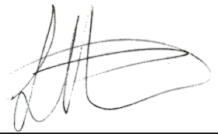
DECLARATION

I declare that this thesis presented for the degree of Master of Science in Engineering (Chemical), has

i. been composed solely by myself,
ii. been solely the result of my work,
iii. not been submitted for any other degree or professional qualification.



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6th day of August 2019 at WITS UNIVERSITY

ABSTRACT

Safe access to safe drinking water is an issue is many developing countries where water treatment service providers are not able to effectively treat water sources impacted by sewage and other pollutants. This research focuses on the quality of urban river water, as well as on the design and use of a small-scale water treatment technology that is aimed at assisting communities lacking access to safe drinking water. The Portable Potable Water Treatment System (PPWTS) is a small-scale system made up of three chambers, namely a constructed wetland (CW), a chamber containing charcoal and a UV chamber utilising UV LEDs for disinfection. River water from the Fairlands Spruit River was passed through the system in an attempt to treat the water to the potable water standards set by the World Health Organization (WHO) and the South African National Standards. The average pH, E.C., turbidity, phosphate, nitrate, sulfate, COD, TOC, total coliform, *E. coli*, aluminium, lead and manganese values obtained through analysis and testing of the feed water were 8.40, 292 µS/cm, 3.27 NTU, 0.052 mg/L, 2.93 mg/L, 27.8 mg/L, 19.5 mg/L, 6.71 mg/L, 56800 MPN/100 mL, 22900 MPN/100 mL, 55.8 µg/L, 2.7 µg/L and 24.91 µg/L respectively. The results obtained through using the PPWTS to treat river water revealed an increase in the average pH of river water to 8.77, a 28.53% increase in E.C., a 36.09% decrease in turbidity, a 92.31% increase of phosphates, a 55.63% removal of nitrates, a 23.51% removal of COD, a decrease in TOC of 16.10%, a total coliform removal rate of 64.86%, a decrease in *E. coli* bacteria of 91.94% and a 34.96% decrease in sulfates. The removal rates for aluminium, lead and manganese achieved using the PPWTS were 47.69%, 10% and 48.37% respectively. Though substantial removal of total coliforms and *E. coli* was achieved, the water collected from the outlet of the PPWTS did not meet the SANS and WHO drinking water standard of 0 MPN/100 mL for total coliforms and *E. coli*. The design of the system could do away with the UV chamber as it is was not effective in removing total coliforms and *E. coli* bacteria.

ACKNOWLEDGEMENTS

This research, conducted at the University of the Witwatersrand was funded by the Centre in Water Research and Development (CIWaRD) in 2016, the National Research Foundation (NRF) Innovation Master's Scholarship in 2017 and Britehouse, a division of Dimension Data in 2018.

I am ever grateful to my mother and father for their continued love and support. I could never fully express how grateful I am for your presence in my life. I dedicate this master's degree to you two, and I hope in my heart that I have made you proud. I love you both!

My sincere and deep gratitude goes to my research supervisor Professor Craig Sheridan for having afforded me the opportunity to pursue this master's degree under his supervision. I am extremely grateful to him for his kindness, openness and willingness to help me through this degree. I would also like to thank Professor Sheridan for all the life advice and guidance he has given me, and for helping me believe in myself and my abilities. Working with him has been both a pleasure and an inspiration. Thank you, Professor, I will never forget you.

A very special Thank You to Mrs. Janet Smith who was always prepared to help me when I needed help with my research. I can safely say that I would not have come as far as I have without you help. I am grateful that you joined us at the WITS School of Chemical and Metallurgical Engineering when you did.

I would like to thank the members of the Industrial and Mining Water Research Unit (IMWaRU) for availing themselves when I needed assistance and for the great social sessions we enjoyed together. It has been an honour to know you all.

I would also take this opportunity to acknowledge and thank Britehouse, a division of Dimension Data for their understanding and cooperation with regards to this master's degree. I would also thank Britehouse for their contribution towards funding for this project in 2018.

Finally, I would like to acknowledge the gentlemen operating the Richard Ward, School of Chemical and Metallurgical Engineering Workshop for their assistance in building my water treatment system. Rodney, Phathu and Hopewell, I thank you all.

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LIST OF ABBREVIATIONS

ABC	Animal Bone Charcoal
AMD	Acid Mine Drainage
AN	Ammonium Nitrate
BOD	Biochemical Oxygen Demand
CBC	Cow Bone Charcoal
CFU	Colony-Forming Units
COD	Chemical Oxygen Demand
CW	Constructed Wetland
CWF	Ceramic Water Filter
DBP	Disinfection By-Products
DWA	Department of Water Affairs
E.C.	Electrical Conductivity
FWS	Free Water Surface
GAC	Granular Activated Carbon
HF	Horizontal-Flow
HRT	Hydraulic Retention Time
HSSF	Horizontal Sub-Surface Flow
ICP-MS	Inductively Coupled Plasma Mass Spectrometry
LED	Light Emitting Diode
MF	Microfiltration
NF	Nanofiltration
PCB	Printed Circuit Board
PET	Polyethylene Terephthalate
PAC	Powdered Activated Carbon

PPWTS	Portable Potable Water Treatment System
RO	Reverse Osmosis
SANS	South African National Standards
SODIS	Solar Water Disinfection
SS	Suspended Solids
TN	Total Nitrogen
TOC	Total Organic Carbon
TP	Total Phosphorus
TSS	Total Suspended Solids
UF	Ultrafiltration
US EPA	United States Environmental Protection Agency
UV	Ultraviolet
VF	Vertical-Flow
WHO	World Health Organization
WSP	Water Services Providers
WWT	Wastewater Treatment
WWTP	Wastewater Treatment Plant

CHAPTER 1 – INTRODUCTION

1.1 BACKGROUND

Currently, South Africa is experiencing the worst drought in 30 years (BBC News, 2015). It is said that the country received its lowest rainfall in the year 2015, since the year 1904 (Groenewald, 2016). South Africa is a water scarce country and is classified as one of the 30 driest countries in the world. The average annual rainfall in South Africa is less than 500 mm while the world average is roughly 850 mm (Government of South Africa, 2015). The drought South Africa is currently experiencing is said to be a result of the El Niño phenomenon. The occurrence of the El Niño climate cycle brought about an abnormal heat wave which increased water evaporation rates in the country, applying further pressure on the limited water resources. The average international water usage a day is 173 litres while South African citizens use 61.8 % more than this daily amount (Government Communications, 2015). In parts of the country, the drought is causing substantial water accessibility problems as communities that rely on collecting rainwater to access fresh drinking water are being driven to use water sources impacted by sewage and other impurities in the vicinity. There have been reports about a community in Senekal, a town in the Free State, drinking water contaminated with sewage as a result of the water scarcity crisis (Singh, 2016). The Senekal community is one that is lacking in water availability and so during this time of drought, the members of this community have been required to wait in line overnight for trucks carrying three 5000 litre water tanks to deliver water to the community. Although the delivery of water to Senekal provides some relief to the community, the amount of water delivered is not enough to sustain the whole community. Every household in the community is only provided with 50 litres of water a day (Chabalala, 2016).

Water pollution is also impacting the available water resources. Water pollution occurs through a number of different mechanisms: urbanisation, industry, mining and agriculture. Urbanisation involves an increase in population as people move into cities and urban areas. An increase in population size results in an increased demand for buildings and homes; increased activity from industries and mines; more intensive sewage collection and treatment facilities; and increased need for growing more crops for a larger population (Rand Water, 2016). These activities all result in additional pollution and place a greater pressure on the available water resources. Urbanisation can lead to the inadequate collection and treatment of sewage which is a major issue in South Africa. In Sakhile, a township near Standerton, the residents have been

living without sewage treatment since 2009. The residents have approached the Lekwa municipality about developing the sewerage infrastructure but have had very little success. As a result, the untreated sewage flows, from houses, onto the streets. In another area of Sakhile, the sewage ends up untreated in the Vaal River that supplies Gauteng with water. The Vaal River flows directly through Standerton and Sakhile township and then into the Vaal Dam at Vereeniging. Vaal River water in Standerton was taken to an SANAS accredited laboratory and the results revealed that the *E. coli* count in the river exceeds 1000 units per 100 millilitre (Batt, 2015). This essentially that means the Vaal Dam is being charged with sewage impacted water as a result of an old and non-functional sewerage infrastructure (Sama Yende, 2016). Mining and Industrial activities are known to produce waste that negatively impact water in a number of ways. Mining and Industrial waste can alter the pH of water, change the colour of water and increase the amount of heavy metals and nutrients in water among other things. Agriculture has the effect of increasing the nutrient content of water as a result of fertiliser use as well as animal excrement. This in turn increases the concentration of nitrates and phosphates in water which can result in eutrophication (Rand Water, 2016).

The rationale of this project is to develop an inexpensive, convenient, user-friendly treatment technology for people who are affected by sewage/industrially impacted drinking water. The aim of this research is to provide a design of a small, portable system that is not only simple and cheap to use, but also inexpensive to make (Leppert & Pepper, 2012). The design of this portable water purifier is also aimed at being environmentally friendly, providing a way of purifying sewage impacted water without causing any further impact to the environment through the use of chemicals. Providing a portable water purification system of this kind would not only provide relief to areas experiencing sewage impacted water, it would also give people the tools, and ultimately the power, to reuse impacted water without having to depend heavily on the municipality and other water purification parastatals to provide drinkable water.

1.2 PROBLEM STATEMENT

The water sources available in rural communities are often impacted as a result of sewage contamination of these water sources, with little to no application of water treatment processes. This results in water that is unsafe for human consumption. Rural communities in situations of this kind do not have access to efficient water treatment systems as a result of various political and socio-economic factors. People in these communities, however, have to make use of the water sources available to them and have very few options as to where they can collect water for use and consumption. Individuals in rural communities make use of the sewage impacted water sources available to them as they are the closest, cheapest and most convenient sources of water.

1.3 AIMS AND OBJECTIVES

This research project aims to answer the following questions.

1.3.1 What are the levels of the impurities in sewage impacted urban river water?

1.3.2 Can the use of a novel three-stage constructed wetland (CW)/charcoal/Ultraviolet (UV) system be used to treat sewage impacted water to potable water standards?

The objectives of this research project are:

1.3.3 To investigate and determine the quality of water in sewage impacted urban rivers.

1.3.4 To determine if the use of a novel CW/charcoal/UV system is viable in the treatment of sewage impacted water sources to potable water standards.

1.4 STRUCTURE OF THESIS

There are 7 Chapters that make up this thesis.

Chapter 1 presents an introduction to the problem being investigated. The research Background, Problem Statement and Aims and Objectives are given in this chapter.

Chapter 2 is an overview of the literature concerning conventional water treatment systems, constructed wetlands and small-scale water treatment systems.

Chapter 3 provides the methods that were used to prepare the samples for testing, analytical techniques used to test for each water quality parameter and how the PPWTS was designed and built.

The results and discussion thereof are presented in Chapter 4. These results are also compared to the SANS and WHO potable water standards.

The conclusions and recommendations are provided in Chapter 5.

The references and appendices sections are presented after Chapter 5.

CHAPTER 2 – LITERATURE REVIEW

2.1 WASTEWATER TREATMENT TECHNOLOGIES AND ALTERNATIVES

The main objective of wastewater treatment (WWT) is to allow for the disposal of sewage and industrial effluent without negatively impacting human health and the environment (Srivastava, 2016). In the past, natural purification systems were used to clean wastewater. Sewage was disposed in waterways where bacteria and other organisms present consumed it and other organic matter in the wastewater. Today, as a result of increased population sizes and the increased volume of both domestic and industrial wastewater, WWT technologies are being used in order to emulate and speed up the natural water purification process (U.S. EPA, 1998). Although conventional WWT processes are most commonly used to treat wastewater, alternatives to these mainstream processes are being looked into. Examples include CWs (American Society of Landscape Architects, 2010) and small-scale purification systems aimed at addressing the lack of potable water in remote and rural areas. A recent invention known as the "Pure" Water Bottle is a simple invention that utilises UV sterilisation to treat impure water to potable standards. (Heimbuch, 2010).

2.2 WASTEWATER TREATMENT PLANTS

The purpose of wastewater treatment plants (WWTP) is to remove impurities from wastewater before releasing it into local waterways (U.S. EPA, 1998). Conventional wastewater treatment processes involve a number of steps, namely: Preliminary Treatment, Primary Treatment, Secondary Treatment, Tertiary (or Advanced) Treatment and Disinfection. Preliminary treatment involves the removal of solids using screens or grit chambers. This step is aimed at preventing the clogging of pipes and pumps in the system. Primary treatment is carried out in a settling tank or clarifier where most of the floating and settling solids are removed. Secondary treatment is made up of an artificial environment that is rigorously controlled to allow for microscopic organisms to consume any waste that is still in the wastewater. These dissolved waste solids are then converted into suspended solids that can be physically removed by settling. Tertiary treatment is a step used to further enhance the quality of the water. This step involves the removal of suspended solids and nutrients but may also involve the removal of specific toxic substances. Disinfection is the last step before releasing treated water back into the environment. This step substantially reduces any remaining viruses and bacteria (The Macon Water Authority, 2011). Common tertiary treatment steps include chlorination, UV

disinfection and ozonation (University of North Dakota Energy & Environmental Research Center, 2016).

In 1956 South Africa made it compulsory that effluent be treated to acceptable water standards and then returned to the water ways from which it was originally obtained through the South African Water Act (Act 54 of 1956) (Morrison et al., 2001). As a result of population and economic growth, the demand for water increased, thus applying increased stress on the existing WWTPs (Mema, 2010). In South Africa today, most rural communities are experiencing water service issues as a result of various socio-economic and political factors. However, the Department of Water Affairs (DWA) has entrusted the Water Service Authorities (WSA) and the Water Services Providers (WSP) with the duty to ensure that every individual in the country, including those in poor communities who cannot afford water services, have free access to a basic level of water service (Mothetha et al., 2013).

In an attempt to motivate municipalities and water service providers to provide high quality water services, South Africa adopted an incentive-based regulation concept that is used to identify, reward, ensure and inspire excellent wastewater management practices. The concept was launched on the 11th of September 2008 and is made up of two programmes, namely the

Blue Drop Certification Programme and the Green Drop Certification Programme. The Blue Drop Certification Programme is aimed at managing the quality of drinking water while the Green Drop Certification Programme exists to manage the quality of wastewater treatment systems. The Blue and Green Drop processes consider the performance results of Water Service Authorities and their providers and in turn, rewards or penalises municipalities based on the evidence of their excellence or failures according to the minimum requirements that have been set (Department of Water Affairs, 2015). Ultimately, each province receives a provincial Blue and Green Drop score reflecting its performance relative to the set standards. According to the Department of Water and Sanitation (2014) the provinces with the highest performance in adhering to Blue and Greed Drop requirements for 2014 were the Western Cape (WC), KwaZulu Natal (KZN) and Gauteng (GP). The Green Drop provincial score is an indication of the risk associated with wastewater treatment systems in a particular province. High provincial Green Drop percentage scores are indicative of poor wastewater treatment systems and facilities while low Green Drop percentage scores indicate effective wastewater treatment. The Green Drop scores obtained by the WC, KZN and GP in 2014 were 57.7%, 66.8% and 66.8% respectively. The bottom three provinces were the Free State (FS) (83.3%),

Mpumalanga (MP) (84.2%) and the North West Province (NW) (86.1%) respectively. The Northern Cape (NC), Eastern Cape (EC) and Limpopo (LP) provinces achieved Green Drop scores of 71.5%, 72.0% and 80.2% respectively. (Department of Water and Sanitation, 2014). These figures can be attributed to the fact that the WC, KZN and GP are the three major provinces in the country. These provinces harbour the highest number of skilled individuals in the country and are thus able to produce high quality water services.

The conventional WWT process is effective in treating wastewater; however, WWTPs are known to negatively impact fresh water sources (Msagati et al., 2015). A study conducted by Ngwenya (2006) on the Eerste River in the WC revealed increasing trends in the electrical conductivity (E.C.) of the river water as well as increases in the nitrate, nitrite and phosphorus concentrations. The Stellenbosch Water Treatment Works is listed, in the study, as one of the contributing factors to the deteriorating water quality in the Eerste River. On the 22nd of August 2005, the town of Delmas in the Mpumalanga Province (MP) was struck by an outbreak of Typhoid. According to the former minister of Water Affairs and Forestry Buyelwa Sonjica, the outbreak was caused by an overloaded sewage system. The overload on the system caused the seepage of sewage into the water sources that the people of Delmas were using. On the 13th of September 2005, statistics revealed that 287 cases of typhoid had been reported while 1406 people suffered from diarrhoea (Motloung, 2005). On the 23rd of September 2005, it was reported that four people had died from typhoid since the outbreak on August the 22nd (Graham, 2005).

WWT is a global problem, however, due to a lack of funding; the high costs associated with conventional wastewater treatment and the progressive increase in wastewater generation, developing countries are unable to ensure that a large percentage of the wastewater generated is correctly and adequately treated (Msagati et al., 2015). Treatment of water also involves treating select water bodies to drinkable standards. Such processes are conducted by potable water treatment plants.

2.3 POTABLE WATER TREATMENT AND WATER BOARDS

The function of Water Boards in South Africa is to distribute raw and potable water across the country. This is a role managed and controlled by the minister of the Department of Water Affairs. Currently South Africa has 9 Water Boards. These are Amatola (EC), Bloem Water (FS), Lepelle North (LP), Magalies (North West Gauteng), Mhlathuze (KZN), Overberg (WC),

Rand Water (GP and MP), Sedibeng Water (FS and North West) and Umgeni Water (KZN). In total, the water boards supply the country with approximately 2.56 billion m^3/annum of bulk drinking water. The water boards together supply potable water to 28 million people (DWS Directorate: Water Macro Planning, 2015). The two largest water boards in the country, Rand Water and Umgeni water supply potable water to 10 million (Ramsden, 2014) and 4 million people respectively (Umgeni Water, 2016), approximately half of the total supplied to by the water boards together.

Rand Water (the largest water authority board in South Africa) draws its water from the Vaal Dam by way of a canal and a gravity pipeline and by pumping water from the Vaal River Barrage Reservoir at Lethabo power station, Zuikerbosch pump station and Vereeniging. A small amount of underground water is also drawn from Zuurbekom. The purification process used to treat the water to potable standards involves seven steps; Coagulation, Flocculation, Sedimentation, Stabilisation, Filtration, Disinfection and Chloramination. Coagulation involves the addition of coagulant (sodium silicate and slaked lime) and the rapid mixing of the water and coagulant to destabilise the particles in the wastewater. Flocculation is the clumping together of the suspended particles destabilised through coagulation. The heavy visible particles, called flock, remain suspended as the water flows at high velocities through spiral flocculates or baffled channel conditioning bays. In the Rand Water sedimentation process, chemical coagulants are added to produce flocs that settle in tanks designed to facilitate sedimentation as well as remove sludge. As a result of using slaked lime as a coagulant, the pH of the water is raised to approximately 10.5 which is conducive to scale formation and corrosion. During stabilisation water is passed through carbonation bays where carbon dioxide is bubbled through the water, reducing the pH of the water to a value of between 8.0 to 8.4. This pH is more stable and prevents scaling and corrosion in the system. The filtration step removes the remaining suspended particles as water is passed through rapid gravity sand filter beds made of finely graded silica sand and pebbles. Disinfection of the water leaving the purification works is achieved using chlorine. Chlorine is used to ensure that any micro-organisms, bacteria or viruses present in the water are killed. The chlorine dosage is between 1.5 to 4.0 mg/L depending on the quality of the raw water being disinfected. This is done to achieve minimum re-growth of micro-organisms during the 6 to 8 hours it takes for the water to make its way to the booster pumping station. There are no chlorine disinfection chambers in the process and so disinfection occurs in the pipelines. A secondary disinfection step is applied through chloramination as chlorine remains active for only 6 to 8 hours. A less

powerful yet longer acting agent is administered at the booster pumping station. This agent is formulated by dosing chlorine and ammonia in a mass ratio of no less than 4:1. This ratio forms the monochloramine which protects water against bacterial re-growth for up to 8 days. The monochloramine protects the water until it gets to the end consumer (Rand Water, 2016).

As the Green Drop Certification Programme is aimed at ensuring high quality WWT plant effluent quality, the Blue Drop Certification Programme is aimed at safeguarding the quality of tap water management as clean water greatly determines the livelihood of humankind. As with the Green Drop Provincial scores, the leading provinces with regard to meeting Blue Drop requirements for 2014 were GP (92%), the WC (89%) and KZN (86%) respectively. The provinces with the lowest Blue Drop scores were the NC (68%), NW (63%) and LP (62%) respectively. The FS, EC and MP achieved Blue Drop scores of 75%, 72% and 69% respectively (Department of Water and Sanitation, 2014). This can again be attributed to the level and availability of skills in the major provinces that other provinces lack. The WC, GP and KZN provinces are also highly urbanised compared to the others and thus harbour individuals who can comfortably pay for water services, unlike in less urbanised areas (Msagati et al., 2015). There are small scale systems that have the potential to address the issues surrounding access to safe drinking water in less urbanised areas. Various water disinfection and purification techniques, such as UV disinfection and heavy metal adsorption using activated carbon, can be used together to create water purification systems on a smaller scale, for individuals, families and communities to use. Other small-scale systems can be made up of one type of purification technique to purify water. One such technique is the Sari filtration method.

2.4 SMALL SCALE SYSTEMS

2.4.1 Small Scale Systems and the Sari Filtration Method

Various small-scale water treatment systems have been developed to target areas where water sources are impacted by sewage and industrial effluent. Such systems include the Sari (woven fibres) filtration method used mainly by women in the Indian Subcontinent. This method involves folding garment cloth 4 to 8 times, then using the folded cloth filter microorganisms and solids from wastewater (Safe Water International Clearinghouse, 2013). A study conducted by Huq et al. (1996) revealed that the use of the Sari filtration method removes 99% of *Vibrio Cholerae* (cells attached to plankton) from water. The study concluded that this simple method

can be used to reduce the number cholera vibrios in raw water from water sources used for drinking water. The Sari method is an attractive method of water purification as the material is readily available to the layperson; however, the method does not address other contaminants like fecal coliforms (Safe Water International Clearinghouse, 2013).

2.4.2 Solar Disinfection (SODIS)

Solar water disinfections (SODIS) is one of the most basic forms of water treatment on a small scale. SODIS is a cost-effective technology that is efficacious with regards to the elimination of water borne pathogens (Islam et al., 2015). The most common manner in which to perform SODIS is to fill a 0.3 – 2.0 litre, transparent polyethylene terephthalate (PET) bottle with low turbidity water and thereafter placing the bottle on a roof or a rack exposed to direct sunlight for at least 6 hours on a sunny day, or for a period of 48 hours when faced with cloudy conditions (Centers for Disease Control and Prevention, 2014). The efficacy of the SODIS water treatment system is dependent on a number of factors, namely the geographic latitude of an area, the altitude of the area, the season and climate of an area, time of exposure to direct sunlight, the time of day, the presence or absence of clouds and cloud cover, volume and material of the bottle being used as well as water colour and turbidity (The Water Treatment Plants, 2011). Disease-causing organisms are destroyed via various mechanisms, namely thermal inactivation, UV-induced DNA alteration and photo-oxidative destruction. SODIS presents a number of benefits, the most important being the inactivation of bacteria, viruses and protozoa that cause diarrheal diseases. SODIS is also simple to use does not bring about a drastic change in the taste of water (Centers for Disease Control and Prevention, 2014). SODIS can be used by small communities around the world that do not have access to a centralised WWTP or access to alternative water treatment methods such as chlorination and filtration. The SODIS treatment method is also economical in that it requires a relatively low investment cost as it requires resources that are often readily available i.e. transparent plastic bottles and sunlight (The Water Treatment Plants, 2011). There are a number of drawbacks associated with the use of SODIS. The SODIS water treatment method can only effectively treat relatively clear water (turbidity below 30 NTU). This means that for highly turbid water, a pretreatment step such as filtration or flocculation is required. SODIS water treatment requires a large supply of intact, clean and suitable plastic bottles (Centers for Disease Control and Prevention, 2014). Another disadvantage of using SODIS technology is that it cannot be used to treat large volumes of water at once. SODIS treatment also requires a suitable climate as the technology is largely wholly dependent on sunlight (The Water Treatment Plants, 2011). Plastic bottles are

said to have the potential to leach compounds into water after being exposed to direct sunlight. Though researchers have been unsuccessful in detecting PET plastic photodegredation products or other harmful substances in SODIS treated water, at concentrations that may be harmful, concerns regarding contaminants of this nature remains high (Boyle et al., 2013).

2.4.3 Pure Water Bottle

An effective small-scale purification system is the Pure Water Bottle. The system is a container, approximately the size of a standard water bottle (500 mL) with a filter design that filters water down to 4 microns in particle size and a UV light system for water disinfection (Heimbuch, 2010). The invention is said to remove 99% of all water impurities and take a maximum of 2 minutes to produce potable drinking water (Transmissions, 2012). The idea was thought up by Loughborough University graduate Timothy Whitehead while travelling in Zambia and witnessing first hand, the water treatment issues that were being faced (Heimbuch, 2010).

2.4.4 LifeStraw®

Another small-scale invention is LifeStraw®. LifeStraw® is a cylindrical device that allows people to safely drink water directly from sewage impacted water sources. LifeStraw® is mainly designed for hikers, hunters and fishermen who may need to drink from the impacted water sources around them. LifeStraw® does however provide great benefits for those living near sewage impacted water. The invention works by removing 99.9999% of waterborne bacteria, and 99.9% of waterborne protozoan parasites from water sources. It also has a capacity to filter 1000 litres of water. LifeStraw® does not, however, remove heavy metals and chemicals from water. It is also an expensive item, costing $24.95 in 2005 which translates to approximately R351.05 today, assuming price stayed the same (The Cutting Edge Catalog, 2005).

2.4.5 Life Sack

The Life Sack, designed by industrial designers Jung Uk Park, Dae Youl, and Myeong Hoon Lee, is an innovative water treatment system in that it is essentially a sack designed for both shipping grain and foodstuffs to people living in remote areas, as well as a solar water treatment system. On completing the design of the Life Sack the three industrial designers approached the NGOs responsible for providing grain and food to people in remote areas to have the NGOs replace the conventional food bags in use with the Life Sack. After removing food supplies from the Life Sack, the bag can then be used as a solar water treatment system. The Life Sack works to treat and filter untreated water using Solar Water Disinfection Process technology.

Life Sack employs UV-A radiation as well as thermal treatment processes to destroy harmful microorganisms in untreated water (CMO Council and BPI Network, 2014).

2.4.6 Ceramic Water Filter (CWF)

Ceramic Water Filters (CWFs) are round bottomed, micro-porous water treatment pots made from a mixture of clay, sawdust and colloidal silver (SAFE Water Now, 2017). The filter itself sits within a plastic or ceramic vessel and can hold approximately 8 to 10 litres of water (Centers for Disease Control and Prevention, 2012). The CWF works to purify water in several different ways. The primary water treatment mechanism in a CWF is the filtering out of harmful bacteria and parasites through the micro pores in the outer ceramic wall of the filter (SAFE Water Now, 2017). The micro pores in the ceramic wall result in 99.99% removal of bacteria and suspended solids larger, or equal to, one micron. The ceramic wall also serves as an effective barrier against pathogens and bacteria including *E. coli* and *Salmonella*. The colloidal silver in the ceramic wall of the filter serves to keep the treated water in the CWF potable by acting as a biocide that inhibits the growth and reproduction of microorganisms within the filter. In some CWFs, the core of the ceramic filter is made up of activated carbon which acts as an adsorbent, removing chlorine, ammonia, fluoride, bad tastes and offensive odours (Southern Cross Pottery, 2017). Some advantages of the CWF include a low one-time cost as well as simplicity of use (Centers for Disease Control and Prevention, 2012). In a study conducted by Brown (2007) CWFs resulted in *E. coli* removal rates of 99.9999%. The drawbacks associated with using CWFs include their lack of effectiveness in treating water of viruses. Recontamination of treated water can also occur as there is no chlorine residual protection applied when using CWF (Centers for Disease Control and Prevention, 2012).

CWs are also an alternative water treatment option to conventional potable water treatment processes as they offer a low tech and low maintenance water treatment solution.

2.5 CONSTRUCTED WETLANDS

2.5.1 Constructed Wetlands General Overview

Wetlands can be defined as land areas that are wet either all year round or periodically, as a result of their location in the landscape (U.S. EPA, 2018). Wetlands are areas of transition between land and water and as such, the boundaries between wetlands and uplands, or deep waters, are not always clear cut (Kenkar & Kadlag, 2016). Wetlands develop conditions that favour plants adapted to the wetland environment (hydrophytes) and encourage the formation

of characteristic (hydric) wetland soils (U.S. EPA, 2018). Historically, wetlands were referred to as swamps, marshes, bogs, sloughs and fens, based on their geographical setting as well as on the plants and the water conditions that constituted the wetland. Constructed wetlands (CWs) are shallow basins filled with a substrate (usually gravel or soil) in which plants that are tolerant of saturated conditions are grown. Wastewater is introduced to the CW at one end of the wetland, flows over or through the substrate and is then collected at the other end (UN-HABITAT, 2008).

