The applicability of advanced treatment processes in the management of deteriorating water quality in the Mid-Vaal river system

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ABSTRACT

The main objective of this study was to determine the applicability of advanced water treatment processes namely granular activated carbon (GAC) adsorption, ultraviolet (UV) light disinfectant and ozone in the management of deteriorating water quality in the Mid-Vaal River system for drinking purposes. Both the scarcity and the deteriorating quality of water in South Africa can be addressed by investigating advanced water treatment processes such as GAC adsorption, UV light disinfectant and ozone. Previously disregarded water resources have the potential to be purified and advanced treatments can improve water quality where conventional water treatments have failed. In addition, advanced treatment processes can be applied to treat used water.

The two sampling sites selected for the study, Rand Water Barrage (RWB) and Midvaal Water Company (MWC), are both located in the Middle Vaal Water Management Area with RWB upstream of MWC. RWB uses GAC adsorption and UV light disinfection and MWC uses ozone as pre- and intermediate treatment process steps for water purification.

The quality of the source water at both sampling sites was determined by analysing the physical and chemical characteristics as well as the algal and invertebrate compositions of the source water. The physical and chemical water quality variables measured included pH, conductivity, turbidity, dissolved organic carbon (DOC), total organic carbon (TOC), total photosynthetic pigments (TPP), microcystin and geosmin.

The source water of both sites was characterised as hypertrophic on account of high chlorophyll concentrations. The water quality of the two sites was distinctly different and a downstream change was observed. The source water of RWB was characterised by high microcystin, geosmin, DOC, TOC and conductivity measurements and dominated by Bacillariophyceae (diatoms) and Cyanophyceae (blue-green bacteria). Problematic species that were present in the source water of RWB included *Aulacoseira* sp., other unidentified centric diatoms, *Pandorina* sp., *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp., *Cryptomonas* sp., *Ceratium* sp. and *Trachelomonas* sp. The source water of MWC was characterised by high pH, turbidity and TPP measurements and was dominated by Chlorophyceae (green algae) and Bacillariophyceae (diatom) species. Problematic algal species that were present in the source water of MWC included *Cyclotella* sp., *Coelastrum* sp., *Pediastrum* sp. and *Scenedesmus* sp. The source water of MWC was deemed to be of a better quality due to the lower Cyanophyceae concentrations and

lower microcystin levels. The invertebrate composition of both sites was similar with Rotatoria as the dominant invertebrate group.

The efficacy of GAC adsorption/UV light disinfection/ozonation on restoring the physical and chemical characteristics of the source water at both sampling sites as well as the algal and invertebrate compositions was determined by ascertaining the nature of the change in or the percentage removal of a water quality variable. The potable water of both sites complied with the standards of water intended for domestic use except for the conductivity at RWB that was slightly elevated. The phytoplankton was removed effectively from the source water of both sites but the removal of invertebrates was unsatisfactory. GAC adsorption and filtration proved to be more effective in the removal of TPP, turbidity, DOC, microcystin and geosmin than ozone. Ozone effected an increase in DOC. UV light disinfection had no or little effect on restoring the water quality variables investigated in this study.

KEYWORDS: advanced water treatment, granular activated carbon (GAC), ultraviolet (UV) light, ozone

OPSOMMING

Die hoofdoel van hierdie studie was om die toepaslikheid van gevorderde waterbehandelingsprosesse, naamlik korrelgeaktiveerde koolstof (GAC) adsorpsie, UV-lig ontsmetting en osoon, in die bestuur van verswakkende watergehalte in die Middel-Vaalrivierstelsel vir drinkdoeleindes te bepaal. Beide die skaarsheid en die verswakkende gehalte van water in Suid-Afrika kan aangespreek word deur gevorderde waterbehandelingsprosesse soos GAC adsorpsie, UV-lig ontsmetting en osoon te ondersoek. Waterbronne wat voorheen onbruikbaar was, het weens die prosesse die potensiaal om gesuiwer te word en gevorderde behandelings kan die watergehalte verbeter waar konvensionele waterbehandelings misluk het. Daarbenewens kan gevorderde behandelingsprosesse ook toegepas word om gebruikte water weer te behandel.

Die twee watersuiweringsaanlegte wat gekies is vir hierdie studie, naamlik Rand Water Barrage (RWB) en Midvaal Water Maatskappy (MWM), is albei in die Middel-Vaal Waterbestuursarea met RWB stroomop van MWM geleë. RWB gebruik GAC adsorpsie en UV-lig ontsmetting, terwyl MWM osoon as voor- en intermediêre behandelingprosesstappe vir watersuiwering gebruik.

Die kwaliteit van die bronwater by beide plekke is bepaal deur die ontleding van die fisiese en chemiese eienskappe, sowel as deur die ontleding van die alg- en soöplanktonsamestellings van die bronwater. Die fisiese en chemiese waterkwaliteitveranderlikes wat gemeet is sluit in pH, geleiding, troebelheid, opgeloste organiese koolstof (DOC), totale organiese koolstof (TOC), totale fotosintetiese pigmente (TPP), mikrosistien en geosmien in.

Die bronwater van beide watersuiweringsaanlegte is gekaraktiseer as hipertrofies op grond van hoë konsentrasies van chlorofil. Die kwaliteit van die water van hierdie twee plekke is duidelik verskillend en 'n stroomaf verandering is waargeneem. Die bronwater van RWB is gekenmerk deur hoë mikrosistien, geosmien, geleiding, DOC en TOC vlakke en die fitoplankton samestelling word oorheers deur Bacillariophyceae (diatome) en Cyanophyceae (blougroenbakterieë). Problematiese algspesies wat in die bronwater van RWB voorgekom het, sluit *Aulacoseira* sp., ongeïdentifseerde sentriese diatome, *Pandorina* sp., *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp., *Cryptomonas* sp., *Ceratium* sp. en *Trachelomonas* sp. in. Die bronwater van MWM is gekenmerk deur hoë pH, troebelheid en TPP en die fitoplankton samestelling word oorheers deur Chlorophyceae (groen alge) en Bacillariophyceae (diatome). Problematiese algspesies wat in die bronwater van MWM voorgekom het sluit *Cyclotella* sp., *Coelastrum* sp., *Pediastrum* sp. en *Scenedesmus* sp in. Die bronwater van MWM was van 'n beter gehalte op grond van die laer konsentrasie van Cyanophyceae en die laer mikrosistienvlakke. Die soöplanktonsamestelling van beide plekke was soortgelyk met Rotatoria as die dominante soöplanktongroep.

Die effektiwiteit van GAC adsorpsie/UV-lig ontsmetting/osonisasie op die herstel van die fisiese en chemiese eienskappe van die bronwater by beide plekke sowel as op die herstel van die alg- en soöplanktonsamestellings is deur die aard van die verandering in ‘n waterkwaliteitveranderlike of die persentasie verwydering van 'n waterkwaliteitveranderlike bepaal. Die drinkwater van beide plekke het voldoen aan die standaard vir water vir huishoudelike gebruik, behalwe vir die geleiding by RWB wat effens hoog was. Die fitoplankton is effektief uit die bronwater van beide plekke verwyder, maar die verwydering van die soöplankton was onbevredigend. GAC adsorpsie was meer doeltreffend in die verwydering van TPP, troebelheid, DOC, mikrosistien en geosmien as osoon. Osoon het 'n toename in DOC bewerkstellig. UV-lig ontsmetting het min of geen effek gehad op die herstel van die waterkwaliteitveranderlikes wat ondersoek was in hierdie studie.

SLEUTELWOORDE: gevorderde waterbehandeling, korrelgeaktiveerde koolstof, ultraviolet-lig ontsmetting, osoon

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LIST OF ABBREVIATIONS

CA	Correspondence analysis
CCA	Canonical correspondence analysis
CMA/s	Catchment Management Agency/Agencies
DOC	Dissolved organic carbon
DNA	Deoxyribonucleic Acid
DWA	Department of Water Affairs
DWAF	Department of Water Affairs and Forestry
ELISA	Enzyme-Linked Immuno Sorbent Assay
GAC	Granular activated carbon
MIB	2-methylisoborneol
MWC	Midvaal Water Company
NWRS	National Water Resource Strategy
PCA	Principal component analysis
RNA	Ribonucleic Acid
RWB	Rand Water Barrage
RWI/s	Regional Water Institute/Institutions
SANAS	South African National Accreditation System
SANS	South African National Standards
THMs	Trihalomethanes
TOC	Total organic carbon
TPP	Total photosynthetic pigments

TWQR	Target Water Quality Range
USEPA	United States Environmental Protection Agency
UV	Ultraviolet
VRS	Vaal River system
WEF	World Economic Forum
WHO	World Health Organisation
WSA/s	Water Service Authority/Authorities
WSP/s	Water Service Provider/Providers
WTW	Water Treatment Works
WUA/s	Water User Association/Associations

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CHAPTER 1: INTRODUCTION

A water supply crisis is a harsh reality not just on a national but on a global scale. The World Economic Forum's (WEF) recent Global Risks report lists a water supply crisis as one of the top five global risks to materialise over the next decade. Water shortage was not considered a risk prior to 2012 by the WEF. Currently water scarcity features as a significant risk to society in terms of likelihood and impact that will require global economic and environmental resilience (WEF, 2013). Based on current population trends and water usage models there is a strong indication that most African countries will surpass the limits of their utilisable water resources by 2025 (Ashton, 2002). The predicted increase in global temperatures with resulting climate changes will place additional demands on over-utilised water resources in the form of droughts.

Water security in South Africa, a water-stressed country, is a topic of concern. This is acknowledged by the second National Water Resource Strategy (NWRS) released in 2013. The NWRS is legally binding on all authorities and other parties responsible for the implementation of the National Water Act (Act 36 of 1998) and provides a framework for the effective management of national water resources (South Africa, 1998). The NWRS indicates that based on Reconciliation Strategies, the availability of surface water and its remaining development potential will not be sufficient to meet the water demands of a growing South African population (DWA, 2013). An increase in population is associated with an increase in urbanisation and agricultural and industrial activities. These anthropogenic activities in turn contribute to a decrease in water quality as a result of eutrophication, increased salinity, acid mine drainage and faecal pollution.

The Department of Water Affairs (DWA) is the responsible entity for the water value chain with the assistance of Catchment Management Agencies (CMAs), Regional Water Institutions (RWIs) and other national entities (Figure 1.1). The primary responsibility for the provision of potable water to consumers remains with the municipalities or the Water Services Authorities (WSAs). This responsibility includes providing an acceptable quality of drinking water at the point of distribution in addition to meeting the demand for drinking water.

In most cases conventional water treatment steps such as coagulation, flocculation, sedimentation, sand filtration and chlorination will provide safe drinking water but the removal of certain harmful organisms cannot be guaranteed (Ewerts, 2010). A consumer's perception of the quality of drinking water is often based on the aesthetic properties of water such as taste, smell and appearance even if the actual risks are low (DWAF, 1998). According to the Drinking Water Quality Framework for South Africa, the WSA is required to undertake specific actions to ensure that drinking water quality standards are met (DWAF, 2009a). In order for a WSA to meet consumer demands to supply not

only enough water but aesthetically pleasing water of a high quality, the investigation and application of alternative advanced water treatment processes will have to be considered.

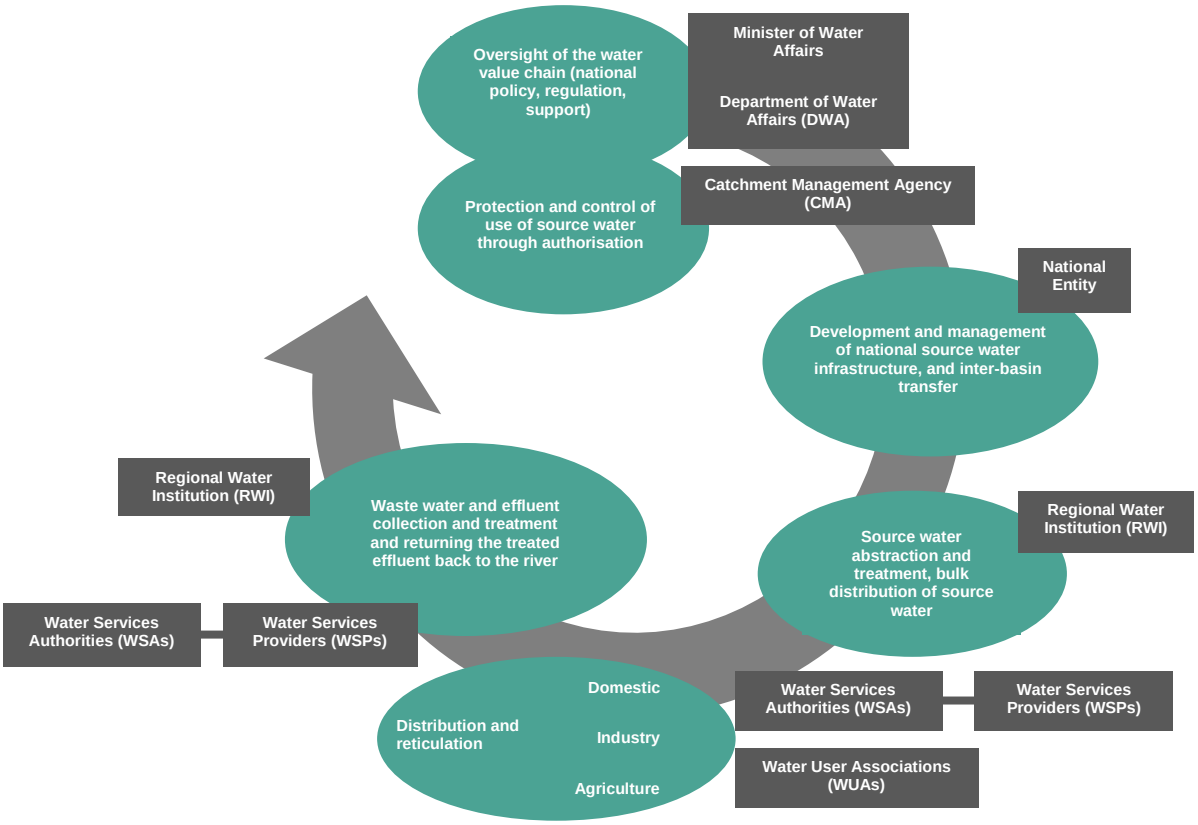


Figure 1.1: Responsible entities for South Africa's water value chain (adapted from DWA, 2013).

In the light of the recurring water scarcity theme, the NWRS advocates a "Source-to-Tap-to-Source" strategy as a sustainable solution for the supply of water. This approach necessitates extensive water re-use and the subsequent advanced treatment of water. The DWA developed a National Strategy for Water re-use that outlines a considered approach to the implementation of water re-use projects. Water re-use can be direct or indirect and reclaimed water can be used for various purposes as illustrated by Figure 1.2. The quality and the intended purpose of the used water will determine the appropriate treatment technology. The direct re-use of water has not been implemented in South Africa but successful potable water re-use schemes are in operation in other countries (DWA, 2011a).

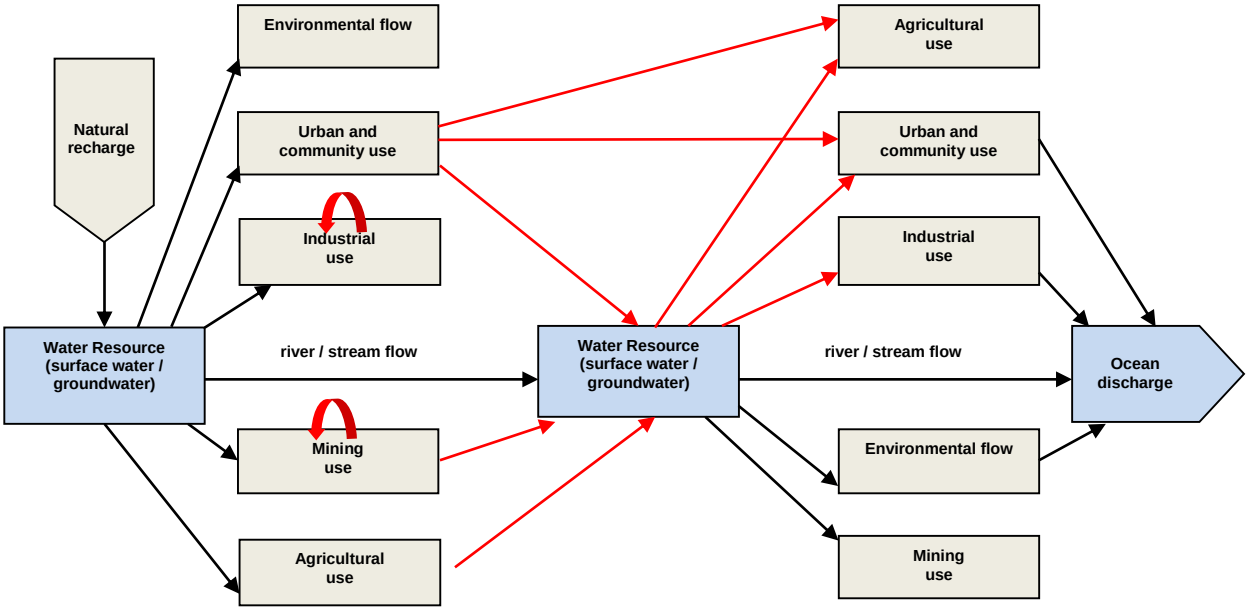


Figure 1.2: The various purposes of reclaimed water (adapted from DWA, 2011a).

Used water can be treated to a standard fit for drinking purposes and the National Strategy for Water re-use provides a list of possible applicable water treatment technologies for water re-use (Table 1.1).

Table 1.1: Applicable water treatment technologies for water re-use (DWA, 2011a).

Category of Pollutants	Applicable Technologies
Macro-organics, COD and BOD5	• Biological treatment (activated sludge, trickling filtration, fixed film reactors, membrane bioreactors) • Chemical coagulation/flocculation and clarification
Particulate and suspended solids	• Chemical coagulation/flocculation and clarification • Granular media filtration • Membrane filtration
Nutrients – Nitrogen	• Biological nitrogen removal (nitrification/denitrification) • Air stripping (ammonia) • Chemical coagulation/flocculation and solids separation
Microbiological Agents: • Bacteria • Viruses • Parasites	• Membrane filtration • Chemical disinfection (chlorine, bromine compounds etc.) • Ultraviolet (UV) radiation
Salinity, inorganic salts	• Precipitation

Category of Pollutants	Applicable Technologies
	• Ion exchange • Membrane desalination (nanofiltration/reverse osmosis)
Metals	• Precipitation • Chemical adsorption • Membrane separation
Micro-organics: • Volatile Organics • Pesticides • Pharmaceuticals • Endocrine Disruptors	• Advanced oxidation (H_2O_2/UV) • Adsorption by activated carbon (granular/powder) • Membrane separation (nanofiltration/reverse osmosis) • Biologically enhanced adsorption (BAC)
Disinfection by-products	• Modify disinfection agent in upstream processes • Advanced oxidation • Adsorption by powdered or granular activated carbon • Membrane separation (nanofiltration/reverse osmosis)
Radionuclides	• Precipitation • Chemically enhanced adsorption • Membrane separation (nanofiltration/reverse osmosis)

Both the scarcity and the deteriorating quality of water in South Africa can be addressed by investigating advanced water treatment processes such as granular activated carbon (GAC) adsorption, ultraviolet (UV) light disinfectant and ozone. Previously disregarded water resources have the potential to be purified and advanced treatments can improve water quality where conventional water treatments have failed. In addition, advanced treatment processes can be applied to treat used water. GAC adsorption removes organic substances such as taste and odour compounds as well as many metals. UV light as a method of disinfection renders microorganisms harmless or kills them through the disruption of deoxyribonucleic acid (DNA) and the membrane structure of the microorganism. The action of ozone can be classified as both an oxidant and a germicidal compound. Ozone is used primarily for taste and odour control in most installations. It enhances coagulation and micro-flocculation (Schutte, 2006).

The main objective of this study is to determine the applicability of advanced water treatment processes namely GAC adsorption, UV light disinfectant and ozone in the management of deteriorating water quality in the Mid-Vaal River system for drinking purposes.The two sampling sites, Rand Water Barrage (RWB) and Midvaal Water Company (MWC), both abstract water from the Mid-Vaal River system. RWB uses GAC adsorption and UV light disinfection and MWC ozone

as pre- and intermediate treatment process steps for water purification. The specific aims of this study are:

- To determine the quality of the source water at both sampling sites by analysing the physical and chemical characteristics as well as the algal and invertebrate compositions of the source water;
- To determine the efficacy of GAC adsorption/UV light disinfection/ozonation on changing the physical and chemical characteristics of the source water at both sampling sites;
- To determine the efficacy of GAC adsorption/UV light disinfection/ozonation on the removal of phytoplankton and invertebrates from the source water at both sampling sites.

This research study will contribute to the current understanding of the applicability and efficacy of different advanced treatment processes and its effect on source water quality. Water utilities that make use of ozonation or UV light disinfection or GAC adsorption or that are in the decision-making stages of which treatment process to use, will find this research beneficial. This study will play a significant role in the determination of which advanced treatment process will have the most significant impact on the quality of local source water.The obtained results can be compared with their own data and thereby assist in the making of well-informed decisions regarding plant optimisation.

The data obtained from this study and the statistical analyses thereof can be used to compile a set of guidelines based on the use of these methods in water purification in the Mid-Vaal River system. Results obtained from this study can furthermore be beneficial in the determination of appropriate treatment technologies for the purification of used water.

CHAPTER 2: LITERATURE REVIEW

2.1. Conventional versus advanced water treatment processes

There is no pure water available for general use in the natural environment as all water contains some contaminants commonly in the form of suspended solids, micro-organisms and dissolved substances (Van Duuren, 1997). As such, the quality of water in its natural, raw state is generally not fit for drinking purposes. The following water quality aspects are essential to consider in the water treatment process:

- Water must not represent a health risk due to chemical or microbiological contamination;
- Water must be aesthetically pleasing;
- Water must not have damaging effects on either the distribution system or on consumer equipment (Van Duuren, 1997).

Consumer health is the single most important aspect to consider in water treatment. In addition, a consumer is also entitled to domestic water of an aesthetically acceptable quality. The primary aim of water treatment for drinking purposes is therefore to produce uncontaminated water by the removal of undesirable elements from source water through selected treatment processes (Schutte, 2006).

Water treatment process selection is determined by the quality of the source water as well as the intended purpose of the treated water. A process can either remove pollutants or change the nature of the source water by the addition of chemicals (Van Duuren, 1997). Water treatment processes are combined to form a process train in order to produce potable water that meets the national drinking water quality standards.

Conventional water treatment methods include coagulation, flocculation, sedimentation and/or flotation, sand filtration and chlorination. These unit processes work to remove particles, naturally occurring organic material and microorganisms. Substances in source water can be suspended, colloidal or in solution. Colloidal particles are electrically negatively charged and will not settle. Coagulation converts these stable particles to unstable particles through the addition of a coagulant to the source water so that flocs can be formed through the process of flocculation (Schutte, 2006). Rand Water uses hydrated lime and activated sodium silicate as coagulants and ferric chloride to aid flocculation (Rand Water, 2014).

Spellman (2003) describes the goal of flocculation as the formation of dense flocs to trap the suspended and colloidal particles that will eventually settle. Coagulation-flocculation contribute to

the removal of microorganisms, colour and turbidity. Sedimentation removes solids by gravity. Water moves slowly through the sedimentation tank while sludge accumulates at the bottom of the tank. Flocs are removed from the water through the process of filtration as the water passes through granular material during this process and suspended and colloidal particles are separated from the water. The flotation process entails the formation of small air bubbles that attach to the flocs causing them to rise to the surface where they are collected as froth. The last unit in the water treatment process train is usually disinfection that is mostly accomplished by adding chlorine or chlorine compounds to the water in order to destroy harmful organisms.

As a result of the deteriorating quality of water sources, conventional water treatment processes do not always succeed in purifying water to a quality that meets drinking water standards. According to Ewerts *et al.* (2013), conventional methods used at South Africa's largest water treatment plant were not effective in removing geosmin as the result of the release of organic compounds by cyanobacteria cells. Although *Ceratium* cells were removed effectively during sand filtration, a large number of these cells can put major strain on sand filters.

Advanced water treatment processes are non-conventional treatment processes that are used for specific purposes other than, or in addition to the clarification and disinfection of water such as the removal of specific substances. These processes can be used individually, in combination with conventional processes or in combination with conventional processes and other advanced processes to address water quality problems (DWAF, 2002).

Table 1.1 provides a list of alternative advanced treatment technologies. For the purposes of this study, overviews of the use of GAC adsorption, UV light disinfection and ozone as advanced treatment technologies are provided.

2.2. Granular activated carbon (GAC) adsorption

Two basic activated carbon adsorption systems are used in water treatment namely granular activated carbon (GAC) that uses carbon as a bed of carbon granules and powdered activated carbon (PAC) that uses carbon in a powdered form (Schutte, 2006). Charcoal or carbon can accomplish multiple functions such as adsorption and filtration when used as a filter medium. Activated carbon in particular is very adaptable as a filter medium in water treatment as it can physically separate suspended solids from water in addition to the adsorption of materials (Cheremisinoff, 2004).

Carbon material is activated through a series of processes namely dehydration, carbonisation and activation. The carbon material to be converted is initially heated to 170°C to remove water. Subsequently the temperature is raised to 275°C to effect carbonisation and the conversion of the

organic matter to elemental carbon. Activation is done with the use of superheated steam, 750-950°C, that burns off by-products and enlarges the surface area by expanding the pores (Cheremisinoff, 2004).

Granular activated carbon particles consist of a highly porous graphite structure over a broad range of pore sizes (Figure 2.1). As a result, activated carbon particles have a large surface area ranging from 450 to 1800 m^2/g that augments the adsorption process. The large surface area and the pore structure of activated carbon are major factors in the adsorption process. The macropores provide a passage to the micropores inside the particle. Micropores are developed during the activation process and contribute considerably to the large surface area. Carbon is known to possess the strongest physical adsorption forces of any material known to mankind (Cheremisinoff, 2004).

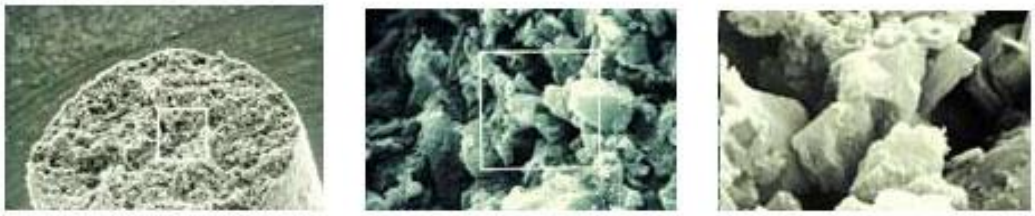


Figure 2.1: The porous structure of activated carbon (Cabot, 2013).

Water contains dissolved organic substances that could be harmful or have negative effects on human health such as substances that cause taste and odour problems, organic pesticides and disinfection by-products. Dissolved organics can only be removed by processes such as activated carbon adsorption and reverse osmosis (Schutte, 2006).

Granular activated carbon is placed in columns through which the water flows at a slow rate during water treatment. As a result of this close contact, organic molecules diffuse into and inside the carbon pores where mainly Van der Waals, chemical and electrical adsorption forces keep the molecules attached to the carbon. Carbon treatment is costly. Therefore granular activated carbon columns are usually the last treatment process to be used after as much as possible of all contaminants have been removed by previous processes, i.e. sedimentation and sand filtration, before chlorination (Schutte, 2006).

2.3. Ultraviolet (UV) light disinfection

UV light lies between X-rays and visible light in the electromagnetic spectrum and is usually invisible to the human eye. UV light consists of four spectrums namely Vacuum-UV, UV-C, UV-B

and UV-A (Figure 2.2). The optimal range of UV light for disinfection is between 200 and 300 nm, UV-B and UV-C, due to its germicidal action on microorganisms (USEPA, 2006).

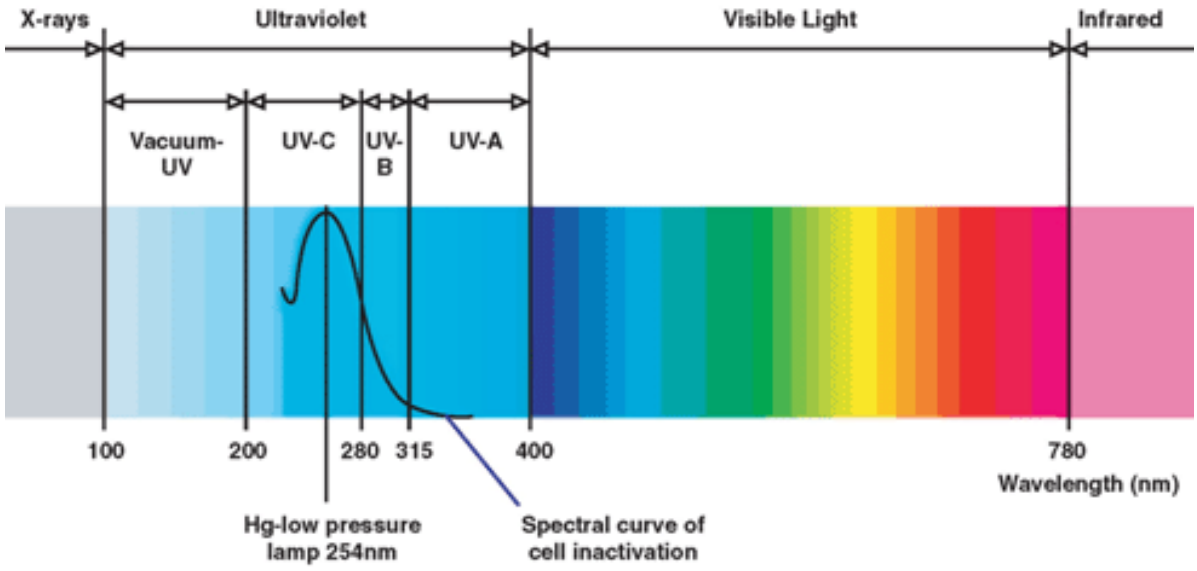


Figure 2.2: Ultraviolet light in the electromagnetic spectrum (UVIR, 2011).

The use of UV light as a disinfectant of drinking water involves the generation of UV light with the required disinfectant properties and the delivery of the light to microorganisms. UV lamps generate UV light by the application of a voltage across a gas mixture that results in the discharge of photons. Mercury gas is most often used for the gas mixture as it releases light in the germicidal wavelength range. Typical UV equipment used in water treatment consists of closed or open-channel UV reactors, UV lamps, lamp sleeves, UV and temperature sensors, ballasts, flow meters, UV transmittance analysers and automatic cleaning mechanisms (Figure 2.3) (USEPA, 2006).

According to the United States Environmental Protection Agency (USEPA) only closed-channel UV reactors where water flows under pressure are in use for the UV light disinfection of drinking water. Mercury arc lamps enclosed in quartz sleeves are housed within the reactor. Lamp configuration is typically perpendicular to water flow to optimise dose delivery. Ballasts provide power to the UV lamps for the generation of an arc which equates to the production of UV light. UV sensors and flow meters, and, in some cases, UV transmission analysers, monitor the reactor's dose delivery (USEPA, 2006).

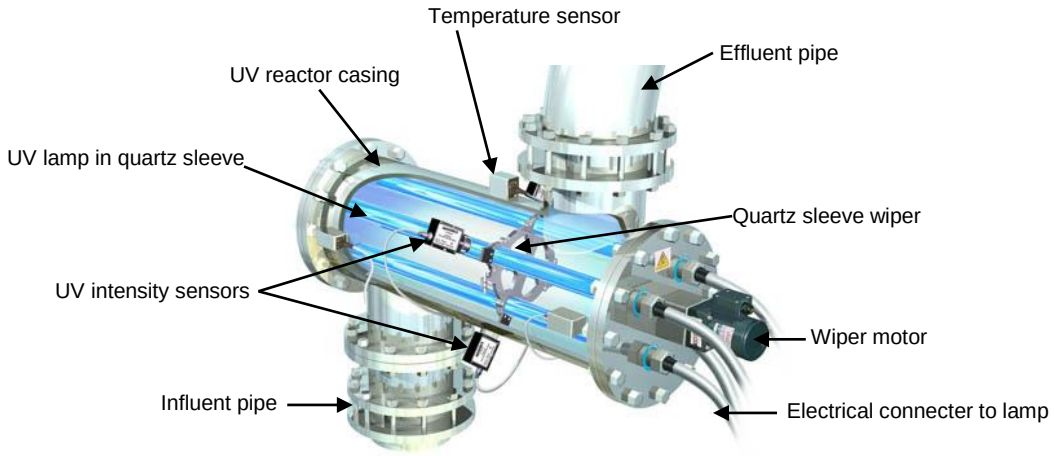


Figure 2.3: Ultraviolet disinfection equipment (adapted from Halma, 2006).

UV light inactivates harmful microorganisms such as *Giardia* and *Cryptosporidium* as result of photochemical damage to their deoxyribonucleic acid (DNA) and ribonucleic acid (RNA). Nucleotides, the building blocks of cellular DNA and RNA, absorb UV radiation that promotes the formation of bonds between adjacent nucleotides creating thymine-thymine dimers. This renders the microorganism incapable of reproducing as a sufficient number of dimers prevents its DNA and RNA from replicating. The microbial repair mechanisms of microorganisms is not regarded as protection against inactivation by UV light. The minimum UV dosage requirements for the inactivation of microorganisms is not universally agreed upon as site specific factors such as source water quality and the level of microbiological contamination have to be considered when determining the optimal UV dosage (Wright & Cairns, 2002).

2.4. Ozonation

Oxygen, in addition to forming the stable O_2 (dioxygen), can also exist in another very reactive molecular form namely ozone (O_3). The passage of an electrical discharge through ordinary O_2 can generate this unstable molecule (Brady & Senese, 2004). Although ozone is regarded as an unstable gas, it is a powerful oxidant. In water treatment, ozone has been effective in a number of applications such as colour and odour removal, the oxidation of iron and manganese, microorganism inactivation and the destabilisation elimination of algae. Ozone is mostly used as a primary disinfectant followed by chlorine as a final disinfectant in a water treatment process train (Rajagopaul *et al*., 2008).

The components of a water treatment ozone system include feed-gas preparation or supply, ozone generation, ozone contacting and ozone destruction as illustrated by Figure 2.4 (USEPA, 1999).

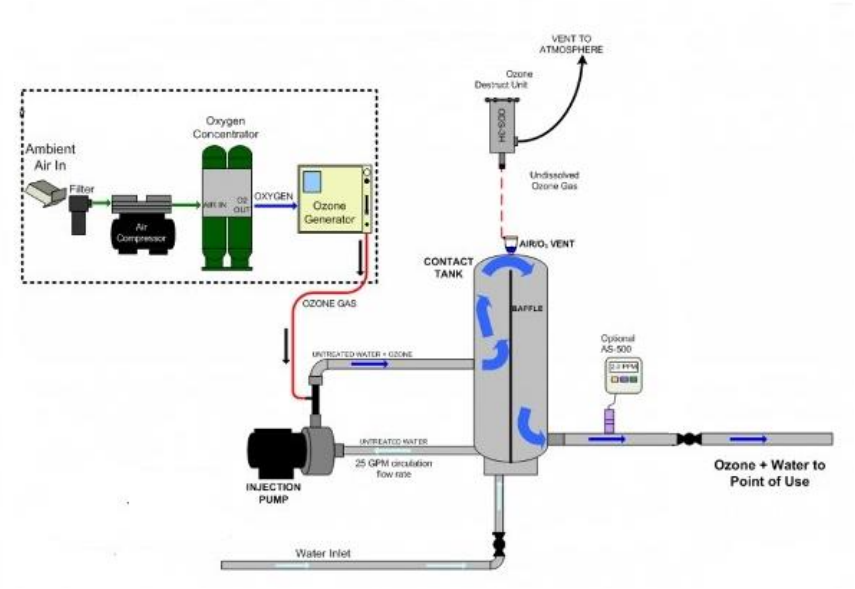


Figure 2.4: Components of a typical ozone system (adapted from Ozone Solutions, 2013).

Ozone should be generated at the point of application due to the instability of the molecule. Ozone is formed through the combination of an oxygen molecule with an oxygen atom (Figure 2.5) and this endothermic reaction requires considerable energy (Van der Walt *et al*., 2009).

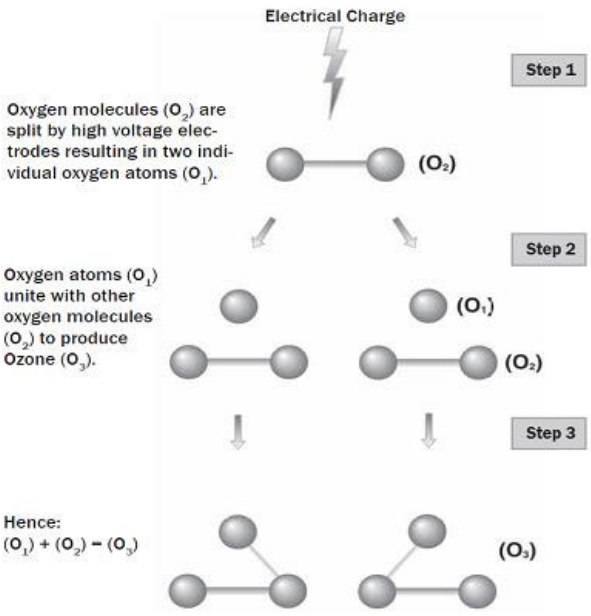


Figure 2.5: The formation of ozone (O_3) (The Pool Shoppe, 2013).

Ozone generators in general use the corona discharge process that entails the circulation of air or oxygen past an electrode charged with a high voltage. The high voltage that is discharged between the electrode nodes generates ozone. The corona discharge process is the preferred method for the treatment of water. An important factor to consider in on-site ozone generation is the oxygen source. The cost of the oxygen source as well as the cost of the energy to produce ozone constitute the primary expenditure in ozone generation (Van der Walt *et al*., 2009). The choice of oxygen source, an air-fed supply or oxygen fed supply, is determined by several factors which include ozone requirements, aspects of the application, operational and maintenance capabilities, financial constraints and logistics. Generally, oxygen fed systems have lower capital and operating costs (Rajagopaul *et al*., 2008).

The function of an ozone contactor system is the transfer of ozone into untreated water. Four types of ozone contactor systems are in use namely side stream venturi injection systems, bubble diffuser systems, deep U-tube systems and turbine mixer systems. The bubble diffuser contactor system that consists of ceramic stones, stainless steel holders and sealable gaskets, is the most extensively used contactor system (Van der Walt *et al*., 2009). The off-gas from the contact system passes through an ozone destructor where ozone is converted to oxygen before it is released to the atmosphere. Off-gas contains ozone concentrations that exceed the allowable maximum concentration. Thermal and catalytic destruction can destroy ozone (Rajagopaul *et al*., 2008).

Mechanisms of disinfection using ozone include: direct oxidation of the cell wall with resulting leaking of cell contents; reactions with radicals formed during ozone breakdown; and damage to the nucleic acids. The free radicals, hydrogen peroxy (HO_2) and hydroxyl are formed when ozone decomposes in water. These radicals play an important part in the disinfection process due to their oxidising capacity that causes protoplasmic oxidation that destroy bacteria. The efficacy of ozone disinfection is dependent upon the concentration and contact time of the ozone in addition to the susceptibility of the microorganisms (USEPA, 1999a). Refer to Table 2.2 for a list of advantages associated with ozone.

2.5. An overview of the use of GAC adsorption, UV light disinfection and ozone in South Africa

The prevalence of the use of advanced water treatment options in water treatment facilities has increased in South Africa in recent years due to the deteriorating water quality of its natural water resources. A number of water treatment plants have introduced the multi-barrier concept where the treatment train consists of more than one advanced treatment process. A brief overview is given in this section of some of the water treatment plants in South Africa that use GAC adsorption, UV light disinfection and ozone.

The Roodeplaat Water Treatment Works (WTW) plant is located in the Pienaars River catchment. Source water is abstracted from Roodeplaat Dam and is highly eutrophic with occasional high levels of taste and odour, iron, manganese and algae concentrations. The first phase of the Roodeplaat WTW was completed in 2005 and included UV light disinfection as one of the treatment processes before chlorination (Van der Walt *et al*., 2009). The second phase was completed in 2012 with the addition of ozonation after rapid sand filtration followed by GAC as treatment processes in anticipation of expected further deterioration of water quality. The ozone treatment equipment comprises of liquid oxygen that is stored onsite and three ozone generators. The GAC treatment equipment comprises of 20 GAC filter bays. Roodeplaat WTW is regarded as one the most advanced WTW plants in South Africa (Mattheus, 2013). Since the implementation of GAC and ozone at the Roodeplaat WTW, a visible improvement in the colour of the water has been observed and full compliance with the targeted water quality has been achieved.

The Vaalkop WTW consists of three water treatment plants located in the Hex- and Eland River catchment. The source water abstracted from Vaalkop Dam is highly eutrophic with taste and odour as a result of high geosmin and 2-methylisoborneol (MIB) concentrations. *Giardia*, *Cryptosporidium* and faecal coliform bacteria are also problematic. Pre-ozonation as well as intermediate ozonation and GAC adsorption treatment processes were introduced with the upgrade of Plant 1. Intermediate ozonation takes place after the Dissolved Air Flotation (DAF) treatment step (Van der Walt *et al*., 2009). Ozone is introduced via an inline static mixer into the pipeline during the pre-oxidation step and via a venturi into two contact basins during the intermediate ozonation step. Ozonation is followed by GAC adsorption (SA Water, 2013). Civil Engineering (2007) reported that the implementation of ozone disinfection and GAC adsorption resulted in the production of an improved quality of potable water by Plant 1 of the Vaalkop WTW, not only from an aesthetic perspective but also from a health perspective due to ozone's capabilities of destroying harmful organisms.

Rietvlei WTW abstracts water from Rietvlei Dam. Problems with increased concentrations of cyanobacteria and the associated taste and odour problems as well as contaminants concerns resulted in GAC adsorption being implemented as a treatment process. (Van der Walt *et al*., 2009). The GAC performance at Rietvlei WTW was closely monitored by De Kloe (quoted by Clements, 2002) for a year after implementation and it was verified that the GAC treatment process yielded potable water of a high quality. Rising concerns with regard to pathogens led to the implementation of ozonation during 2008 to supplement GAC adsorption. Ozone is used prior to the GAC adsorption process as oxidised organic compounds are more readily adsorbed. Pilot plant studies before the implementation of ozonation confirmed the successful removal of pathogens from the source water by ozone (CSV Water, 2011).

The Roodefontein Dam and Keurbooms River supply water to the Plettenberg Bay Central WTW. The conventional sedimentation process proved to be unsuccessful in treating source water quality challenges such as high levels of colour, iron, manganese, taste and odour. Ozonation was implemented in 2005 to address these problems and has been effective in eliminating taste and odour problems in addition to reducing high iron and manganese levels and high degrees of colour (McPherson & Lombard, 2006).

MWC uses ozonation and RWB uses GAC adsorption and UV light disinfection as treatment processes. These two sampling sites were chosen for this study and details of the water treatment process trains in use by these two sites respectively are provided in Chapter 4.

Sigudu (2010) tested a generic monitoring protocol for the management of the protozoan parasites, *Giardia* and *Cryptosporidium*, in drinking water at the RWB treatment plant. The results of the study confirmed that the multi-barrier concept of GAC adsorption and UV light disinfection in use by RWB is effective in removing *Giardia* and *Cryptosporidium* cysts. The study furthermore recommended that water treatment utilities can reduce the risk of *Giardia* and *Cryptosporidium* contamination by introducing advanced treatment options such as ozone or UV light disinfection.

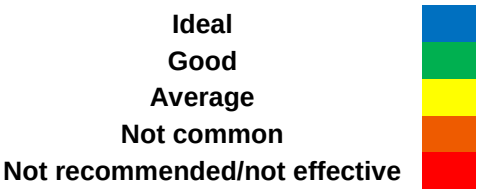
A study conducted by Morrison (2009) at MWC indicated that intermediate ozonation (refer to Figure 4.5) either reduced or had a fluctuating influence on the composition of algal species as well as on the physical and chemical characteristics of the source water and proved to be a beneficial treatment step in the purification process.

