan municipal wastewater systems.

Given that WSP Scenario 3 proved to be the most effective in terms of solid and sludge removal, and WSP Scenario 4 the most effective in terms of organics and nutrient removal, a hybrid technique that incorporates the advantages of both scenarios is suggested to acknowledge the limitations of single-objective optimisations. In contrast to the findings of this research (*cf.* Chapters 4 and 5), which examines performance indicators separately, this multi-scenario synthesis is a revolutionary approach in WSP engineering.

The following guidelines should be considered when employing the re-engineered WSP configurations for the efficient removal of contaminants:

- (a) **Hydraulic retention time (HRT):** The simulation results (*cf.* Chapter 6) indicated that a minimum HRT of 24 days is required in Scenario 3 to

- achieve efficient solids removal, whereas for Scenario 4, significant nutrient reductions were achieved within 22 days. These retention benchmarks provide specific targets for plant designers working in similar semi-arid, temperature-variable environments.
- (b) **Seasonal performance optimisation:** The essence of the temperature dependency model (Equation 2.1) was discussed and highlighted in this research. The impact of meteorological parameters was not considered in the simulation owing to the software license restrictions applicable to the Hydromantis GPS-X. However, if meteorological parameters were to be incorporated in the simulations, it would result in a reduction of treatment efficiency in winter. To mitigate this, thermal buffering zones and flexible baffling (*cf.* Chapter 3) are recommended to reduce stratification and preserve microbial activity.
- (c) **Sludge distribution management:** The effects of inlet/outlet orientation, flow regime, and geometry at the Botshabelo WSP were highlighted in this research. Unfortunately, the model does not cater for sludge distribution patterns for WSPs. However, the use of staggered inlets, increased flow path lengths, and compartmentalisation (*cf.* Chapters 3 and 6) to limit dead zones and ensure more uniform sludge dispersion, are recommended.
- (d) **Surface volume-area (V:A) ratio:** Although findings showed that the treatment efficiency of WSPs improve when the V:A ratio is optimised to prevent short-circuiting and enhance retention, the surface area of the existing Botshabelo WSP (*cf.* Chapter 3) with a maximum allowable depth up to 3 m was adopted. Furthermore, by implementing WSP Scenarios 3 and 4, the effective treatment zones can be maximised to improve solids and nutrient removal efficiencies.
- (e) **Bridging impact studies and engineering solutions:** By embarking on a simulation-based study (*cf.* Chapter 6) owing to identified treatment failures (*cf.* Chapters 2, 4, and 5), the research also offered additional insights, i.e., from diagnosing pollution sources to the re-engineering of WSPs. This progression from impact assessment to design reform is not only methodologically innovative, but also regionally urgent, given ongoing compliance challenges in many South African municipalities.

Therefore, in summary, the contributions to new WSP knowledge include the following:

- (a) translating theory into site-specific solutions via simulations;
- (b) introducing hybrid scenario optimisation to reconcile trade-offs in treatment goals;
- (c) establishing actionable hydraulic and geometric design thresholds; and
- (d) expanding the engineering toolkit for WSP design in semi-arid, resource-limited regions.

These contributions are of particular importance for municipalities facing structural and operational problems, thereby setting a precedent for performance-based, data-driven wastewater infrastructure planning in sub-Saharan Africa and beyond.

#### **7.4 Recommendations**

Based on the literature reviewed and the results obtained from this research, the following aspects are recommended, some of which need to be considered during future research:

- (a) The best performing WSP design should be incorporated into existing WWTPs in the FS province and across South Africa for efficient contaminant removal. In small towns, the re-engineered WSP can be used as a standalone system for treating wastewater.
- (b) Nutrient removal should be integrated as a sustainable approach to mitigate potential environmental concerns such as algal blooms and as a low-cost alternative to produce nitrogen and phosphorus.
- (c) Managers should encourage proactive monitoring and taking adequate steps to prevent surface water pollution.
- (d) Further research in deploying advanced CFD modelling is necessary to better understand sludge interaction with pond geometry.
- (e) More research is required on trapping greenhouse gas (methane) from WSPs as an alternative energy source.
- (f) The inclusion of municipal-level strategies, cost–benefit considerations, and scalability to other regions in South Africa.

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