Wetland ecosystems are made up of various structural components, namely underlying strata, hydric soils, detritus, water and emergent vegetation. The underlying strata can be made up of lithic or mineral strata which are either impermeable or saturated with water. Typically, the underlying strata lie below the active rooting zone of the vegetation occupying the wetland. Hydric soils lie above the underlying strata are made up of the mineral and organic soil layer of the wetland. This layer is often saturated with water and serves the purpose of containing the rhizomes, roots, tubers, burrows and other connections to the surface environment (Kadlec et al., 2000). The permeability of the hydric soil layer plays a role in the movement of water in the wetland (Kenkar & Kadlag, 2016). The detritus within a CW is the accumulation of both live and dead organic matter in a wetland. This includes dead algae, dead emergent plant matter, live and dead animals as well as bacteria and fungi (Kadlec et al., 2000). Hydrology (water) is the most important factor in a CW as it ties together all the functions of a wetland (Kenkar & Kadlag, 2016). Standing water allows for the existence of a habitat for floating and submerged plant species, aquatic animals, microbes and living algae. Macrophytic plants also play a vital role in the functioning of a wetland as they assist in the achievement of high levels of water quality improvement that is characteristic of most wetland treatment systems (Dhote, 2007). Natural Wetlands often have all of the characteristics mentioned above while CWs tend to have components that are less mature in nature. An example of this is the soil organic matter. In natural wetlands, this component of the wetland is well established as it takes an extended period of time to form. In CWs however, the natural processes responsible for the development of a mature soil organic matter layer are forgone (Kadlec et al., 2000).

Experiments involving the use of CWs were first carried out in Germany by Käthe Seidel in the 1950s. Full scale systems were then put into use in the 1960s (Vymazal, 2007) and since then, CWs have become a reliable technology for treatment of wastewater (Vymazal, 2010). The effectiveness of CWs can be attributed to the various mechanisms that take place within the CWs to bring about and facilitate WWT. These include, but are not limited to; settling of

suspended solids; breakdown and conversion of water impacting substances by microorganisms and plants and the predation and natural die-off of pathogens; chemical transformation; ion exchange and adsorption on plant surfaces, substrate, sediment and litter; natural die-off and predation of pathogens (Kadlec et al., 2000); chemical precipitation and filtration as water comes into contact with the substrate and litter (Kenkar & Kadlag, 2016), aerobic and anaerobic microbial degradation, denitrification and plant uptake (UN-HABITAT, 2008). As a result of the wastewater treatment mechanisms presented by CWs, CWs are currently perceived as a natural, low cost water treatment technology aimed at imitating the water treatment processes characteristic of natural wetlands (UN-HABITAT, 2008).

2.5.2 Constructed Wetland Types

There exists four main types (or configurations) of CWs, namely Free Water Surface (FWS), Horizontal Sub-Surface Flow (HSSF), Vertical-Flow (VF) and Hybrid CWs (Vymazal, 2008). FWS, HSSF and VF CWs are classified according to the flow regime of the water passed through them while Hybrid treatment wetlands are made up of a combination of these three different CW types (Stauffer & Spuhler, 2018).

FWS wetlands can be defined as systems where the water surface is exposed to the atmosphere (U.S. EPA, 2000). FWS CWs have characteristic areas of open water, emergent aquatic vegetation and floating vegetation. The flow and infiltration of water in FWS CWs can be controlled using dikes, berms and liners depending on the condition of the soil as well as local regulations (Kadlec & Wallace, 2009). In FWS CWs water flows from an inlet point to an outlet over a vegetated soil surface (U.S. EPA, 2000). Wastewater flowing through FWS treatment wetlands is treated via various processes, namely sedimentation, oxidation/reduction, filtration, precipitation and adsorption. Because FWS CWs are an imitation of natural wetlands, they attract a variety of different organisms, namely insects, fish, reptiles, mammals and birds. FWS treatment wetlands are most commonly used in the advanced treatment of effluent that has undergone secondary or tertiary treatment. FWS wetlands present the potential for exposure to pathogens and as such, are seldom used for secondary treatment (Kadlec & Wallace, 2009). FWS wetland technology began in North America at the end of the 1960s along with the ecological engineering of natural wetlands for WWT (Vymazal, 2008).

HSSF wetlands are made up of soil or gravel beds with wetland vegetation. HSSF wetlands are generally made up of inlet piping, a synthetic or clay CW liner, emergent, filter media, berms and outlet piping with a level control mechanism (Kadlec & Wallace, 2009). HSSF CWs are

referred to as ‘horizontal flow’ wetlands due to the fact that the wastewater fed to the CW at the inlet flows through the porous medium below the surface of the gravel/soil bed in a horizontal fashion. The wastewater is the collected and discharged at the outlet of the CW (UN-HABITAT, 2008). The water in a HSSF CW is not exposed to the environment and thus human and animal exposure to pathogens is much lower in HSSF CWs compared to FWS wetlands. These CWs are commonly used to provide secondary treatment to single-family homes and small communities (Kadlec & Wallace, 2009). The concept of using CWs characterised by a horizontal subsurface flow for the purpose of water treatment was developed in Germany during the 1970s. The first operational CW was established in 1974 and was located in Othfresen, Germany (Kadlec et al., 2000).

VF CWs, designed by Käthe Seidel, were originally intended to serve as pretreatment units before feeding wastewater into HSSF beds. Early VF systems were comprised of several stages with the first stage having 2 – 4 beds that were fed wastewater in rotation. These multistage VF wetland systems were often served as the first water treatment stage in hybrid systems (Vymazal, 2008). VF CWs are often planted with common reed though other emergent macrophytes such as bulrush or cattails are viable options. VF treatment wetlands are made up of a flat gravel bed topped with sand as well as reeds (Kadlec et al., 2000). CWs of this nature are fed with large batches of wastewater from the top of the CW at regular intervals, resulting in the gradual percolation of the wastewater down through the gravel bed (UN-HABITAT, 2008). A drainage network exists at the base of the VF CW to allow for collection of the wastewater (Kadlec et al., 2000). The first form of VF CW was that of Seidel in Germany. The VF system was developed in the 1970s and was occasionally referred to as the Max Planck Institute Process (MPIP) or the Krefeld Process (Vymazal, 2008).

Different types of CWs can be combined so as to achieve greater removal rates by taking advantage of the effects offered by different kinds of CWs and their configurations (Stauffer & Spuhler, 2018). Systems of this nature are commonly referred to as Hybrid CWs (Vymazal, 2008). Hybrid CWs are commonly made up of VF and HF CWs arranged in a staged fashion to achieve the desired removal efficiency (UN-HABITAT, 2008). More recent Hybrid CWs however can have FWS CWs included as part of the Hybrid system. An example of such a system is the Hybrid CW system located in Kõo village of Viljandi County, Estonia. This Hybrid CW is comprised of two VF beds, followed by a HSSF bed and two FWS CWs (Vymazal, 2008).

2.5.3 Constructed Wetland Plants and Selection

A large number of plant species are able to grow in CWs. Plants that are used in CWs should ideally be capable of thriving in nutrient-rich conditions and be tolerant of saturated soils. Plants used in CWs also have to be able to grow in anaerobic environments (Kadlec & Wallace, 2009). Kadlec and Wallace (2009) have compiled a list of factors that need to be taken into account when selecting a plant species to plant in a CW. These factors include hydrology, climate, cost and availability of planting stock, plant size, the rate at which the plant spreads within the wetland, water quality and the maintenance requirements of the plant. Wood (1999) conducted an investigation on 27 CWs in South Africa. In this study, the vegetation planted in 23 of the 27 South African CWs is explicitly stated. The locations and the type of vegetation planted in each of the 23 CWs are listed in Table 1.

Table 1: South African CWs and their associated plant species

Location / Name of CW	**Plant Species / Name**
Mpophomeni	*Frogmouths Sp*
Letlhabile	Sedges
Ladybrand	*Typha Sp*
Bethlehem	*Frogmouths Sp*
Kruger National Park	*Frogmouths Sp*
Pilansburg Game Reserve	*Frogmouths S.*
Pietersburg Truck Stop	*Typha Sp*
Klipdrift	*Frogmouths Sp*
Middelburg	*Typha Sp*
Reddersburg	*Typha Sp*
Kranskop	*Frogmouths Sp*
Friemersheim	*Frogmouths Sp*
Warmbaths Cotton Gin	*Typha and Frogmouths Sp*
Karbochem Newcastle	*Typha Sp*
Midrand Residential Home	*Frogmouths Sp*
Mount Grace Hotel	Various Plant Species
CSIR Campus Wetland	*Eichhornia, Lemna, Typha* and *Frogmouths Sp*
Potchefstroom	*Lemna, Typha, Frogmouths Sp,* rushes and sedges
Makwane	*Frogmouths Sp*
Felixton Sugar Mill	*Frogmouths Sp*
Milnerton	*Typha Sp*
Oil Industry Wetlands (Pretoria and Secunda)	*Typha Sp*
Bannockburn Minewater	*Typha Sp*

Kadlec et al. (2000) list a total of 32 plant species that are appropriate for use in CWs. An adapted table of this list can be viewed in Table D1, Appendix D. Given the number of available plant species that are viable for use in a CW, it is evident that plant selection is dependent on various factors such as those given by Kadlec and Wallace (2009). Other elements that determine the successful selection of CW vegetation include selecting indigenous flora and

avoiding the use of invasive plant species (Sheridan, 2012). Xu et al. (2019) conducted an investigation on the impact of different CW plant species on CH_4 emissions. As part of their investigation, Xu et al. (2019) analysed data from 62 papers related to CH_4 fluxes in CWs. Their findings revealed that the most common plants used in CWs are *P. australis* (35.29%), *T. latifolia* (17.65%), *J. effusus* (8.82%), *P. arundinacea* (7.84%) and *Z. latifolia* (5.88%) respectively.

2.5.4 Advantages and Disadvantages of Constructed Wetlands

The use of CWs offers considerable advantages while presenting limitations as well. CWs present attractive advantages such as being less expensive to build in comparison to other treatment methods (Kadlec et al., 2000). Expenses associated with operations and maintenance of CWs are a lot lower compared to conventional wetland treatment systems (UN-HABITAT, 2008). On-site labour occurs periodically on a CW as opposed to the continuous on-site labour required on conventional WWT plants (Kenkar & Kadlag, 2016). CWs have a high tolerance for fluctuations in the flow of water entering the system and are also capable of treating wastewater with a low organic load. CWs also provide the advantage of creating a habitat for organisms living within or around the wetland. This in turn provides a level of aesthetic enhancement of the environment in which the CW is situated (Kadlec et al., 2000). Overall, CWs are systems that emulate natural systems and can thus work harmoniously with nature, making them an environmentally sensitive alternative to treating wastewater (UN-HABITAT, 2008). CWs also present a number of limitations. CWs generally require larger areas of land relative to conventional treatment plants (Kenkar & Kadlag, 2016). This means that treatment of wastewater using CWs can only be considered cost effective, relative to other options, only where land is available and affordable (UN-HABITAT, 2008). Wetland performance may be less consistent relative to the performance achieved in conventional WWTPs as CWs are affected by seasonal changes and environmental phenomena such as rainfall and drought. CWs require a minimum amount of water to operate and cannot survive complete drying. Another disadvantage with CWs is that the biological elements of CWs are sensitive to toxic chemicals like pesticides (Kenkar & Kadlag, 2016).

2.5.5 Constructed Wetland Findings from Previous Studies

Despite the mainstream use of conventional water treatment systems, CWs have been shown to substantially improve the quality of wastewater. Ligang et al. (2011) conducted an experiment in Luqiao Village of Xiake Town in Jiangyin Province, located within the Peoples Republic of China. The experiment consisted of a 20-acre CW designed to accommodate a load

of 700 tons of rural sewage wastewater per day. The CW was designed to treat sewage wastewater from 600 households in rural communities. The CW was monitored for a year with the main test indicators being total nitrogen (TN), total phosphorus (TP), suspended solids (SS), BOD_5, and the number of fecal coliforms. The average percentage removal rates of TN, TP, SS, BOD_5 and fecal coliforms were 81.23, 84.41, 62.50, 84.73 and 99.96 respectively. The removal rates of the test indicators are quite high except for that of the SS. Ligang et al. (2011) explains that this is as a result of plant debris not being cleaned timeously, thus contributing to the effluent concentration of SS. The CW has a fecal coliform reduction efficiency of over 99% which essentially means that the CW can effectively prevent the spread of pathogen contamination.

In Iran a subsurface flow CW of 150 m^2 comprised of *P. australis* reed was used in a test to treat municipal wastewater. The CW was subjected to an organic loading rate of 200 kg/ha/day and according to Badkoubi et al. (1998) the removal efficiencies of BOD, COD, total suspended solids (TSS), TN, TP and fecal coliforms in this system were 90, 86, 89, 34, 56 and 99% respectively (Kivaisi, 2000). The removal efficiencies of TN and TP are heavily affected by the substrate constituting a CW. Lu et al. (2016) revealed that the substrate medium in a CW plays an important role in the functioning of the wetland. In a consistent CW system where only the substrate was changed, Lu et al. (2016) found that at a hydraulic retention time (HRT) of 6 days a steel slag substrate gave the highest TP removal efficiency (83.5%) of 4 substrates (steel slag, maifanite, limestone and bamboo charcoal) while a maifanite substrate gave the highest TN removal efficiency (79.2%). In Nairobi, Kenya, a 0.5-hectare CW exists that is aimed at supporting wildlife and providing potential for recreation. The CW was designed for 1200 people and functions well with regards to wastewater treatment. The removal efficiencies of BOD, COD, TSS, TN, TP and fecal coliforms obtained using the system were recorded as 98, 94-98, 67-99, 87-90, 88 and 99.4-99.9% (Nyakang'o & van Bruggen, 1999).

In a study conducted by Sheoran (2017), a system of three equally sized (1 m × 1 m × 1 m) but independent CWs were used to treat synthetic AMD. The study was made up of three sets of experiments where the CWs were fed synthetic AMD to three different heights per experiment, namely 100 m, 150 m and 200 m respectively. In each experiment, a one litre sample of water was collected from each CW after 24, 48, 72, 96 and 168 hours for chemical examination. The results obtained from the three experimental runs conducted during this study showed an increase in pH from 2.93 to 7.22, a decrease in E.C. of 27.94 - 35.93%, a decrease in turbidity

of 30.14 - 66.65%, a 21.52 - 28.09% decrease in sulfates and a 35.90 - 76.44% decrease in manganese.

Song et al. (2006) conducted a 6-year investigation on a CW system situated in Rongcheng, Shandong Province, China. The CW system had a total area of 80 ha and a treatment capacity of 2.0×10^4 m^3 d^{-1}, thus allowing for the system to meet the water treatment needs of approximately 1.2×10^5 people. The treatment system was made up of a grill separation tank, a holding bank, distribution ditches and a system of CWs. Domestic wastewater and a small amount of industrial wastewater was collected in the grill-separation tank. The grill separation tank was set in place to remove large, solid waste particles, larger than 10 mm, from the wastewater. From the holding bank, the wastewater was then pumped into a distribution system that assisted in the transportation of the wastewater into the CW portion of the system. The CW portion of the system was comprised of a system of 108 treatment beds each having a length of 150 m, a width of 30 m and a maximum depth of 0.5 m. The CW treatment beds were all planted with common reed (*Phragmites australis*), however a few of the beds germinated other kinds of wetland plants naturally. Harvesting of above-ground biomass was conducted by hand on October of each year while soil tillage was carried out biyearly. Soil tillage was conducted to prevent the accumulation of organic loading and SS near the influent end of the treatment beds as this may have led to the disruption of flow distribution as well as head losses. During the 6-year investigation (January 1999 to December 2004) water samples were collected from the CW system and analysed for pH, BOD_5, COD, SS, ammonium nitrate (AN), TP, total coliforms and fecal coliforms. Over the 6-year investigation period, the pH showed a small increase between the inlet and the outlet. The mean inlet pH was 7.17 ± 0.19 while the outlet pH was recorded as 7.36 ± 0.23. The overall mean BOD_5 removal rate over the 6-year investigation period was 70.4 ± 9.6%. BOD_5 removal was however, affected by seasonal changes as the mean percentage removal of BOD_5 was notably higher in summer (74.1%) and spring (72.8%) than it was in winter (67.8%) and autumn (66.6%). The same can be said of the mean percentage removal rates of COD in this CW system. Seasonal changes had an effect on the COD removal in the CW system, resulting in higher COD removal rates in summer (66.3%) and spring (65.4%) compared to the COD removal rates achieved during winter (59.4%) and autumn (61.1%). The overall mean percentage removal rate for COD discovered during this investigation was 62.2 ± 10.1%. The CW system exhibited a relatively high mean percentage reduction of SS, removing 70 ± 8.9% of all SS entering the CW. A SS removal rate drop was observed during October of every year and coincided with the harvesting of above-ground

biomass that was conducted on every October during this investigation. SS mean removal rates dropped to 63.8 ± 20.6% in October. The harvesting of the reeds compromised the capacity of the CW to remove SS solids as SS removal is the result of physical processes involving aquatic plants. AN mean percentage removal was relatively low at 40.6 ± 15.3% removal. AN percentage removal was also subject to seasonal changes, as it was found that percentage removal rates associated with AN were much higher in summer (54.5%) compared to spring (33.7%), winter (32.4%) and autumn (43.3%). The pollutant that was the least effectively removed during this study was TP. TP presented a total mean percentage removal rate of 29.6 ± 12.8%. TP was also affected by seasonal changes, and as such, the highest TP removal rates were observed during summer (35.0%) and autumn (34.0%) while the lowest TP removal rates were observed in winter (28.9%) and spring (28.2%). The highest mean removal percentages observed during this study were those associated with total coliforms and fecal coliforms. The mean percentage removal rates of the total and fecal coliforms found in this investigation were 99.7% and 99.6% respectively.

Comparing the above studies, it appears that CWs have relatively high removal efficiencies for BOD/BOD_5 and fecal coliforms (above 70%). TN and TP removal are dependent on the substrate being used and TSS removal is dependent on the influent solids content of the water, the aquatic vegetation present in the CW, as well as on the number of SS the wetland itself, contributes to the water in the system. It is also apparent that the removal efficiency of a CW is affected by both seasonal changes and the manner in which the CW is maintained. To address possible microbial contamination of water leaving a CW system, UV disinfection can be used to inactivate harmful pathogens present in drinking water.

2.6 DISINFECTION TECHNOLOGIES AND UV DISINFECTION

Disinfection is the final step in the WWT process before treated water can be released back into the environment or made available for use and consumption (The Macon Water Authority, 2011). The disinfection process ensures that a substantial percentage of pathogens are inactivated. This provides humans, as well as the environment, with a degree of protection against pathogenic organisms including those causing cholera, typhoid, polio and hepatitis (Norweco, 2018). There are various methods of disinfection including chlorination, ozonation and UV disinfection.

2.6.1 Chlorination

Chlorination entails the addition of measured doses of chlorine to water in order to produce a residual (low level of chlorine) sufficient to destroy viruses, cysts and bacteria. Chlorine is usually added to public drinking water as a final treatment step, often after an upstream filtration step aimed at removing sediment that may shield pathogenic organisms from the effects of the chlorine (WaterProfessionals, 2018). Chlorine inactivates microorganisms by causing damage to the cell membrane of the microorganism. Once the cell membrane is compromised, the chlorine enters the cell and disrupts the cell respiration and DNA activity of the cell. Chlorination has proven effective against viruses and bacteria and is a popular choice for water disinfection as it is inexpensive yet effective in water treatment (Safe Drinking Water Foundation, 2018). A considerable disadvantage associated with chlorination is the formation of chlorine disinfection by-products (DBPs). Chlorine is able to combine with organic compounds in wastewater to form DBPs such as haloacetic acids (HAAs) and trihalomethanes (THMs). It is said that the continuous exposure to such DBPs over an extended period of time contributes towards the number of cancer risk factors an individual may have (Minnesota Department of Health, 2005). Another disadvantage of chlorine is that it is not capable of inactivating all microbes. There exist some protozoan cysts that are immune to the inactivating effects of chlorine (Safe Drinking Water Foundation, 2018).

2.6.2 Ozonation

Ozonation (also known as ozonisation or ozone disinfection) is a water disinfection technique involving the infusion of ozone into water. Ozone is made up of three oxygen atoms (O_3) and is one of the strongest oxidants. Ozonation is a kind of advanced oxidation process that entails the production of a very reactive oxygen species (Mazille & Spuhler, 2018). The rearrangement of atoms when oxygen molecules are subjected to high-voltage discharge results in the formation of ozone (Cullen et al., 2011). For water disinfection purposes, ozone is typically produced conventionally using either a Corona-Discharge Ozone generator or a UV Ozone generator (Oram, 2014). Ozone is an unstable gas that readily decomposes to form a free radical (free oxygen atom) as well as molecular oxygen (O_2). This free oxygen radical is extremely reactive and oxidises the cell membranes of pathogens, effectively destroying the cell (WPM SA, 2017). There are a number of advantages associated with the use of ozonation. Ozonation presents far greater germicidal properties than chlorination. Ozonation results in the rapid destruction of bacteria, viruses and protozoa (such as Giardia and Cryptosporidium) over a wide pH range (Mazille & Spuhler, 2018). Though ozonation results in the production of DBPs,

these are mainly organic DBPs such as ketones and aldehydes that can easily be degraded using a bio filter (Lenntech, 2019). A disadvantage of concern in the use of ozonation is the formation of bromate during the disinfection of untreated water containing high levels of bromide. Bromate has been identified as a DBP of the ozonation process and poses a great heath risk as it is suspected that bromate is carcinogenic (Jarvis et al., 2007). Ozonation does not offer any germicidal or disinfection residual effects, meaning should any pathogenic microorganisms survive during the ozonation disinfection process, downstream users may acquire the diseases associated with the surviving microorganism (Oram, 2014).

2.6.3 UV Disinfection

UV disinfection in water treatment can be defined as a physical process whereby microorganisms are neutralised as they pass by UV lamps submerged in untreated water (Trojan Technologies, 2019). UV disinfection is also applied in the healthcare industry to sterilise medical operating rooms and surgical instruments (Kodoth & Jones, 2015). The UV disinfection process in WWT involves the exposure of untreated water to radiation from UV light (National Drinking Water Clearinghouse, 2000). UV disinfection treatment works through the penetration of UV light through the cell walls of pathogenic organisms. The UV radiation from UV light disrupts the genetic material within the cell, causing inactivation of the organism (National Drinking Water Clearinghouse, 2000).

2.6.3.1 UV Radiation Types and Pathogen Inactivation UV Dosages

There are different types of UV radiation in existence, namely UV-A (315-400 nm), UV-B (280–315 nm), UV-C (200-280 nm) and Vacuum-UV (100-200 nm) (Lui et al., 2014). Conflicting information exists regarding the UV radiation type deemed effective for the inactivation of microorganisms. There are those, like LeChevallier and Kwok-Keung (2004), and Lui et al. (2014) that assert that UV-B and UV-C UV radiation are responsible for microorganism neutralisation, while others such as Brian Oram (2014) claim that only UV-C type UV radiation possesses the germicidal properties necessary to destroy pathogenic microorganisms. The consensus, however, is that UV-C radiation is effective in the inactivation of pathogenic microorganisms. Protozoa, bacteria and viruses exposed to the germicidal wavelengths (primarily 254 nm) of UV light are incapacitated from reproducing and infecting as a result of UV treatment. UV light has been shown to be effective against pathogenic organisms that cause polio, typhoid, cholera and other viral, parasitic and bacterial diseases (Trojan Technologies, 2010). The germicidal effect of UV disinfection is determined by the UV energy dosage absorbed by a microorganism. The UV dosage received by a microorganism

(commonly expressed in mJ/cm^2 or $\mu Ws/ cm^2$) is the product of the UV light intensity and the length of time it is exposed to the set UV light intensity (Moore, 2014). The required log-reduction (i.e. extent of disinfection) will dictate the necessary UV dosage for disinfection. Water exposure time is mainly influenced by the design of the reactor as well as the flow rate of water through the system (Trojan Technologies, 2010). UV intensity is affected by the type of equipment used (e.g. lamp type) as well as water quality parameters such as TSS and UV transmittance in the water (URS Corporation, 2004). Log-reduction values are used to express the percentage pathogen disinfection of water. The log inactivation values of 1-log, 2-log, 3-log and 4-log correspond to disinfection percentage values of 90, 99.9, 99.99 and 99.999% respectively (Microchem Laboratory, 2015).

2.6.3.2 UV Disinfection Advantages and Disadvantages

According to Trojan Technologies (2010), the average doses for the log inactivation values of 1-log, 2-log, 3-log and 4-log associated with disinfecting water of *E. coli* bacteria are 1.5, 2.8, 4.1 and 5.6 mJ/cm^2 respectively. UV disinfection has a number of advantages. A major advantage in the use of UV disinfection is that it does not form any DBPs. Another advantage of this treatment method is that there are no potentially harmful chemicals used that may be consumed by downstream users. It has also been found that, unlike ozonation, UV disinfection does not form bromines or bromates from bromide. A disadvantage that UV disinfection shares with ozonation is the lack of residual disinfection effects after treatment. Pathogenic organisms that survive the UV treatment step can cause harm to downstream consumers (National Drinking Water Clearinghouse, 2000). Another disadvantage with UV disinfection is that if the light intensity is not powerful enough to destroy the pathogenic microorganisms in untreated water, these pathogens can undergo a process of repairing themselves. Microorganisms can repair themselves in three via three mechanisms, namely Photoreactivation/Dark Repair, Nucleotide Excision Repair and Recombination repair (URS Corporation, 2004). Distance also plays a role in UV disinfection, as the further away a UV light source is from wastewater, the lower the UV dosage will be that the wastewater receives (Lorcheim, 2015).

2.6.3.3 UV Disinfection Findings from Previous Studies

Kodoth and Jones (2015) conducted an experiment that involved observing the phenotypic changes of *E. coli* (strain K12) cells after exposure to short wavelength UV light (254 nm). The study was specifically aimed at investigating the disinfection effects of small UV lights used in high school laboratories in order to assess whether or not common households could make use of UV radiation on a small scale. The experiment was made up of four petri dishes

inoculated with *E. coli* using an isolated streak derived from a plate of healthy bacteria. One petri dish was exposed to UV light for 24 hours and another was exposed to UV light for 48 hours. The remaining two petri dishes served as control plates for the two dishes that were exposed to UV light; one control for the dish that was exposed to UV light for 24 hours and the other for the dish that experienced 48 hour UV light exposure. Three trials were performed in order to verify the results obtained during this experiment. After three experimental trials it was apparent that the number of colony-forming units (CFUs) in the petri dishes exposed to UV radiation was visibly lower compared to the control petri dishes. It was also found that there were fewer CFUs in the petri dishes exposed to 48 hours of UV radiation compared to the petri dishes exposed to 24 hours of UV radiation. The conclusions drawn from this experiment were that: UV radiation was able to inhibit the growth of K12 strain *E. coli* with the possibility that UV radiation had killed off the bacteria; Longer UV exposure times result in a greater decrease in the number of observable CFUs; and UV high school laboratory lights of short UV wavelengths are able to bring about the inhibition of bacterial growth.

Despite how effective UV disinfection is at inactivating microorganisms, it does not address the presence of pollutants such as heavy metals in drinking water. Heavy metals are a major environmental and health concern with regards to water human and animal consumption. Despite the existence of multiple technologies capable of treating wastewater containing heavy metals such as Chemical Precipitation, Ion Exchange and Membrane Filtration, activated carbon is the most commonly used method in environmental treatment processes addressing heavy metal removal worldwide (El Zayat & Smith, 2010).

2.7 HEAVY METAL REMOVAL TECHNOLOGIES AND ADSORPTION

2.7.1 Heavy Metals in Drinking Water; Causes and Effects on Human Health

The presence of heavy metals in wastewater is a major environmental pollution concern (Gunatilake, 2015). Heavy metals are also known as trace elements and are characterised by an atomic density exceeding 5 g/cm^3 (Akpor et al., 2014). The majority of elements that fall under this category have a high water solubility and are well-known toxic or carcinogenic agents. Examples of heavy metals are copper, zinc, gold, mercury, silver, lead, manganese, aluminium and tin (Gunatilake, 2015). Some heavy metals such as copper, manganese, cobalt and zinc are essential for human health (Blakemore & Jennett, 2001); however, excessive amounts of these metals can have negative effects. Heavy metals are released into the

environment through natural processes and anthropogenic activities (Malik & Khan, 2017). Natural sources that contribute toward heavy metal pollution in wastewater include volcanic activities, soil erosion, aerosol particles and urban runoff (Akpor et al., 2014). Electroplating, metal smelting, agriculture and manufacturing processes are all examples of human activities contributing towards heavy metal pollution in wastewater (Malik & Khan, 2017). Each individual heavy metal presents specific signs when consumed by human beings in toxic doses, however the general signs associated with cadmium, lead, mercury, zinc, arsenic, aluminium and/or copper poisoning include diarrhoea and other gastrointestinal problems, tremor, haemoglobinuria, stomatitis, ataxia as well as pneumonia (Akpor et al., 2014). Inorganic arsenic and cadmium have been classified as human carcinogens, according to the International Agency for Research on Cancer (IARC). While both arsenic and cadmium are considered carcinogenic, arsenic is associated with skin damage while cadmium is associated with kidney damage (Malik & Khan, 2017). As a result of the harmful effects associated with the consumption of heavy metals in water by humans, various methods and technologies have been identified and developed in an attempt to ensure heavy metals are effectively removed from water sources. These methods include chemical precipitation, ion exchange processes, membrane filtration and adsorption (Gunatilake, 2015).