2.6. The selection of an appropriate treatment process

The characteristics of the source water and by implication the associated problems determine the selection of an appropriate treatment process. Based on the characterisation of the water source, appropriate treatment objectives can be developed and the treatment processes required to meet these objectives can be identified (Van der Walt *et al*., 2009).

Table 2.1 provides a list of typically encountered water treatment challenges and the respective appropriate treatment. For the purposes of this study only three advanced treatment technologies, namely GAC adsorption, UV light disinfection and ozone, are included.

Table 2.1: Selection of an appropriate treatment technology (adapted from Van der Walt *et al*., 2009).



	High colour	High taste and odour	High turbidity	High chlorophyll-a	High algae	High cyanobacterial toxins	High bacteria and virus	High *Cryptosporidium*	High *Giardia*	High DOC	High Manganese	High iron
GAC adsorption												
UV light disinfection												
Ozone												

The advantages and disadvantages of each treatment process (Table 2.2) have to be taken into account as these, in addition to space and budgetary constraints, can play a major role in the decision-making process.

Table 2.2: Advantages and disadvantages of GAC adsorption, UV light disinfection and ozonation.

TREATMENT	ADVANTAGES	DISADVANTAGES
GAC adsorption	• Proven reliable for the removal of dissolved solids; • Equipment doesn't utilise much space; • Technology that can be incorporated into an existing treatment plant without difficulty (USEPA, 2000).	• Wet GAC can be highly corrosive and abrasive; • Fluctuations in pH, temperature and flow rate can affect the efficacy of adsorption; • Bacterial growth in granular carbon beds can occur resulting in hydrogen sulphide generation (USEPA, 2000).

TREATMENT	ADVANTAGES	DISADVANTAGES
UV light disinfection	• Effective in the deactivation of most viruses, spores, cysts; • Physical process that eliminates the need to generate, handle, store and transport potentially toxic chemicals; • No residual effect; • Operator-friendly; • Short contact time; • Equipment doesn't utilise much space (USEPA, 1999**a**).	• Some viruses, spores and cysts may not be inactivated by low dosages; • The destructive effects of UV can occasionally be repaired by microorganisms (USEPA, 1999**a**).
Ozone	• Ozone is more effective than chlorine in destroying viruses and bacteria. Trihalomethanes (THMs) are formed as a disinfection by-product when chlorine is used as a disinfectant. The concentration of THMs with its associated health risks can be reduced with the use of ozone; • Short contact time; • Ozone decomposes rapidly which eliminates the need to remove harmful residues; • Regrowth of microorganisms doesn't occur after ozonation; • Fewer handling and transport safety issues as ozone is generated onsite; • The dissolved oxygen concentration of the effluent is raised by ozone which can eliminate the reaeration process (USEPA, 2000).	• Some viruses, spores and cysts may not be inactivated by low dosages; • Requires complex equipment and efficient contacting systems; • Equipment must be corrosion-resistant as ozone is very reactive; • Off-gases must be destroyed due to potential toxicity; • Costs associated with ozone treatment can be relatively high (USEPA, 2000).

Any institution responsible for the provision of potable water that is considering to either build a new water treatment facility or upgrade an existing one to introduce alternative water treatment options, not only needs to determine the quality of the source water but also needs to ascertain whether future anthropogenic influences can have an impact on the water source.

CHAPTER 3: MATERIALS AND METHODS

3.1. Introduction

The Accreditation for Conformity Assessment, Calibration and Good Laboratory Practice Act (Act No. 19 of 2006) provides for an internationally recognised national accreditation and monitoring system for South Africa. The South African National Accreditation System (SANAS) is recognised as the only national accreditation authority that can provide conformity assessment, calibration and good laboratory practise accreditations to calibration, testing and verification laboratories (SA, 2006).

SANAS accredits testing and calibration laboratories according to internationally agreed standards that are developed and published by the International Organisation for Standardisation (ISO). Methods used by Rand Water (RW) Analytical Services in Vereeniging and Midvaal Water Company (MWC) Analytical Services in Stilfontein are accredited in accordance with the recognised International Standard ISO/IEC 17025:2005 (SANAS, 2013).

This chapter serves to describe the sampling regime followed for the collection of water samples from both sampling sites, namely Rand Water Barrage (RWB) and MWC, as well as the materials, methods and statistical procedures used for the analysis of the collected water samples.

3.2. Sampling regime

3.2.1. Site 1: Rand Water Barrage (RWB)

RWB water samples were collected bimonthly according to RW working instructions for the period January 2009 to December 2010 at the following sampling points (refer to numbers in Figure 4.3):

- source water (1);
- secondary water after coagulation-flocculation, sedimentation and sand filtration (2);
- after granular activated carbon (GAC) adsorption (3);
- after ultraviolet (UV) light disinfection (4);
- potable water after chlorination disinfection (5).

The following water quality variables were measured:

(i) pH (in pH units);

(ii) Conductivity (in mS/m);

(iii) Turbidity (in NTU);

(iv) Dissolved organic carbon (DOC) (in mg/L);

(v) Total organic carbon (TOC) (in mg/L);

(vi) Total photosynthetic pigments (TPP) (in µg/L);

(vii) Microcystin (in µg/L);

(viii) Geosmin (in ng/L);

(ix) Algal composition (in cells/ml) by means of identification and enumeration;

(x) Invertebrate composition (in org/m^3) by means of identification and enumeration.

The invertebrate composition of the source water was not determined due to high turbidity. The invertebrate sampling during the period June 2009 to November 2009 at the other sampling points was not conducted due to cost restrictions. All water samples were analysed at RW Analytical Services according to RW accredited methods.

3.2.2. Site 2: Midvaal Water Company (MWC)

The following sampling points (refer to numbers in Figure 4.5) at MWC were utilised for the purposes of this study:

- source water (1);
- after pre-ozonation (2);
- after chemical dosing (coagulation and flocculation) (3);
- after Dissolved Air Flotation (DAF) (4);
- after intermediate ozonation (5);
- after settling (sedimentation) (6);
- after filtration (7);
- potable water after chlorination disinfection (8).

MWC water samples were collected according to MWC working instructions for the period January 2010 to December 2011 for the measurement of the following water quality variables:

(i) pH (in pH units);

(ii) Conductivity (in mS/m);

(iii) Turbidity (in NTU);

(iv) DOC (in mg/L);

(v) TOC (in mg/L);

(vi) TPP (in µg/L);

(vii) Microcystin (in µg/L);

(viii) Geosmin (in ng/L);

(ix) Algal composition (in cells/ml) by means of identification and enumeration;

(x) Invertebrate composition (in org/m^3) by means of identification and enumeration.

Water samples for the measurement of pH, conductivity, turbidity, TPP, DOC and TOC were collected weekly at all sampling points and analysed at MWC Analytical Services according to MWC accredited methods.

Geosmin, microcystin, algal and the invertebrate samples were collected every three weeks according to RW working instructions. Sampling for geosmin and microcystin was done at all sampling points. The invertebrate sampling was done at the source water, after pre-ozonation and potable water sampling points. Geosmin and microcystin samples were analysed and invertebrate identification and enumeration were conducted at RW Analytical Services according to RW accredited methods. The algal sampling was done at the source water, after pre-ozonation and after intermediate ozonation sampling points. The candidate conducted algal identification and enumeration for MWC at North-West University according to RW accredited methods.

3.3. Materials and methods

(i) pH

pH (in pH units) of 120 ml of the sample was measured with an autotitrator instrument.

(ii) Conductivity

Conductivity (in mS/m) of 120 ml of the sample was measured with a Conductometer.

(iii) Turbidity

Turbidity (in NTU) was measured with an HACH-2100AN Turbidimeter.

(iv) DOC

DOC (in mg/L): An adequate volume of sample was filtered through GF/C filter paper. The filtrate was transferred into a small glass bottle and the opening covered with aluminium foil. An adequate volume of deionised water was also filtered to be used as the blank. A plastic tube protruding from the instrument was inserted into the sample to begin the analysis. The following calculation was used to determine DOC results:

DOC (mg/L) = Sample value – Blank value (The instrument was calibrated weekly.)

(v) TOC

TOC (in mg/L): The sample was transferred into a small glass bottle and the opening covered with aluminium foil. A plastic tube protruding from the instrument was inserted into the sample to begin the analysis and the total organic carbon concentration was measured. (The instrument was calibrated weekly.)

(vi) TPP

Total chlorophyll (in µg/L) was measured with the Chlorophyll-665 method as described in Swanepoel *et al*. (2008a). A measured volume of sample was filtered, with the aid of gentle suction, thereby concentrating the phytoplankton onto a filter paper. Chlorophyll-655 (total pigment) was extracted from the concentrated phytoplankton in a known volume of methanol. After 1 hour extraction in a water bath, the extract was clarified by centrifugation. The absorbance of the extract at 665 nm (corrected for “background” interferences using the absorbance at 750 nm) was undertaken by using a spectrophotometer. The concentration of the total chlorophyll (µg/L) was then calculated using the formula below derived from Sartory (1982) and Steynberg (1986).

$E = 10^6 \times A(A_{665}-A_{750}) \times Ve \div Vm \times L$

Where

E = Chlorophyll (phaeophytin)

A = Absorption coefficient of 0.0133

A665 = Absorbance at 665 nm

A750 = Absorbance at 750 nm

Ve = Volume of solvent (mL)

Vm = Volume of sample (mL)

L = Path length of cu cuvette (cm)

x = Multiplication

(vii) Microcystin

Microcystin (in µg/L) analysis was done using the Enzyme-Linked Immuno Sorbent Assay (ELISA) technique where the microcystins in the sample and the enzyme compete for binding sites on the walls of the micro wells as described in Swanepoel *et al*. (2008a). The presence of total chlorine was determined and sodium thiosulphate added to the sample if total chlorine was present (>0.1 mg/L). The sample was shaken to ensure uniform distribution. Polypropylene tubes were prepared for every sample to be analysed. The sample was agitated to ensure homogeneity and the marked polypropylene tube destined for freeze thawing (approximately 1.5 mL) was filled with the sample. Algal cells were lysed to release the microcystin by freeze thawing the sample with liquid nitrogen. A minimum of 50 µL of sample was extracted and filtered. The microtiter plate reader was calibrated. 125 µL of microcystin assay, 20 µL of negative control, 20 µL of calibrator, 20 µL of sample, 100 µL of microcystin-enzyme conjugate, 100 µL of substrate and 100 µL of stop solution were added to a well after the required incubation period. The plate was read with the microplate reader within 30 minutes of the addition of stop solution. The method's limit of detection was 0.18 µg/L.

(viii) Geosmin

Geosmin (in ng/L) was analyzed by using the Purge and Trap method coupled to Gas Chromatography-Mass Spectrometry as described in Swanepoel *et al*. (2008a). An inert gas was bubbled through a 25 ml water sample contained in a purging chamber at 70˚C. The analytes were transferred from an aqueous phase to a vapor phase and swept through a sorbent trap where the analytes were trapped. The trap was heated and back-flushed with the inert gas to desorb the purgeables onto a gas chromatographic column. The gas chromatograph separated the analytes as it was temperature programmed and the analytes were detected with a mass spectrometer. Standard solutions were treated similarly and results were compared with results obtained from these standard solutions. The method's limit of detection was 5 ng/L.

(ix) Algal identification and enumeration

Algal identification and enumeration (in cells/ml) were done by means of the sedimentation technique using gravity as originally described by Utermöhl (1932). A 100 ml sample was preserved

with 2 ml formaldehyde. The sample was shaken and transferred to a stainless steel cylinder and closed with a rubber stopper. The cylinder was hit with a mechanical hammer to deflate the gas vacuoles of the cyanobacteria to ensure sedimentation. Up to 5 ml of the sample, depending on the turbidity, was pipetted into a Perspex sedimentation chamber. The remaining volume of the sedimentation chamber was filled with distilled water and covered with a glass cover slip. The sample was placed in a desiccator with water in the bottom for 48 hours to avoid evaporation of the sample.

The candidate identified and counted the algae to genus level using an inverted light microscope and a Whipple grid in the eyepiece (Figure 3.1). Identification was done using a guide for the identification of microscopic algae (Janse van Vuuren *et al*., 2006). A minimum of 200 cells were identified if the sample contained 200 or more cells by moving one grid at a time from left to right and rotating the chamber to a cross section and moving from right to left if the count was less than 200 at the end of the first lane. The volume of the sample, number of lanes, date, name and the number of cells counted for each genus were recorded. The same procedure was followed for all samples.

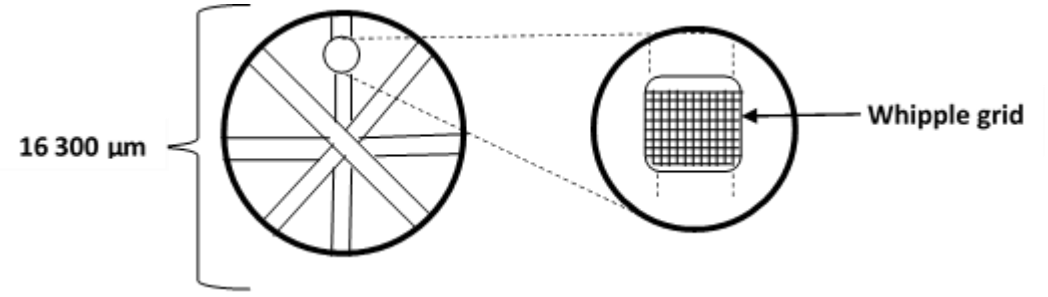


Figure 3.1: A line diagram showing the orientation of the lanes and the Whipple grid used for algal enumeration (Swanepoel et al., 2008a).

The following calculations as described by Lund *et al.* (1958) were used to calculate the number of algal cells per millilitre:

- The area of the sedimentation chamber floor was calculated:

Area = πr^2

- The area of the Whipple grid was calculated:

Area of a field = length x width

- The area of one rectangular lane was calculated:

Lane area = diameter of sedimentation chamber x width of Whipple grid

- The conversion factor was calculated:

Conversion factor = sedimentation chamber floor area ÷ total lane area

- The final conversion factor was calculated:

Final conversion factor = conversion factor ÷ volume of sample used

- The biomass was calculated:

Biomass (cells/ml) = count x final conversion factor

- The percentage composition of a taxon was calculated:

% composition = (biomass concentration of the taxon in cells/ml) x 100 ÷ total biomass concentration in cells/ml

The method's limit of detection for source water was 36 cells/ml and for potable water 1 cell/ml.

(x) Invertebrate identification and enumeration

Invertebrates were sampled using a simple sampling device which consisted of a pump or tap connection, an Anjet filter holder with a 50 µm mesh cylindrical filter or a 50 µm mesh dolphin bucket, a flow meter, and Gardenia fittings attached to the Anjet system and the flow meter, to facilitate the connection of a water hose (Ferreira & Du Preez, 2012). The start and end flow meter readings were recorded. In the laboratory the invertebrates were washed from the cylindrical filters and concentrated using a Millipore filter system fitted with a 50 µm circular mesh. The mesh was washed with 70% ethanol to dislodge and preserve the invertebrates. The invertebrates were stained with a Rose Bengal solution and left to stand for 12 to 24 hours. Invertebrates (in org/m^3) were identified and counted using a stereomicroscope and an electronic K microprocessor.

3.4. Statistical analysis

Statistica version 10 software (Statsoft Inc.) was used to determine the basic statistics namely the mean, maximum, minimum and standard deviation of the water quality variables. Differences between the water quality variables of the source water of the two sampling sites as well as differences between the sampling points of each site were determined through the use of Statistica. The Kolmogorov-Smirnov test for normality was used to determine if the variables were distributed parametrically. The data did not meet the assumptions of normality in the distribution of all

variables. The Kruskal-Wallis ANOVA for non-parametric data was used to compare multiple independent samples to determine differences between the variables in the source water of each site as well as differences between the variables from sampling point to sampling point. The Kolmogorov-Smirnov test for comparing two independent samples were used when appropriate. Spearman Rank correlation tests were conducted to determine correlations between the water quality variables of the source water. Box-and-whisker plots were created to illustrate the differences between the variables in the source water of each site as well as to illustrate the changes of the variables between sampling points.

Microsoft Excel 2013 was used to create pie charts of the algal composition and line charts to illustrate the differences between the sampling locations of the algal and invertebrate compositions.

CANOCO version 4.5 software was used to perform multivariate and ordination analyses (Ter Braak and Smilauer, 1998). Only the datasets that contained all the variables were used for multivariate analysis. Environmental variables were regarded as positively correlated with each other if their arrows subtended a small angle, uncorrelated if their arrows were at 90° and negatively correlated if their arrows were in opposite directions. The same was true for the species variables in correlation with the environmental variables. Environmental variables or sampling localities on the ordination diagrams that were closer together had more in common than variables or localities that were apart.

The source water quality as well as the efficacy of the advanced treatment steps in use at both sampling sites were determined by means of the sampling regime, materials and methods and the statistical procedures described in this chapter.

CHAPTER 4: SOURCE WATER QUALITY

4.1. Introduction

The first aim of this study was to determine the source water quality of both sampling sites. The assessment of the quality of the source water of both sampling sites served a dual purpose. Firstly, the source water was characterised and secondly, a dataset was produced that served as a baseline in order to determine the efficacy of the advanced treatment processes currently in use at Midvaal Water Company (MWC) and Rand Water Barrage (RWB).

The source water quality was determined by analysing the physical and chemical characteristics, the algal and invertebrate composition and by taking into account the integrated geography of the study area.

The following source water variables were measured at RWB:

(i) pH (in pH units);

(ii) Conductivity (in mS/m);

(iii) Turbidity (in NTU);

(iv) Dissolved organic carbon (DOC) (in mg/L);

(v) Total organic carbon (TOC) (in mg/L);

(vi) Total photosynthetic pigments (TPP) (in µg/L);

(vii) Microcystin (in µg/L);

(viii) Geosmin (in ng/L);

(ix) Algal composition (in cells/ml) by means of identification and enumeration.

The following source water variables were measured at MWC:

(i) pH (in pH units);

(ii) Conductivity (in mS/m);

(iii) Turbidity (in NTU);

(iv) DOC (in mg/L);

(v) TOC (in mg/L);

(vi) TPP (in µg/L);

(vii) Microcystin (in µg/L);

(viii) Geosmin (in ng/L);

(ix) Algal composition (in cells/ml) by means of identification and enumeration;

(x) Invertebrate composition (in org/m^3) by means of identification and enumeration.

Refer to Sections 3.2, 3.3 and 3.4 for an overview of the sampling regime that was followed and an explanation of the material and methods that were used to determine the source water quality.

4.2. An overview of the study area

The integrated water resources of the Vaal River system (VRS) sustain the water requirements of 20 million people in South Africa. The Vaal River not only serves as a conduit to transfer water among the three Vaal Water Management Areas (WMAs), the Upper, Middle and Lower Vaal, but also for transfers via the distribution system of Rand Water to the Crocodile West and Marico Rivers. Significant transfers occur from the Usutu, Thukela, Olifants and Orange Rivers into the VRS (Figure 4.1). An interdependency exists among the Upper, Middle and Lower Vaal WMAs due to their cascading orientation (DWAF, 2009b). Many of the water quality problems in the Middle Vaal originate from the Vaal Barrage that is located in the Upper Vaal WMA It is therefore important to consider inherited source water quality issues when investigating alternative treatment options for the Middle Vaal WMA.

The Upper Vaal WMA is the most important WMA in South Africa from a water resource management perspective due to large quantities of water being transferred into and out of this WMA and large quantities of water being released into the Middle and Lower Vaal WMAs.

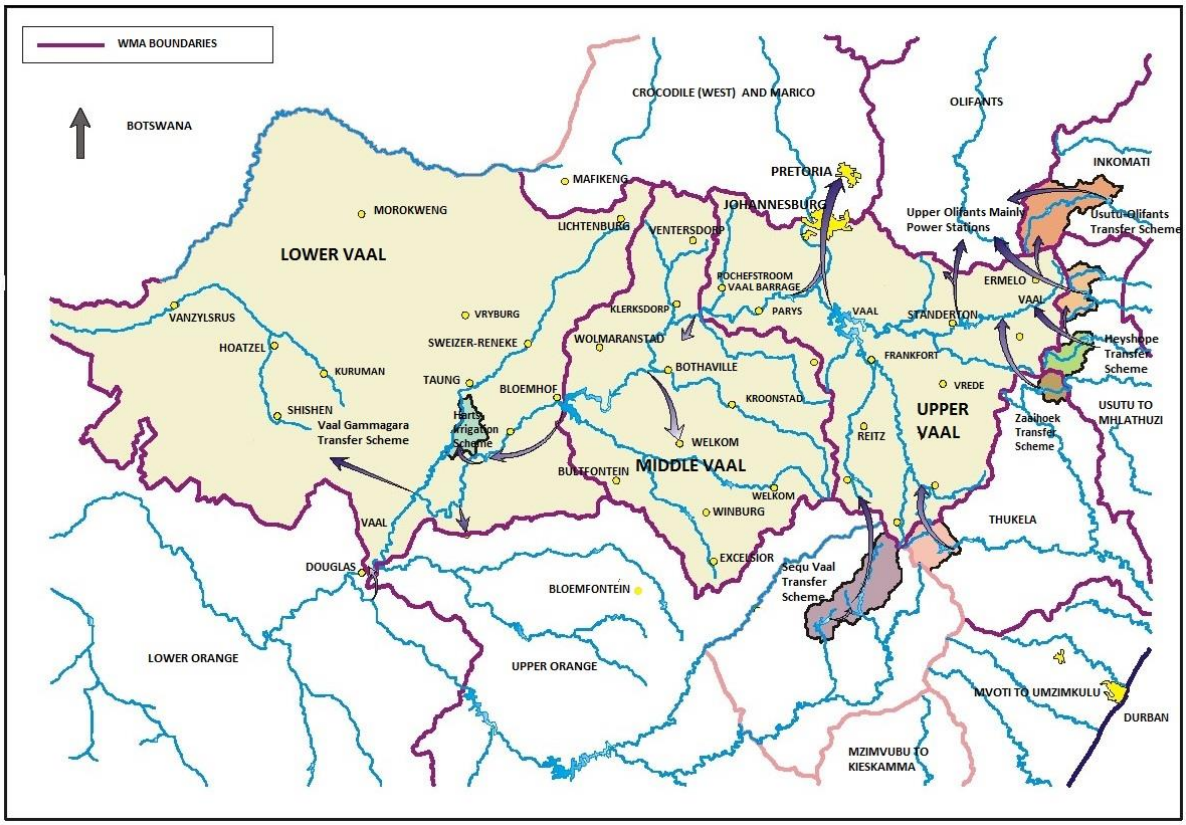


Figure 4.1: A map illustrating the water resource infrastructure of the Vaal River system and the location of the Upper, Middle and Lower Vaal Water Management Areas (adapted from DWAF, 2009b).

The study area, the Middle Vaal WMA, is located in the Free State and North West Provinces of South Africa between the Upper and Lower Vaal WMAs (Figure 4.1). The Vaal River flows in a westerly direction from the Upper Vaal WMA through the Middle Vaal WMA, joined by the Rhenoster, Vals, Skoonspruit and Vet Rivers as main tributaries, before flowing into the Lower Vaal WMA. The climate is semi-arid with an evaporation rate that exceeds the mean annual rainfall. Only three percent of the South African population reside in this WMA with the majority concentrated in the urban areas of Klerksdorp, Orkney, Stilfontein, Welkom, Virginia and Kroonstad. Agriculture with associated significant irrigation, mining and urban activities are the predominant land uses in the Middle Vaal WMA. Agricultural use comprises the majority of the total water requirements. (DWAF, 2003).

Naturally occurring surface water in the Middle Vaal WMA is fully utilised with no possibility for future development (DWAF, 2003). Furthermore, according to DWAF (2003), the majority of the surface water in this WMA is provided by inflows from the Upper Vaal WMA, most of which flows through the Middle Vaal WMA into the Lower Vaal and only a small portion of this yield is used locally. Water that enters the Middle Vaal WMA along the Vaal River contains a large proportion of

urban and industrial return flows from the Johannesburg area. As a result, salinity levels can be very high. The large domestic component of return flows into the surface water increases nutrient concentrations and promotes excessive algal growth.

Human activities affect surface water resources and local decisions in a WMA related to land use and development planning can contribute to the overall impact on the quality of a water resource. The analysis of water quality data provides more information on human actions in a WMA. This was one of the aims of the national Water Quality Planning Level review of South Africa's surface water resources conducted by DWA (DWA, 2011b). This review identified several water quality issues of concern in the Middle Vaal WMA (Table 4.1).

Table 4.1: Water quality issues identified in the Middle Vaal Water Management Area by the Department of Water Affairs (DWA, 2011b).

Water Quality Issue	Driver	Effect
Eutrophication	• Poor wastewater treatment works; • Urbanisation and informal settlements (un-serviced sewage); • Intensive use of fertilizers for agriculture.	• Increasing algal blooms; • Health risk associated with toxic cyanobacteria; • Taste and odour problems; • Aesthetically unpleasing; • Increased water treatment costs.
Microbial contamination	• Wastewater treatment works; • Dense informal settlements.	• Health risk to recreational users; • Health risks associated with drinking water, bathing and washing.
Salinisation	• Mining; • Wastewater treatment works; • Irrigation.	• Lower crop yield; • Irrigation system clogging; • Increased water treatment costs.
Altered flow regime	• Dams; • Weirs.	• Turbidity; • Seasonal flow changes; • Ecological water requirement changes; • Algal growth.
Radioactivity	• Discarded mine dumps.	• Bioaccumulation of pollutants; • Carcinogenic effects.

4.3. Sampling sites

4.3.1. Site 1: Rand Water Barrage (RWB)

Figure 4.2: A satellite image indicating the location of the Rand Water Barrage water treatment plant as well the source water abstraction point. Latitude: -26.759769; Longitude: 27.682328 (Google Earth, 2014).

The RWB water treatment plant is located at the Vaal Barrage weir (Figure 4.2). The plant currently only treats approximately 20 000 litres of water per day and supplies the small community situated within the vicinity of the plant with potable water. The Vaal River flows in a westerly direction from the RWB abstraction point for approximately a kilometre before it reaches the Middle Vaal WMA (Swanepoel, 2011). Due to the cascading nature of the three Vaal WMAs, the RWB source water is regarded as being abstracted from the Middle Vaal MWA for the purposes of this study as the quality of this water is representative of the quality of water in the upper reaches of the Middle Vaal WMA.

Conventional water treatment processes implemented at RWB water treatment plant include coagulation-flocculation using hydrated lime and activated sodium silicate as coagulants and ferric chloride to aid flocculation, sedimentation, pressurised sand filtration and chlorination. In addition, a multi-barrier approach has been adopted through the use of granular activated (GAC) adsorption followed by ultraviolet (UV) light disinfection as advanced treatment processes (Swanepoel, 2011). UV light intensity is maintained above 80%. Water is tested on a grab sample basis and any

deterioration in the quality of the UV output is rectified by the replacement of the UV light tubes (Sigudu, 2010). The sequence of the water treatment processes at RWB is illustrated by Figure 4.3.

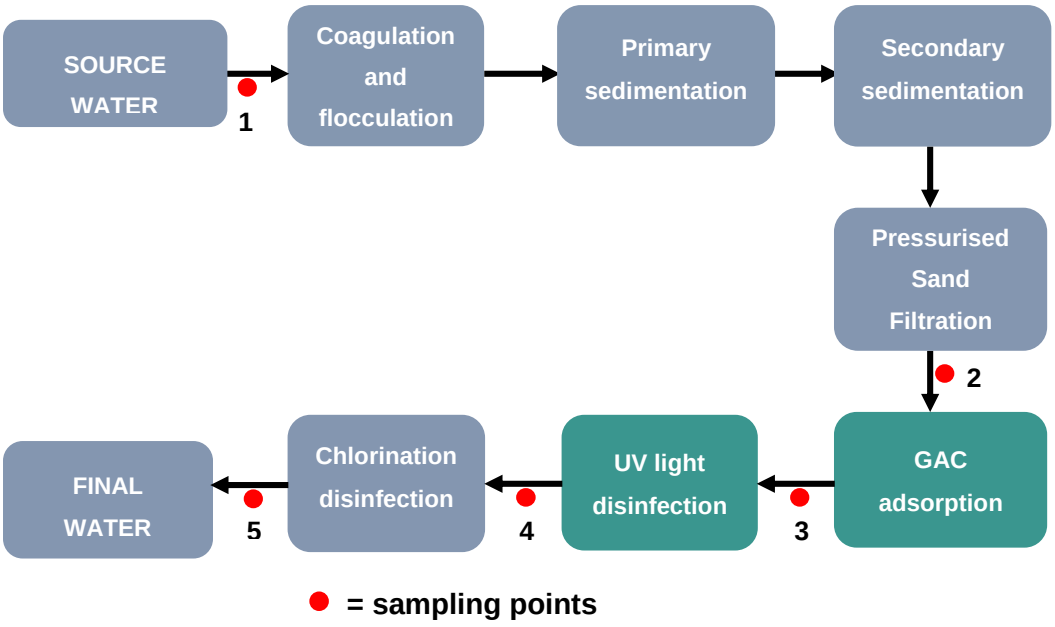


Figure 4.3: A flow diagram illustrating the sequence of the water treatment steps at the Rand Water Barrage treatment plant (adapted from Swanepoel, 2011).

4.3.2. Site 2: Midvaal Water Company (MWC)

Figure 4.4: A satellite image indicating the location of Midvaal Water Company as well the source water abstraction point. Latitude: -26.9310434205; Longitude: 26.7971001431 (Google Earth, 2014).

MWC is situated in the North West province of South Africa and is the bulk potable water supplier for the Orkney, Klerksdorp and Stilfontein area. MWC abstracts water from the Middle Vaal River, 150 kilometres downstream of the Vaal Barrage (Water & Sanitation Africa, 2006).

The sequence of the water treatment processes at MWC is illustrated in Figure 4.5. Large pumps situated in the intake tower (Figure 4.4), abstract source water from the Vaal River. Ferric chloride, aluminium sulphate and hydrated lime are used for the coagulation-flocculation process. During the Dissolved Air Flotation (DAF) process, flocs are removed through the formation of air bubbles that attach to the light-weighted flocs resulting in the flocs rising to the surface. The flocs not removed during the DAF process, settle from the water by gravitation during sedimentation. Circular clari-flocculators, a horizontal-flow dam and a pulsator are the three different types of sedimentation units that are used at MWC. Small, remaining flocs are removed through the rapid gravity type sand filters that use silica sand as a filtration medium. In the final treatment step, chlorine is added as a chemical disinfection agent. Potable water is distributed to approximately 500 000 consumers per day (MWC, 2007).

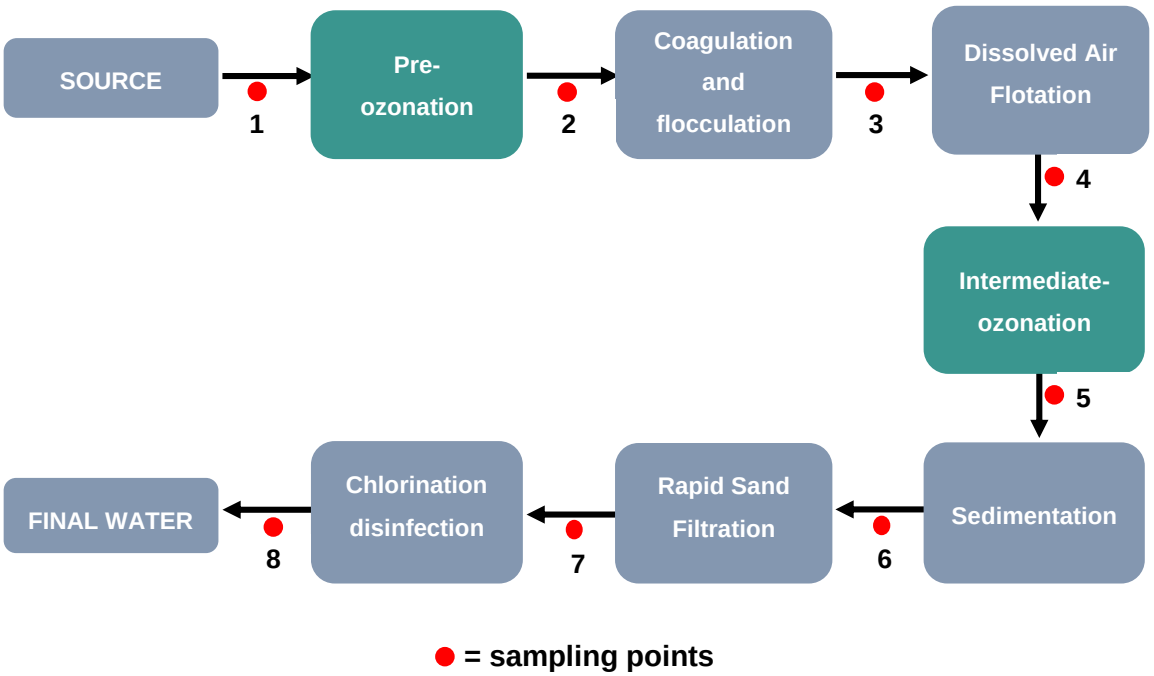


Figure 4.5: A flow diagram illustrating the sequence of the water treatment steps at Midvaal Water Company.

Ozonation is used as an advanced treatment process in two stages, pre-ozonation and intermediate ozonation. In the pre-ozonation process, ozone is added to the source water before

any treatment process and in the intermediate ozonation process, ozone is added after the DAF process. In 1985, two U-tube ozone reactors were constructed at MWC for the application of pre-ozone. The DAF plant was commissioned in 1985 to remove excessive organic material and the pre-ozonation process moved up in the treatment train to become the intermediate ozonation process. The MWC plant was subsequently upgraded in 2007 to include a pre-ozonation step. Ozonation equipment at MWC currently comprises of an ozone dose of 24 kilograms per hour through a pressure swing absorption (PSA) oxygen supply system, a radial flow type pre-ozone reactor and two intermediate U-tube reactors (Water & Sanitation Africa, 2006).

4.4. Results

4.4.1. Introduction

Water quality variables not only characterise a water source but also provide an indication of water quality-related problems that can result in health, aesthetic and economic impacts (DWAF, 1996). The most applicable water treatment option for a water source is ascertained by determining the source water quality as most water quality variables influence the water treatment processes.

The South African National Standard for Drinking Water (SANS 241) is the official guideline in South Africa for the assessment of the quality of domestic water. This standard specifies the quality of acceptable domestic water in terms of a water quality range for each water quality variable (SABS, 2006). SANS 241 describes two classes of domestic water with Class I as the recommended operational limit for lifetime consumption.

The South African Water Quality Guidelines for Domestic Use, a single set of water quality criteria for domestic water, specifies a No Effect Range, referred to as the Target Water Quality Range (TWQR), for each water quality variable (DWAF, 1996). This range indicates the levels of concentration at which a particular water quality variable will not have an adverse effect on the suitability of water for long-term domestic use. The goal of the TWQR is to specify ideal water quality and therefore water quality that falls outside of this range can still be deemed acceptable.

The source water quality of Sites 1 (RWB) and 2 (MWC) was determined by analysing:

- the physical and chemical characteristics of both sites and interrelationships;
- the algal compositions of both sites and interrelationships;
- the invertebrate composition of Site 2.

The physical and chemical source water quality variables of Sites 1 and 2 were compared to the recommended TWQR alternatively to SANS 241. The limit recommended by Rand Water (RW) was used if the TWQR or SANS 241 limit for a particular variable was unavailable.

Knowledge of the source water quality of Sites 1 and 2 provided a basis to determine the efficacy of the advanced water treatment processes in use at both sites.

4.4.2. Physical and chemical characteristics of the source water

Descriptive statistics namely the mean, minimum, maximum and standard deviation (SD) for each physical and chemical variable measured in the source water of Sites 1 and 2 are provided in Tables 4.2 and 4.3 respectively. TPP was regarded as one of these variables for the purposes of the study.

Table 4.2: Descriptive statistics for the physical and chemical water quality variables measured in the source water of Site 1 (Rand Water Barrage) for the sampling period January 2009 to December 2010. SD = Standard Deviation; (✓) = within SANS241/TWQR/RW guidelines; (x) = not within SANS241/TWQR/RW guidelines.

Variable	Unit	SANS/RW/ TWQR	Mean	Minimum	Maximum	SD
pH	pH units	✓	8.04	6.95	9	0.46
Conductivity	mS/m	x	61.533	23.5	86	15.172
Turbidity	NTU	x	23.505	4.21	129	22.959
DOC	mg/L	x	6.842	4.171	11.196	1.175
TOC	mg/L	x	7.892	0	31.549	3.957
TPP	µg/L	x	62.196	11.080	166.637	41.485
Microcystin	µg/L	x	0.523	< 0.18	2.339	0.626
Geosmin	ng/L	x	8.423	< 5	80.37	14.804

Table 4.3: Descriptive statistics for the physical and chemical water quality variables measured in the source water of Site 2 (Midvaal Water Company) for the sampling period January 2010 to December 2011. SD = Standard Deviation. (✓) = within SANS/TWQR/RW guidelines; (x) = not within SANS/TWQR/RW guidelines.

Variable	Unit	SANS/RW/ TWQR	Mean	Minimum	Maximum	SD
pH	pH units	✓	8.73	7.41	9.67	0.58
Conductivity	mS/m	x	55.7	18	134	18.548
Turbidity	NTU	x	37.731	10	142	34.833
DOC	mg/L	✓	6.331	3.6	10	1.09
TOC	mg/L	x	7.028	3.8	12	1.2
TPP	µg/L	x	78.531	2.7	221	55.23
Microcystin	µg/L	x	< 0.18	< 0.18	0.25	0.038
Geosmin	ng/L	x	< 5	< 5	12	2.347

The SANS 241 Class I operational limit for pH is between 5 and 9.7 pH units (SABS, 2011). The pH of Site 1 did not exceed this range during the sampling period although measurements remained in the upper range with a mean pH of 8.04 and a maximum value of 9 (Table 4.2). The mean pH of Site 2 was 8.73 during the sampling period (Table 4.3). The pH measurements of Site 2 did not exceed the SANS 241 limit. A maximum value of 9.67 was reached during June 2010. Seasonal fluctuations were observed at Site 2 with higher pH values during autumn and winter and lower pH values, although still in the upper range, during spring and summer. A comparison of the pH measurements of Site 1 and Site 2 indicated a significant difference (p = 0.000) between the pH values of the two sites with lower measurements recorded at Site 1.

The TWQR for conductivity is 0 to 70 mS/m. The upper limit of this range takes into account higher water consumption in warmer climates (DWAF, 1996). The conductivity for both sites was in the upper limit of the TWQR with means of 61.533 mS/m (Table 4.2) and 55.7 mS/m (Table 4.3) for Sites 1 and 2 respectively. The conductivity measurements of Site 1 exceeded the TWQR from June to November 2009 reaching a maximum of 86 mS/m during October 2009. Values higher than the recommended range were recorded during August 2010 and from October to December 2011 at Site 2 reaching a maximum of 134 mS/m during December 2011. Similar trends were observed at both sites with lower conductivity values from February to April and higher values from May to January during the respective sampling periods. Figure 4.6 provides a comparison of the conductivity values between Sites 1 and 2 and illustrates the lower values recorded at Site 2 (p = 0.0123).

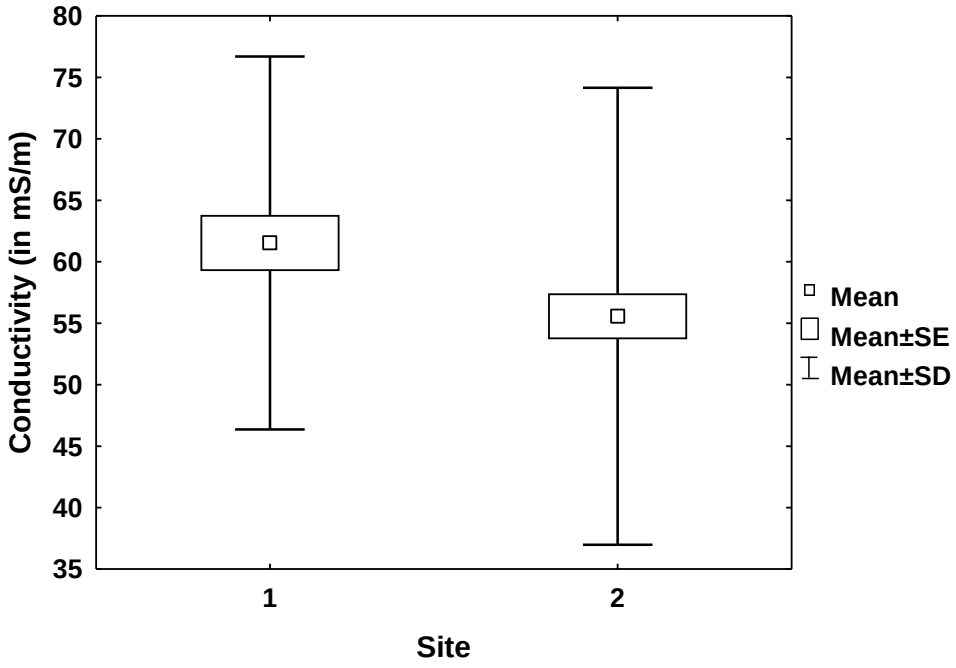


Figure 4.6: A box-and-whisker plot illustrating the difference in the mean conductivity values (in mS/m) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

The turbidity measurements for both Sites 1 and 2 were extremely high during the respective sampling periods and all the values recorded exceeded the SANS 241 Class I operational limit of ≤ 1 NTU (SABS, 2011). The mean turbidity for Site 1 was 23.505 NTU (Table 4.2) and 37.731 NTU (Table 4.3) for Site 2. Turbidity values increased from December to March for both sites during the respective sampling periods with a maximum of 129 NTU (Table 4.2) recorded for Site 1 during January 2010 and a maximum of 142 NTU (Table 4.3) recorded for Site 2 during February 2010. A comparison of the turbidity measurements of the two sites as illustrated by Figure 4.7, indicated significantly higher values for Site 2 (p = 0.0012).

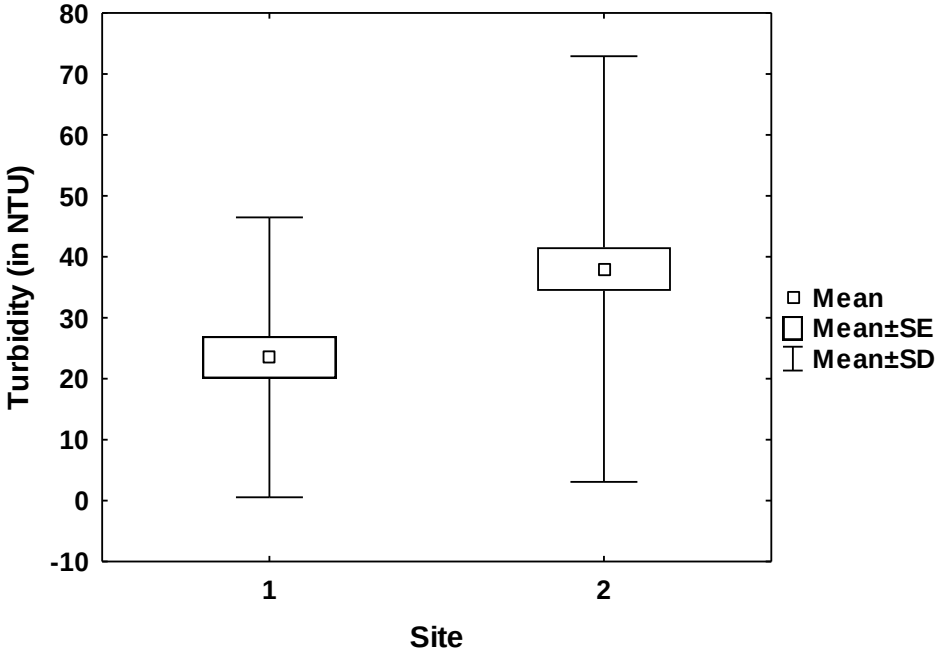


Figure 4.7: A box-and-whisker plot illustrating the difference in the mean turbidity values (in NTU) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

The SANS 241 (SABS, 2006) Class I operational limit for DOC is < 10 mg/L as is the limit for TOC (SABS, 2011). The DOC values of Site 2 did not exceed this limit during the sampling period with a mean of 6.331 mg/L (Table 4.3). The DOC measurements of Site 1 were higher than the DOC values of Site 2 with a mean of 6.842 mg/L although the operational limit was only exceeded once during the sampling period with a maximum of 11.196 mg/L (Table 4.2) measured during May 2010. A significant difference was observed between the DOC values of the two sites (p = 0.006) as illustrated by Figure 4.8. Similarly, Figure 4.9 shows that a significant difference was observed

between the TOC measurements of Site 1 and Site 2 (p = 0.004). A mean of 7.892 mg/L (Table 4.2) was recorded for Site 1 and a mean of 7.028 mg/L (Table 4.3) for Site 2. The TOC limit was exceeded by the maximum measurements of each site namely 31.549 mg/L (Table 4.2) for Site 1 and 12 mg/L (Table 4.3) for Site 2.

