Water quality and potential toxicity assessment of desalinated seawater for drinking purposes in the City of Cape Town, South Africa

By

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Abstract

The Western Cape is progressively becoming threatened by resultant water shortages caused by the frequent drought conditions, necessitating the need to explore alternative water supplies through seawater desalination to produce reliable drinking water to meet demand. Desalination involves the removal of dissolved salts from seawater to generate saline free drinking water to meet various human needs. The study investigated the water quality levels and potential toxicity of seawater desalination processes from intake water, to the final treated water intended for drinking, with the purpose of ascertaining its fitness for consumption. The discharge effluent from these plants was also assessed to determine its potential toxicity on the environment using aquatic test organisms. The microbiological and physico-chemical water quality of the raw and final treated water samples of the Strandfontein and Monwabisi desalination plants, Cape Town, South Africa, and their efficiency were investigated. The raw, final treated water and brine effluent of the Strandfontein and Monwabisi desalination plants were analysed for ecotoxicity using the test organisms, namely: marine algae (*Phaeodactylum tricornutum*), marine crustacean (*Artemia franciscana*) and marine bacterium (*Vibrio fischeri*). The monitoring studies were conducted over a 12 months period from December 2018 to November 2019. The raw and treated final water quality from seawater samples were determined and assessed against the South African National Standard (SANS) 241: 2015 limits for drinking water pertaining to microbiological, physical, aesthetic and chemical determinants related to long-term consumption.

The study findings showed trends of highest bacterial counts for *Escherichia coli* (*E. coli*) and enterococci in the raw water from these two desalination plants during the winter period, which may be associated with rainfall periods within the City of Cape Town that flushes faecal contaminants from wastewater effluents into the rivers and ultimately into the sea. Higher trends of *E. coli* in the raw water from Monwabisi were also observed during the summer period which may be associated with increased recreational use of this beach during the hot summer months and favourable temperatures for bacterial growth. Enterococcus and *E. coli* were determined in the raw water from both desalination plants and the t-test results for the bacteria showed a p value > 0.05, thus there was no significant difference for *E. coli* and enterococcus in the raw water samples. Increased heterotrophic plate counts (HPC) of 324 CFU/mL for Monwabisi and 175 CFU/mL for Strandfontein were observed during the summer period in the treated water. The HPC CFU/mL from the two desalination plants was

less than the set standard limit of SANS 241: 2015 of ≤1 000 CFU/mL for treated water. The compliance of HPC by both desalination plants indicates the effectiveness of the reverse osmosis treatment process and the adequacy of the residual chlorine used. Also, highest *E. coli* bacterial populations of 1 CFU/100 mL for Strandfontein and 6 CFU/100 mL for Monwabisi were observed during summer period, which may be associated with proliferation of bacteria during warmer conditions. There was a significant difference $p < 0.001$ in *E. coli* between the raw and treated water for both plants showing treatment efficiency in removal of *E. coli* initially found in the raw water sources, and also indicating absence of faecal pollution in the treated water. Increased bacterial counts of total coliforms (TC) in the treated water from both plants were detected during warmer periods of spring and summer when compared to other periods. High TC counts of 201 CFU/100 mL in Strandfontein could have resulted from localized run-off hard surfaces and ablution facilities at the beach. In Monwabisi, high TC counts of 201 CFU/100 mL were suggested to have emanated from storm-water detention pond located near the plant and had an influence on the presence of these bacteria in the treated water. High significant variation ($p < 0.001$) was observed for pH, total dissolved solids, conductivity, alkalinity, nitrates, and chlorides from the raw and treated water from Strandfontein and Monwabisi desalination plants. A significant reduction to acceptable levels of these parameters from the raw to the treated drinking water samples is regarded as an indication of the effectiveness of treatment process applied at the two desalination plants. These physico-chemical parameters were mostly all compliant with the standard guideline limits throughout the study period. In terms of potential toxicity of the raw and treated water as well as the brine effluent, the raw water samples from both plants showed the least toxicity with the growth inhibition (algae) and mortality (crustacean) test compared to the treated water samples and brine effluent. The treated water and brine effluent showed some toxicity to *P. tricornutum* and *A. franciscana*. The addition of chemicals during the desalination treatment process was suggested to have influenced the detected toxicity on the treated water and the brine effluent. The *V. fischeri* bioluminescence test results for the three matrices (raw, treated and brine water samples) showed some bacterial stimulation, indication of no toxicity presence.

Overall, the results of the study showed that the final treated water product from both plants was of high quality and in compliance to SANS 241: 2015 and depicting limited toxicity against test organisms. Findings suggest that regular water quality monitoring of the desalination plants is an essential component. In conclusion, the desalination technology offers a great benefit in the augmentation of water supplies and narrowing the gap of diminishing freshwater resources.

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List of abbreviations used

AQCs	Analytical Quality Controls
CFU	Colony forming units
DWAF	Department of Water Affairs and Forestry
DWTP	Desalination water treatment plant
EC	Electrical conductivity
E.coli	*Escherichia coli*
ENSO	El Niño Southern Oscillation phenomenon
HPC	Heterotrophic plate count
MPN	Most probable number
MPS	Multi probe system
MUG	4-methyl-umbelliferyl-β-D-glucorinide
MW	Monwabisi samples
ONPG	o-nitrophenyl-β-D-galactopyranoside
m-TEC	Modified thermotolerant *Escherichia coli*
RO	Reverse osmosis
SANS	South African National Standard
SF	Strandfontein samples
SWRO	Seawater reverse osmosis
WET	Whole effluent toxicity
WHO	World Health Organisation

CHAPTER 1: STUDY BACKGROUND

1.1 Introduction

The lack of water resources and the decline of water quality constitute one of the most pressing challenges for municipalities, businesses, farming, domestic usage and the environment worldwide (Wei *et al.*, 2011; Gosling and Arnell, 2016; Damkjaer and Taylor, 2017; Greve *et al*., 2018). South Africa is also confronted with an emergency of inadequate provision of water as a result of a combination of minimal and erratic rainfall, increased evaporation rates, global warming, the El Nino phenomenon and an expanding economy coupled with an increasing society whose topographical requirements for water are not aligned with the distribution and allocation of usable water supplies (Masante *et al*., 2018).

Worldwide, approximately a third of the human population live in countries with water scarcities and future projections predict an increase by a further two thirds by 2025 (Elimelech and Phillip 2011; Ibrahim *et al*., 2017). Future projections of climate models suggest that the trend of increasing water scarcity across the globe will continue as climate change impacts become more severe. These predicted changes will cause more unreliable precipitation patterns, increased evaporation and transpiration rates and higher temperatures, resulting in water resources being more scarce and less reliable (Olsson, 2012).

In the global rankings for exceedingly water deficient regions, South Africa is ranked 30th, as a country with irregularly scattered precipitation and runoff. In the year 2017 for the first time in 113 years, the Western Cape Province experienced the most severe water scarcities and the worst drought since 1904 (Botai *et al*., 2017; Wolski, 2018). The influence of additional climate factors and the effects of drought led to the province being announced as a disaster region in the year 2017 as the drought prompted the City of Cape Town to estimate a “day zero” when surface water storage reservoirs supplying the municipal area reached a low of 13.5 % volume capacity (Botai *et al*., 2017; Parks *et al*., 2019). The increasing demand for water necessitates the development of alternative water resources such as seawater desalination (Ghaffour, 2012; Voutchkov, 2018; Jones *et al*., 2019). The inadequacies in the supply of freshwater resources have led South Africa to evaluate other possible water supplies for future use including seawater desalination (Gude, 2015; Blersch, 2014). The assessment of seawater desalination at a practicability level has been evaluated in numerous coastal cities, including Cape Town (Blersch and Du Plessis, 2017). The City of Cape Town has since installed two seawater desalination plants along the coast to produce potable water as a measure to augment existing water supplies and mitigate the impacts caused by the drought particularly during 2017/2018 period (Petrik *et al*., 2017).

Desalination is defined by Darre and Toor (2018) as the removal of dissolved salts from water to generate potable water of reduced salinity to meet various human needs. Desalination can be in the form of brackish, seawater or contaminated groundwater and requires the application of water treatment processes for water reuse (Pangarkar *et al*., 2011; Blersch, 2014; Yousefi *et al*., 2014; Subramani and Jacangelo, 2015). In South Africa, desalination of seawater, brackish water and groundwater is used and these forms of desalination were reviewed as potential supply sources across numerous key cities, including Cape Town (Blersch and Du Plessis, 2017). Seawater desalination, compared to traditional resources such as freshwater and groundwater has increased climate-resilience and thus it is almost 100 % assured to be readily available throughout the year. However, the drawback is that the elevated reliability comes with great costs in terms of energy requirements and operational costs as well some environmental concerns particularly, the production of increased volumes of highly saline effluent (brine) (Wilder *et al*., 2016; Gude, 2017; Blersch and Du Plessis, 2017).

Source of water and associated physical, chemical and biological properties influences how it should be treated to achieve final acceptable quality for potable water. Water quality is regarded as a set of physical, chemical and biological characteristics of water that must be met in relations to a particular standard, so that the water deemed safe for consumption by the consumer or specific use (Cazares-Mendez and Alcantara-Araujo, 2014; Pule *et al*., 2017). South Africa has developed several acts and guidelines that are in place in order to safeguard water quality (DWAF, 2005). Thus, regular monitoring of drinking water, involves the assessment of the levels of some chemical, physical and microbiological water quality properties in relationship to set standards (USEPA, 2003).

Toxicity testing, commonly referred to as bioassays is an essential tool in the assessment of potential hazards produced by a particular test substance in an organism (Barceló *et al*., 2020). Toxicity testing assays are regularly used to supplement physicochemical and microbiological water quality tests as a means to assess the interaction between the living organisms and the concentration of toxicants in the aquatic environment (Wieczerzak *et al*., 2016; Barceló *et al*., 2020). Numerous bioassays have been developed to assess the potential toxicity of wastewater and drinking water; these bioassays use living organisms such as fish, protozoa, algae, bacteria and others (Harbi *et al.*, 2017, Xu *et al*., 2020). The organisms used to assess toxicity vary in composition, their respective sensitivity and their response to pollutants, thus a battery of assays with representatives of the food chain from the level of consumers, producers and decomposers are commonly used in order to cover a wide range of sensitivities (Hale *et al*., 2019). Commonly, *Phaeodactylum tricornutum*, *Artemia franciscana* and *Vibrio fischeri* are used as test organisms in the assessment of marine environments for the detection of potential toxicity effects due to their sensitivity and response when exposed to toxicants. The three test organisms were used in the present study and represented three trophic levels namely; crustacean, algae and bacteria which is essential to assess the toxicity effects on the ecosystem using organisms which represent various trophic levels.

The occurrence of drought in the Western Cape resulted in the installation along the coastline of Monwabisi and Strandfontein seawater reverse osmosis desalination plants, to serve as alternative water supplies for producing drinking water from seawater to augment the drinking water supplies (Petrik *et al*., 2017; Parks *et al*., 2019). Seawater, like other water source supplies for drinking water, is prone to anthropogenic pollution. This study monitored the quality of drinking water produced by these desalination plants using the reverse osmosis technology over a period of 12 months. In this study, the drinking water quality was monitored in terms of the set national standard (SANS 241: 2015) that specifies guideline requirements in terms of physical, aesthetic, chemical and microbiological parameters. The study also assessed the potential toxicity effects of the water and effluent using marine test organisms from different trophic levels. Water quality screening of drinking water produced by seawater desalination is important to ensure adequate disinfection and efficiency of the treatment process in order to determine the suitability of the water and minimize public health risks.

1.2 Statement of problem

Many cities across the world are challenged with water shortages (Djuma *et al*., 2016; Damania *et al*., 2017). Globally, studies have demonstrated that 40% of the human population encounter critical water shortages; by 2025 this figure is anticipated to escalate to 60% (Schewe *et al*., 2014). Additionally, approximately 4 billion people who amount to 66% of the global population presently encounter a state of severe water scarcity for at least one month per year (Mekonnen and Hoekstra, 2016). Statistics further show that “traditional” water resources such as precipitation, snowmelt and surface runoff retained in lakes, rivers, and aquifers have become inadequate in meeting human requirements particularly in water-scarce regions (Jones *et al*., 2019). As the need for developing alternative ways to produce freshwater continues to increase, seawater desalination is continuing to expand as an alternative water resource supply across the globe (Missimer and Maliva, 2018).

Despite the water supply benefits that come with seawater desalination, there are some disadvantages in terms of environmental impacts. Of particular concern are the potential effects resulting from the wastewater discharges (brine) which are being released back to the marine environment (Panagopoulos and Haralambous, 2020; Elsaid *et al*., 2020). These discharges can pose negative effects on the aquatic ecosystem due to their highly saline nature and coupled with the fact that they may contain numerous chemicals which are added during the treatment operations some of which include; the adjustment of pH, chlorination, coagulation, cleaning of membranes, flocculation, dechlorination and antiscaling (Kress *et al*., 2019; Elsaid *et al*., 2020).

Desalinated water has the potential to be detrimental to human health as the by-products of the chemicals used in the desalination process can filter through to the "treated" water and thus potentially endangering the health of the consumer (WHO, 2007; Sharmila and Darun, 2013; Darwish *et al*., 2013). Pure desalinated water is characterized by unpleasant and undesirable properties that can affect the water distribution system negatively (Birnhack *et al*., 2011). As a result, desalinated water is often subjected to post treatment processes to stabilize and reduce its corrosivity, which may result in the contamination of the treated final water (Birnhack *et al*., 2011). Very few regulatory guidelines exist that are specifically designed to regulate water quality of potable water produced using desalination (Nriagu *et al*., 2016), thus more work needs to be done to have well evolved desalinated potable water specific guidelines.

The assessment of the water quality of the final treated drinking water produced from the seawater desalination process is important and requires the determination of physical, chemical and microbiological determinants in order to monitor whether the final treated water complies with the SANS 241:2015 set for acceptable human consumption. Furthermore, a need exists in order to determine the possible impacts of the seawater desalination and its discharge effluent, by using bio-toxicity testing using a battery of marine test organisms to assess the potential effects of the resulting wastewater discharges and the raw and final treated water. Seawater desalination plants produce highly saline and chemical containing waste effluents which can have negative impacts on the neighbouring environments by deteriorating the quality of the water and sediment quality, harming the normal operations of the marine ecosystem (Sadhwani *et al*., 2005; Dawoud and Al Mulla, 2012; Missimer and Maliva, 2018; Elsaid *et al*., 2020; Panagopoulos and Haralambous, 2020). Importantly, the different contaminants not only have a particular impact (i.e. salt content or complex chemicals), but when these conditions are combined in the water column, their effects may be aggravated as the effluent from desalination plants is a complex effluent comprising of different contaminants (Darling and Côté, 2008; Jones *et al*., 2019). The study sites of this research were Monwabisi and Strandfontein desalination plants in the City of Cape Town. The two plants fall under the False Bay region in Cape Town. Monwabisi and Strandfontein desalination plant were designed to supply about 7 MI/day of drinking water to the municipal area.

1.3 Rationale/ Justification of the study

Drinking water quality assessments play an essential role in supplying and ensuring safe drinking water for the consumers. Coastal environments are prone to anthropogenic pollution from municipal effluents, industrial effluents as well as agricultural run-off and river discharges. These pollution sources can affect the water quality which may in turn result in environmental and potential human health risks. Previous studies have highlighted the importance of continuous monitoring of the quality of coastal waters using sampling and assessment in terms of water quality parameters namely; microbiological, chemical and physical. The City of Cape Town in the year 2017, introduced seawater desalination plants as part of the water resilience plan due to the severe drought the city encountered which led to low dam levels for supplying the municipal region. The impacts of climate change globally have resulted in the rapid decline of freshwater sources and the Western Cape Province is no exception in this challenge.

Assessment of water quality of product water from seawater desalination plants is thus essential to safe guard human health and ensuring that the treatment processes applied are adequate in removing undesired contaminants from the water prior to distribution. Due to the inefficiency of processing high volumes of sewage received daily by the City of Cape Town, sewage is regularly discharged into the ocean daily (Petrik *et al.*, 2017). This sewage comprises of high microbial levels as well as complex chemicals which may pose a risk to the near shore coastal environment, as well as may affect the desalination plants' intake water. Thus, it is important to have a continuous monitoring program for drinking water produced by desalination plants not only for the traditional water quality assessments namely; microbial, chemical and physical, but also for potential toxicity using marine test organisms from different trophic levels. Toxicity tests are regularly used as complementary assays to assess the disinfection adequacy as well as the efficiency of the treatment process to guarantee complete decomposition of potent chemicals in the water. Few regulatory guidelines exist that are designed to monitor quality of desalinated water that reaches the consumer's tap (WHO, 2005).

The unpleasant and undesirable characteristics of pure desalinated which often lead to post treatment processes of treating desalinated water can result in the introduction of by-products of varying chemical characteristics into the treated water thus affecting the overall water quality (Birnhack *et al.*, 2011). The health effects of chemicals added in desalinated water treatment processes have not yet being fully explored, necessitating the need to review current guidelines. Furthermore, the brine effluent may be toxic to marine biota due to its hypersaline nature and may contain chemicals that are used in the treatment process and these have a potential of negatively affecting water quality and aquatic organisms (Missimer and Maliva, 2018). Since the City of Cape Town has ventured extensively into exploring the use of alternative water sources for the metro, there is a need for the municipality to undertake initial research into analysing the necessary aspects relating to the water quality, environmental impacts of abstraction and effluent discharge. In the case of desalination, there is a need to advance understanding of seawater desalination by assessing the water quality of the product water generated using the desalination method and evaluating the potential toxicity effects on the marine test organisms.

1.4 Research questions:

- What is the physicochemical and microbiological water quality of the raw and final treated water of the Strandfontein and Monwabisi desalination plant?
- How effective is the treatment efficiency of the Strandfontein and Monwabisi plant in achieving the set standard by comparing the raw and final product water?
- What is the potential toxicity effect of effluent and the desalinated water on test organism; using marine algae, marine crustacean and marine bacterium?

1.5 Hypothesis/thesis statement

The water quality of the final treated water produced for drinking water from the desalination plants (Strandfontein and Monwabisi in Cape Town) complies with the South African National Standard (241:2015) which regulates the quality of acceptable drinking water and poses no toxicity effects to marine test organisms.

1.6 Aim of the study

The aim of the study was to monitor the water quality and potential toxicity of the seawater desalination process from intake water, to the final treated water intended for drinking in order to determine its fitness for consumption using microbiological, chemical and physicochemical tests and a battery of marine test organisms. The discharge effluent released into the marine environment was also assessed for its quality and potential toxicity impacts on the environment.

Research Objectives were to;

- Assess the physical and chemical water quality of the raw water and final treated water of the Strandfontein and Monwabisi desalination plant.
- Assess the microbiological quality of the raw water and final treated water of the Strandfontein and Monwabisi desalination plant.

- Investigate the treatment efficiency of the Strandfontein and Monwabisi plant by comparing the quality of the raw and final product water, and against the SA drinking water quality standard (SANS 241: 2015)
- To perform ecotoxicological analysis on the raw water, final treated water and brine of the Strandfontein and Monwabisi desalination plant using *marine algae*, *marine crustacean* and *marine bacterium.*

1.7 Scope of the study and limitations

In South Africa, the quality standard of drinking water is governed by a number of acts and regulations which are aimed at safeguarding water quality for human health protection. The scope of the study was to evaluate microbiological, physical and chemical quality of seawater desalination, and check it for compliance of the drinking water produced according to the South African National Standard (SANS 241:2015) which is a standard that specifies the quality of acceptable drinking water, as shown in Table 1.

SANS 241 is used for ensuring safety of public health in relationship to drinking water. This standard is used as a definitive reference on acceptable numerical limits for drinking water quality based on physical, microbiological, aesthetic and chemical water quality. Three sampling sites located in Monwabisi and Strandfontein desalination plants respectively were sampled *i.e.* raw wa b ter, treated water and brine effluent and water quality assessments were carried out. The microbiological, physical and chemical samples were sampled bi-weekly from December 2018 to November 2019. The toxicity tests samples were sampled monthly from February 2019 to August 2019 for a period of seven months.