2.7.2 Chemical Precipitation

Chemical precipitation is a conventional and proven technology for the removal of heavy metals in wastewater. It is also used for the removal of other inorganic substances, fats, oils and suspended solids (U.S. EPA, 2000). The goal in using the Chemical Precipitation process is to produce an insoluble metal precipitate by reacting the dissolved metals in wastewater with a precipitant. Once the heavy metals are in precipitate form they are removed from the wastewater which in turn brings down the heavy metal concentration in the water (Gunatilake, 2015). Lime is often used as the precipitant of choice in chemical precipitation processes due to its low cost; however, large quantities of lime are necessary for this process to be effective (U.S. EPA, 2000).

2.7.3 Ion Exchange

Ion Exchange processes are reversible chemical reactions used in the removal of dissolved ions from a solution while replacing them with ions of a similar charge. Ion exchange is mainly used in the removal of calcium and magnesium ions from water as a means to soften hard water; however, it is steadily being used more often in the removal of other dissolved ionic species (U.S. EPA, 2018). The ion exchange process makes use of an ion exchange resin. This

resin is a synthetically polymerised organic compound containing either positive or negative sites that have the capability to attract ions of an opposite charge from the solution surrounding the resin (Editors of Encyclopaedia Britannica, 2018). The ion exchange process obeys the law of Electroneutrality, in that the total number of ionic charges leaving the solution surrounding the resin is equivalent to the total number of ionic charges entering the solution (Al-Enezi et al., 2004). Ion exchange processes present the advantages of utilising low-cost materials and user-friendly operations. The technology has also been proven to be effective in the removal of heavy metals from aqueous solutions. Though ion exchange has been shown to be effective in removing heavy metals from aqueous solutions, it is only effective in treating solutions with a low concentration of heavy metals (Gunatilake, 2015).

2.7.4 Membrane Filtration

Membrane filtration is a technique used to separate particles in liquid solutions or gaseous mixtures (Synder Filtration, 2018). In WWT, this separation technique is used to remove suspended solids, organic compounds and inorganic components like heavy metals, from wastewater (Gunatilake, 2015). Membrane filtration makes use of a semi-permeable membrane that plays the role of a barrier. This membrane works by retaining larger particles while permitting smaller molecules to pass through into the permeate. There are four main pressure-driven membrane filtration processes. In order of decreasing pore size, these four processes are Microfiltration (MF), Ultrafiltration (UF), Nanofiltration (NF) and Reverse Osmosis (RO) (Synder Filtration, 2018). MF is a physical separation process that removes suspended solids from wastewater by means of a membrane with a small pore size. Membranes used in the MF process typically have a pore size ranging from 0.1 to 10 µm. Unlike NF and RO, which are processes that can be used to remove dissolved species from a solution, MF employs only physical filtration to remove particles (WaterProfessionals, 2018). Such particles include bacteria, large particulates and colloidal particles. MF is often used to treat wastewater in the food and beverage industry before it is discharged to a municipal sewer. UF is similar to MF, the only difference being that UF makes use of a permeable membrane with smaller pores ranging from 0.01 to 0.1 µm (Synder Filtration, 2018). UF is capable of separating macromolecules, heavy metals and suspended solids from inorganic solutions (Gunatilake, 2015). UF is a separation process widely used in WWT as well as in protein concentration. NF membranes are similar to RO membranes as they are made up of a thin film-like composite layer with a pore size of less than 1 µm, on top of a porous layer with a pore size ranging from 50 to 150 µm. The NF processes is capable of rejecting multivalent salts as well as uncharged

solutes (Synder Filtration, 2018); however, NF is not as effective at removing monovalent ions from water as RO is. RO can achieve 98-99% removal of monovalent ions while NF typically removes 50-90%, depending on the material and the design of the membrane (WaterProfessionals, 2018). RO is a separation process that employs pressure to force a solution through a membrane causing the retention of the solute on one side of the membrane and allowing the solvent to pass through to the other side (Gunatilake, 2015). RO membranes are capable of rejecting all monovalent ions while permitting the passage of water molecules in aqueous solutions. RO is commonly used in the treatment of industrial water and in the desalination of seawater (Synder Filtration, 2018). Though there exist four membrane filtration processes, only UF, NF and RO have pore sizes small enough to bring about effective heavy metal removal in wastewater (Gunatilake, 2015). The overall yield obtained through using NF and RO is relatively lower than that obtained using MF and UF due to the small pore sizes and high operating pressures associated with the NF and RO processes (Synder Filtration, 2018). Membrane separation processes present a limitation in that membranes often become clogged as a result of fouling. Membrane fouling can be defined as the deposition of dissolved or suspended solids on the surface of membranes. Fouling results in decreased flux across membranes thereby decreasing the effectiveness of separation processes involving membranes. In order to use membrane separation as an effective separation process in the long term, membranes need to be washed using cleaning agents such as acids alkalis, disinfectants and surfactants (Tomaszewska & Białończyk, 2012).

2.7.5 Heavy Metal Adsorption and Findings from Previous Studies

Adsorption is a process that is widely accepted in environmental treatment processes around the world. The adsorption process is a phase transfer process used to remove substances from a fluid phase (liquids or gases). It can also be described as the enrichment of chemical species from a fluid phase on to a liquid or solid surface. With regard to WWT, adsorption has proven to be an effective removal process for a myriad of solutes. In WWT, ions or molecules are removed from the aqueous solution by adsorption onto solid surfaces (Worch, 2012). A variety of different adsorbents have been developed and tested for a variety of applications. Adsorbents used in WWT include activated carbon, alumina, bentonite, clays, fly ash, ferric oxide, magnesium oxide, saw dust, silica, zeolites and activated anthracite volcanic ash (Rathi, 2001).

Heavy metal adsorption is most commonly performed using activated carbon as processes like RO, ion exchange and chemical precipitation are much more expensive (El Zayat & Smith, 2010). Moreno et al. (2010) used an activated carbon made from cow bone char (CBC) as a

heavy metal adsorbent and found that it is able to retain Mn^{2+}, Fe^{2+}, Cu^{2+} and Ni^{2+} heavy metal ions from an aqueous solution. The quantities of Mn^{2+}, Fe^{2+}, Cu^{2+} and Ni^{2+} adsorbed per gram of CBC at equilibrium were 29.56 mg, 31.43 mg, 35.44 mg and 32.54 mg respectively. Agarwal and Gupta (2015) used animal bone charcoal (ABC), made from the bones of animals to remove Cr (VI) ions from an aqueous solution. The results from their experiments revealed that at a pH of 2, 92% of the Cr (VI) ions, in a solution with a concentration of 100 mg/L, were removed.

Experiments conducted using coconut coir pith activated carbon, as an adsorbent, point toward coir pith being a potentially effective way of carrying out heavy metal adsorption in real industrial wastewaters. The removal rates of heavy metal ions recorded during this experiment are 73% for Cu (II) at a pH of 5.0, 100% for Hg (II) at a pH of 3.5, 100% for Pb (II) at a pH of 4.0, 100% for Cd (II) at a pH of 4.0 and 92% for Ni (II) at a pH of 3.5 (Kadirvelu et al., 2001).

Jearmeen (2015) conducted a study that was aimed at assessing the efficiency of Bird Bone Char powder at adsorbing Chromium (Cr) from Titanium Industry effluent. The experiment involved using two different kinds of effluent, namely raw and treated titanium industry effluent. Powdered Bird Bone Char was mixed with the two different effluent types in 5 different concentrations for each effluent. The five concentrations of the Bird Bone Char to Effluent mixtures used for this experiment were 1, 2, 3, 4 and 5 g/L respectively. Each Bird Bone Char and Effluent mixture was maintained at six different pH levels (0.5, 2, 4, 7, 8 and 9). The Bird Bone Char adsorbents were thoroughly mixed with the effluent solutions and the mixtures were agitated every 2 hours. The adsorption process was carried out for a period of 60 days and the removal efficiency of Cr achieved by the different Bird Bone Char –Effluent mixtures was recorded thereafter. Jearmeen (2015) discovered that the removal rates of Cr in raw titanium industry effluent by Bird Bone Char were highest in the Bird Bone Char-Effluent mixtures having a pH of 7. Cr removal rates of 47.79%, 51.83%, 55.96%, 62.42%, and 68.87% were achieved at a pH of 7 for Bird Bone Char to raw titanium industry effluent concentrations of 1, 2, 3, 4 and 5 g/L respectively. Removal efficiencies associated with treated titanium industry effluent were also found to be highest at a pH of 7. The Cr removal rates achieved using Bird Bone Char to adsorb Cr in treated titanium industry effluent are given by Jearmeen (2015) as 56.65%, 60.12%, 67.09%, 69.51% and 76.96% for Bird Bone Char to Effluent concentrations of 1, 2, 3 ,4 and 5 g/L respectively.

In the context of this literature, an opportunity to provide an effective, cheap, small-scale, water treatment system for rural communities, and communities affected by sewage impacted water,

exists. The opportunity itself is a means to equip people affected by sewage impacted water sources with the knowledge and technology to treat water themselves and thereby circumvent the socio-economic issues surrounding improper water treatment practices.

CHAPTER 3 – METHODOLOGY

3.1 SAMPLE COLLECTION AND PREPARATION

The sample collection and preparation process involved feeding the system with river water as well as collecting liquid samples from the feed water and from the CW, charcoal and UV chambers. On each sample collection and feeding day, a volume of approximately 5 litres of river water was fed to the CW chamber, displacing the water in the CW chamber and pushing it into the charcoal chamber. The water in the charcoal chamber was then displaced by the water coming from the CW chamber and flowed into the UV chamber. The water in the CW and charcoal chambers was left to stand for at least a day (24 hours) before collecting samples from these chambers. Sample water in the UV chamber was collected after 1 hour of exposure to UV light on each sampling day. The UV light dosage in the UV chamber was 28800 mJ/cm^2. This dosage, however, represents the dosage provided by the LEDs on their surfaces and not throughout the UV chamber. Because the PPWTS was situated outdoors and exposed to the elements, the volume of the water in the system decreased due to evaporation and evapotranspiration. For the purpose of this project however, it was assumed that 5 L is fed to the system and thus 5 L exits the system.

Before feeding the system on a sampling day however, sample aliquots were collected from the feed water to be fed to the system, and from the CW and charcoal chambers. This sampling procedure was carried out 2 to 3 times a week (Monday to Friday) allowing for a minimum time of 24 hours before collecting the next set of samples and feeding the system. The system was not fed on weekends as coliform measurements must be done shortly after the collection of samples and total coliform and *E. coli* testing equipment was unavailable on weekends. Sample aliquots were collected from the CW chamber using a syringe fitted with a metallic extension that could penetrate the wetland gravel. This was done to retrieve a representative water sample from within the wetland. Sample aliquots collected from the charcoal chamber were collected as an overflow coming from the outlet pipe of the charcoal chamber. Sample aliquots were retrieved from the UV chamber by using the valve regulating the flow of water from the UV chamber outlet pipe. The samples collected were stored in 50 mL Centrifuge tubes. A total of 8 separate water samples were collected from each chamber and from the feed water for analysis. The samples were labelled pH, E.C and turbidity; heavy metals; COD; TOC; phosphate; nitrate; sulfate and total coliforms and *E. coli*.

It was noted, on each sampling day, that the body of water in the CW chamber was the water from the last feed to the system (+24 hours before sampling), while water occupying the charcoal chamber was water that had been fed to the CW chamber the second last time the system was fed (+48 hours before sampling). The water that occupied the UV chamber was water coming from the charcoal chamber. The water in the UV chamber was exposed to 1 hour of UV light on each sampling day. All Heavy Metal samples were acidified with a dilute Nitric Acid solution to a pH of < 2 to prevent precipitation of the metal ions in solution. All samples collected from the CW chamber were filtered (using Munktell Ahlstrom filter paper, Grade 393, Diameter = 150 mm) save for the pH, E.C and turbidity CW sample. Filtration of the CW samples was necessary as the samples retrieved from the CW chamber were high in particulates which would have affected the accuracy of the analyses that were performed on the water. The pH, E.C and turbidity CW samples were not filtered as this would have produced an inaccurate turbidity reading.

A volume of 5 L of feed water was added to the system at a time as this is a volume of water sufficient to provide one person with enough potable water to consume in a day. The length of time the UV LEDs were switched on was part of the design of the system. The idea behind exposing the sewage impacted water to 1 hour of UV light was to achieve substantial inactivation of pathogens while providing potable water in relatively short span of time and conserving the battery pack that powers the UV LEDs.

Weather details were recorded on each sampling day to monitor the effect of weather on the system and its operation. These weather details can be found in Appendix B.

3.2 TESTING OF THE SAMPLES

The pH, E.C. and turbidity of the sample water collected from each chamber was measured. The samples were also tested for phosphates, nitrates, chemical oxygen demand (COD), sulfates, *E. coli* bacteria and total coliforms, total organic carbon (TOC) and (heavy) metals (aluminium, manganese and lead). Below is a description of the equipment, materials and processes that were used to obtain the experimental results given as part of this research.

3.2.1 pH Measurement

The pH of the water samples (feed, CW, charcoal and UV) was measured using the OHAUS Starter 3100 pH Meter. Calibration of the pH meter was carried out using three prepared buffer solutions of pH 4.01, 7.00 and 10.01 respectively. A control buffer (of pH 4.01, 7.00 or 10.01) was also used to assess the effectiveness of the calibration.

3.2.2 E.C.

The E.C. of the water samples collected from the water treatment system was measured using the OHAUS Starter 3100c Conductivity Bench meter. The conductivity meter was calibrated using a Wirsam Conductivity Solution of E.C 1413 µS/cm.

3.2.3 Turbidity

Turbidity was measured using the HACH 2100N Laboratory Turbidimeter. Three HACH control samples (1.8 NTU, 18 NTU and 180 NTU, Cat no. 22543-01, 22543-02 and 22543-03 respectively) were measured for turbidity to assess accuracy of the turbidity readings given by the turbidimeter. HACH Sample cells (Cat. no 2084900) were used to contain the water samples to be analysed. A Lint Free cloth was used to wipe the outside of the sample cells used while a HACH Silicone Oil (Cat no. 126936) was applied to the sample cells before measuring turbidity using an Oiling Cloth.

3.2.4 Chemical Oxygen Demand (COD)

The COD of the water samples was measured using the Merck COD Solutions A+B Spectroquant Method for samples with a COD concentration range of 4.0 – 40.0 mg/L. COD Solution A Cat no. 1.14538.0065 and 2.85 mL of COD Solution B Cat no. 1.14681.0495 were used. The COD test kit reagents and the water samples were brought together inside Merck 16 mm spectroquant vials (Cat no. 1.14724.0001). Merck COD Standard Solution, CRM of concentration 2000 mg/L in H_2O (Cat no. 1.25033.0100) was used to prepare COD standard samples of 10 ppm, 20 ppm and 40 ppm respectively. A blank COD sample was made up using deionized water to achieve a zero COD reading. A control standard of 1176 mg/L was prepared using 0.85 grams of Potassium Hydrogen Phthalate Salt ($C_8H_5KO_4$) and deionised water in a 1000 mL volumetric flask. The COD samples were heated in a Merck Spectroquant® TR 320 Thermoreactor at 148˚C for 120 minutes. The Merck Spectroquant Pharo 300 Spectrophotometer was used to measure the COD sample absorbance values at a wavelength of 340 nm.

3.2.5 Phosphate

Phosphate water tests were conducted using the Merck phosphate test kit (Cat no. 1.14848.0002). A phosphate standard solution of 1000 mg/L was prepared using 0.3582 g of KH_2PO_4 and deionised water in a 250 mL volumetric flask. The phosphate test involved the use of Merck 16 mm spectroquant vials. The Merck Spectroquant Pharo 300 Spectrophotometer was used to measure the concentration of the phosphate samples using a built-in phosphate test method (PO_4-P, 0.05 – 5.00 mg/L).

3.2.6 Nitrate

Nitrate water testing was done using the Merck nitrate test kit (Cat no. 1.14773.001). A nitrate standard solution of concentration 1000 mg/L was prepared with 0.4077 g of KNO_3 deionised water in a 250 mL volumetric flask. A control standard of 1000 mg/L was also prepared using 0.3427 g of $NaNO_3$ with deionised water in a 250 mL volumetric flask. The Merck Spectroquant Pharo 300 Spectrophotometer was used to measure the absorbance of the nitrate samples using a built-in nitrate test method (NO_3-N, 0.5 – 18.0 mg/L). The test was conducted using Merck 16 mm spectroquant vials.

3.2.7 Total Coliforms and *E. coli*

Total coliforms and *E. coli* tests were carried out using the IDEXX Colilert-18 test kit made up of the 120 mL sterile vessels, IDEXX Defined Substrate Technology (DST) nutrient indicator reagent packs and IDEXX Quanti-Tray/2000 sample trays. Samples were diluted with deionised water before testing to keep readings within the limits of the test kit. Dilutions were carried out relative to the results of a previous test. The IDEXX Quanti-Trays containing the diluted sample and DST nutrient indicator reagent mixture were sealed using The IDEXX Quanti-Tray Sealer Model 2X and then incubated at 35 ± 0.5˚C for 18 hours. To obtain a most probable number (MPN) value for total coliforms, the number of yellow wells on the IDEXX Quanti-Tray/2000 trays were counted and the MPN table provided by IDEXX was consulted. To obtain an MPN value for *E. coli* the IDEXX Quanti-Tray/2000 trays were observed under a 6-watt, 365-nm UV light to count the number of fluorescent wells. The dilution factor associated with each sample was multiplied by the MPN value obtained to achieve an actual MPN value. A comparator sample was prepared using deionized water to allow for comparison between wells that have turned yellow, or wells that fluoresce, with the comparator sample that was negative for both total coliforms and *E. coli*.

3.2.8 Total Organic Carbon (TOC)

TOC of water samples was measured using the Teledyne Tekmar Torch Combustion TOC Analyzer. Calibration standards (5, 10, 25, 50 and 100) were used in the calibration of the instrument prior to the commencement of the overall sample batch run.

In this technique (catalytic combustion), samples were injected by syringe into a catalyst packed tube which is enclosed in a high-temperature (680˚C -1000˚C). The carbon in the sample is oxidized into carbon dioxide. The CO_2 is then swept by the carrier gas to the non-dispersive infrared (NDIR) detector. Once the gases inside the detector have reached equilibrium, the detector seals off. A single reading is used to determine the amount of CO_2 in the detector cell as it correlates directly to the concentration of the carbon contribution from the sample. This method (catalytic combustion) is best suitable for use in wastewater applications with 10 - 20 000 ppm range. The combination of temperature, an oxygen rich environment from the carrier gas (generally Ultra Zero Air or Oxygen), and a catalyst is used to oxidize the carbon in the sample to CO2. The CO2 is then swept to the Non-Dispersive, Infrared (NDIR) detector.

3.2.9 Sulfate

Sulfate tests were conducted using a photometrics method to calculate the concentration of water samples. Sulfate standards were prepared using Sulfate Standard Solution 1000 mg/L (Cat no. 1.19813.0500). A sulfate control standard was prepared with 0.4538 g K_2SO_4 and deionised water in a 250 mL volumetric flask. A gelatine solution was prepared with 17.5 g of gelatine, 190 mL of 32% HCl and deionised water in a 1000 mL volumetric flask. The sulfate test also required the preparation and use of a Barium Chloride (BaCl) solution made using 0.3 g of BaCl dissolved in 1 mL of deionised water. The Merck Spectroquant Pharo 300 Spectrophotometer was used to measure the absorbance readings of the sulfate samples at a wavelength of 880 nm. The test was conducted using Merck 16 mm spectroquant vials.

3.2.10 Heavy Metals

Heavy metals testing (aluminium, manganese and lead) was conducted by the Instrument Scientist at the Analytical Facility situated at the University of Johannesburg (UJ). Inductively Coupled Plasma Mass Spectrometry (ICP-MS) was the method of heavy metals testing used for this research. The Perkin Elmer 300X quadrupole ICP-MS was used to perform the ICP-MS analyses. The mass-to-charge ratio values (m/z) that were used to measure aluminium, manganese and lead

were 27, 55 and 208 respectively. Before testing, sample water was acidified to a pH of below 2 to suppress the precipitation of metallic ions from the water.

3.3 DESIGN AND BUILD OF THE SYSTEM

The Portable Potable Water Treatment System (PPWTS) is made up of three chambers through which wastewater flows; the CW, charcoal chamber and UV chamber respectively. The system is designed such that by feeding water to the CW chamber, the water occupying the CW is pushed into the charcoal chamber while the water occupying the charcoal chamber flows into the UV chamber. The CW chamber is the first chamber the wastewater occupies as the CW is aimed at holistically improving the quality of the water while bringing down the turbidity of the water that will occupy the following chamber. The charcoal chamber follows the CW for the adsorption of metals. The UV chamber is positioned last to treat any *E. coli* and coliform bacteria that may still be present in the water. The design of the system was inspired by the idea to provide an effective, affordable and low maintenance water treatment device for individuals in areas lacking adequate and effective water treatment services. A small CW was made part of the system as the CW only requires water to continue functioning. Charcoal was selected as the heavy metal adsorbent of choice for this project as it is much easier to access for individuals living in rural and remote areas compared to activated carbon. Charcoal is also more affordable relative to activated carbon which is commonly used in the adsorption of heavy metals (El Zayat & Smith, 2010). UV light was considered as part of this design due to the lack of DBPs produced using UV disinfection. The design of the system was aimed at effectively treating sewage impacted water of various water impurities safely at low cost. The following section is a breakdown of the materials that constitute each chamber, a schematic diagram of the system as well as images of the system components.

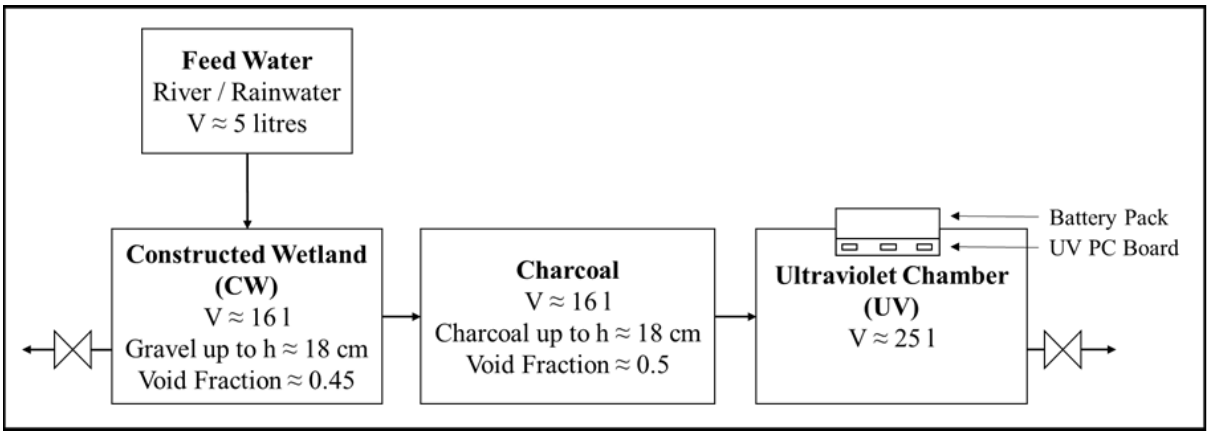

Figure 3.1: Portable Potable Water Treatment System Schematic Diagram

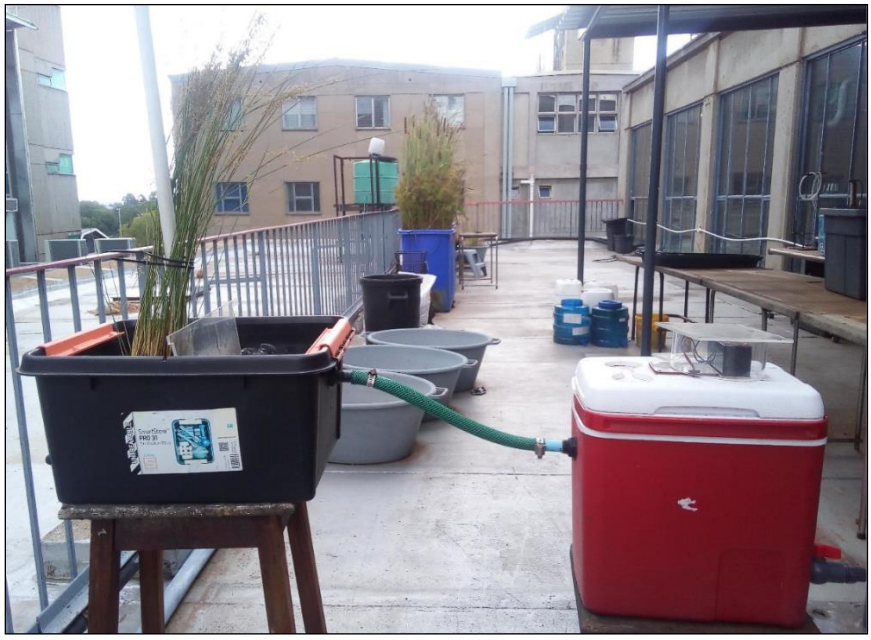

Figure 3.2: Portable Potable Water Treatment System

3.3.1 CW Chamber

The CW chamber is made up of half of a 32-litre plastic rectangular tub (Length = 50 cm, Breadth = 39 cm, Height = 26 cm), a clear sheet of Perspex (Thickness ≈ 0.5 cm, Height ≈ 23 cm, Breadth ≈ 37 cm) dividing the length of the tub into two sections, gravel rocks filling the tub to a height of approximately 18 cm and a *Juncus effusus* plant. This chamber has an outlet valve that is fitted 3 cm from the base of the tub to allow for the collection of water samples at that level. The CW chamber has a void fraction of ≈ 0.45. The CW chamber has a total volume of approximately 16 L, however, taking into account the void fraction and the height of the gravel filling, the CW chamber can hold a maximum feed water volume of approximately 8 L. The CW chamber is shown in Figure 3.2 and Figure 3.3. *Juncus effusus* was selected as the plant of choice as the plant thrives in sunny conditions and shallow standing water. These conditions can readily be found across South Africa. The plant is well distributed in the country as it can be found in five provinces, namely, LP, GP, MP, KZN and EC (Random Harvest, 2019). *Juncus effusus* is a hardy plant, it is pest resistant and it is unpalatable to herbivores (New Moon Nursery, 2019). *Juncus effusus* was thus chosen for this project based on these advantages.

3.3.2 Charcoal Chamber

The charcoal chamber is made up of the other half of the 32-litre plastic tub. The Perspex sheet forming the CW chamber separates the charcoal chamber from the CW chamber and sits ≈ 3 cm from the base of the plastic tub to allow for the flow of water from the CW chamber to the charcoal chamber. The charcoal chamber has a small outlet pipe (Length ≈ 3.5 cm, Diameter ≈ 1.5 cm) located 6 cm from the top of the tub where water from the charcoal chamber is collected. The charcoal chamber if filled with “braai” charcoal to a height of approximately 18 cm and has a void fraction of ≈ 0.5. The charcoal chamber has a total volume of approximately 16 L, however, taking into account the void fraction and the height of the charcoal filling, the chamber can hold a maximum water volume of approximately 8.78 L. The charcoal chamber is shown in Figure 3.2 and Figure 3.3.

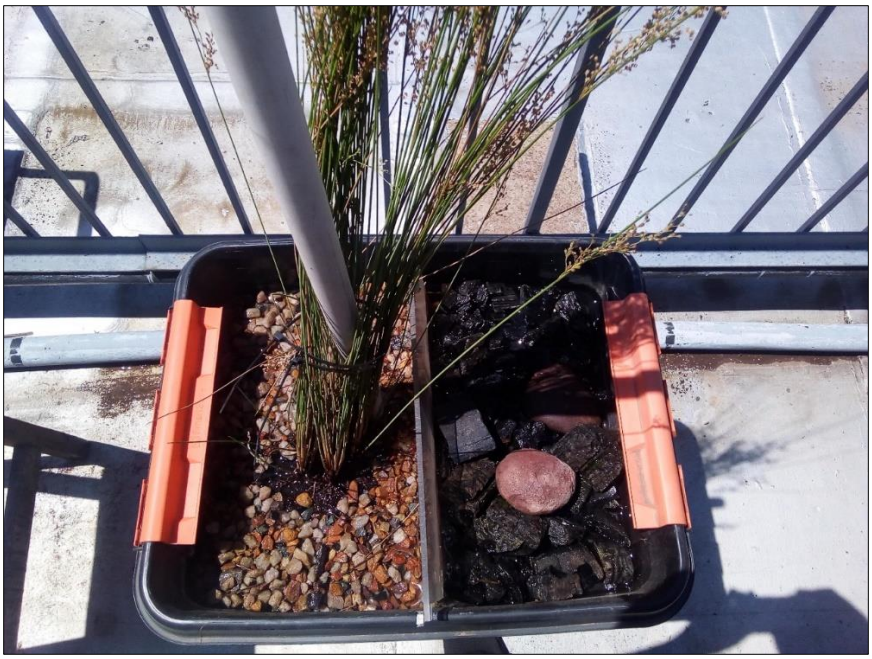

Figure 3.3: CW and Charcoal Chambers (Top-View)

3.3.3 UV Chamber

The UV chamber is made up of a 25-litre Cooler Box (Length = 44 cm, Breadth = 25 cm, Height = 38 cm) with an inlet pipe fitted on one end of the cooler box, 7.5 cm from the top of the cooler box. An outlet valve of diameter 2 cm was fitted to the bottom of the cooler box, 3 cm from the base to allow for outflow of water. A piece of hose 36 cm long (Diameter ≈ 1.7 cm) connects the outlet pipe of the charcoal chamber to the inlet pipe of the UV chamber to allow water to flow from the charcoal chamber to the UV chamber. The inward facing side of the cooler box lid has a UV LED Printed Circuit Board (PCB) attached to it with three LEDs (LG Innotek 278nm LED 1in1, Model Name: LEUVA66B00HF00) mounted on it. The PCB was fastened onto the inside of the Cooler box lid by means of 2 screws that drilled into the lid of the Cooler Box that pass through the corners of the UV PCB. On the top of the cooler box lid (outside the cooler box) sits a 12-watt rechargeable battery that serves to power the UV LEDs. The LEDs have a peak wavelength of 278 nm and a radiant flux of approximately 2.0 mW with dimensions L = 6.0 mm, W = 6.0 mm and H = 1.35 mm (LG Innotek, 2013). The UV chamber is shown in Figure 3.2 and in Figure 3.4, and the UV LED PCB is shown in Figure 3.5.