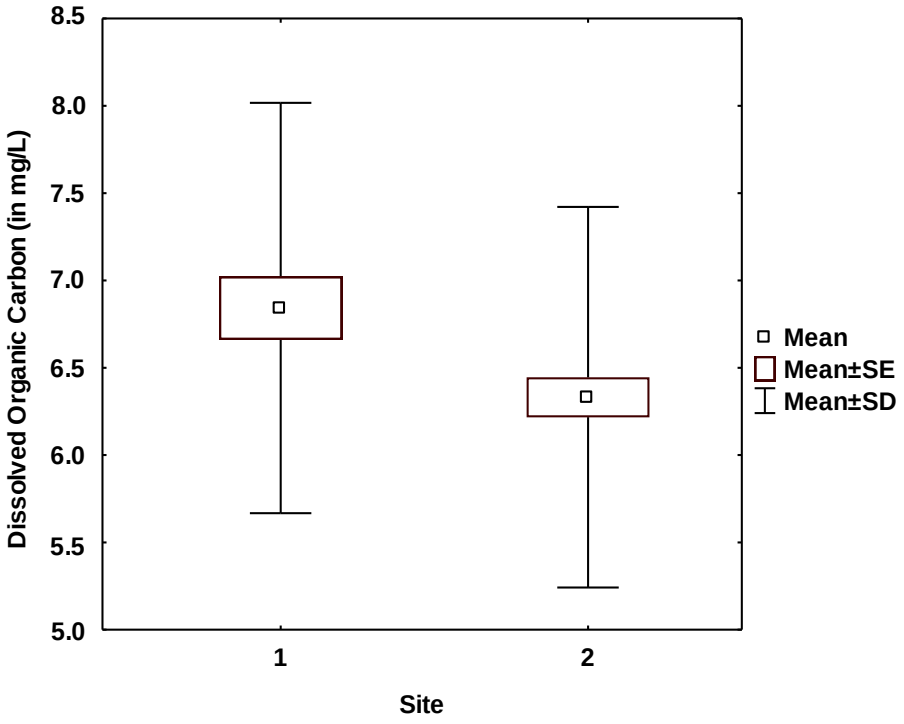


Figure 4.8: A box-and-whisker plot illustrating the difference in the mean DOC values (in mg/L) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

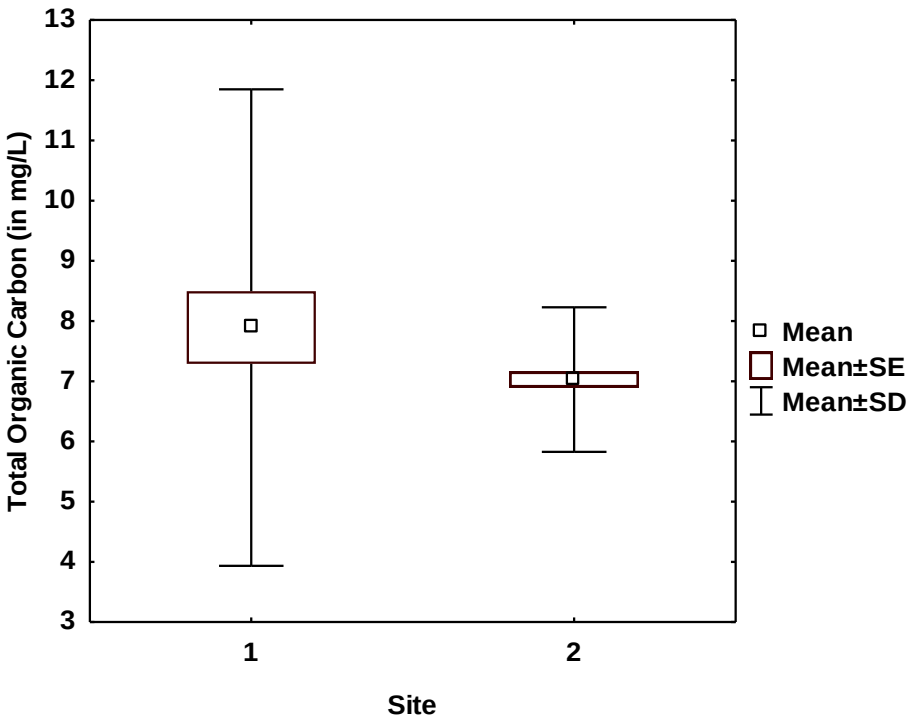


Figure 4.9: A box-and-whisker plot illustrating the difference in the mean TOC values (in mg/L) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

No provision is made for a TPP range by either DWAF or SANS 241.The recommended maximum limit for source water set by Rand Water is 15 µg/L (Swanepoel *et al*., 2008a). This limit was exceeded by all measurements except between January and March 2011 at Site 2 and between February and March 2009 and April 2010 at Site 1. The mean TPP value for Site 1 was 62.196 µg/L (Table 4.2) and for Site 2 was 78.531 µg/L (Table 4.3). A significant difference was observed between the TPP values of the two sites ($p < 0.01$) upon comparison (Figure 4.10) with higher values recorded for Site 2. As illustrated by Figure 4.11, similar seasonal fluctuations in TPP measurements were observed during the respective sampling periods of both sites with higher values recorded during winter and spring and lower values recorded during summer and autumn. The maximum TPP value for Site 1, 166.64 µg/L (Table 4.2), was recorded during July 2009 and the minimum TPP value, 11.08 µg/L (Table 4.2), was recorded during April 2010. The maximum TPP value for Site 2 of 221 µg/L (table 4.3) and the minimum value of 2.7 µg/L (Table 4.3) were recorded during late May 2010 and February 2011 respectively.

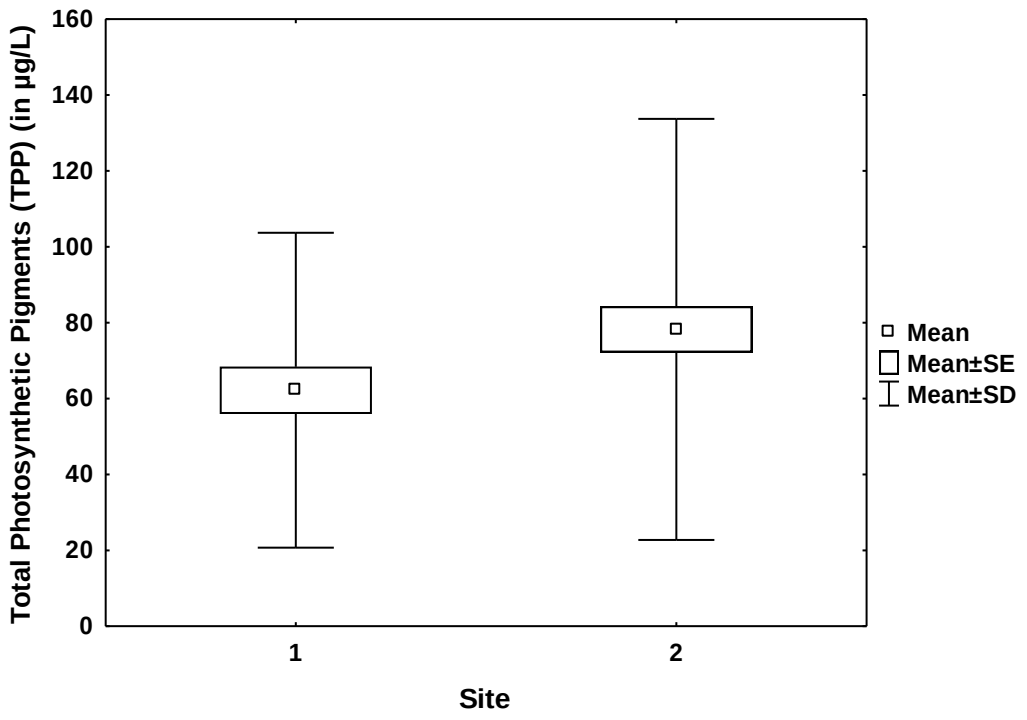


Figure 4.10: A box-and-whisker plot illustrating the difference in the mean TPP values (in µg/L) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

Figure 4.11: A line chart illustrating the seasonal fluctuations in the TPP values (in µg/L) of the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011).

The SANS 241 limit for microcystin for potable water is ≤ 1 µg/L (SABS, 2011). The ELISA method's (refer to Chapter 3.3) limit of detection is 0.18 µg/L (Swanepoel *et al*., 2008a). The microcystin values of Site 2 remained low during the sampling period with a mean of < 0.18 µg/L (Table 4.3) and with a maximum of 0.25 µg/L reached during December 2010. The microcystin values of Site 1 varied considerably from the mean of 0.523 µg/L (Table 4.2) as illustrated by Figure 4.12 with a maximum of 2.339 µg/L (Table 4.2) recorded during January 2009. Several measurements exceeded the limit for potable water. A significant difference was observed between the microcystin measurements of Sites 1 and 2 ($p < 0.025$) with higher microcystin values recorded at Site 1 (Figure 4.12).

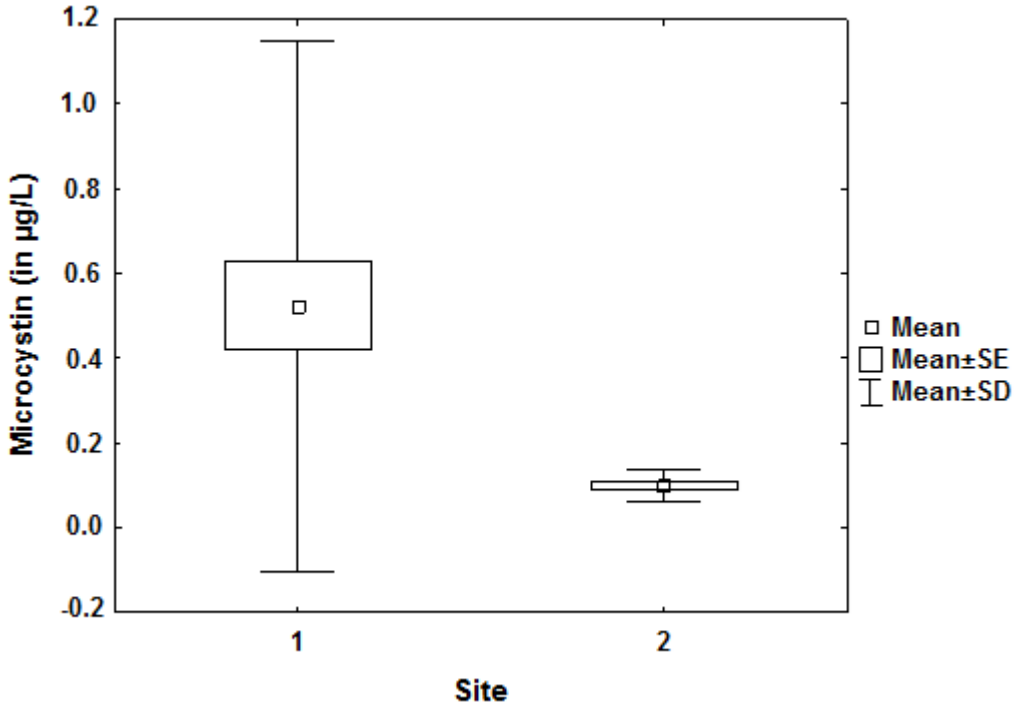


Figure 4.12: A box-and-whisker plot illustrating the difference in the mean microcystin values (in µg/L) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

Geosmin samples were analysed with a Rand Water accredited method using the Purge and Trap method coupled to Gas Chromatography-Mass Spectrometry (Refer to Chapter 3.3). The method's limit of detection is 5 ng/L. As neither DWAF nor SANS241 specify a limit for geosmin, Rand Water set a limit of 30 ng/L for geosmin in source water (Ewerts, 2015). The mean geosmin value of the source water of Site 1 was 8.423 ng/L (Table 4.2). As illustrated by Figure 4.13, considerable variation was observed in the spread of the data with a maximum of 80.37 ng/L reached during November 2010 and a minimum of < 5 ng/L (Table 4.2). A mean geosmin value of < 5 ng/L was recorded at Site 2 with a maximum of 12 ng/L reached during December 2010 (Table 4.3). The geosmin values of Site 1 exceeded the limit of 30 ng/L twice but the geosmin values of Site 2 remained well below the limit. A significant difference was noted between the geosmin values of the two sites ($p < 0.025$) with higher values recorded at Site 1.

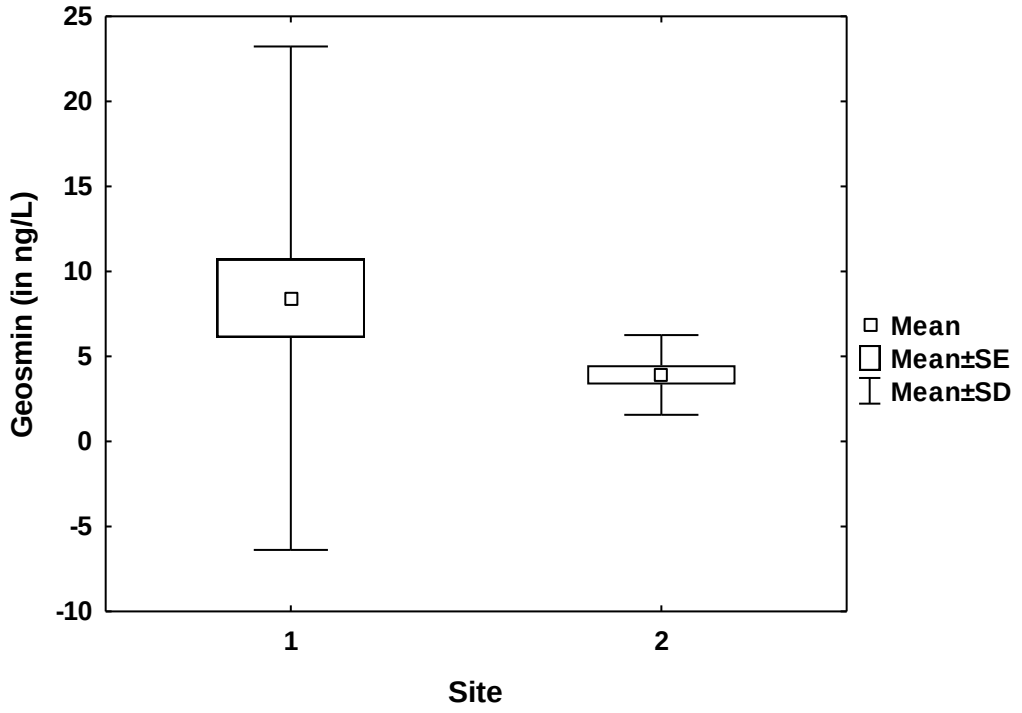


Figure 4.13: A box-and-whisker plot illustrating the difference in the mean geosmin values (in ng/L) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

4.4.3. Algal characteristics of the source water

An algal genus list was compiled for Site 1 and Site 2 for all species recorded in the source water to determine the species diversity and to illustrate possible differences in the species composition

of each site (Table 4.4). Seven algal classes were identified in the source water namely Bacillariophyceae (diatoms), Chlorophyceae (green algae), Cryptophyceae, Cyanophyceae (blue-green bacteria), Euglenophyceae, Dinophyceae (dinoflagellates) and Chrysophyceae.

Table 4.4: A complete list of algal genera identified in the source water of Site 1 and Site 2 (names in brackets pertain to ordination diagrams).

	SITE 1	SITE 2
BACILLARIOPHYCEAE (BACIL)		
Asterionella sp.	✓	x
Aulacoseira sp. (*Aulac*)	✓	✓
Cocconeis sp.	✓	✓
Cyclotella sp. (*Cyclo*)	✓	✓
Cymbella sp.	✓	✓
Unidentified centric diatoms	✓	✓
Fragilaria sp.	✓	✓
Gomphonema sp.	✓	✓
Melosira sp. (*Melos*)	✓	✓
Navicula sp.	✓	✓
Nitzschia sp. (*Nitzs*)	✓	✓
Unidentified pennate diatoms	✓	✓
Pinnularia sp.	x	✓
Surirella sp. (*Surir*)	✓	✓
Ulnaria sp.	x	✓
Total number of BACILLARIOPHYCEAE genera	**13**	**14**
CHLOROPHYCEAE (CHLORO)		
Actinastrum sp. (*Actin*)	✓	✓
Carteria sp. (*Carte*)	✓	✓
Chlamydomonas sp. (*Chlam*)	✓	✓
Chlorella sp.	✓	✓
Coccomonas sp. (*Coccom*)	✓	x
Coelastrum sp. (*Coela*)	✓	✓
Coenocystis sp.	x	✓
Closterium sp.	x	✓
Cosmarium sp. (*Cosma*)	x	✓
Crucigenia sp. (*Cruci*)	✓	✓
Eudorina sp.	✓	x
Gonatozygon sp.	x	✓
Gonium sp.	x	✓
Micractinium sp.	x	✓
Monoraphidium sp. (*Monap*)	✓	✓
Oocystis sp.	✓	✓
Pandorina sp. *(Pando)*	✓	✓
Pediastrum sp. (*Pedia*)	✓	✓
Pteromonas sp.	x	✓
Scenedesmus sp. (*Scene*)	✓	✓
Schroederia sp.	x	✓
Sphaerocystis sp.	x	✓
Staurastrum sp.	x	✓
Tetraedron sp.	✓	✓
Tetrastrum sp.	x	✓
Total number of CHLOROPHYCEAE genera	**14**	**23**

	SITE 1	SITE 2
CRYPTOPHYCEAE (CRYPTO)		
Cryptomonas sp. (*Crypt*)	✓	✓
Total number of CRYPTOPHYCEAE genera	**1**	**1**
CYANOPHYCEAE (CYANO)		
Anabaena sp. (*Anab*)	✓	✓
Microcystis sp. (*Micro*)	✓	✓
Oscillatoria sp. (*Oscil*)	✓	✓
Pseudanabaena sp. (*Pseud*)	✓	✓
Total number of CYANOPHYCEAE genera	**4**	**4**
EUGLENOPHYCEAE (EUGLENO)		
Euglena sp.	✓	✓
Phacus sp.	✓	✓
Strombomonas sp.	✓	✓
Trachelomonas sp. (*Trach*)	✓	✓
Total number of EUGLENOPHYCEAE genera	**4**	**4**
DINOPHYCEAE (DINO)		
Ceratium sp. (*Cerat*)	✓	✓
Peridium sp. (*Perid*)	✓	✓
Total number of DINOPHYCEAE genera	**2**	**2**
CHRYSOPHYCEAE		
Mallomonas sp.	✓	x
Total number of CHRYSOPHYCEAE genera	**1**	**0**
TOTAL NUMBER OF GENERA	**39**	**48**

The source water of Site 2 with a total number of 48 algal genera exhibited more diversity than the source water of Site 1 with a total number of 39 algal genera (Table 4.4). The major difference in phytoplankton diversity can be attributed to the number of Chlorophyceae in the source water of Site 2 namely 23 genera compared to 14 recorded in the source water of Site 2.

The statistical analysis of the algal composition of both sites excluded algae that occurred twice or less in the source water. The Chrysophyceae algal class was excluded from statistical analysis as only one genus was identified in the source water of Site 1 and enumerated on only one occasion.

The percentage compositions of the algal classes identified in the source water of Sites 1 and 2 are illustrated by Figures 4.14 and 4.15 respectively.

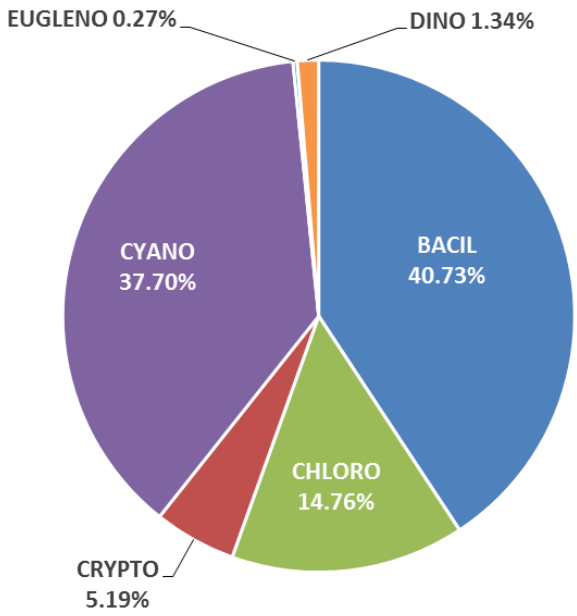


Figure 4.14: A pie chart illustrating the percentage composition of the algal classes identified in the source water of Site 1 for the sampling period January 2009 to December 2010.

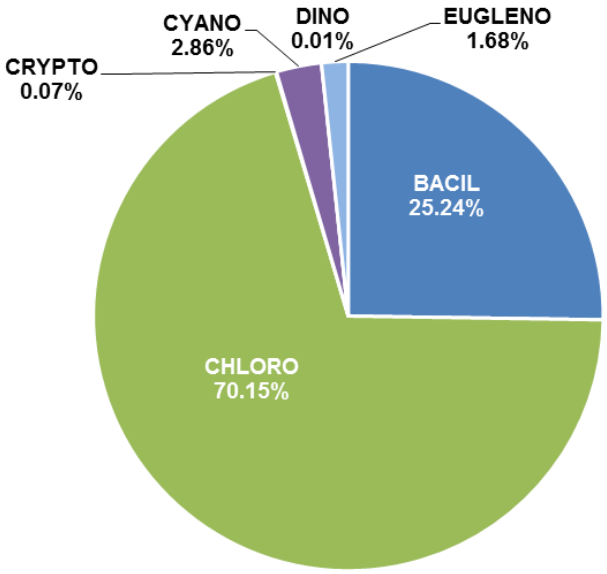


Figure 4.15: A pie chart illustrating the percentage composition of the algal classes identified in the source water of Site 2 for the sampling period January 2010 to December 2011.

As illustrated by Figure 4.14, Bacillariophyceae was the most abundant algal class of Site 1 (40.73%) followed closely by Cyanophyceae (37.70%). Chlorophyceae comprised only 14.76% and

Cryptophyceae 5.19% of the algal composition in the source water of Site 1. There was a low percentage, only 0.27%, of Euglenophyceae present in the source water of Site1.

It is clear from Figure 4.15 that Site 2 was dominated by Chlorophyceae (70.15%). Bacillariophyceae comprised a quarter, 25.24%, of the algal composition of Site 2. The Cyanophyceae and Euglenophyceae percentages of Site 2 were low, 2.86% and 1.68% respectively (Figure 4.15). Dinophyceae comprised a negligible percentage of the algal composition of Site 2.

Descriptive statistics namely the mean, minimum, maximum and standard deviation (SD) for the algal classes that were present in the source water of Sites 1 and 2 are provided in Tables 4.5 and 4.6 respectively.

Table 4.5: Descriptive statistics for the algal classes that were present in the source water of Site 1 for the sampling period January 2009 to December 2010. SD = Standard Deviation.

Algal class	Unit	Mean	Minimum	Maximum	SD
Bacillariophyceae	cells/ml	3285	0	16467	4293
Chlorophyceae	cells/ml	1192	0	9688	2103
Cryptophyceae	cells/ml	419	0	2727	632
Cyanophyceae	cells/ml	2949	0	43806	7860
Dinophyceae	cells/ml	108	0	775	189
Euglenophyceae	cells/ml	< 36	0	323	71
Total cells	cells/ml	7975	0	50802	9874

Table 4.6: Descriptive statistics for the algal classes that were present in the source water at Site 2 for the sampling period January 2010 to December 2011. SD = Standard Deviation.

Algal class	Unit	Mean	Minimum	Maximum	SD
Bacillariophyceae	cells/ml	5167	< 36	22380	5883
Chlorophyceae	cells/ml	14363	140	125769	31738
Cryptophyceae	cells/ml	< 36	0	86	28
Cyanophyceae	cells/ml	585	0	7150	1844
Dinophyceae	cells/ml	< 36	0	36	9
Euglenophyceae	cells/ml	344	0	4362	1116
Total cells	cells/ml	20475	229	140712	34346

No significant difference ($p > 0.05$) was observed between the Bacillariophyceae concentration of Sites 1 and 2 (Figure 4.16). The mean concentration for Site 1 was 3285 cells/ml (Table 4.5) and for Site 2 the mean was 5167 cells/ml (Table 4.6). An increase in the abundance of Bacillariophyceae species was observed in the source water of both sites during winter with Site 1 reaching a maximum concentration of 16467 cells/ml (Table 4.5) during August 2009 and Site 2 reaching a maximum concentration of 22380 cells/ml (Table 4.6) during July 2011. *Aulacoseira* was the most abundant Bacillariophyceae genus in the source water of Site 1 and *Cyclotella* and other unidentified centric diatoms were dominant in the source water of Site 2.

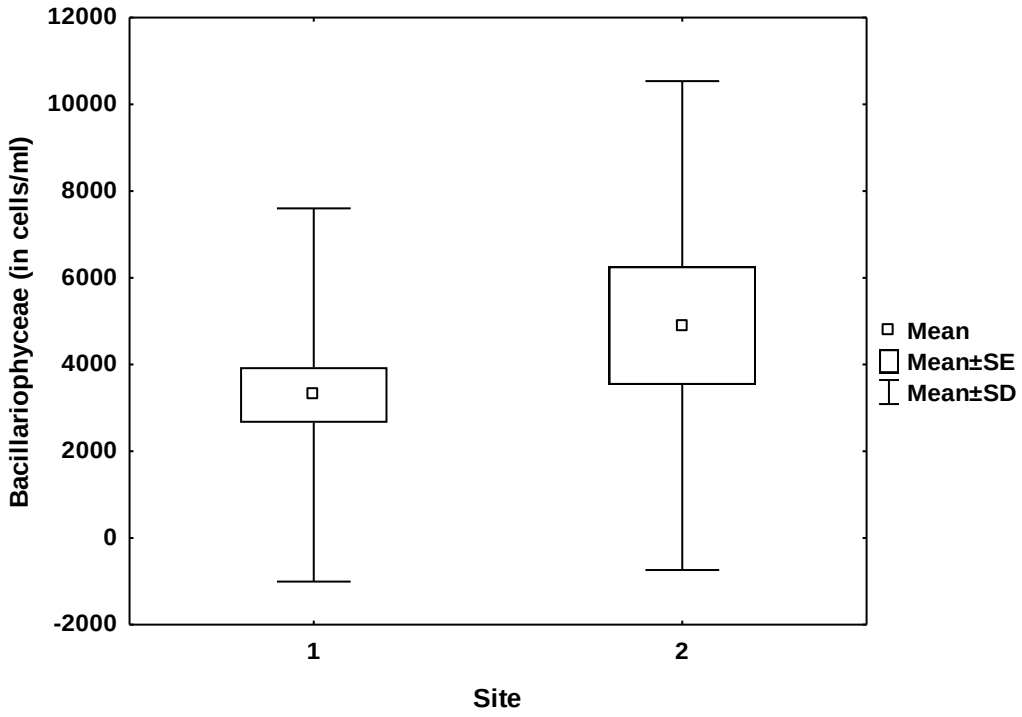


Figure 4.16: A box-and-whisker plot illustrating the difference in the mean Bacillariophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

An increase in the abundance of Chlorophyceae in the source water of both sites was observed from August to October. The maximum Chlorophyceae concentration for Site 1 was 9688 cells/ml (Table 4.5) and 125769 cells/ml (Table 4.6) for Site 2, both recorded during the month of October. The minimum Chlorophyceae concentration in the source water Site 2 was 126 cells/ml (Table 4.6) with a mean count of 14363 cells/ml (Table 4.6). The mean Chlorophyceae concentration for Site 1 was 1192 cells/ml (Table 4.5). Figure 4.17 illustrates the significant difference (p = 0.0001) between the Chlorophyceae concentrations of Sites 1 and 2 with a much lower concentration in the source water of Site 1. *Scenedesmus* sp. was abundant in the source water of Site 2 as well as *Coelastrum* sp. *Pandorina* sp. occurred frequently in the source water of both sites.

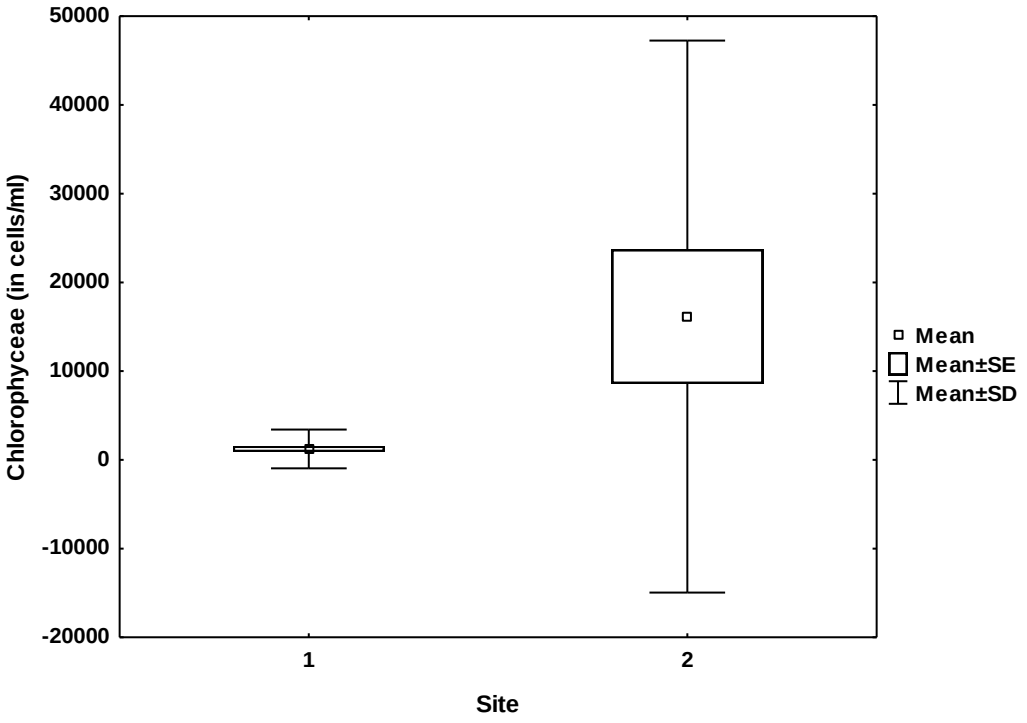


Figure 4.17: A box-and-whisker plot illustrating the difference in the mean Chlorophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

A significant difference (p = 0.0002) was observed between the Cryptophyceae concentration of Sites 1 and 2 (Figure 4.18). The mean concentration for Site 1 was 418 cells/ml (Table 4.5) whereas the mean concentration for Site 2 was < 36 cells/ml (Table 4.6). The maximum Cryptophyceae concentration for Site 2 was 86 cells/ml (Table 4.6) which was considerably lower than the maximum concentration of 2691 cells/ml (Table 4.5) for Site 1. It was observed that the maximum concentrations for both sites were reached during the winter months namely August and July for Sites 1 and 2 respectively. *Cryptomonas* sp. occurred frequently in the source water of Site 1 and to a lesser extent in the source water of Site 2.

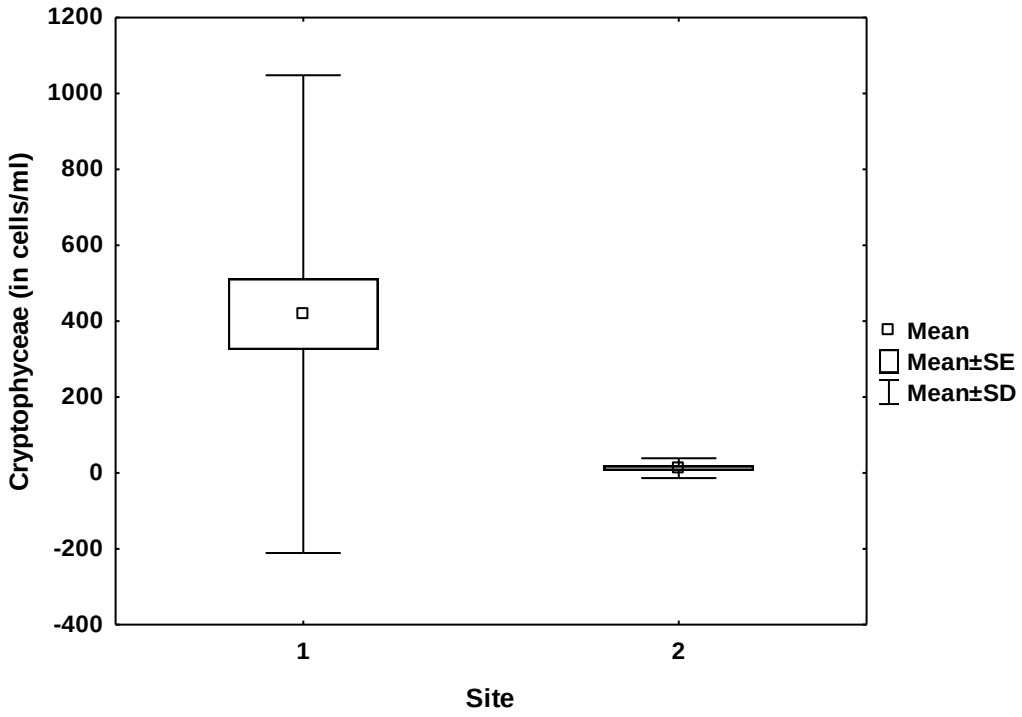


Figure 4.18: A box-and-whisker plot illustrating the difference in the mean Cryptophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

Cyanophyceae occurred far more frequently in the source water of Site 1 than in the source water of Site 2. The mean Cyanophyceae concentration for Site 1 was 2949 cells/ml (Table 4.5) with a maximum concentration of 43806 cells/ml (Table 4.5) during November 2010 and a minimum of 0 cells/ml. This high standard deviation of 7841 cells/ml (Table 4.5) is clearly illustrated by Figure 4.19 as is the difference ($p < 0.05$) between the Cyanophyceae concentration of Sites 1 and 2. *Microcystis* sp. featured abundantly in the source water of Site 1. The mean concentration for Site 2 was 585 cells/ml (Table 4.6) with a maximum of 7150 cells/ml (Table 4.6) during June 2010.

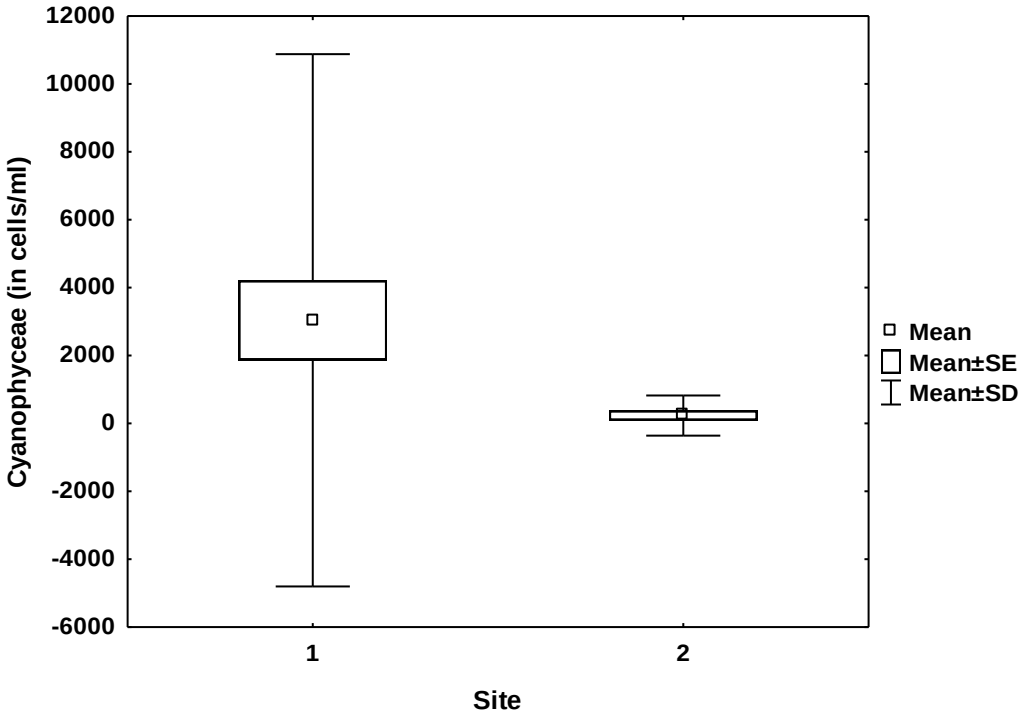


Figure 4.19: A box-and-whisker plot illustrating the difference in the mean Cyanophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

No Dinophyceae species were recorded in the source water of Site 1 during the second half of the respective sampling period namely March to December 2010. Similarly, no Dinophyceae species were recorded in the source water of Site 2 during the second half of the respective sampling period namely January to November 2011. Dinophyceae species were more abundant in the source water of Site 1 with a mean concentration of 108 cells/ml (Table 4.5). The problematic species, *Ceratium* sp., occurred frequently in the source water of Site 1 but not in large numbers. Dinophyceae were encountered in only three of the samples of Site 2 with a mean concentration of < 36 cells/ml (Table 4.6). A maximum count of 775 cells/ml (Table 4.5) was recorded for Site 1 and a maximum count of 36 cells/ml (Table 4.6) for Site 2. A significant difference was observed between the Dinophyceae concentration of Sites 1 and 2 (p = 0.016) as illustrated by Figure 4.20.

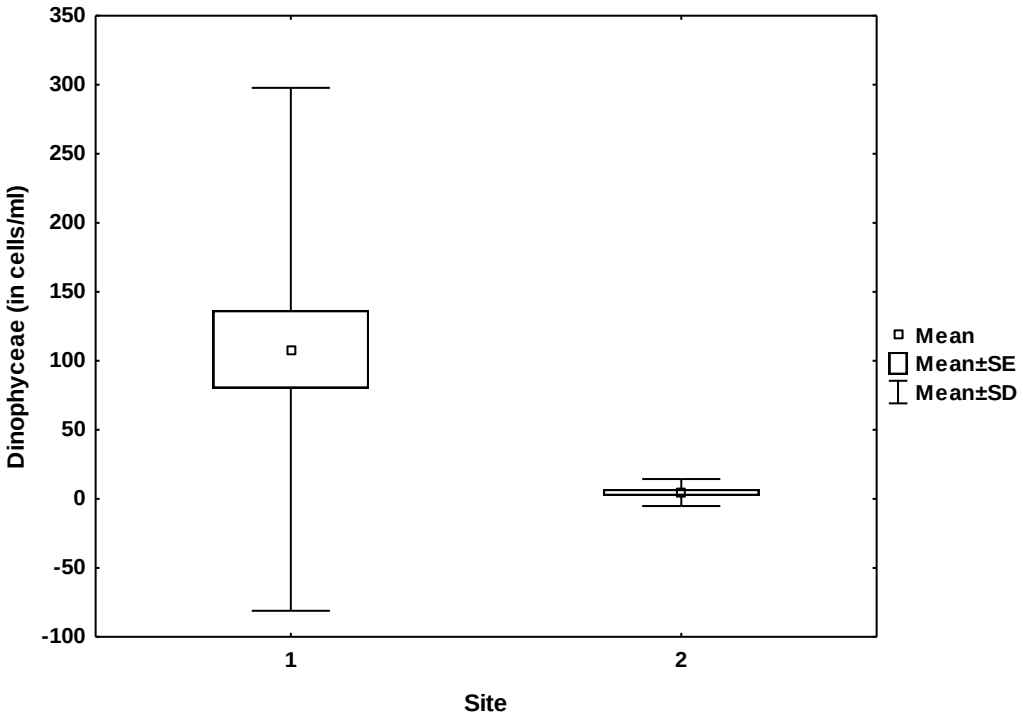


Figure 4.20: A box-and-whisker plot illustrating the difference in the mean Dinophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

The Euglenophyceae concentration for Site 1 was significantly less (p = 0.0003) than the concentration for Site 2 as illustrated by Figure 4.21. The mean Euglenophyceae concentration for Site 2 was 344 cells/ml (Table 4.6) with a maximum of 4362 cells/ml (Table 4.6) recorded during October 2011 with *Trachelomonas* sp. as the dominant species. In contrast, the mean concentration for Site 1 was < 36 cells/ml (Table 4.5) with a maximum of 323 cells/ml (Table 4.5) recorded during August 2008.

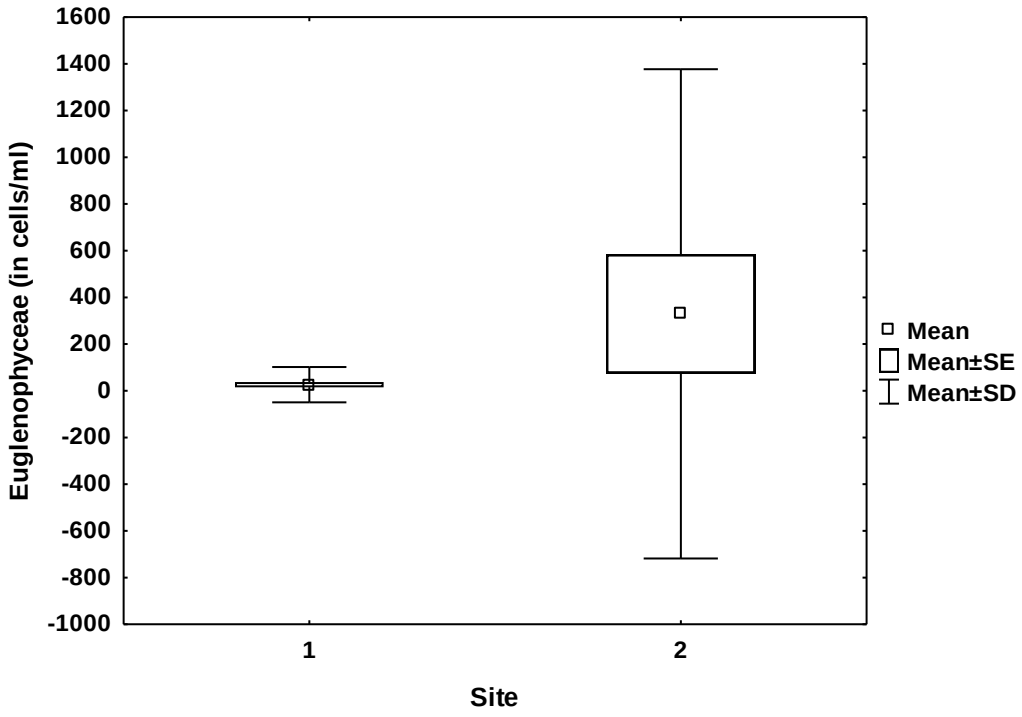


Figure 4.21: A box-and-whisker plot illustrating the difference in the mean Euglenophyceae concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

A significant difference was observed between the total phytoplankton concentration of Sites 1 and 2 (p = 0.027) as illustrated by Figure 4.22 with a much higher concentration recorded in the source water of Site 2. The mean of the total algal cells of Site 1 was 7975 cells/ml (Table 4.5) whereas the mean for Site 2 was 20475 cells/ml (Table 4.6). The maximum total algal cells for Site 1 was 50802 cells/ml (Table 4.5) recorded during November 2010. Site 2 reached a maximum of 140712 cells/ml (Table 4.6) during October 2011. Higher concentrations were recorded for both sites during spring.