Marine toxicity bioassays were used as a supplementary tool to assess for any potential toxicity effects of the raw and treated water as well as the brine effluent using marine test organisms since the drinking water was produced using seawater. The test organisms that were chosen for this study are of marine habitat, thus were preferable for assessing desalinated seawater as they are adapted to saline ecosystems compared to other test organisms which are found abundantly in freshwater systems. This was done to mimic their natural habitat in order to eliminate “false potential toxicity effects” due to stress caused by the unfamiliar environmental matrices (seawater). The selection of the ecotoxicological test organisms was based on similar studies which used marine organisms to test saline water. The final water could have been assessed using freshwater organisms however as a shortcoming, that was not evaluated in this study. The three test organisms were selected to represent three trophic levels namely invertebrate, algae and bacteria. This is important for better evaluation of integrative effects on these three trophic levels. Marine organisms are known to be sensitive to detect micro-pollutants in water. The tests were also selected based on test sensitivity for detecting contaminant effects in the water. The effects on parameters such as growth, reproduction and mortality based on specific biochemical endpoints were assessed. Test organisms from different trophic levels have varied sensitivity when exposed to pollutants in water, thus the present study assessed toxic endpoints in marine species such as; *A. franciscana*, *P. tricornutum* and *V. fischeri*.

The limitations of the study included sampling consistency due to the nature of operations of desalination plants which depended on weather conditions. Natural phenomenon like algal blooms causes temporary closure or the plants to be offline and unable to produce desalinated in extreme weather conditions that inhibit the intake of water from the sea. The chemical, physical and microbiological data was sampled twice per week for a year, which was sufficient to accumulate data for all the seasons in order to determine and understand the influence of seasons in water quality. The toxicity bioassays were conducted for seven months. Since toxicity testing was done as a supplementary tool, its limited monitoring time did not affect the study significantly since the prescribed water quality parameters specified in the regulatory standards for drinking water (SANS 241; 2015) were covered. Effects of climatic and environmental variables which may influence the water quality were not considered. Several other determinants as specified by SANS 241; 2015 were not assessed due to limited resources. The effects of climatic and environmental variables did not affect the study since the objective of the study was to determine the water quality of desalinated drinking water hence emphasis on environmental factors did not form part of the study. Few of limited importance determinants were not assessed and most of the determinants were assessed. Particularly determinants which are of concern for desalinated water were evaluated thus this did not significantly compromise the data. Table 1 shows the assessed and not assessed determinants in terms of SANS 241 (2015) and additional assessments not specified on SANS 241 (2015) standard.

Table 1: SANS 241 (2015) Microbiological, Physical and Chemical standard determinants (assessed and not assessed)

Microbiological determinants		Physical determinants		Chemical determinants	
Assessed	Not assessed	Assessed	Not assessed	Assessed	Not assessed
E. coli Total Coliforms Heterotrophic plate count Faecal Coliforms	Protozoan Parasites Somatic coliphages	Turbidity Electrical conductivity pH Colour Total Dissolved Solids	-	Free Chlorine Ammonia (N) Chloride (Cl) Fluoride (F) Manganese(Mn) Nitrate (N) Sodium (Na) Sulphate (SO_4) Aluminium (Al) Arsenic (As) Cyanide (CN-) Iron (Fe) Lead (Pb) Antimony (Sb) Cadmium (Cd) Chromium(Cr) Cobalt (Co) Copper (Cu) Nickel (Ni) Selenium (Se) Vanadium (V) Zinc (Zn)	Mercury (Hg) Dissolved Organic Carbon (DOC) Total Trihalomethanes (THM) Phenols
Additional analyses (not specified on SANS 241: 2015)					
Enterococcus		Alkalinity Total hardness		Calcium (C) Magnesium (Mg) Potassium (K)	

1.8 Ethical considerations

Consent to access the study area was obtained from the City of Cape Town. Project ethical clearance was granted from University of South Africa, College of Agriculture and Environmental Science (letter in appendix A). Work was carried out in compliance with set field and laboratory standard guidelines in an accredited laboratory. Personnel Protective Equipment (PPE) such as gloves, lab coats, masks, boots etc. were used by all personnel handling the sampling and laboratory testing to ensure that all health and safety requirements were adhered to. Disposal of used experimental material was appropriately handled and stored in tightly closed hazardous drums and removed to the hazardous room for safe and suitable disposal.

1.9 Thesis outline

This study is divided into five chapters as follows:

Chapter 1 Introduction- provides the general introduction of the study where the problem statement is described; the researched question asked; rationale of the study, study aim and objectives presented.

Chapter 2 Literature Review- presents the theoretical component that informed the study in addition to presenting the reviewed literature related to the study topic in South Africa and across the Globe.

Chapter 3 Study design and Methodology- describes how the research was conducted detailing the study design, methods and procedures for data collection and analysis in addition to discussing the research design that was used and the methodology that was followed.

Chapter 4 Results and Discussions- focuses on the results and discussion for the study.

Chapter 5 Summary, conclusions and recommendations- provides summary of findings concluding remarks and recommendations of the study.

Chapter 6 References- provides references used in the present study

Appendices

CHAPTER 2: LITERATURE REVIEW

2 Introduction

2.1 Global water problems

Water plays an essential function in the normal functioning and maintenance of human health and sustainable ecosystem development (Sun *et al*., 2016). 71% of the surface of the earth is covered by water and 97% of this is distributed in oceans as salt water (Wimalawansa, 2013). The demand for freshwater resources to cater for population growth, extreme urbanization, industrialization and variable consumption patterns has increased tremendously (Bagatin *et al*., 2014; UNESCO, 2015). Increasing population dynamics and its related forms of consumption variability are placing great pressures on water resources, predominantly freshwater (Roberts, 2010). Subsequently, there is growing need for exploring alternative approaches for enhancing the accessibility of surface water. Seawater desalination is receiving great attention particularly in coastal areas as way of augmenting other water resources (Missimer and Maliva, 2018; Voutchkov, 2018).

In 2019, there were about 15,906 estimated desalination plants in operation, found in 177 countries across the globe (Jones *et al*., 2019; Berenguel-Felices *et al*., 2020). Currently, desalination treatment is operational in approximately 174 countries across the world (Berenguel-Felices *et al*., 2020), and most of these plants are located in countries such as the United States, China, Australia, and other European regions, North Africa and the Middle East. A few desalination plants are located in South America and Africa (Voutchkov, 2018; Jones *et al*., 2019; Pistocchi *et al*., 2020; Ligaray *et al*., 2020).

2.2 The impacts of drought on availability of surface water

Drought is defined by Zhang *et al*. (2018) and Zhao *et al*. (2020) as a lengthy lack or deficit of rainfall that leads to water shortages for some activities or groups. Frequent and recurrent drought events are reportedly posing severe economic, environmental and social adverse effects for South Africa (Zargar *et al*., 2011; Zhang *et al*., 2018). The El Niño Southern Oscillation phenomenon (ENSO) is regarded as the contributing factor towards the inconsistencies and unstable rainfall in South Africa (Meque and Abiodun, 2014). ENSO is more prevalent when the Earth’s atmosphere and the Pacific Ocean react together, with resultant inconsistency of numerous climate and oceanic patterns noted (Holloway *et al*., 2012). The South Pacific Ocean events affect the temperature, wind, pressure and rainfall over South Africa (Holloway *et al*., 2012). Furthermore, Tyson and Preston-Whyte (2000) reported that 30 % of rainfall variability is attributed to El Niño events.

The Western Cape Province recently faced severe drought dating back to 2015 when the Province experienced dry, hot summers and extremely low winter rainfalls, subsequent to that was a moderately dry 2016 followed by record-breaking low rainfall in 2017 as presented in Figure 1 below (Wolski, 2018; Enqvist and Ziervogel, 2019). This drought led to the significant drop of the City's dam levels, and thus the City pursued alternative water supply of seawater by using reverse osmosis (RO) and desalination plants located at several sites across the coastline (Petrik *et al*., 2017). Figure 1 shows Western Cape's dam levels for water stored from July 2015 to July 2019. These dams are used by the City of Cape Town for water storage and are mainly fed by rainfall. As shown in Figure 1, Cape Town's surface water supply system is operated from these six major reservoirs, namely; Theewaterskloof, Voelvlei, Upper and Lower Steenbras, Wemmershoek and Berg River. These dams had the lowest water in 2017 due to the severe drought. The dam levels dropped significantly reaching 25.1 % total storage with the last 10 % being the unusable water normally consisting of silt and other materials, which led to the terminology of "day zero" which was a predicted day that there would be no water supplying the municipal region if the City of Cape Town continued not receiving rain (CoCT, 2019).

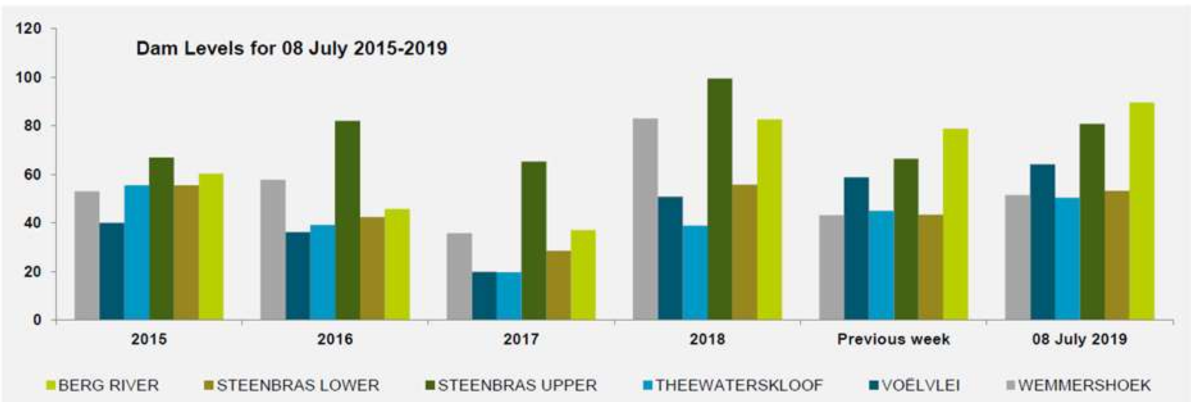

Figure 1: Western Cape's Dam levels during the period July 2015- July 2019 (retrieved from CoCT website, dated 12 July 2019).

2.3 Solutions to water supply issues in water scarce areas

Several approaches have been developed in many countries to mitigate the issues related with water shortages. These approaches include demand mitigation, water conservation, and improvements in technology, controlling the pricing of water and applying water restrictions (Cosgrove *et al*., 2015; Gude, 2017; Ghernaout *et al*., 2019). Other solutions to water shortages include the use of wastewater reclamation and desalination technologies which have been used in many coastal regions as a supply of drinking water (Gude, 2017; Sepehr *et al*., 2017).

2.3.1 Alternative water resources

2.3.1.1 Desalination

Historically, South Africa did not regard desalination as a feasible alternative because desalination plants are expensive to operate and the country only opted for using available surface and groundwater resources that are less costly in meeting the country's water supply demands (Ghaffour *et al*., 2013; Virgili, 2016; Blersch and Du Plessis, 2017). The decline of available surface water resources, declining desalination expenses and advancements in desalination techniques have however, led to seawater desalination receiving extensive recognition as an alternative to closing the gap between water provision, conservation and demand management (Ghaffour *et al*., 2013; Gude, 2016, Missimer and Maliva, 2018; Berenguel-Felices *et al*., 2020). In South Africa, population growth, urbanization, economic growth, climate change and deteriorating water quality are some of the factors that have played a role in putting pressure on surface water resources. It is for this reason that in recent years desalination could be regarded as a viable potential water resource for meeting water demands, due to the rapid technological advancements intertwined with a decrease in production costs, making seawater desalination a competitor with other conventional water resources treatment processes (Blersch, 2014; Ziolkowska *et al.*, 2015; Blersch and Du Plessis, 2017).

Desalination was evaluated at feasibility level in various coastal cities across South Africa including Cape Town, Durban, Port Elizabeth and Saldanha bay (DWA, 2013; Blersch, 2014). Furthermore, it is highlighted in the country's water development reports as a potential future supply source (DWAF, 2008; DWA, 2010; DWA, 2013). Unsurprisingly, numerous coastal cities in South Africa are considering exploiting seawater not only at pilot scale but rather in a large-scale desalination production of potable water using seawater (Blersch, 2014; Jones *et al*., 2019). Seawater desalination is climate resistant and the oceans allow access to the inexhaustible water supply. It offers a consistent, independent source of potable water for many coastal regions. Desalination first became commercially available in the 1960s, and since then it has spread continually and constitutes a major constituent of fresh water supply in many arid countries (Qar and Abdel-Monem, 2014; Voutchkov, 2018; Pistocchi *et al*., 2020; Elsaid *et al*., 2020).

2.3.2 The water quality of desalinated seawater

Some studies done by Duranceau *et al*. (2012), Gacem *et al*. (2012) and Jones (2019) denoted that the water quality variables of desalinated final treated water fluctuates within the distribution system prior to reaching the consumer`s taps irrespective of the desalination technology, design and operational conditions of particular plant used, as well as the subsequent post-treatment processes of the desalinated water. It is of utmost importance to note that water produced by desalination contains low levels of minerals, alkalinity, pH and total organic carbon (Gacem *et al*., 2012). However it is often found to be corrosive towards metal materials used in distribution pipes, storage and plumbing which leads to deterioration of the water distribution systems (Gacem *et al*., 2012). It is for this reason that Gacem *et al*. (2012) suggested matching the treated desalinated water with the existing water supply to mitigate aggressiveness of water and excess gas challenges. The quality and aesthetics of the desalinated water is also influenced by the chemicals components and other materials used during desalination processes (Shomar and Hawari, 2017). In addition, pure desalinated water has an unpleasant and undesirable characteristic, this results in post-treatment activities which are often carried out for stabilization and reduction of corrosivity by applying lime or blending with other water sources (Birnhack *et al*., 2011). These post-treatment efforts can adequately result in products of varying chemical characteristics seeping through to the final water and thus affecting the water quality. Although, Seawater reverse osmosis (SWRO) is regarded as a well-established technology, SWRO lacks reliable and effective standardized universal strategies for water results analysis or results clarification aimed at monitoring and controlling this anthropogenic source of drinking water (Saeed *et al*., 2019). Very few regulatory guidelines have been developed specifically to determine the quality of drinking water produced by desalination (Nriagu *et al*., 2016). Thus, there is a need to advance knowledge and regulatory guidelines that are specific to drinking water generated through seawater desalination. Table 2 shows a set of water quality parameters which were proposed for assessing drinking water produced by SWRO desalination (Saeed *et al*., 2019).

Table 2: Water quality parameters assessment needs for drinking water produced from desalination

Microbiological	**Physical**	**Chemical**	**Disinfectant**
Aerobic microorganisms Cyst Heterotrophic plate count Legionella Total coliforms (faecal coliform and *Escherichia Coli*) *Vibrio cholerae*	Turbidity Colour Alkalinity Hardness Conductivity pH Silt density index Total dissolved solids	Bicarbonate Boron Cadmium Chloride Copper Fluoride Iron Iodide Lead Potassium Sodium Sulfate Zinc Magnesium Nitrite Manganese Mercury Bromate Bromide Haloacetic acids Bromophenol Iodinated trihalomethanes Total Trihalomethanes	Chloramines (as Cl_2) Chlorine dioxide (as ClO_2) Residual chlorine (as Cl)

2.3.2.1 Desalination technologies

Desalination refers to the process of passing water through a particular treatment process to reduce the salinity (Sepehr *et al*., 2017). There are two basic approaches that are used for desalination; thermal and physical separation (Youssef *et al.*, 2014). Thermal treatment refers to the distillation processes where the saline water is boiled to cause a phase change which results in the driving out water vapour from the salt solution (Winter *et al*., 2002). The two commercialised thermal processes include: the multi-stage-flash (MSF) and multi-effect distillation (MED) techniques. These techniques are easy to manage and operate, can treat very salty water and require less pre-treatment, however these thermal methods have higher heat demands, additional electric energy requirements and higher production costs (Al-Karaghouli & Kazmerski, 2013). Physical separation refers to the isolation of the dissolved salts from the saline solution. This is regularly done using a membrane, which can be considered to act as an extremely fine "sieve" (Shatat and Riffat, 2014). There are two membrane based desalination processes used as part of desalination technology, namely: the reverse osmosis (RO) and electrodialysis (Pontius, 2018; Kavitha *et al*., 2019). These methods are highly used because of their lower energy consumption; vast improvement in membrane efficiency (higher recovery ratio of water) and low unit water production costs (Ghaffour *et al*., 2013). Figure 2 illustrates the desalination process involved in the production of desalinated water.

Figure 2: Schematic representation of the desalination process (Adapted from WRC Report No TT 637/15).

2.3.2.2 Reverse osmosis

Reverse Osmosis (RO) is conducted by applying pressure driven membrane process connected to high hydraulic pressure pumps to provide the suitable energy (Kavitha *et al*., 2019). In the process, seawater is pressed against a semi-permeable membrane by reversing the natural osmotic process. The membrane thus permits the flow of water and entraps dissolved salts (Malaeb and Ayoub, 2011). RO is the most widely used process around the world, as it has lower energy and space requirements, and produces drinking water at lower prices (Semiat and Hasson, 2012; Kurihara and Takeuchi, 2018). The price of using RO as a desalination technology has since decreased since the inception of the RO technology in the early 1970s, due to the advancements in membrane efficiency and quality using RO (Takabatake *et al*., 2021). RO is the most utilised desalination technology of choice in South Africa (Semiat and Hasson, 2012). The City of Cape Town has also installed desalination plants across the coast that use reverse osmosis to produce drinking water taking cognisant the advancements of membranes and high recoveries of drinking water from seawater. In water quality, RO has been used due to its effectiveness to remove minerals, volatile organic compounds, fluoride and other chemical pollutants in drinking water sources (Gao *et al*., 2020).

2.4 Impacts of desalination plant on aquatic ecosystem

One of the challenges of desalination is to generate drinking water whilst minimizing the stresses on the marine environment (Panagopoulos and Haralambous, 2020; Elsaid *et al*., 2020). Seawater desalination can pose adverse environmental effects; particularly the possible effects from the wastewater effluents (brine) that is released back into the marine environment which can detrimentally impact the physiochemical and ecological functioning of the receiving environment (Roberts *et al*., 2010; Jones *et al*., 2019). The desalination process discharges a concentrated salt solution as an effluent, with up to double the strength of salinity compared to the original seawater and can have significant adverse effects for marine environment (Del-Pilar-Ruso *et al*., 2015). The increased salinity in the environment can affect some biota as well as marine organisms with varied sensitivity to salinity (Missimer and Maliva, 2018). For example, other organisms are unable to regulate osmosis thus increased salinity causing the water to flow out of their cells leading to cell dehydration and subsequent death due to dessication (Voutchkov, 2018).

Sessile organisms such as plants and corals may also be affected by salinity changes as they are unable to move and escape the high salinities (Laird *et al*., 2017). Additionally, high salinity can affect water chemistry (dissolved oxygen saturation and turbidity) and water column structure (stratification) (Laird *et al*., 2017). The desalination wastewater (brine) effluent is comprised of various chemicals such as biofouling control additives, antiscalants, biocides, neutralized acids and bases used for cleaning the membranes, coagulants and chlorine by-products which are added during desalination pre-treatment and post-treatment processes (Missimer and Maliva, 2018; Kress, 2019).

The discharge of brine effluent in ocean can pose potential negative impacts to marine life (predominantly larvae), which include mortality, fluctuations in seawater quality, effects on fish resources, degradation of marine habitats as a result of the toxic concentrations of brine, anoxic or hypoxic settings, and pressure from turbulent mixing at the discharge point ((Latterman, 2009; El wahab and Hamoda, 2012; Cooley *et al*., 2013; Gong *et al*., 2019). The lack of adequate dilution can result in a highly saline wastewater discharge plume that may spread out for a substantial distance (Van der Merwe *et al*., 2014). Thus, the impact of the plume is dependent on the characteristics of the desalination plant and its brine effluent. Several studies have indicated that the use of diffusers (Del-Pilar-Ruso, 2015) or by-passing seawater (Fernández-Torquemada *et al*., 2009) may assist in the dilution of the effluent and thereby reducing the impacts of the brine in the receiving water.