Figure 3.4: UV Chamber with UV LED PC Board

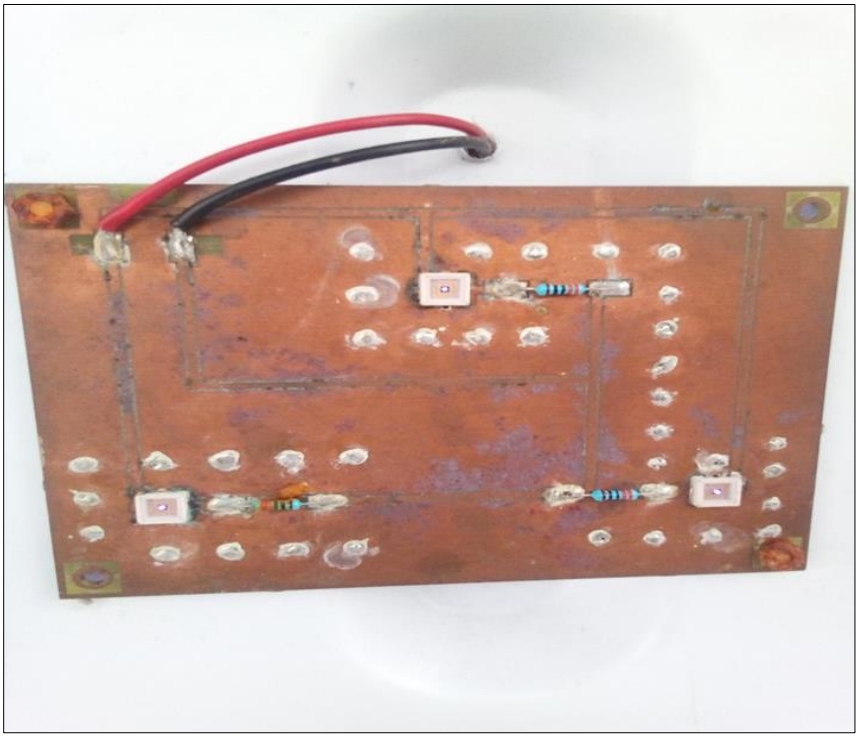

Figure 3.5: UV LED PCB

CHAPTER 4 – RESULTS AND DISCUSSION

4.1 RESULTS

The following results section consists of two sections. The first section is a presentation of the results obtained from the analysis of the feed water used to feed the PPWTS. The second section is a presentation of the results obtained from the analysis of all the water samples collected from the feed water as well as from the CW, charcoal and UV chambers on each sampling day. The second section provides a comparison of the results obtained to the WHO and SANS drinking water limits.

The PPWTS was fed with water from two different sources at different stages of the experimental process, namely River Water and Rainwater. The river water was collected from Fairlands Spruit, located in Fairland, Johannesburg, South Africa at the following coordinates: 26˚8'8" S, 27˚56'60" E, Elevation = 1560 m.

Stormwater runoff was used as the rainwater for this research. The stormwater was collected from the roof of the Richard Ward building at the University of the Witwatersrand in Braamfontein, Johannesburg, South Africa.

River water was used on three different occasions, namely the 4th to the 18th of June, the 4th to the 10th of July and the 17th to the 26th of July 2018. Rainwater was used on the 20th of June to the 3rd of July and again on the 12th and the 16th of July 2018. Rainwater was used as feed water to the system as the river from which water was collected for the purpose of feeding the PPWTS was impacted heavily by a sewage discharge that occurred between the 14th and the 20th of June 2018. Rainwater was used a second time as there were issues in accessing river water from the 12th to the 16th of July 2018. The Figures 4.1, 4.2 and 4.3 show the impact of the sewage discharge on the Fairlands Spruit River water used to feed the PPWTS.

Figure 4.1: Fairlands Spruit Sewage Impacted River Water - A

Figure 4.2: Fairlands Spruit Sewage Impacted River Water – B

Figure 4.3: Fairlands Spruit Sewage Impacted River Water - C

Use of the sewage impacted river water shown in Figure 4.1, 4.2 and 4.3 was avoided from the 20th of June to the 3rd of July 2018 as the impact of the sewage discharge on the river water was visibly high. The degree to which the river water was impacted by sewage was too high and could have negatively affected the capacity of the CW to treat water passing through the system. Using the sewage impacted river water as the feed water to the PPWTS between the 20th of June and the 3rd of July 2018 presented another risk in that the degree of the sewage impact on the river water may have led to the destruction of the *Juncus effusus* plant constituting the CW chamber. Rainwater collected from the roof of the Richard Ward Building at the University of the Witwatersrand was used on the 20th of June to the 3rd of July 2018 as the feed water to the system to allow the Fairlands Spruit river time to recover from impact of the sewage discharge that took place between the 14th and the 20th of June 2018.

In the following graphs illustrating the results obtained from analysing water collected from the PPWTS, a rectangle with a thick, dark border has been used to mark the dates on which rainwater was used as the feed water to the system (20th June to 3rd July and 12th to 16th July 2018). Tables containing the experimental data used to plot the following graphs can be accessed in Appendix A. The weather did not appear to have a considerable impact on the results obtained from analysis

of the water collected from the PPWTS, however, details regarding the weather on each sampling day can be viewed in Appendix B. For the purpose of this research project, the removal of aluminium, lead and manganese was addressed to observe the effects of the PPWTS on heavy metals. These metals were selected based on a preliminary Inductively Coupled Plasma Mass Spectrometry (ICP-MS) analysis that was conducted for samples of both river water and rain water. The concentrations of aluminium, lead and manganese were relatively high compared to the concentrations of other metals identified in the water samples. These metals were also selected as their consumption can result in adverse health effects in humans. The results of this preliminary ICP-MS analysis can be viewed in Table C1 Appendix C.

4.1.1 Feed Water Quality Results

4.1.1.1 pH and E.C.

The pH and E.C. of the feed river and the rain water are shown in Figure 4.4.

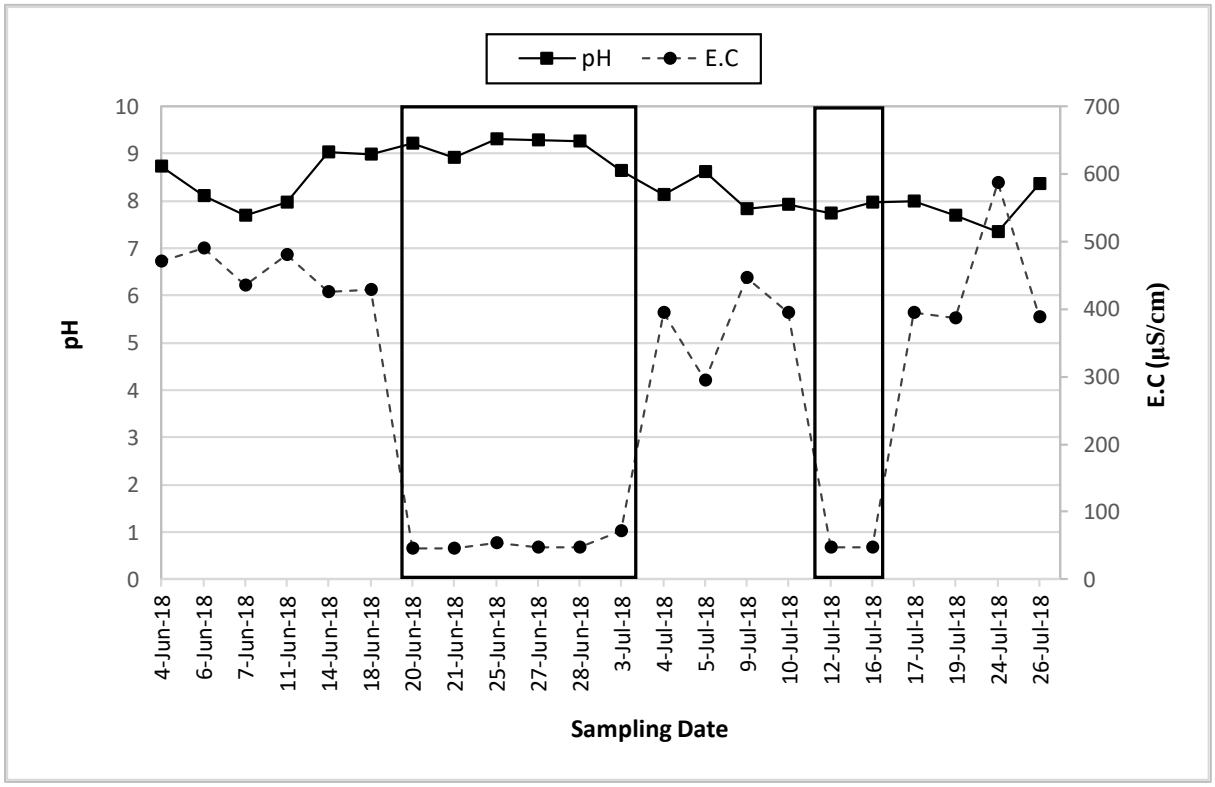

Figure 4.4: pH and E.C. of Feed Water vs. Sampling Date (Blocks represent rain water)

pH

The pH of the feed water used for this research had a value of between 7 and 9.5. The lowest pH was 7.34 and the highest pH was 9.31. The highest pH readings in Figure 4.4 are those associated with the rainwater fed to the system from 20 June to 3 July 2018. The pH of the rainwater that was used as feed to the water treatment system was considerably higher than that of normal rainwater which is approximately 5.6 (Dowdey, 2018). This may have been caused by the gravel and the soil that occupied the rainwater collection tub as soil and gravel are known to increase the pH of water (Flickerton, 2017). The pH values associated with river water in Figure 4.4 (4 June to 18 June; 4 July to 10 July and 17 July to 26 July) are noticeably lower relative to those of the rainwater. According to Rand Water (2018) however, the pH of healthy rivers ranges between 6.5 and 8.5. Approximately 71.43 % (10 out of 14) of the river water pH values were within this pH range. Considering the river had been impacted by a discharge of sewage between the 15th to the 20th of June 2018, the river cannot be considered healthy and the experimental data gathered in relation to this river water is not reflective of a healthy river. Despite this however, the pH profile of the river water in Figure 4.4 coincides well with what Rand Water (2018) considers a healthy river with respect to pH.

E.C.

As is evident in Figure 4.4 the E.C. of the feed river water ranged between the 300 – 500 µS/cm range, with the only points outside this range being the E.C. readings associated with the river water fed to the water treatment system on the 5th of July and the 24th of July 2018. As shown in Figure 4.4, the river water had a minimum E.C. of 295 µS/cm (5 July 2018) and a maximum E.C. of 587 µS/cm (24 July 2018), while the maximum E.C. for rainwater was 72.2 µS/cm (3 July 2018). The results shown in Figure 4.4 provide a view of how the E.C. of river water was different to that of rainwater. According to the Grand River Conservation Authority (2018), ground water can carry salts into river water and thereby raise the conductivity of the river water. Furthermore, the nature of the discharges that are made into rivers influence the E.C. of river water (U.S. EPA, 2012). The highest E.C. reading obtained for river water is shown in Figure 4.4 as 587 µS/cm. This value is

reflective the E.C. of the river water that was fed to the water treatment system on the 24th of July 2018. This high E.C. reading may be the result of the sewage discharge that was made into the river between the 15th to the 20th of June 2018. Rainwater, however, is relatively free of impurities, save for the substances picked up by the rain water in the atmosphere (World Health Organization, 2011). This offers a reason as to why the rainwater E.C. readings are considerably lower relative to those given by the river water.

4.1.1.2 Phosphates and Nitrates

Phosphate and nitrate concentrations for the river and the rain feed water are shown in Figure 4.5.

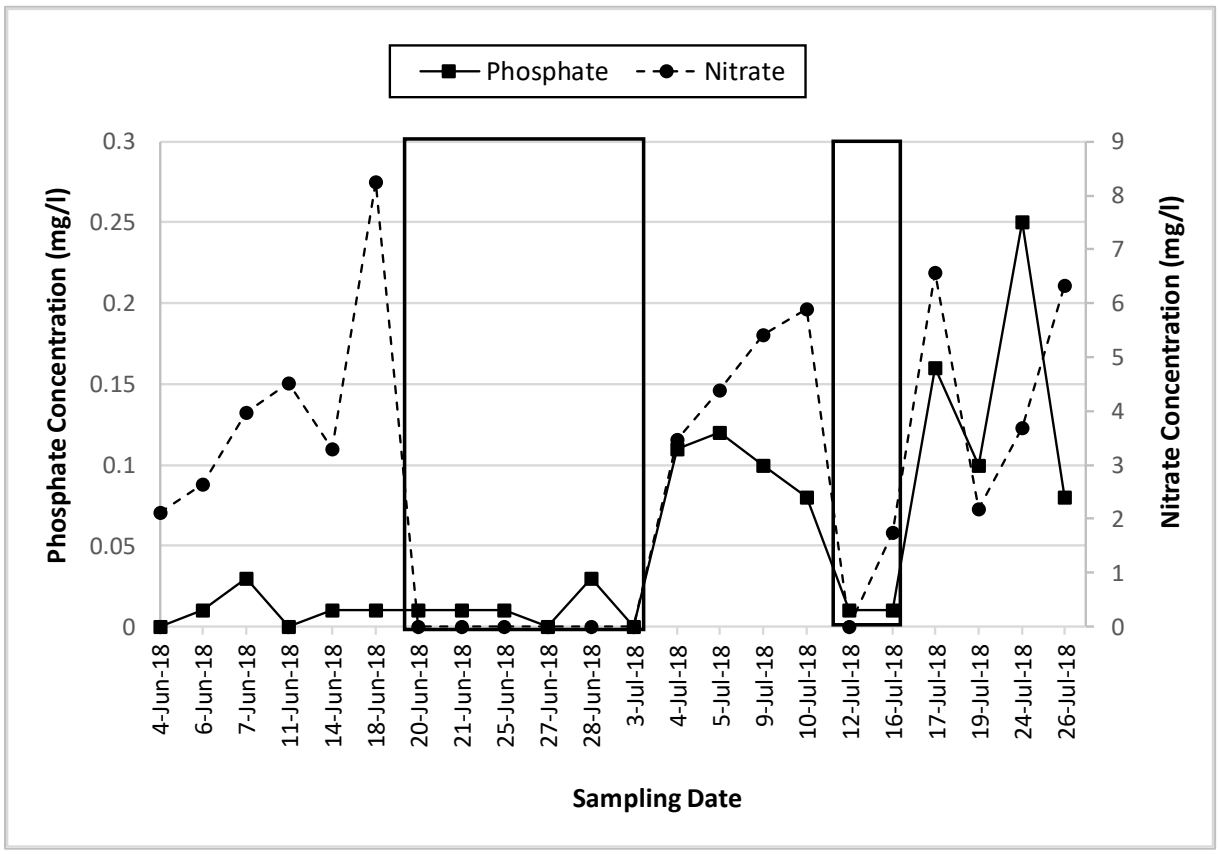

Figure 4.5: Phosphate and Nitrate Concentration of Feed Water vs. Sampling Date (Blocks represent rain water)

Phosphates

The concentration of phosphates in the river water used for this research was unexpectedly low with the highest concentration of phosphates shown in Figure 4.5 as 0.25 mg/L. The concentration of phosphates in the feed water (both river water and rainwater) was below 0.05 mg/L until a new batch of river feed water was introduced to the system on the 4th of July 2018, after the Fairland' Spruit experienced a discharge of sewage into its water. The new river water was visibly impacted by sewage and as such, the phosphate concentration of this water was slightly higher. A phosphate concentration of 0.11 mg/L was recorded on the 4th of July 2018 from analysis of the sewage impacted river water. From the 4th of July onwards, no river water samples had a phosphate concentration of below 0.05 mg/L. The river water had a maximum phosphate reading of 0.25 mg/L on the 24th of July 2018.

Nitrates

In Figure 4.5 it is can be seen that nitrate concentrations of river water river water ranged between 2 mg/L to 7 mg/L, save for the maximum feed nitrate concentration on the 18th of July 2018 with a value of 8.26 mg/L. The rainwater feed samples all returned a nitrate reading of 0 mg/L, except for the rainwater feed used on the 16th of July 2018 of concentration 1.74 mg/L. The nitrate concentrations of the river water used on the 9th, 10th, 17th and 26th of July 2018 were higher than 5 mg/L, which is not seen in the feed river water used before the 20th of July 2018, save for the nitrate concentration of the feed water used on 18th of July 2018. This may be the result of the river having been impacted by sewage which most likely resulted in an increased nitrate concentration.

4.1.1.3 COD and TOC

The COD and TOC of the feed river and rain water is shown in Figure 4.6.

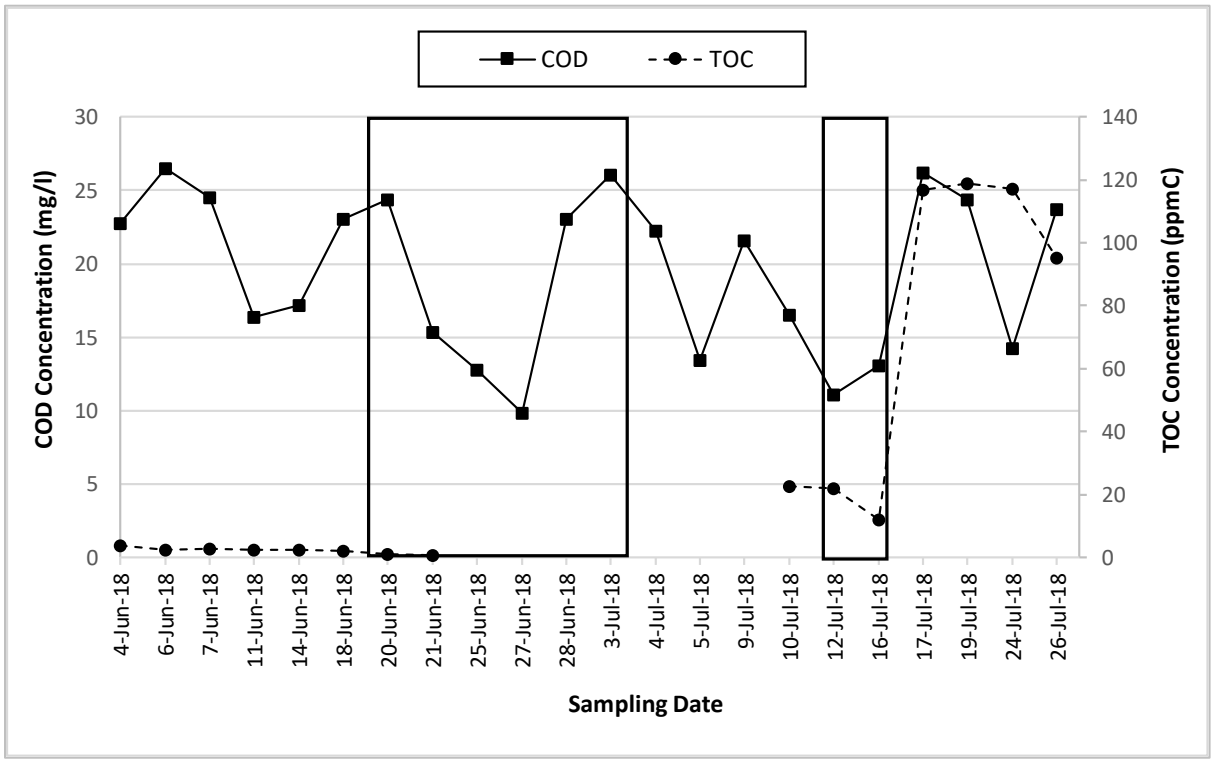

Figure 4.6: COD and TOC Concentration of Feed Water vs. Sampling Date (Blocks represent rain water)

COD

Figure 4.6 shows the COD concentrations of the feed water to the system. The COD concentration of the feed water was between the minimum and maximum COD concentrations of 9.80 mg/L and 26.52 mg/L respectively. This was the case for both the river water and the rainwater. The rainwater collection tub contained soil, pebbles and various suspended solids, along with the collected rainwater. This may be the reason rainwater had comparable COD results to the river water. There is however, a noticeable drop in the COD concentration from the 20th to the 27th of June 2018, which relates to the rainwater that was fed to the system during that period. The COD of the feed water fluctuated with time; however, the majority of the feed water COD concentrations ranged between the 10 to 25 mg/L.

TOC

As is evident in Figure 4.6, feed water TOC concentrations were initially low, though the TOC of river water (4 to 18 of June 2018) was slightly higher than that of rainwater (20 and 21 June 2018). TOC results for both river water (10 July 2018) and rainwater (12 and 16 July 2018) were considerably higher from the 10th of July 2018 as can be seen in Figure 4.6. TOC results for water samples collected from the 25th of June to the 9th of July 2018 could not be recorded as this sample batch was corrupted during the TOC analysis preparation process. Water samples collected from the 25th of June to the 9th of July 2018 returned experimental TOC values close, or equal to, zero, which is inconsistent with prior TOC findings. The TOC readings of the feed samples collected from the 17th to the 26th of July 2018 were considered invalid and an inaccurate representation of the TOC profile of the feed water as they were abnormally high. It is suspected that a contamination of the said samples occurred due to a rupture of the internal structures within the deionized water tank; contamination of the deionized water used to rinse the TOC vials; or carbon contamination of the sample water before analysis. The most reliable representation of the TOC profile of the feed water used in this research is represented by the results obtained from the 4th to the 21st of June 2018 and from the 10th to the 16th of July 2018.

4.1.1.4 Total Coliforms and *E. coli*

Figure 4.7 shows the feed river, and rainwater total coliform and *E. coli* most probable number (MPN).

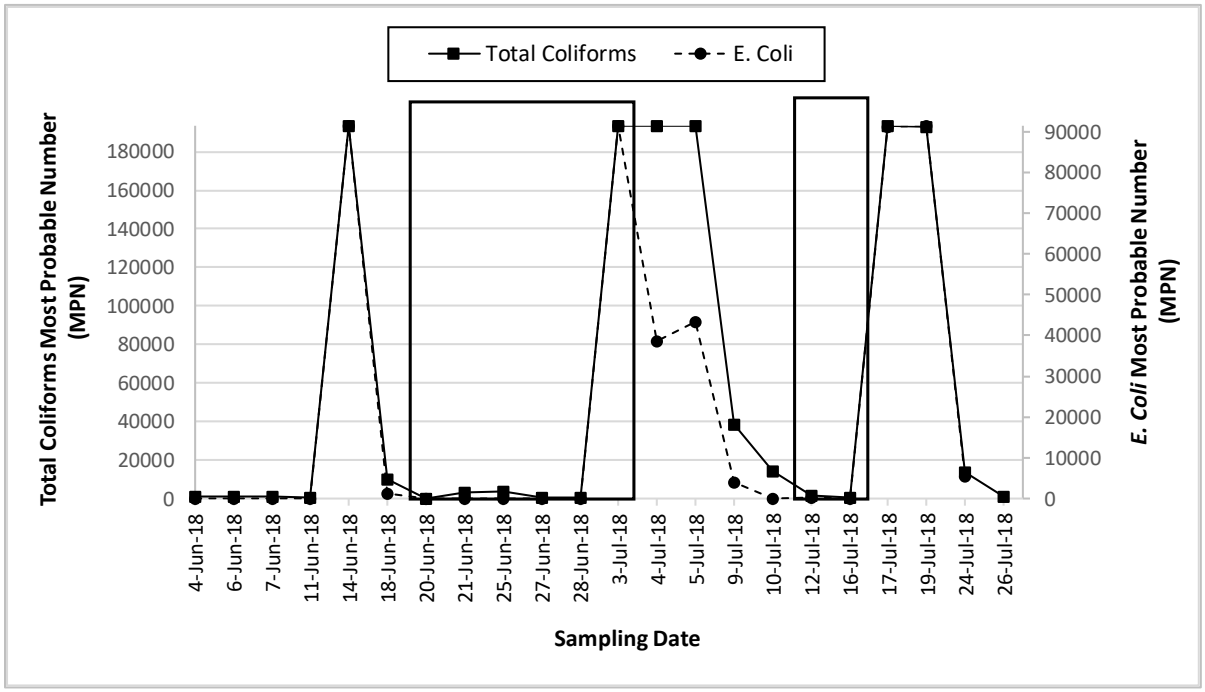

Figure 4.7: Total Coliforms and *E. coli* Most Probable Number (MPN) of Feed Water vs. Sampling Date (Blocks represent rain water)

Total Coliforms

Figure 4.7 depicts the total coliform most probable number (MPN) of the feed water to the water treatment system. The five data points associated with the 14th of June 2018 and the 3rd, 4th, 5th and 17th of July 2018 represent water samples that had an MPN for total coliforms that was beyond the reading limits of the IDEXX Colilert-18 test kit. A maximum MPN value of 193500 was assigned to total coliform readings exceeding the limits of the test kit. Dilutions were performed on the feed water for testing purposes as the water used to feed the system resulted in MPN values that were beyond the limits of the IDEXX Colilert-18 test kit. Dilutions were kept constant from the 20th of June up to the 5th of July 2018. It was evident however, that the new river water fed to the system from the 4th of July 2018 was impacted by sewage to a great degree, relative to the feed water that had been used before the 4th of July 2018. This was apparent in that MPN number related to the

feed water fed to the system on the 4th and 5th of July 2018 gave MPN readings that exceeded the limits of the IDEXX Colilert-18 test kit. This increase in total coliform MPN was the result of the sewage discharge that was made into the river between the 14th to the 20th of June 2018. Overall, it is clear that the river water used as the feed water was sewage impacted at the time it was collected and used. According to (World Health Organization, 2011), rainwater is relatively free of impurities and as such, rainwater was expected to give a low MPN result. The MPN results for total coliforms in rainwater were very low. This is evident in the results obtained between the 18th and the 28th of June 2018 and on the 12th and the 16th of July 2018 when rainwater was used as the feed to the system.

E. coli

As can be seen in Figure 4.7, the trend of the feed water *E. coli* MPN curve resembles that of the total coliforms however, the MPN for *E. coli* is substantially lower. The maximum MPN reading for *E. coli* is set at 91500 for graphical representation purposes as some water samples that were analysed using the IDEXX Colilert-18 testing kit produced results that exceeded the limits of the test kit. The feed water *E. coli* maximum is depicted in Figure 4.7 by the data points associated with the 14th of June, the 3rd of July and the 19th of July 2018. The rainwater feed produced an *E. coli* MPN reading of 0 from the 20th to the 28th of June and again on the 16th of July 2018 when rainwater was used. The results shown in Figure 4.7 confirm the presence of *E. coli* in the river water.

4.1.1.5 Turbidity

The turbidity of the feed river and rain water is shown in Figure 4.8.

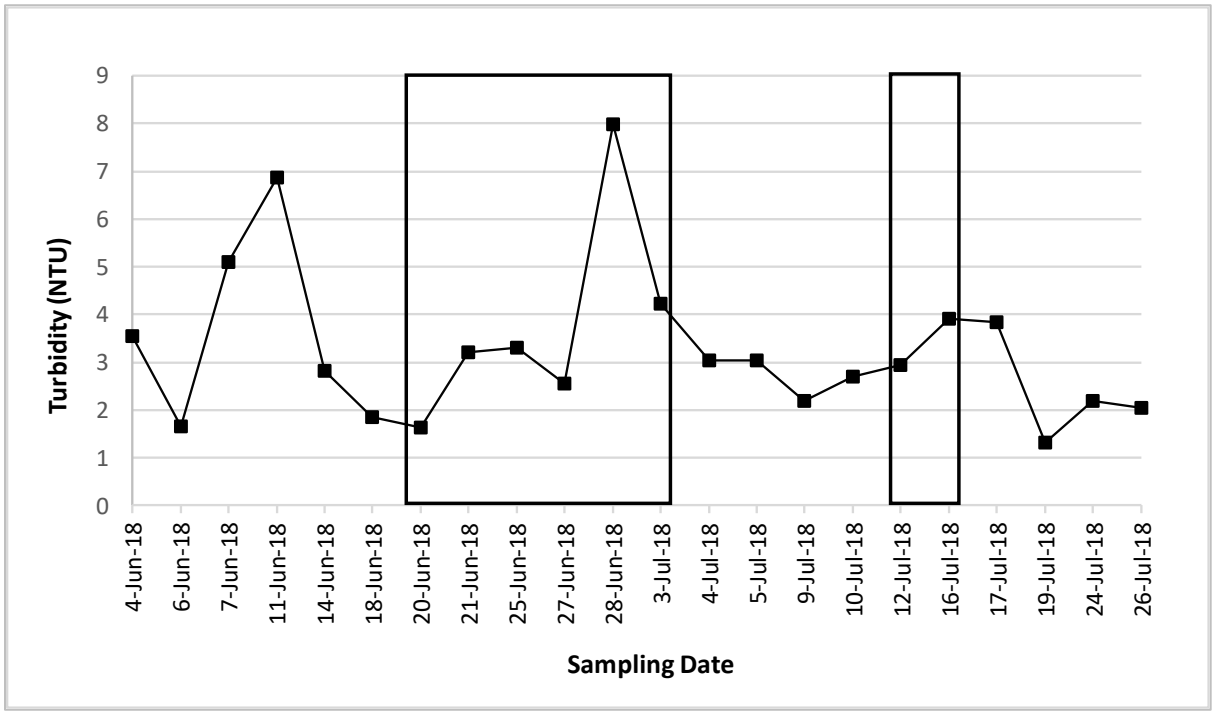

Figure 4.8: Turbidity of Feed Water vs. Sampling Date (Blocks represent rain water)

As is evident in Figure 4.8 the turbidity of the feed water (both river and rainwater) ranged between 1 to 8 NTU, with a minimum turbidity of 1.32 NTU (19 July 2018) and a maximum turbidity of 7.98 NTU (28 June 2018). The maximum turbidity reading was that of the rainwater fed to the treatment system on the 28th of June 2018. This can be attributed to the fact that, between the 18th and the 28th of June 2018, there was no rainfall in Braamfontein, Johannesburg (Gauteng, South Africa) and because the rainwater was collected in a container that had soil particles, pebbles and various suspended solids, using more of the top surface rainwater increased the ratio of solid particles to rainwater in the tub. This may have caused an increase in the turbidity of the rainwater in the tub. The majority of the feed water turbidity results, however, were within the 1 to 4 NTU range. From the results shown in Figure 4.8 the turbidity of both the rainwater and river water was relatively low.