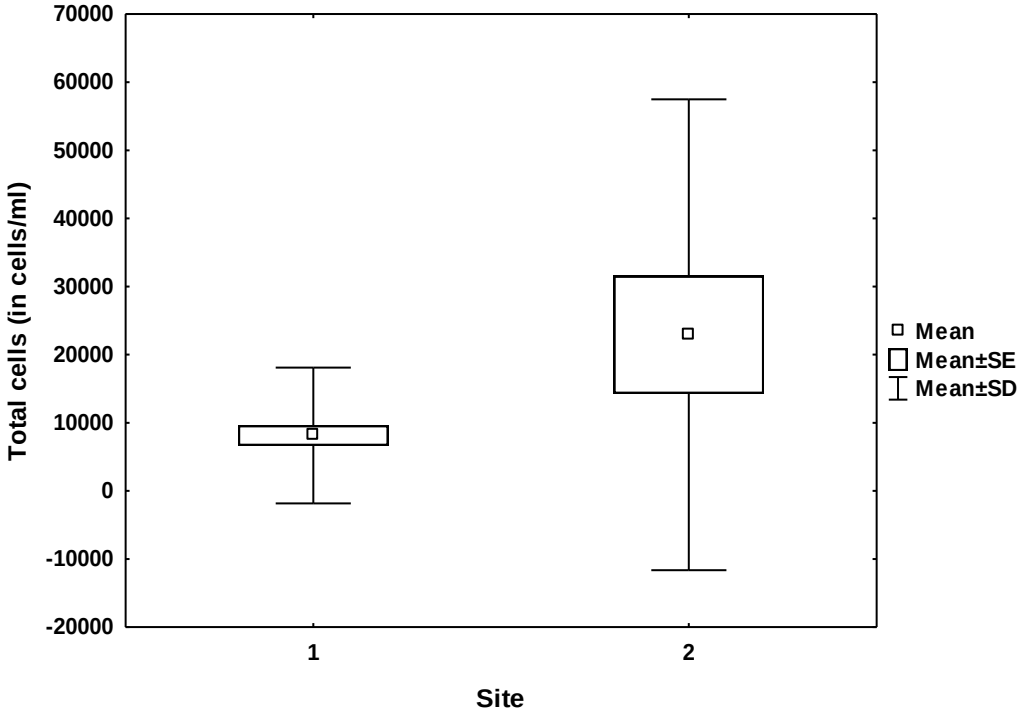


Figure 4.22: A box-and-whisker plot illustrating the difference in the mean total algal cells concentration (in cells/ml) between the source water of Site 1 (January 2009 to December 2010) and the source water of Site 2 (January 2010 to December 2011). SE = Standard Error; SD = Standard Deviation.

4.4.4. Invertebrate composition of Site 2

The source water of Site 1 was not monitored for invertebrates due to high turbidity. Table 4.7 provides a list of the invertebrate groups as well as descriptive statistics for the invertebrate groups that were present in the source water of Site 2. Eight invertebrate groups were enumerated namely Hydracarina, Cladocera, Copepoda, Diptera, Ephemeroptera, Nematoda, Ostracoda and Rotatoria. The remainder of the commonly occurring invertebrate groups were grouped under "Other groups of invertebrates" and a group was created for unknown invertebrates.

Table 4.7: Descriptive statistics for the invertebrate groups that were present in the source water of Site 2 for the sampling period January 2010 to December 2011. SD = Standard Deviation.

Invertebrate group	Unit	Mean	Minimum	Maximum	SD
Hydracarina	org/m^3	< 1	0	1	0
Cladocera	org/m^3	28	0	285	73
Copepoda	org/m^3	18	0	117	30
Diptera	org/m^3	10	0	75	19
Ephemeroptera	org/m^3	< 1	0	3	1
Nematoda	org/m^3	< 1	0	3	1
Ostracoda	org/m^3	1	0	5	2
Rotatoria	org/m^3	89	0	975	248
Unknown invertebrates	org/m^3	22	0	180	56
Other groups of invertebrates	org/m^3	136	2	1003	253
Total invertebrates	**org/m^3**	**302**	**6**	**2008**	**503**

Figure 4.23 illustrates the percentage composition of the invertebrate groups identified in the source water of Site 2. Other groups of invertebrates (44.63%) and Rotatoria (29.27%) were the most abundant groups and combined comprised 75% of the invertebrate composition. Cladocera (9.11%), Unknown invertebrates (7.19%), Copepoda (5.85%) and Diptera (3.29%) comprised 25% of the invertebrate composition. Ostracoda (0.31%), Nematoda (0.15%), Ephemeroptera (0.14%) and Hydracarina (0.06%) occurred only in very small percentages in the source water.

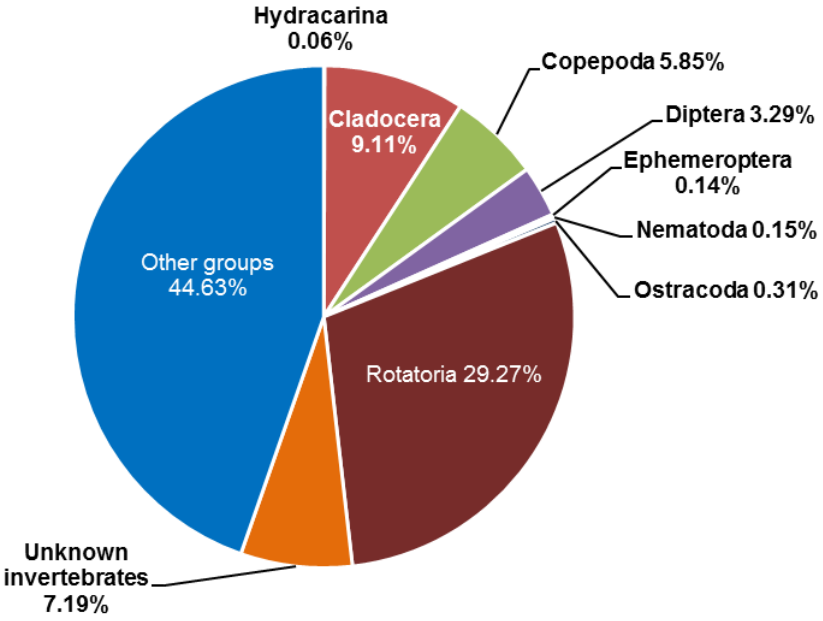


Figure 4.23: A pie chart illustrating the percentage composition of the invertebrate groups that were present in the source water of Site 2 for the sampling period January 2010 to December 2011.

Table 4.7 verifies the abundance of Other groups of invertebrates with a mean concentration of 136 org/m^3 and Rotatoria with a mean concentration of 89 org/m^3 during the sampling period. The maximum concentrations for Other groups of invertebrates (1003 org/m^3) and Rotatoria (975 org/m^3) were both reached during summer 2010 whereas maximum concentrations for Cladocera, Copepoda and Diptera were reached during spring. Other groups of invertebrates were present in all source water samples taken during the sampling period. The mean total number of invertebrates in the source water of Site 2 was 302 org/m^3.

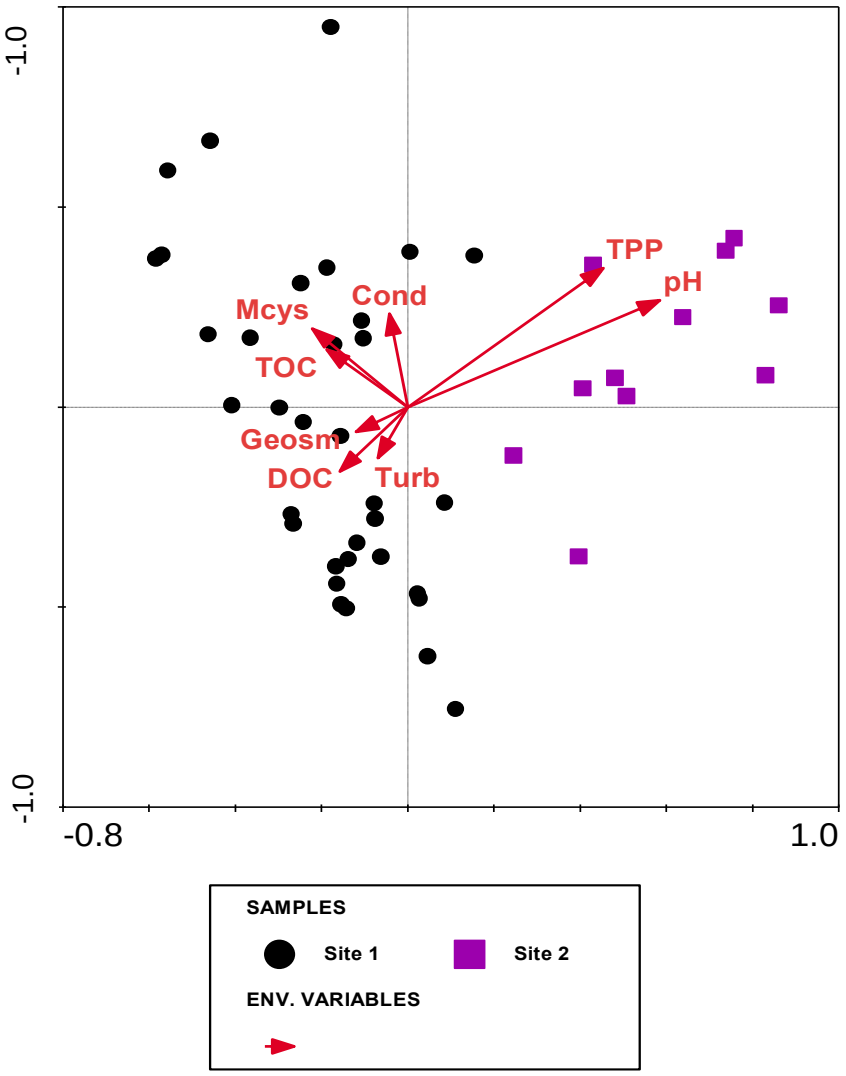
Rand Water's recommended limit for invertebrates in potable water is 20 org/m^3 with a maximum permissible limit of 100 org/m^3 and a crisis limit of 250 org/m^3 (Shaddock, 2006). A more stringent recommended limit of 1 org/m^3 is in place for Diptera due to consumer complaints with a maximum permissible limit of 4 org/m^3 and a crisis limit of 7 org/m^3. The mean concentration of Diptera in the source water of Site 2, 10 org/m^3, as well as the mean total number of invertebrates, 302 org/m^3, exceeded the crisis limit considerably.

4.4.5. Multivariate analysis

The multivariate analysis methods that were used in this study are described in Section 3.4.

An indirect linear gradient analysis, the principal component analysis (PCA), was used to determine the principal physical and chemical source water quality variables associated with Sites 1 and 2. The resulting PCA ordination plot (Figure 4.24) clearly illustrates the differences between the physical and chemical characteristics of Sites 1 and 2.

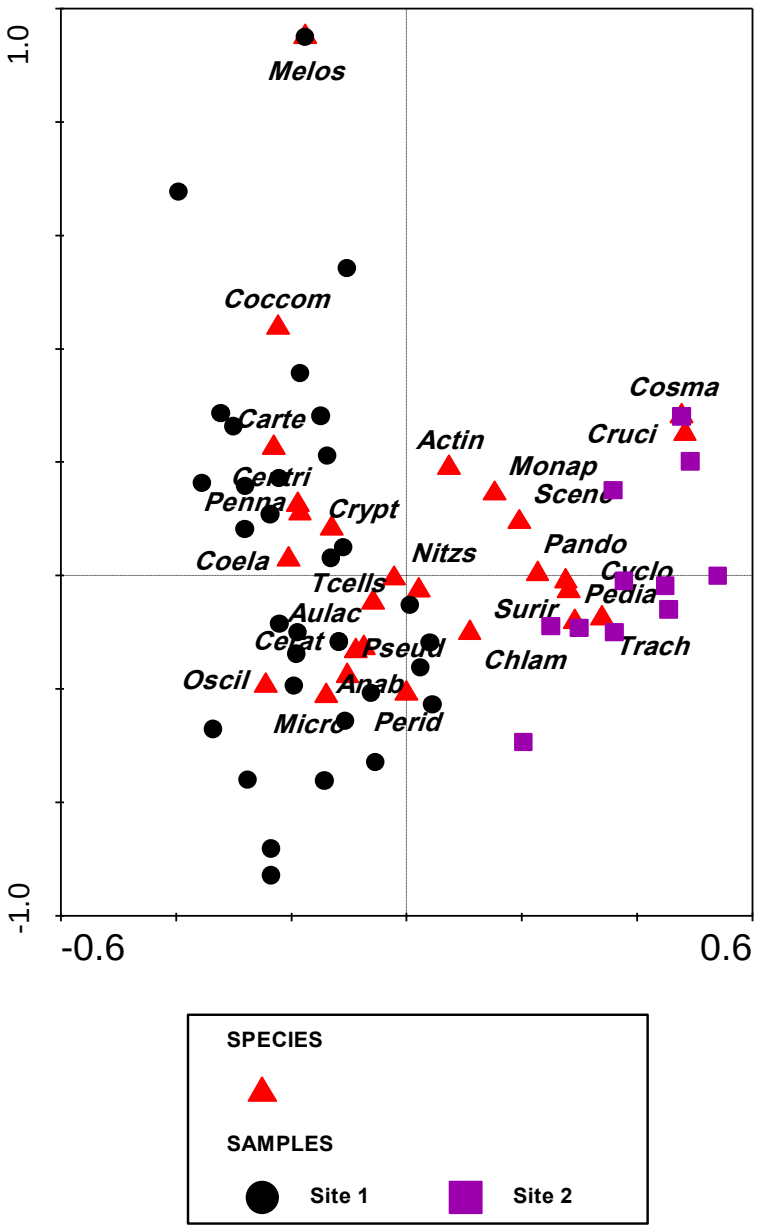
The first axis explained 34.8 % of the variance in the data and the second axis explained 17.4% of the variance. TPP and pH had the biggest influence on the data and the samples of site 2 were associated closely with these components. Samples of site 1 were associated mostly with microcystin and geosmin, DOC, TOC, conductivity and turbidity.



Axes	1	2	3	4	Total variation
Eigenvalues:	0.209	0.178	0.101	0.069	1.000
Species-environment correlations:	0.707	0.541	0.595	0.447	
Cumulative percentage variance					
of species data:	20.9	38.7	48.7	55.6	
of species-environment relation:	34.8	52.2	64.0	68.6	
Sum of all eigenvalues:					1.000
Sum of all canonical eigenvalues:					0.300

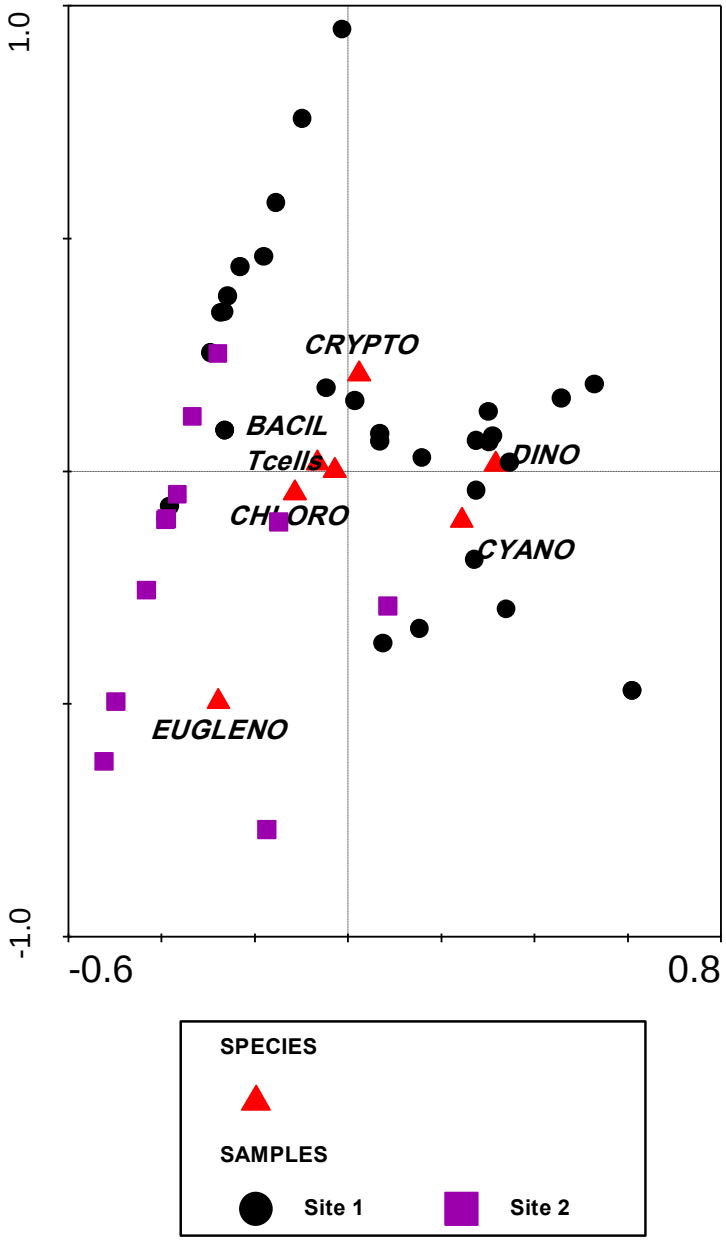
Figure 4.24: A PCA site plot showing the correlation between the principal physical and chemical water quality variables and the source water samples of Sites 1 and 2.

An indirect gradient analysis, the correspondence analysis (CA), was used to determine the similarities or dissimilarities between the algal compositions of Sites 1 and 2. Figure 4.25 is a CA ordination plot showing the algal species compositions of Site 1 and 2 and Figure 4.26 is a CA ordination plot showing the algal classes compositions of Site 1 and 2. The results of both ordination plots clearly illustrate the differences observed in the phytoplankton compositions of the respective sites. The Cyanophyceae, in particular the problematic species *Microcystis* sp., *Anabaena* sp. and *Oscillatoria* sp., Dinophyceae and Cryptophyceae algal classes were associated with the samples of the source water of Site1. The Chlorophyceae and Euglenophyceae algal classes were mostly associated with the samples of the source water of Site 2. The Bacillariophyceae algal class was associated with the samples of both sites. The overall algal species variance was 2.294 and a total variance of 24.7% in the algal species data was explained by the first two canonical axes (Figure 4.25). A total variance of 60% in the algal classes data was explained by the first two canonical axes (Figure 4.26).



Axes	1	2	3	4	Total inertia
Eigenvalues:	0.341	0.226	0.203	0.182	2.294
Cumulative percentage variance of species data:	14.9	24.7	33.6	41.5	
Sum of all eigenvalues:					2.294

Figure 4.25: A CA site plot showing the associations between the algal species and the source water samples of Sites 1 and 2.



Axes	1	2	3	4	Total inertia
Eigenvalues:	0.177	0.111	0.086	0.056	0.480
Cumulative percentage variance of species data:	36.9	60.0	77.9	89.6	
Sum of all eigenvalues:					0.480

Figure 4.26: A CA site plot showing the associations between the algal classes and the source water samples of Sites 1 and 2.

A direct unimodel gradient analysis, the canonical correspondence analysis (CCA), was used to determine the relationships between the physical and chemical characteristics, referred to as the environmental variables, and the species data of Sites 1 and 2.

Figure 4.27 is a CCA ordination biplot that illustrates the relationships between the environmental variables and samples of Sites 1 and 2. Figure 4.28 is a CCA triplot that illustrates the relationships between the environmental variables, samples and algal species compositions of Sites 1 and 2. Monte Carlo Permutation test results for both CCA ordination plots indicated a statistically significant relationship between the environmental variables and samples and between the environmental variables, samples and algal species composition (p <0.05).

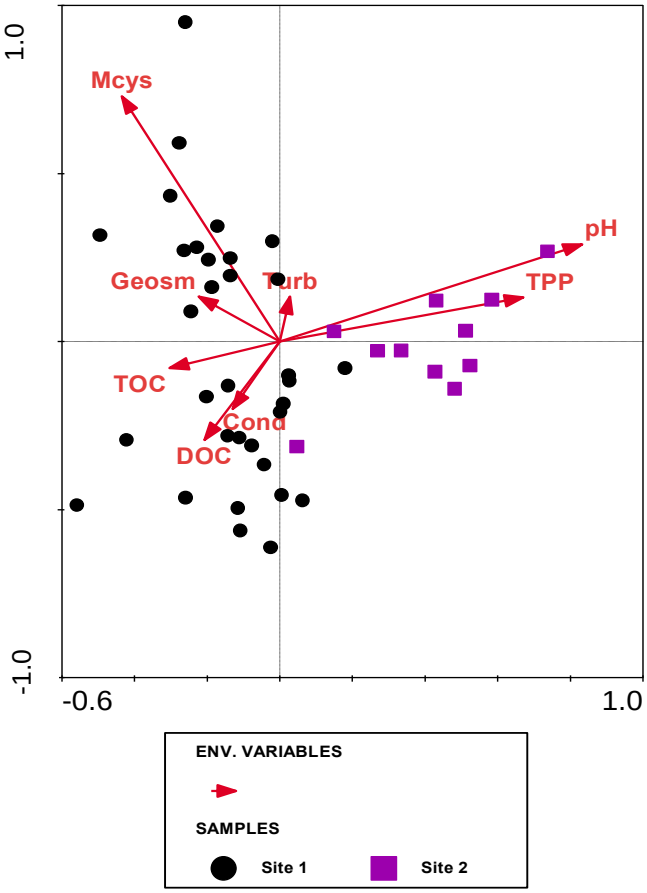
The relationships between the environmental variables and samples illustrated by Figure 4.24 are observed again in Figure 4.27. Figure 4.28 reflects the same environmental and sample relationships with the added dimension of algal species. Although the first four canonical axes of both ordination diagrams only explained 21.9% of the algal species data variance, 82.4% of the variance in species-environment relation data was explained. Microcystin, pH and TPP were the principal contributors to the variance in species-environment relation data. The samples of Site 1 were closely associated with microcystin, geosmin, conductivity, DOC and TOC. The samples of Site 2 were closely associated with TPP and pH and to a lesser extent conductivity. Turbidity appeared to associate with both sites.

As illustrated by Figures 4.27 and 4.28, the following environmental variables correlations were observed:

- pH was positively correlated with TPP but negatively correlated with TOC and DOC;
- Turbidity was negatively correlated with conductivity and DOC;
- Microcystin and geosmin were positively correlated;
- Conductivity, DOC and TOC were positively correlated;
- TPP and pH were negatively correlated with TOC.

Figure 4.28 illustrates the preference of the majority of Chlorophyceae species that ocurred in the source water namely *Pandorina* sp., *Chlamydomonas* sp., *Crucigenia* sp., *Monoraphidium* sp., *Pediastrum* sp., *Scenedesmus* sp. and *Cosmarium* sp. for a more alkaline environment. Two Bacillariophyceae species, *Cyclotella* sp. and *Surirella* sp., and *Trachelomonas* sp, an Euglenophyte, were also strongly associated with high pH values. As these algal species were closely associated with the samples of Site 2, a positive correlation with TPP was also reflected.

As was expected, *Microcystis* sp. was associated with microcystin and geosmin and similarly *Anabaena* sp. and *Pseudanabaena* sp. with geosmin. These Cyanophyceae species were grouped relatively in close proximity to each other as illustrated by Figure 4.28 although *Anabaena* sp. clearly displayed more tolerance for high turbidity. *Oscillatoria* sp., also a Cyanophyceae species, was observed to be further away though possibly suggesting a slightly different preferred environment than the other Cyanophyceae species that were present in the source water. *Actinastratum* sp., a Chlorophyceae species, *Cryptomonas* sp., a Cryptophyceae species, as well as centric diatoms were strongly correlated with conductivity.

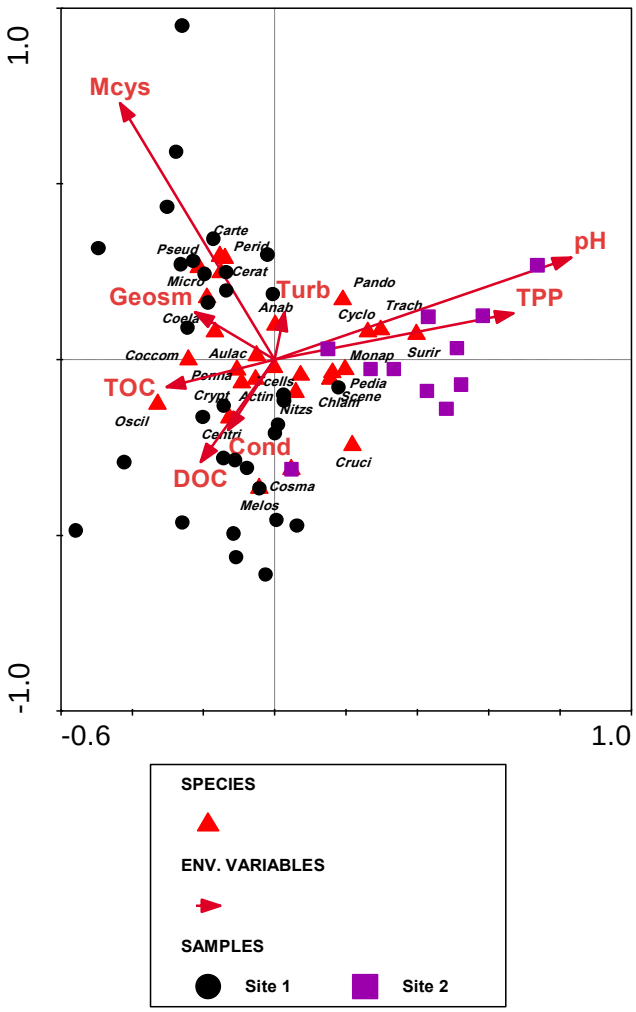


Axes	**1**	**2**	**3**	**4**	**Total inertia**
Eigenvalues:	0.218	0.118	0.098	0.068	2.294
Species-environment correlations:	0.840	0.847	0.753	0.750	
Cumulative percentage variance					
of species data:	9.5	14.7	18.9	21.9	
of species-environment relation:	35.9	55.2	71.2	82.4	
Sum of all eigenvalues:					2.294
Sum of all canonical eigenvalues:					0.609

Monte Carlo test

Test of significance of first canonical axis:	eigenvalue	=	0.218
	F-ratio	=	3.685
	P-value	=	0.0020
Test of significance of all canonical axes:	Trace	=	0.609
	F-ratio	=	1.583
	P-value	=	0.0040

Figure 4.27: A CCA bi-plot showing the relationships between the environmental variables and the source water samples of Sites 1 and 2.



Axes	1	2	3	4	Total inertia
Eigenvalues:	0.218	0.118	0.098	0.068	2.294
Species-environment correlations:	0.840	0.847	0.753	0.750	
Cumulative percentage variance					
of species data:	9.5	14.7	18.9	21.9	
of species-environment relation:	35.9	55.2	71.2	82.4	
Sum of all eigenvalues:					2.294
Sum of all canonical eigenvalues:					0.609

Monte Carlo test

Test of significance of first canonical axis:	eigenvalue	=	0.218	F-ratio	=	3.685
	P-value	=	0.0020			
Test of significance of all canonical axes:	Trace	=	0.609	F-ratio	=	1.583
	P-value	=	0.0040			

Figure 4.28: A CCA tri-plot showing the relationships between the environmental variables and algal species compositions and the source water samples of Sites 1 and 2.

4.4.6. Significant correlations

Spearman's Rank Order Correlations were run as an additional statistical tool to determine correlations between the water quality variables of the source water at both sites. The results of these nonparametric correlation tests substantiated whether a relationship that was established with the multivariate analysis of the data was indeed significant. In addition, the results also indicated significant relationships between the variables that were not observed with the multivariate analysis of the data. As only complete datasets were used for the multivariate analysis, significant correlations with some algal species were observed in these results that were not reflected in the multivariate ordination diagrams.

Correlations were significant at $p < 0.05$. The following significant positive and negative correlations were established between the water quality variables of the source water of both sites for the purposes of this study:

- The strongest correlation was the negative correlation between conductivity and turbidity followed by the positive correlation between pH and TPP;
- TPP and total cells were positively correlated;
- pH was negatively correlated with turbidity and DOC and positively with conductivity;
- Conductivity and DOC were positively correlated;
- Turbidity and TPP were negatively correlated;
- DOC and TOC were positively correlated;
- *Microcystis* sp. and microcystin were positively correlated;
- pH was positively correlated with Bacillariophyceae, Chlorophyceae and Euglenophyceae;
- DOC was positively correlated with Cryptophyceae;
- TPP was positively correlated with Bacillariophyceae, Chlorophyceae and Euglenophyceae.

4.5. Discussion

Studies conducted by Janse van Vuuren (2001), Kruskopf (2002), Carrim (2006) and Morrison (2009) determined the characteristics of the source water quality variables and correlations between the variables at different periods in the Middle Vaal WMA. Compatible data, when available, from these studies was used as a comparative tool for the current study as Site 2 was included as a sampling site in all these studies. Janse van Vuuren's (2001) and Kruskopf's (2002) studies also included upstream sampling sites namely Parys and Barrage that were used as a comparison for Site 1. Janse van Vuuren's (2001) study was conducted from 1992 to 1997 and Kruskopf's (2002) study was conducted from 1999 to 2001. Means were determined for the

respective periods for purposes of this study. Carrim's (2006) study was conducted from 2005 to 2006 and Morrison's (2009) study was conducted from 2007 to 2008.

4.5.1. Physical and chemical characteristics of the source water

A clear distinction was observed between the samples of Site 1 and the samples of Site 2 (Figure 4.24) with a downstream change in the source water. The samples of Site 2 were associated with TPP and pH whereas the samples of Site 1 were associated mostly with microcystin, geosmin, DOC, TOC, conductivity and turbidity. Statistical significant differences between all the physical and chemical variables of the source water of Sites 1 and 2 were recorded.

Table 4.8 provides a comparison between the physical and chemical characteristics of the source water that were determined in previous studies and in the current study.

Table 4.8: A comparison of the physical and chemical characteristics of the source water of Sites 1 and 2 as determined in this study with previous studies.

Variable	Unit	Janse van Vuuren (2001)		Kruskopf (2002)		Carrim (2006)	Morrison (2009)	Current study	
		Parys	Site 2	Barrage	Site 2	Site 2	Site 2	Site 1	Site 2
pH	pH units	8.2	8.2	7.9	8.6	8.6	8.64	8.04	8.73
TPP	µg/L						105	62.196	78.531
Chlorophyll-*a*	µg/L	37.98	38.65	36.65	42.15	90.75	64.64		
Conductivity	mS/m	62	65	46.2	68.05		76	61.533	55.7
Turbidity	NTU	29.73	51.93	26.1	31.95	45	12	23.505	37.731
DOC	mg/L					6	6.2	6.842	6.331
Geosmin	ng/L						<6	8.423	4.083

Numerous aspects of water quality are influenced by pH such as taste, corrosivity and the solubility and speciation of metal ions. Water tastes sour at a pH below 6 and bitter at a pH above 9. The most important significance of pH though in terms of water purification concerns its effects on water treatment processes (DWAF, 1998).

According to Schutte (2006), pH is an important factor in the coagulation-flocculation treatment process. Coagulants such as aluminium sulphate, ferric chloride, hydrated lime and activated silica are added during this process to achieve destabilisation of colloidal particles in the water in order to form flocs. Aluminium sulphate requires a controlled pH of between 6 and 7.4 as aluminium must precipitate completely. The optimum pH for ferric chloride is between 5 and 8. pH increases when

lime is added and therefore the pH must be lowered when lime is used (Schutte, 2006). Hydrated lime and activated sodium silicate are used as coagulants and ferric chloride to aid flocculation at Site 1 and ferric chloride and aluminium sulphate are used as coagulants at Site 2 with hydrated lime for pH correction. According to Spellman (2003), one of the most important treatment processes is disinfection. The contact time required if chlorine is used for disinfection is related to increases in pH.

High pH values were recorded for both sites during the respective sampling periods. A statistically significant difference was observed between the pH values of Sites 1 and 2 with a downstream increase. The mean pH of the source water of Site 1 was 8.04 and for Site 2 was 8.73 in accordance with previous studies. Janse van Vuuren (2001) recorded mean pH values of 8.2 for both sites. A downstream increase was also observed with Kruskopf's (2002) recorded mean pH values of 7.9 and 8.6 (Table 4.8). Carrim (2006) and Morrison (2009) recorded mean pH values of 8.6 and 8.64 respectively.

The chlorophyll-665 method was used to determine the TPP in the source water of both sites. TPP includes the chlorophyll-*a* pigment, phaeophytin-*a* as well as other pigments. The mean TPP value for Site 1 was 62.196 µg/L and for Site 2 was 78.531 µg/L with a downstream increase. The mean TPP value of 105 µg/L as observed by Morrison (2009) was higher than both these values. The seasonal fluctuations that were observed by Morrison (2009) differed from this study as TPP values increased during winter and spring and decreased during summer and autumn (Figure 4.11). These seasonal fluctuations were also observed by Janse van Vuuren (2001). Chlorophyll-*a* measurements were used as a comparison when TPP measurements were not available. It appears from Table 4.8 upon comparing chlorophyll-*a* or TPP that previous studies found a downstream increase. Janse van Vuuren's (2001) recorded chlorophyll-*a* measurement for Parys increased from 37.98 µg/L to 38.65 µg/L for Site 1. Kruskopf's (2002) recorded Barrage chlorophyll-*a* value of 36.65 µg/L increased to 42.15 µg/L. It was also observed from Table 4.8 that not only was there a downstream increase of chlorophyll-*a* between sites but also a general increase in chlorophyll-*a* in the Middle Vaal WMA.

A strong positive correlation between pH and TPP was observed (Figure 4.24) that was confirmed as statistically significant by a Spearman test. pH increases when algae utilise carbon dioxide and carbonate levels are depleted during the second step of the photosynthesis process (Knappe, 2004). High photosynthesis rates were further confirmed by the significant positive correlation between TPP and the total algal biomass. Janse van Vuuren (2001) showed that high algal biomass correlated positively with high chlorophyll-*a* concentrations. This positive relationship can be observed in Table 4.8 as higher pH values correspond to higher TPP or chlorophyll-*a* values. The

increase in algal biomass in the Middle Vaal WMA is indicative of an increase in the development of nuisance conditions. Site 2 makes use of a pre-ozonation step to assist with the reduction of TPP.

Conductivity is used as an estimate of the Total Dissolved Solids (TDS) in water. TDS is comprised of various dissolved inorganic salts. The total hardness and the corrosive and scaling potential of water are closely associated with TDS. TDS is most likely to accumulate in water moving downstream as salts are continuously being added through surface runoff (DWAF, 1998). The mean conductivity for Site 1 was 61.533 mS/m and 55.7 mS/m for Site 2. Conductivity values of both sites exceeded the TWQR for conductivity periodically. The lower conductivity values that were recorded for Site 2 were unexpected as Site 2 is located downstream from Site 1. Janse van Vuuren (2001) observed a negative relationship between discharge and conductivity and an increase in discharge at Site 2 can possibly explain the lower conductivity values. Janse van Vuuren reported a mean conductivity of 62 mS/m for Parys and 65 mS/m for Site 2 over the six year study period (Janse van Vuuren, 2001). Kruskopf (2002) recorded mean conductivity values of 46.2 mS/m for Barrage and 68.05 mS/m for Site 2. These values reflect the downstream increase of TDS and this is indicative of increased salinity (Table 4.8). Morrison (2009) found a TDS mean of 76 mS/m.

The removal of turbidity from source water requires an effective coagulation- flocculation process as turbidity is caused by suspended inorganic and organic matter. Turbidity is strongly associated with water colour and the presence of micro-organisms and turbid source water also requires more chlorine for disinfection purposes (DWAF, 1998). Site 1 uses GAC adsorption and Site 2 DAF to enhance the removal of turbidity. All the turbidity measurements of both sites exceeded the SANS 241 limit for turbidity with a mean of 23.505 NTU for Site 1 and a mean of 37.731 NTU for Site 2. The turbidity values of Site 2 were significantly higher than the values of Site 1 indicating a downstream increase in turbidity. The maximum turbidity values of both sites, 129 NTU and 142 NTU respectively, were reached during the rainy season confirming Carrim's (2006) and Morrison's (2009) findings that turbidity and rainfall are positively correlated. Kruskopf (2002) recorded mean turbidity values of 26.1 NTU for Barrage and 31.95 NTU for Site 2 and Janse van Vuuren (2001) recorded mean turbidity values of 29.73 NTU and 51.93 NTU for Site 2 (Table 4.8) showing a clear downstream increase. Morrison (2009) recorded a mean turbidity of 12 NTU.

A strong, statistical significant negative correlation was observed between turbidity and conductivity (Figures 4.24 and 4.27) that was confirmed with a Spearman's test. Janse van Vuuren (2001) found a positive correlation between discharge and turbidity. Therefore when discharge and turbidity increased, conductivity decreased as discharge has a dilution effect on the TDS in the water.

Turbidity also displayed a statistical significant negative relationship with pH and TPP. This negative correlation was as a result of impeded light penetration due to high turbidity that led to reduced rates of photosynthesis (Dallas & Day, 2004).

The presence of TOC (total organic carbon in solution and suspended form) and DOC (the dissolved organic carbon) in source water can have aesthetic implications. In source water that receives organic waste from agricultural runoff high concentrations of DOC can occur with toxicity implications. If the source water DOC concentrations are only slightly elevated, DOC may be removed through coagulation-flocculation, settlement and filtration. A GAC treatment step, such as the GAC adsorption step in use by Site 1, may assist with the removal of DOC (DWAF, 1996). Site 2 uses DAF as an additional treatment step to aid conventional treatment steps in the removal of DOC. The DOC values of Site 2 with a mean of 6.331 mg/L did not exceed the operational limit for DOC and was similar to the mean values recorded by Carrim (2006) of 6 mg/L and by Morrison (2009) of 6.2 mg/L. A mean TOC value of 7.028 mg/L was recorded for Site 2. The DOC and TOC values of Site 1 were significantly higher with means of 6.842 and 7.892 mg/L.

Positive correlations were observed between TOC, DOC and conductivity (Figures 4.24 and 4.27). Conductivity represents TDS in the source water. TOC and DOC are major components of TDS which is comprised of all the dissolved organic and inorganic material in water (Dallas & Day, 2004).

Microcystins are cyanobacterial toxins produced by certain frequently occurring cyanobacterial species such as *Microcystis* sp., *Anabaena* sp. and *Oscillatoria* sp. Substantial amounts of microcystins are released into the water through cell lysis as microcystins are usually cell-bound (WHO, 2011). According to Swanepoel *et al.*, (2008a), the occurrence of microcystin in source and domestic water poses a health risk to consumers. Microcystin monitoring enables pro-active actions to avoid possible illnesses arising from the consumption of water that contains the toxin. The microcystin values recorded at Site 1 were relatively high with a mean of 0.523 µg/L while the mean microcystin value recorded at Site 2 was < 0.18 µg/L. The maximum values of Sites 1 and 2, 2.34 µg/L and 0.25 µg/L respectively, were both reached during summer. Optimised coagulation-flocculation, filtration or DAF processes can control cell-bound microcystins effectively. Activated carbon, ozonation and chlorination can effectively remove microcystins that are dissolved in the water as a result of cell lysis (WHO, 2011). Site 1 uses GAC adsorption and Site 2 uses DAF and ozonation in addition to conventional treatment steps for the removal of microcystin. The correlation observed between microcystin and *Microcystis* sp. (Figure 4.28), indicates that this species was primarily responsible for the production of microcystin in the source water and it was confirmed with a Spearman's test as statistically significant.

Geosmin, a secondary metabolite, is one of the most important compounds responsible for the earthy or musty odour of water. In addition to actinomycetes, certain Cyanophyceae species, *Microcystis* sp., *Anabaena* sp. and *Oscillatoria* sp., are known producers of geosmin (Swanepoel *et al*., 2008a). The geosmin values recorded at Site 1 with a mean of 8.423 ng/L were considerably higher than the values of Site 2 with a mean of < 5 ng/L. The geosmin values of Site 1 exceeded the limit of 30 ng/L twice with a high maximum of 80.37 ng/L reached during November 2010. The geosmin values of Site 2 remained well below the limit. Morrison (2009) also found that the geosmin levels of the source water of Site 2 did not exceed the detection limit. An advanced water treatment process such as activated carbon is required to remove geosmin from water (Swanepoel *et al*., 2008a). Site 1 uses GAC adsorption and Site 2 uses intermediate ozonation as treatments for taste and odour producing compounds such as geosmin. Positive correlations between geosmin and two known producers of geosmin, *Anabaena* sp. and *Microcystis* sp., were observed but did not prove to be statistically significant. A statistically significant correlation between *Ceratium* sp. and geosmin was found though. Swanepoel *et al*. (2008b) found that high concentrations of Cyanophyceae and *Ceratium* sp. occur together and therefore *Ceratium* sp. may be correlated with the products, such as geosmin, of certain Cyanophyceae species. High geosmin concentrations corresponded to high concentrations of *Anabaena* sp. and *Microcystis* sp. that indicated that these two species were major contributors to the geosmin production in the source water of Site 1.

4.5.2. Algal characteristics of the source water

The determination and analysis of the algal compositions of the source water play a vital role in the purification of water for domestic purposes. Phytoplankton and related extra-cellular products can affect the physico-chemical reactions of water purification processes. Phytoplankton may circumvent the purification processes resulting in the production of aesthetically unacceptable potable water. Some algal species produce toxins that can be detrimental to consumers' health if not removed completely (Swanepoel *et al.*, 2008a). Different algal species also cause different problems in the purification process. Knowledge of the algal characteristics of source water is therefore imperative in order to adapt purification processes accordingly.

Seven algal classes were identified in the source water of Sites 1 and 2 namely Bacillariophyceae (diatoms), Chlorophyceae (green algae), Chrysophyceae, Cryptophyceae, Cyanophyceae (blue-green bacteria), Dinophyceae and Euglenophyceae. Chrysophyceae species were scarce as only one species was recorded in the source water of Site 1. The limited occurrence of Chrysophyceae species was in accordance with the findings of Janse van Vuuren (2001), Kruskopf (2002), Carrim (2006) and Morrison (2009) in the Vaal River. Eutrophic conditions and a pH above 7.5 are not

conducive to the growth of Chrysophyceae species (Van den Hoek *et al*., 1995) which would explain the absence of these species in the source water.

The source water of Sites 1 and 2 were clearly differentiated by their respective phytoplankton compositions (Figures 4.25 and 4.26). Different algal classes and species were associated with each site. Site 1 was dominated by Bacillariophyceae and Cyanophyceae whereas Site 2 was dominated by Chlorophyceae and to a lesser extent Bacillariophyceae (Figures 4.14 and 4.15). Euglenophyceae species were more abundant in the source water of Site 2 whereas Cryptophyceae and Dinophyceae species were more abundant in the source water of Site 1. The statistical significant differences between the algal class concentrations of Site 1 and 2 further substantiated the dissimilar algal composition of each site. The Cyanophyceae, Cryptophyceae and Dinophyceae concentrations were significantly more in the source water of Site 1 whereas the Chlorophyceae and Euglenophyceae concentrations were significantly more in the source water of Site 2. The total algal biomass also differed significantly with more algal cells recorded in the source water of Site 2. There was no significant difference between the Bacillariophyceae species concentration of Sites 1 and 2.