2.5 Water quality

Water quality refers to a set of biological, physical and chemical characteristics of water that must be met according to a particular standard, as to deem the water safe for use (Cazares-Mendez and Alcantara-Araujo, 2014; Pule *et al*., 2017). The consumption of drinking water contaminated with human or animal waste has the greatest microbial risk to public health (Cabral, 2010; Wen *et al*., 2020). Wastewater effluents in freshwater and coastal seawater are the largest sources of faecal microorganisms, including pathogenic microorganisms (Cabral, 2010; Korajkic *et al*., 2019). The assessment of quality of drinking water in terms of microbiology is determined by screening bacteria of faecal origin often referred to as faecal indicator organisms. *E. coli*, faecal coliforms and enterococcus spp. members are regularly used as hygiene indicator bacteria (Rodrigues and Cuhna, 2017; Rock *et al*., 2019; Offenbaume *et al*., 2020).

2.5.1 Drinking water quality in South Africa

In South Africa, drinking water (also referred to as potable water) is defined in SANS 241: 2015 as water with acceptable quality criteria based on physical, chemical determinants, aesthetic and microbiological properties and should meet the intended purposes of consumption by humans. Most of these properties are characterized by constituents that are either dissolved or suspended in the water (DWAF, 1996). Water quality is regulated by these two guidelines, namely: South African National Standard (SANS) 241: 2015 and the Target Water Quality Requirements (TQWR) (DWAF, 1996). The water quality assurance of drinking water in South Africa is legislated by the SANS 241: 2015. Compliance with the standard is regarded as an acceptable health risk to consumers of water for a lifetime usage, thus implying the water will not pose a significant health risk for an average consumption of 2L of water per day for 70 years for a person weighing 60 kg. The risk categories of determinants include acute health, chronic health, aesthetic and operational (SANS 241:2015).

Drinking water quality by in terms of physical properties is characterised by aesthetic properties, including, taste, odour, and the colour or cloudiness of water (DWAF, 2005). Physical properties themselves are not associated with a direct health effect; however they are able to show potential problems with the water quality (WHO, 2008). Chemical quality of drinking water is characterised by the type and amount of dissolved materials such as metals, salts and organic compounds, which can have adverse effects in health when they exceed acceptable limits (Akter *et al*., 2016; Gutiérrez-Lucas *et al*., 2017). SANS 241 specifies the acceptable ranges and limits of many chemicals which are categorised as macro, micro and organic determinants (SANS 241, 2015). Microbiological quality is characterised by the presence of pathogenic microorganisms in drinking water (Rocks *et al*., 2019; Wen *et al*., 2020).

2.6 Microbiological determinants

Several pathogenic microbes are found in human and other warm-blooded animal faeces (Prinsloo, 2014; Alipour *et al*., 2014; Offenbaume *et al*., 2020; Wen *et al*., 2020). Surface runoff, soil leachate and wastewater discharges lead to the introduction of these faeces into the marine environment (Prinsloo, 2014; Devane *et al*., 2020; McKee and Cruz, 2021). The human health risks associated with these pathogens are related to the uses of the water as well as the amount of the pathogens found in the water (Quattara *et al.*, 2009; Januário *et al*., 2020). Since, seawater is prone to contamination, chemical compounds and pathogens are introduced to the marine environment daily mostly through anthropogenic activities, it is important to assess the raw and treated water produced by seawater desalination plants to ensure that the pathogens are not filtered through to the product water thus posing a potential human health risk for consumers.

2.6.1 Heterotrophic plate count (HPC)

The HPC method refers to the detection of all heterotrophic microbes with the ability to be cultured on a non-selective solid medium under specified conditions (Prinsloo, 2014; Rygala *et al*., 2020). The sources of HPC include various water resources, soil, food, air and plants (Shifat-E-Raihan *et al*., 2017). These heterotrophic bacteria include all the bacteria that utilize organic material as food sources for development. The occurrence of bacteria in drinking water represents one of the greatest human health risks and is associated with diseases such as gastroenteritis, cholera, cryptosporidiosis and giardiasis amongst other diseases (Liu and Liu, 2017). The HPC method is done to measure bacterial colonies in drinking water (Amanidaz *et al*., 2015). The presence of these bacteria may suggest the presence of other opportunistic non-faecal pathogens (Bedada *et al*., 2018). The determination of HPCs is an important tool in water quality assessments as changes in the bacterial community in the water can affect the water aesthetically by influencing the taste, odour and colour through formation of a sticky and slimy layer (Amanidaz *et al*., 2015).

Furthermore, these bacteria can cause corrosion to water distribution systems (Bedada *et al*., 2018). The presence of high numbers of these bacteria may show; a failure in the treatment process of the water, presence of biofilm (Rygala *et al*., 2020), which can thus increase the risk of gastroenteritis (Prévost *et al*., 1998) and poor sanitation conditions (Sarker *et al*., 2019). An increase of HPC bacteria in the treated water compared to the raw water may indicate: post-treatment contamination, growth within the distributed water and biofilms that are present in the distribution system (Payment & Robertson, 2004). Thus, HPC is a good indicator of the efficiency of the water treatment process and sanitation of the distribution system (WHO, 2018; Yi *et al*., 2019). SANS 241: 2015 specifies that good quality drinking water should not exceed HPC bacterial counts of 1 000 CFU/mℓ.

2.6.2 Total coliforms (TC)

Total coliforms refer to a family of microorganisms that are widely used as the indicator of potable water quality (Niyoyitungiye *et al*., 2020). Total coliforms are found in the aquatic environment, soil, vegetation as well as the intestines of humans and some mammals (Cabral, 2010; Seo *et al*., 2019). Coliforms thus serve as indicators of potential faecal contamination in water quality assessments as their presence in water gives an indication of faecal contamination (Wen *et al*., 2020). Thus they are reliably used to evaluate the general sanitary quality of water (Niyoyitungiye *et al*., 2020). Furthermore, some species of the total coliform group have been demonstrated higher resistance to disinfection than *E. coli* and are thus more suitable indicators of poor disinfection (Saingam *et al*., 2020).

2.6.3 *E. coli*

E. *coli* is present in human and animal intestines. *E. coli* in water quality is mainly used to detect for faecal pollution (Odonkor and Ampofo, 2013; Cho *et al*., 2020). *E. coli* belongs to the coliform group; however it is more specifically an indicator of faecal pollution compared to other faecal coliforms (Francy *et al*., 2013; Offenbaume *et al*., 2020), thus *E. coli* is a more reliable indicator of faecal pollution in drinking water (Levy *et al*., 2012; Niyoyitungiye *et al*., 2020; Nowicki *et al*., 2021). The presence of *E. coli* in water is often associated the presence of other faecal pathogenic bacteria and viruses, representing a health risk to humans (Carrillo-Gómez *et al*., 2019). Contaminated drinking water has been documented in many countries around the world as a cause of some illnesses due to the presence of microbial pathogens in water (Carillo-Gomez *et al*., 2019). Thus, *E. coli* is used as an indicator organism to determine the acceptable concentration of faecal contaminants in water (Ibrahim, 2019). In drinking water monitoring *E. coli* is used to assess the efficiency of the disinfection process (Carrillo-Gómez *et al.*, 2019). In drinking water *E. coli* must not be detected as recommended by WHO (2011) and SANS 241 (2015).

2.6.4 Enterococci

The presence of enterococci is a useful indicator for determining the presence of faecal contamination of waters (Alipour *et al*., 2014; D'Ugo *et al*., 2018; Waideman *et al*., 2020). Enterococci belong to the intestinal microbiota group found in humans and animals, however it can also be found in soil and surface water through human and animal faecal matter pollution (Offenbaume *et al*., 2020). Enterococci are able to survive longer in marine environments as they are able thrive in highly salty environments (Harwood *et al*., 2005; Rodrigues and Cunha, 2017). The set limit for enterococci bacterial counts in marine water is less than 100 CFU/100 mL.

The use of enterococcus as a hygiene indicator of water is well documented in previous studies (Alipour *et al*., 2014; Offenbaume *et al*., 2020). Enterococcus has been effectively used as indicators of health risk and is used to assess the water quality of estuarine and marine environments (Rothenheber and Jones, 2018). The presence of enterococcus in water has been correlated with point source contamination with faecal matter in coastal regions (Soller *et al*., 2015; Rothenheber and Jones, 2018). WHO recommends the use of enterococcus as a supplementary tool for assessing drinking water quality (WHO, 2018). The assessment of enterococci in drinking water is important as they are associated with human health risks including; urinary tract infections, wound infections, bacteraemia and pelvic infections amongst other ailments (Arias and Murray, 2012; Waideman *et al*., 2020; Saingam *et al*., 2020).

2.6.5 Blue-flag guideline

The blue flag guideline is a programme set by an international body for the monitoring and management of marine environments. This programme facilitates the integration of sustainable development between freshwater and marine environments. The blue-flag guideline is responsible for ensuring that municipalities achieve standards in four categories of: water quality, environmental management, environmental education and safety. The compliance of beaches is monitored according to the criteria set by the blue flag guideline. The Blue Flag programme aims at ensuring beaches are monitored regularly to achieve good bathing water quality based on the most suitable international; national standards and legislation. Seawater was used as the source/feed water for the production of drinking water using the RO desalination technology at Monwabisi and Strandfontein DWTPs. The raw water from both desalination plants was assessed and checked for compliance against limits set by the blue flag status used for beaches and marinas.

2.7 Physical and aesthetic determinants

A number of physical and aesthetic parameters are used to detect potential problems or to determine the water quality, thus forming an integral component in standard water quality testing (Prinsloo, 2014). Physical water quality determinants, such as electrical conductivity, pH and turbidity are traditionally used; however these influence the aesthetic quality of water but are not associated with any direct health issues (WHO, 2008).

Physical parameters such as pH, conductivity, turbidity, TDS and suspended solids, can affect desalination feed-water and these parameters vary from different regions (Yang *et al*., 2010). These parameters are important for determining the efficiency of the water treatment and were used in the present study to assess the desalination plant's treatment efficiency.

pH - is used for measuring the hydrogen ions, thus indicating the acidity or alkalinity scale of a solution. pH of the water is an important variable as it influences many chemical reactions in water and also affects the treatment process including disinfection (Ibrahim, 2019; Hung *et al*., 2020). The buffering nature of seawater causes the pH of seawater to be fairly stable, and is normally around 8 (Saeed *et al*., 2019).

Electrical conductivity m S/cm (EC) - is used to measure the strength of a solution to pass electrical flow through conductive ion contents. The electrical conductivity is regarded as an important parameter for assessing the clarity of water which is influenced by the nature and amounts of ionized materials that are in the water (Chughtai *et al*., 2014).

Total dissolved solids g/L (TDS) - indicate dissolved content, whether organic or inorganic substances in liquid solution. Regularly, seawater contains an average of approximately 30 000 – 45 000 mg/L of TDS concentration (Qiu and Davies, 2012; Nthunya *et al*., 2018). Previously due to the high costs of the removal of salts in desalination, saline water was not considered for producing drinking water (Yousefi *et al*., 2014). TDS determination in water quality is important as the concentration of TDS affects the palatability of the water (Akoto *et al*., 2017).

Turbidity- measures the light transmitting properties of water. Turbidity measures the clarity of water and its concentration in water is indicative of the amount of residual suspended and colloidal matter (USEPA, 1999). Turbidity is important in SWRO to check for pre-treatment requirements by the desalination plant (Saeed *et al*., 2019).

Total Alkalinity-The alkalinity of water is a measure of its capacity to neutralize acids. Alkalinity is mostly associated with salts of weak acids, although weak or strong base may also contribute. Alkalinity is usually influenced by bicarbonate, carbonate and hydroxide (Boyd *et al*., 2016). Alkalinity determination is also important in water quality assessments since highly alkaline water is normally unpalatable.

Total Hardness- Water hardness is regularly measured as calcium carbonate (CaCO3). Water hardness depends on the presence of some major anions and cations, such as bicarbonate, sulfate, chloride, calcium and magnesium. There are no known effects of water hardness in humans; however, it is an essential parameter for domestic, agricultural and industrial use (Chidya *et al*., 2011; Sharmar and Kumar, 2017).

Colour- colour of drinking water is normally due to the presence of organic matter (Oyedeji *et al,* 2010). There are no health based effects of colour in drinking water, however its assessment is important as the coloration of water can be influenced the presence of irons, some heavy metals and some impurities (WHO, 2011).

It should be noted however that EC, TDS, turbidity, total hardness, colour and total alkalinity are all dependant on the concentration of dissolved salts that are present in seawater. The numerical limits of these parameters are specified in Table 3.

Table 3 : SANS 241: 2015 Physical and aesthetic determinants limits for drinking water

Physical and aesthetic determinants	**Standard limits**
Colour mg/L PI-Co	≤ 15
Conductivity mS/m	≤ 170
Total dissolved solids mg/L	≤ 1200
Turbidity NTU	≤1 Operational ≤ 5 Aesthetic
pH	≥ 5 pH ≤ 9.7

2.8 Chemical determinants

Chemical quality parameters assessments are important in that they influence the microbial quality of water as they supply the microbes with some nutrients (Prinsloo, 2014). Chemical water quality parameters refer to concentration of dissolved or suspended substances in water. Several chemical components ranging from nutrients (nitrates and phosphate), micro and macro determinants including metals serve as indicators of water pollution. Inadequate treatment and waste disposal from humans and livestock and exploitation of limited water resources leads to pollutants such as heavy metals, nitrates and salt entering the water supplies (Nsor *et al.*, 2016; Nayar, 2020). SANS 241: 2015 stipulates the chemical determinants that must be analysed for drinking water, as shown in Table 4.

2.8.1 The presence of nutrients in drinking water

Nutrients such as nitrates and phosphates are commonly found in drinking water. The presence of nitrates and phosphates in drinking water is normally associated with organic pollution from anthropogenic activities such as agricultural runoff (fertilizers, pesticides) livestock farming and effluents from municipal and industrial wastewaters (Batool *et al.*, 2018; Maguvu *et al.*, 2020). High values of these nutrients in drinking water have been linked with various human health risks (Taneja *et al.*, 2019). In the present study nutrient determination of the raw water and treated drinking water included these parameters; nitrates, nitrites, phosphates and sulphates.

2.8.2 The presence of heavy metals in drinking water

Most heavy metals occur naturally in the environment, however anthropogenic activities has been linked with an increase of heavy metals in water sources (Agoro *et al.*, 2020). Other anthropogenic sources of heavy metals in water include; non-point source run-off, untreated domestic and industrial wastewater effluents, accidental chemical spills and precipitation, among others (Anticó *et al.*, 2017). In South Africa and many other countries such as Kenya, China, Iran and United states amongst others, the contamination of marine ecosystems with contaminants such as heavy metals is widely documented (Mekki, and Sayadi, 2017; Nyamukamba *et al.*, 2019; Iloms *et al.*, 2020; Kinuthia *et al.*, 2020). Municipal wastewater is largely known as a source of pollution in numerous aquatic environments (Agoro *et al.*, 2020). Consumption of heavy metal contaminated water can result in infectious diseases such as cancer, acute nausea, central nervous system impairment, reduced growth and development, fetal abnormalities and skin rashes (Chowdhury *et al*., 2016; Aloke *et al*., 2019). The ability of heavy metals to bio-accumulate in tissues can cause detrimental health issues over time and some have carcinogenic and mutagenic properties amongst other health risks (Titilawo *et al*., 2018).

2.8.3 Dissolved salts in drinking water

In humans dissolved salts such as potassium, calcium, magnesium, sodium, and chloride play an essential role in numerous essential cell functions, regulating metabolism, development, cell repair and volume regulation (Akoto *et al*., 2017). The concentration of these dissolved salts in drinking water can affect the aesthetic quality of drinking water. Increased concentration of dissolved salts such as sodium and chloride can impart a taste to the water by making the water taste salty. Dissolved salts; sodium, chloride, potassium, magnesium and calcium were determined in the present study.

2.8.4 Fluoride in drinking water

Fluoride in low concentration in drinking water has been regarded as beneficial for dental health since it provides protection against dental cavities for both children and adults (Akuno *et al*., 2019). However, concentrations of fluoride exceeding the limit can lead to dental fluorosis (tooth discoloration and/or pitting) and more seriously skeletal fluorosis (with adverse changes in bone structure) (WHO, 2011). The sources of fluoride in drinking water include weathering of fluorine rich minerals and anthropogenic sources such as mining, usage of pesticides and brick kilns (Sankhla and Kumar, 2018), thus it is important for drinking water sources to check for compliance of fluoride.

2.8.5 Ammonia in drinking water

Ammonia is another pollutant that is found in drinking water sources, ammonia is found in water through municipal effluent discharges and nitrogenous waste from animals and nitrogen fixation, air deposition and run-off from agricultural lands (Fu *et al*., 2012). Ammonia in drinking water may induce adverse effects on taste, odour and also increase heterotrophic bacteria (Fu *et al*., 2012). Thus, adequate treatment is required to ensure that the aesthetic quality of the treated water is achieved and ammonia concentrations in the drinking water were determined.

2.8.6 Free chlorine used in the disinfection process for water treatment

The disinfection of water includes chlorination which is responsible for the inactivation of microorganisms that cause numerous waterborne diseases and involves maintaining free chlorine concentration in the water to minimize water pollution and subsequent regrowth (Collivignarelli *et al*., 2018). Chlorine is effective in killing most pathogenic bacteria and viruses (WHO, 2008; Abbas, 2011). Chlorine is widely used as a disinfectant due to various advantages which include; it is relatively cheap, efficient, and ease of measurement both on the field and in the laboratories. An additional advantage of chlorine is that it leaves a disinfectant residual that helps to avoid recontamination along the distribution system, including transportation and household storage of water (Abbas, 2011). Free chlorine of the treated water was determined. SANS 241: 2015 recommends chlorine levels of ≤5 mg/L for treated drinking water. Table 4 shows the chemical determinants which were assessed in the present study and their respective numerical limits specified by SANS 241: 2015.

Table 4: SANS 241: 2015 Chemical determinants limits for drinking water

Chemical determinants	Standard limits
Chlorides mg/L	≤ 300
Nitrates mg/L	≤ 11
Nitrites mg/L	≤ 0.9
Sulphates mg/L	≤ 500 Acute health & ≤ 250 Aesthetic
Fluoride mg/L	≤ 1.5
Ammonia mg/L	≤ 1.5
Sodium mg/L	≤ 200
Zinc mg/L	≤ 5
Antimony µg/L	≤ 20
Arsenic µg/L	≤ 10
Barium µg/L	≤ 700
Boron µg/L	≤ 2400
Cadmium µg/L	≤3
Total Chromium µg/L	≤ 50
Copper µg/L	≤ 2000
Cyanide µg/L	≤ 200
Iron µg/L	≤ 2000 Acute health ≤ 300 Aesthetic
Lead µg/	≤ 10
Manganese µg/L	≤ 400 Acute health ≤ 100 Aesthetic
Nickel µg/L	≤ 70
Aluminium µg/L	≤ 300
Selenium µg/L	≤ 40
Uranium µg/L	≤ 30

2.9 Toxicity testing in water quality

Toxicity tests are often used as complementary tools to traditional water quality assessments (Xu *et al*., 2020). Toxicity testing refers to quantification of the toxic effects of a pollutant, which can also be in the form of an environmental matrix (water, sediment etc.) on a living organism (Wadhia and Thomson, 2007; Barceló *et al*., 2020). Conventional approaches for assessing water quality include conducting costly and complex physicochemical parameters methods for measuring the degree of pollution in aquatic environments to acquire information about the water quality of certain water bodies (McKnight *et al*., 2012).

Whole effluent toxicity (WET) testing methods involve the use of test organisms such as *V. fischeri* bioluminescence assay, *P. tricornutum* growth inhibition test and *A. franciscana* mortality assay. In South Africa the Direct Estimation of Ecological Effect Potential system (DEEEP) is used, this is a monitoring tool that uses a suite of ecotoxicological methods to monitor water quality of effluent discharges. These toxicity tests are able to provide an economical alternative for measuring and determining the effects of toxic unidentified pollutants (Carbonell *et al*., 2010; Cruzeiro *et al*., 2017; Xu *et al*., 2020). Toxicity testing was done in this study to measure the integrative effects of the desalinated water and its brine effluent on the three test organisms to assess suitability of the desalinated water and the potential effects of the brine effluent on the receiving marine environment.