4.1.1.6 Sulfates

The river water and rainwater feed Sulfate concentration is shown in Figure 4.9.

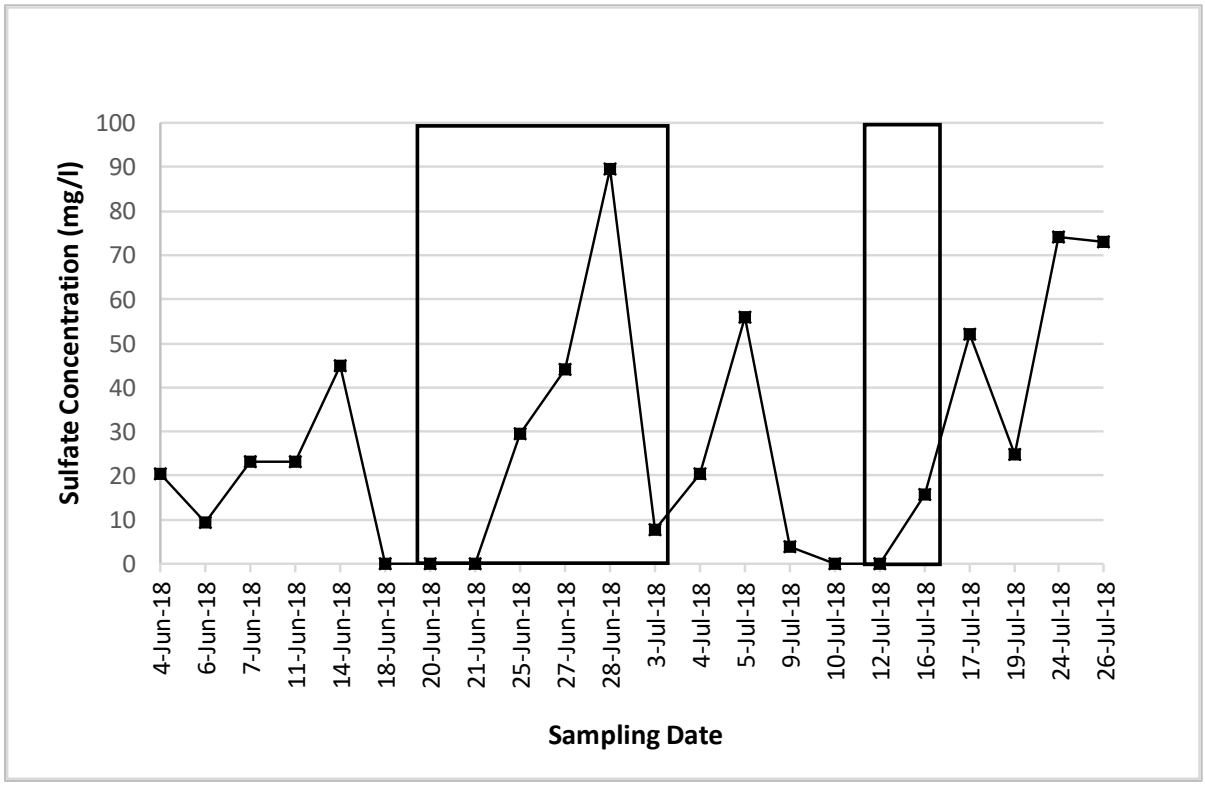

Figure 4.9: Sulfate Concentration in Feed Water vs. Sampling Date (Blocks represent rain water)

In Figure 4.9 above the Sulfate concentration of the feed water ranged between 0 to 90 mg/L. The highest Sulfate concentration was given by the rainwater that was fed to the system on the 28th of June 2018. From the 25th of June 2018, the rainwater Sulfate concentration began to rise. This may have been the result of having used up a substantial volume of the surface water in the rainwater collection tub, resulting in the use of more turbid rainwater with a high level of SS as the feed to the PPWTS. The high Sulfate concentration of the rainwater on the 28th of June 2018 coincides with a high turbidity reading of 7.98 NTU (Figure 4.8), an increase in the COD of the feed rainwater (Figure 4.6) and a small increase in the phosphate concentration (Figure 4.5).

4.1.1.7 Heavy Metals: Aluminium and Manganese

The aluminium and manganese concentrations in the river and rain water feed are shown in Figure 4.10.

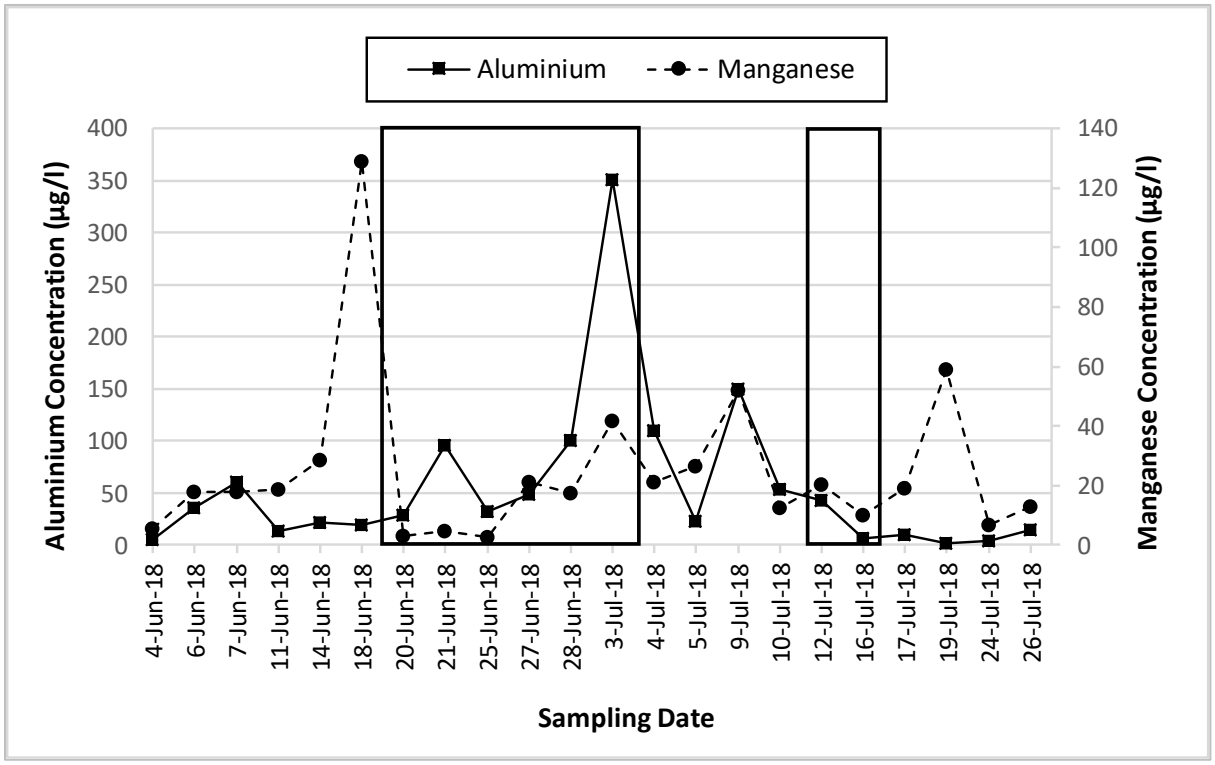

Figure 4.10: Aluminium and Manganese Concentration of Feed Water vs. Sampling Date (Blocks represent rain water)

Aluminium

As can be seen in Figure 4.10, the aluminium concentration of the feed water to the PPWTS ranged between 1 to 355 µg/L. The minimum and maximum feed water aluminium concentrations were 1.92 µg/L (19 July 2018) and 351 µg/L (3 July 2018). All of the feed water aluminium concentrations in Figure 4.10 have a value of 150 µg/L or less, with the only exception being the maximum feed aluminium concentration of 351 µg/L. The highest aluminium feed water concentration is associated with the rainwater feed on the 3rd of July 2018. This may be as a result of the soil and SS in the rainwater collection tub that may have contributed to the aluminium content of the rainwater. The leaching of heavy metals from roof building materials may have also been a contributing factor in the aluminium concentration on the 3rd of July 2018. Apart from the

maximum aluminium concentration value of 351 µg/L, the river and rainwater feed water concentrations are well below the SANS Operational Risk aluminium concentration limit of 300 µg/L (Vinlab, 2016).

Manganese

As is evident in Figure 4.10, the feed water manganese concentration ranged between 3 to 130 µg/L. The lowest manganese concentration was 3.12 µg/L (20 June 2018) and the highest was 129 µg/L (18 June 2018). The majority of the manganese concentration values have a concentration of below the 60 µg/L save for the manganese concentration associated with the 18th of June 2018. The average concentration of the feed river water (30.47 µg/L) is higher than that of the rainwater feed water (15.18 µg/L). The three highest manganese concentrations (18 June, 9 July and 19 July 2018) are values associated with river water fed to the PPWTS. All the manganese feed water concentration values in Figure 4.10 are below the SANS Health Risk concentration limit 400 µg/L (Vinlab, 2016).

4.1.1.8 Heavy Metals: Lead

The lead concentration of the feed river and rain water is shown in Figure 4.11.

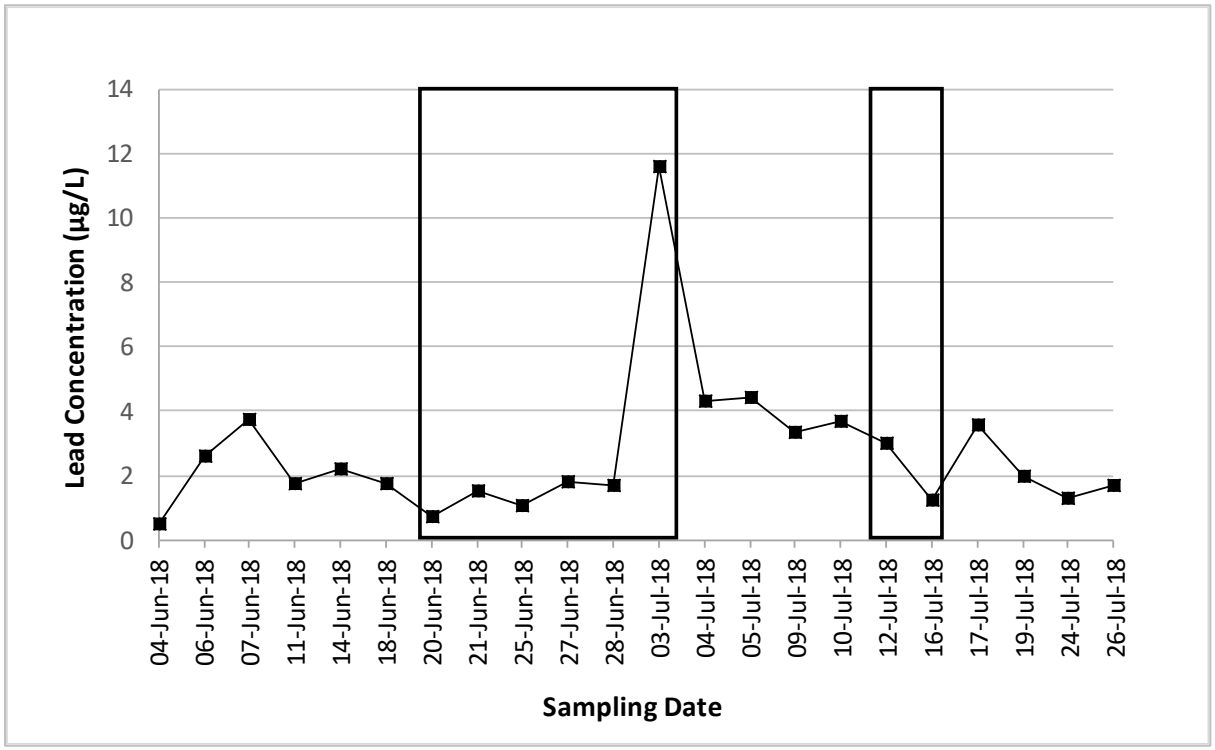

Figure 4.11: Lead Concentration of Feed Water vs. Sampling Date (Blocks represent rain water)

The lead concentration of the feed water used for this research had a value of between 0 and 12 µg/L. The lowest feed water lead concentration was 0.48 µg/L (4 June 2018) and the highest concentration obtained was 11.6 µg/L (3 July 2018). The majority of the feed water lead concentration values are of a concentration below 5 µg/L. The only exception is the concentration of the rainwater feed on the 3rd of July 2018, which, according to Figure 4.11, had a concentration of 11.6 µg/L. This spike in the lead feed water concentration coincides with the spikes seen on the 3rd of July 2018 for total coliforms, *E. coli* and aluminium. The COD feed water concentration is also particularly high on the 3rd of July 2018. The lead feed water concentrations obtained during this research have concentrations that are below the SANS and WHO drinkable water limit of 10 µg/L (Vinlab, 2016; World Health Organization, 2011).

4.1.2 Portable Potable Water Treatment System Removal Results

4.1.2.1 pH

The pH values of the feed water and the water in the CW, charcoal and UV chambers are shown in Figure 4.12.

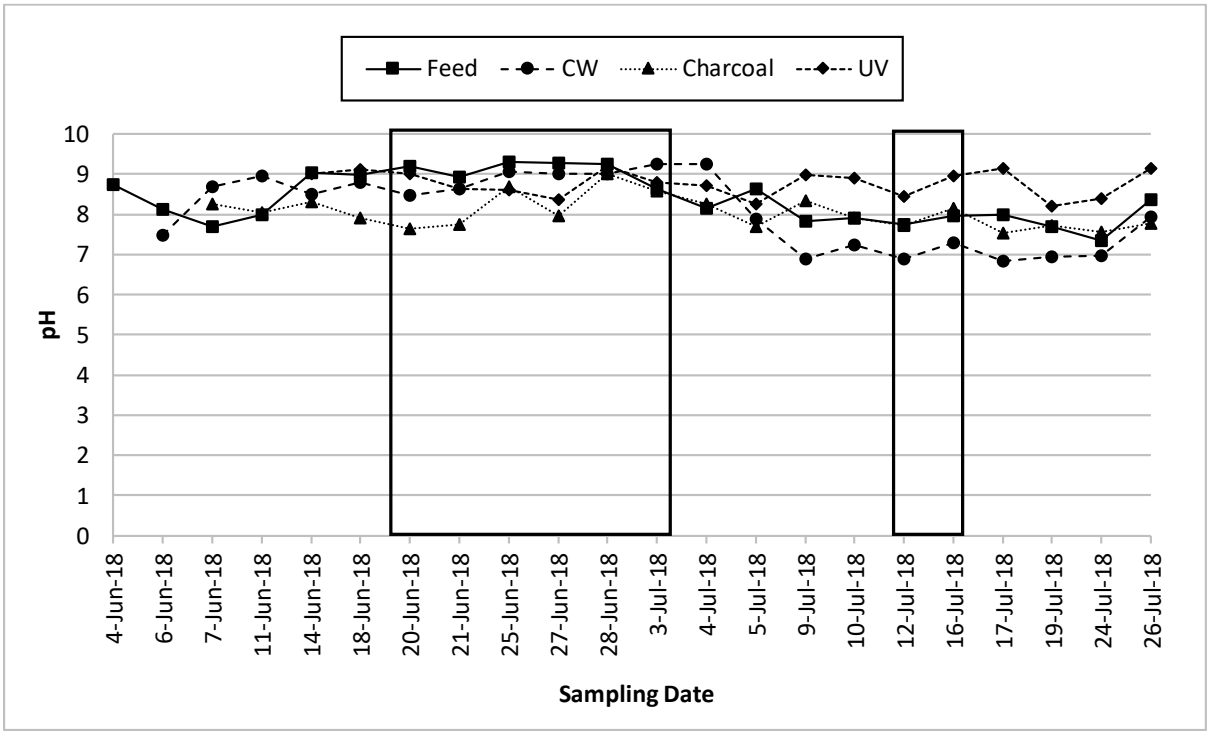

Figure 4.12: pH of Water in the System vs. Sampling Date (Blocks represent rain water)

Figure 4.12 shows the pH values of the water fed to the water treatment system as well as the pH of the water that occupied the CW, charcoal and UV chambers of the system. The pH of the water in the system generally fell between the 7 to 9 pH range. The pH of the water collected from the charcoal chamber was well within the 7 to 9 pH range, with only one charcoal water sample collected on the 28th of June 2018 having a pH of 9.01. Multiple water samples collected from the feed as well as from the CW and the UV chambers had a pH above 9 while only CW water samples had a pH below 7, from the 9th to the 24th of July 2018. Table 2 shows the percentage decrease in the average pH across the PPWTS relative to the feed.

Table 2: Decrease in Average pH across the Potable Water Treatment System

	Average Value (pH)
Feed	8.40
CW	8.10
Charcoal	8.04
UV	8.77

It is evident from the data in Table 2 that the average pH of the water in the system was lowest in the charcoal chamber. The average pH of the water in the system increased beyond that of the feed water in the UV chamber. The outlet was always greater than the feed.

4.1.2.2 E.C.

The E.C. of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.13.

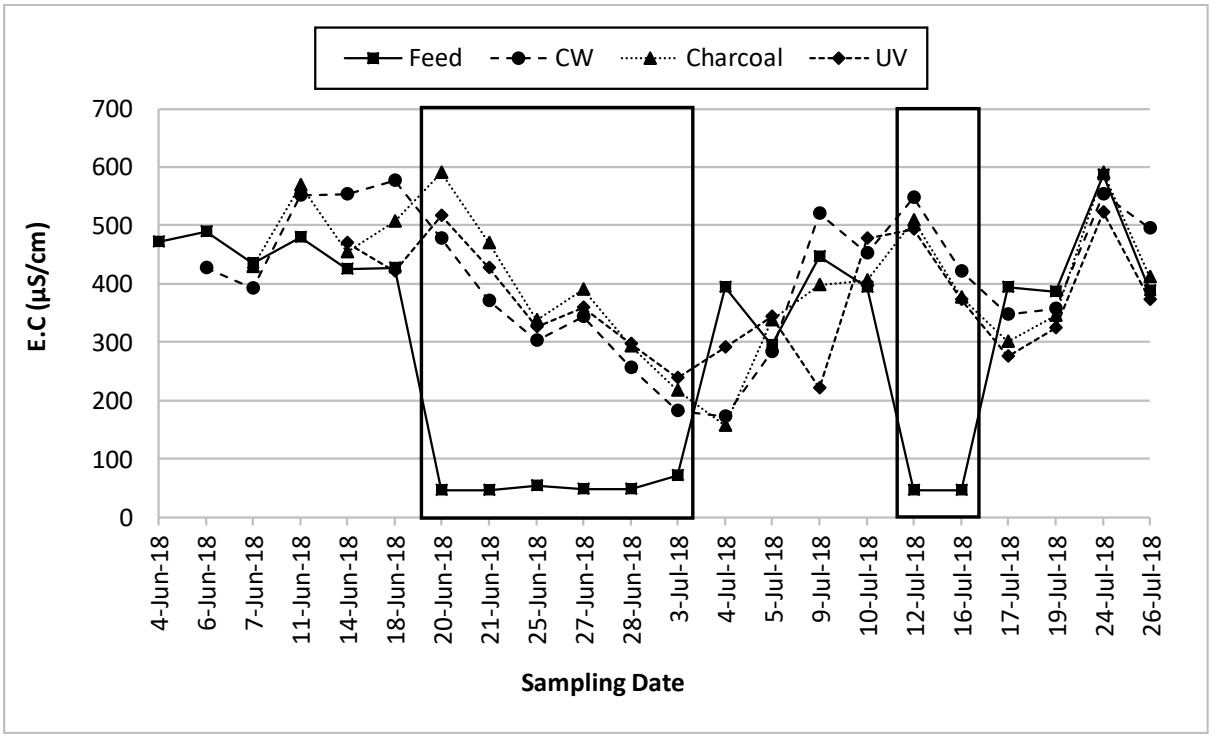

Figure 4.13: E.C. of Water in the System vs. Sampling Date (Blocks represent rain water)

The E.C. of the water passed through the water treatment system fell between the 100 – 600 µS/cm range, save for the rainwater feed (20 June to 3 July and 12 to 16 July 2018). The CW and charcoal chambers had the effect of increasing the E.C. of the water passing through the system as the water collected from the CW and charcoal chambers had the highest number of points situated above the 500 µS/cm mark. The only time the E.C. of the feed and UV water was above 500 µS/cm was on the 24th of July 2018. Water from the CW and charcoal chambers may have had a higher E.C. in general as evapotranspiration would have concentrated any ionic species present. Water from the CW chamber moved directly into the charcoal chamber, carrying through the salts and minerals from the CW chamber into the charcoal chamber. Looking at the dates on which rainwater was used as feed water, it is evident that as more rainwater was fed to the water treatment system, the E.C. of the water collected from the CW and charcoal chamber decreased. In both instances when rainwater was fed to the system, an accompanying downward trend in the E.C. values of the water in the system chambers was observed. This phenomenon is the result of a rainwater feed with a low E.C. and thus a low concentration of salts and/or minerals. The rainwater fed to the system appeared to have been collecting salts and minerals from the CW and charcoal chambers in the system and flushing them out of the system, thus reducing the E.C. of the water passing through the system. Table 3 shows the percentage decrease in the E.C. across the chambers of PPWTS.

Table 3: Percentage Decrease in Average E.C. across the Potable Water Treatment System

	Average Value (E.C. - µS/cm)	% Decrease / %Increase*
Feed	292	-
CW	409	40.07*
Charcoal	404	38.36*
UV	375	28.42*

The results in Table 3 show the percentage by which the average E.C. of the water fed to the system increased relative to the feed. The E.C. of the water was higher than that of the feed in all three chambers (CW, charcoal and UV). The highest increase in average E.C. occurred in the CW chamber.

4.1.2.3 Turbidity

The turbidity of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.14.

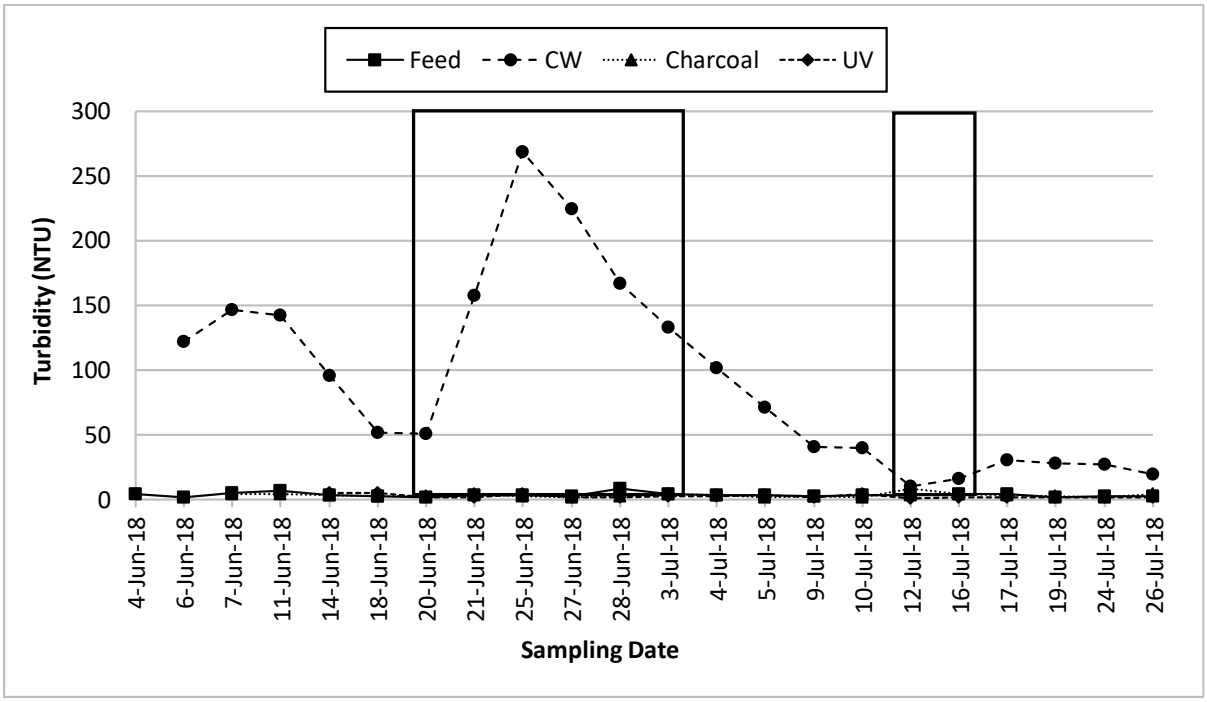

Figure 4.14: Turbidity of Water in the System vs. Sampling Date (Blocks represent rain water)

As is evident in Figure 4.14, the turbidity of the feed water and that of the water samples collected from the charcoal and UV chambers was low, with no turbidity readings of more than 8 NTU. The water collected from the CW chamber had the highest average turbidity, with a maximum turbidity of 268 NTU on the 25th of June 2018. The turbidity of the water collected from the CW chamber began to decrease after this maximum value as more feed water was passed through the system over time. The turbidity of the water collected from the CW chamber may have increased from the 20th of June 2018 due to having moved the PPWTS shortly after experimental work was commenced. The system was moved to a sheltered area to keep the UV chamber battery pack from being rained on. The system was also moved to prevent any sun damage that would negatively affect the system's components, as a result of having the PPWTS exposed to sunlight indefinitely. This movement may have caused the loosening of the soil particles in the CW chamber, thereby increasing the turbidity of the water collected from the system. The high turbidity of the sample

water collected from the CW chamber, however, did not cause a rise the turbidity of the water collected from the subsequent chambers of the system as the CW chamber served as a filter, lowering the turbidity of the water that entered the charcoal and UV chambers to turbidity readings of below 5 NTU. The only anomaly with regards to turbidity was the water sample collected from the charcoal chamber on the 12th of July 2018. This water sample returned a turbidity reading of 7.90 NTU. Table 4 shows the percentage decrease in the turbidity of the feed water across the three chambers of the PPWTS.

Table 4: Percentage Decrease in Average Turbidity across the Portable Potable Water Treatment System

	Average Value (Turbidity - NTU)	% Decrease / % Increase*
Feed	3.27	-
CW	92.24	2721*
Charcoal	2.86	12.54
UV	2.09	36.09

As is evident in Table 4, the turbidity of the water in the PPWTS increased substantially in the CW chamber but decreased as it passed through the CW to the charcoal and UV chambers. The average turbidity in the water treatment system was lowest in the UV chamber and this was lower than the feed.

4.1.2.4 Phosphates

The phosphate Concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.15.