Janse van Vuuren (2001), Kruskopf (2002), Carrim (2006) and Morrison (2009) found Bacillariophyceae and Chlorophyceae to be the dominant algal classes in the source water of Site 2 in accordance with the findings of this study. In addition, Janse van Vuuren (2001) and Kruskopf (2002) also found Bacillariophyceae and Chlorophyceae to be the dominant algal classes in the source water of the upstream sites. It was found in this study that Cyanophyceae replaced Chlorophyceae as the second most dominant algal class in the source water of Site 1. According to Walmsley (2000) an increasing dominance by Cyanophyceae is one of the problems associated with excessive eutrophication. The prevalence of Cyanophyceae in the source water of Site 1 is indicative of high levels of nutrients that can lead to an increased frequency of water quality problems.

The algal composition changed significantly in a downstream direction. Cyanophyceae changed from being a dominant class in the source water of Site 1 to comprising only a negligible percentage of the algal composition of the source water of Site 2. Chlorophyceae increased dramatically in the source water of Site 2 (Figure 4.14 & 4.15) from 14.76% to 70.15%. Bacillariophyceae decreased but remained a dominant algal class. Dinophyceae and Cryptophyceae decreased and Euglenophyceae increased. These downstream changes of the algal compositions confirmed the observed changes in the source water from upstream to downstream. A downstream decrease in the percentage composition of Cyanophyceae and increases in Chlorophyceae and Euglenophyceae percentage compositions were observed by Janse van Vuuren (2001) as well.

A list of problematic algal species that were present in the source water of Sites 1 and 2 was compiled (Table 4.9) for the purposes of this study. These species can potentially have an impact on the water treatment processes. Table 4.9 also provides information on the types of problems that can be caused by these species and the site association. A complete list of algal species that were present in the source water of Sites 1 and 2 is provided in Table 4.5.

Table 4.9: A list of problematic algal species that were present in the source water of Sites 1 and 2 during the respective sampling periods and associated potential water treatment problems (adapted from Janse van Vuuren *et al.*, 2006).

Species	Site mostly associated with	Taste and odour	Filter clogging	Toxins	Blooms
Bacillariophyceae					
Aulacoseira sp.	1	✓	✓		✓
Other centric diatoms	1	✓	✓		✓
Cyclotella sp.	2		✓		✓
Chlorophyceae					
Coelastrum sp.	2				✓
Pandorina sp.	1	✓			✓
Pediastrum sp.	2				✓
Scenedesmus sp.	2				✓
Cyanophyceae					
Anabaena sp.	1	✓	✓	✓	✓
Microcystis sp.	1	✓	✓	✓	✓
Oscillatoria sp.	1	✓	✓	✓	✓
Cryptophyceae					
Cryptomonas sp.	1				✓
Dinophyceae					
Ceratium sp.	1	✓	✓		✓
Euglenophyceae					
Trachelomonas sp.	2				✓

Bacillariophyceae species featured prominently in the source water of both sites. *Cyclotella* sp. occurred in the source water of both sites but was far more abundant in the source water of Site 2 and formed blooms during autumn and spring. *Aulacoseira* sp. featured as a dominant species in the source water of Site 1 and occurred from autumn to spring forming blooms during spring. Other

unidentified centric diatoms also displayed a tendency to form blooms during spring in the source water of Site 1. According to Knappe (2004) members of Bacillariophyceae are frequently associated with filter clogging and taste and odours problems in source water (Table 4.9). At low water temperatures Bacillariophyceae species display an advantage over other algal species due to their ability to perform oxygenic photosynthesis. *Cyclotella* sp. was closely associated with the samples of Site 2 and *Aulacoseira* sp. and other unidentified centric diatoms with the samples of Site 1 (Figures 4.25 & 4.28). A statistical significant relationship was found between pH and *Cyclotella* sp. (Figure 4.28). Logsdon (2002) found that large numbers of *Cyclotella* sp. can influence the pH of source water causing pH to increase due to high photosynthetic rates.

The majority of the Chlorophyceae species were associated with Site 2. *Scenedesmus* sp. was the dominant species in the source water of Site 2 and large numbers were recorded during the summer months. *Scenedesmus* sp. is an indicator of nutrient-enriched water as a result of anthropogenic contributions (Janse van Vuuren *et al.*, 2006). Similarly, *Pediastrum* sp. and *Coelastrum* sp. also occurred abundantly during the same periods as *Scenedesmus* sp. in the source water of Site 2 and are also indicative of eutrophic water. *Pandorina* sp. occurred frequently in the source water of both sites and formed blooms during spring. This species can impart a fishy odour to the water when present in large numbers (Janse van Vuuren *et al.*, 2006).

Cryptophyceae was represented by *Cryptomonas* sp. that occurred in moderate numbers in the source water of Site 1 during winter periods. *Cryptomonas* sp. is indicative of organically enriched water (Janse van Vuuren *et al*., 2006) and this was confirmed with a statistically significant positive relationship between *Cryptomonas* sp. and DOC.

Some Cyanophyceae species are among the most problematic from a water treatment perspective. Many common Cyanophyceae species produce toxins, cause filter clogging and taste and odour problems. A high standard deviation of 7860 cells/ml for Cyanophyceae species was observed in the source water of Site 1. This was an indication of the variability in the Cyanophyceae concentrations and can be attributed to the presence of algal blooms. Cyanophyceae blooms mostly occur during summer when water temperatures are higher in nutrient-enriched water (Knappe, 2004). *Microcystis* sp., *Anabaena* sp. and *Oscillatoria* sp. were present in the source water of Sites 1 and 2. These species were considerably more abundant in the source water of Site 1 with *Microcystis* sp. as the dominant species in the source water of Site 1. *Microcystis* sp. blooms did not only occur during summer in the source water of Site 1 but also during the winter of 2009. This unusual occurrence can be attributed to the fact that 2009 was the eighth warmest year globally since 1880 (NOAA, 2013). According to the Council for Scientific and Industrial Research (CSIR, 2009), if the current climate change trends continue, an increase in toxic algal blooms can

be expected as warmer surface waters are conducive to the formation of these blooms. According to Harding and Paxton (2001), cyanobacterial blooms usually occur at temperatures above 20˚C. Different Cyanophyceae species have wide ranges of temperature extremes and *Microcystis* spp., in particular, can tolerate temperatures over a wide range. *Anabaena* spp. were abundant during the summer months in the source water of Site 1. It was observed that *Anabaena* sp. clearly displayed more tolerance for high turbidity. Cyanophyceae can tolerate lower levels of light intensity due to the presence of phycobiliproteins that can absorb light in the green spectrum (Knappe, 2004).

Ceratium sp. is a well-known member of Dinophyceae and is associated with taste and odour problems. *Ceratium* sp. is tolerant of a wide range of aquatic chemistry conditions (Knappe, 2004). This species frequently occurred in the source water of Site 1 although not in large numbers. According to Ewerts *et al*. (2013), the large cells of *Ceratium* sp. are known to cause extensive water treatment problems even in small numbers. In addition, Ewerts *et al*. (2013) found that conventional water treatment processes are inadequate to remove large quantities of *Ceratium* sp. Swanepoel *et al.* (2008b) found that *Ceratium hirundinella* can penetrate into the potable water in a convention water treatment plant. *Ceratium* sp. are also known for causing widespread blooms. Van Ginkel *et al.* (2001) recorded *Ceratium hirundinella* in bloom forming conditions for the first time in the Hartbeespoort Dam in 1999. Hart and Wragg (2009) recorded a lake-wide bloom of *Ceratium hirundinella* in the Albert Falls Dam in October 2006.

Euglenophyceae species were relatively scarce with the exception of *Trachelomonas* sp. that was mostly associated with Site 2. *Trachelomonas* sp. is not associated with any known problems in source water, but is more common in source water where the concentration of organic matter is higher (Janse van Vuuren *et al*., 2006). *Trachelomonas* sp. impregnate iron and manganese in their loricas and Janse van Vuuren (2001) suggested that a possible explanation of the relative abundance of this species in the source water of Site 2 was that the water was polluted with heavy metals and organic substances originating from the mines adjacent to the Middle Vaal WMA.

Table 4.10 indicates whether the problematic species encountered in the source water during this study were present in the source water during previous studies.

Table 4.10: A comparison of the problematic algal species that were present in the source water of Sites 1 and 2 during the respective sampling periods with the problematic algal species found in previous studies (✓ = present; x = absent).

Species	Janse van Vuuren (2001)		Kruskopf (2002)		Carrim (2006)	Morrison (2009)	Current study	
	Parys	Site 2	Barrage	Site 2	Site 2	Site 2	Site 1	Site 2
Bacillariophyceae								
Aulacoseira sp.	✓	✓	✓	✓	✓	✓	✓	✓
Other centric diatoms	✓	✓	✓	✓	✓	✓	✓	✓
Cyclotella sp.	✓	✓	x	x	✓	✓	✓	✓
Chlorophyceae								
Coelastrum sp.	✓	✓	✓	✓	✓	✓	✓	✓
Pandorina sp.	✓	✓	✓	✓	✓	✓	✓	✓
Pediastrum sp.	✓	✓	x	✓	✓	✓	✓	✓
Scenedesmus sp.	✓	✓	✓	✓	✓	✓	✓	✓
Cyanophyceae								
Anabaena sp.	✓	✓	✓	x	x	x	✓	✓
Microcystis sp.	✓	✓	✓	x	x	✓	✓	✓
Oscillatoria sp.	✓	✓	✓	✓	x	✓	✓	✓
Cryptophyceae								
Cryptomonas sp.	✓	✓	✓	✓	x	✓	✓	✓
Dinophyceae								
Ceratium sp.	x	x	✓	x	x	✓	✓	✓
Euglenophyceae								
Trachelomonas sp.	x	x	✓	✓	✓	✓	✓	✓

It appears from Table 4.10 that the occurrence of *Ceratium* sp. was more frequent since 2008. Although the occurrence of *Cyclotella* sp. seemed to have increased, this cannot be confirmed Kruskopf (2002) did not identify centric diatoms to genus level.

Spearman's correlation tests confirmed that Bacillariophyceae, Chlorophyceae and Euglenophyceae species were the major contributors to TPP. Furthermore, these three algal classes were positively correlated with pH. As the samples of Site 2 were mostly associated with TPP and pH, this observation serves to confirm the association of different algal classes and species with different aquatic environmental conditions.

4.5.3. Invertebrate composition of the source water of Site 2

The occurrence of aquatic invertebrates in drinking water decreases its aesthetic value. A common user complaint is the presences of little crustaceans and worms in domestic water as it is associated with low hygiene. It is therefore necessary to maintain invertebrate numbers at an undetectable level to consumers. In addition to affecting the aesthetic quality of water, invertebrates can also interfere with treatment processes. Large numbers of invertebrates can block filters and disrupt the disinfection process (Shaddock, 2006).

Eighteen invertebrate groups commonly occur in potable water distribution systems (AWWA, 1995, as quoted by Ferreira and Du Preez, 2012). This includes the dominant groups Diptera, Copepoda and Rotatoria and other common groups such as Amphipoda, Cladocera, Nematoda, Ostracoda, Turbellaria, Gastrotricha, Tardigrada, Annelida, Gastropoda, Hydracarina, Bryozoa, Ephemeroptera, Protozoa, Collembola and Nematomorpha. In this study eight invertebrate groups were enumerated namely Hydracarina, Cladocera, Copepoda, Diptera, Ephemeroptera, Nematoda, Ostracoda and Rotatoria. The remainder of the commonly occurring invertebrate groups were grouped under “Other groups of invertebrates” and a group was created for unknown invertebrates. Other groups of invertebrates and Rotatoria were the dominant invertebrate groups in the source water of Site 2. These two invertebrate groups were present in all the samples and both reached their maximum concentrations during summer. Cladocera, Diptera and Copepoda reached their maximum concentrations during spring. This seasonal succession was in accordance with the findings of Ferreira and Du Preez (2012). Steynberg *et al.* (1996), found that the sand filtration step is capable of removing the majority of invertebrates from the source water. According to Logsdon (2002), members of Rotatoria have the ability to penetrate through to the filter by using their appendages to move through the sand. Pre-disinfection is required to prevent this.

4.6. Conclusions

The quality of the source water of the two sampling sites, RWB (Site 1) and MWC (Site 2), was established by analysing the physical and chemical characteristics and the algal compositions of the two sites as well as the invertebrate composition of Site 2. Correlations between the physical and chemical water quality variables and between the physical and chemical variables and algal compositions provided further insight into the quality of the source water. In determining the quality of the source water, water quality problems that can potentially interfere with, or have an impact on treatment processes, were also identified.

High TPP concentrations indicate high chlorophyll concentrations. The source water of both sites can be regarded as hypertrophic as the mean TPP concentrations exceeded 25 µg/L (OECD,

1982). According to Walmsley (2000), “hypertrophic” indicates excessive levels of nutrient-enrichment where physical factors control plant growth and water quality problems are experienced on a continuous basis. A comparison of this study to previous studies confirmed the same trend of downstream changes in the aquatic environment. It also appeared that there has been a general increase in chlorophyll-*a* concentrations over the past decade which is an indication of increased eutrophication. All the physical and chemical water quality variables measured in the source water exceeded the TWQR’s or alternatively the SANS 241 limits or the guidelines stipulated by Rand Water at some point.

The invertebrate composition of the source water of Site 2 was dominated by Rotatoria and Other groups of invertebrates. The total number of invertebrates as well as the number of Diptera organisms far exceeded the crisis limit set for invertebrates in potable water.

The source water of Site 1 differed significantly from the source water of Site 2 not only in terms of physical and chemical characteristics but also in terms of algal composition:

- Site 1 was characterised by significantly higher microcystin, geosmin, conductivity and DOC values;
- Site 2 was characterised by significantly higher pH, TPP and turbidity values;
- The dominant algal classes in the source water of Site 1 were Bacillariophyceae and Cyanophyceae;
- The dominant algal classes in the source water of Site 2 were Chlorophyceae and Bacillariophyceae;
- Euglenophyceae species were more abundant in the source water of Site 2 whereas Dinophyceae and Cryptophyceae species were more abundant in the source water of Site 1;
- Potentially problematic species that were mostly associated with the samples of the source water of Site 1 included *Aulacoseira* sp., other unidentified centric diatoms, *Pandorina* sp., *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp., *Cryptomonas* sp., *Ceratium* sp. and *Trachelomonas* sp.;
- Potentially problematic species that were mostly associated with the samples of the source water of Site 2 included *Cyclotella* sp., *Coelastrum* sp., *Pediastrum* sp. and *Scenedesmus* sp.;
- *Microcystis* sp. was positively correlated with microcystin as well as geosmin and *Anabaena* sp. with geosmin;
- The problematic dinoflagellate, *Ceratium* sp., seemed to occur more frequently in Middle Vaal WMA;

- The Bacillariophyceae, Chlorophyceae and Euglenophyceae algal classes were the major contributors to the total phytoplankton biomass and therefore to TPP and were responsible for increases in pH.

It can be concluded from the above that there is a distinct difference between the source water of Sites 1 and 2 and each site would require different treatment processes and approaches to produce potable water. It can also be concluded that the source water of the downstream site, Site 2, appeared to be of a slightly better quality mainly due to the absence of more problematic algal species.

CHAPTER 5: THE EFFICACY OF ADVANCED TREATMENT PROCESSES

5.1. Introduction

The objective of water treatment is to accomplish the type of change in source water quality that will render the source water suitable for its intended purpose. The suitability of the treated water quality for a specific purpose is determined by the compliance of the water quality with a set of guidelines formulated for that purpose (Hendricks, 2006). The guidelines for the quality of water intended for domestic use are much more stringent than, for example, the guidelines for the quality of water intended for industrial purposes. It is to be expected that any water source intended for human consumption will require some treatment and the determination of the quality of the source water will provide an indication of the type of treatment required. The source water quality of Sites 1 and 2, Rand Water Barrage (RWB) and Midvaal Water Company (MWC), far exceeded the domestic water guidelines and requires intensive treatment.

RWB utilises granular activated carbon (GAC) and ultraviolet light (UV) disinfection and MWC utilises ozone as advanced treatment processes in addition to conventional water treatment processes (refer to Section 4.2.1). The efficacy of these advanced treatment processes was determined by ascertaining the nature of the change in the source water quality of both sites. The change in the source water quality was determined by:

- Measuring the physical and chemical water quality variables of Sites 1 and 2 at the sampling points as indicated in Table 5.2;
- The identification and enumeration of the algal compositions of Sites 1 and 2 at the sampling points as indicated in Table 5.2;
- The identification and enumeration of the invertebrate compositions of Sites 1 and 2 at the sampling points as indicated in Table 5.2.

Table 5.1 provides a list of the integers assigned to the sampling sites and points. A sampling point equates to a treatment step in the treatment process. Refer to Figures 4.3 and 4.5 for flow diagrams of the treatment processes in use at RWB and MWC.

Table 5.1: A list of integers assigned to sampling sites and sampling points (treatment steps).

Sampling site	Integer	Sampling point	Integer
RWB	**1**	Source water	1
		Secondary water after sand filtration	2
		After GAC adsorption	3
		After UV light disinfection	4
		Potable water after chlorination disinfection	5
MWC	**2**	Source water	1
		After pre-ozonation	2
		After chemical dosing (coagulation and flocculation)	3
		After Dissolved Air Flotation (DAF)	4
		After intermediate ozonation	5
		After settling (sedimentation)	6
		After sand filtration	7
		Potable water after chlorination disinfection	8

The chemical and physical water quality variables were measured at all sampling points. Total photosynthetic pigments (TPP) were included for the purposes of this study. Table 5.2 provides a list of the chemical and physical water quality variables measured as well as the sampling points where algal and invertebrate identification and enumeration were done. Refer to Sections 3.2 and 3.3 for an overview of the sampling regime that was followed and for an explanation of the material and methods that were used to determine the efficacy of the advanced water treatment processes.

Table 5.2: Sampling points where physical and chemical water quality variables were measured and where algal and invertebrate identification and enumeration were done;(✓) = done; (x) = not done.

	Site 1					Site 2							
Sampling points	**1**	**2**	**3**	**4**	**5**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Physical and chemical variables													
pH (in pH units)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Conductivity (in mS/m)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Turbidity (in NTU)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Dissolved organic carbon (DOC) (in mg/L)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Total organic carbon (TOC) (in mg/L)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Total photosynthetic pigments (TPP) (in µg/L)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Microcystin (in µg/L)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Geosmin (in ng/L)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Algal identification and enumeration (in cells/ml)													
	✓	✓	✓	✓	✓	✓	✓	x	x	✓	x	x	x
Invertebrate identification and enumeration (in cells/ml)													
	x	✓	✓	✓	✓	✓	✓	x	x	x	x	x	✓

Kruskal-Wallis ANOVA tests were conducted on all the samples to determine statistical significant differences. The mean values of the water quality variables were used to calculate the percentage change or removal between the sampling points as well as between the source and potable water order to determine the efficacy of the advanced water treatment processes. In some instances, mean values for water quality variables were below the detection limit and the percentage change or removal could not be determined. It is important to note that even though some of the mean values of the source water quality variables were within the guidelines for water intended for domestic use, individual measurements often exceeded these guidelines.

5.2. Results

5.2.1. Site 1: Rand Water Barrage (RWB)

5.2.1.1. Physical and chemical characteristics

Table 5.3 provides the mean values of the physical and chemical variables measured at the different sampling points of Site 1.

Table 5.3: The mean values of the physical and chemical variables measured at the different sampling points of Site 1. (✓) = potable water within SANS241/TWQR/RW guidelines; (x) = potable water not within SANS241/TWQR/RW guidelines. Sec = secondary water after coagulation-flocculation; GAC = granular activated carbon; UV = ultraviolet light disinfection.

	Sampling points					
Variable	**1 Source**	**2 Sec**	**3 GAC**	**4 UV**	**5 Potable**	**SANS/RW/ TWQR**
pH (in pH units)	8.04	8.31	8.01	8.03	8.07	✓
Conductivity (in mS/m)	61.533	65.834	65.731	64.970	71.430	x
Turbidity (in NTU)	23.505	1.263	0.231	0.266	0.361	✓
DOC (in mg/L)	6.842	5.469	4.483	4.446	4.064	✓
TOC (in mg/L)	7.892	6.127	4.498	4.681	4.143	✓
TPP (in µg/L)	62.196	4.608	0.383	0.345	0.086	✓
Microcystin (in µg/L)	0.523	< 0.18	< 0.18	< 0.18	< 0.18	✓
Geosmin (in ng/L)	8.423	5.076	< 5	< 5	< 5	✓

Kruskal-Wallis ANOVA multiple comparisons tests (2-tailed) were conducted on all the samples to determine whether there were statistical significant differences between the physical and chemical variables measured at the different sampling points of Site 1. These results are presented in Table 5.4.

Table 5.4: Kruskal-Wallis ANOVA results indicating statistically significant differences between the physical and chemical variables measured at the sampling points of Site 1. The level of significance for statistical analysis was set at *p* = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference.

Variable	Valid n	Sampling points					
			1	2	3	4	5
pH (in pH units)	220	1		✓	x	x	x
		2	✓		✓	✓	✓
		3	x	✓		x	x
		4	x	✓	x		x
		5	x	✓	x	x	

Variable	Valid n	Sampling points					
Conductivity (in mS/m)	220		1	2	3	4	5
		1		x	x	x	✓
		2	x		x	x	x
		3	x	x		x	x
		4	x	x	x		✓
		5	✓	x	x	✓	
Turbidity (in NTU)	220		1	2	3	4	5
		1		✓	✓	✓	✓
		2	✓		✓	✓	✓
		3	✓	✓		x	x
		4	✓	✓	x		x
		5	✓	✓	x	x	
DOC (in mg/L)	107		1	2	3	4	5
		1		✓	✓	✓	✓
		2	✓		✓	✓	✓
		3	✓	✓		x	x
		4	✓	✓	x		x
		5	✓	✓	x	x	
TOC (in mg/L)	214		1	2	3	4	5
		1		x	✓	✓	✓
		2	x		✓	✓	✓
		3	✓	✓		x	x
		4	✓	✓	x		x
		5	✓	✓	x	x	
TPP (in µg/L)	175		1	2	3	4	5
		1		✓	✓	✓	✓
		2	✓		✓	✓	✓
		3	✓	✓		x	✓
		4	✓	✓	x		✓
		5	✓	✓	✓	✓	
Microcystin (in µg/L)	180		1	2	3	4	5
		1		✓	x	✓	x
		2	✓		x	x	x
		3	x	x		x	x
		4	x	x	x		x
		5	x	x	x	x	
Geosmin (in ng/L)	205		1	2	3	4	5
		1		x	✓	✓	✓
		2	x		✓	✓	✓
		3	✓	✓		x	x
		4	✓	✓	x		x
		5	✓	✓	x	x	

Significant differences were observed between the mean pH value at sampling point 2 and the mean pH values at sampling points 1, 3, 4, and 5 (Table 5.4). The mean pH noticeably increased from 8.04 at sampling point 1 to 8.31 at sampling point 2 with a significant decrease to 8.01 at sampling point 3 (Table 5.4). The mean pH of 8.07 at sampling point 5 was higher than the mean pH of 8.04 at sampling point 1 (Table 5.3) although not significantly higher. As illustrated by Figure 5.1, the pH of Site 1 increased after the coagulation-flocculation, sedimentation and sand filtration treatment steps, decreased after GAC adsorption and increased again after UV light and chlorination disinfection. The pH of the potable water namely 8.07 (Table 5.3) was high but still within the SANS 241 Class I operational limit of between 5 and 9.7 pH units (SABS, 2011).

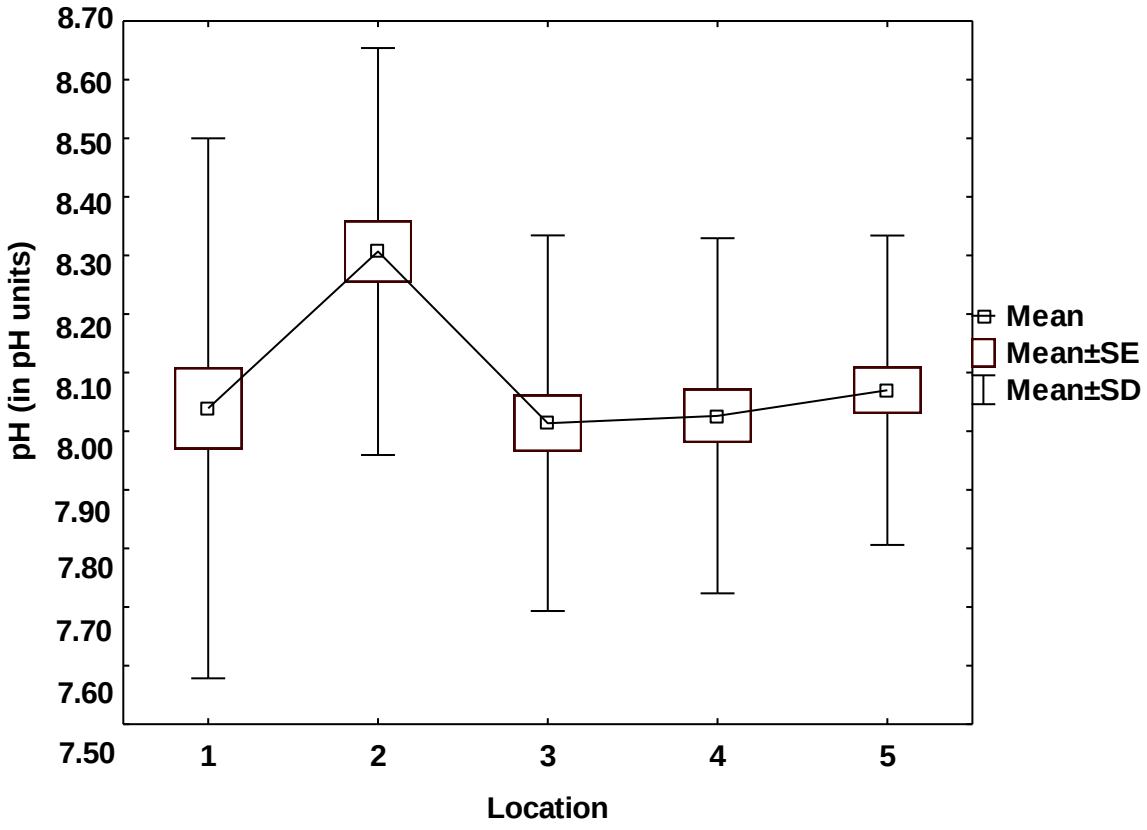


Figure 5.1: A box-and-whisker plot illustrating the change in the mean pH values (in pH units) between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

The mean conductivity value increased significantly (Table 5.4) from 61.533 mS/m at sampling point 1 to 71.430 mS/m at sampling point 5 (Table 5.3). This increase resulted in an overall increase of 16.08% in conductivity as indicated by the negative percentage change in Figure 5.2. Conductivity increased from sampling point 1 (61.533 mS/m) to sampling point 2 (65.834 mS/m) and decreased slightly at sampling points 3 and 4 (Table 5.3). A significant difference was recorded between sampling points 4 and 5 with the mean conductivity value increasing from 64.970 mS/m to 71.430 mS/m (Table 5.3). The mean conductivity value increased after the coagulation-flocculation, sedimentation and sand filtration treatment steps, decreased after GAC adsorption and increased significantly after chlorination disinfection. The mean conductivity value of 71.430 mS/m measured in the final water exceeded the TWQR of 0 to 70 mS/m (DWAF, 1996).

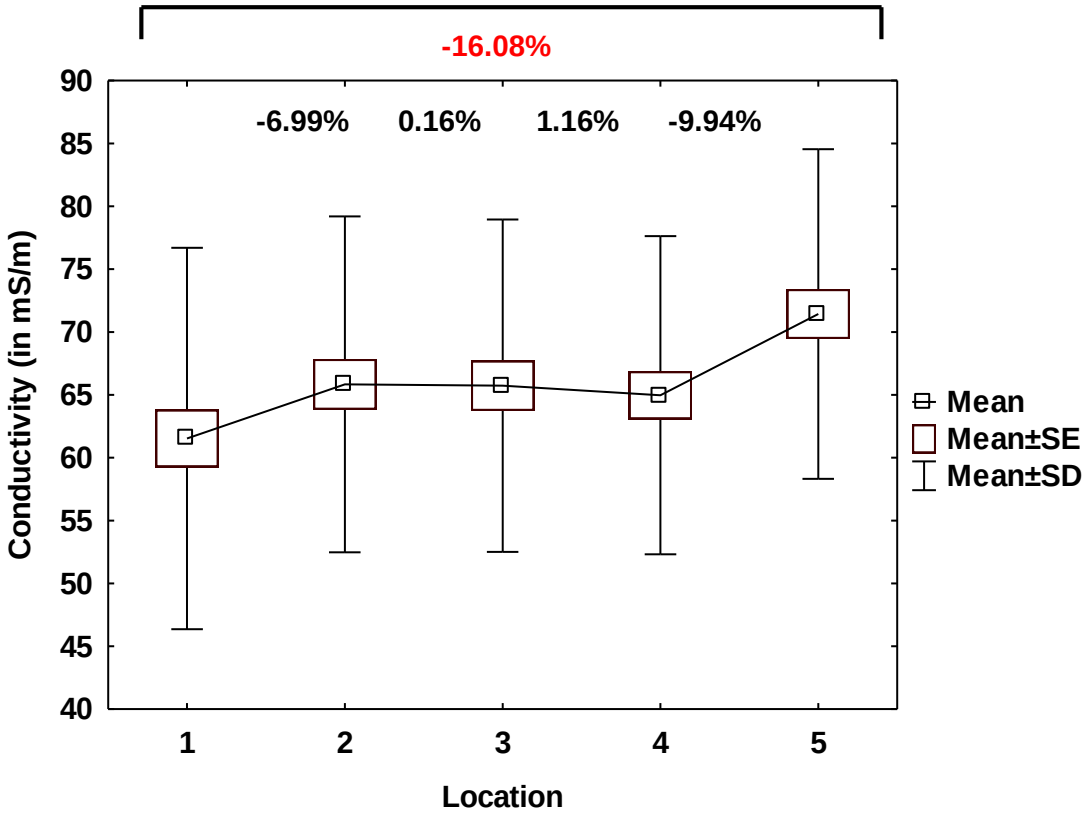


Figure 5.2: A box-and-whisker plot illustrating the change in the mean conductivity values (in mS/m) as well as the percentage change between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

Turbidity was removed successfully as indicated by the 98.46% removal (Figure 5.3). The mean turbidity value decreased significantly from 23.505 NTU at sampling point 1 to 0.361 NTU at sampling point 5 (Table 5.3). The most significant removal of turbidity occurred between sampling points 1 and 2 (Table 5.4) with a decrease from a mean of 23.505 NTU at sampling point 1 to a mean of 1.263 NTU at sampling point 2. A subsequent decrease at sampling point 3 occurred to a mean of 0.232 NTU with an increase at sampling point 5 to a mean of 0.361 NTU (Table 5.3). The mean value of 0.361 NTU at sampling point 5 was well within the SANS 241 Class I operational limit of ≤ 1 NTU (SABS, 2011). As illustrated by Figure 5.3, the most significant removal of turbidity occurred after the coagulation-flocculation, sedimentation and sand filtration treatment steps. GAC adsorption removed the remainder to an acceptable level. Chlorination disinfection caused an increase in turbidity but not above the SANS 241 limit.

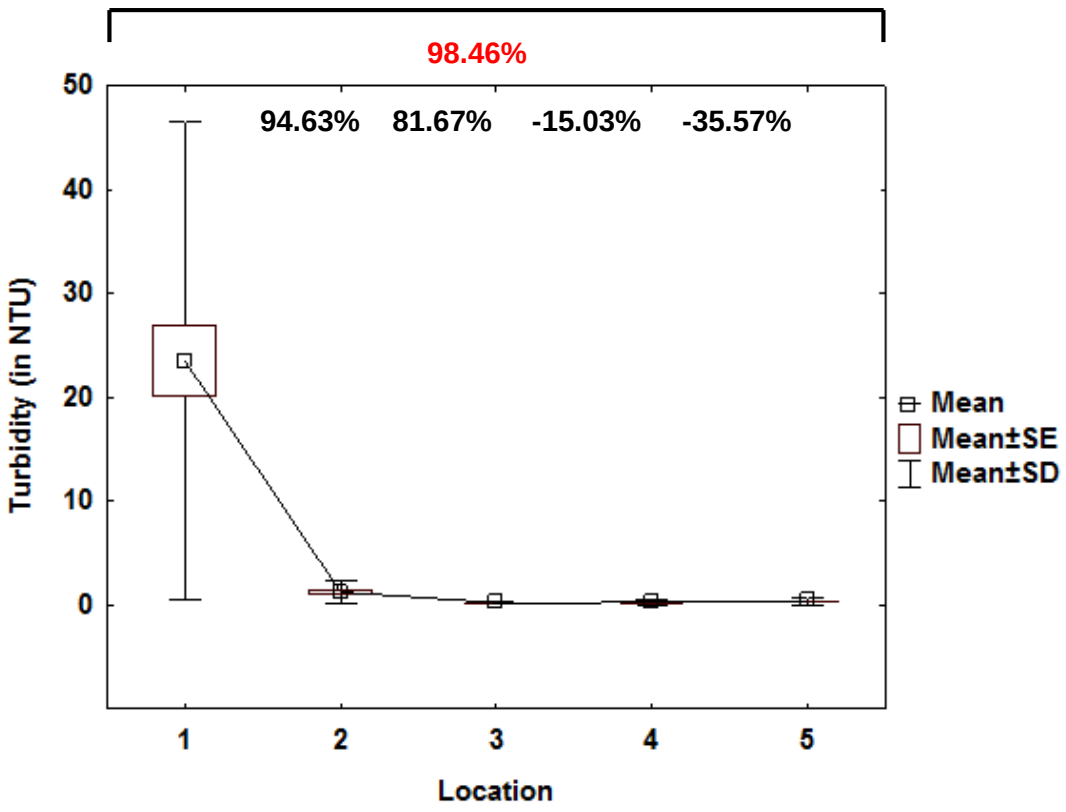


Figure 5.3: A box-and-whisker plot illustrating the change in the mean turbidity values (in NTU) as well as the percentage removal between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

Similar DOC and TOC removal trends were observed with a 40.81% removal of DOC and a 47.51% removal of TOC (Figures 5.4 & 5.5). Significant differences were recorded between sampling points 1 and 5 for both variables (Table 5.4). The mean DOC value of 6.842 mg/L at sampling point 1 decreased to 4.064 mg/L at sampling point 5 and the mean TOC value of 7.892 mg/L (sampling point 1) decreased to 4.143 mg/L (sampling point 5). The mean DOC value decreased significantly between sampling points 1, 2 and 3 from 6.842 mg/L to 5.459 mg/L to 4.483 mg/L The mean TOC value showed a significant decrease from 6.127 mg/L at sampling point 2 to 4.498 mg/L at sampling point 3. (Tables 5.3 & 5.4). A slight increase in TOC at sampling point 4 was recorded. As illustrated by Figures 5.4 and 5.5, the removal of DOC and TOC was the most effective after the coagulation-flocculation, sedimentation, sand filtration and GAC adsorption treatment steps. GAC adsorption with a 26.58% removal proved to be very effective in the removal of TOC. TOC increased slightly but not significantly after UV light disinfection and decreased again after chlorination disinfection. The DOC and TOC means recorded in the potable water were within the SANS 241 Class I operational limit of < 10 mg/L (SABS, 2011).

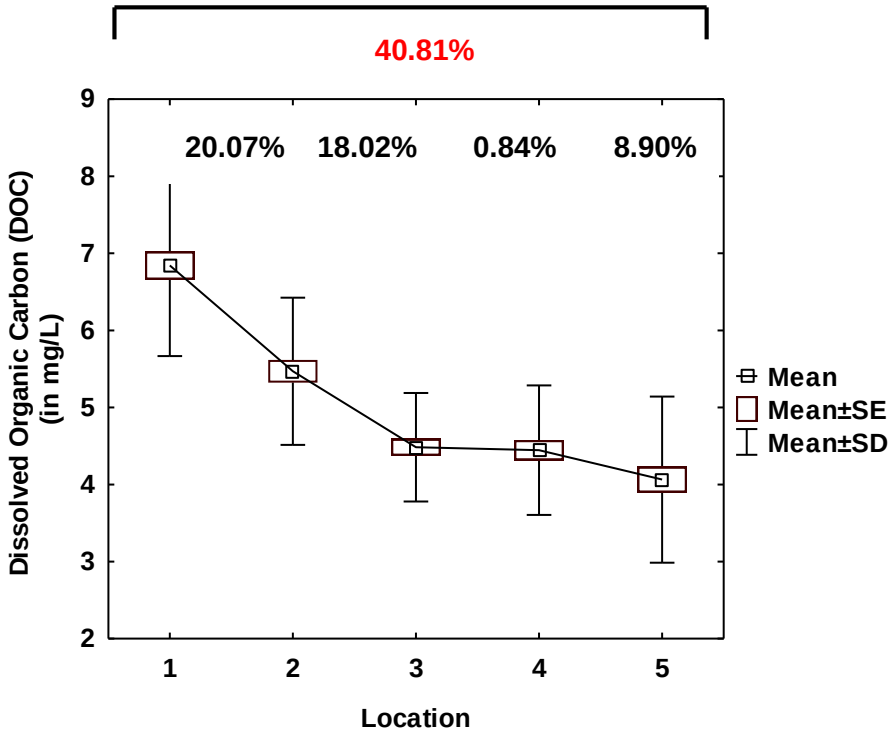


Figure 5.4: A box-and-whisker plot illustrating the change in the mean DOC values (in mg/L) as well as the percentage removal between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

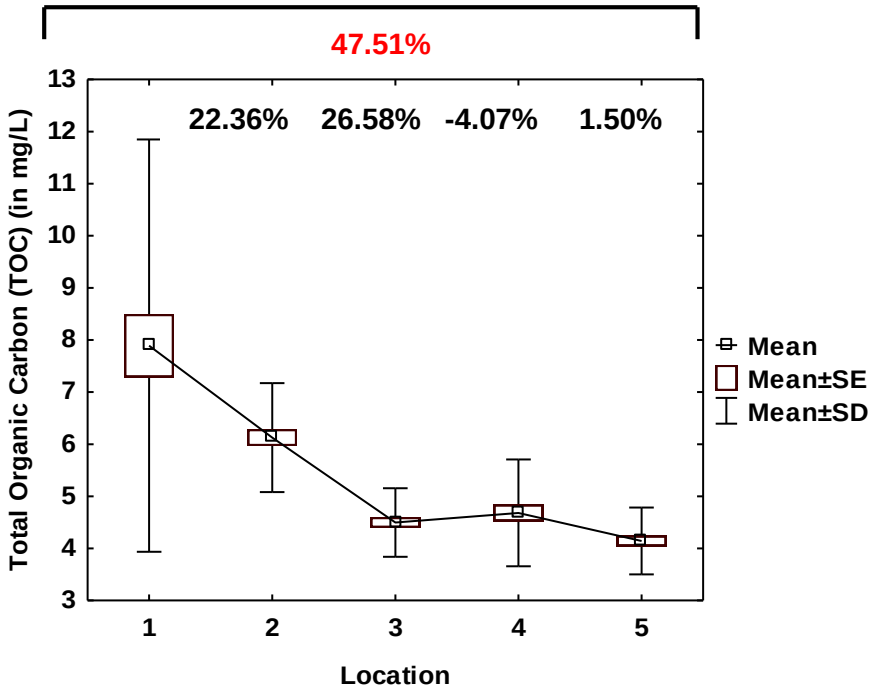


Figure 5.5: A box-and-whisker plot illustrating the change in the mean TOC values (in mg/L) as well as the percentage removal between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

An almost complete removal, 99.86%, of TPP from the source water was found (Figure 5.6) with a significant decrease in the mean TPP value from 62.196 µg/L at sampling point 1 to 0.086 µg/L at sampling point 5. Significant decreases in TPP were observed between all the sampling points except between sampling points 3 and 4 (Table 5.4). The mean TPP value decreased from 62.196 µg/L at sampling point 1 to 4.608 µg/L at sampling point 2 to 0.383 µg/L at sampling point 3. A subsequent decrease from 0.345 µg/L at sampling point 4 to 0.086 µg/L at sampling point 5 occurred (Table 5.3). The coagulation-flocculation, sedimentation and sand filtration treatment steps effectively removed the majority of TPP as illustrated by Figure 5.6. GAC adsorption further reduced TPP to a mean of 0.383 µg/L which is lower than the recommended limit for potable water as set by Rand Water, namely 1 µg/L (Swanepoel *et al*., 2008a). After the chlorination disinfection treatment step the TPP value was negligible.

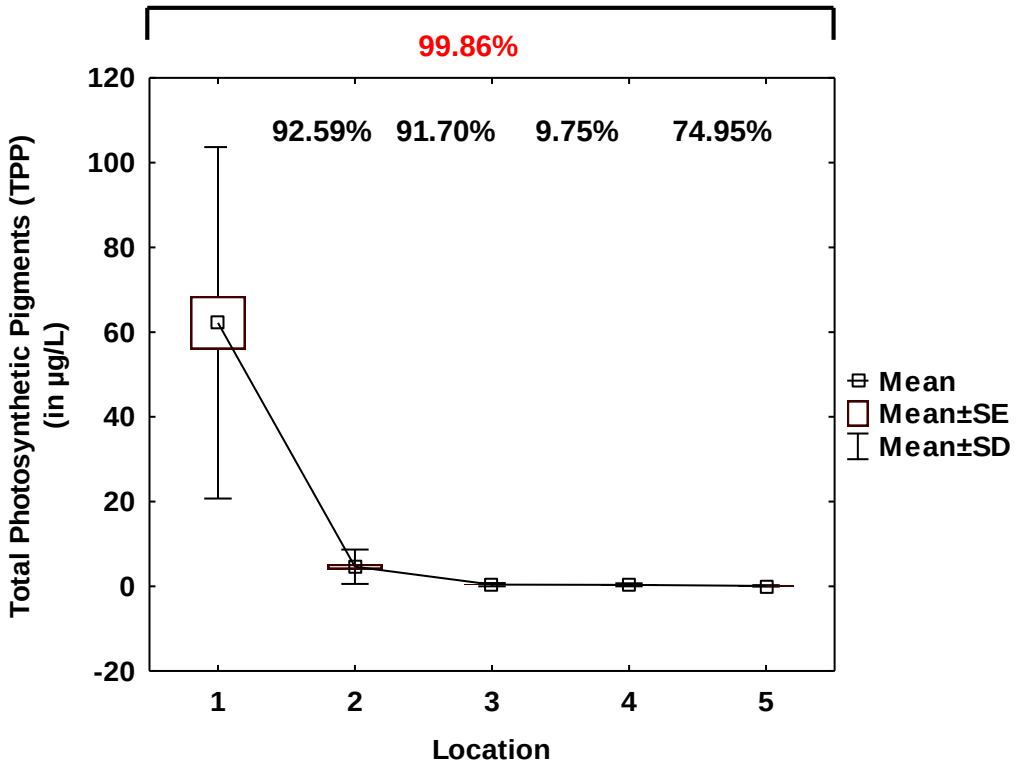


Figure 5.6: A box-and-whisker plot illustrating the change in the mean TPP values (in µg/L) as well as the percentage removal between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

The mean microcystin value decreased significantly from 0.523 µg/L at sampling 1 to < 0.18 µg/L at sampling point 5 (Table 5.4). The coagulation-flocculation, sedimentation and sand filtration treatment steps achieved a complete removal of microcystin from the source water of Site 1 as illustrated by Figure 5.7. Mean values below the detection limit of the method namely 0.18 µg/L were halved for graphical representation purposes. The mean microcystin concentration of < 0.18 µg/L in the potable water was well below the SANS 241 limit of ≤ 1 µg/L for microcystin (SABS, 2011).