2.9.1 Significance of toxicity testing

WET is an integrative approach that incorporates numerous influences such as physical, chemical and biological effects on organisms. These can include parameters such as pH, compound solubility and bioavailability in order to provide the whole picture of the possible effects when everything is taken into account (Libralato *et al.,* 2010; Cruzeiro *et al.,* 2017; Rotini *et al.,* 2017). Thus, WET assays using living organisms are an all-inclusive method that enables toxicity assessments and combine all the possible effects of the components, including possible additive, synergistic and antagonistic effects to be determined (Whadhia and Thomson, 2007; Žaltauskait *et al.,* 2014; Kocbus, 2015). Due to the inadequacy of evaluating all the individual pollutants in complex effluents and their respective toxicity, this necessitates the use of WET, using aquatic organisms such as fish, crustaceans, algae and microorganisms representing different trophic levels (USEPA, 2004; Rotini *et al.,* 2017).

Details on the test organisms used are further described from section 2.9.3 to 2.9.5. These methods are relatively inexpensive and can be used to regulate contaminants commonly not detected by analytical procedures (Cruzeiro *et al.,* 2017).

2.9.2 Types of toxicity tests

Two kinds of toxicity tests exist, which are acute and chronic toxicity tests (Arome and Chinedu, 2013; Libralato *et al*., 2016). Acute toxicity refers to the adverse effects of a test substance (pollutant) over a short-term period (e.g. mortality is used as the measured parameter on the 48 *Daphnia magna* mortality test). A chronic toxicity test refers to the adverse effects of a test substance over long term exposure to a pollutant (e.g. inhibition of normal reproduction and growth is the measured parameter for the 21 day *D. magna* reproduction toxicity test (Gobi *et al*., 2012).

2.9.3 *A. franciscana*

Artemia spp. belongs to the crustacean family and are adapted to extreme conditions including those found in hypersaline lakes (Gajardo and Beardmore, 2013; Veeramani *et al*., 2019). *Artemia* spp. particularly feed on phytoplankton and are an essential major consumer (Triantaphyllidis *et al*., 1998). These organisms are closely related to zooplankton such as Daphnia and copepods and are normally used for toxicity testing assessments as test organisms. *Artemia* spp. including *Artemia salina, A. franciscana, Artemia urmiana* and *Thamnocephalus platyurus* have been used in toxicity testing and are regarded a useful instrument for initial assessment of toxicity (Veni and Pushpanathan, 2014). *Artemia* spp. are widely used as laboratory test organism as a result of their tiny body size and relatively short lifespan as well as its accessibility as dry cysts (Fichet *et al.,* 1998; Lish *et al*., 2019) and their ability to respond to stress caused by contamination by altering their molecular, cellular, and physiological levels (Marigoˊmez *et al*., 2004). Figure 3 shows an image of the marine crustacean, *A. Franciscana* which was used in study.

Figure 3: ***A. franciscana*** **(Adapted from Hamidi** ***et al*****., 2014), Accessed 2020/08/05.**

2.9.4 *P. tricornutum*

Algae are regularly used in aquatic toxicity tests because they can be easily grown in the laboratory and can be exposed to water soluble compounds with ease. Additionally, the sensitivity to respond to numerous pollutants makes this specie an ideal test organism for monitoring aquatic environments (Choi *et al*., 2012). *P. tricornutum* (Bohlin), is a diatom (*Bacillaroficeae*) belonging to the family *Phaeodactilaceae* found in marine water (Butler *et al*., 2020). Diatoms are largely found in marine and terrestrial ecosystems (Costas-Gil *et al*., 2015) where they play an important role in photosynthetic production (Benoiston *et al*., 2017). *P. tricornutum* is an ideal test organisms because of its ease to be cultured in the lab, has valued physiological and genetic characteristics (Soto *et al*., 2005) and its genome has been fully sequenced (Bowler *et al*., 2008; Feijão *et al*., 2020). *P. tricornutum* is often used in ecotoxicological assessments to determine pollution (Liu *et al*., 2019; Feijão *et al*., 2020). Figure 4 shows an image of the marine diatom, *P. tricornutum* which was used in study.

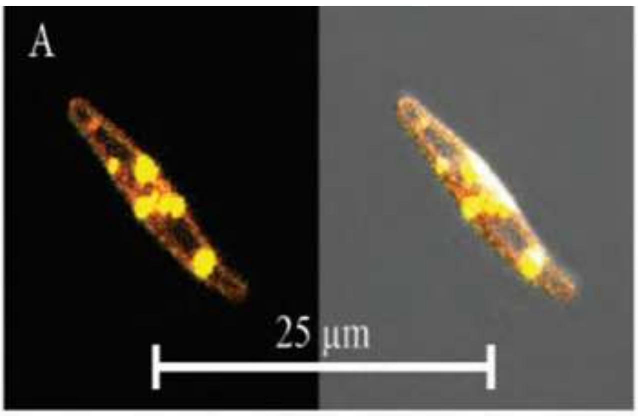

Figure 4: ***P. tricornutum*** **(Adapted from Costas-Gil** ***et al*****., 2015), Accessed: 2020/08/08**

2.9.5 *V. fischeri*

Microorganisms represent an essential constituent of the ecosystem. Toxicity tests using microorganisms are regularly performed as they are relatively rapid, reproducible, cheap, and are not linked with ethical issues (Rotini *et al.,* 2017). *V. fischeri* is a heterotrophic gram-negative, rod shaped bacterium found in marine environments which has bioluminescent properties (Abbas *et al*., 2018; Drzymała and Kalka, 2020). This bacterium is widely used as it can produce results rapidly and with ease, the test organisms are available in freeze-dried form and thus can be readily reconstituted and used, additionally biotox tests have been validated and are widely applied for wastewater and environmental monitoring, thus they can be used for continuous remote initial screening (Faria *et al*., 2004; Abbas *et al*., 2018). Figure 5 shows an image of the marine bacterium, *V. fischeri* which was used in the current study.

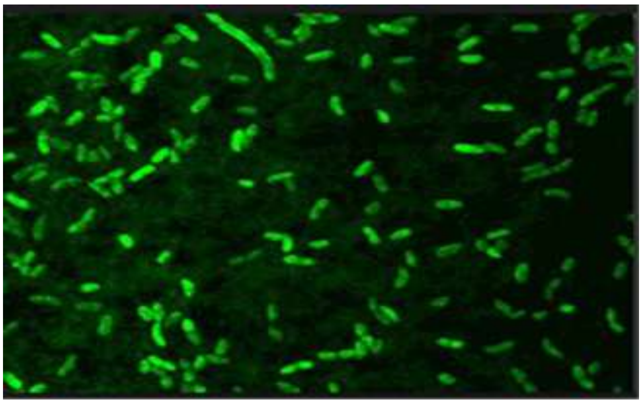

Figure 5: Micrograph of *V. fischeri*

From (https://microbewiki.kenyon.edu/index.php/Vibrio_fischeri Accessed: 2020/08/09)

A similar study in Cape Town was conducted in 2017 by Petrik *et al.* (2017), which monitored water produced by the Victoria and Alfred (V&A) waterfront desalination plant and also evaluated the impact of the nearby marine sewage outfalls. The aim of the study was to assess the sewage pollution affecting seawater and marine organisms in Table Bay, Cape Town and to evaluate their implications for the governance of urban water as well as sewage treatment and desalination. The findings revealed high microbial loads of *E. coli* and enterococcus in the seawater during the study period. Toxicity testing of the seawater and marine organisms was also conducted. The findings showed that the seawater was also polluted with trace amounts of organic chemical pollutants which included; perfluorinated compounds, caffeine, Bisphenol-A (BPA), pharmaceuticals, industrial chemicals and personal care products. Higher levels of these organic pollutants were however found in marine aquatic organisms which included; limpets, mussels, sea urchins, starfish, sea snails, seaweeds and some sediment samples. This was suggested to be a result of bioaccumulation of the pollutants overtime by marine organisms which are constantly exposed to the presence of these chemicals in the seawater.

Additionally, results further showed the presence of various contaminants in the intake water used for desalination closer to the marine outfall in Green Point (Petrik *et al*., 2017). The authors also noted that the water recovered from desalination may still be contaminated with trace amounts of pollutants even after the reverse osmosis process, which was supported by previous research by Patterton, (2013) as well. Therefore, the findings revealed that drinking water produced from seawater desalination may represent a public health risk and further illustrated the need for continuous monitoring to ensure efficiency of the desalination process treatment technology by including toxicity testing together with routine traditional assessments. It further indicated the need of treatment protocols for desalinated seawater with the ability to ensure the complete elimination of bacterial loads and organic chemical compounds. Thus, the findings revealed a need for terms of reference for the intake and recovered water from desalination plants which would specify appropriate testing and monitoring of chemical compounds and microorganisms, including toxicity testing for drinking water produced by seawater desalination plants (Petrik *et al*., 2017).

The present study also demonstrates the need for similar research to bridge the wide gap in information pertaining to drinking water quality assessments of water produced by seawater desalination and toxicity assessments on the receiving environment to which these desalination plants discharge their effluent. Furthermore, it is highlighted that there is a need for development of specific guidelines aimed at monitoring drinking water produced by seawater desalination to ensure the safe delivery of drinking water and protection of the environment.

CHAPTER 3: RESEARCH DESIGN AND METHODOLOGY

3.1 Research design

The study made use of a quantitative, experimental study design. The quantitative research studies were used to determine the cause and effect relationships between studied variables.

3.1.1 **Description of the study site**

The City of Cape Town is located at the Northern tip of Cape Peninsula of the Western Cape Province, South Africa. The study areas of focus in this study were Monwabisi and Strandfontein, which are located in the City of Cape Town represented in Figure 6. Raw water, treated water and plant effluent for the Monwabisi and Strandfontein desalination plants were studied.

Monwabisi and Strandfontein fall under the False Bay region in Cape Town. False Bay marine environment is of particular importance for commercial and maintenance of the fishing industry (fish and shellfish) and the South African Navy (Van der Merwe *et al*., 1991). The False Bay is distinguished by the sandy and rocky beaches famously known as regular tourist attraction areas and are extensively used for recreation and water sports (Taljaard *et al*., 2000). Furthermore, False Bay has numerous areas of conservation importance, with the vast majority being selected and legislated (Mdzeke, 2004).

Strandfontein beach is found approximately 12 km west of the Monwabisi beach along the northern, sandy, wave-exposed perimeter of False Bay. Monwabisi is a beach resort found on the northern wave-exposed perimeter of False Bay. This site is an essential recreational area for the community of Khayelitsha as well as for people who live further inland. Monwabisi offers visitors both a sandy beach and a very popular tidal pool, which was built in 1987 on the mixed rocky/sandy shoreline and a 170m longshore breakwater was built seaward of the existing tidal pool to further enhance safely.

Monwabisi and Strandfontein desalination plant were built to supply about 7 MI/day of drinking water to the City of Cape Town municipality. Impacts of desalination plant effluent on the receiving environment were evaluated. This was done to add onto the information relevant to the potential impacts of the desalination plants to the False Bay marine environment as well as assess the overall water quality of the drinking water produced by these two desalination plants.

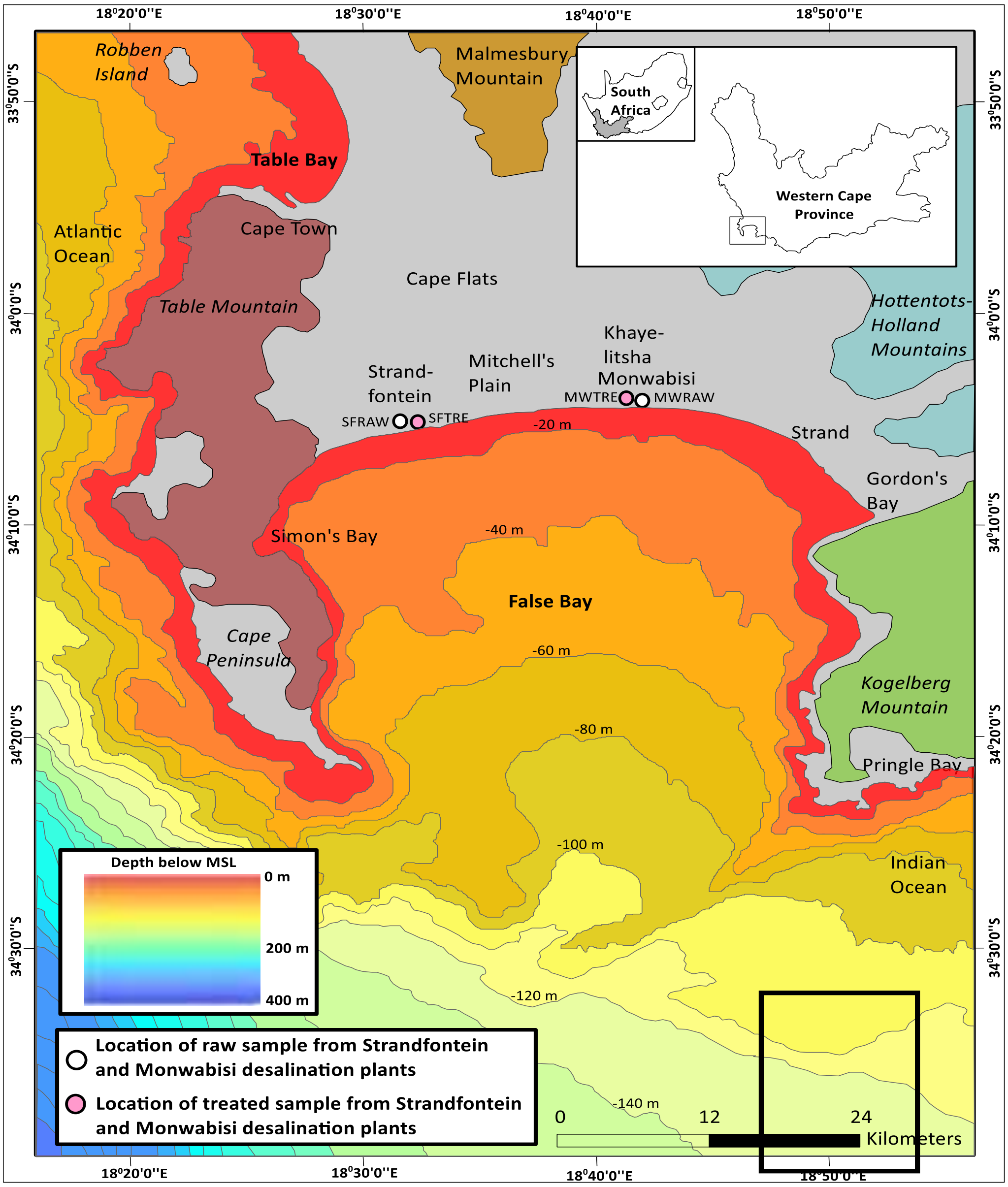

Figure 6: Location of sampling sites in the City of Cape Town.

3.1.1.1 Strandfontein desalination plant

Figure 7: Strandfontein desalination plant (Laird *et al*., 2017)

3.1.1.2 Monwabisi desalination plant

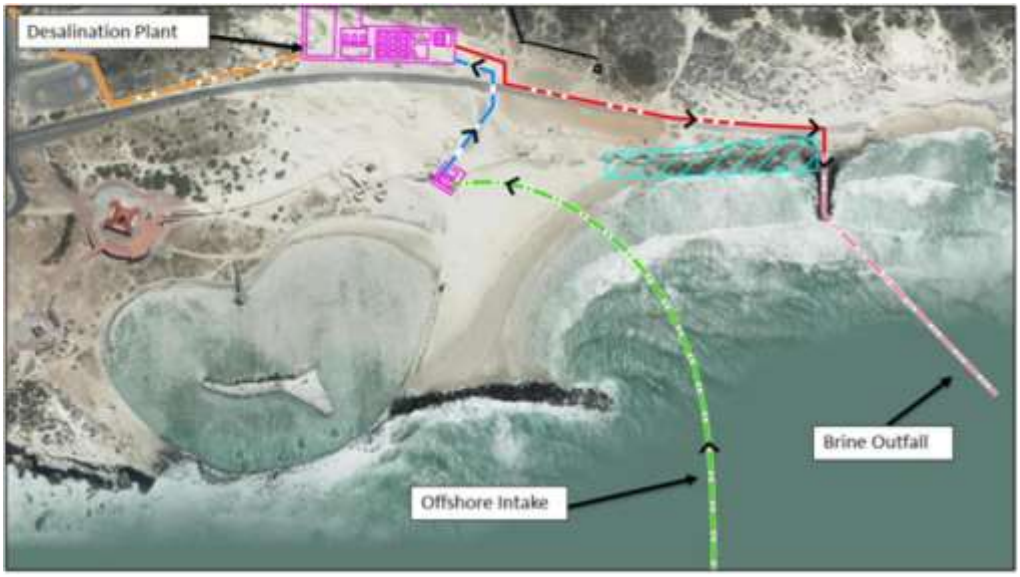

Figure 8: Monwabisi desalination plant (Laird *et al*., 2017)

Figure 7 & 8 shows the layout of the two desalination plants respectively. The two images depict the desalination plants' abstraction points for the raw intake water and the brine effluent outfall discharge points and the treated drinking water.

Table 5: GPS coordinates of sampling points

Sampling site name	Description	Latitude	Longitude
Monwabisi (Khayelitsha)	Monwabisi beach	34°4’4.763”S	18°42’28.974”E
DWTP_MW_RAW	Monwabisi Desalination Plant Raw Sample	34°4’17.753”S	18°52’21.382”E
DWTP_MW_TREATED	Monwabisi Desalination Plant Treated Sample	34°4’18.311”S	18°41’20.068”E
Strandfontein (Mitchell’s Plain)	Strandfontein beach	34°5’17.817”S	18°33’15.668”E
DWTP_SF_RAW	Strandfontein Desalination Plant Raw Sample	34°5’11.202”S	18°33’23.507”E
DWTP_SF_TREATED	Strandfontein Desalination Plant Treated Sample	34°5’11.249”S	18°33’22.582”E

3.2 Data collection

3.2.1 Sampling sites

Samples for microbiological, physical, chemical and toxicological analysis were collected from the following sampling points at Monwabisi and Strandfontein desalination plants.

- Raw incoming (influent) water and Final product water (treated)
- Wastewater discharge (Brine)

3.2.2 Sampling procedure

Water quality samples were collected from the Strandfontein and Monwabisi desalination plants for a period of 12 months starting from December 2018 to November 2019. For the water samples, the grab sampling technique was used. Sterile plastic bottles were used to collect 100 mL water samples for microbiological analysis. For chemical and toxicity analysis, a sample volume of 2 L were collected using 2L plastic bottles for each of the sampling point (raw water, treated water & brine effluent) from the two desalination plants. The sampling was conducted monthly for seven months for toxicity with a total of 6 samples per month. The total number of the samples at the end of the study was 42 composite grab samples for toxicity test results. Microbiological and chemical samples were collected twice on a weekly basis with a total of 8 samples per month. Limited sampling was conducted in December 2018 and March 2019 due to the temporary closure of these two plants as a result extreme weather conditions and an algal bloom on the False Bay catchment. The total number of samples at the end of the study was 83 grab samples for the microbiological, physical and chemical samples. The sample bottles were filled to the top and tightly sealed to avoid leaking and loss of volatiles. The samples were kept in a cooler box with ice packs during transportation refrigerated at 4 °C before commencement with sample analysis.

3.3 Water sample Analyses

3.3.1 Microbiological Analysis

3.3.1.1 Heterotrophic plate count (HPC)

The HPC method (CFU/mL) (Pour plate method) (SANS 5221:2007) also sometimes referred to as the total bacterial count is used as a method for measuring the sum of bacteria present in a sample. The pour plate methodology was used to measure the total bacteria per sample using plate count agar. The water sample (1 ml), was aseptically pipetted into a sterile petri dish and about ± 15 mL standard plate count agar was added. The sample in the petri dish was swirled on the bench top for a few seconds, before the agar solidified. The plates were then placed on the bench until the agar solidified and the plates were inverted and incubated for 48 hours at 35ºC. Following incubation, the colonies on the plates were counted and the CFU/mL were recorded. Two replicates per sampling site and run where performed for the HPC method. The HPC method was only used to assess the water quality of the treated drinking water and it was excluded for the raw water samples. Heterotrophic microorganisms are found in large numbers in raw water sources. The HPC method is not normally done raw samples which would result in overgrowth of colonies thus making plate counting difficult to carry out as there would be too numerous results more frequently (WHO, 2008). Following disinfection with chlorine, HPC numbers are expected to decrease. However, high numbers are often observed in treated water which could indicate; deterioration in sanitation, possible stagnation and the potential growth of biofilms (WHO, 2008). Thus, due to the large amount of distribution points in the City of Cape Town, HPC was only tested on treated potable water as per SANS 241 (2015) requirement.