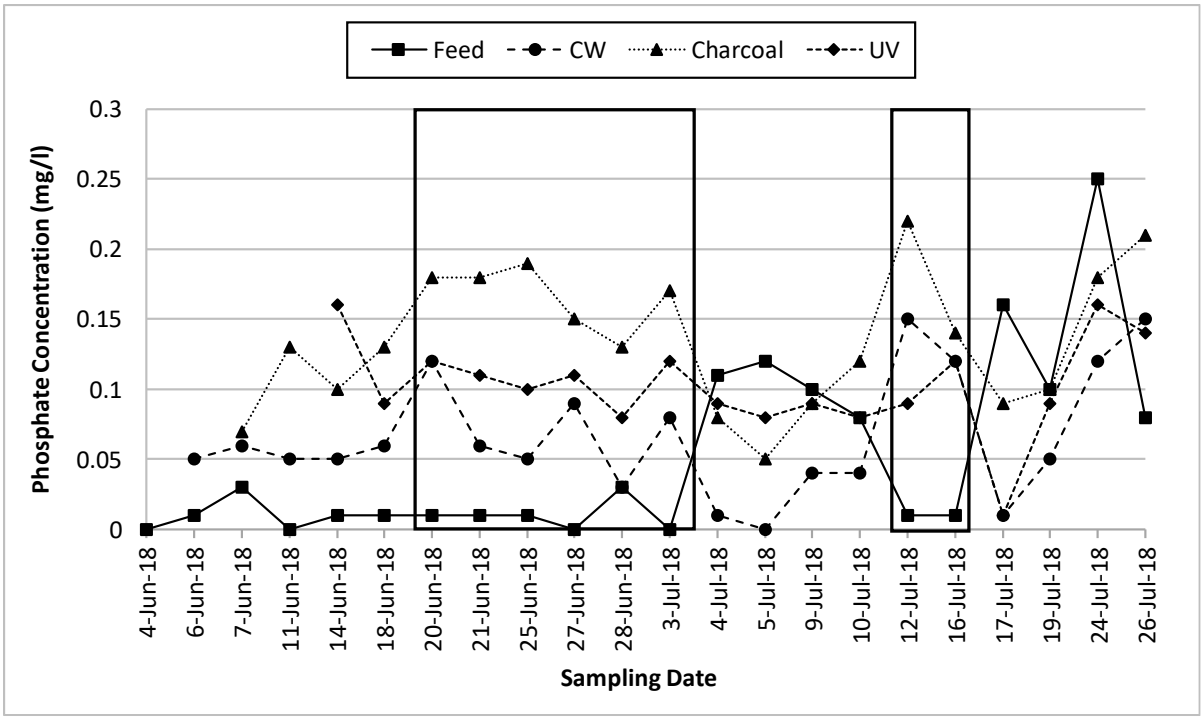

Figure 4.15: Phosphate Concentration of Water in the System vs. Sampling Date (Blocks represent rain water)

From Figure 4.15 it is evident that the phosphate concentration in the feed water and in the water treatment system was very low. The maximum phosphate concentration obtained from the system was 0.25 mg/L which was the concentration of the feed on the 24th of July 2018. The feed phosphate concentrations remained well below 0.05 mg/L before the new feed river water was introduced to the treatment system on the 4th of July 2018. The new river water was impacted by sewage and thus the concentration of this feed water increased to above 0.05 mg/L. The phosphate concentrations of the water samples collected from the CW, charcoal and UV chambers were higher than that of the feed water. This is apparent when looking at the phosphate concentration profile of the system from the 4th of June to the 3rd of July 2018. The phosphate concentration of the water collected from the charcoal chamber did not drop to below 0.05 mg/L while only one sample point associated with the UV chamber water profile had a concentration of below the 0.05 mg/L (17 July 2018). The phosphate concentration is also highest in the charcoal chamber when

river water was introduced to the system between the 20th of June and the 3rd of July 2018. As is evident in Figure 4.15 above, the phosphate concentration was lowest in the feed water overall, which means the system itself caused an increase in the phosphate concentration of the water that was passed through it. The following table shows the percentage removal rates of phosphates in the PPWTS.

Table 5: Percentage Removal of Average Phosphate Concentration across the Portable Potable Water Treatment System

	Average Value (Phosphates - mg/L)	% Removal / % Increase*
Feed	0.052	-
CW	0.066	26.92*
Charcoal	0.14	169*
UV	0.10	92.31*

As can be seen in Table 5, the phosphate concentration in the water fed to the PPWTS increased in the chambers of system relative to the concentration of the feed water. The highest percentage increase in the average phosphate concentration occurred in the charcoal chamber. This indicates that the system had no remedial capacity on phosphate, and due to evapotranspiration may have actually increased the concentration. The levels of phosphate, however, remained in orders of magnitude below the discharge limit of 10 mg/L as given in the General Authorisations (Department of Water Affairs and Forestry, 2003).

4.1.2.5 Nitrates

The nitrate Concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.16.

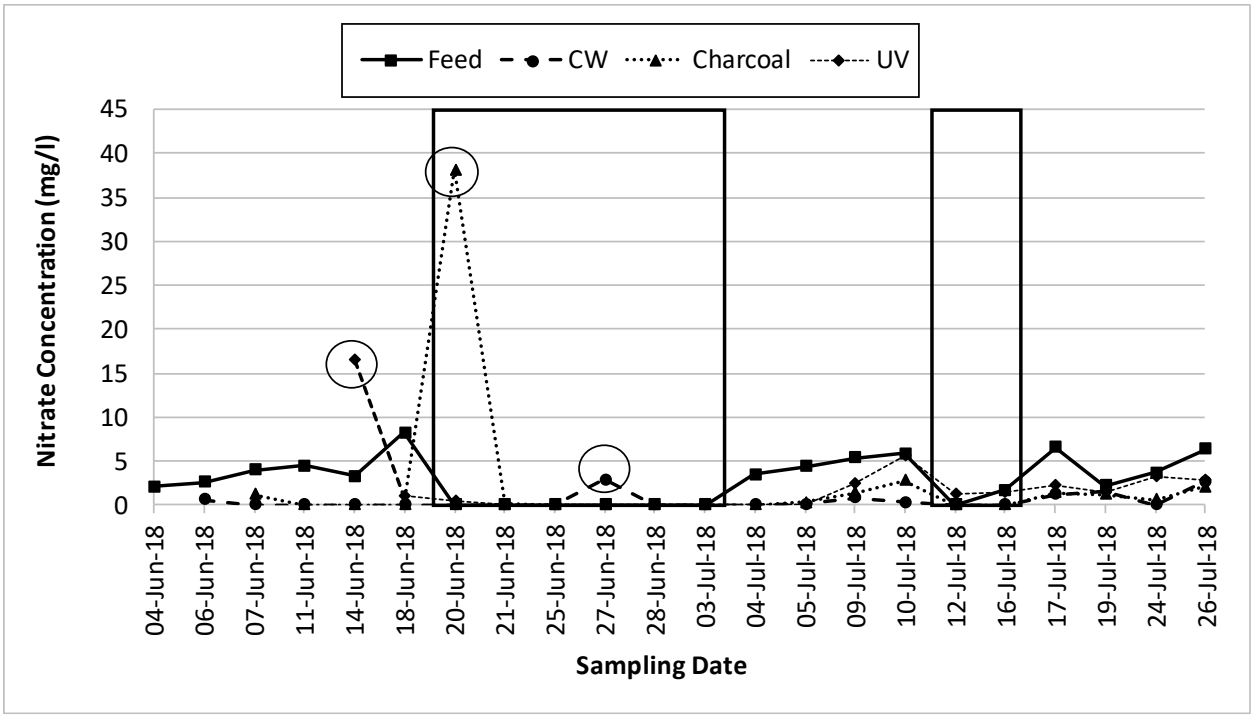

Figure 4.16: Nitrate Concentration of Water in System vs. Sampling Date (Blocks represent rain water)

From the nitrate concentration profile of the system in Figure 4.16 it can be seen that the feed nitrate concentration was generally greater than that of the water in the rest of the system (CW, charcoal and UV chambers). The results in Figure 4.16 show that nitrate removal took place within the system. A maximum feed river water nitrate concentration of 8.26 mg/L was obtained on the 18th of June 2018. Despite this maximum feed water concentration, no water samples collected from the CW, charcoal and UV chambers were found having a concentration of over 3.50 mg/L, save for the sample water collected from the UV chamber on the 10th of July 2018 with a concentration of 5.54 mg/L. Analysis of the water collected from the system revealed that the water collected from the CW chamber had the lowest nitrate concentration overall. This is evident in Figure 4.16 where the CW nitrate concentration curve remains below the other curves, indicating a lower concentration relative to the water in the charcoal and UV chambers and in the feed. On each occasion where rainwater was fed to the water treatment system, the nitrate concentration of

the feed water was 0 mg/L, except for the rainwater feed on the 16th of July 2018, which had a nitrate concentration of 1.74 mg/L. The points in Figure 4.16 that are circled, namely the UV chamber water concentration obtained on the 14th of June; the charcoal chamber concentration obtained on the 20th of June; and the CW chamber concentration obtained on the 27th of June 2018, are considered to be experimental errors based on the high number without any corresponding reason or link to any other parameters assessed (such as phosphates or COD). Table 6 presents the percentage removal rates of nitrates achieved by the PPWTS.

Table 6: Percentage Removal of Average Nitrate Concentration across the Portable Potable Water Treatment System

	Average Value (Nitrates - mg/L)	% Removal / % Increase*
Feed	2.93	-
CW	0.35	88.05
Charcoal	0.56	80.89
UV	1.30	55.63

As is evident in Table 6, the average nitrate concentration of the feed water decreased in the chambers of the PPWTS relative to the feed water. The CW chamber presented the highest nitrate removal rate. The nitrate results presented in Table 6 are presented with the omission of the nitrate concentrations obtained on the 14th of June (UV chamber), 20th of June (charcoal chamber) and the 27th of June 2018 (CW chamber) as these values are considered erroneous.

4.1.2.6 COD

The COD of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.17.

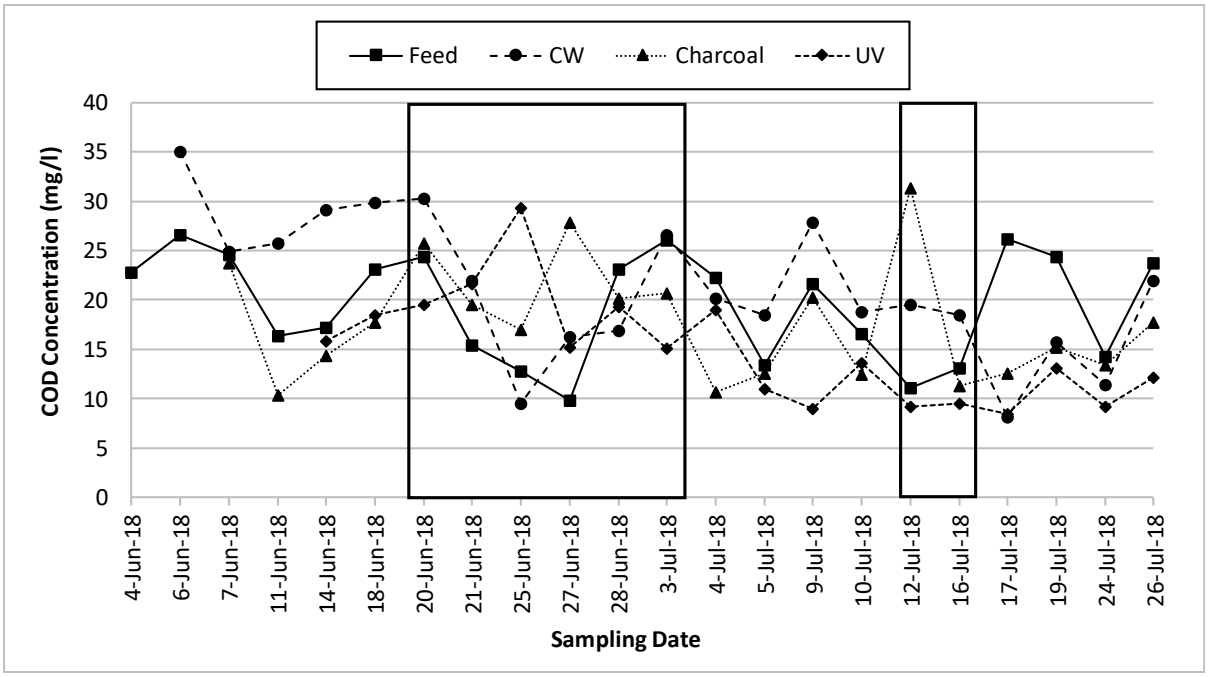

Figure 4.17: COD of Water in the System vs. Sampling Date (Blocks represent rain water)

Figure 4.17 shows the COD concentrations obtained for the feed water as well as for water in the CW, charcoal and UV chambers. It is evident in Figure 4.17 that the COD concentration of the water contained in the system chambers, and that of the feed, fell within the 10 – 25 mg/L range. Fourteen water samples had a COD concentration of above 25 mg/L and only three samples had a concentration above the 30 mg/L. Half of the samples with a concentration above 25 mg/L (7/14 points) were samples related to the COD concentration of the CW chamber sample water. Both the highest and the lowest COD concentrations recorded (35.05 mg/L on the 6th of June and 8.16 mg/L on the 17th of July, respectively) were obtained through analysis of sample water collected from the CW chamber. From the 4th of June to the 20th of June 2018, sample water collected from the CW chamber was shown to have a higher COD concentration than that of the feed. This phenomenon took place from the 4th to the 20th of June 2018 only and is most likely the result of the CW adjusting to the new river water feed as before the 4th of June 2018, the system was fed tap water. The UV chamber presented the lowest COD concentrations as from the 27th of June to the 26th of July 2018, all COD concentrations associated with the UV Chamber were below 20

mg/L. Furthermore, from the 10th to the 26th of July 2018, UV chamber sample water COD concentrations remained below 13.60 mg/L. The following table presents the percentage removal rates of COD achieved by the PPWTS.

Table 7: Percentage Removal of Average COD Concentration across the Portable Potable Water Treatment System

	Average Value (COD - mg/L)	% Removal / % Increase*
Feed	19.48	-
CW	21.26	9.14*
Charcoal	17.71	9.09
UV	14.90	23.51

As per Table 7, the average COD concentration of the water in the PPWTS increased in the CW chamber, but decreased in the charcoal and the UV chambers with the water in the UV chamber having the lowest COD average concentration compared to the charcoal and CW chambers. This indicates that the system is effective at removing COD.

4.1.2.7 TOC

The TOC of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.18.

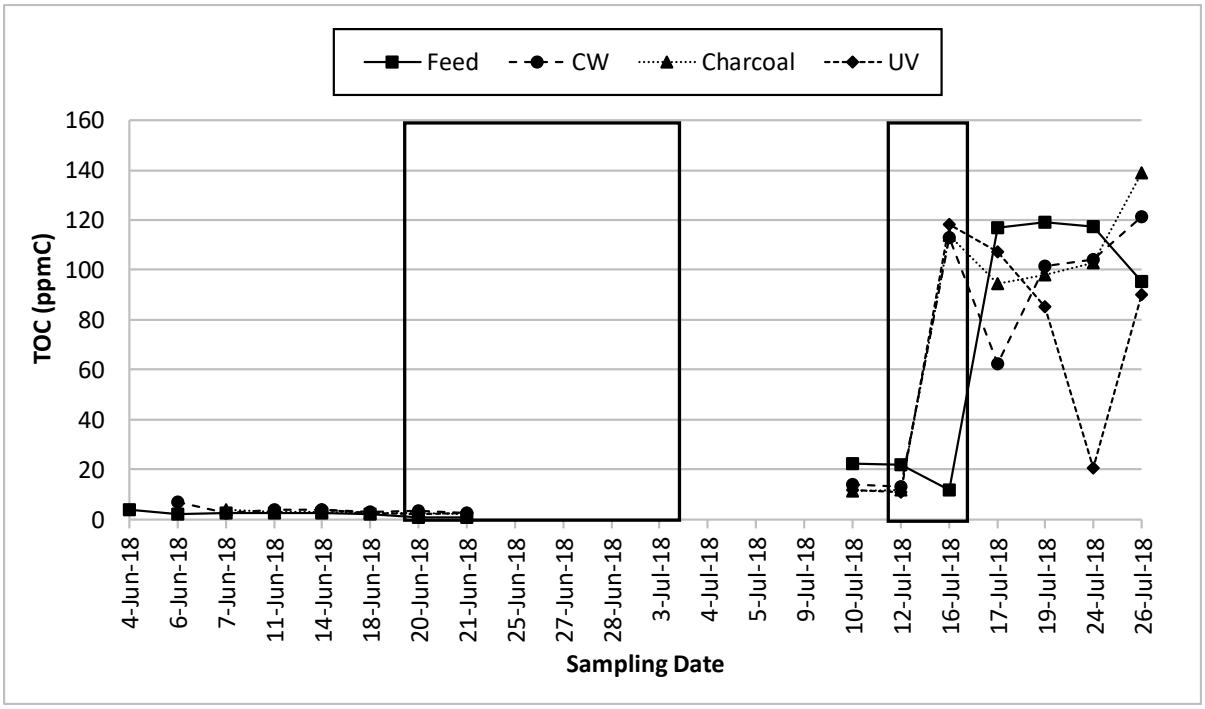

Figure 4.18: TOC of Water in the System vs. Sampling Date (Blocks represent rain water)

In Figure 4.18, it is evident that from the 4th to the 21st of June 2018, the TOC of the sample water collected from the system collectively was relatively low, as no sample returned a TOC concentration of above 7.0 parts per million Carbon (ppmC) during this period. From the 4th to the 21st of June 2018 however, the TOC concentration of the feed was lower than that of the water associated with the CW, charcoal and UV chambers. This is different to what was observed on the 10th and the 12th of July 2018, where the concentration of the feed water was higher than that of the chambers that follow. It is likely that there is a TOC feed concentration below which the treatment system increases the TOC concentration of the water passing through the system. TOC results for water samples collected from the 25th of June to the 9th of July 2018 could not be recorded as the machine software was corrupted and the samples were lost. TOC analysis of the water samples collected from the 25th of June to the 9th of July 2018 produced TOC concentrations of zero (0 ppmC), which is inconsistent with prior TOC findings. The TOC readings of the samples collected from the 16th to the 26th of July 2018 were found to be unusually high and are thus

regarded as inaccurate and not representative of the processes taking place within the water treatment system with regards to TOC. Carbon contamination of the samples collected from the 17th to the 26th of July 2018 is suspected to have caused the sharp spike in TOC concentration that can be seen in Figure 4.18 on the 16th of July 2018. Contamination of the water samples may have occurred due to a rupture of the internal structures inside the deionized water tank; contamination of the deionized water used to rinse the TOC vials; or carbon contamination of the sample water before the TOC analysis. The most reliable representation of the TOC profile of the water used in this research is represented by the results obtained from the 4th to the 21st of June 2018 and from the 10th to the 12th of July 2018. Table 8 shows the average TOC concentrations across the PPWTS as well as the TOC removal rates relative to the feed water

Table 8: Percentage Removal of Average TOC Concentration across the Portable Potable Water Treatment System

	Average Value (TOC - ppmC)	% Removal / % Increase*
Feed	6.71	-
CW	5.93	11.62
Charcoal	5.10	24.00
UV	5.63	16.10

As can be seen in Table 8, TOC concentration decreased in the chambers of the PPWTS. The highest TOC removal occurred in the charcoal chamber, relative to the feed. The results presented in the above table were obtained using the concentration of the samples collected from the 4th to the 21st of June and samples collected on the 10th and the 12th of July 2018. As was previously mentioned, the samples collected from the 25th of June to the 9th of July, and the samples collected from the 16th of July onwards, were corrupted during the TOC analysis process.

4.1.2.8 Total Coliforms

The total coliform MPN count of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.19.

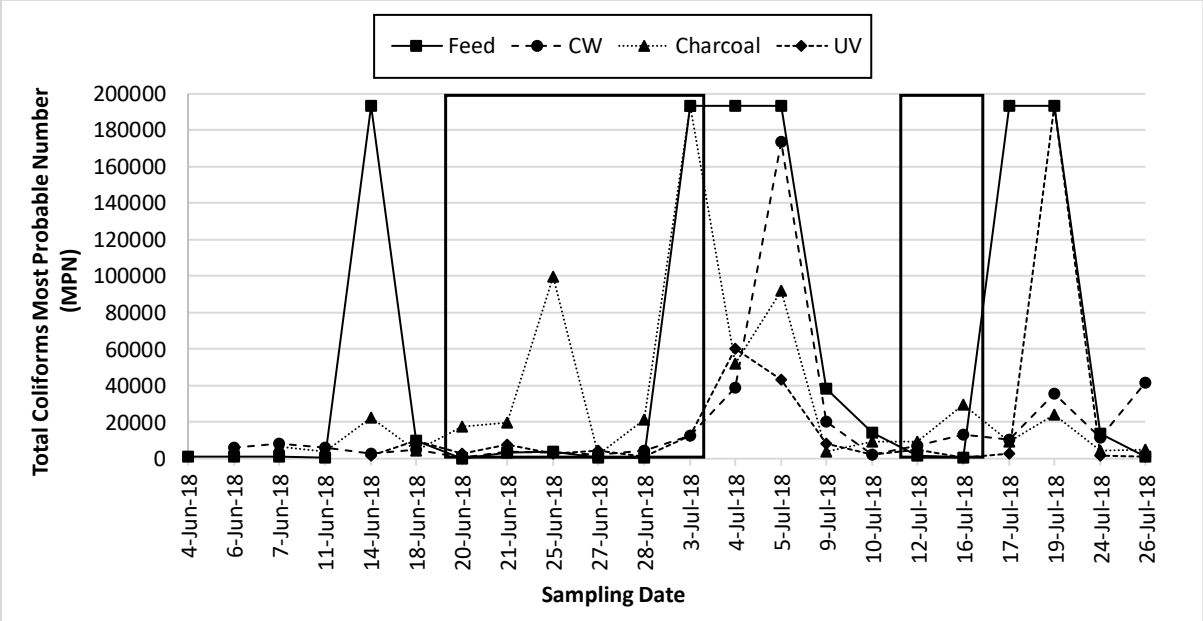

Figure 4.19: Total Coliform MPN in System Water vs. Sampling Date (Blocks represent rain water)

Figure 4.19 conveys the most probable number (MPN) of total coliforms in the feed water and in the CW, charcoal and UV chambers of the PPWTS. The feed water had the highest total coliform count relative to the water collected from the CW, charcoal and UV chambers. As in Figure 4.7, the highest total coliform MPN value was set at 193500 to represent readings that were beyond the testing limits of the IDEXX Colilert-18 test kit. The data point representing the water sample collected from the CW chamber on the 5th of July 2018 is considered an invalid representation of the total coliform MPN as it deviated greatly from the rest of the CW chamber data points that had an MPN value below the 60000 MPN mark. The charcoal chamber also presented an invalid point on the 25th of June 2018 as it can be seen in Figure 4.19 that the total coliform MPN values of the water in the feed and in the CW and UV chambers were below the 20000 MPN mark between the 20th of June to the 3rd of July 2018 when rainwater was used as the feed to the system. On the 28th of June 2018, it was discovered that a bird had defecated in the charcoal chamber. This is the most likely reason for the sharp increase in the total coliform MPN of the water in the charcoal chamber on the 3rd of July 2018. The total coliform MPN in the charcoal chamber remained relatively high

from the 3rd to the 5th of July 2018 as the total coliform MPN associated with the charcoal chamber was highest during this time. The total coliform MPN of the sample collected from the UV chamber on the 19th of July 2018 was considered an invalid representation of the MPN of total coliforms in the UV chamber as the total coliform MPN values of the CW and charcoal chambers leading to the 19th of July 2018 (from the 10th to the 17th July 2018) were not high and would not likely cause as drastic an increase in the total coliform MPN of the water in the UV chamber as what can be seen in Figure 4.19.

Table 9: Percentage Removal of Average Total Coliforms (MPN) across the Portable Potable Water Treatment System

	Average Value (Total Coliforms - MPN)	% Removal / % Increase*
Feed	57000	-
CW	19300	66.14
Charcoal	31300	45.09
UV	20000	64.91

The results depicted in Table 9 convey the average total coliform removal rates that were achieved by each chamber of the PPWTS relative to the feed. The highest total coliform removal rate was achieved in the CW chamber. In general, however, the system was effective in removing two thirds of the total coliforms.

4.1.2.9 *E. coli*

The *E. coli* MPN count of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.20.

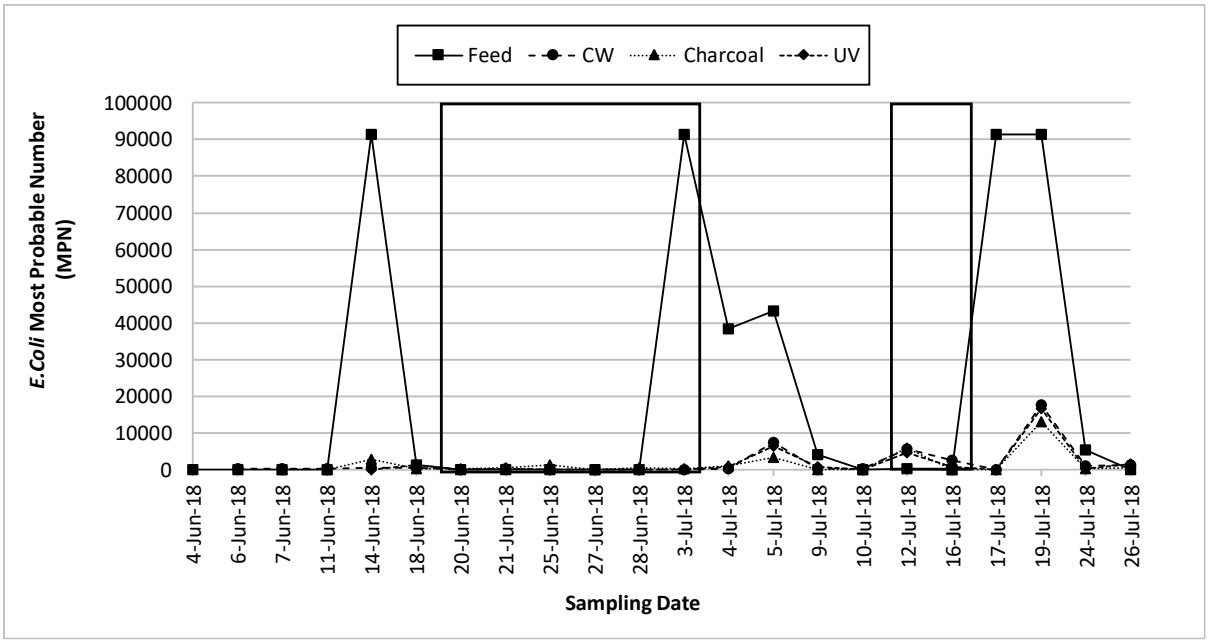

Figure 4.20: *E. coli* MPN in System Water vs. Sampling Date (Blocks represent rain water)

As can be seen in Figure 4.20, the feed water had the highest MPN value for *E. coli* relative to water in the CW, charcoal and UV chambers of the system. A maximum *E. coli* MPN value was set at 91500 to represent the MPN values of the water samples with MPN readings that were beyond the limits of the IDEXX Colilert-18 testing kit. Looking at Figure 4.20, it is evident that the PPWTS effectively treated the feed water passing through the chambers of the system as only the feed water was recorded as having high *E. coli* MPN readings, while the *E. coli* MPN readings of the sample water collected from the CW, charcoal and UV chambers did not exceed the 2000 MPN. The following table presents the *E. coli* removal rates across the chambers of the PPWTS.

Table 10: Percentage Removal of *E. coli* (MPN) across the Portable Potable Water Treatment System

	Average Value (*E. coli* - MPN)	% Removal / % Increase*
Feed	23000	-
CW	1800	92.17
Charcoal	1500	93.48
UV	1800	92.17

Table 10 presents the *E. coli* removal rates associated with each chamber in the PPWTS relative to the feed. The highest *E. coli* removal rate was achieved by the CW chamber. The system was able to remove 90% of the *E. coli*. These results are similar to those reported by Barbagallo et al. (2013) and Mairi et al. (2012) who obtained *E. coli* removal rates of 99-99.99% and 92.9 ± 6.05% respectively.

4.1.2.10 Sulfates

The Sulfate concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.21.

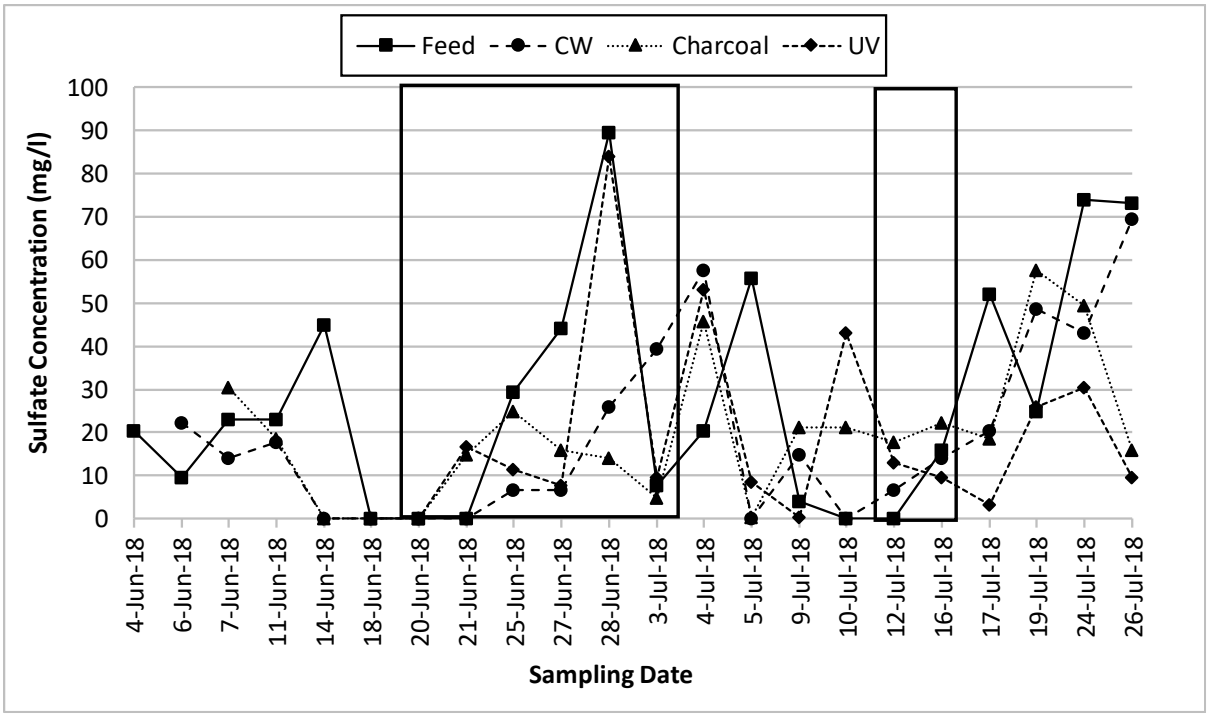

Figure 4.21: Sulfate Concentration of Water in the System vs. Sampling Date (Blocks represent rain water)

Figure 4.21 presents the Sulfate concentration in the feed water as well as in the CW, charcoal and UV chambers. Although an apparent trend in the change of the sulfate concentration across the system is not clearly visible in Figure 4.21, the highest sulfate concentrations in the PPWTS are associated with the feed water. Feed water sulfate concentrations on the 28th of June (89.45 mg/L), 24th of July (74.00 mg/L) and 26th of July 2018 (73.10 mg/L) are the highest concentrations in Figure 4.21. The data point representing the water sample collected from the UV chamber with a concentration of 84 mg/L (28th of June 2018) is considered an invalid point as no other UV chamber water samples gave a concentration reading as high as this sample. Additionally, the concentrations of the water samples in the CW and charcoal chamber from the 20th to the 27th of June 2018 do not point towards the potential of a sharp increase in the sulfate concentration of the sample water in the UV chamber of the 28th of June. Table 11 shows the sulfate removal rates achieved in each chamber of the PPWTS.