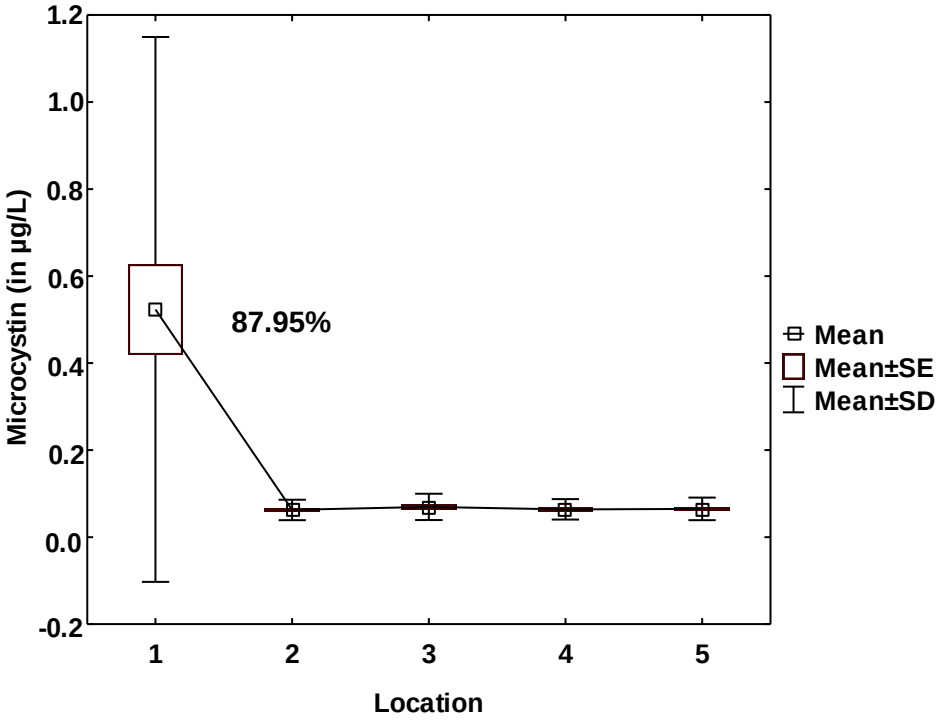


Figure 5.7: A box-and-whisker plot illustrating the change in the mean microcystin values (in µg/L) between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

A significant difference was recorded between the mean geosmin values of sampling points 1 and 5 as the geosmin value decreased from 8.423 ng/L at sampling point 1 to < 5 ng/L at sampling point 5 (Table 5.3) indicating that geosmin was completely removed from the source water of Site 1. The mean geosmin value decreased from 8.423 ng/L at sampling point 1 to 5.076 ng/L at sampling point 2 with a further significant decrease to < 5 ng/L at sampling point 3 (Tables 5.3 & 5.4). Mean values below the detection limit of the method, namely 5 ng/L, were halved for graphical representation purposes. A considerable variation from the mean geosmin value at sampling point 4 can be observed in Figure 5.8 due to a maximum of 66.59 ng/L that was reached after UV light disinfection. The significant decrease in geosmin can be attributed to the GAC adsorption treatment step after the initial removal of geosmin by the coagulation-flocculation, sedimentation and sand filtration treatment steps (Figure 5.8). The mean geosmin value of < 5 ng/L measured in the potable water was well below the limit of 30 ng/L for potable water as specified by Rand Water (Swanepoel *et al.*, 2008a).

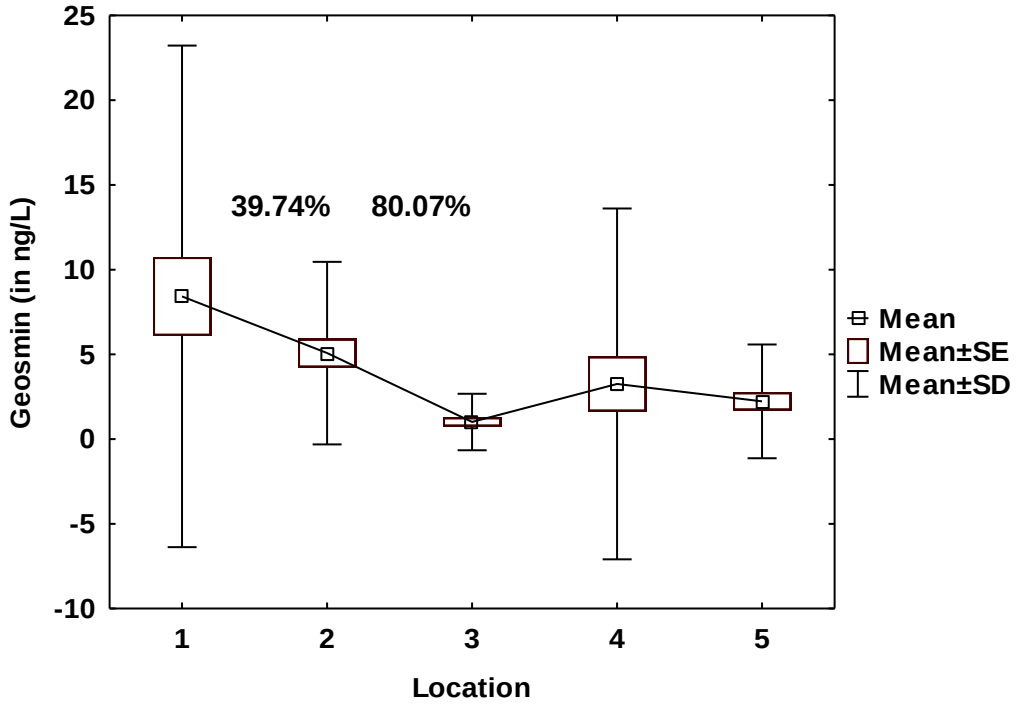


Figure 5.8: A box-and-whisker plot illustrating the change in the mean geosmin values (in ng/L) between the sampling points of Site 1. SE = Standard Error; SD = Standard Deviation.

5.2.1.2. Algal characteristics

Table 5.5 provides the mean concentration of the algal classes identified and enumerated at the different sampling points of Site 1.

Table 5.5: The mean concentrations of the algal classes identified and enumerated at the different sampling points of Site 1. Sec = secondary water after coagulation-flocculation; GAC = granular activated carbon; UV = ultraviolet light disinfection.

		Sampling points				
Algal class	**Unit**	**1 Source**	**2 Sec**	**3 GAC**	**4 UV**	**5 Potable**
Bacillariophyceae (BACIL)	**cells/ml**	3285	8	1	1	2
Chlorophyceae (CHLORO)	**cells/ml**	1192	8	4	3	1
Cryptophyceae (CRYPTO)	**cells/ml**	419	1	< 1	< 1	< 1
Cyanophyceae (CYANO)	**cells/ml**	2949	194	13	35	30
Dinophyceae (DINO)	**cells/ml**	108	4	< 1	< 1	< 1
Euglenophyceae (EUGLENO)	**cells/ml**	< 36	1	18	40	34
Total cells	**cells/ml**	7975	216	37	79	67

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on all the algal samples to determine whether there were statistical significant differences between the algal classes at the different sampling locations of Site 1. These results are presented in Table 5.6.

Table 5.6: Kruskal-Wallis ANOVA results indicating statistically significant differences between the algal class concentrations at the sampling points of Site 1. The level of significance for statistical analysis was set at *p* = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference. Valid n = 221.

Algal class (in cells/ml)	**Sampling points**					
Bacillariophyceae		**1**	**2**	**3**	**4**	**5**
	1		✓	✓	✓	✓
	2	✓		✓	✓	✓
	3	✓	✓		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	
Chlorophyceae		**1**	**2**	**3**	**4**	**5**
	1		x	✓	✓	✓
	2	x		x	✓	✓
	3	✓	x		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	

Algal class (in cells/ml)	Sampling points					
Cryptophyceae		1	2	3	4	5
	1		✓	✓	✓	✓
	2	✓		x	x	x
	3	✓	x		x	x
	4	✓	x	x		x
	5	✓	x	x	x	
Cyanophyceae		1	2	3	4	5
	1		x	✓	✓	✓
	2	x		x	x	x
	3	✓	x		x	x
	4	✓	x	x		x
	5	✓	x	x	x	
Dinophyceae		1	2	3	4	5
	1		x	✓	✓	✓
	2	x		✓	✓	✓
	3	✓	✓		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	
Euglenophyceae		1	2	3	4	5
	1		x	✓	✓	✓
	2	x		✓	✓	✓
	3	✓	✓		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	
Total cells		1	2	3	4	5
	1		✓	✓	✓	✓
	2	✓		x	✓	✓
	3	✓	x		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	

Significant differences were found for all the algal classes and for the total phytoplankton biomass between sampling points 1 and 5 (Table 5.6). Bacillariophyceae decreased from 3285 cells/ml to 2 cells/ml, Chlorophyceae from 1192 cells/ml to 1 cell/ml, Cryptophyceae from 419 cells/ml to < 1 cells/ml, Cyanophyceae from 2949 cells/ml to 30 cells/ml and Dinophyceae from 108 cells/ml to < 1 cell/ml. The total phytoplankton biomass decreased from 7975 cells/ml to 67 cells/ml (Table 5.5). Euglenophyceae increased from < 36 cells/ml to 34 cells/ml.

As illustrated by Figure 5.9, the changes in the mean algal class concentrations and total phytoplankton biomass displayed a similar trend except for Euglenophyceae. All the mean algal class concentrations decreased from sampling points 1 to 3 except for Euglenophyceae that showed an increase (Table 5.5). Significant decreases were recorded for Bacillariophyceae, Cryptophyceae and the total phytoplankton biomass between sampling points 1, 2 and 3. Cyanophyceae, Chlorophyceae and Dinophyceae significantly decreased between sampling points 1 and 3 (Table 5.6). Although not significant, increases in the mean concentrations were observed for Bacillariophyceae, Cryptophyceae, Cyanophyceae, Dinophyceae, Euglenophyceae and the total phytoplankton biomass at sampling point 4 (Table 5.5). The Bacillariophyceae, Cryptophyceae and Dinophyceae values showed a slight subsequent increase at sampling point 5. Cyanophyceae,

Euglenophyceae and the total phytoplankton biomass concentrations subsequently decreased slightly at sampling point 5. The Chlorophyceae concentration decreased at every sampling point (Table 5.5).

The total percentages' removal, ranging from 98.56% to 99.99%, as well as the mean algal class concentrations at sampling point 5 (Table 5.5) indicated that the majority of the phytoplankton biomass was removed effectively by the water treatment processes. It was evident that there was an increase in the Bacillariophyceae, Cryptophyceae, Cyanophyceae, Dinophyceae and Euglenophyceae concentrations as well as in the total phytoplankton biomass between sampling points 3 and 4. It was found that the percentages' removal of the phytoplankton recorded at sampling point 3 were more or similar than the percentages' removal recorded at sampling point 5. This indicated that the majority of the phytoplankton was effectively reduced by the coagulation-flocculation, sedimentation, sand filtration and GAC adsorption treatment steps. Significant decreases indicated that GAC filtration was effective in the removal of Cyanophyceae, Chlorophyceae and Dinophyceae. UV light disinfection appeared to be less effective even disadvantageous to a certain extent as the majority of the mean phytoplankton concentrations increased after this treatment step.

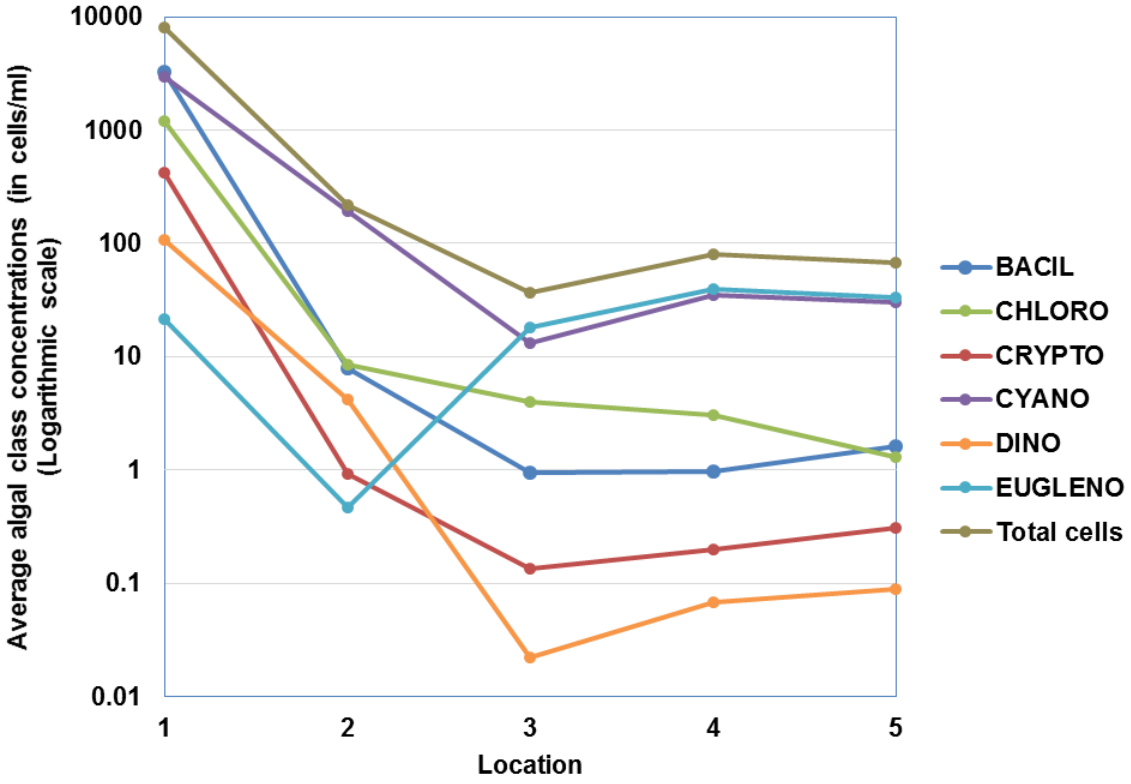


Figure 5.9: A line chart illustrating the change in the mean algal class concentration (in cells/ml) between the sampling points of Site 1.

Problematic algal species with the potential to negatively impact on water treatment processes were identified in the source water of Site 1 (Refer to Table 4.8). Table 5.7 provides the mean concentrations of the problematic algal species identified and enumerated at the different sampling points of Site 1.

Table 5.7: The mean concentrations of the problematic algal species identified and enumerated at the different sampling points of Site 1. Sec = secondary water after coagulation-flocculation; GAC = granular activated carbon; UV = ultraviolet light disinfection.

		Sampling points				
Species	**Unit**	**1 Source**	**2 Sec**	**3 GAC**	**4 UV**	**5 Potable**
Anabaena sp.	**cells/ml**	569	16	1	1	2
Aulacoseira sp.	**cells/ml**	1513	4	< 1	< 1	< 1
Other centric diatoms	**cells/ml**	1177	3	< 1	< 1	1
Ceratium sp.	**cells/ml**	100	4	0	< 1	< 1
Cryptomonas sp.	**cells/ml**	419	1	< 1	< 1	< 1
Microcystis sp.	**cells/ml**	2153	63	9	28	24
Oscillatoria sp.	**cells/ml**	226	115	3	7	4
Pandorina sp.	**cells/ml**	182	< 1	< 1	< 1	< 1

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on the all the samples to determine statistical significant differences between the problematic algal species at the different sampling points of Site 1. These results are presented in Table 5.8.

Table 5.8: Kruskal-Wallis ANOVA results indicating statistically significant differences between the problematic algal species concentrations at the different sampling points of Site 1. The level of significance for statistical analysis was set at *p* = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference. Valid n = 220.

Algal species (in cells/ml)	Sampling points					
***Anabaena* sp.**		1	2	3	4	5
	1		x	x	x	x
	2	x		x	x	x
	3	x	x		x	x
	4	x	x	x		x
	5	x	x	x	x	
***Aulacoseira* sp.**		1	2	3	4	5
	1		✓	✓	✓	✓
	2	✓		✓	x	✓
	3	✓	✓		x	x
	4	✓	x	x		x
	5	✓	✓	x	x	
Other centric diatoms		1	2	3	4	5
	1		✓	✓	✓	✓
	2	✓		x	x	x
	3	✓	x		x	x
	4	✓	x	x		x
	5	✓	x	x	x	
***Ceratium* sp.**		1	2	3	4	5
	1		x	✓	✓	✓
	2	x		✓	✓	✓
	3	✓	✓		x	x
	4	✓	✓	x		x
	5	✓	✓	x	x	
***Cryptomonas* sp.**		1	2	3	4	5
	1		✓	✓	✓	✓
	2	✓		x	x	x
	3	✓	x		x	x
	4	✓	x	x		x
	5	✓	x	x	x	
***Microcystis* sp.**		1	2	3	4	5
	1		x	x	x	x
	2	x		x	x	x
	3	x	x		x	x
	4	x	x	x		x
	5	x	x	x	x	
***Oscillatoria* sp.**		1	2	3	4	5
	1		x	x	x	x
	2	x		x	x	x
	3	x	x		x	x
	4	x	x	x		x
	5	x	x	x	x	
***Pandorina* sp.**		1	2	3	4	5
	1		x	x	x	x
	2	x		x	x	x
	3	x	x		x	x
	4	x	x	x		x
	5	x	x	x	x	

Significant differences were observed between sampling points 1 and 5 for *Aulacoseira* sp., other centric diatoms, *Ceratium* sp. and *Cryptomonas* sp. (Table 5.8). *Aulacoseira* sp. decreased from

1513 cells/ml to < 1 cell/ml, other centric diatoms from 1177 cells/ml to 1 cell/ml, *Ceratium* sp. from 100 cells/ml to < 1 cell/ml, and *Cryptomonas* sp. from 419 cells/ml to < 1 cell/ml (Table 5.7). Decreases, although not significantly, in the mean concentrations of *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp. and *Pandorina* sp. occurred between sampling points 1 and 5. *Anabaena* sp. decreased from 569 cells/ml to 2 cells/ml, *Microcystis* sp. from 2153 cells/ml to 24 cells/ml, *Oscillatoria* sp. from 227 cells/ml to 4 cells/ml and *Pandorina* sp. from 182 cells/ml to < 1 cell/ml.

As illustrated by Figure 5.10, the changes in the mean problematic algal species concentration displayed a similar trend. This trend was also observed with the mean algal class concentration (Figure 5.5). All the concentrations decreased from sampling points 1 to 3 (Table 5.7). Significant decreases were recorded for *Aulacoseira* sp., other centric diatoms and *Cryptomonas* sp. between sampling points 1, 2 and 3. *Ceratium* sp. decreased significantly between sampling points 2 and 3 (Table 5.8) and was removed completely. Although not significantly, *Aulacoseira* sp., *Ceratium* sp., *Cryptomonas* sp., *Microcystis* sp. and *Oscillatoria* sp. increased at sampling point 4. *Ceratium* sp. and *Cryptomonas* sp. subsequently increased slightly and *Aulacoseira* sp., *Microcystis* sp. and *Oscillatoria* sp. decreased slightly at sampling point 5. *Anabaena* sp. and other centric diatoms decreased slightly at sampling point 4 and showed an increase at sampling point 5. *Pandorina* sp. decreased at every sampling point (Table 5.7).

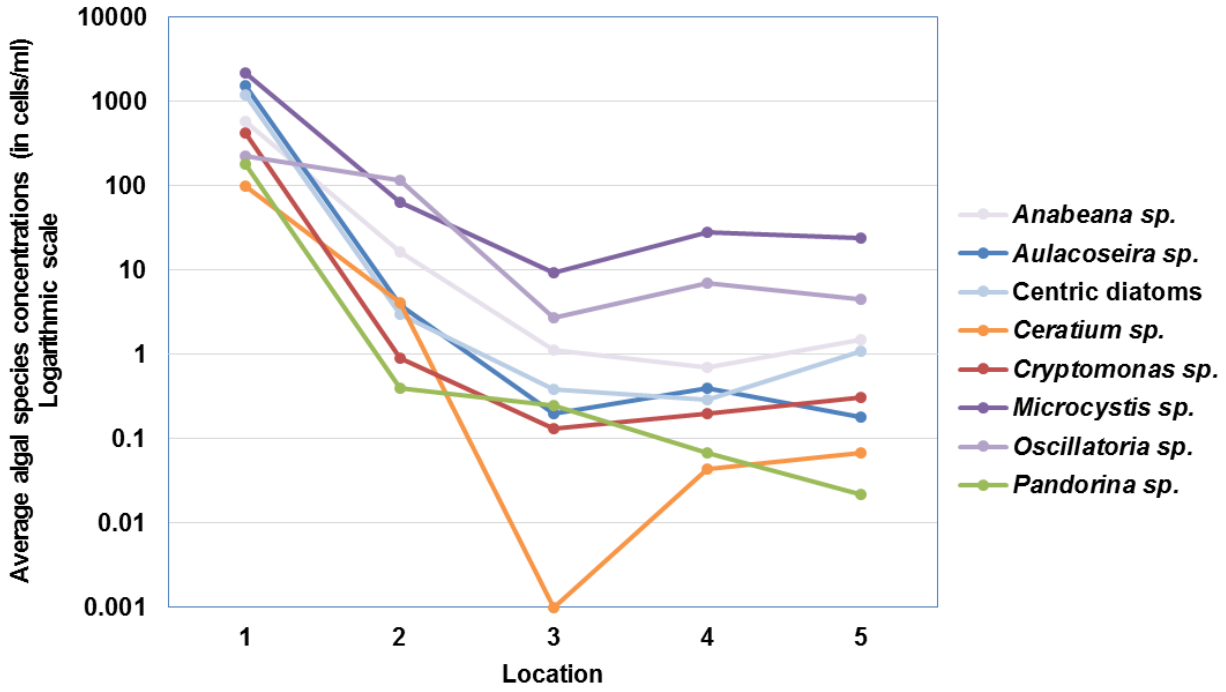


Figure 5.10: A line chart illustrating the change in the mean problematic algal species concentrations (in cells/ml) between the different sampling points of Site 1.

The total percentages' removal ranging from 98.04% to 99.99% as well as the mean problematic algal species concentrations at sampling point 5 (Table 5.7) indicated that the problematic algal species were effectively removed by the water treatment processes. It is clear though that all the problematic algal species except *Pandorina* sp. increased between sampling points 3 and 5. A noticeably lower percentage removal, 49.43%, was recorded for *Oscillatoria* sp. between sampling points 1 and 2 in comparison with the other problematic species. It was found that the percentages' removal of the algal species recorded at sampling point 3 were slightly higher or similar than the percentages' removal recorded at sampling point 5 indicating that the problematic algal species were effectively removed by the coagulation-flocculation, sedimentation, sand filtration and GAC filtration treatment steps. GAC filtration appeared to be effective in the removal of *Ceratium* sp. The algal concentrations of the problematic species appeared to increase after UV light and chlorination disinfection.

5.2.1.3. Invertebrate characteristics

Table 5.9 provides the mean concentration of the invertebrate groups identified and enumerated at the different sampling points of Site 1.

Table 5.9: The mean concentrations of the invertebrate groups identified and enumerated at the different sampling points of Site 1. Sec = secondary water after coagulation-flocculation; GAC = granular activated carbon; UV = ultraviolet light disinfection.

		Sampling points			
Invertebrate group	**Unit**	**2 Sec**	**3 GAC**	**4 UV**	**5 Potable**
Hydracarina (Acari)	**org/m^3**	< 1	< 1	< 1	< 1
Cladocera (Clado)	**org/m^3**	1	< 1	< 1	< 1
Copepoda (Copep)	**org/m^3**	146	89	129	31
Diptera (Dipt)	**org/m^3**	1	1	1	1
Ephemeroptera (Ephem)	**org/m^3**	< 1	< 1	< 1	< 1
Nematoda (Nemat)	**org/m^3**	1	6	5	7
Ostracoda (Ostra)	**org/m^3**	2	3	2	1
Rotatoria (Rotat)	**org/m^3**	1081	358	146	47
Unknown invertebrates (Uninv)	**org/m^3**	< 1	< 1	< 1	< 1
Other invertebrates (Other)	**org/m^3**	1236	456	282	85
Total invertebrates (Total inv)	**org/m^3**	2469	913	539	171

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on the all the samples to determine statistical significant differences between the invertebrate groups at the different sampling points of Site 1. These results are presented in Table 5.10.

Table 5.10: Kruskal-Wallis ANOVA results indicating statistically significant differences between the invertebrate concentrations at the different sampling points of Site 1. The level of significance for statistical analysis was set at p = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference. Valid n = 87.

Invertebrate group (in org/m^3)	Sampling points				
Hydracarina		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Cladocera		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Copepoda		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Diptera		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Ephemeroptera		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Nematoda		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Ostracoda		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Rotatoria		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Unknown invertebrates		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	

Invertebrate group (in org/m^3)	Sampling points				
Other invertebrates		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	
Total invertebrates		2	3	4	5
	2		x	x	x
	3	x		x	x
	4	x	x		x
	5	x	x	x	

The mean total invertebrate concentrations at sampling points 2, 3 and 4 of 2469 org/m^3, 913 org/m^3 and 539 org/m^3 respectively exceeded the crisis limit of 250 org/m^3 for invertebrates in potable water as set by Rand Water (Shaddock, 2006). The mean total concentration of 171 org/m^3 at sampling point 5 exceeded the maximum permissible limit of 100 org/m^3. No significant differences were observed between the mean concentrations of the invertebrate groups enumerated at the different sampling points of Site 1 (Table 5.10). The mean concentrations of Hydracarina, Ephemeroptera and Unknown invertebrates were insignificant as these concentrations were below 1 org/m^3. Cladocera was completely removed from the source water.

Nematoda increased from 1 org/m^3 to 6 org/m^3 between sampling points 2 and 3 with a subsequent increase to 7 org/m^3 at sampling point 5 (Table 5.9). The changes in the mean counts of Rotatoria, Other groups of invertebrates and Total invertebrates displayed a similar trend with a decrease at each sampling point. Copepoda increased at sampling point 4 and the mean concentration of Diptera remained unchanged at 1 org/m^3 (Figure 5.11).

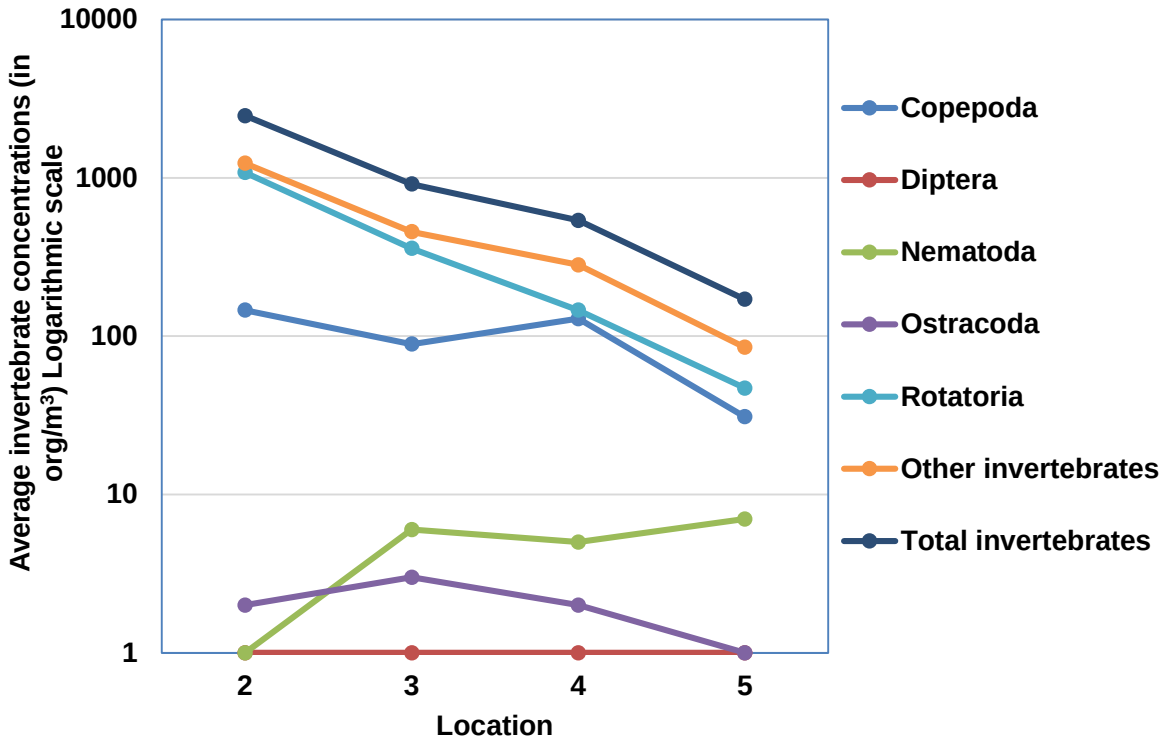


Figure 5.11: A line chart illustrating the change in the mean invertebrate concentrations (in org/m^3) between the different sampling points of Site 1.

The percentages' removal ranging from 93% to 98% and the concentrations at sampling point 5 (Table 5.10) indicated the effective removal of Rotatoria, Other invertebrate groups and Total invertebrates. Lower percentages' removal were recorded for Copepoda and Ostracoda. Unknown invertebrates only reflected a 37.33% decrease. The removal of Nematoda and Diptera was ineffective. Nematoda increased in the potable water. All the invertebrate groups except Nematoda showed a decrease between sampling point 4 and 5 (Figure 5.11 and Table 5.10) suggesting that chlorination disinfection was the most effective treatment step.

5.2.2. Site 2: Midvaal Water Company (MWC)

5.2.2.1. Physical and chemical characteristics

Table 5.11 provides the mean values of the physical and chemical variables measured at the different sampling points of Site 2.

Table 5.11: The mean values of the physical and chemical variables measured at the different sampling points of Site 2. (✓) = potable water within SANS241/TWQR/RW guidelines; (x) = potable water not within SANS241/TWQR/RW guidelines. C/F = coagulation/flocculation; DAF = Dissolved Air Flotation; Sedi = sedimentation; S/F = sand filtration.

	Sampling points								
Variable	**1 Source**	**2 Pre-ozone**	**3 C/F**	**4 DAF**	**5 Inter-ozone**	**6 Sedi**	**7 S/F**	**8 Potable**	**SANS/ RW/ TWQR**
pH (in pH units)	8.73	8.71	9.17	9.06	8.90	8.74	8.35	8.27	✓
Conductivity (in mS/m)	55.700	55.464	58.060	57.672	57.798	57.980	58.052	57.716	✓
Turbidity (in NTU)	37.731	37.229	37.668	14.020	16.546	3.104	0.554	0.479	✓
DOC (in mg/L)	6.331	6.340	5.757	5.534	5.607	5.245	4.855	4.600	✓
TOC (in mg/L)	7.028	7.385	6.476	6.151	6.290	5.739	5.316	5.122	✓
TPP (in µg/L)	78.531	52.588	54.133	17.582	16.531	6.485	0.271	0.237	✓
Microcystin (in µg/L)	< 0.18	< 0.18	< 0.18	< 0.18	< 0.18	< 0.18	< 0.18	< 0.18	✓
Geosmin (in ng/L)	< 5	< 5	< 5	< 5	< 5	< 5	< 5	< 5	✓

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on all the samples to determine whether there were statistical significant differences between the physical and chemical variables measured at the different sampling points of Site 2. These results are presented in Table 5.12.

Table 5.12: Kruskal-Wallis ANOVA results indicating statistically significant differences between the physical and chemical variables measured at the different sampling points of Site 2. The level of significance for statistical analysis was set at *p* = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference.

Variable	Valid n	Sampling points								
pH (in pH units)	764		1	2	3	4	5	6	7	8
		1		x	✓	✓	x	x	✓	✓
		2	x		✓	✓	x	x	✓	✓
		3	✓	✓		x	✓	✓	✓	✓
		4	✓	✓	x		x	✓	✓	✓
		5	x	x	✓	x		x	✓	✓
		6	x	x	✓	x	x		✓	✓
		7	✓	✓	✓	✓	✓	✓		x
		8	✓	✓	✓	✓	✓	✓	x	
Conductivity (in mS/m)	748		1	2	3	4	5	6	7	8
		1		x	x	x	x	x	x	x
		2	x		x	x	x	x	x	x
		3	x	x		x	x	x	x	x
		4	x	x	x		x	x	x	x
		5	x	x	x	x		x	x	x
		6	x	x	x	x	x		x	x
		7	x	x	x	x	x	x		x
		8	x	x	x	x	x	x	x	
Turbidity (in NTU)	782		1	2	3	4	5	6	7	8
		1		x	x	✓	✓	✓	✓	✓
		2	x		x	✓	✓	✓	✓	✓
		3	x	x		✓	✓	✓	✓	✓
		4	✓	✓	✓		x	✓	✓	✓
		5	✓	✓	✓	x		✓	✓	✓
		6	✓	✓	✓	✓	✓		✓	✓
		7	✓	✓	✓	✓	✓	✓		x
		8	✓	✓	✓	✓	✓	✓	x	
DOC (in mg/L)	703		1	2	3	4	5	6	7	8
		1		x	✓	✓	✓	✓	✓	✓
		2	✓		✓	✓	✓	✓	✓	✓
		3	✓	✓		x	x	x	✓	✓
		4	✓	✓	x		x	x	✓	✓
		5	✓	✓	x	x		x	✓	✓
		6	✓	✓	x	x	x		x	✓
		7	✓	✓	✓	✓	✓	x		x
		8	✓	✓	✓	✓	✓	✓	x	
TOC (in mg/L)	706		1	2	3	4	5	6	7	8
		1		x	✓	✓	✓	✓	✓	✓
		2	x		x	✓	✓	✓	✓	✓
		3	✓	x		x	x	✓	✓	✓
		4	✓	✓	x		x	x	✓	✓
		5	✓	✓	x	x		x	✓	✓
		6	✓	✓	✓	x	x		x	✓
		7	✓	✓	✓	✓	✓	x		x
		8	✓	✓	✓	✓	✓	✓	x	
TPP (in µg/L)	627		1	2	3	4	5	6	7	8
		1		x	x	✓	✓	✓	✓	✓
		2	x		x	✓	✓	✓	✓	✓
		3	x	x		✓	✓	✓	✓	✓
		4	✓	✓	✓		x	✓	✓	✓
		5	✓	✓	✓	x		x	✓	✓
		6	✓	✓	✓	✓	x		✓	✓
		7	✓	✓	✓	✓	✓	✓		x
		8	✓	✓	✓	✓	✓	✓	x	

Variable	Valid n	Sampling points								
Microcystin (in µg/L)	144		1	2	3	4	5	6	7	8
		1		x	x	x	x	x	x	x
		2	x		x	x	x	x	x	x
		3	x	x		x	x	x	x	x
		4	x	x	x		x	x	x	x
		5	x	x	x	x		x	x	x
		6	x	x	x	x	x		x	x
		7	x	x	x	x	x	x		x
		8	x	x	x	x	x	x	x	
Geosmin (in ng/L)	142		1	2	3	4	5	6	7	8
		1		x	x	x	x	x	x	x
		2	x		x	x	x	x	x	x
		3	x	x		x	x	x	x	x
		4	x	x	x		x	x	x	x
		5	x	x	x	x		x	x	x
		6	x	x	x	x	x		x	x
		7	x	x	x	x	x	x		x
		8	x	x	x	x	x	x	x	

A significant difference was observed between the mean pH values of sampling points 1 and 8 (Table 5.12) with a decrease in pH from 8.73 to 8.27 (Table 5.11). The pH increased significantly from 8.71 at sampling point 2 to 9.17 at sampling point 3. The increase in pH was followed by significant decreases between sampling points 4 and 7 and 8 (Table 5.12). As illustrated by Figure 5.12, the pH of Site 2 increased noticeably after the coagulation-flocculation treatment step and subsequently decreased after each treatment step. The mean pH of the potable water namely 8.27 (Table 5.11) was high but still within the SANS 241 operational limit of between 5 and 9.7 pH units (SABS, 2011).

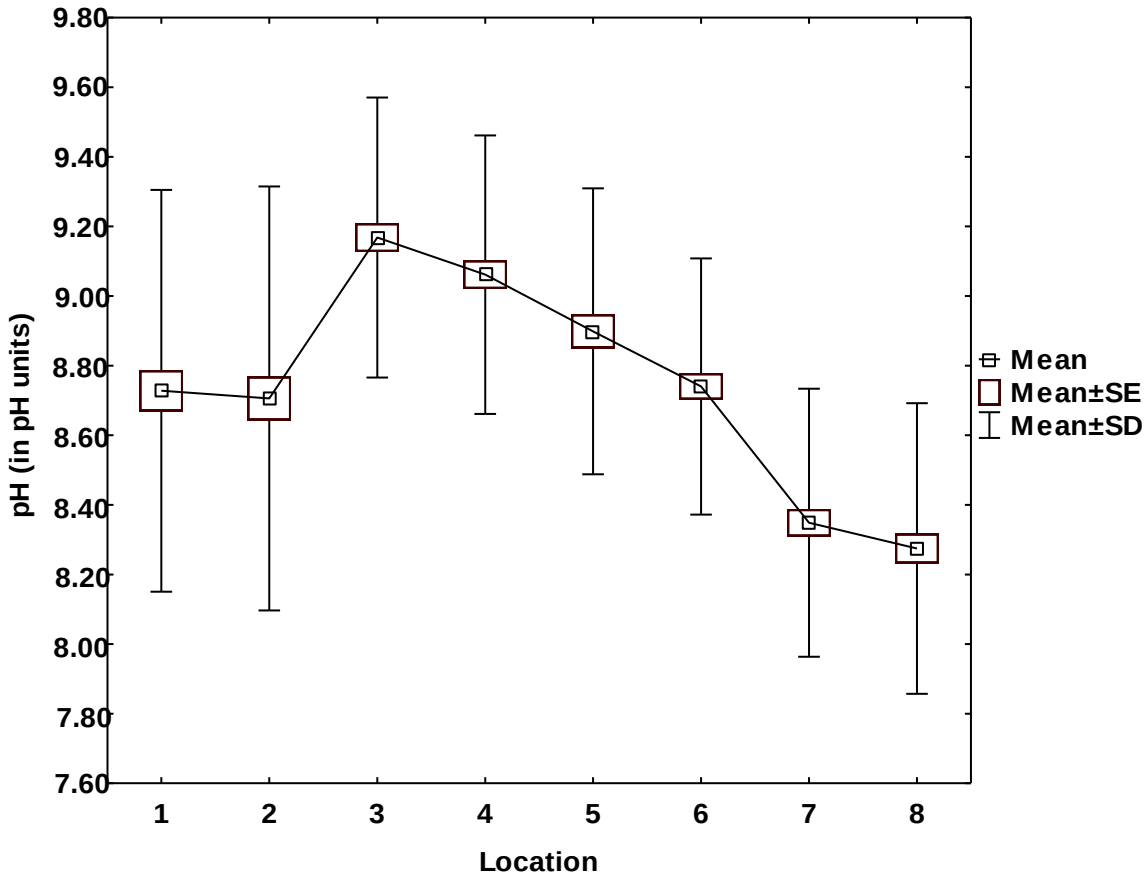


Figure 5.12: A box-and-whisker plot illustrating the change in the mean pH values (in pH units) between the sampling points of Site 2. SE = Standard Error; SD = Standard Deviation.

No significant changes were observed in the mean conductivity values of Site 2 (Table 5.12). The mean conductivity increased from 55.7 mS/m at sampling point 1 to 57.716 mS/m at sampling point 8 (Table 5.11) resulting in an overall increase in conductivity of 3.62% as indicated by the negative percentage value (Figure 5.13). A noticeable increase in conductivity was recorded from 55.464 mS/m at sampling point 2 to 58.060 mS/m at sampling point 3. A slight subsequent decrease was observed at sampling point 4 with subsequent increases at sampling points 5, 6 and 7. Conductivity decreased again at sampling point 5. Figure 5.13 indicates these changes in the mean conductivity values of Site 2. After the coagulation-flocculation treatment step, the mean conductivity increased followed by a decrease after the Dissolved Air Flotation (DAF) treatment step. Conductivity increased slightly subsequently and a decrease was recorded again after chlorination disinfection. The mean conductivity of 57.716 mS/m measured in the final water was within the TWQR of 0 to 70 mS/m (DWAF, 1996).

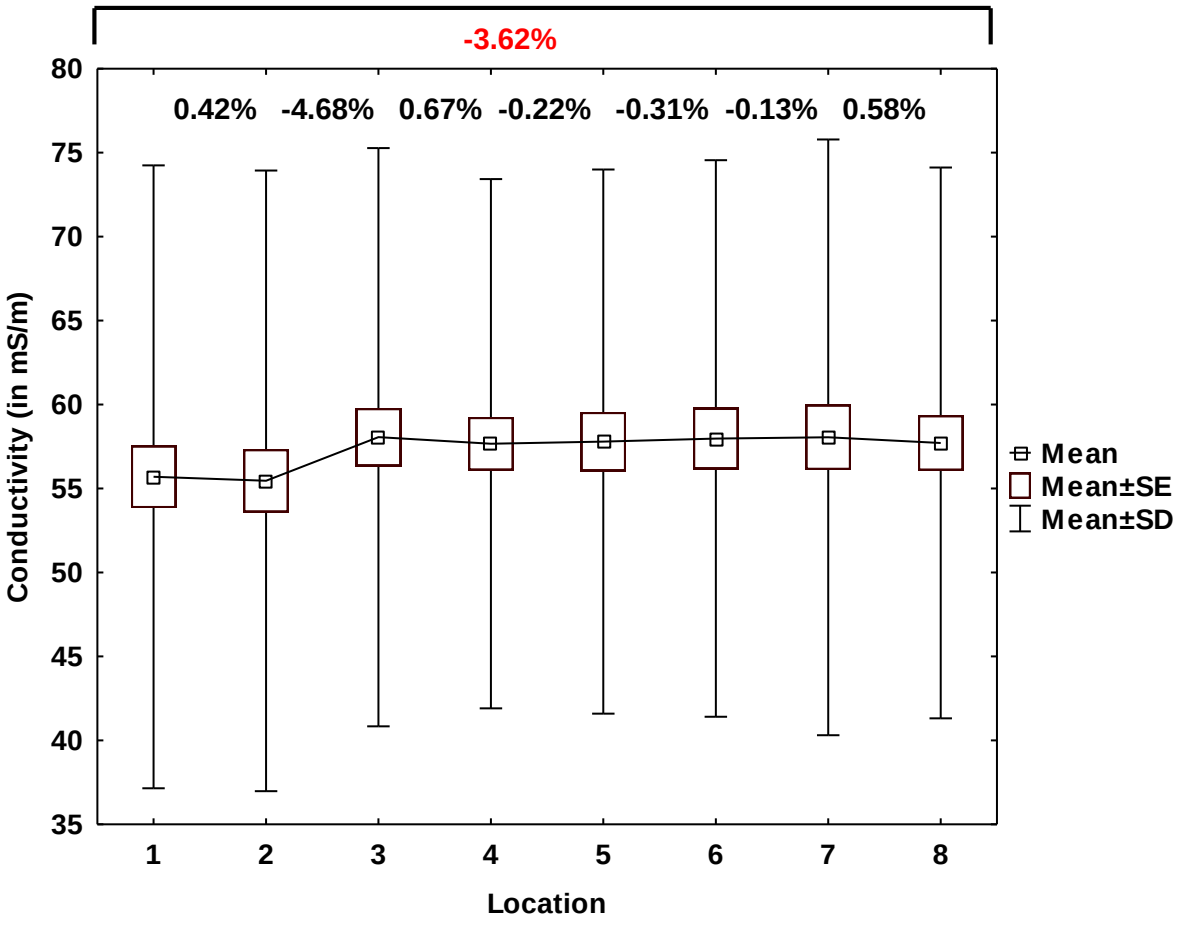


Figure 5.13: A box-and-whisker plot illustrating the change in the mean conductivity values (in mS/m) as well as the percentage removal between the sampling points of Site 2. SE = Standard Error; SD = Standard Deviation.