3.3.1.2 Faecal coliforms

The coliform method (Colilert- simultaneous enumeration of coliforms and *E. coli* using the Idexx Quanti-tray) (PHLS, 1999) is an extensively used test method as the indicator for faecal contamination of water. The Quanti-Tray™ multi-well most probable number (MPN) method uses a substrate medium that contains o-nitrophenyl-β-D-galactopyranoside (ONPG) and 4-methyl-umbelliferyl-β-D-glucorinide (MUG) (PHLS, 1999). Following incubation at 37 ºC for a minimum of 18 hours, a yellow colour is produced due to the formation of β-galactosidase which indicates the presence of faecal coliforms (PHLS, 1999). The reaction of β-glucoronidase causes *E. coli* when present in the sample to fluorescence (PHLS, 1999). Water sample (100 ml) was decanted into a bottle and the substrate was added. The substrate was allowed to dissolve before the sample was dispersed into a multi-well tray and incubated at 37 ºC for 18–22 hours. Following incubation, the numbers of yellow cells were counted (and those that fluoresce under UV light (these represent *E. coli* presence) and these were also counted and recorded.

3.3.1.3 *E. coli*

The test for the presence of *E. coli* was done by the Membrane filtration method using Membrane- thermotolerant *E. coli* agar (modified m-TEC) (Oshiro, 2002). A volume of 100 mL of the water sample was filtered and the filter paper was placed on Modified m-TEC agar plates and incubated for 24 hours ±4 hours. Initially the plates were incubated for 2 hours at 37 °C and then further 22 Hours at 44.5 °C. After incubation all red coloured colonies were counted and recorded. The modified m-TEC method was used for the raw untreated water samples.

3.3.1.4 Detection and enumeration of intestinal enterococci

The presence of enterococci was determined using the membrane filtration method and m-Enterococcus agar) (SANS 7899-2:2004). A volume of 100 mL of the water sample was filtered and the filter paper was placed on m-Enterococcus agar and incubated for 48 hours. After incubation the pink/reddish colonies were counted as enterococci (CFU/100 mL) and the results recorded.

3.4 Physical and chemical water quality parameters analysis

3.4.1 pH and Electrical Conductivity (EC)

pH and EC were measured using the Metrohm TIAMO system.

3.4.2 Turbidity

Turbidity was measured using a HACH ratio Turbidimeter (Nephelometer).

The determination of pH, EC and turbidity included immersing of the electrode for each parameter into the water samples and recording of the measurements.

3.4.3 Total dissolved solids (TDS)

TDS were measured using TDS/EC conversion factor formula. Conductivity in water is made up of dissolved ions; a directly proportional relationship between EC and TDS exist. This relationship allows estimation of TDS using a conversion factor *f* from a measured EC concentration and is commonly used for the determination of TDS (Van Niekerk *et al*., 2014).

The following formula was used TDS = EC. *f*

Where *f = 6.5*

3.4.4 Total Hardness

Total hardness was determined by calculation using measurements of calcium and magnesium. The following formula was used: Hardness was expressed as $CaCO_3$/L = 2.497 [Ca] + 4.118 [Mg].

3.4.5 Total alkalinity

Total alkalinity was measured by titrating the sample with hydrochloric acid. Once the endpoint is reached in the sample, three main forms of alkalinity (bicarbonate, carbonate and hydroxide) have been neutralized. There are two "equivalence points", where pH changes rapidly as the acid is added and these points lie near pH 4.9 and 8.3. These points are then determined by measuring the pH as the acid is added using the TIAMO system electrode.

3.4.6 Metal determination using Inductively Coupled Plasma Optical Emission Spectroscopy (ICP-OES)

The sample preparation included filtering water samples using a 0.45 µm syringe filter into labelled ICP test tubes for analysis. Acidification with concentrated nitric acid was done to the water samples. Acidification was necessary for modification of the sample into a suitable form for ICP aspiration. Following this, the water samples were analysed using the ICP instrument.

The following metals were determined using the ICP instrument: Heavy metals: Aluminium (Al); Chromium (Cr); Antimony (Sb); Cobalt (Co); Arsenic (As) Copper (Cu); Nickel (Ni) Uranium (U); Barium (Ba) Iron (Fe) Selenium (Se); Boron (B) ;Lead (Pb); Zinc (Zn); Cadmium (Cd) and Manganese (Mn).

Light metals: Calcium (Ca); Potassium (K); Magnesium (Mg) and Sodium (Na)

3.4.7 Ammonia determination

Ammonia is measured using the Flow injector analyser (FIA). The method is based on the Berthelot reaction. Sample preparation includes the addition of the following reagents to the beakers; sodium phenolate, sodium hypochlorite solution (3.5%), ammonia buffer, sodium nitroprusside (0.05%), HCl (20%), carrier (sulphuric acid). A reaction between ammonia and phenol takes place, then with sodium hypochlorite to form indophenol blue. To improve test sensitivity Sodium nitroprusside (nitroferricyanide) is added to the sample complex. The reaction product is measured using absorbance and this is directly proportional to the original concentration.

3.4.8 Cyanide (CN) determination

A liquid sample is first mixed with phosphoric acid, heated to 140 °C and then UV treated to break down metal CN and organic complexes. The Hydrogen Cyanide (HCN) (g) from the sample matrix passes a through Teflon membrane and trapped into sodium hydroxide solution. The CN reacts with pyridine – barbituric acid and the absorbance is determined colourmetrically.

3.4.9 Discreet Analyser (DA)

The principle of this instrument is that the sample is incubated with reagents in single cuvettes respectively for as per test method time frames and is then moved through the analyzer where absorbance is measured. Table 5 shows a list of reagents used for each determinant and their concentration ranges which were measured using the discreet analyser.. Chlorides, nitrates, nitrites, sulphates, fluorides, colour were measured using the discreet analyser.

Table 6: Chemical parameters measured using the discreet analyser

Parameter	Reagents
Chlorides	Stock solution – Mercuric thiocyanate Stock solution – Ferric nitrate working solution – colour reagent chloride calibration standards Chloride AQC Standards
Nitrates + Nitrites	Copper sulphate stock solution Zinc sulphate stock solution Sodium hydroxide Hydrazine sulphate (reductant) Sulphanilamide reagent Nitrate calibration standards Nitrate AQC standards
Sulphates	Precipitating solution Sulphate stock solution Sulphate working standards Sulphate calibration standards Sulphate AQC standards
Fluoride	Certified alizarin Certified cerous nitrate Certified acetate buffer Fluoride stock solution Fluoride working standards Fluoride AQC standards
Colour	Certified Potassium chloroplatinate Certified cobaltous chloride Certified platinum-cobalt standard 500 mg/L Pt-Co /color units (CU)

	Hydrochloric acid De-ionised water

3.5 Toxicity bioassays

Toxicity screening bioassays were done for all the grab samples using three test species *A. franciscana*, *P. tricornutum* and *V. fischeri*. Potassium dichromate was used as a positive control (reference toxicant) for all the toxicity screening assays to ensure reliability and accuracy of the data obtained from the toxicity tests. Table 6 provides an overview of the toxicity tests used in the present study.

Table 7: Summary of toxicity testing methods used in this study

Test species	***V. fischeri***	***P. tricornutum***	***A. franciscana***
Toxicity test	**15 and 30 min bioluminescent inhibition**	**72-hour growth inhibition test**	**24 hr acute toxicity test**
Standard method	ISO 1998	ISO10253, 2016	ASTM, 2012 Standard Guide E1440-91.
Test type	Static non-renewal	Static non-renewal	Static non-renewal
Exposure period	15 and 30 minutes	72 hr	24 hr
Test temperature	15 ± 1 °C	20°C (+/- 2°C)	25 °C
Number of test organisms per well	-	10^4 cells/ml	10
Replicate number per sample	3	3	3
Test sample volume	0.5 ml	25 ml	1 ml
Test endpoint	% bioluminescent inhibition or stimulation compared to the unexposed control	% growth inhibition or stimulation compared to the control	% mortality
Measuring equipment	Luminometer	Spectrophotometer	Visual observation under light box

3.5.1.1 *Artemia toxicity screening test principle*

A. franciscana larvae which are hatched from cysts are exposed to test water samples over a 24-hour period in a static test. After 24 hours of exposure % mortality is recorded (ASTM, 2012).

3.5.1.2 Marine algal growth inhibition bioassay principle

The marine algal growth inhibition bioassay involved exposing the marine diatom *P. tricornutum* to test water samples over duration of 72 hours. The effects on the growth or inhibition of the algae are measured by contrasting the algae with the average growth of the unexposed control cultures. Growth and growth inhibition are determined by means of measuring the algal biomass over time (ISO, 2016). The optical density of algal cells was measured daily at the same time from day 0 (before incubation) until day 3 (72 hrs.). Optical density of the cell cultures was measured at 670 nm in a spectrophotometer with 10 cm cell path. The measured growth of the algae in the test substance was then compared to the growth of the control during the same duration and thus the effects on growth inhibition due to exposure to the test substance were calculated. Exposure time of 72 hrs was used because afterwards, algal growth ceases which may due to the depletion of nutrient in the medium and accumulation of toxic by-products (Stevenson, 2014).

3.5.1.3 Vibrio Bioluminescence test principle

V. fischeri which is a light producing bacterium found in warm marine conditions is exposed to test sample, to determine the inhibitory effect of the water samples on the light emission of *V. fischeri* after 15 and 30 minutes of exposure (ISO, 1998).

3.6 Data analysis

Data generated from the three toxicity tests as conducted in the laboratory was recorded on a computer program and the EC50 (concentration causing 50 % effect inhibition or mortality) was determined. Statistical analysis were performed using statistica software to explore whether there was variation of water quality between the raw and treated samples from Monwabisi and Strandfontein desalination plants using microbiological, physical and chemical properties. The application of statistics on the data was done to determine treatment efficiency of the desalination plants in removing some substances/materials in the raw water, to evaluate the pollution load from the raw water to the final treated drinking water and lastly, to check the water quality of the treated drinking water. Significance of variation or differences in the concentration of the determinants was tested by using the independent samples t-test.

3.7 Data validity and reliability

Sampling of the water samples was conducted consistently using standard sampling method by skilled and competent personnel to ensure consistency. Positive controls were also used to ensure the reliability of the results for all the methods. Each analysis per sample was done using two or more replicates (method specific) to ensure accuracy and serve also as method AQCs (Analytical Quality Controls). Water quality parameters were measured using calibrated equipment and the same type of equipment was used during the course of the project. The water samples were analysed in a controlled laboratory environment and analysis of the samples were conducted using recognized internationally standardised methods (SANS 241, 2015; ISO, 1998; ISO 10253, 2016; ASTM, 2012 Standard Guide E1440-91). All this was done as a means of ensuring precision and reliability.

CHAPTER 4 RESULTS AND DISCUSSION

3.8 *E. coli* and enterococcus water quality results for the raw water from SF and MW DWTP.

The source seawater (raw water) samples from Strandfontein and Monwabisi desalination plants were assessed for two microbiological determinants; enterococcus and *E. coli* and checked for compliance against limits set by the blue flag status. The guideline specifies limits of *E. coli* values of 250 CFU/100 mL and 100 CFU/100 mL of enterococci for seawater. While the raw water samples were compared against the set criteria of blue-flag guideline and the treated final water was checked against the national standard SANS 241:2015 which specifies compliance of drinking water. HPC and total coliforms were excluded in the raw water analyses as these parameters are often used as microbiological indicators of final drinking water quality. The abbreviations used in the data presentation and discussion are represented as; MWRAW and SFRAW (Monwabisi and Strandfontein raw water); MWTRE and SFTRE (Monwabisi and Strandfontein treated water). The results presented in the bar graphs used mean values (n=83), error bars representing the standard deviation (SD).

The results for Monwabisi raw water as presented in Figure 9 show that samples had some enterococci counts during most of the study period, with the exception of December 2018. In December, analysis was done only once as compared to other months where analyses were performed bi-weekly with a total of eight samples each month. The reason for limited sampling in December was due to a natural phenomenon of algal blooms in the False Bay region that occurred during the month of December 2018 which led to the temporary closure of the two desalination plants; Monwabisi and Strandfontein. High algal cells can damage the membranes thus the plants were put offline temporarily and unable to produce desalinated water (CoCT, 2018). Often desalination plants are temporarily shut down as a result of extreme weather, for example during periods of high tides and high turbidity of the seawater.

The results for raw water from Monwabisi DWTP with seasonal variations fluctuated throughout the study, however the results that had high enterococcus were observed during July (winter season) with a sample count of 126 CFU/100 mL. The results for Strandfontein raw water presented in Figure 9 had some counts of enterococci throughout the study period. Similarly, to Monwabisi desalination plant raw water; Strandfontein raw water results showed elevated bacterial counts during winter and had enterococcus levels in June with a maximum sample count of 101 CFU/100 ml, compared to other months which had low enterococci bacterial counts. These high results from both plants did not comply with the blue-flag guideline which recommends less than 100 CFU/100 mL for enterococci for seawater. In Monwabisi desalination plant, *E. coli* was found throughout the study period in the raw water. Highest *E. coli* counts in Monwabisi raw water were observed in April 2019 at a maximum single count of 125 CFU/100 mL which is during the autumn season. In Strandfontein desalination plant high *E. coli* values in raw water were observed mostly in June (winter), with highest single count of 100 CFU/100 mL. The presence of enterococci and *E. coli* in the raw water from both plants suggests faecal contamination from the sea into which untreated sewage among other human inputs find way into the water.

Monwabisi beach is an important recreational area for the community of Khayelitsha. In 2013, Khayelitsha Township, a partial informal settlement in Cape Town was stated to encompass a large proportion (38%) of Cape Town's shacks in informal settlement (Seekings, 2013). Informal settlements are largely known for their lack in basic services such as sanitation, pollution, overcrowding and poor waste management, which can influence water quality in neighbouring water sources (Msimang, 2017). Strandfontein found in Mitchell's plain is an area forming part of the Cape Flats. The Cape Flats is a predominantly an urban area comprising of formal and informal settlements (Mauck, 2017). The challenges associated with the density of these areas found close to False Bay catchment include ageing storm-water and sewage infrastructure leading to systems overload and an increase in levels of pollution in the sea, thus introducing pathogenic bacteria like *E. coli* into the seawater (CoCT, 2018). The lack of proper sewage system results in the liquid waste containing faeces polluting rivers, groundwater and marine water environments; furthermore this is a major cause of waterborne diseases (Msimang, 2017). This could explain the abundance of some pathogenic bacteria of faecal origin in the raw water abstracted from Monwabisi and Strandfontein.

The effect of temperature on survival of *E. coli* in water is strain dependent (Abberton *et al*., 2016); therefore, strain type influences the persistence of *E. coli* in the environment than temperature, which may explain the inconsistencies of distribution of *E. coli* in all the seasons throughout the study period. Rainfall periods are known to influence enterococcus and *E. coli* concentrations in raw water and in this study high concentration of enterococcu*s* and *E. coli* were observed in winter. Seasonal changes in temperature and rainfall has been known to affect microbial abundance in water (Prinsloo, 2014). Winter in Cape Town is the rainy period, rainfall may lead to increased microbial levels in surface water and precipitation events flush off the effluent from rivers into the ocean (Prinsloo, 2014; Gil *et al*., 2015; Laureano-Rosario *et al*., 2017). During the dry seasons less enterococci bacterial counts were observed in the raw water from both plants which may be due to the absence of rainfall transportation of faecal contaminants into the seawater via freshwater discharges which is in agreement with the findings of the study by Rothenheber and Jones, (2018). Additionally, high concentrations of *E. coli* were observed for this study in warmer periods similar to the research study by Lamine *et al*. (2019). Furthermore, during summer, trends of higher *E. coli* counts in the raw water from Monwabisi DWTP were observed. Lamine *et al.* (2019) suggested that an increase in touristic activities during summer may constitute a source of potential contamination of the seawater, thus the use of this marine environment by the community of Khayelitsha particularly in summer may have constituted as a source of pollution for the *E. coli* and enterococci present in the raw water from Monwabisi DWTP. Lastly, the current study showed trends of higher enterococci counts than *E. coli* counts in the raw water from both plants. Enterococci have been shown to demonstrate higher resistance to environmental stress and occur in abundance and are a larger group compared to *E. coli***,** thus these might have influenced the enterococci concentrations detected in the current study, which has been shown by other studies as well (Laureano-Rosario *et al*., 2017; Saingam *et al*., 2020).

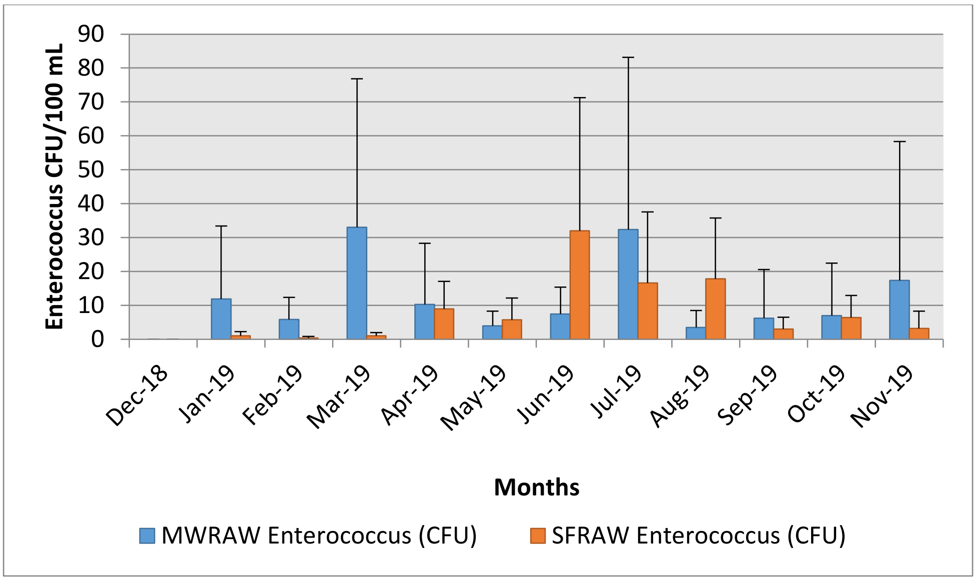

Figure 9: Enterococcus concentrations in the raw water from MW and SF DWTPs.

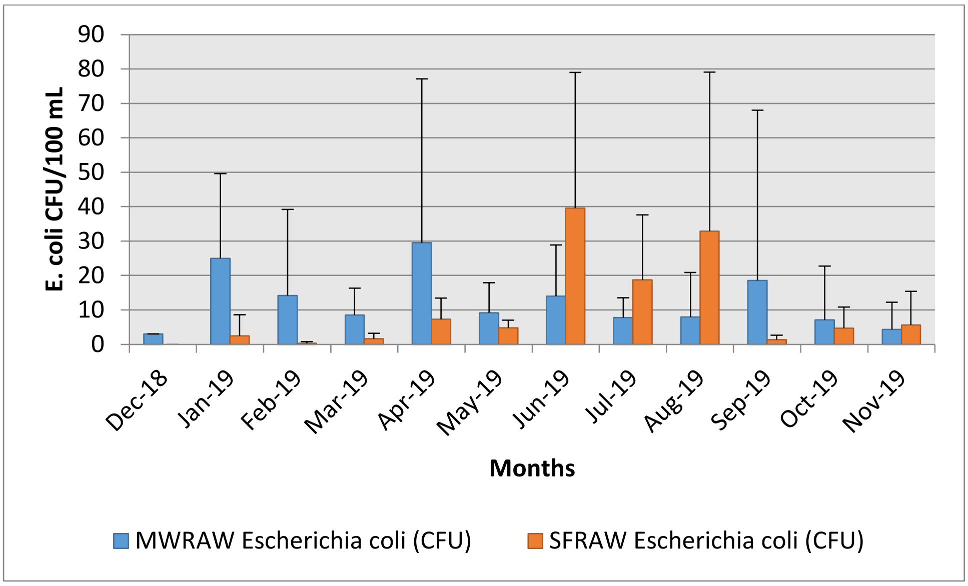

Figure 10: *E. coli* concentrations in the raw water from MW and SF DWTPs.