Table 11: Percentage Removal of Average Sulfates Concentration across the Portable Potable Water Treatment System

	Average Value (Sulfates - mg/L)	% Removal / % Increase*
Feed	27.80	-
CW	19.40	30.22
Charcoal	19.67	29.24
UV	18.08	34.96

The removal rates listed in Table 11 show that the average sulfate concentration in the PPWTS decreased in the chambers of the system relative to the feed water. Average Sulfate concentration was lowest in the UV chamber. Whilst sulfate is not a chemical of concern in this locality, it may be if these systems were deployed in areas where drinking water was impacted by AMD.

4.1.2.11 Heavy Metals: Aluminium

The aluminium concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.22.

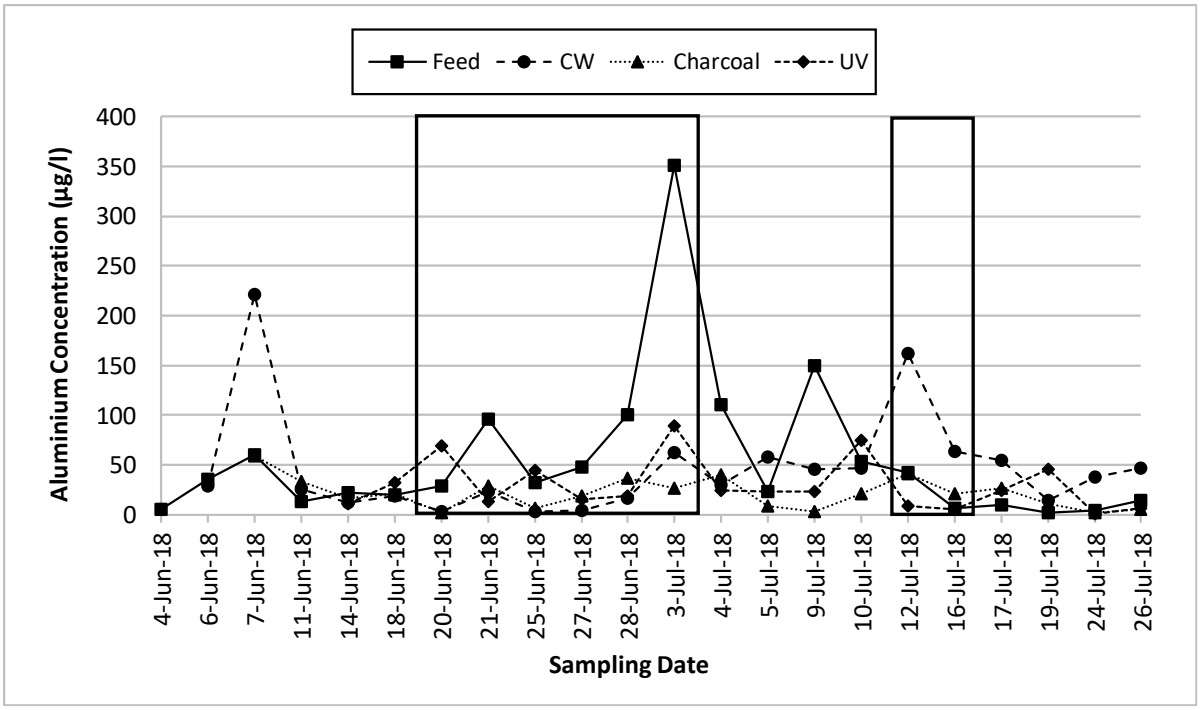

Figure 4.22: Aluminium Concentration of Water in the System vs. Sampling Date (Blocks represent rain water)

In Figure 4.22, it is evident that the feed water aluminium concentrations are generally higher than those obtained from the CW, charcoal and UV chambers. This is evidenced by the feed water concentrations obtained from the feed water fed to the PPWTS on the 21st and the 28th of June and on the 3rd, 4th and 9th of July 2018. The highest aluminium concentration obtained during this research was 351 µg/L, which was the aluminium concentration of the feed water to the PPWTS on the 3rd of July 2018. The lowest aluminium concentration obtained was 0.33 µg/L. This concentration was that of the sample water collected from the UV chamber on the 24th of July 2018. It is clear from Figure 4.22 that the concentration of the feed water and that of the water contained in the system chamber fell within the 0 to 100 µg/L range as only 3 feed concentrations (3rd, 4th and 9th July 2018) and 2 CW sample concentrations (7th June and 12th July 2018) had a concentration of above 100 µg/L. There is a sharp increase in the concentration of the water collected from the CW on the 7th of June 2018 which may have been the CW chamber's response

to the introduction of sewage impacted river water to the system. The CW may have been in a phase where it was attempting to equilibrate with the newly introduced river water. The average aluminium concentrations of the feed water and the water in the CW, charcoal and UV chambers obtained as a result of this research are 55.80, 46.50, 21.38 and 29.19 µg/L respectively. The decrease in the average aluminium concentration across the PPWTS chambers shows that the system is effective in reducing the aluminium concentration of impacted water to some degree. Table 12 shows the average aluminium removal rates obtained in each chamber of the PPWTS.

Table 12: Percentage Removal of Average Aluminium Concentration across the Portable Potable Water Treatment System

	Average Value (Aluminium - µg/L)	% Removal / % Increase*
Feed	55.80	-
CW	46.50	16.67
Charcoal	21.38	61.68
UV	29.19	47.68

It is evident in Table 12 that the average concentration of aluminium in the feed water decreased across the system chambers, with the highest decrease taking place in the charcoal chamber. The average aluminium concentration in all 3 chambers of the PPWTS satisfy the WHO drinkable water limit of 0.10 mg/L (100 µg/L) and the SANS Operational Risk aluminium concentration limit of 300 µg/L (Vinlab, 2016; World Health Organization, 2011).

4.1.2.12 Heavy Metals: Manganese

The manganese concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.23.

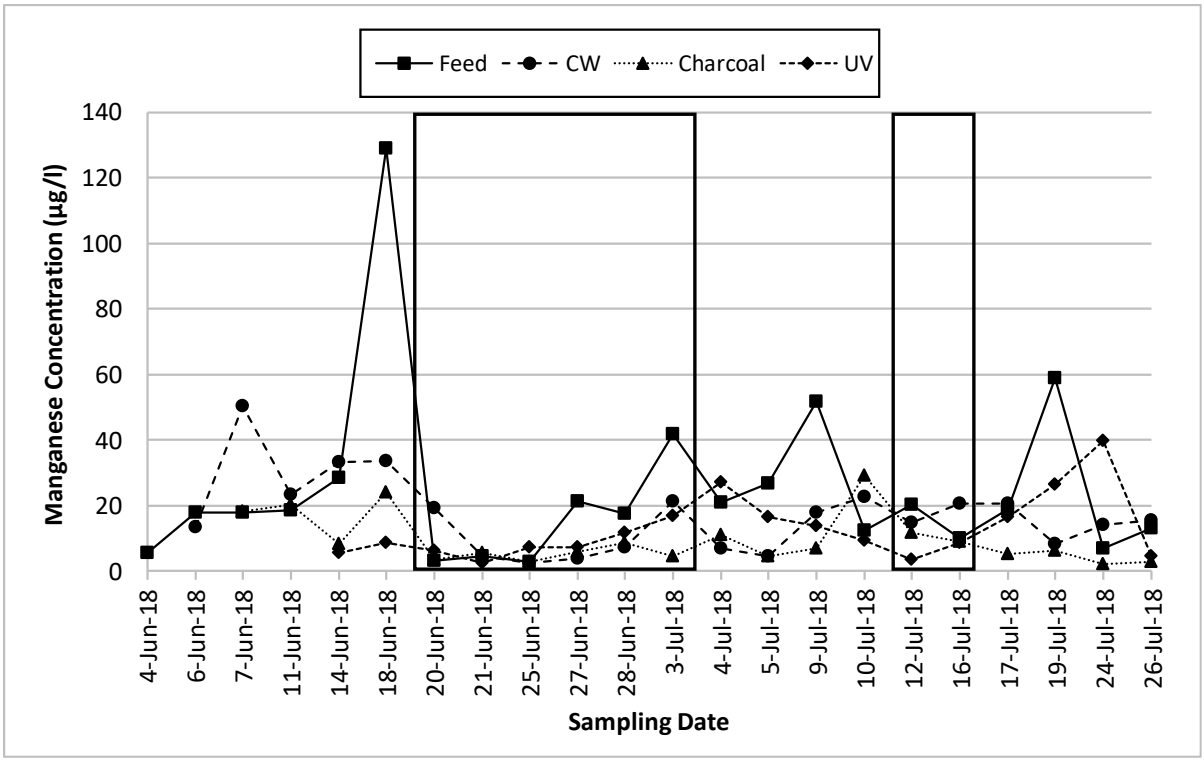

Figure 4.23: Manganese Concentration of Water in the System vs. Sampling Date (Blocks represent rain water)

Figure 4.23 shows the manganese concentrations in the PPWTS. The majority of the samples collected from the water treatment system are shown in Figure 4.23 to have had a concentration of below 60 µg/L. The only water sample that had a concentration of over 60 µg/L was the feed water sample collected on the 18th of June 2018. A concentration of 129 µg/L was obtained for the feed water sample obtained on the 18th of June 2018. This was the highest manganese concentration obtained during this research. The feed water curve generally shows higher concentrations of manganese relative to the other curves showing the concentrations in the CW, charcoal and UV chambers. The lowest concentration in Figure 4.23 is 2.16 µg/L which is the concentration of the sample water collected from the charcoal chamber on the 24th of July 2018. The charcoal chamber is the only chamber with manganese concentrations that do not exceed 30.00 µg/L. There is a sharp increase in the concentration of manganese in the CW chamber on the 7th of June 2018. This

coincides with the spike seen in Figure 4.22 on the 7th of June 2018, corresponding to an increase in the concentration of aluminium in the CW chamber. Table 13 shows the average manganese removal rates achieved through the use of the PPWTS.

Table 13: Percentage Removal of Average Manganese Concentration across the Portable Potable Water Treatment System

	Average Value (Manganese - µg/L)	% Removal / % Increase*
Feed	24.91	-
CW	17.02	31.68
Charcoal	9.50	61.89
UV	12.86	48.38

The decrease in the average manganese concentration across the chambers of the PPWTS shows that manganese is being removed by the system. As was the case with aluminium, the highest manganese removal rate was achieved by the charcoal chamber. The average manganese concentrations in the PPWTS chambers satisfy both the SANS (≤ 400 µg/L) and the WHO (≤ 0.05 mg/L i.e. ≤ 50 µg/L) drinking water limits.

4.1.2.13 Heavy Metals: Lead

The lead concentration of the feed water and the water in the CW, charcoal and UV chambers is shown in Figure 4.24.

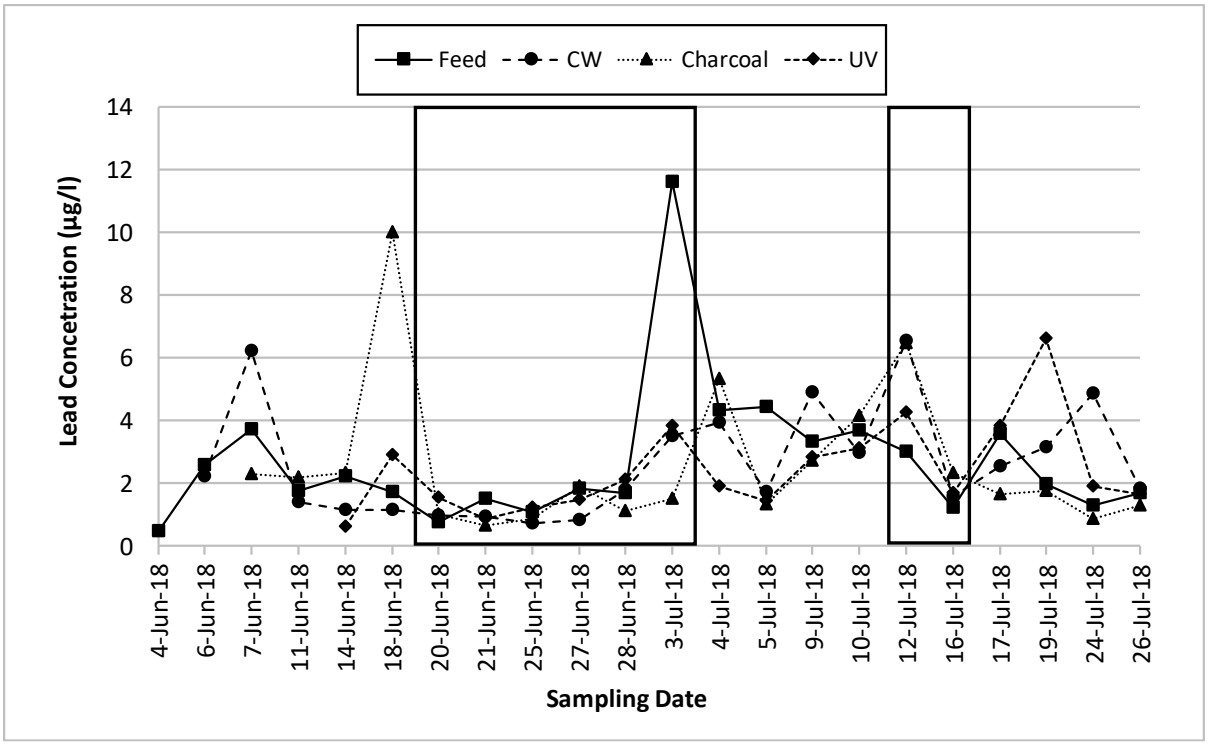

Figure 4.24: Lead Concentration of Water in the System vs. Sampling Date (Blocks represent rain water)

Figure 4.24 shows the lead concentrations of the feed water and of the water in the PPWTS chambers. It can be seen in Figure 4.24 that the majority of the water samples collected from the feed and from the system chambers had concentrations of below 6 µg/L. The only exceptions are concentrations of the samples collected from the feed water on the 3rd of July, the CW sample collected on the 7th of June and the 12th of July, the charcoal chamber samples collected on the 18th of June and the 12th of July and the UV chamber sample collected on the 19th of July 2018. There is a sharp increase in the lead concentration of the sample water in the CW chamber on the 7th of June 2018. This coincides with the spike in aluminium and manganese on the 7th of June 2018. The sample water collected from the CW chamber on the 7th of June 2018 seems to have had a relatively high concentration of minerals and heavy metals. It is also apparent that the lead concentration in the system increased after the 3rd of July 2018. As can be seen in Figure 4.24, the concentration of the water in system before the 4th of July 2018 was generally less than 4 µg/L.

After the 3rd of July 2018 however, the concentration of the water in the PPWTS rises beyond 4 µg/L. This is the time during which the impacted water from the river was reintroduced to the PPWTS following the sewage discharge that occurred between the 14th and the 20th of June 2018. In Table 14 the average lead removal rates achieved through use of the PPWTS are presented.

Table 14: Percentage Removal of Average Lead Concentration across the Portable Potable Water Treatment System

	Average Value (Lead - µg/L)	% Removal / % Increase*
Feed	2.70	-
CW	2.61	3.29
Charcoal	2.58	4.52
UV	2.43	10.12

Table 14 shows the average lead removal rate in each of the chambers of the PPWTS relative to the feed, as well as the average concentration in each chamber. The average lead concentration in the system is considerably low, as are the lead removal rates. The highest average lead removal rate is achieved in the UV chamber. Although the lead removal rates are low, the lead concentrations in the PPWTS are below the SANS and WHO drinking water lead limit of 10 µg/L.

4.1.3. Comparison of Results to SANS and WHO Drinking Water Standards

In Table 15 the average results obtained from analysis of the water samples collected from the PPWTS chambers and feed water are compared to the WHO and the SANS drinking water limits/standards. The sample labelled UV represents the outlet of the system. It is evident that not all parameters conform to WHO or SANS standards, however overall, the system had a positive impact on the river water. PPWTS outlet water parameters conforming to the SANS drinkable water standards based on average values are the nitrates, sulfates, TOC, manganese and lead concentrations. Outlet parameters conforming to the WHO drinkable water standards based on average values are pH and aluminium. Turbidity, total coliforms and *E. coli* do not conform to SANS or WHO potable water standards. E.C., phosphates and COD do not have explicit SANS or

WHO drinkable water limits. In Table 15 below, values that are double-underlined indicate water quality parameter average values that exceed the SANS/WHO drinking water limits.

Table 15: Results Comparison to WHO and SANS Water Standards (Vinlab, 2016; World Health Organization, 2011)

Parameter	Feed	CW	Charcoal	UV	WHO	SANS
pH	8.40	8.10	8.04	8.77	8.2 - 8.8	
E.C. (µS/cm)	292	409	404	375		1500
Turbidity (NTU)	3.27	92.24	2.86	2.09	1.5	1
Phosphates (mg/L)	0.052	0.066	0.14	0.10		
Nitrate as N (mg/L)	2.93	0.35	0.56	1.30	50	11
Sulfates (mg/L)	27.80	19.40	19.67	18.08		500
COD (mg/L)	19.48	21.26	17.71	14.90		
TOC (ppmC)	6.71	5.93	5.10	5.63		10
Total Coliforms (MPN / 100 mL)	57000	19300	31300	20000	0	
***E. coli* (MPN / 100 mL)**	23000	1800	1500	1800	0	0
Aluminium (µg/L)	55.80	46.50	21.38	29.19	100	
Manganese (µg/L)	24.91	17.02	9.50	12.86	50	400
Lead (µg/L)	2.70	2.61	2.58	2.43	10	10

4.2 DISCUSSION

The experiments carried for this research investigated river water quality as well as the enhancement of the quality of river water through the use of a novel CW/charcoal/UV system. The focus of the investigation was to determine the levels of the impurities in river water while making use of a CW/charcoal/UV system to try and bring the level of these impurities down to potable water standards. In this discussion, each water quality indicator that was considered during this research is discussed with regards to the quality of the river water used for this research and the results obtained from the use of the PPWTS.

4.2.1 pH

The average pH of the river water used for this research was 8.40. This pH value falls within the WHO and SANS domestic water limits. This pH value also falls within the Rand Water (2018) pH range for healthy rivers (6.5 – 8.5). The average pH of the river water collected during this research falls within a similar pH range as the Grand and Speed river water bodies (Grand River Conservation Authority, 2018). The pH of the feed water used in this investigation falls within international river norms.

The average pH of the water fed to the PPWTS decreased in the CW and charcoal chambers while an increase in pH of the water was recorded in the UV chamber. The pH of water in CWs appears to be variable as Mwanyika et al. (2016) reported an increase in pH from 7.17 ± 0.19 to 7.36 ± 0.23, while Sheoran (2017) reported an increase in pH from 2.93 to 7.22. Notwithstanding this, the pH is still within the required potable water limits.

4.2.2 E.C.

The average E.C. of the river water used as the feed for this research was 292 µS/cm. This value is well below the SANS limit of 1500 µS/cm. According to the US EPA (2012) the conductivity of rivers in the United States generally range from 50 to 1500 µS/cm and thus the river studied here falls within international norms.

The average E.C. of the water fed to the treatment system used for this research increased in the CW and charcoal chambers. There was then a decrease in E.C. when the water in the system moved

from the charcoal chamber to the UV chamber. The increase in E.C. did not cause the E.C. of the water to fall outside the SANS limits. Mwanyika et al. (2016) saw a decrease in the E.C. of the influent water that was passed through the CW used as part of their study and as such contradicts the findings of this investigation. Notwithstanding this however, the E.C. of the water collected from the PPWTS still falls within the E.C. potable water limits.

4.2.3 Turbidity

The average turbidity of the feed water used for this research was 3.27 NTU. This value exceeds both the WHO and SANS drinking water turbidity limits. Rivers and lakes are said to have a turbidity of less than 10 NTU during low flow periods therefore this finding is to be expected (Fondriest Environmental, Inc, 2014).

The turbidity of the water fed to the PPWTS decreased as it passes through the system. The average turbidity of the water collected from the CW chamber is high relative to the other chambers as the sample water collected from the CW was collected using a syringe with a metallic extension that penetrated the gravel of the CW. This resulted in the collection of CW sample water containing soil particles. It is clear however, that the water flowing into the charcoal and UV chambers does not carry with it the soil particles from within CW. The CW acts as a filter and removes soil particles and SS from feed water into the system. Others [Sheoran (2017), Ligang et al. (2011), Nyakang'o and van Bruggen (1999)] found similar behaviour, where drinking water limits were not reached. If the design goal of a system is to reach drinking water limits, other treatment methods may need to be added.

4.2.4 Phosphates

The average phosphate concentration of the feed water used in this research was 0.052 mg/L. Phosphate limits in drinking water are not explicitly given as part of the SANS and WHO standards listed in Table 15. The feed phosphate concentration, however, is considerably low taking into account the sewage discharge that occurred between the 14th and the 20th of June 2018.

The phosphate concentration of the water in the PPWTS increases after occupying the CW chamber. This occurrence is contrary to many other experiments that were carried out of this

nature. Song et al. (2006), Ligang et al. (2011) and Khisa et al. (2014) all recorded a decrease in TP/phosphates to some extent. Literature points towards CWs being effective with regards to phosphorus removal, however the PPWTS brings about an increase in the phosphate concentration of the water passed through the system. This occurrence may be the result of a low phosphate concentration in the feed water, causing phosphates to diffuse from a high concentration in the CW chamber to a low phosphate concentration in the feed water. Notwithstanding the increase in phosphate concentration, the phosphate concentration in the water collected from the UV chamber of the PPWTS does not pose a great human health risk as it is very low.

4.2.5 Nitrates

An average nitrate feed water concentration of 2.93 mg/L was obtained through analysis. This concentration falls below the SANS and the WHO drinking water standards. According to the World Health Organization (2011) surface water typically has a nitrate concentration of 0-18 mg/L and the nitrate feed concentration obtained falls within this range.

The average nitrate concentration decreases in the CW chamber, indicating that there is nitrate removal occurring. The nitrate concentration then increases slightly in the charcoal chamber while the greatest increase is seen in the UV chamber. The nitrate concentration, however, is well below the SANS and WHO drinking water limits throughout the water treatment system. Chang et al. (2013) and Beutel et al. (2009) recorded a decrease in nitrate concentrations when wastewater was passed through a CW treatment setup. The rise in nitrate concentration observed in the UV chamber may be the result of an accumulation of nitrates over time. The UV chamber was emptied after each experimental run, however, nitrates may have remained on the sides of the UV chamber thus contributing to the nitrate concentration of the new water entering the system. It is also possible that the rise in nitrate concentration observed occurring in the charcoal and UV chambers is the result of nitrates escaping the CW chamber as water moves from the CW chamber into the charcoal and UV chambers.

4.2.6 Sulfates

The feed sulfate concentration obtained through this experiment was 27.80 mg/L. This concentration is well below the SANS drinking water limit.

The sulfate concentration of the water fed to the PPWTS decreased across the system chambers and is lowest in the UV chamber. The removal rates achieved in the CW, charcoal, and UV chambers (30.22, 29.24 and 34.96% respectively) are in the neighborhood of the results obtained by Sheoran et al. (2017) in treated AMD using CWs (21.52 to 28.09% sulfate removal rate). Song et al. (2001) reported near-complete removal of sulfates using CWs to treat synthetic mine water. The water collected from the PPWTS met the SANS drinking water standards.

4.2.7 COD

An average feed COD of 19.48 mg/L was obtained through analysis of the feed water.

A decrease in the COD of the feed water passed through the PPWTS was observed. A maximum decrease of 23.51% was observed in the UV chamber. Sudarsan et al. (2015) carried out experiments on a pilot scale CW located on the SRM university campus. Wastewater obtained from a sewage treatment plant was passed through the CW as part of the investigation. The results of the investigation show a 76.16% decrease in COD of the wastewater that was passed through the CW system. Villalobos et al. (2013) constructed a pilot plant comprised of six HSSF CWs which was fed with domestic wastewater. The experimental procedure was carried out for 280 days with the aim being the removal of organic matter and nutrients. This experiment achieved a 59% COD removal efficiency. Albalawneh et al. (2016) achieved a 51% decrease in the COD of partially treated municipal wastewater. The results obtained during this research and those obtained from literature indicate that CWs have a capacity to remove COD from wastewater.

4.2.8 TOC

In this study, TOC was assessed and in general, it decreased. The feed water TOC concentration obtained through analysis was below the TOC SANS drinking water limit. Other studies describing this parameter were not found. Notwithstanding this, the decrease in TOC is positive and beneficial.

4.2.9 Total Coliforms

The average feed water MPN value obtained through testing was 57000. The feed water used during this research does not meet the World Health Organization total coliform drinking water limit of 0 MPN / 100 mL.

The average level of total coliforms in the feed water passed through the CW/charcoal/UV treatment system decreased in the CW chamber by 66.14%. The average total coliform level increased in the charcoal chamber relative to the CW chamber. In the UV chamber, the average total coliform level dropped relative to the charcoal chamber. Relative to the feed water, the average total coliforms MPN value dropped by 64.91% in the UV chamber. Song et al. (2006). Nyakang'o and van Bruggen (1999) reported a considerable decrease in the level of total and fecal coliforms using CWs to treat wastewater (above 99% removal). It is evident that CWs have a positive effect with regards to total and fecal coliform removal. The UV chamber used for this research did not have a substantial impact on total coliform removal. According to Extension (2016), it is not advisable to perform UV disinfection on wastewater having a total coliform count of above 1000. This may provide a reason for perceived lack of effectiveness of the UV LEDs in substantially decreasing the total coliform count. PPWTS was not able to bring the level of total coliforms down to WHO potable water standards. To reach drinking water standards using this system, alternative treatment methods may need to be introduced.

4.2.10 *E. coli*

The average feed MPN *E. coli* value obtained through the testing of the feed water was 23000. This value falls outside both the WHO, and the SANS drinking water limits.

The average *E. coli* bacteria levels in the water passed through the system decreased by 92.17% across the entire water treatment system. Barbagallo et al. (2013) achieved *E. coli*. Removal rates of 99% to 99.99% while Mairi et al. (2012) reported a decrease in *E. coli* bacteria of 92.9 ± 6.05%. The studies mentioned, including this research project, provide evidence of the capacity that CWs have a notable capacity to remove *E. coli* in wastewater sources. Looking at the results achieved during this experiment, most of the *E. coli* inactivation can be attributed to the CW chamber as the charcoal chamber is not expected to have a great influence on the removal of *E. coli*. The UV chamber does not have a considerable effect on the inactivation of *E. coli*. As is the case the total

coliforms, if this system is aimed at treating river water to potable standards, alternative water treatment methods should be considered.

4.2.11 Heavy Metals: Aluminium

The average aluminium concentration of the feed water fed to the PPWTS was 55.80 µg/L. According to WHO (2003), the concentration of aluminium in waters with a near-neutral pH commonly ranges between 0.1 to 0.5 mg/L. Though the average pH of the feed water used to feed the PPWTS was alkaline (pH = 8.40), it presented an aluminium concentration similar to that of near-neutral water (0.0558 mg/L). The feed water aluminium concentration met the WHO drinkable water limit of 0.1 mg/L.

The aluminium removal rate in the PPWTS was highest in the charcoal chamber, with a removal rate of 61.68%. Ghazy & El-Morsy (2007) conducted a study in which powdered activated carbon (PAC) was used to remove aluminium from aqueous solutions of concentrations ranging from 1.35 to 8.1 mg/L. In this study, approximately 100% aluminium removal was achieved for aqueous solutions of concentrations 1.35 to 2.75 mg/L. Goher et al. (2015) used GAC to remove aluminium from pollution sources draining into the Ismailia Canal in Egypt and achieved a 99.2% aluminium removal rate. In both investigations, aluminium removal was close to 100%. This may be the result of using a powdered activated carbon and thus providing more adsorption sites for aluminium to adsorb onto. Another reason for the high removal rate relative to the PPWTS charcoal chamber may be that activated carbon was used instead of regular "braai" charcoal. It is evident from the results however, that regular charcoal is able to serve as an adsorbent for aluminium removal.

4.2.12 Heavy Metals: Manganese

The average manganese feed water concentration was 24.91 µg/L. This average concentration is below the SANS and WHO drinking water limits of 400 µg/L and 50 µg/L respectively.