Turbidity was removed successfully as indicated by the overall removal of 98.73% (Figure 5.14). The mean turbidity value decreased significantly from 37.731 NTU at sampling point 1 to 0.479 NTU at sampling point 8 (Table 5.11). Turbidity increased slightly at sampling points 3 and 5 but not significantly. Significant decreases were recorded between sampling points 3 and 4 as well as from sampling points 5 to 7 (Table 5.12). The mean turbidity decreased from 37.668 NTU at sampling point 3 to 14.020 NTU at sampling point 4. A decrease from 16.546 NTU to 0.554 NTU was recorded between sampling points 5 and 7 (Table 5.11). The mean value of 0.479 NTU at sampling point 8 was well within the SANS 241 Class I operational limit of ≤ 1 NTU (DWAF, 1996). As illustrated by Figure 5.14, the DAF treatment step was effective in the removal of turbidity as well as the sedimentation and sand filtration treatment steps.

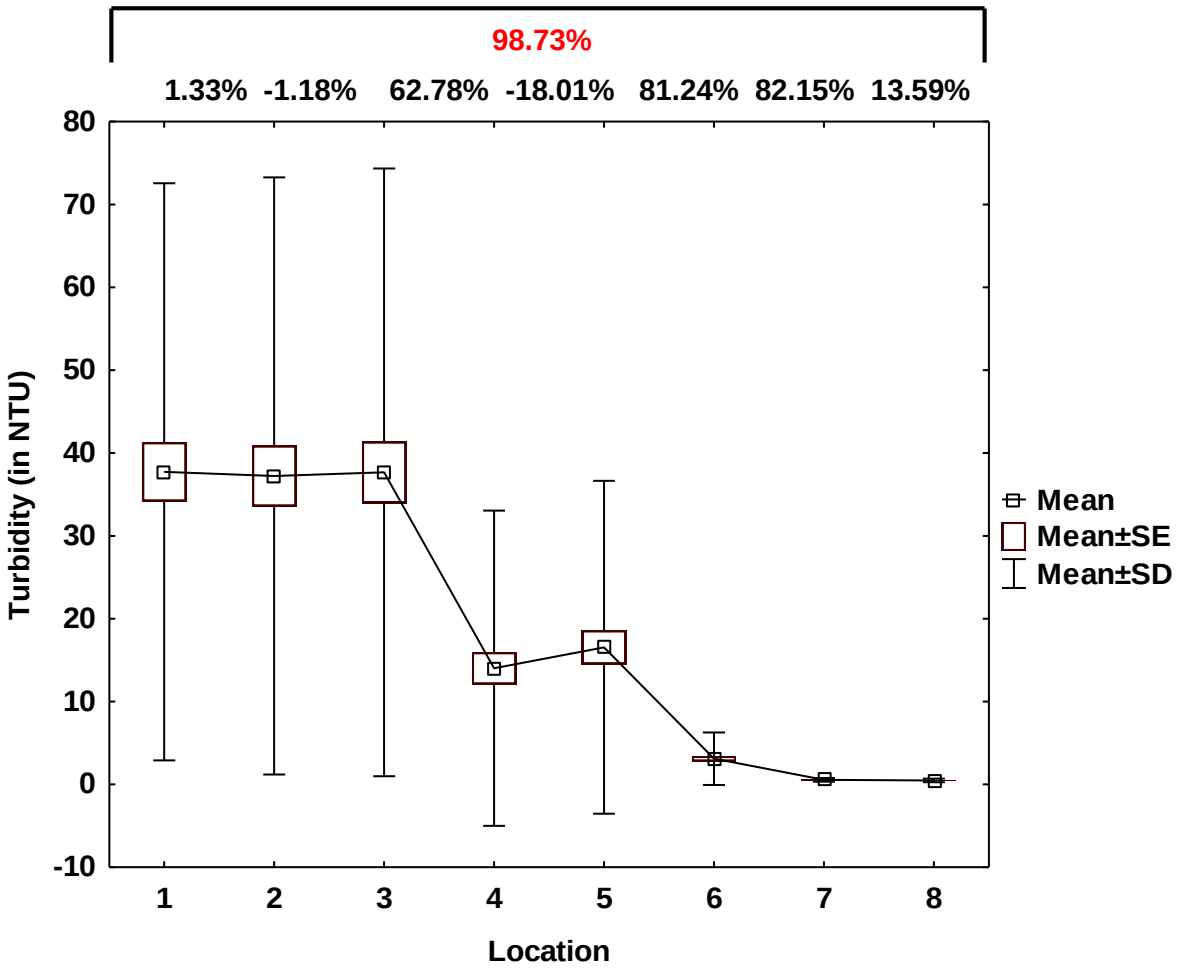


Figure 5.14: A box-and-whisker plot illustrating the change in the mean turbidity values (in NTU) as well as the percentage removal between the sampling points of Sites 2. SE = Standard Error; SD = Standard Deviation.

Similar DOC and TOC removal trends were observed with a 27.34% removal of DOC and a 27.11% removal of TOC (Figures 5.15 & 5.16). Significant differences were recorded for both variables between sampling points 1 and 8 (Table 5.12). The mean DOC value of 6.331 mg/L at sampling point 1 decreased to 4.600 mg/L at sampling point 8 and the mean TOC value of 7.028 mg/L decreased to 5.122 mg/L (Table 5.11). The mean DOC values showed a significant decrease from 6.340 mg/L to 5.534 mg/L between sampling points 2 and 4 and the mean TOC values from 7.028 mg/L to 6.476 mg/L between sampling points 1 and 3. Significant decreases were also observed for both DOC and TOC between sampling points 5 and 7 (Table 5.12). Increases in the mean DOC and TOC values were observed at sampling points 2 and 5. As illustrated by Figures 5.15 and 5.16, the removal of DOC and TOC was the most effective after the coagulation-flocculation, sedimentation and sand filtration treatment steps with the DAF treatment step also contributing to the removal. The DOC mean value of 4.600 mg/L and the TOC mean value of 5.122 mg/L recorded in the potable water were within the SANS 241 Class I operational limit of < 10 mg/L (SABS, 2011).

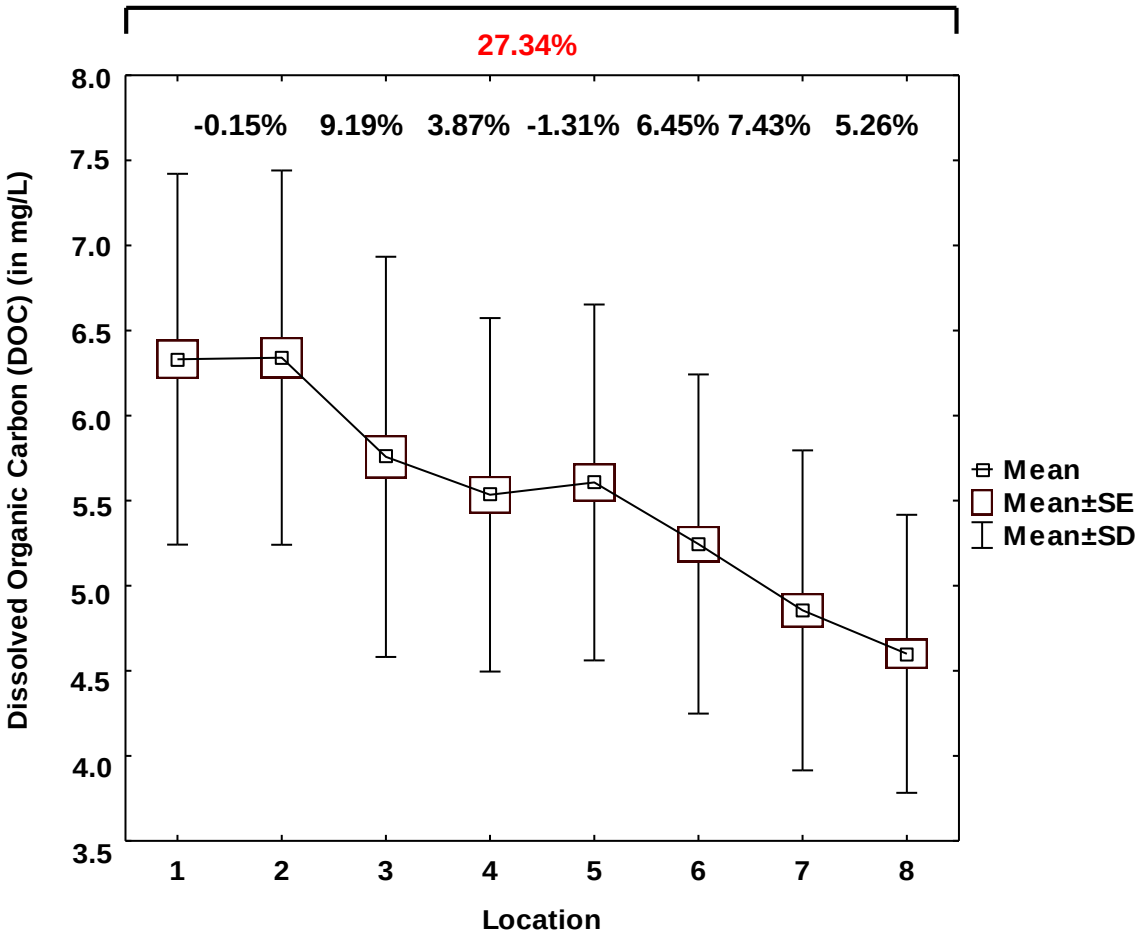


Figure 5.15: A box-and-whisker plot illustrating the change in the mean DOC values (in mg/L) as well as the percentage removal between the sampling points of Sites 2.SE = Standard Error; SD = Standard Deviation.

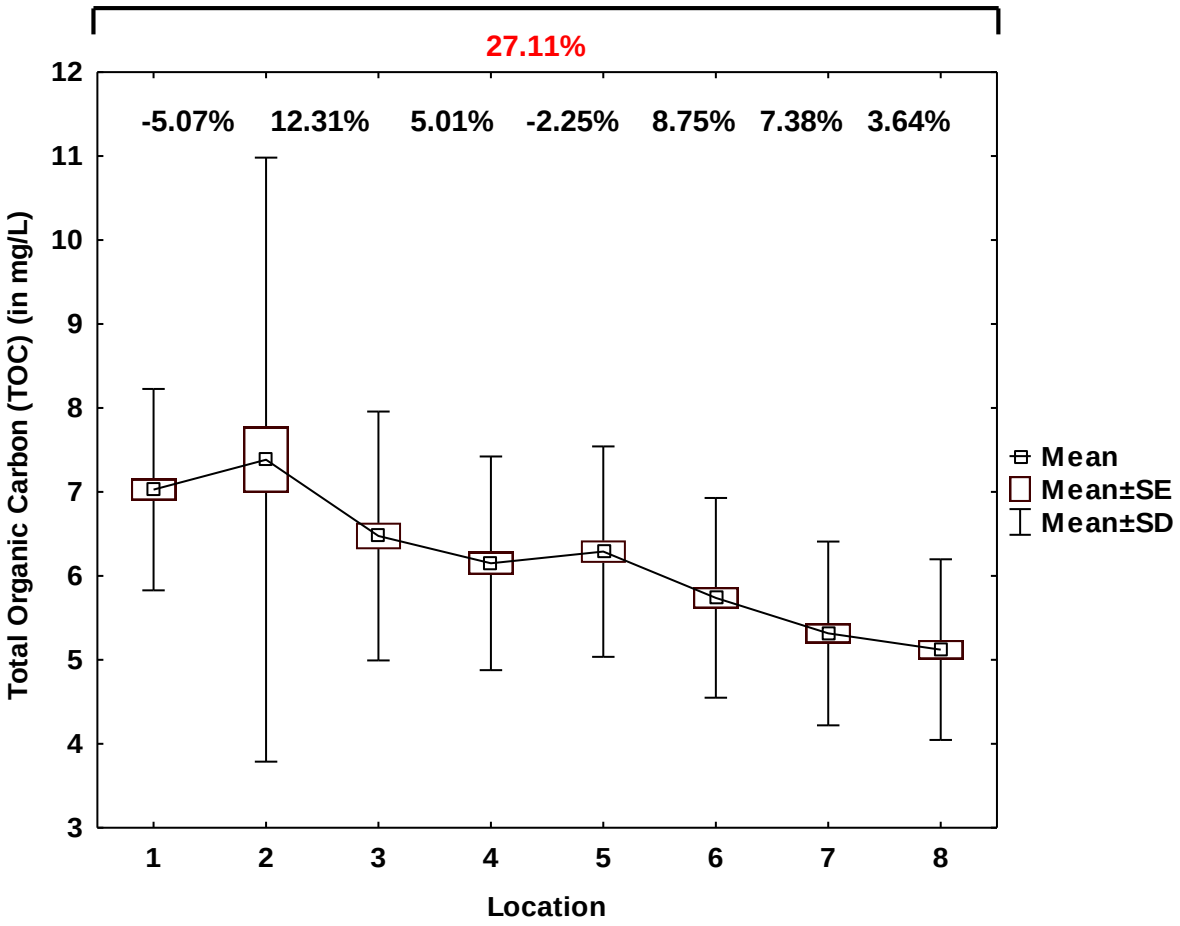


Figure 5.16: A box-and-whisker plot illustrating the change in the mean TOC values (in mg/L) as well as the percentage removal between the sampling points of Sites 2. SE = Standard Error; SD = Standard Deviation.

A complete removal, 99.70%, of TPP from the source water was found (Figure 5.17) with a significant decrease in the mean TPP value from 78.531 µg/L at sampling point 1 to 0.237 µg/L at sampling point 8 (Table 5.12). A decrease in TPP was observed at all the sampling points except for a slight increase at sampling point 3 (Table 5.11). A significant decrease occurred between sampling points 1 and 4 from 78.531 µg/L to 17.583 µg/L. The mean TPP value of 6.485 µg/L decreased significantly at sampling point 6 to 0.271 µg/L at sampling point 7. The pre-ozonation, DAF, sedimentation and sand filtration treatment steps effectively removed the majority of TPP as illustrated by Figure 5.17. The mean TPP value of 0.237 µg/L in the potable water was within the recommended limit for potable water as set by Rand Water namely 1 µg/L (Swanepoel *et al*., 2008a).

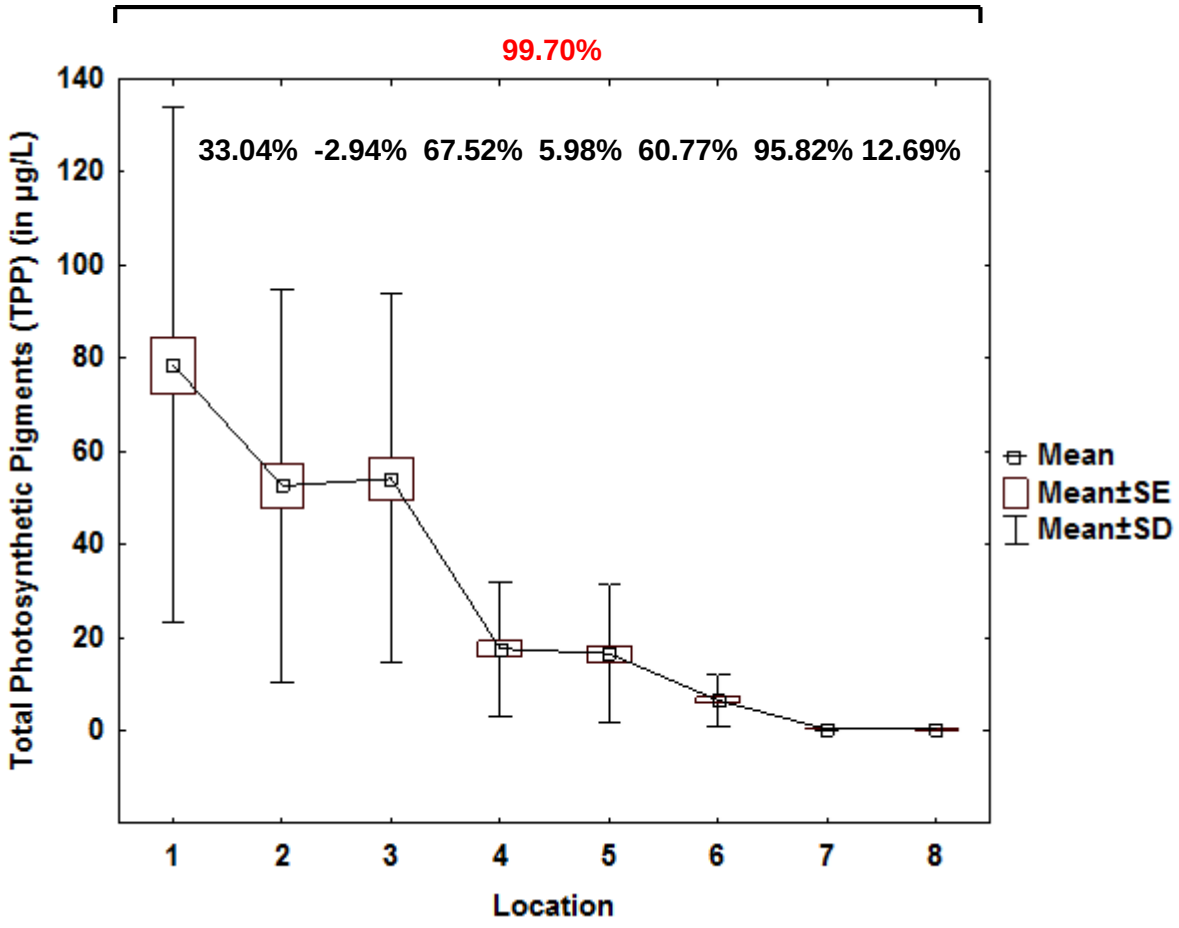


Figure 5.17: A box-and-whisker plot illustrating the change in the mean TPP values (in µg/L) as well as the percentage removal between the sampling points of Sites 2. SE = Standard Error; SD = Standard Deviation.

As the mean microcystin values recorded were < 0.18 µg/L (Table 5.11) at all the sampling points of Site 2, the most effective treatment step for the removal of microcystin from the source water could not be established. Mean values below the detection limit of the method namely 0.18 µg/L were halved for graphical representation purposes. A considerable variation from the mean microcystin value at sampling point 3 can be observed in Figure 5.18 though due to a maximum value of 0.23 µg/L that was reached after the pre-ozonation and coagulation-flocculation treatment steps. The mean microcystin value of < 0.18 µg/L recorded for the potable water was well below the SANS 241 limit of ≤ 1 µg/L for the microcystin concentration in potable water (SABS, 2011).

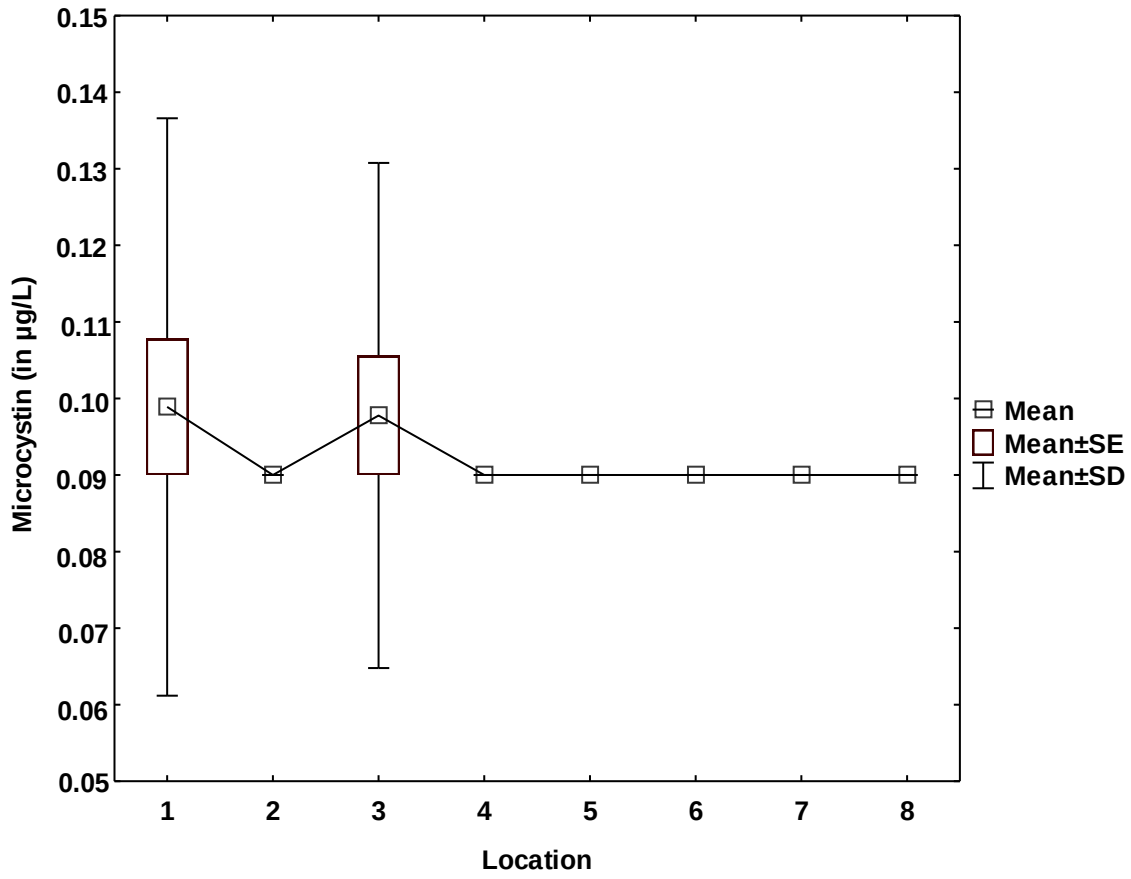


Figure 5.18: A box-and-whisker plot illustrating the change in the mean microcystin values (in µg/L) between the sampling points of Site 2. SE = Standard Error; SD = Standard Deviation.

As the mean geosmin values recorded were < 5 ng/L (Table 5.11) at all the sampling points of Site 2, the most effective treatment step for the removal of geosmin from the source water could not be established. Mean values below the detection limit of the method namely 5 ng/L were halved for graphical representation purposes. Figure 5.19 illustrates a considerable variation from the mean geosmin value recorded at sampling point 3 due to a maximum value of 31 ng/L that was reached following the pre-ozonation and coagulation-flocculation treatment steps. The mean geosmin value of < 5 ng/L recorded in the potable water was well below the geosmin limit of 30 ng/L for potable water as specified by Rand Water (Swanepoel *et al.*, 2008a).

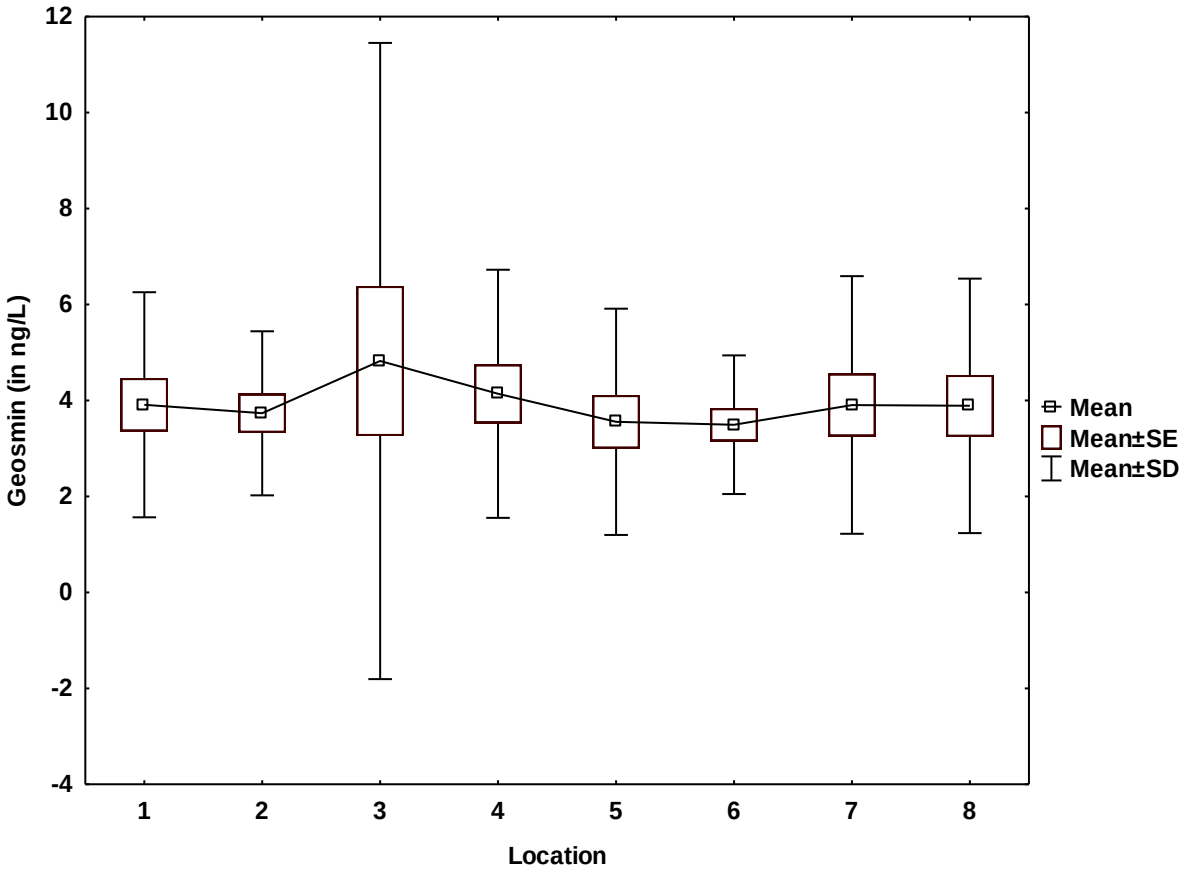


Figure 5.19: A box-and-whisker plot illustrating the change in the mean geosmin values (in ng/L) between the sampling points of Site 2. SE = Standard Error; SD = Standard Deviation.

5.2.2.2. Algal characteristics

The phytoplankton sampling of Site 2 was done at sampling points 1, 2 and 5. Table 5.13 provides the mean concentrations of the algal classes identified and enumerated at the different sampling points of Site 2.

Table 5.13: The mean concentrations of the algal classes identified and enumerated at the different sampling points of Site 2.

		Sampling points		
Algal class	**Unit**	**1 Source**	**2 Pre-ozone**	**5 Inter-ozone**
Bacillariophyceae (BACIL)	cells/ml	5167	5482	2173
Chlorophyceae (CHLORO)	cells/ml	14363	12394	4248
Cryptophyceae (CRYPTO)	cells/ml	< 36	11	0
Cyanophyceae (CYANO)	cells/ml	585	97	0
Dinophyceae (DINO)	cells/ml	< 36	0	0
Euglenophyceae (EUGLENO)	cells/ml	344	61	15
Total cells	cells/ml	20475	18046	6436

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on all the algal samples to determine whether there were statistical significant differences between the algal class concentrations at the different sampling locations of Site 2. These results are presented in Table 5.14.

Table 5.14: Kruskal-Wallis ANOVA results indicating statistically significant differences between the algal classes' concentration at the different sampling points of Site 2. The level of significance for statistical analysis was set at *p* = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference. Valid n = 50.

Algal class (in cells/ml)	Sampling points			
Bacillariophyceae		1	2	5
	1		x	✓
	2	x		x
	5	✓	x	
Chlorophyceae		1	2	5
	1		x	x
	2	x		x
	5	x	x	
Cryptophyceae		1	2	5
	1		x	✓
	2	x		✓
	5	✓	✓	
Cyanophyceae		1	2	5
	1		x	✓
	2	x		✓
	5	✓	✓	
Dinophyceae		1	2	5
	1		x	x
	2	x		x
	5	x	x	
Euglenophyceae		1	2	5
	1		x	x
	2	x		x
	5	x	x	
Total cells		1	2	5
	1		x	✓
	2	x		x
	5	✓	x	

As indicated by Table 5.14, significant differences were found for the Bacillariophyceae, Cryptophyceae and Cyanophyceae algal classes between sampling points 1 and 5 as well as for the total phytoplankton biomass (Table 5.13). Bacillariophyceae decreased from 5167 cells/ml at sampling point 1 to 2173 cells/ml at sampling 5 and the total phytoplankton biomass decreased from 20475 cells/ml to 6436 cells/ml. Cryptophyceae and Cyanophyceae were completely removed at sampling point 5. As illustrated by Figure 5.20, decreases were recorded at every sampling point for all the mean algal class concentrations except for Bacillariophyceae that increased slightly at sampling point 2. The total phytoplankton biomass decreased at every sampling point as well. Dinophyceae species were completely removed at sampling point 2. Percentage removals ranging from 95% to 100% indicated that Cryptophyceae, Cyanophyceae, Dinophyceae and Euglenophyceae species were effectively removed from the source water by ozonation. The mean concentrations of Chlorophyceae, Bacillariophyceae and the total phytoplankton biomass were 4248 cells/ml, 2173 cells/ml and 6436 cells/ml at sampling point 5 (Table 5.13) respectively. As a

result, lower percentage removals were recorded for Chlorophyceae (70.42%), Bacillariophyceae (57.94%) and for the total phytoplankton biomass (68.57%).

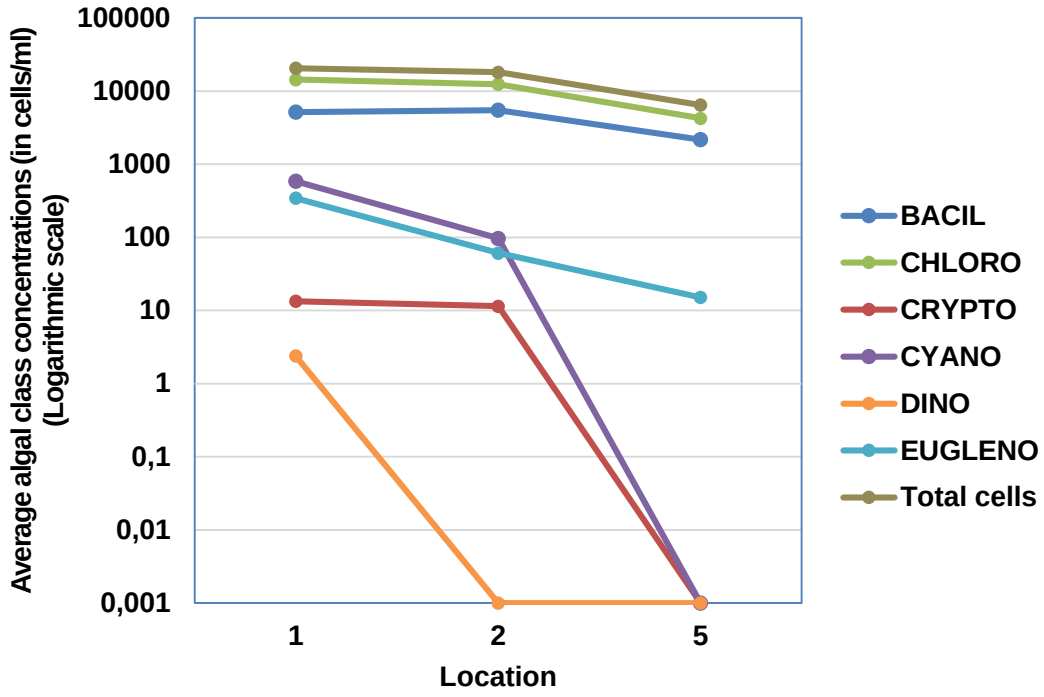


Figure 5.20: A line chart illustrating the change in the mean algal class concentrations (in cells/ml) between the sampling points of Site 2.

Problematic algal species with the potential to negatively impact on water treatment processes were identified in the source water of Site 2 (Refer to Chapter 4.4.2; Table 4.8). Table 5.15 provides the mean concentrations of the problematic algal species identified and enumerated at the different sampling points of Site 2.

Table 5.15: The mean concentrations of the problematic algal species identified and enumerated at the different sampling points of Site 2.

		Sampling points		
Species	**Unit**	**1 Source**	**2 Pre-ozone**	**5 Inter-ozone**
Coelastrum sp.	**cells/ml**	1346	424	318
Cyclotella sp.	**cells/ml**	4878	4601	1984
Pediastrum sp.	**cells/ml**	938	627	381
Scenedesmus sp.	**cells/ml**	10205	9885	2988
Trachelomonas sp.	**cells/ml**	333	54	9

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on the all the samples to determine statistical significant differences between the problematic algal species at the different sampling points of Site 2. These results are presented in Table 5.16.

Table 5.16: Kruskal-Wallis ANOVA results indicating statistically significant differences between the problematic algal species' concentrations at the different sampling points of Site 2. The level of significance for statistical analysis was set at *p* = 0.05. (✓) Statistically significant difference; (x) No statistically significant difference. Valid n = 49.

Algal species (in cells/ml)	**Sampling points**			
		1	**2**	**5**
***Coelastrum* sp.**	**1**		**x**	**x**
	2	**x**		**x**
	5	**x**	**x**	
***Cyclotella* sp.**				
	1		**x**	✓
	2	**x**		**x**
	5	✓	**x**	
***Pediastrum* sp.**				
	1		**x**	**x**
	2	**x**		**x**
	5	**x**	**x**	
***Scenedesmus* sp.**				
	1		**x**	**x**
	2	**x**		**x**
	5	**x**	**x**	
***Trachelomonas* sp.**				
	1		**x**	**x**
	2	**x**		**x**
	5	**x**	**x**	

No significant differences were observed between the algal species concentrations at the selected sampling points of Site 2 except for *Cyclotella* sp. that decreased significantly from 4878 cells/ml at sampling point 1 to 1984 cells/ml at sampling point 5 (Table 5.16). As illustrated by Figure 5.21, a reduction in the mean algal species concentration occurred at each sampling point. Ozonation appeared to be effective in the removal of *Trachelomonas* sp. with a removal of 97.28% recorded for this species but less effective in the removal of *Coelastrum* sp., *Cyclotella* sp., *Pediastrum* sp. and *Scenedesmus* sp. The percentage removal of *Cyclotella* sp. was low namely 59.33% even though there was a significant decrease recorded for this species. The percentages' removal of *Coelastrum* sp., *Pediastrum* sp. and *Scenedesmus* sp. ranged from 59% to 76%.

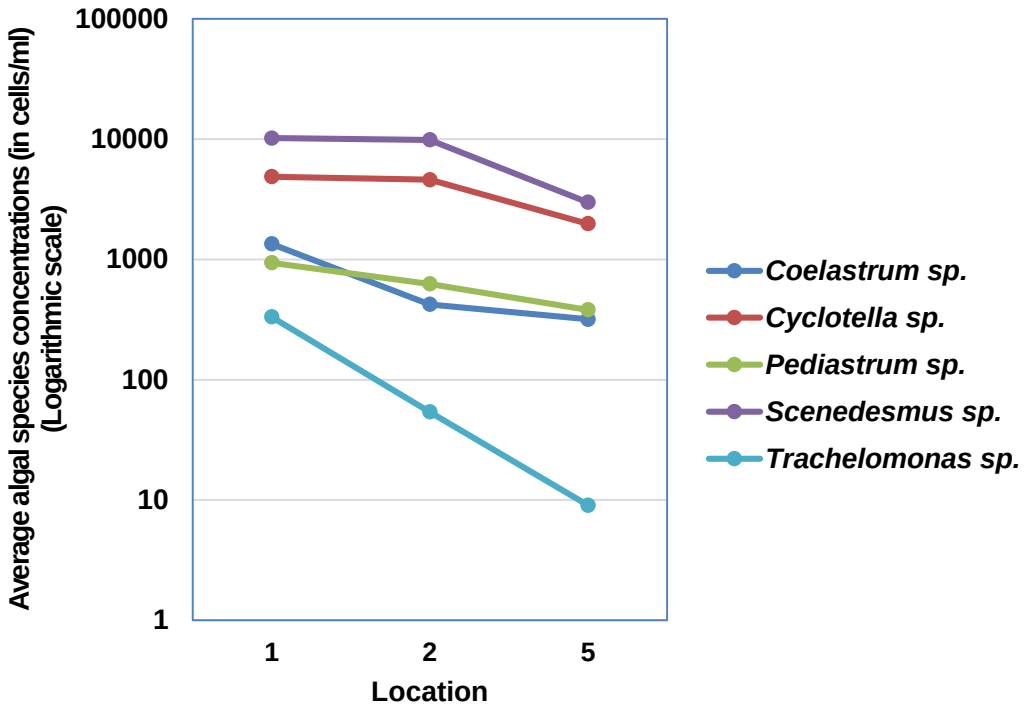


Figure 5.21: A line chart illustrating the change in the mean concentrations of the problematic algal species (in cells/ml) between the sampling points of Site 2.

5.2.2.3. Invertebrate characteristics

The invertebrate sampling of Site 2 was done at sampling points 1, 2 and 8. Table 5.17 provides the mean concentration of the invertebrate groups identified and enumerated at the different sampling points of Site 2.

Table 5.17: The mean concentrations of the invertebrate groups identified and enumerated at the different sampling points of Site 2.

		Sampling points		
Invertebrate group	**Unit**	**1 Source**	**2 Pre-ozone**	**8 Potable**
Hydracarina (Acari)	org/m^3	< 1	< 1	< 1
Cladocera (Clado)	org/m^3	28	1	< 1
Copepoda (Copep)	org/m^3	18	4	1
Diptera (Dipt)	org/m^3	10	6	4
Ephemeroptera (Ephem)	org/m^3	< 1	< 1	< 1
Nematoda (Nemat)	org/m^3	1	1	< 1
Ostracoda (Ostra)	org/m^3	1	1	< 1
Rotatoria (Rotat)	org/m^3	89	2	9
Unknown invertebrates (Uninv)	org/m^3	22	160	9
Other invertebrates (Other)	org/m^3	136	168	19
Total invertebrates (Total inv)	org/m^3	302	336	41

Kruskal-Wallis ANOVA multiple comparisons (2-tailed) tests were conducted on the all the samples to determine statistical significant differences between the concentrations of the invertebrate groups at the different sampling points of Site 2. These results are presented in Table 5.18.

Table 5.18: Kruskal-Wallis ANOVA results indicating statistically significant differences between the mean concentrations of the invertebrate groups at the different sampling points of Site 2. The level of significance for statistical analysis was set at p = 0.05. (✓) = Statistically significant difference; (x) = No statistically significant difference. Valid n = 52.

Invertebrate group (in org/m^3)	Sampling points			
Hydracarina		1	2	8
	1		x	x
	2	x		x
	8	x	x	
Cladocera		1	2	8
	1		x	✓
	2	x		x
	8	✓	x	
Copepoda		1	2	8
	1		x	✓
	2	x		x
	8	✓	x	
Diptera		1	2	8
	1		x	x
	2	x		x
	8	x	x	
Ephemeroptera		1	2	8
	1		x	x
	2	x		x
	8	x	x	
Nematoda		1	2	8
	1		x	x
	2	x		x
	8	x	x	
Ostracoda		1	2	8
	1		x	x
	2	x		x
	8	x	x	
Rotatoria		1	2	8
	1		✓	✓
	2	✓		x
	8	✓	x	
Unknown invertebrates		1	2	8
	1		✓	x
	2	✓		x
	8	x	x	
Other invertebrates		1	2	8
	1		x	✓
	2	x		✓
	8	✓	✓	
Total invertebrates		1	2	8
	1		x	✓
	2	x		✓
	8	✓	✓	

The mean total invertebrate concentrations at sampling points 1 and 2 of 302 org/m^3 and 336 org/m^3 respectively exceeded the crisis limit of 250 org/m^3 as set by Rand Water. The mean total invertebrate concentration at sampling point 8 of 41 org/m^3 was below the maximum permissible limit but exceeded the recommended limit of 20 org/m^3. The recommended limit of 1 org/m^3 for Diptera in potable water was also exceeded by the mean value of 4 org/m^3 at sampling point 8.

Significant decreases between sampling points 1 and 8 were observed in the mean concentrations of Cladocera, Copepoda, Rotatoria, Other groups of invertebrates and Total invertebrates (Table 5.18). Cladocera, Nematoda and Ostracoda were completely removed from the source water. The mean concentrations of Hydracarina and Ephemeroptera were insignificant as these concentrations were below 1 org/m^3. As illustrated by Figure 5.22, noticeable increases were observed at sampling point 2. Unknown invertebrates increased significantly between sampling points 1 and 2 from 22 org/m^3 to 160 org/m^3, Other groups of invertebrates from 136 org/m^3 to 168 org/m^3 and Total invertebrates from 302 org/m^3 to 336 org/m^3 (Table 5.17). Other groups of invertebrates and Total invertebrates subsequently decreased again significantly between sampling points 2 and 8. Rotatoria decreased significantly between sampling points 1 and 2 from 89 org/m^3 to 2 org/m^3 and increased again at sampling point 8. Copepoda and Diptera decreased at every sampling point.

Unknown invertebrates only reflected a 59.09% removal from the source water and Diptera a low 60%. A 94.44% removal was observed for Copepoda and 89.89% for Rotatoria. Other groups of invertebrates and Total invertebrates reflected an 86% removal. Generally the invertebrates were reduced between the pre-ozonation and the chlorination disinfection treatment steps indicating that sand filtration contributed the most to the removal of invertebrates. Increases were observed though for some groups after pre-ozonation.

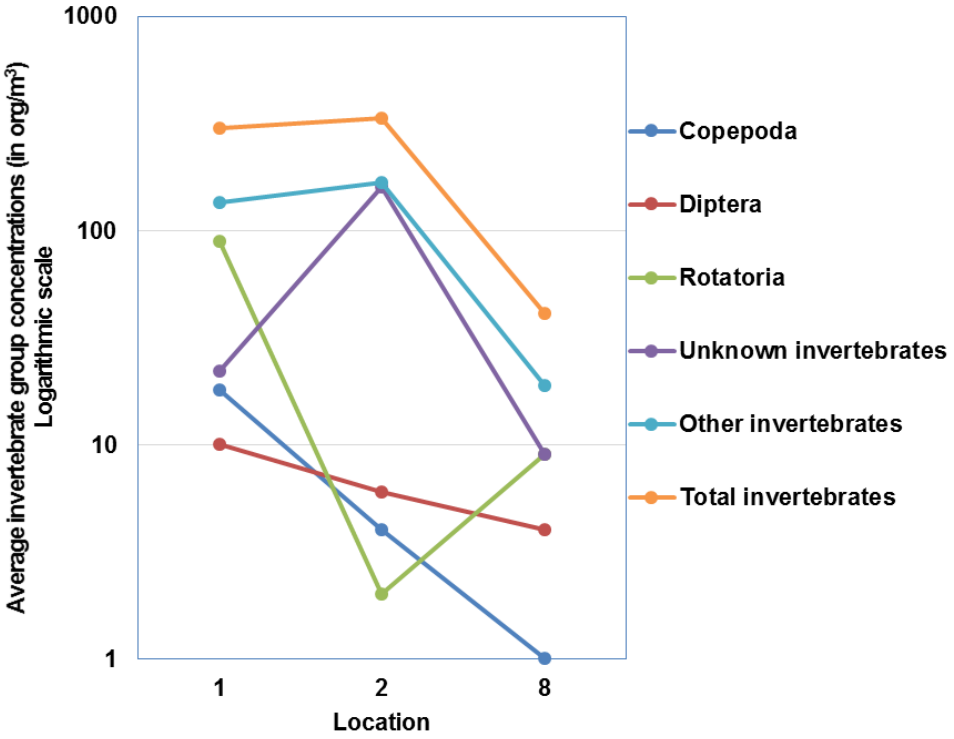


Figure 5.22: A line chart illustrating the change in the mean invertebrate group concentrations (in org/m^3) between the sampling points of Site 2.

5.3. A comparison of the efficacy of GAC, UV light disinfection and ozonation in the removal of turbidity, DOC and TPP

Sampling occurred during different time periods at the two sites, namely from January 2009 to December 2010 at Site 1 and from January 2010 to December 2011 at Site 2. Therefore, in order to directly compare the effectiveness of GAC, UV light disinfection and ozonation in the removal of turbidity, DOC, and TPP from the source water of Sites 1 and 2, the turbidity measurements and the DOC and TPP concentrations were grouped into classes consisting of appropriate denominations. Table 5.19 provides a list of the classes and the denominations assigned to each class.