3.9 Microbiological water quality results for the treated water from Strandfontein and Monwabisi Desalination plants

The treated water from Strandfontein and Monwabisi desalination plants were assessed for three microbiological indicators namely; total coliforms, *E. coli* and HPC, results are shown in Figures 11, 12 & 13. These are microbiological indicators required by SANS 241:2015 for ensuring safe drinking water.

3.9.1 HPC in treated water from Strandfontein and Monwabisi desalination plants

In the present study the microbiological determinant HPC CFU/mL was assessed for the treated final water from the two desalination plants. The results were compared to that of the drinking water standard SANS 241:2015 to assess for compliance.

The results for the treated water from Monwabisi showed that HPC counts were found in the drinking water throughout the study period. An increase in HPC bacteria in September 2019 with a single maximum count of 324 CFU/mL HPC counts was observed in the treated water from Monwabisi DWTP. Highest HPC results were observed in March 2019 with a maximum HPC count of 175 CFU/mL for the treated water from Strandfontein DWTP. The present study showed trends of increased HPC counts in the warmer seasons, spring and summer for Monwabisi and Strandfontein DWTPs. In the summer months after conventional treatment, it is regularly observed that the HPC populations increase from the regular 100 CFU/mL in the distribution system to values ranging from 500 – 1000 CFU/mL (Shifat-E-Raihan *et al*., 2017) of which the current study showed similar findings. However, although HPC bacterial counts were found in the treated water from both plants, these results were still compliant with the standard as they were less than 1000 CFU/mL. HPCs in water are influenced by temperature, residence time, availability of residual disinfection and availability of organic molecules as food sources (Blokker *et al*., 2016). Increased levels of HPC have been correlated with seasonal changes, increasing during hot summer periods and during rainy season as shown by previous studies (Shifat-E-Raihan *et al*., 2017; Shakoor *et al*., 2018), including the present study.

The oceans in the Western Cape are prone to a diverse range of pressures (Pfaff *et al*., 2019). The False Bay catchment where Monwabisi and Strandfontein are found is no exception as it is also affected by human influences including; municipal discharges via rivers, fertilizers, faulty sewer pipes, leaking water mains, contaminated storm-water and groundwater, and natural organic matter which all constitute as potential nutrient sources load to the bay (Pfaff *et al*., 2019). These factors may have played a role in the presence of HPC bacteria in Monwabisi and Strandfontein treated water which influenced the water quality fluctuations for these two desalination plants as bacterial and nutrient loading were identified as main threats to the water quality in the False Bay Catchment which Monwabisi and Strandfontein fall under. Desalinated water is no exception to challenges for desalination plants in maintaining the microbial water quality during storage and distribution (WHO, 2003). High HPCs in the distribution system is normally linked with ineffective disinfection processes (Mokhosi and Dzwairo, 2015). The acceptable levels and compliance of these two plants with all the HPC results is indicative of efficient water treatment process and shows the adequacy of the residual disinfectant used that was added during water treatment.

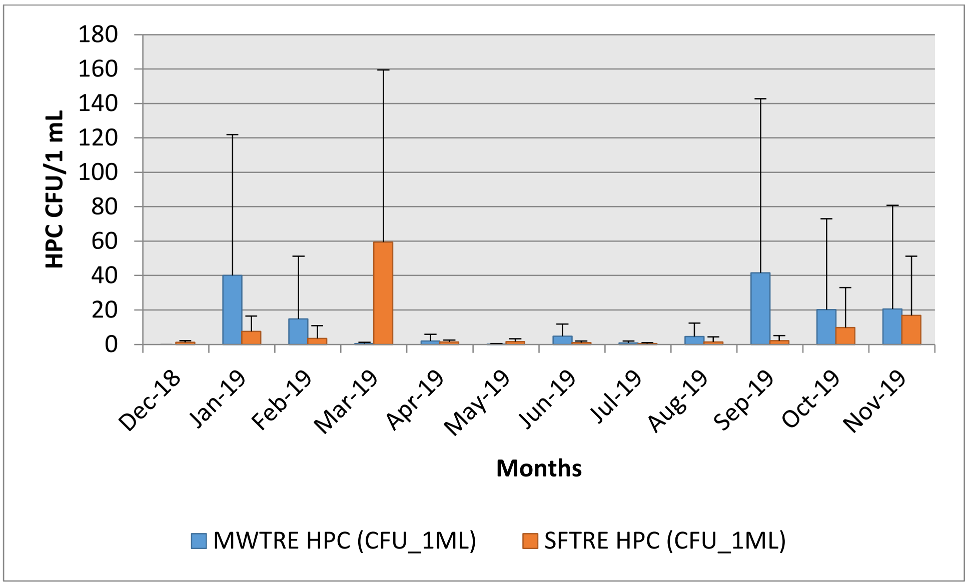

Figure 11: Levels of HPC in the treated water from MW and SF DWTPs.

3.9.2 *E. coli* in treated water from Strandfontein and Monwabisi desalination plants

The results for *E. coli* in the final drinking water from Strandfontein plant presented in Figure 12 mostly complied with the SANS 241: 2015, DWAF (1996) and WHO (2011), which specifies that for drinking water *E. coli* must not be detected, as presented in Figure 12 no *E. coli* was detected in the treated water, except in October 2019 where *E. coli* count of 1

CFU/100 mL was detected. *E. coli* results for the treated water from Monwabisi complied with the standard except for two occasions where *E. coli c*ounts of 6 CFU/100 mL in January 2019 and 2 CFU/100 mL in September 2019 were detected as shown in Figure 12, exceeding the SANS 241: 2015 limit specifying that *E. coli* must not be detected in drinking water.

The presence of *E. coli* in drinking water from Monwabisi and Strandfontein DWTPs suggests faecal contamination and as recommended by WHO requires further investigation on the potential sources; such as inadequate treatment as well as request for subsequent sampling to determine whether the pollution is persistent. Resamples, as per standard operating procedures for non-compliant *E. coli* results in drinking water were done for these three non-compliant results and the resample results showed that no *E. coli* was detected. Since the resamples of the initially non-compliant results passed and detected no *E. coli*, this may suggest possible contamination of the sample tap/ point since these are known factors that can influence the presence of *E. coli* (Gizaw *et al*., 2018). Other studies have shown that the presence of *E. coli* in drinking water can be influenced by poor neighborhood sanitation and hygiene practices around water sources and failure to protect water sources (Gwimbi *et al*., 2019). This may be the cause of the two non-compliant *E. coli* results in the treated drinking water from Monwabisi and Strandfontein DWTPs, since these areas have a large number of informal settlements and thus the presence of *E. coli* in the drinking water may be due to poor neighborhood sanitation.

The results on the treated drinking water from both plants show that *E. coli* was detected during the warmer months as higher temperatures are commonly known to enhance *E. coli* growth (Petersen and Hubbart, 2020). The compliance of most of the results points to the good quality of the final treated water and the effectiveness of the treatment process of the plants, particularly the disinfection and absence of faecal pollution on the treated water.

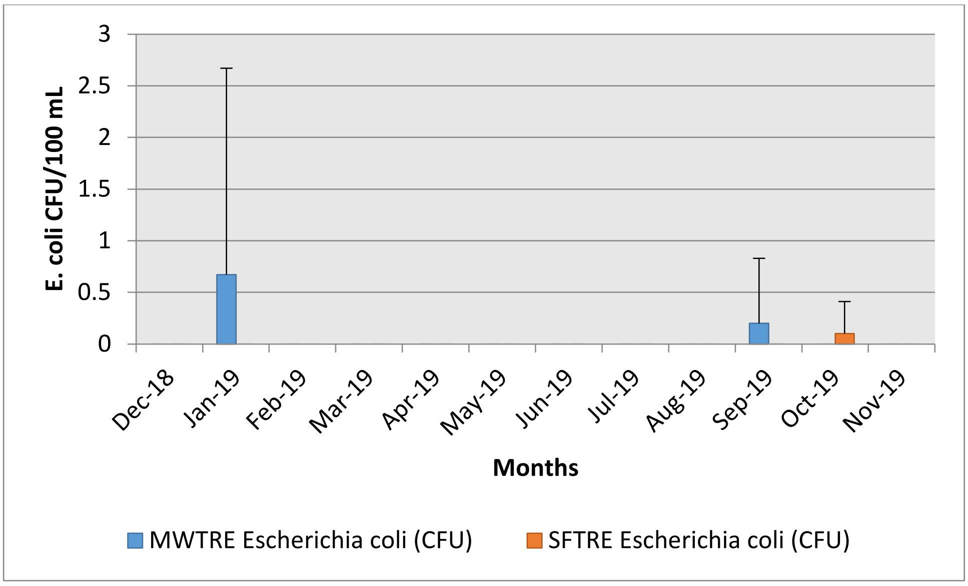

Figure 12: Levels of *E. coli* in the treated water from MW and SF DWTPs.

3.9.3 TC in treated water from Strandfontein and Monwabisi desalination plants

In water quality monitoring, total coliforms (TCs) are referred to as "indicator organisms" as their presence may indicate the presence of other disease causing organisms (Offenbaume *et al*., 2020). SANS 241:2015 recommends TC counts of ≤ 10 CFU/100 mL.

The treated water from Monwabisi desalination plant had seven TC counts that exceeded the limit of ≤ 10 CFU/100 mL TC, with maximum single counts of 201 CFU/100 mL for (January/September/October); in February/November non-compliant TC counts were detected once with a single count of 24 CFU/100 mL counts respectively; in July TC exceeding the limit were detected once with a count of 12 CFU/100 mL. The treated water from Strandfontein desalination plant had four TC counts that exceeded the limit of ≤ 10 CFU/100 mL TC with maximum count of 201 CFU/100 mL for the month of September; in November a maximum single count of 24 CFU/100 mL was detected and in October one non-compliant TC value of 165 CFU/100 mL was detected. Similar to the non-compliant TC values in Monwabisi, Strandfontein maximum counts were also recorded during spring and summer which are the warmer months with increased temperatures that support total coliforms proliferation than in winter months (Yadav *et al*., 2019). TC in drinking water has been shown to survive even when there is adequate free chlorine in the treated water, which may be due to the ability of these organisms to withstand chlorination, leading to their abundance and persistence in the environment even prior treatment (Fakayode and Ogunjobi, 2018; Waideman *et al*., 2020), these findings are similar to the current study as well which showed TC counts even though adequate chlorine levels were found in the treated water from both plants.

The increasing human population in the Western Cape has been associated with anthropogenic challenges on coastal environments including; pollution and eutrophication, and the False Bay region where Monwabisi and Strandfontein lie in the coast is also prone to these challenges (Pfaff *et al*., 2019). In a 2020 report for the City's' coastline water quality, Strandfontein water quality was rated as poor. In Strandfontein, there is no storm-water drainage or direct effluent discharges to the tidal pool where the desalination water is abstracted. The poor water quality at Strandfontein therefore is suggested to be associated with localized run-off from hard surfaces and ablution facilities adjacent to the pool and the recreational use of the pool particularly during periods of high usage, and lastly, the great number of birds that roost on the tidal pool wall may also be contributing factors that influenced the poor water quality rating score at Strandfontein (CoCT, 2019). Strandfontein tidal pool comprises of a semi-enclosed waterbody, tidal pools are often incapable of adequate dispersion properties and are thus as a result prone to numerous pollutant loads (CoCT, 2019). All this factors may explain the presence of high TC counts which were found in the treated water from Strandfontein DWTP.

The 2020 water quality report for Monwabisi (CoCT, 2019) showed that the water at this beach was rated as poor water quality as well. Water quality at this beach has been rated "poor" for the past four years (CoCT, 2019). Monwabisi beach experiences spikes in bacterial counts and not necessarily persistent high bacterial counts. The sudden rise in bacterial counts at this beach is suggested to be due to a storm-water detention pond of poor water quality that is situated East of Monwabisi (CoCT, 2019). This pond water is known to comprise of the highest bacterial counts in Cape Town, worsening during precipitation events as it contaminants the shoreline in Monwabisi (CoCT, 2019). The effects of the recreational use of these facilities during the hotter season may have also influenced the higher TC counts during warmer season as all non-compliances in both were detected in during spring and summer seasons.

The findings of the current study also showed TC counts which were more than *E. coli* counts in the treated water from both desalination plants. It is often observed that TC concentration in water is much more than *E. coli*. Mahmud *et al.* (2019) evaluated the concentration of TC and *E. coli* in drinking water; his findings showed results of TC more than *E. coli* in drinking water which is similar to other research studies including the present study. Also TC are a large group comprising of coliforms, faecal coliforms and *E. coli* (Niyoyitungiye *et al*., 2020), which may explain the higher counts of TCs compared to *E. coli* found in the treated water from both plants. Since TC are a broad group of bacteria found in soil, decaying vegetables, water and faeces, their presence in water does not always represent a threat to health but could point to a problem within treatment operations or a breach in the distribution system (Ellis *et al*., 2018). Furthermore, it is well documented in literature that TC can survive in water for longer periods than *E. coli*, which may also explain why there were more TC counts than *E. coli* counts in the current study (Mahmud *et al*., 2019).

The bacterial fluctuations observed in the current study for *E. coli*, HPC, total coliforms and enterococcus might be influenced by irregular ocean high tides which caused sudden spikes in bacterial counts and some chemical determinants such as TDS, EC, and turbidity amongst other parameters which were detected in both plants. The increased counts associated with such spikes may be responsible for the observed varied water quality fluctuations and high standard deviations between the bacterial counts in samples amongst other factors which contributed to the contamination of these two coastal environments including neighboring conditions, treatment processes, environmental and climatic conditions.

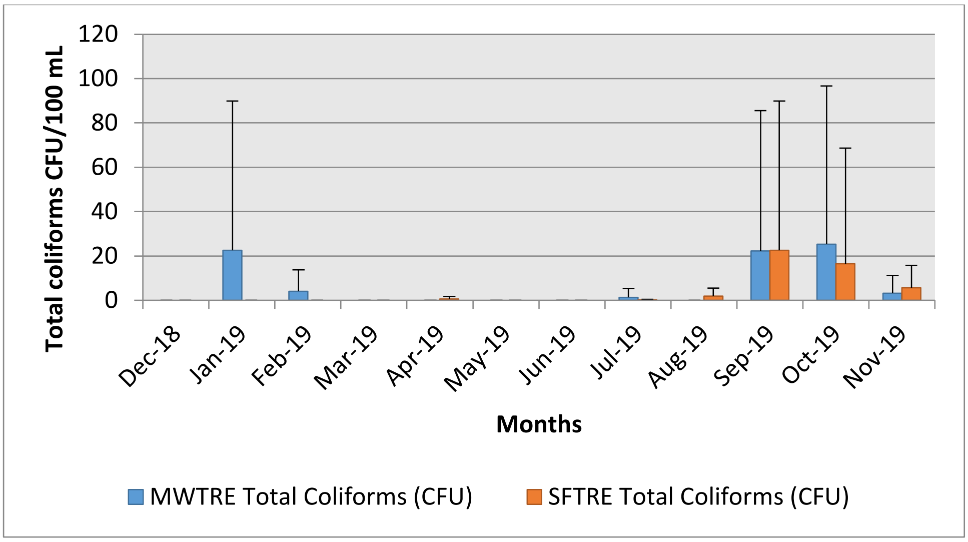

Figure 13: Levels of total coliforms in the treated water from MW and SF DWTPs.

3.10 Physical and aesthetic properties of the raw and treated drinking water

3.10.1 Electrical conductivity (EC) levels of the raw and treated drinking water from Strandfontein and Monwabisi DWTPs.

The EC for the raw water from Strandfontein desalination plant during the course of the study ranged from 4521 mS/m to 5260 mS/m. The EC for the treated water respectively ranged from 41.8 mS/m to 223.2 mS/m with the highest value of 223.2 mS/m observed in summer exceeding the compliance limit recommended by SANS 241: 2015 limit of ≤ 170 mS/m for the treated water. Untreated water is associated with high EC values as seen in Figure 14, due to the presence of high dissolved solids associated with raw water. The results from Strandfontein for EC of treated water were within the standard limit value except one result recorded in January which was 223.2 mS/m. Highest mean EC value of 110 ± 56 mS/m for the treated water were recorded in January 2019 which is during summer in Cape Town. The increased temperature in summer may have influenced the increase in EC as higher temperatures facilitate the movement of ions in water thus resulting in increased EC and TDS (Shrestha *et al*., 2017).

The EC for the raw water from Monwabisi desalination plant during the course of the study ranged from minimum of 4444 mS/m to a maximum of 5179 mS/m. The EC for the treated water from Monwabisi ranged from 22 to 88.9 mS/m. The results from Monwabisi for EC were within the value recommended by SANS 241: 2015 limit of ≤ 170 mS/m for the treated water. Highest EC mean value of 79.63 ± 4.74 mS/m were recorded in October which is during the spring season.

In the present study low EC values of the treated water were observed during autumn and the winter season coinciding with a previous similar study (Akpe *et al*., 2018) which showed that in rainy season the EC values were low due to the dilution effect caused by rainfall. The results in Figure 15 demonstrate that there was a significant decrease of EC from the raw water and final treated water for both the plants. Low EC in drinking water is a characteristic of good quality of water. The reverse osmosis technology used at these desalination plants removes solids, turbidity, colloidal matters, and others, and thus it gives lowest conductivity value following treatment. The variations in EC in drinking water are suggested to be influenced by numerous factors such as agricultural and industrial activities and other land uses, which affect the quantity of dissolved material and subsequently the electrical conductivity of the water after treatment (Rahmanian *et al*., 2015). Conductivity is not known to a threat on human health, it is normally screened for purposes such as determining the mineralization rate of minerals such as potassium, calcium and sodium and for estimating the quantity of chemical reagents required to treat such water (Khan *et al*., 2013). High EC values may lead to lowering the aesthetic value of the water by giving mineral taste to the water (Rahmanian *et al*., 2015). The conductivity of the treated drinking water from both plants can be considered aesthetically good with increased palatability since low dissolved salts in drinking water increase the palatability of water. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in EC concentrations between the raw and treated samples from both plants existed. T-test results showed significant differences for EC concentration between the raw and treated water ($p < 0.001$).

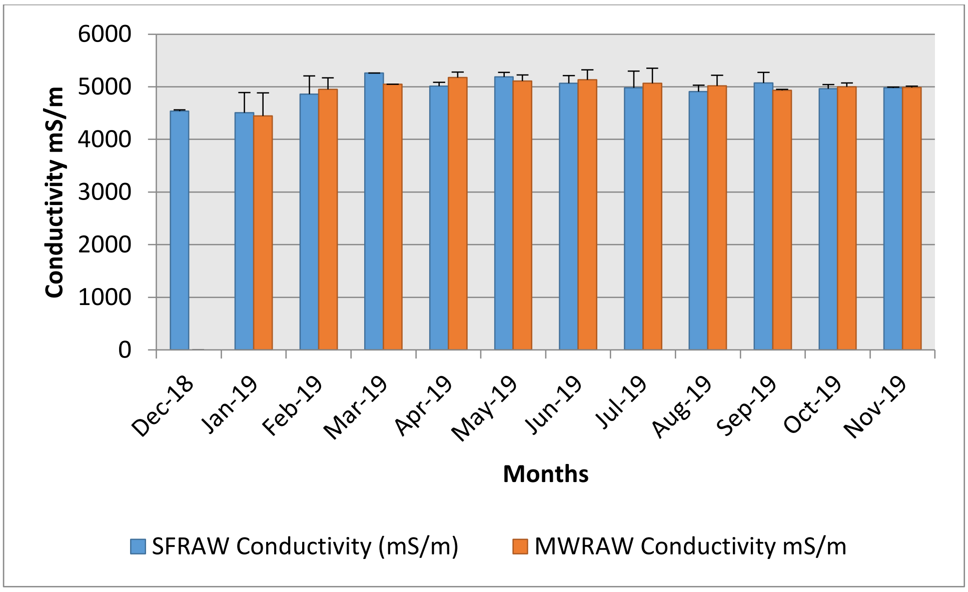

Figure 14: Conductivity (mS/m) levels for raw water from MW and SF DWTPs.