The highest manganese removal rate in the PPWTS was achieved in the charcoal chamber. Experimental results show a 61.89% manganese removal rate in the charcoal chamber. Reddy & Reddy (2014) conducted an investigation in which they used powdered activated carbon to perform manganese adsorption. Using one gram of powdered activated carbon (particle size 82.5µm),

Reddy & Reddy (2014) achieved 90% manganese removal from a 20 mg/L manganese aqueous solution of 30 mL in 30 minutes. Abustan et al. (2017) achieved 99.39% manganese removal using GAC as an adsorbent to remove manganese from riverbank filtration water. The difference in the manganese percentage removal rates obtained using the PPWTS and those obtained by Reddy & Reddy (2014) and Abustan et al. (2017) can be attributed to the fact that the PPWTS makes use of regular "braai" charcoal while activated carbon was used in the investigations conducted by Reddy & Reddy (2014) and Abustan et al. (2017). Another reason may be the fact that the charcoal chamber of the PPWTS was made up of charcoal briquettes and not finely ground charcoal. Crushed or powdered charcoal would provide a larger surface area for adsorption to occur. Despite the large difference in the manganese percentage removal however, it is evident that regular charcoal is capable of removing manganese from wastewater.

4.2.13 Heavy Metals: Lead

The average feed water lead concentration obtained during this research was 2.70 µg/L. This average concentration is well below the SANS and the WHO drinkable water limit of 10 µg/L.

The average lead concentration of the feed water decreased by 10.12% across the three chambers of the PPWTS. A lead removal of 3.29% was achieved across the CW chamber while the charcoal chamber resulted in a 4.52% lead concentration decrease. Khisa et al. (2014) achieved a decrease in lead concentration of 18.5% in wastewater that was passed through a constructed wetland system. Mwanyika et al. (2016) saw a decrease of 94% in the lead concentration of the effluent passed through a constructed wetland system at Egerton University in Kenya. Both Khisa et al. (2014) and Mwanyika et al. (2016) reported higher lead removal rates compared to the results obtained through use of the PPWTS. This may be partly due to the incoming lead concentrations of the wastewater being fed to each respective system. Khisa et al. (2014) and Mwanyika et al. (2016) made use of feed water with lead concentrations of 11.9 and 1.25 mg/L respectively. These inlet concentrations are much higher than the average 2.70 µg/L (i.e. 0.0027 mg/L) lead concentration of the feed water that was fed the PPWTS. The low lead concentration of the feed water to the PPWTS lowered the potential for lead removal by the system as the concentration of lead was already exceptionally low.

In a study conducted by Rosli et al. (2015), Granular Activated Carbon (GAC) and jackfruit activated carbon was used to investigate lead adsorption in wastewater. Rosli et al. (2015) reported a 98% decrease of lead in wastewater using a GAC adsorbent and an 80% decrease in lead when jackfruit was used. These removal rates are also much higher than the 10.12% that was achieved using the PPWTS. This may be again, due to the low inlet lead concentration of the water fed to the PPWTS.

CHAPTER 5 – CONCLUSIONS AND RECOMMENDATIONS

The focus of this research was the treatment of sewage impacted river water using a novel three-stage CW/charcoal/UV water treatment system. An experimental investigation demonstrated the capacity of the CW/charcoal/UV system to remediate the river water to potable standards. Analyses performed on the feed water revealed the changes in the properties of the water passing through the system chambers. Analyses of the water in the CW/charcoal/UV system revealed the extent to which the system can treat river water as well as which water properties the treatment system could bring to potable standards according to WHO and SANS.

This research project focused on two research objectives. These were

- To investigate and determine the quality of an urban, sewage impacted river.
- To determine if a novel CW/charcoal/UV system is effective in treating urban sewage impacted river water to potable standards.

The first objective was achieved through analysis of the feed water passed through the system. The feed water was tested for pH, E.C, turbidity, phosphates, nitrates, sulfates, COD, TOC, total coliforms, *E. coli* as well as the metals aluminium, manganese and lead. It was discovered through the experiments carried out that total coliforms and *E. coli* had the highest concentration of the water quality indicators that were addressed in this study. The levels of total coliforms and *E. coli* bacteria of the feed water are substantially greater than the drinking water limits set by WHO and SANS. Turbidity of the feed water was also higher than the limits set by WHO and SANS. It was evident through this study that urban storm-water rivers are highly impacted by sewage especially as indicated by the concentration of coliform bacteria. This is indicative of aging and poorly maintained sewage infrastructure around this river. Based on this finding, and visual observation of other areas of Johannesburg, it implies that most rivers within the Johannesburg metropolitan region would be highly impacted.

The second objective was achieved through the design and usage of the CW/charcoal/UV system. The results obtained demonstrated that the system was effective, although it was not able to treat sewage impacted river water SANS/WHO standards. The water quality of the outlet from the system was, however, in all cases except phosphate, better than the influent. This implies that such a system could potentially be used for persons without access to piped potable water to improve

the quality substantially. Whilst not meeting all the SANS/WHO codes, the water quality would lessen the risk to persons using such a system than if they were using the river water untreated.

Based on the results obtained during this research, a number of design improvements, that may enhance the performance of the PPWTS, can be recommended.

The UV disinfection chamber was not effective in destroy total coliforms and *E. coli* bacteria in water. Replacing the UV disinfection unit with SODIS may result in more efficient total coliform and *E. coli* disinfection. Making use of a SODIS unit for disinfection would be beneficial in South Africa as the country experiences a high degree of sunshine, which is necessary for SODIS to work. SODIS is a more cost-effective water treatment method, relative to UV disinfection, as only large transparent, PET containers are needed for disinfection. Such a resource is more accessible to rural communities as opposed to UV LEDs and battery packs.

The UV chamber could be enhanced in a way that allows for the UV LEDs to be in contact with the water. The distance between the UV LEDs and the water in the UV chamber of the PPWTS is approximately 30 cm. This may have played a role in the lack of effectiveness of the UV chamber in destroying total coliforms and *E. coli* bacteria. To achieve a chamber where the UV LEDs are in contact with the water being treated, a different UV chamber design would be required. This new design would need to allow for the LEDs to be in contact with the water, while ensuring the UV PCB remains protected from the water in the chamber. Such a design may require positioning the UV PCB on the side or underneath the UV chamber and making small perforations in the chamber to fit the LED through, while keeping water away from the PCB. Bringing the UV LEDs closer to the water in the UV chamber may have a positive disinfection effect.

The PPWTS was not able to bring the turbidity of water down to potable standards. In order to address this, an extra chamber made up of gravel and various soils could be set up between the CW and charcoal chambers. This chamber would only serve as a filter to lower the turbidity of the water before it enters the charcoal chamber.

There are a number of unanswered questions regarding this research project that future work may be able to address. Performing experimental work until the system fails would help to identify when (or if) the system breaks and which component of the system breaks first i.e. which water quality indicator does the system stop removing/treating first. This information would assist in

knowing when to replace the components (such as charcoal, for example) of the system to ensure effective water treatment. Performing experiments during the various seasons of the year would also prove useful as this would provide a view of how removal rates change with respect to each season and varying weather conditions. This research project could also be further improved by examining the removal rates of the water quality parameters in each chamber individually. This may provide a clearer view of the removal rates of each water quality parameter relative to each chamber i.e. which chamber results in the highest and lowest removal rates for each water quality parameter. This information would allow for a better understanding of the relationship between the water quality parameters and the different water treatment methods associated with each chamber of the PPWTS.

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APPENDIX

Appendix A: Experimental Results Tables

Table A1: Experimental pH Results

pH Readings				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	8.73	-	-	-
06 June 2018	8.11	7.48		-
07 June 2018	7.70	8.68	8.25	-
11 June 2018	7.98	8.96	8.05	-
14 June 2018	9.04	8.50	8.31	9.00
18 June 2018	8.98	8.80	7.90	9.12
20 June 2018	9.21	8.48	7.65	9.01
21 June 2018	8.92	8.64	7.75	8.64
25 June 2018	9.31	9.05	8.68	8.61
27 June 2018	9.28	9.02	7.96	8.37
28 June 2018	9.26	9.02	9.01	9.20
03 July 2018	8.64	9.26	8.59	8.80
04 July 2018	8.14	9.26	8.26	8.71
05 July 2018	8.62	7.89	7.69	8.25
09 July 2018	7.83	6.90	8.35	8.98
10 July 2018	7.92	7.24	7.91	8.89
12 July 2018	7.75	6.88	7.73	8.45
16 July 2018	7.96	7.28	8.16	8.96
17 July 2018	7.99	6.84	7.54	9.13
19 July 2018	7.69	6.95	7.72	8.20
24 July 2018	7.34	6.96	7.56	8.38
26 July 2018	8.36	7.94	7.78	9.13

Table A2: Experimental Electrical Conductivity (E.C.) Results

EC Readings (uS/cm)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	471	-	-	-
06 June 2018	490	427	-	-
07 June 2018	435	393	429	-
11 June 2018	480	551	568	-
14 June 2018	426	554	454	470
18 June 2018	428	577	506	422
20 June 2018	45.9	478	590	517
21 June 2018	46.5	370	469	428
25 June 2018	53.8	302	337	327
27 June 2018	47.4	344	391	360
28 June 2018	47.6	256	293	296
03 July 2018	72.2	183	218	239
04 July 2018	394	172.7	157.7	292
05 July 2018	295	283	338	344
09 July 2018	447	520	397	222
10 July 2018	395	452	406	478
12 July 2018	47.2	548	509	494
16 July 2018	46.7	422	376	372
17 July 2018	394	348	300	275
19 July 2018	387	357	345	325
24 July 2018	587	553	590	522
26 July 2018	388	495	412	373

Table A3: Experimental Turbidity Results

Turbidity (NTU)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	3.56	-	-	-
06 June 2018	1.65	122.00	-	-
07 June 2018	5.11	146.00	4.07	-
11 June 2018	6.87	142.00	4.17	-
14 June 2018	2.81	95.10	2.71	4.91
18 June 2018	1.86	51.00	2.01	5.13
20 June 2018	1.63	50.20	2.29	1.39
21 June 2018	3.21	157.00	4.05	1.39
25 June 2018	3.31	268.00	2.03	3.49
27 June 2018	2.56	224.00	1.30	1.42
28 June 2018	7.98	167.00	1.99	1.04
03 July 2018	4.22	133.00	3.57	2.05
04 July 2018	3.05	101.00	3.05	2.09
05 July 2018	3.03	71.00	1.47	2.99
09 July 2018	2.20	40.30	2.13	1.11
10 July 2018	2.71	39.60	1.27	3.51
12 July 2018	2.95	9.88	7.90	0.44
16 July 2018	3.91	15.40	3.76	1.15
17 July 2018	3.85	30.50	2.69	1.47
19 July 2018	1.32	28.00	2.00	1.24
24 July 2018	2.19	27.20	1.22	1.54
26 July 2018	2.05	18.80	3.50	1.20

Table A4: Experimental Phosphate Results

Phosphate Test (mg/L)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	0.00	-	-	-
06 June 2018	0.01	0.05	-	-
07 June 2018	0.03	0.06	0.07	-
11 June 2018	0.00	0.05	0.13	-
14 June 2018	0.01	0.05	0.10	0.16
18 June 2018	0.01	0.06	0.13	0.09
20 June 2018	0.01	0.12	0.18	0.12
21 June 2018	0.01	0.06	0.18	0.11
25 June 2018	0.01	0.05	0.19	0.10
27 June 2018	0.00	0.09	0.15	0.11
28 June 2018	0.03	0.03	0.13	0.08
03 July 2018	0.00	0.08	0.17	0.12
04 July 2018	0.11	0.01	0.08	0.09
05 July 2018	0.12	0.00	0.05	0.08
09 July 2018	0.10	0.04	0.09	0.09
10 July 2018	0.08	0.04	0.12	0.08
12 July 2018	0.01	0.15	0.22	0.09
16 July 2018	0.01	0.12	0.14	0.12
17 July 2018	0.16	0.01	0.09	0.01
19 July 2018	0.10	0.05	0.10	0.09
24 July 2018	0.25	0.12	0.18	0.16
26 July 2018	0.08	0.15	0.21	0.14

Table A5: Experimental Nitrate Results

Nitrates (mg/L)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	2.121405751	-	-	-
06 June 2018	2.632587859	0.651757188	-	-
07 June 2018	3.974440895	0.000000000	1.226837061	-
11 June 2018	4.517571885	0.000000000	0.000000000	-
14 June 2018	3.303514377	0.000000000	0.000000000	16.49840256
18 June 2018	8.255591054	0.000000000	0.000000000	1.035143770
20 June 2018	0.000000000	0.000000000	38.12779553	0.555910543
21 June 2018	0.000000000	0.000000000	0.076677316	0.000000000
25 June 2018	0.000000000	0.000000000	0.000000000	0.000000000
27 June 2018	0.000000000	2.920127796	0.000000000	0.000000000
28 June 2018	0.000000000	0.000000000	0.000000000	0.000000000
03 July 2018	0.000000000	0.000000000	0.000000000	0.000000000
04 July 2018	3.463258786	0.000000000	0.000000000	0.000000000
05 July 2018	4.389776358	0.076677316	0.300319489	0.000000000
09 July 2018	5.412140575	0.843450479	1.386581470	2.504792332
10 July 2018	5.891373802	0.268370607	2.792332268	5.539936102
12 July 2018	0.000000000	0.000000000	0.000000000	1.162939297
16 July 2018	1.738019169	0.000000000	0.000000000	1.418530351
17 July 2018	6.562300319	1.194888179	1.38658147	2.217252396
19 July 2018	2.185303514	1.57827476	1.194888179	1.450479233
24 July 2018	3.686900958	0.000000000	0.619808307	3.335463259
26 July 2018	6.338658147	2.664536741	2.121405751	2.888178914

Table A6: Experimental Sulfate Results

Sulfates (mg/L)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	20.36363636	-	-	-
06 June 2018	9.45454546	22.18181818	-	-
07 June 2018	23.09090909	14.00000000	30.36363636	-
11 June 2018	23.09090909	17.63636364	18.54545455	-
14 June 2018	44.90909091	0.00000000	0.00000000	0.00000000
18 June 2018	0.00000000	0.00000000	0.00000000	0.00000000
20 June 2018	0.00000000	0.00000000	0.00000000	0.00000000
21 June 2018	0.00000000	0.00000000	14.90909091	16.72727273
25 June 2018	29.45454545	6.72727273	24.90909091	11.27272727
27 June 2018	44.00000000	6.72727273	15.81818182	7.63636364
28 June 2018	89.45454545	25.81818182	14.00000000	84.00000000
03 July 2018	7.636363636	39.45454545	4.909090909	9.454545455
04 July 2018	20.36363636	57.63636364	45.81818182	53.09090909
05 July 2018	55.81818182	0.00000000	0.363636364	8.545454545
09 July 2018	4.00000000	14.90909091	21.27272727	0.363636364
10 July 2018	0.00000000	0.00000000	21.27272727	43.09090909
12 July 2018	0.00000000	6.72727273	17.63636364	13.09090909
16 July 2018	15.81818182	14.00000000	22.18181818	9.454545455
17 July 2018	52.18181818	20.36363636	18.54545455	3.090909091
19 July 2018	24.90909091	48.54545455	57.63636364	25.81818182
24 July 2018	74.00000000	43.09090909	49.45454545	30.36363636
26 July 2018	73.09090909	69.45454545	15.81818182	9.454545455

Table A7: Experimental Chemical Oxygen Demand (COD) Results

COD (mg/L)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	22.75409836	-	-	-
06 June 2018	26.52459016	35.04918033	-	-
07 June 2018	24.55737705	24.88524590	23.73770492	-
11 June 2018	16.36065574	25.70491803	10.29508197	-
14 June 2018	17.18032787	29.14754098	14.39344262	15.86885246
18 June 2018	23.08196721	29.80327869	17.67213115	18.49180328
20 June 2018	24.39344262	30.29508197	25.70491803	19.47540984
21 June 2018	15.37704918	21.93442623	19.47540984	21.60655738
25 June 2018	12.75409836	9.47540984	17.01639344	29.31147541
27 June 2018	9.80327869	16.19672131	27.83606557	15.21311475
28 June 2018	23.08196721	16.85245902	20.13114754	19.31147541
03 July 2018	26.03278689	26.52459016	20.62295082	15.04918033
04 July 2018	22.26229508	20.13114754	10.62295082	18.98360656
05 July 2018	13.40983607	18.49180328	12.59016393	10.95081967
09 July 2018	21.60655738	27.83606557	20.29508197	8.98360656
10 July 2018	16.52459016	18.81967213	12.42622951	13.57377049
12 July 2018	11.11475410	19.47540984	31.27868852	9.14754098
16 July 2018	13.08196721	18.49180328	11.27868852	9.47540984
17 July 2018	26.19672131	8.16393443	12.59016393	8.49180328
19 July 2018	24.39344262	15.70491803	15.21311475	13.08196721
24 July 2018	14.22950820	11.44262295	13.40983607	9.14754098
26 July 2018	23.73770492	21.93442623	17.67213115	12.09836066

Table A8: Experimental Total Organic Carbon (TOC) Results

TOC (ppmCarbon)				
Date/Day	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	3.8654	-	-	-
06 June 2018	2.3309	6.797	-	-
07 June 2018	2.5506	2.6689	3.7493	-
11 June 2018	2.4227	3.862	3.2278	-
14 June 2018	2.4088	3.8228	2.9112	3.7306
18 June 2018	2.0347	2.8237	2.5719	2.4871
20 June 2018	0.9734	3.3309	2.6837	2.3657
21 June 2018	0.7851	2.7293	2.3361	2.5014
25 June 2018				
27 June 2018				
28 June 2018				
03 July 2018				
04 July 2018				
05 July 2018				
09 July 2018				
10 July 2018	22.4575	14.2134	11.3181	11.6367
12 July 2018	22.0147	13.1281	12.0344	11.0451
16 July 2018	11.9737	112.8783	114.0108	118.1901
17 July 2018	116.8655	62.3013	94.5178	107.2631
19 July 2018	118.9974	101.5232	98.1393	85.3367
24 July 2018	117.2633	104.2803	102.8933	20.5464
26 July 2018	95.2475	121.4494	138.9858	90.0014

Table A9: Experimental Total Coliforms Results

Total Coliforms MPN				
Date	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	809	-	-	-
06 June 2018	733.5	5818	-	-
07 June 2018	741.5	8164	6488	-
11 June 2018	519.5	5848	3448	-
14 June 2018	193500	2410	22398	1565
18 June 2018	9804	4925	4380	9768
20 June 2018	20	520	17240	2628
21 June 2018	3232	3830	19365	7746
25 June 2018	3744	2880	99315	2780
27 June 2018	196	1810	2260	4065
28 June 2018	205	4280	21420	955
03 July 2018	193500	12340	193500	12260
04 July 2018	193500	38790	51720	60165
05 July 2018	193500	173290	92080	43320
09 July 2018	38166.66667	20280	3900	8280
10 July 2018	14200	1960	9000	2120
12 July 2018	1300	7180	9040	4660
16 July 2018	500	13260	29400	400
17 July 2018	193500	10240	8960	2440
19 July 2018	193133.3333	35640	23740	193500
24 July 2018	13750	11100	4220	1680
26 July 2018	775	41280	4600	820

Table A10: *E. coli* Experimental Results

***E. coli* MPN**				
Date	**Feed**	**CW**	**CHAR**	**UV**
04 June 2018	41.00	-	-	-
06 June 2018	60.50	170.00	-	-
07 June 2018	67.00	243.00	62.00	-
11 June 2018	7.50	332.00	10.00	-
14 June 2018	91500.00	410.00	2740.00	0.00
18 June 2018	1112.00	535.00	173.3333333	1166.00
20 June 2018	0.00	0.00	205.00	82.00
21 June 2018	0.00	100.00	425.00	268.00
25 June 2018	0.00	0.00	1130.00	0.00
27 June 2018	0.00	0.00	50.00	50.00
28 June 2018	0.00	0.00	520.00	0.00
03 July 2018	91500.00	0.00	300.00	0.00
04 July 2018	38505.00	100.00	860.00	730.00
05 July 2018	43320.00	7330.00	3360.00	6480.00
09 July 2018	4033.333333	200.00	0.00	820.00
10 July 2018	0.00	200.00	0.00	0.00
12 July 2018	250.00	5560.00	5740.00	4660.00
16 July 2018	0.00	2600.00	0.00	820.00
17 July 2018	91350.00	0.00	0.00	0.00
19 July 2018	91500.00	17720.00	13000.00	16520.00
24 July 2018	5325.00	1040.00	200.00	200.00
26 July 2018	0.00	1200.00	400.00	1460.00

Table A11: Aluminium Concentration Experimental Results

Al (µg/L)				
Date	**Feed**	**CW**	**Char**	**UV**
04 June 2018	5.12	-	-	-
06 June 2018	35.50	28.60	-	-
07 June 2018	60.50	221.00	58.90	-
11 June 2018	13.10	25.40	32.80	-
14 June 2018	21.60	12.00	16.80	11.00
18 June 2018	19.50	19.00	21.10	32.30
20 June 2018	29.10	2.46	1.39	68.50
21 June 2018	96.20	23.20	28.60	13.60
25 June 2018	31.80	2.99	6.19	44.50
27 June 2018	48.30	4.06	18.40	15.20
28 June 2018	100.00	16.30	37.10	18.40
03 July 2018	351.00	62.70	26.40	88.80
04 July 2018	110.00	30.20	39.80	23.80
05 July 2018	23.20	57.90	8.85	22.70
09 July 2018	150.00	45.10	3.52	22.70
10 July 2018	53.10	47.20	21.10	74.40
12 July 2018	42.70	162.00	40.80	8.85
16 July 2018	6.19	63.20	21.10	5.12
17 July 2018	9.38	54.10	26.40	23.80
19 July 2018	1.92	14.70	10.40	45.10
24 July 2018	4.59	37.60	2.35	0.33
26 July 2018	14.70	46.70	5.65	6.36

Table A12: Manganese Concentration Experimental Results

Manganese (µg/L)				
Date	**Feed**	**CW**	**Char**	**UV**
04 June 2018	5.67	-	-	-
06 June 2018	17.90	13.30	-	-
07 June 2018	17.70	50.20	18.20	-
11 June 2018	18.60	23.20	20.10	-
14 June 2018	28.40	33.10	8.26	5.56
18 June 2018	129.00	33.70	24.00	8.58
20 June 2018	3.12	19.30	3.49	6.30
21 June 2018	4.60	4.39	5.45	2.43
25 June 2018	2.75	2.59	2.70	7.20
27 June 2018	21.30	3.81	5.88	7.26
28 June 2018	17.50	7.26	8.64	11.80
03 July 2018	41.70	21.20	4.34	16.80
04 July 2018	21.00	6.83	11.10	27.00
05 July 2018	26.60	4.60	4.66	16.50
09 July 2018	51.80	17.90	6.83	13.70
10 July 2018	12.50	22.70	29.20	9.22
12 July 2018	20.40	14.70	11.70	3.54
16 July 2018	10.10	20.50	8.90	8.64
17 July 2018	18.90	20.70	5.08	16.60
19 July 2018	58.90	8.16	6.30	26.30
24 July 2018	6.73	14.00	2.16	39.70
26 July 2018	12.90	15.30	2.91	4.34

Table A13: Lead Concentration Experimental Results

Lead (µg/L)				
Date	**Feed**	**CW**	**Char**	**UV**
04 June 2018	0.48	-	-	-
06 June 2018	2.59	2.23	-	-
07 June 2018	3.73	6.23	2.30	-
11 June 2018	1.76	1.38	2.18	-
14 June 2018	2.22	1.16	2.31	0.62
18 June 2018	1.73	1.16	10.00	2.90
20 June 2018	0.74	0.95	0.99	1.54
21 June 2018	1.52	0.94	0.65	0.87
25 June 2018	1.08	0.70	0.85	1.22
27 June 2018	1.81	0.84	1.91	1.47
28 June 2018	1.67	1.79	1.10	2.11
03 July 2018	11.60	3.51	1.49	3.81
04 July 2018	4.33	3.92	5.31	1.89
05 July 2018	4.44	1.73	1.34	1.43
09 July 2018	3.32	4.88	2.70	2.84
10 July 2018	3.69	2.95	4.15	3.11
12 July 2018	3.01	6.54	6.46	4.25
16 July 2018	1.23	1.58	2.31	1.68
17 July 2018	3.56	2.53	1.65	3.81
19 July 2018	1.97	3.15	1.75	6.61
24 July 2018	1.29	4.87	0.86	1.91
26 July 2018	1.68	1.84	1.29	1.65

Appendix B: Weather Recorded on Sampling Days – Table

Table B1: Weather Recorded on Sampling Days

Weather - University of the Witwatersrand, Braamfontein, Johannesburg					
Date	**Min Temp (˚C)**	**Max Temp (˚C)**	**Humidity (%)**	**Precipitation (%)**	**Wind (km/hr)**
04 June 2018	2	15	55	0	23
06 June 2018	4	17	30	0	8
07 June 2018	4	18	33	0	23
11 June 2018	1	18	43	0	9
14 June 2018	6	14	52	0	16
18 June 2018	7	20	86	0	8
20 June 2018	6	20	44	0	5
21 June 2018	9	18	66	0	17
25 June 2018	6	18	36	0	5
27 June 2018	7	17	59	0	5
28 June 2018	6	18	66	0	10
03 July 2018	1	14	43	0	26
04 July 2018	2	15	37	0	5
05 July 2018	1	15	69	0	5
09 July 2018	5	15	84	40	11
10 July 2018	6	16	55	24	14
12 July 2018	4	14	55	0	16
16 July 2018	0	13	36	0	10
17 July 2018	2	13	28	0	18
19 July 2018	2	14	72	0	8
24 July 2018	4	18	27	0	8
26 July 2018	7	20	35	0	11

Appendix C: Preliminary ICP-MS Heavy Metal Test Results – Table

Table C1: Preliminary ICP-MS Heavy Metal Test Results

Metal / Element	River Water (Conentration in µg/L)	Rain Water (Conentration in µg/L)
Ag	6	8
Al	386	624
As	<2	<2
B	58	27
Ba	46	13
Be	<2	<2
Bi	<2	<2
Ca	13967	8257
Cd	1	1.8
Ce	4	6.3
Co	52	58
Cr	45	54
Cs	9	9
Cu	8	17
Dy	5	4.6
Er	6	5.6
Eu	5	4.6
Fe	801	233
Ga	2.2	2
Gd	4.6	5
Ge	<5	<5
Hf	2	1.7
Hg	<5	<5
Ho	2	2
In	<2	<2

K	1956	291
La	4	10
Li	<2	<2
Lu	0.1	0.1
Mg	13914	668
Mn	319	149
Mo	<10	<10
Na	10006	254
Nb	21	20
Nd	5	8
Ni	48	76
P	<2000	<2000
Pb	24	18
Pr	4	4.8
Rb	0.83	<0.3
S	10190	3837
Sb	1.5	4.4
Sc	<2	<2
Se	<5	<5
Si	4716	980
Sm	4.7	5
Sn	2.8	4.3
Sr	157	32
Ta	<2	<2
Tb	2.4	2.5
Te	2	1.87
Th	1.25	7
Ti	111	60
Tl	<2	<2
Tm	0.57	0.6

U	2.23	2.2
V	<5	<5
W	6.5	6.9
Y	<2	<2
Yb	3.1	3.1
Zn	6	7
Zr	13	7

Appendix D: Emergent, Herbaceous Wetland Plant Species - Table

Table D1 below presents a list of various emergent plant species that are appropriate for use in CWs. This table has been adapted from Kadlec et al. (2000).

Table D1: Emergent, Herbaceous Wetland Plant Species

Scientific Name	Common Name	Max. Water Depth (m)	Flooding Duration (%)
Alternanthera philoxeroids	Alligator Weed	1.0	70 - 100
Canna spp.	Canna Lilies	0.25	50 -100
Carex spp.	Sedges	0.25	50 – 100
Ceratophyllum spp.	Coontail	>3	75 – 100
Caladium jamaicense	Sawgrass	0.25	50 – 100
Colocasia esculenta	Wild Taro	0.5	25 – 100
Cyperus spp.	Sedges	0.5	50 -100
Eleocharis spp.	Spikerushes	0.5	50 – 100
Elodea spp.	Waterweed	>3	75 - 100
Glyceria spp.	Mannagrass	0.3	0 – 100
Hydrocloa caroliniensis	Watergrass	1.0	75 – 100
Iris spp.	Iris	0.2	50 - 100
Juncus spp.	Rushes	0.25	50 – 100
Lemna spp.	Duckweed	None	75 – 100
Ludwigia spp.	Water primroses	0.5	70 – 100
Panicum hemitomon	Maidencane	0.3	50 – 100
Panicum repens	Torpedo grass	0.5	50 – 100
Peltandra spp.	Spoon flowers	0.25	50 – 100
Phalaris arundinacea	Reed canarygrass	0.3	13 - 100
Phragmites australis	Common reed	0.5	70 – 100
Polygonum spp.	Smartweeds	0.25	50 – 100
Pontederia spp.	Pickerelweeds	0.25	70 – 100
Rhynchospora spp.	Beak-rush	0.5	50 – 100

Sagittaria spp.	Arrowheads	0.5	50 – 100
Saururus cernuus	Lizard's-tail	0.2	50 – 100
Scirpus spp.	Bulrush	1.5	75 – 100
Sparganium spp.	Bur-reed	0.5	70 - 100
Sphagnum spp.	Sphagnum mosses	0.1	75 – 100
Thalia geniculata	Arrowroot	0.75	75 – 100
Typha spp.	Cattail	0.75	70 – 100
Zizania aquatica	Wild rice	1.0	70 – 100
Zizaniopsis milacea	Southern wild rice	1.0	70 - 100