Table 5.19: The classes and the denominations assigned to the source water turbidity measurements and the DOC and TPP concentrations of Sites 1 and 2.

Variable	Class 1	Class 2	Class 3	Class 4	Class 5
Turbidity (in NTU)	0 - 39	39 - 78	> 78	-	-
DOC (in mg/L)	0 - 6	> 6	-	-	-
TPP(in µg/L)	0 - 39	39 - 78	78 - 117	117 - 156	> 156

The percentages' removal of turbidity, DOC and TPP from the source water of Site 1 were calculated between the source water and the GAC adsorption treatment step, the source water and UV light disinfection treatment step as well as between the source and potable water. The percentages' removal of turbidity, DOC and TPP from the source water of Site 2 were calculated between the source water and the intermediate ozonation treatment step and between the source and potable water. The percentages' removal of turbidity, DOC and TPP are provided in Table 5.20.

Table 5.20: The percentages' removal of turbidity, DOC and TPP from the source water of Site 1 after GAC adsorption and UV light disinfection and from the source water of Site 2 after ozonation.

	Site 1			Site 2	
Source	**GAC**	**UV**	**Potable**	**Ozone**	**Potable**
Turbidity					
Class 1	98.43%	98.15%	97.56%	64.32%	96.95%
Class 2	99.58%	99.54%	99.26%	55.07%	99.29%
Class 3	99.80%	99.84%	99.84%	48.86%	99.55%
DOC					
Class 1	23.22%	18.69%	36.52%	0.14%	23.98%
Class 2	36.45%	37.89%	41.32%	21.23%	31.81%
TPP					
Class 1	98.70%	98.95%	99.82%	66.63%	99.31%
Class 2	99.35%	99.34%	99.80%	72.12%	99.62%
Class 3	99.51%	99.70%	99.95%	75.13%	99.72%
Class 4	99.49%	99.47%	99.88%	83.93%	99.76%
Class 5	99.84%	99.88%	99.99%	92.30%	99.88%

As indicated by Table 5.20, turbidity was removed effectively from the source water of both sites with percentages' removal ranging from 96% to more than 99%. The filtration properties of the GAC treatment step proved to more effective in the removal of turbidity than ozonation as indicated by the percentages' removal effected by ozone (Table 5.20). UV light disinfection appeared to have no influence on the removal of turbidity.

The source water of Site 1 reflected higher DOC percentages' removal than the source water of Site 2. GAC adsorption removed 23.22% of the Class 1 DOC concentration and 36.45% of the Class 2 DOC concentration whereas ozonation only removed 0.14% of the Class 1 DOC concentration and 21.23% of the Class 2 DOC concentration (Table 5.20). It appeared that both GAC adsorption and ozonation were more effective in the removal of higher concentrations of DOC. GAC adsorption was more effective than ozonation in the removal of DOC. UV light disinfection appeared to have no effect on the removal of DOC.

TPP was removed effectively from the source water of both sites with percentages' removal of more than 99%. It was observed that GAC adsorption was more effective than ozonation in the removal of TPP (Table 5.20). Percentages' removal by ozone ranged from 66% to 92% whereas percentages' removal by GAC adsorption were more than 98%. It appeared that ozonation was more effective at higher concentrations of TPP as indicated by the parallel increases in the percentage removals. The remainder of TTP was removed from the source water of Site 2 by the treatment steps following intermediate ozonation namely sedimentation, sand filtration and chlorination disinfection. As was illustrated by Figure 5.6, these treatment steps also played an important role in the removal of TPP from the source water of Site 1. UV light disinfection appeared to have no effect on the removal of TPP.

5.4. Multivariate analysis

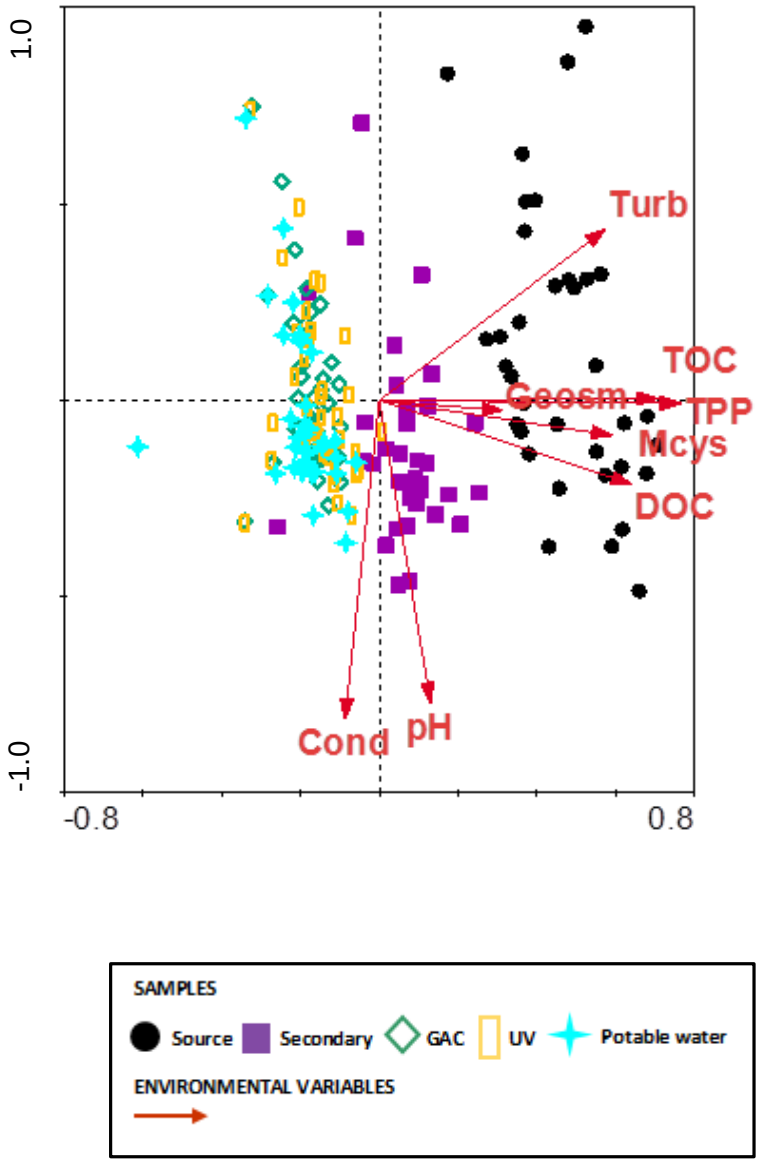
The multivariate analysis methods that were used in this study are described in Section 3.3.

An indirect linear gradient analysis, the principal component analysis (PCA), was used to determine the principal physical and chemical water quality variables associated with the samples of Sites 1 and 2 during the different water treatment steps. The resulting PCA ordination plots (Figures 5.23 & 5.24) clearly illustrate the differences between the samples after each treatment step.

The source water of Site 1 was clearly distinguishable from the secondary water after the coagulation-flocculation, sedimentation and sand filtration treatment steps (Figure 5.23). The first axis of the PCA ordination plot explained 37.5% of the variance in the data and the second axis 18.3% of the variance. Samples taken after GAC adsorption, UV light and chlorination disinfection were found to be distinctly different from the source and secondary water. The GAC adsorption, UV light disinfection and potable water samples were not clearly separated from each other. The potable water samples were negatively correlated with turbidity, TPP, DOC, TOC, microcystin and geosmin indicating the successful removal of these variables. There was an indication that microcystin and geosmin were released during the purification process. Secondary water samples were closely associated with pH whereas conductivity was associated with the GAC adsorption, UV light disinfection and potable water samples.

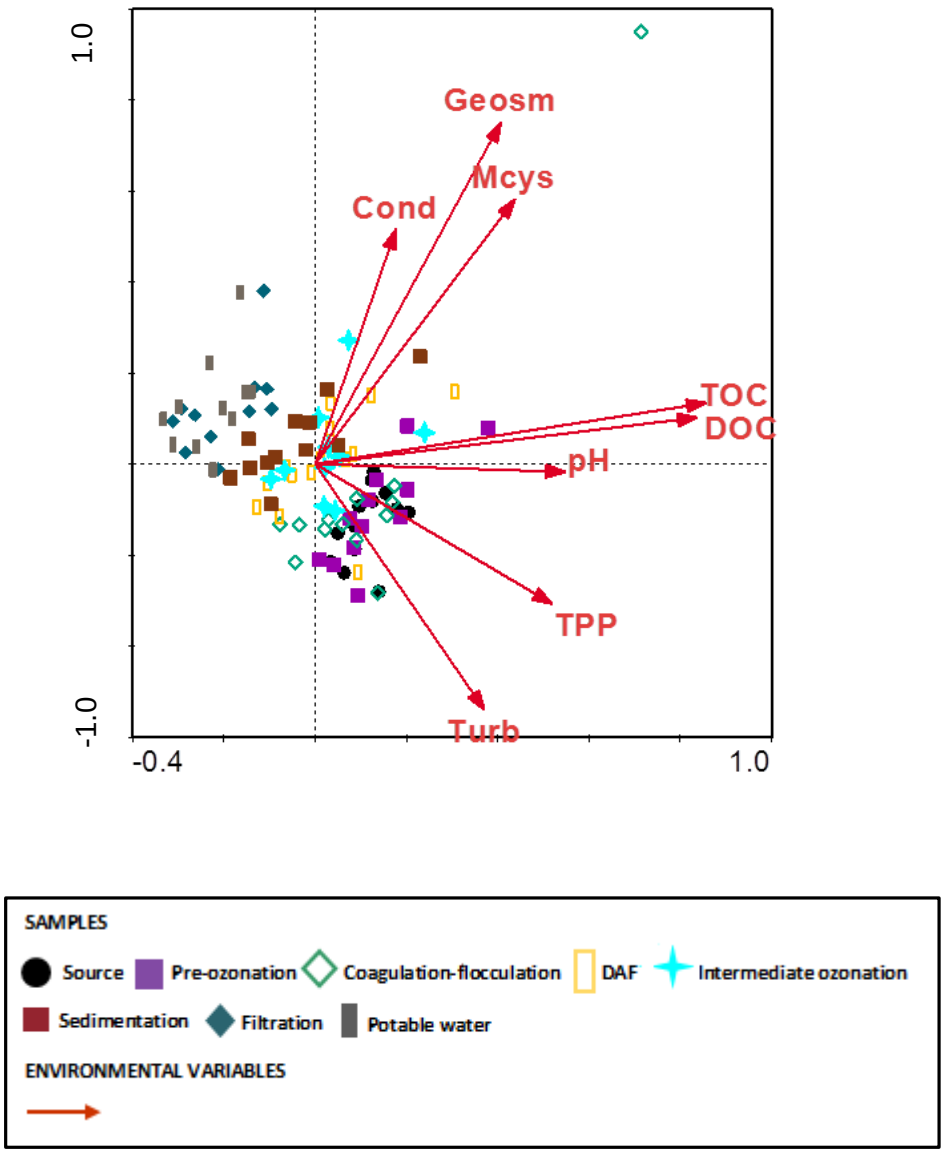
As observed in the PCA ordination plot of Site 2 (Figure 5.24), the potable water samples and the filtration treatment step samples were grouped together and a distinct difference was observed between these samples and the sedimentation treatment step samples. The DAF and intermediate ozonation treatment steps samples appeared to be grouped together and were distinguishable from the source water, pre-ozonation and coagulation-flocculation treatment steps samples that were grouped together. The first axis of the PCA ordination plot explained 38.9% of the variance in the

data and the second axis 20.2% of the variance. The source water, pre-ozonation and coagulation-flocculation treatment steps samples were closely associated with TPP and turbidity. TPP and turbidity were negatively correlated with the potable water indicating the effective removal of these variables. It appeared that microcystin and geosmin were released into the water during the water treatment process. In addition, pH, DOC, TOC and conductivity values appeared to increase during the water treatment process.



Axes	1	2	3	4	Total variation
Eigenvalues:	0.375	0.183	0.125	0.111	1.000
Species-environment correlations:	0.962	0.991	0.878	0.895	
Cumulative percentage variance					
of species data:	37.5	55.8	68.4	79.5	
of species-environment relation:	39.6	60.1	71.1	81.2	
Sum of all eigenvalues:					1.000
Sum of all canonical eigenvalues:					0.878

Figure 5.23: A PCA site plot showing the correlation between the principal physical and chemical water quality variables and the water samples sampled at the different locations of Site 1



Axes	1	2	3	4	Total variation
Eigenvalues:	0.389	0.202	0.150	0.120	1.000
Species-environment correlations:	0.975	0.969	0.995	0.980	
Cumulative percentage variance of species data:	38.9	59.1	74.1	86.1	
of species-environment relation:	39.2	59.3	75.1	87.3	
Sum of all eigenvalues:					1.000
Sum of all canonical eigenvalues:					0.942

Figure 5.24: A PCA site plot showing the correlation between the principal physical and chemical water quality variables and the water samples sampled at the different locations of Site 2.

5.5. Discussion

The physical and chemical characteristics as well as the algal and invertebrate compositions of the source water of Sites 1 and 2 were altered by the respective treatment processes at each site. The effect of a particular treatment step on a water quality variable was recorded as either a change in the variable or a percentage removal of a variable. The efficacy of GAC adsorption, UV light disinfection and ozonation, in addition to the efficacy of conventional treatment processes, were determined by the changes in the pH and conductivity and by the removal of turbidity, DOC, TOC, TPP, microcystin, geosmin, phytoplankton and invertebrates.

5.5.1. Physical and chemical characteristics

The mean pH of the potable water of Sites 1 and 2 namely 8.07 and 8.27 respectively, were high but within the SANS 241 limit for potable water. A slight increase in the mean pH from 8.04 measured in the source water to 8.07 measured in the potable water of Site 1 was observed (Table 5.3). The mean pH of Site 2 decreased significantly from 8.73 in the source water to 8.27 in the potable water (Table 5.11). These changes in pH are most probably due to the different coagulation-flocculation treatment processes in use at each site. Site 1 uses hydrated lime and activated sodium silicate as coagulants and ferric chloride to aid flocculation and Site 2 uses ferric chloride and aluminium sulphate as coagulants followed by the addition of hydrated lime. The mean pH of both sites increased significantly after the coagulation-flocculation process. This increase in pH can be attributed to the addition of hydrated lime during the coagulation-flocculation treatment step as the addition of an alkali will increase the pH (Schutte, 2006). The use of hydrated lime in combination with coagulants enhances the coagulation-flocculation process as it results in a “pH shock” that renders algal cells immobile (Ewerts *et al.*, 2013). This observation was also confirmed by PCA ordination diagrams (Figures 5.23 & 5.24).

A significant decrease in the pH of Site 1 was observed after GAC adsorption. This decrease in pH corresponded to a significant decrease in TPP after GAC adsorption (Table 5.3). High rates of photosynthesis increase pH resulting in a positive correlation between TPP and pH. The total phytoplankton biomass was effectively removed from the source water of Site 1 by the coagulation-flocculation, sedimentation, sand filtration and GAC adsorption and filtration treatment steps. This removal resulted in a decrease in TPP and therefore in pH as well. It must be noted though that fluctuations in pH can affect the efficacy of GAC adsorption (USEPA, 2000). An increase in pH, although not significant, was observed after UV light disinfection. This could be an indication of algal growth as the photosynthesis process increased pH again. The pH of Site 2 displayed decreases after pre- and intermediate ozonation (Table 5.11). Carrim (2006) also observed a

decrease in pH after pre-ozonation. These decreases cannot be attributed to ozonation as ozone has no effect on pH (USEPA, 2006).

The mean conductivity of the potable water of Site 2 namely 57.716 mS/m was within the TWQR for potable water but the mean conductivity of the potable water of Site 1, 71.430 mS/m, exceeded the TWQR guidelines. The mean conductivity values of Sites 1 and 2 increased from the source water to the potable water. A significant increase from 61.533 mS/m to 71.430 mS/m (Table 5.3) was found for Site 1 with an overall increase of 16.08%. The mean conductivity of Site 2 increased from 55.7 mS/m to 57.716 mS/m (Table 5.11) with an overall increase in conductivity of 3.62%. The mean conductivity values of both sites increased after the coagulation-flocculation treatment steps. A significant increase in conductivity was also found after the chlorination disinfection treatment step of Site 1. These increases in conductivity were as a result of the addition of coagulants such as ferric chloride and aluminium sulphate during the coagulation-flocculation process and chlorine during the chlorination disinfection process. Conductivity is a measure of dissolved material in water which is comprised of inorganic ions such as chloride, iron and aluminium amongst others. The greater the number of ions in solution, the higher the conductivity (Dallas & Day, 2004). Figures 5.23 and 5.24 illustrate the positive correlation between pH and conductivity as well the increase in conductivity during the purification process. A slight decrease in conductivity was observed after GAC adsorption at Site 1 most probably due to the adsorption of dissolved ions. Ozonation at Site 2 did not have an effect on conductivity. Increases in conductivity at MWC were also observed by Mamba *et al.* (2009) who also recorded no impact on conductivity by ozonation.

Turbidity was removed successfully from the source water of Sites 1 and 2 with a 98.46% removal recorded for Site 1 (Figure 5.3) and a 98.73% removal for Site 2 (Figure 5.14). Significant decreases from the source to the potable water were observed for both sites and the mean potable water turbidity values of 0.361 NTU for Site 1 and 0.479 NTU for Site 2 were well within the TWQR for potable water.

GAC adsorption preceded by the conventional coagulation-flocculation, sedimentation and sand filtration treatment steps were effective in the removal of turbidity from the source water of Site 1. This is an indication of an effective coagulation-flocculation process. The effectiveness of GAC adsorption in the removal of turbidity was expected as its reliability in the removal of dissolved solids has been proven (USEPA, 2000). GAC also acts as a filter that traps the particulate material in the granular activated carbon particles. DAF removed turbidity to a large extent from the source water of Site 2 and the sedimentation and sand filtration treatment steps removed the remainder. Ozonation and UV light disinfection did not appear to have an effect on turbidity as can be expected.

Contrary to the findings of this study, Mamba *et al.* (2009) and Morrison (2009) found that ozonation reduced turbidity. If turbidity is caused by organic material, ozone might oxidise the organic material which will reduce the turbidity. However, similar to the findings of this study, Van der Walt *et al.* (2009) also indicated that ozone and UV light disinfection were not effective in the removal of turbidity. A direct comparison of the percentage turbidity removals between Sites 1 and 2 confirmed the findings that GAC adsorption and filtration were more effective than ozone in the removal of turbidity and that UV light disinfection did not have an impact on the removal of turbidity.

Significant differences between the mean DOC and TOC values of the source and potable water were observed for both sites although the overall percentages' removal of DOC and TOC were low. The mean DOC value of Site 1 decreased from 6.842 mg/L in the source water to 4.064 mg/L in the potable water and the mean TOC value decreased from 7.892 mg/L to 4.143 mg/L (Table 5.3). The mean DOC value of Site 2 decreased from 6.331 mg/L in the source water to 4.6 mg/L in the potable water and the mean TOC value decreased from 7.028 mg/L to 5.122 mg/L (Table 5.11). The overall percentage removal of DOC for Site 1 was 40.81% (Figure 5.4) and for Site 2 only 27.34% (Figure 5.15). The overall percentage removal of TOC for Site 1 was 47.51% (Figure 5.5) and for Site 2 a low 27.11% (Figure 5.16). Mamba *et al.* (2009) found that a high pH value after the coagulation-flocculation treatment step resulted in a low percentage removal of DOC and TOC. Ashery *et al.* (2010) observed that the removal of DOC is the most effective at a pH of between 5 and 6 when aluminium sulphate is used as a coagulant. The low percentages' removal of DOC observed at high pH values after coagulation-flocculation at Site 2 confirmed these findings.

The DOC of Site 1 increased after UV light disinfection indicating the presence of organic material probably in the form of substances released by algal cells and bacteria. Increases in the mean DOC and TOC values of Site 2 were observed after pre- and intermediate ozonation. Ramseier (2010) found that ozonation led to a rapid increase in assimilable organic carbon. The PCA ordination plot (Figure 5.24) illustrates these increases. Carrim (2006) observed a similar low percentage removal of DOC for Site 2 as well as increases in DOC. According to the Environmental Protection Agency (1999b), ozonation increases biodegradable DOC as large organic molecules are converted into smaller organic molecules that are easily biodegradable. Increases in DOC can result in cost implications for a treatment plant as the chlorine demand increases accordingly. The mean DOC and TOC values recorded in the potable water of both sites were still within the SANS 241 Class I operational limit though. The coagulation-flocculation, sedimentation, sand filtration and GAC adsorption treatment steps proved to be the most effective in the removal of DOC and TOC from the source water of Site 1.

A direct comparison of the DOC removal from the source water of Sites 1 and 2 indicated higher percentages' removal of DOC from the source water of Site 1 (Table 5.20). GAC adsorption removed 23.22% and 36.45% of the Class 1 and 2 DOC concentrations respectively whereas ozonation only removed 0.14% and 21.23% of the Class 1 and 2 DOC concentrations. It appeared that both treatment processes were more efficient at removing higher concentrations of DOC as indicated by the higher percentages' removal of the Class 2 DOC concentration. GAC adsorption in combination with the coagulation-flocculation, sedimentation and sand filtration treatment steps were more effective in the removal of DOC than ozone and it follows that it will be more effective in the removal of TOC as well. As expected, UV light disinfection made no contribution to the removal of DOC and TOC as the aim of UV light disinfection is the inactivation of protozoan parasites such as *Cryptosporidium* and *Giardia* (USEPA, 2006).

TPP was removed effectively from the source water of both sites with an overall removal of 99.86% for Site 1 (Figure 5.6) and an overall removal of 99.70% for Site 2 (Figure 5.17). The mean TPP values in the potable water of both sites were within the recommended limit for potable water as set by Rand Water. TPP was removed from the source water of Site 1 throughout the treatment process with the coagulation-flocculation, sedimentation, sand filtration and GAC adsorption treatment steps contributing significantly to the removal. Removal of TPP by UV light disinfection was not significant. TPP was reduced significantly from 62.196 µg/L in the source water of Site 1 to 0.383 µg/L after GAC adsorption to 0.086 µg/L in the potable water (Table 5.3). TPP was removed from the source water of Site 2 after every treatment step except for a slight increase after the coagulation-flocculation treatment step. This increase was probably due to the presence of Chlorophyceae species. Algal sampling done prior to the coagulation-flocculation treatment step and not directly after this step indicated the presence of Chlorophyceae. TPP decreased significantly from 78.531 µg/L in the source water of Site 2 to 0.237 µg/L in the potable water. Significant removals of TPP from the source water of Site 2 can be attributed to the pre-ozonation, DAF, sedimentation and sand filtration treatment steps. As illustrated by Figure 5.17, DAF fulfilled an important role in the removal of TPP. This finding was confirmed by Morrison (2009). Intermediate ozonation effected a removal of TPP but not significantly.

A direct comparison of the percentages' TPP removal between Sites 1 and 2 indicated that GAC adsorption was more effective than intermediate ozonation in the removal of TPP. Percentages' removal by GAC adsorption were more than 98% and percentages' removal by ozone ranged from 66% to 92%. The removal of TPP by UV light disinfection was not significant. Increases in the TPP Class concentrations corresponded to increases in the percentages' removal by ozonation indicating that ozone was more efficient at higher TPP concentrations.

The coagulation-flocculation, sedimentation and sand filtration treatment steps effected the complete removal of microcystin from the source water of Site 1 (Figure 5.7). The mean microcystin value of Site 1 decreased from 0.523 µg/L in the source water to < 0.18 µg/L in the potable water (Table 5.3). The mean potable water microcystin values of both sites were well below the SANS 241 limit of ≤ 1 µg/L. The percentage removal of microcystin from the source water of Site 2 could not be established as the microcystin concentrations were too low. A high maximum microcystin value was recorded after the pre-ozonation and coagulation-flocculation treatment steps though (Figure 5.18). Ozonation can cause damage to the cell walls of cyanobacteria resulting in the leaching of toxins such as microcystin into the water. The high maximum microcystin value following pre-ozonation was probably due to the damage caused to the cell membranes of *Microcystis* spp. (Ramseier *et al.*, 2003). Figure 5.24 illustrates the release of microcystin into the water during the purification process.

The geosmin concentrations of < 5 ng/L in the potable water of both sites were well below the recommended limit of 30 ng/L as specified by Rand Water (Tables 5.3 & 5.11). As expected, GAC adsorption effected the removal of geosmin from the source water of Site 1. GAC adsorption is regarded as the ideal treatment process for taste and odour (Van der Walt, *et al.*, 2009). A high maximum value of 66.59 ng/L was observed after the UV light disinfection treatment step. The UV system at RWB is an in-line system where the UV lamps are placed perpendicular to the water flow. In these systems the walls of the pipes are irradiated by visible light only causing the regrowth of algae on the walls of the pipes (Ijpelaar *et al*., 2007). This variation from the mean geosmin value of < 5 ng/L can be attributed to the regrowth of geosmin-producing algal species in the UV system. The PCA ordination plot (Figure 5.23) illustrates the release of geosmin during the treatment process. The geosmin concentrations of Site 2 were < 5 ng/L at all sampling points. A variation from this mean value was observed with a maximum value of 31 ng/L that was reached after the pre-ozonation and coagulation-flocculation treatment steps. This was an indication of the release of geosmin within the treatment process due to cell lysis.

5.5.2. Algal characteristics

The algal classes Bacillariophyceae, Chlorophyceae, Cryptophyceae, Cyanophyceae and Dinophyceae that occurred in the source water of Site 1 as well as the total phytoplankton biomass were significantly reduced in the source water of Site 1. The conventional treatment steps of coagulation-flocculation, sedimentation and sand filtration in combination with the GAC filtration step were effective in the removal of phytoplankton from the source water of Site 1 as percentage removals ranged from 98.56% to 99.99%. GAC adsorption in particular was effective in the removal of Cyanophyceae, Chlorophyceae and Dinophyceae. Chlorophyceae concentrations decreased

throughout the treatment process. All the algal classes and the total phytoplankton biomass, with the exception of Chlorophyceae, increased after UV light disinfection. This confirms the finding of algal regrowth on the walls of the pipes of the UV system. The corresponding insignificant removal of TPP after UV light disinfection compared to significant removals of TPP after the other treatment steps further served to confirm this finding. Bacillariophyceae, Cryptophyceae and Dinophyceae showed a slight subsequent increase after chlorination disinfection.

Percentage removals ranging from 98.04% to 99.99% as well as the mean concentrations in the potable water (Table 5.7) indicated the effective removal of the problematic algal species namely *Anabaena* sp., *Aulacoseira* sp., other unidentified centric diatoms, *Ceratium* sp., *Cryptomonas* sp., *Microcystis* sp., *Pandorina* sp. and *Oscillatoria* sp. from the source water of Site 1. The coagulation-flocculation, sedimentation, sand filtration and GAC filtration treatment steps significantly reduced *Aulacoseira* sp., other unidentified centric diatoms, *Ceratium* sp. and *Cryptomonas* sp. These treatment steps also reduced *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp. and *Pandorina* sp. effectively although these removals were not significant (Table 5.8). *Ceratium* sp. can disrupt the flocculation process (Ewerts *et al.*, 2013) resulting in the breakthrough of algal cells which can explain that the removal of these species was not perceived to be significant. The removal of Cyanophyceae from the source water of Site 1 was significant even though the removal of Cyanophyceae species was not observed to be significant. This is probably due to the significant correlation that was found between Cyanophyceae and *Ceratium* sp. and the fact that *Ceratium* sp. was significantly removed from the source water of Site 1. *Aulacoseira* sp., *Ceratium* sp., *Cryptomonas* sp., *Microcystis* sp. and *Oscillatoria* sp. increased slightly after UV light disinfection confirming the finding of algal regrowth in the UV system. The increase of *Oscillatoria* sp. corresponded to the increase in geosmin after UV light disinfection.

Algal sampling at Site 2 was conducted in the source water and after pre- and intermediate ozonation. The results therefore did not include the algal composition of the potable water. The algal classes Chlorophyceae, Cryptophyceae, Cyanophyceae, Dinophyceae and Euglenophyceae and the total phytoplankton biomass were reduced throughout the water treatment process except for Bacillariophyceae that increased slightly after pre-ozonation (Table 5.15). Percentages' removal ranging from 96% to 100% indicated that ozonation was effective in the removal of Cryptophyceae, Cyanophyceae, Dinophyceae and Euglenophyceae species from the source water of Site 2. Ozone removed 70.42% of the Chlorophyceae species, 57.94% of the Bacillariophyceae species and 68.57% of the total phytoplankton biomass. Ozonation appeared to be effective in the removal of the problematic algal species *Trachelomonas* sp. but less effective in the removal of *Coelastrum* sp., *Cyclotella* sp., *Pediastrum* sp. and *Scenedesmus* sp. Morrison (2009) and Carrim (2006)

similarly found that ozonation was less effective in the removal of Bacillariophyceae species such as *Cyclotella* sp. due to these species' resistant frustule.

The removal of TPP appeared to be more significant than the removal of phytoplankton throughout the treatment process. The Chlorophyceae algal class was the dominant class in the source water of Site 2 and contributed the most to the TPP in the source water. Upon comparison of the removal of TPP and Chlorophyceae between the source water and the water after ozonation, the percentage removal was similar. As TPP was effectively removed from the source water this was indicative of the effective removal of Chlorophyceae as well as the total phytoplankton biomass from the source water of Site 2. It would appear that the treatment steps following intermediate ozonation, namely sedimentation, sand filtration and chlorination disinfection, contributed significantly to the removal of Chlorophyceae, Bacillariophyceae and the total phytoplankton biomass. It is important to note however that chlorine and ozone oxidise chlorophyll but the algal cells do not necessarily disintegrate. Therefore in order to determine the efficacy of a water treatment plant to remove algae, TPP analysis alone is not sufficient.

5.5.3. Invertebrate characteristics

The majority of invertebrates originate at the treatment plant (Shaddock, 2006). Invertebrates infest sections of the pipeline that provide a supply of food. Food is a limiting factor in the pipeline and bacteria form the basis of the food available. Increases in invertebrate numbers can be also be an indication of insufficient backwashing, problems at filter houses and broken pipes as invertebrates can penetrate the water purification system through these means. In addition, invertebrates can also form breeding populations within the different water treatment steps.

Rand Water's recommended limit for invertebrates in potable water is 20 org/m^3 with a maximum permissible limit of 100 org/m^3 and a crisis limit of 250 org/m^3 (Shaddock, 2006). A more stringent recommended limit of 1 org/m^3 is in place for Diptera due to consumer complaints with a maximum permissible limit of 4 org/m^3 and a crisis limit of 7 org/m^3.

The mean total invertebrate concentration of 171 org/m^3 in the potable water of Site 1 exceeded the maximum permissible limit of 100 org/m^3. The mean total invertebrate concentration of 41 org/m^3 in the potable water of Site 2 was below the maximum permissible limit but exceeded the recommended limit of 20 org/m^3. The recommended limit of 1 org/m^3 for Diptera in potable water of Site 2 was also exceeded by the mean value of 4 org/m^3.

Sampling of the invertebrates of Site 1 was not conducted in the source water but after every treatment step. The secondary water after coagulation-flocculation of Site 1, displayed a similar invertebrate composition as the invertebrate composition of the source water of Site 2 with

Rotatoria and Other groups of invertebrates as the dominant invertebrate groups. The invertebrate concentrations in the secondary water following coagulation-flocculation, sedimentation and sand filtration appeared to be quite high with a mean total invertebrate concentration of 2469 org/m^3. Sand filtration should remove invertebrates from the water as it removes suspended particles not removed through coagulation-flocculation. This is an indication of "dead zones" in the sand beds that provide ideal breeding areas for invertebrates (Shaddock, 2006).

Rotatoria, Other invertebrate groups and the total number of invertebrates were removed effectively from the source water of Site 1 with percentage removals ranging from 93% to 98%. Lower percentage removals were recorded for Copepoda and Ostracoda. The removal of Nematoda and Diptera was ineffective. A spatial trend was observed as some invertebrate groups increased after GAC adsorption while other groups increased after UV light disinfection. Nematoda and Ostracoda increased after GAC adsorption whereas Copepoda increased after UV light disinfection. This suggests that the habitat and food sources in a particular location are conducive towards the formation of breeding colonies of specific groups of invertebrates. A spatial trend in invertebrate abundance was also observed by Ferreira and Du Preez (2012). According to Wang *et al.* (2014), the invertebrate colonisation of GAC filters is a common occurrence in water treatment plants and the mean invertebrate abundance in the GAC filtrate was 8 times that of the invertebrate abundance in the inlet water. An increase in invertebrate abundance after UV light disinfection corresponded to increases in DOC which is indicative of the presence of organic material and bacteria that can be utilised as a food source by invertebrates. It appeared that chlorination disinfection was the most effective in the removal of invertebrates as all the groups except Nematoda displayed a decrease after this treatment step. This was probably due to the removal of bacteria as a food source for invertebrates through the addition of chlorine. Nematodes are highly adaptable and resistant to adverse environmental conditions (Shaddock, 2006).

The invertebrate sampling of Site 2 was conducted in the source water, after pre-ozonation and in the potable water. Cladocera, Nematoda and Ostracoda were completely removed during treatment from the source water of Site 2. A low 59.09% removal was recorded for Unknown invertebrates and Diptera similarly reflected only a 60% removal. Percentages' removal for Copepoda, Rotatoria, Other groups of invertebrates and the total number of invertebrates ranged from 86% to 94.44%. Other groups of invertebrates, Unknown invertebrates and the total number of invertebrates increased after pre-ozonation. These increases corresponded to an increase in DOC. The availability of carbon contributes to the presence of bacteria as a food source and therefore increases the abundance of invertebrates. Cladocera, Copepoda, Diptera and Rotatoria decreased after pre-ozonation. Dissolved oxygen is a limiting factor for these invertebrate groups (Shaddock, 2006) and elevated DOC concentrations deplete oxygen resulting in the reduction of

these invertebrates. As invertebrate sampling was not conducted after the intermediate ozonation treatment step, it could not be established whether this treatment step effected similar changes in the invertebrate composition of Site 2. All the groups decreased between the coagulation-flocculation and chlorination disinfection treatment steps except for Rotatoria that increased again, indicating either a breeding colony or the evasion of the filtration process.

5.6. Conclusions

The efficacy of the advanced treatment processes in combination with conventional treatment processes in use at the two samplings sites. RWB and MWC, was determined by ascertaining the change in the source water quality. RWB uses GAC adsorption and UV light disinfection and MWC uses ozonation as advanced treatment steps.

In general, it was found that the quality of the potable water of both sites complied with the guidelines set for the quality of water intended for domestic use with the exception of the presence of invertebrates. The physical and chemical water quality variables of the potable water of both sites were within the domestic water use limits with the exception of the conductivity of Site 1 that exceeded the limit slightly. The removal of phytoplankton from the source water of both sites was effective.

The following conclusions were drawn in terms of the efficacy of the advanced treatment processes:

- A direct comparison between GAC adsorption and ozonation indicated that GAC adsorption and filtration treatment steps were more effective in the removal of turbidity, DOC and TPP than ozone;
- GAC adsorption was effective in the removal of geosmin from the source water of Site 1;
- GAC adsorption, in combination with conventional treatment steps, was effective in the removal of phytoplankton from the source water of Site 1;
- Ozone was effective in the removal of Cryptophyceae, Cyanophyceae, Dinophyceae and Euglenophyceae from the source water of Site 2 but less effective in the removal of Bacillariophyceae;
- The conventional water treatment processes were not optimised for the removal of invertebrates;
- GAC adsorption and UV light disinfection effected an increase in invertebrate abundance due to a lack of equipment maintenance;
- Ozonation increased DOC and promoted bacterial growth.

The advanced treatment steps aided by the conventional water treatment steps facilitated the production of potable water of an acceptable quality at both sites. There are measures though that can be put into place to augment the change effected in a water quality variable or the percentage removal of a variable.

CHAPTER 6: CONCLUSIONS AND RECOMMENDATIONS

"The situation currently in South Africa is that we have 98% of the water in the country being considered 'fully allocated', this means that my child and your child that is being born tomorrow has 2% of water for use going into the future" (Molewa, 2013). This statement by the Minister of Environmental Affairs encapsulates the extent of the water supply crisis in South Africa. This is further exacerbated by a demanding South African population with an average per capita water consumption of 235 litres per day compared to the world average of 173 litres per day (Hedden & Cilliers, 2014). Furthermore, South Africa's freshwater resources are over-exploited and subject to increasing pollution resulting in deteriorating water quality. Advanced water treatment processes such as granular activated carbon (GAC) adsorption and ultraviolet (UV) light disinfection and ozonation used, in combination with conventional water treatment processes, have the potential to address source water quality problems.

The efficacy of the treatment processes in use at a water treatment plant can only be determined if the quality of the source water is known. The characterisation of the source water provides insight into the physical and chemical characteristics as well as the biological composition of the source water. Changes effected in the source water by the treatment processes indicate the applicability and efficiency of the various treatment processes. The compliance of the produced potable water with a set of guidelines for water intended for domestic use is a pivotal factor in determining the efficacy of the treatment processes. It must be emphasised though that compliance with the guidelines indicate potable water of an acceptable quality and not necessarily potable water of the best attainable quality.

The two sampling sites selected for this study, Rand Water Barrage (RWB) and Midvaal Water Company (MWC), are both located in the Middle Vaal Water Management Area with RWB situated upstream of MWC. The source water characterisation confirmed that the water quality in the Mid-Vaal River system has deteriorated as an increase in eutrophication was evident over the past decade upon comparison with previous studies.

Although the source water of both sites can be characterised as hypertrophic on account of high chlorophyll concentrations, the water quality of the two sites was distinctly different and a downstream change was observed. The source water of RWB was characterised by high microcystin, geosmin, dissolved organic carbon (DOC) and conductivity measurements and dominated by Bacillariophyceae (diatoms) and Cyanophyceae (blue-green bacteria). Problematic species that occurred in the source water of RWB included *Aulacoseira* sp., unidentified centric diatoms, *Pandorina* sp., *Anabaena* sp., *Microcystis* sp., *Oscillatoria* sp., *Cryptomonas* sp., *Ceratium*

sp. and *Trachelomonas* sp. The source water of MWC was characterised by high pH, turbidity and total photosynthetic pigments (TPP) measurements and dominated by Chlorophyceae (green algae) and Bacillariophyceae (diatoms). Problematic algal species that occurred in the source water of MWC included *Cyclotella* sp., *Coelastrum* sp., *Pediastrum* sp. and *Scenedesmus* sp. The source water at MWC was deemed to be of a better quality due to lower Cyanophyceae concentrations and lower microcystin levels. The invertebrate composition of both sites was similar with Rotatoria as the dominant invertebrate group.

The change in or the percentage removal of the physical and chemical water quality variables as well as the percentage removal of phytoplankton and invertebrates were determined to gauge the effectiveness of the water treatment processes in use at the respective sampling sites. RWB uses GAC adsorption and UV light disinfection in addition to conventional water treatment processes and MWC uses pre- and intermediate ozonation. MWC also utilises Dissolved Air Filtration (DAF) to assist with the removal of DOC, suspended solids and algae.

All the physical and chemical water quality variables complied with the standards of water intended for domestic use except for the conductivity at RWB that was slightly elevated. This was caused by the addition of excessive salts during the coagulation-flocculation process. The phytoplankton was removed effectively from the source water of both sites but the removal of invertebrates was unsatisfactory. The number of invertebrates in the potable water of RWB was above the maximum permissible limit and above the recommended limit in the potable water of MWC. It was clear that invertebrates penetrated both plants during the purification processes and established breeding colonies. The sand filtration processes at each site are not optimised for the removal of invertebrates and food sources, most probably in the form of bacteria, are available.

Although TPP and turbidity were effectively removed from the source water of MWC, a direct comparison between the efficacies of GAC adsorption and ozonation indicated that the GAC adsorption and filtration steps were more effective in the removal of these variables than ozone. DOC was reduced to an acceptable level in the potable water of both sites but with considerably lower percentage removals from the source water of MWC. GAC adsorption proved to be more effective than ozone in the removal of DOC. Furthermore, DOC increased after ozonation. Ozonation typically increases the biodegradable DOC and if this not removed in the treatment plant, it can lead to accelerated bacterial growth and regrowth in the distribution system (USEPA, 1999b).

Van der Walt *et al*. (2009) recommend UV light disinfection as the ideal treatment for the inactivation of bacteria and viruses as well as the protozoan parasites, *Giardia* and *Cryptosporidium*. As these variables were not included in this study the effectiveness of UV light disinfection in this regard could not be established. Sigudu (2010) found that UV light disinfection

was effective in the removal of *Giardia* and *Cryptosporidium*. There were indications of algal regrowth on either the walls of the pipes of the UV system or on the lamp sleeves. The UV system at RWB is an in-line system where the UV lamps are placed perpendicular to the water flow and the walls of the pipes and the lamp sleeves are irradiated by visible light only causing the regrowth of algae on the walls of the pipes. Surface regrowth can also be a source of food for invertebrates.

GAC adsorption also appeared to be more effective in the removal of geosmin and microcystin from the source water of RWB than ozone as indicated by the percentages' removal.

It can also be concluded that in order to determine the efficacy of a water treatment plant to remove phytoplankton, TPP analysis should be conducted in conjunction with algal enumeration.

The following recommendations and suggestions for future research are made based on these findings:

- Conductivity: An adjustment to be made to the dosage of the chemicals that are added during the coagulation-flocculation process at RWB to reduce conductivity;
- Invertebrates: Invertebrate sampling is currently not conducted at MWC. A regular sampling regime must be implemented at MWC to monitor invertebrates in the plant. Further investigation is required to establish the entry point as well the nature and location of the food sources. Proper filter bed and filter house maintenance must be conducted at both sites. In particular, control measures should be introduced to reduce invertebrate abundance in the GAC filters at RWB. Wang *et al.* (2014) recommend backwashing with chlorine or drying of the filter beds. The soaking of the filter beds in salt water is recommended as an emergency procedure but it is costly.
- UV light disinfection: The UV system at RWB should be inspected for surface algal regrowth or any surface fouling. If observed, the surface fouling should be manually cleaned according to the manufacturer's procedures. There is scope for future studies to determine the efficacy of UV light disinfection in the inactivation of bacteria and viruses within the RWB treatment plant.
- GAC adsorption: MWC can benefit from adding a GAC adsorption step after intermediate ozonation. GAC adsorption will assist the conventional treatment processes in the removal of excess DOC and thus prevent bacterial growth that can serve as a food source for invertebrates. The addition of a biological activated GAC filter should be investigated for the enhanced removal of DOC. A study of the extent of biofilm growth in the treatment plant can also be considered. A GAC adsorption step will also assist in the removal of microcystin and geosmin.
- Chlorine disinfection: The chlorine residual in both systems may not be enough to prevent algal regrowth and should be investigated.

- pH: The pH of the potable water of both sites was within the recommended limit but can be considered high. Both sites can potentially benefit from lowering the pH more after the initial pH shock. pH shock is necessary to render algal cells immobile and inactivate invertebrates but lowering the pH sufficiently subsequently will result in enhanced DOC removal and chlorination disinfection. According to Polasek (2012), there is a higher trihalomethane formation potential at pH values greater than 8.1 and a higher potential for calcium carbonate scaling in the distribution pipe system at pH above 7.9. There is also an increased risk of the formation of other disinfection by-products at pH levels above 7.8. The optimum pH at which water has a fresh taste is in the range of 7.2 to 7.5.

The use of GAC adsorption and UV light disinfection at RWB and the use of ozone at MWC in combination with the respective conventional treatment processes at each site, proved to be effective in addressing the deteriorating water quality in the Mid-Vaal River system. In addition to addressing the efficacy of advanced treatment processes, this study also addressed potential concerns with the treatment processes. It is also evident from this study that the RWB treatment plant is underutilised and capable of producing potable water of a good quality. In the current scenario of the water supply and demand crisis in South Africa and the advocacy of water re-use, the RWB plant can be utilised to its full potential by Rand Water.

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