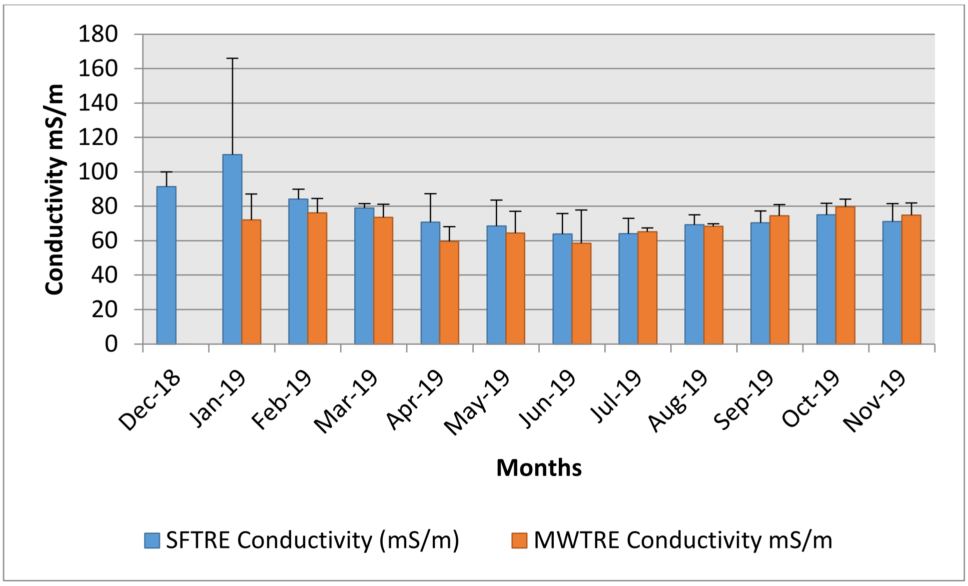

Figure 15: Conductivity (mS/m) levels for the treated water from MW and SF DWTPs.

3.10.2 Total dissolved solids (TDS) of the raw and treated drinking water from Strandfontein and Monwabisi DWTPs.

The TDS of the raw water from Strandfontein desalination plant during the course of the study ranged from of 29369.5 to 34190 mg/L. The TDS for the treated water from Strandfontein respectively ranged from 395.32 to 686.61 mg/L. SANS 241: 2015 recommends TDS values of ≤ 1200 mg/L for drinking water. Highest TDS mean value of 686.61 ± 288.79 mg/L in treated water from Strandfontein were recorded in January 2019. The TDS detected in the treated water from Strandfontein in January showed a positive relationship with the highest EC detected for the same month. Total Dissolved Solids (TDS) are known to have a positive correlation with conductivity since dissolved salts in water generally determine the electrical conductivity, thus the higher the TDS, the higher the conductivity (Islam *et al*., 2017). This relationship is depicted in Figure 17 as well, since dissolved solids in water generally determine the EC (Meride and Ayenew, 2016).

The TDS of the raw water from Monwabisi desalination plant during the course of the study ranged from 28887.4 to 33663.6 mg/L. The TDS for the treated water from Monwabisi ranged from 391.28 to 533.63 mg/L. Highest TDS mean value of 533.63 ± 29.97 mg/L were recorded in October for the treated water from Monwabisi. Similar to Strandfontein, the high TDS showed a positive relationship with the high EC recorded in the same month (Oct). Water treatment involves the reduction of TDS in the raw water. The treated water samples from Monwabisi and Strandfontein showed significant reduction in TDS following reverse osmosis treatment process which includes removal or reduction of solid matter using a membrane. The treated samples from both desalination plants complied with the SANS 241:2015 standard as all the results were less than the limit of 1200 mg/L TDS in drinking water. The presence of dissolved solids in water may affect its taste (Islam *et al*., 2017). The palatability of TDS concentration less than 600 mg/L is regarded as good, and that TDS levels exceeding 1,200 mg/L is regarded as unpalatable (Akoto *et al*., 2017). Since the results from both plants produced water that was less than 1,200 mg/L for TDS the treated water from these plants can be considered good and palatable. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in TDS concentrations between the raw and treated samples from both plants existed. T-test results showed significant differences for TDS concentration between the raw and treated water ($p < 0.001$).

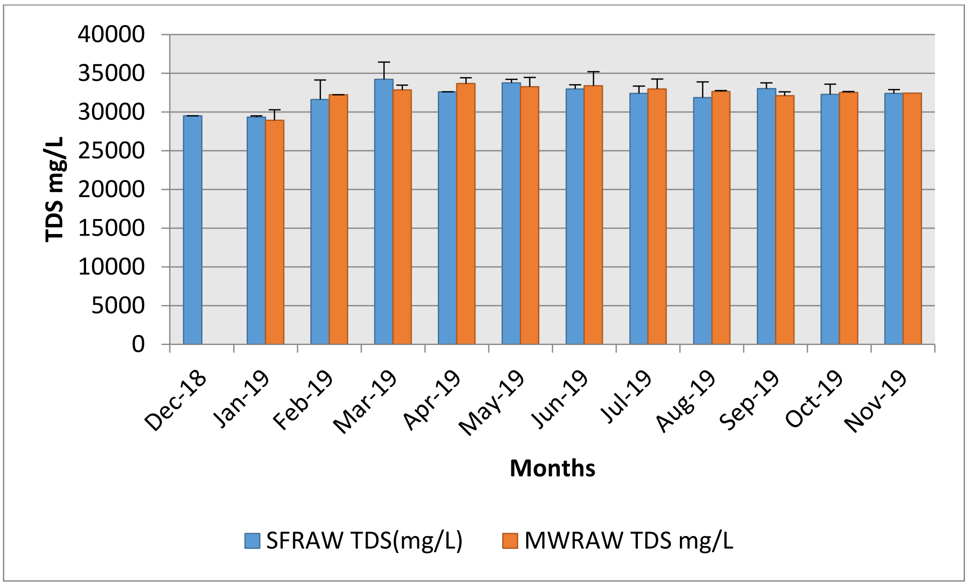

Figure 16: TDS (mg/L) levels in the raw water from MW and SF DWTPs.

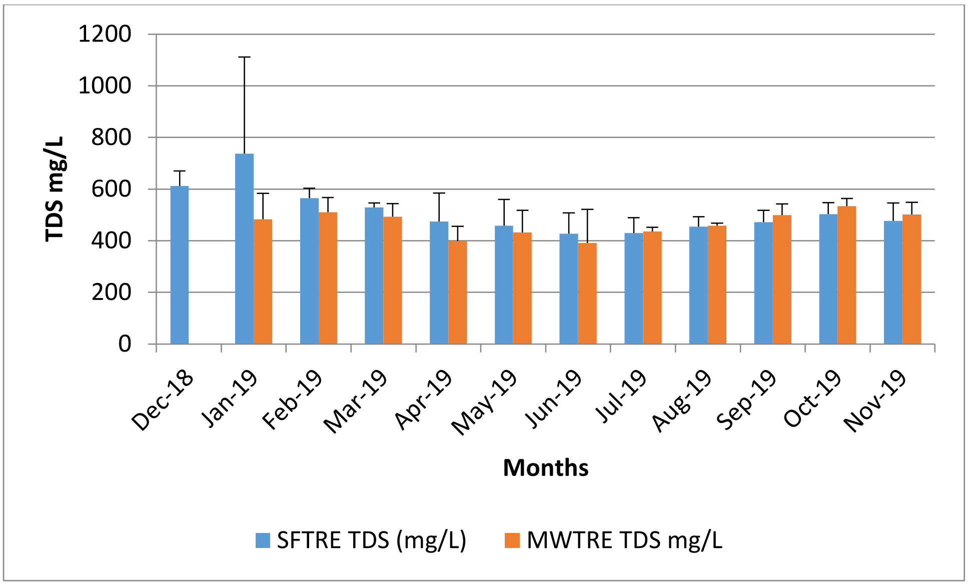

Figure 17 : TDS (mg/L) levels in the treated water from MW and SF DWTPs.

3.10.3 Alkalinity of the raw and treated drinking water from Strandfontein and Monwabisi desalination plant

Alkalinity of the raw water from Strandfontein ranged from 46.67 mg/L to 78.5 mg/L. Alkalinity for the treated water from Strandfontein ranged from 13.3 mg/L to 34.15 mg/L. The alkalinity concentrations of the raw water from Monwabisi ranged from 41.4 mg/L to 83.5 mg/L. The treated water from the same plant had alkalinity that ranged from 14.35 mg/L to 30.19 mg/L. Drinking water alkalinity levels are not specified in SANS 241:2015; however, the WHO standard specifies alkalinity in terms of total dissolved solids (TDS) of 500 mg/L. The alkalinity for the treated samples from Monwabisi and Strandfontein were relatively low with the highest alkalinity recorded value of 34.19 mg/L for Strandfontein and 30.19 mg/L for Monwabisi. Highest alkalinity levels in the treated samples from both plants were observed during warmer periods of summer and spring, which may be due to the high salts (EC, sodium and chlorides) observed in the plants during the same period of Jan for SFTRE and Oct for MWTRE. Alkalinity is not a contaminant; however, it is used as a measure of substances that have acid-neutralizing capacity. Elevated alkalinity concentrations in drinking water may cause numerous problems such as being unpalatable to the consumer. The results from both plants show that the treatment process worked adequately to reduce alkalinity levels which were initially higher in the raw water. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in alkalinity concentrations between the raw and treated samples from both plants existed. T-test results showed significant differences for alkalinity concentration between the raw and treated water ($p < 0.001$).

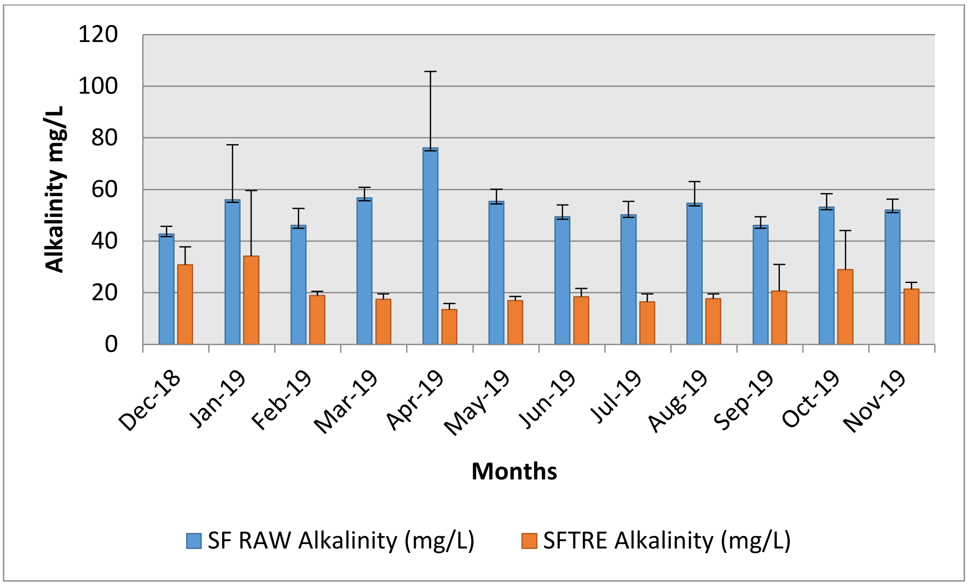

Figure 18: Alkalinity (mg/L) levels in the raw and treated water from SF DWTP.

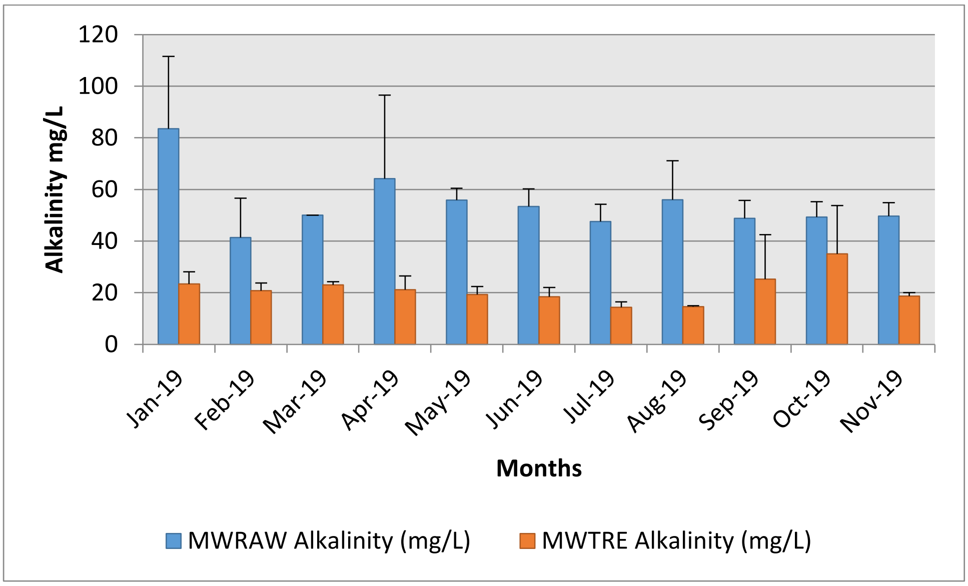

Figure 19: Alkalinity (mg/L) levels in the raw and treated water from MW DWTP.

3.10.4 pH for the raw and treated drinking water from Monwabisi and Strandfontein desalination plant

The pH in the raw water samples from Strandfontein ranged from 7.85 to 8.06. The pH of the treated water samples from Strandfontein desalination plant ranged from 8.54 to 9.19. The pH in the raw samples from Monwabisi ranged from 7.9 to 8.1 and the treated samples from Monwabisi ranged from 8.59 to 9.15. The observed variation in pH between the raw and treated water from both plants may be a good reason for constant monitoring to detect the fluctuations in pH. The treated water samples from Strandfontein and Monwabisi complied with the SANS 241:2015 standard which recommends pH values of $\geq 5 \leq 9.7$ at temperature of 25 °C. The increased pH for the treated water compared to the raw water from Monwabisi and Strandfontein desalination plants can be linked with pH adjustment using lime in the desalination water treatment process which increases the pH (Shatat and Riffat, 2014). In seawater reverse osmosis membranes, pH is a very stable parameter as a result of the huge buffering capacity of seawater, therefore the pH fluctuates only slightly and remains basic (Saeed, *et al*., 2019). It was observed in this study that pH was fairly stable and remained basic throughout all the seasons as shown in Figure 20 and 21. The pH of drinking water is normally not associated with any direct effects on human health; however, it has some indirect health effects such as influencing other water quality parameters such as solubility of some metals, affecting the taste and survival of pathogens (Kim *et al*., 2011). The pH of the treated water from both plants showed that the treatment significantly impacted the pH through the production of treated water with increased pH showing sufficient post-treatment efforts of using lime as an alkalizing agent to increase the pH in treated water. The final pH of the treated water complied with SANS 241: 2015 requirements of ≥ 5 to ≤ 9.7. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in pH concentrations between the raw and treated samples from both plants existed. T-test results showed significant differences for pH between the raw and treated water ($p < 0.001$).

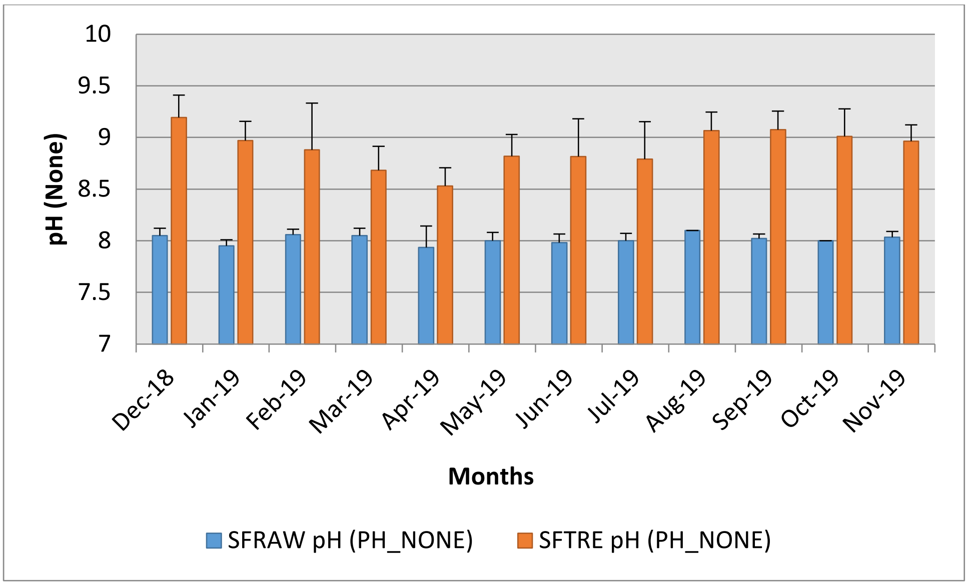

Figure 20: pH levels in the raw and treated water from SF DWTP.

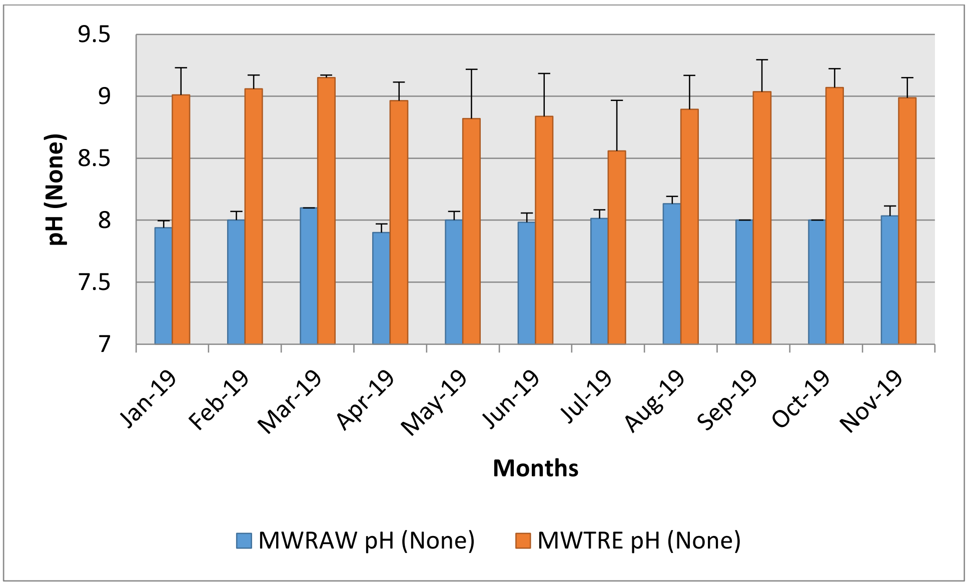

Figure 21: pH levels in the raw and treated water from MW DWTP.

3.10.5 Turbidity and colour in the treated water from Monwabisi and Strandfontein desalination plants

The turbidity of the treated water from Strandfontein desalination plant was mostly consistently below 1 NTU and showed compliance with the standard, except for January where turbidity levels spiked from 0.6 to 77.5 NTU. Furthermore, the colour was also consistently low throughout the study period and was below 5 mg/ L Pt-Co, except the increased colour value observed also in January ranging from 5 to 74 mg/ L Pt-Co. The high standard deviation shown in Figure 23 was influenced by the sudden increase (spike) in both turbidity and colour from normally constant low compliant values throughout the study period except for the samples collected in January which exceeded the standard limit and therefore this variance led to higher standard deviations. The acceptable limits for colour and turbidity in drinking water are ≤ 15 mg/ L Pt-Co and ≤ 5 NTU respectively. Figure 23 showed abrupt increases in turbidity and colour for January for the treated water from Strandfontein DWTP. Since colour is influenced by the presence of inorganic ions and turbidity by solids and colloidal matter, a positive relationship between turbidity and colour was observed. The increase in turbidity in January led to an increase in colour as well, since the colour of water is strongly related to turbidity. Another study which assessed the water quality from seawater reverse osmosis desalination plants also reported stable turbidity results with occasional abrupt increases which may be associated with rough sea conditions (Saeed *et al*., 2019), similar to the findings of the current study.

The turbidity of treated water from Monwabisi desalination plant was mostly consistent throughout the study period with an average below 1 NTU and was within compliance requirement which states ≤ 5 NTU. The colour for treated water from Monwabisi was below 15 mg/ L Pt-Co limit required by the standard. Following the desalination process using reverse osmosis it is expected that the water must have low turbidity due to the removal of unwanted suspended solids and unwanted pathogens (Rahmanian *et al*., 2015). The results from Monwabisi show that treatment process used at the desalination plant which includes reverse osmosis membranes sufficiently removed all the undesired solids which resulted in the reduction of colour and turbidity in the samples.

Colour in water is normally as result of the presence of inorganic ions (such as iron and manganese), humus and peat materials, plankton and weeds (Akoto *et al*., 2017). High colour of water from natural organic carbon can also indicate a high tendency to produce by-products from disinfection process. However, there is no health-based guideline value for colour in drinking water (WHO, 2011). The observed turbidity could be a result of silt or soil deposition, organic matter or microorganisms which are often associated with high turbidity of drinking water. Since the treated water from Strandfontein had high turbidity and colour in January which is during summer, increasing temperature is known to lead to an increase in turbidity which may have influenced the spike in turbidity and colour during the same month (Tan *et al*., 2017).

Increased turbidity in water presents challenges with water treatment processes often leading to escalated treatment costs (DWAF, 1998). Furthermore, previous studies have often linked high turbidity in water with the presence microbiological contamination and thus makes disinfection of the water difficult, since some of the colloids have adsorptive characteristics and thus may shield some microorganisms from the disinfectant and when not treated properly prior to the distribution system may cause waterborne diseases such as gastroenteritis (Kale, 2015). The turbidity values in both plants were consistently low throughout the study period, and there was no direct relationship observed between turbidity and biological water quality parameters including *E. coli*, HPC and total coliforms. High bacterial counts in the drinking water from both plants were observed even when there was low turbidity. Highest turbidity levels exceeding the maximum standard limit in Strandfontein DWTP in January also did not show a direct relationship with microbial loads in the drinking water, as there was no *E. coli* and total coliforms detected for the same month and the HPC bacterial counts were low in January. The low bacterial load even when turbidity levels were exceedingly high in the treated water from Strandfontein might be due to the effect of the chlorine disinfectant applied which worked well to supress bacterial growth in the treated water. Sources of increased turbidity the raw water is often associated with soil erosion and domestic, industrial runoff and agricultural runoff from catchment (Naubi *et al.*, 2016). The treatment process at the two desalination plants including pre-treatment and filtration using a membrane resulted in treated water samples with reduced colour and turbidity, since seawater is known to comprise of large amounts of dissolved salts which can impact the colour and turbidity.

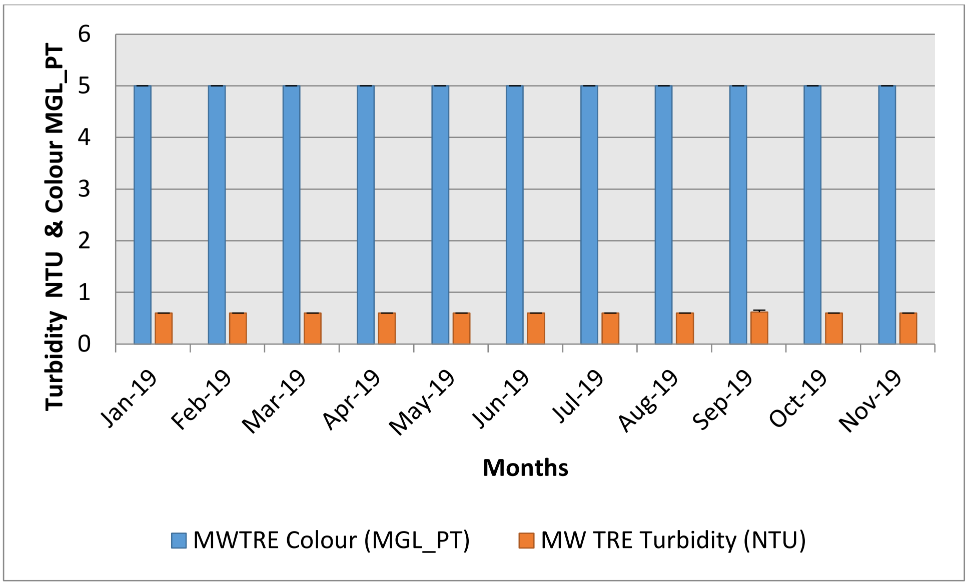

Figure 22: Turbidity (NTU) and colour mg/L PT levels in the treated water from MW DWTP.

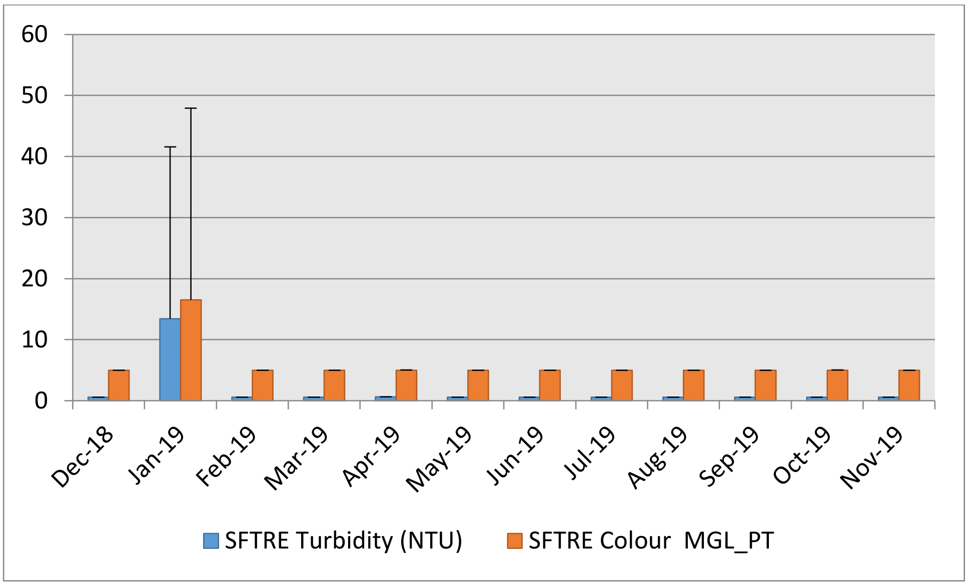

Figure 23: Turbidity (NTU) and colour mg/L_ PT levels in the treated water from SF DWTP.

3.10.6 Total hardness of the treated drinking water from Strandfontein and Monwabisi desalination plant

The total hardness for the treated water from Strandfontein desalination plant ranged from 4 to 601 mg/L with the highest total hardness of 601 mg/L in the treated water from Strandfontein recorded in January as shown in Figure 24. The total hardness for the treated water from Monwabisi desalination plant ranged from 5 to 122 mg/L with the highest total hardness of 122 mg/L in the treated water from Monwabisi recorded in April. The treated water samples from Monwabisi and Strandfontein desalination plant were both lower than 300 mg/L and thus were within the limit specified by WHO, except the one-recorded result of 601 mg/L at Strandfontein DWTP. The low amount of total hardness in the treated water from both plants suggests low amounts of cations are dissolved in water showing efficiency of the reverse osmosis treatment process. Water hardness ($CaCO_3$) is influenced by the presence of major anions and cations, such as bicarbonate, sulphate, chloride, calcium and magnesium (Sharmar and Kumar, 2017). The total hardness of drinking water is not specified in SANS 241:2015 standard, however, WHO recommends a limit of ≤ 300 mg/L for total hardness. Currently, there is no health-based guideline value proposed for hardness in drinking-water, however high hardness can cause problems for daily human uses (WHO, 2011). Water hardness is not known to induce any effects in human health, however high hardness can cause problems for daily human uses (Chidya *et al*., 2019). The treated water from both plants produced treated water with low total hardness showing that the water was aesthetically good.

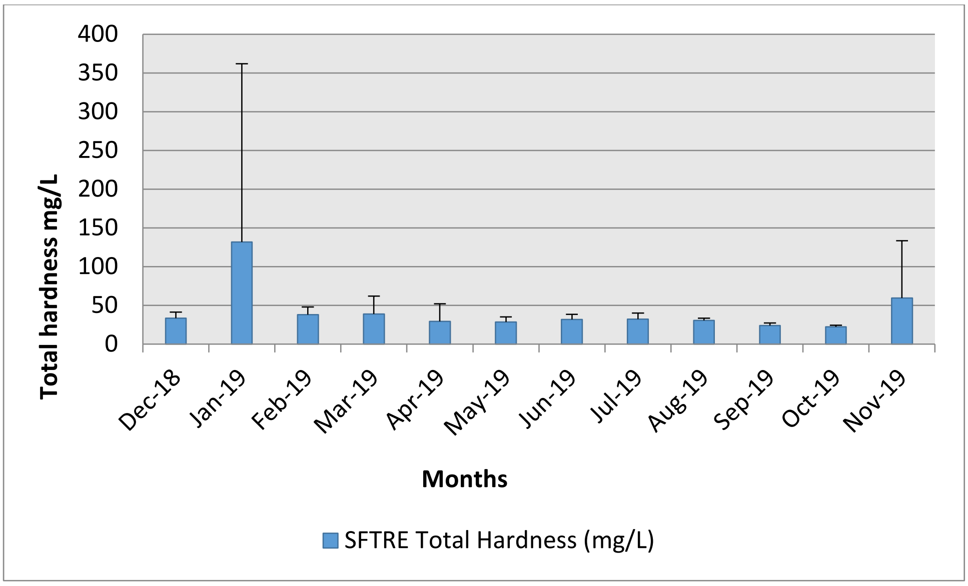

Figure 24: Total hardness (mg/L) levels in the treated water from SF DWTP

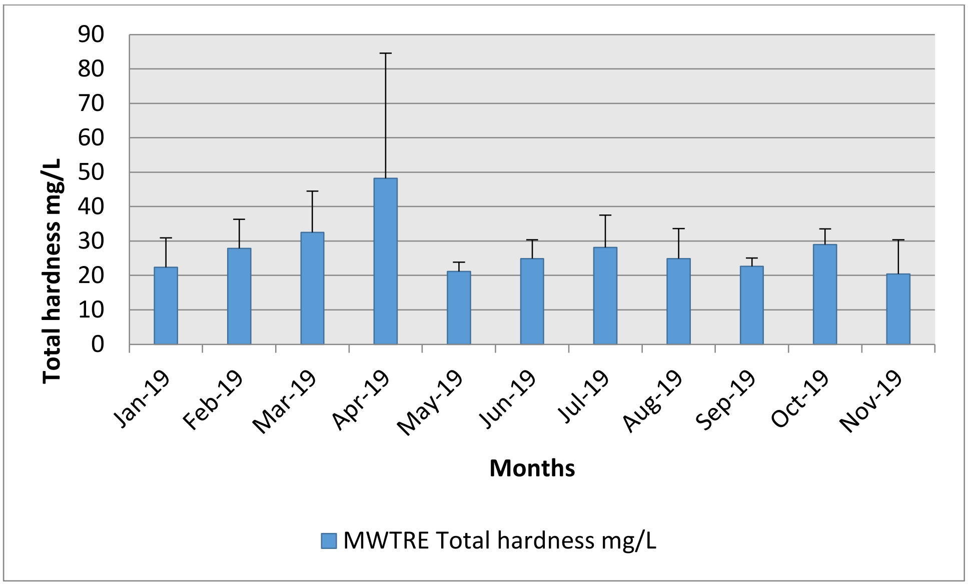

Figure 25: Total hardness (mg/L) levels in the treated water from MW DWTP.

3.11 Chemical properties of the raw and treated water

3.11.1 Sodium and chloride in treated water from Monwabisi and Strandfontein DWTPs

Chloride levels of the raw and treated water from Strandfontein ranged from 15777 to 25936 mg/L and 36 to 714 mg/L respectively. The highest value of 714 mg/L of chloride for the treated water were recorded in January, which is during the summer period and this exceeded the limit of ≤ 300 for chloride in treated water. The sodium in the treated water from Strandfontein ranged from 16.6 to 288.2 mg/L. The highest sodium value of 288.2 mg/L were also recorded in January which was higher and exceeded the recommended limit of ≤ 200 mg/L for sodium in treated water. The highest recorded chloride and sodium values at Strandfontein DWTP for the treated water in January also coincided with the highest values for EC, TDS, total hardness, colour and turbidity, thus showing that dissolved solids such as chlorides and sodium influenced the concentration of other parameters such as EC, TDS, colour, turbidity, alkalinity and total hardness. These values recorded in January for CI and Na were the only non-compliant results as the other values for these parameter throughout the study were lower than the recommended limit of ≤ 300 mg/L for CI and ≤ 200 mg/L for Na which may be due to water quality fluctuations associated with sudden rough sea conditions.

The raw water from Monwabisi DWTP ranged from 2092 to 38849 mg/L for chloride levels. The treated water from the same plant ranged from 18 to 251 mg/L for chloride. The chloride results for the treated water from Monwabisi were all less than recommended limit of ≤ 300 mg/L for chlorides and complied with the SANS 241 standard. The sodium in the treated water from Monwabisi ranged from 25.9 to 259.2 mg/L throughout the sampling period. The sodium from the treated samples from Monwabisi were also compliant with SANS 241 and were less than the ≤ 200 mg/L limit except the recorded value of 259.2 recorded in January.

Since the source water used to produce the treated water is seawater, this may be associated with the increased concentration of CI and Na in the treated water. Sodium and Chloride are the main constituents of seawater and makeup 85 % of the dissolved salts that give seawater its brackish taste (Loganathan *et al*., 2017), thus the seawater used as source water for this current study contributed to the concentration of sodium and chloride detected in the raw and treated water from both plants. Chloride is formed by the dissolution of salts of hydrochloric acid as table salt (NaCl), Na_2CO_3. The sources of chloride in drinking water include industrial waste, sewage and seawater (Meride and Ayenew, 2016). Following membrane treatment that allows only “pure” water to pass through, the chloride levels showed a marked decrease. Chloride and sodium levels in the water does not pose any significant risk to the users, however they can impart taste to the water for levels exceeding the compliance limit (Edokpayi *et al*., 2018). The low chloride and sodium in majority of the treated water from both plants suggests treated water with good aesthetic qualities since values exceeding the limits are related with salty water. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in chloride concentrations between the raw and treated samples from both plants existed. T-test results revealed significant differences for chlorides between the raw and treated water ($p < 0.001$).

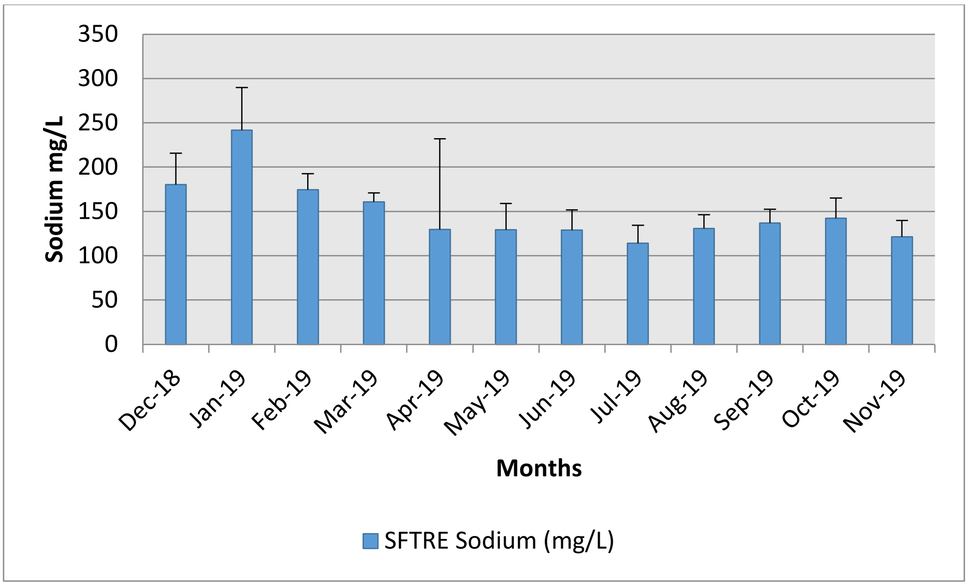

Figure 26: Sodium (mg/L) levels in the treated water from SF DWTP.

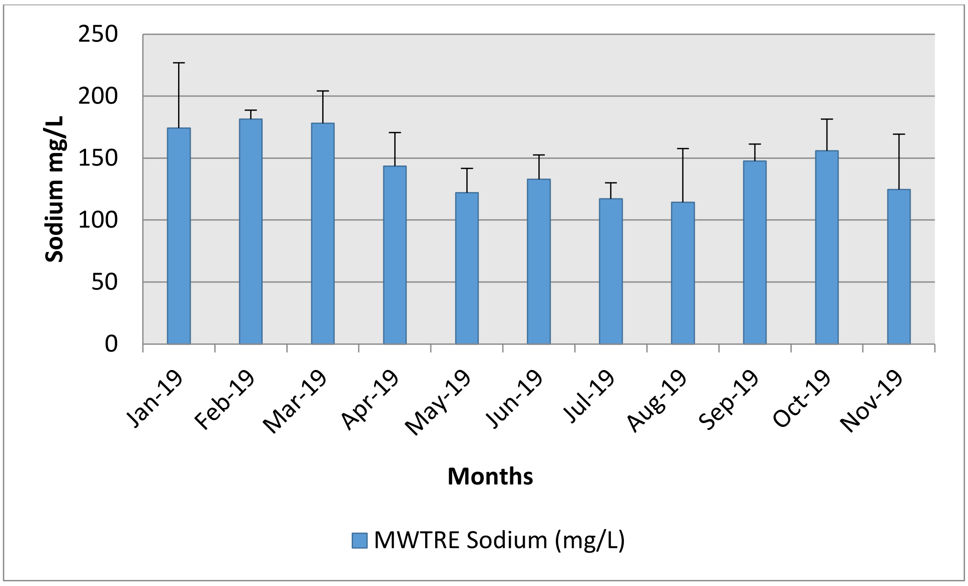

Figure 27: Sodium (mg/L) levels in the treated water from MW DWTP.

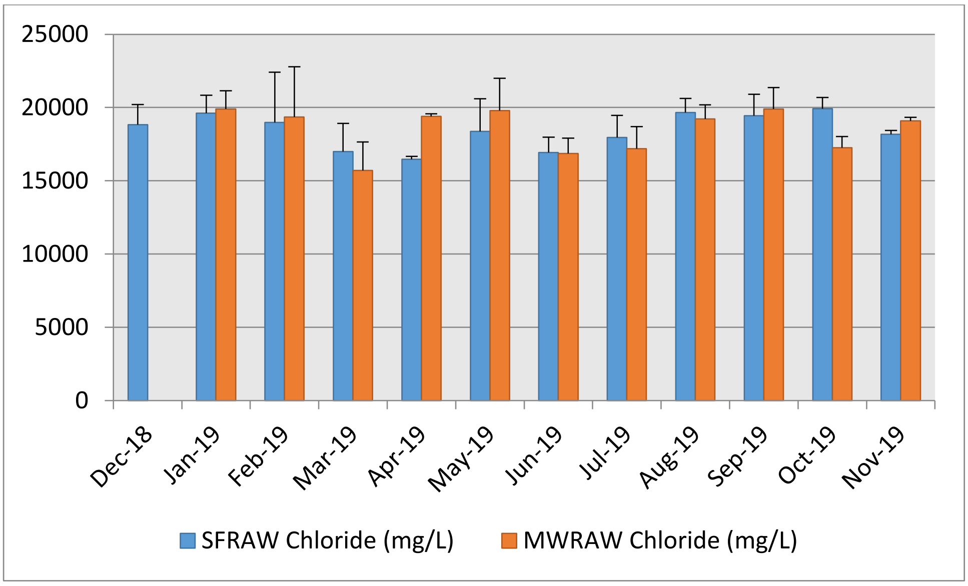

Figure 28: Chloride (mg/L) levels of the raw water from SF ad MW DWTPs.

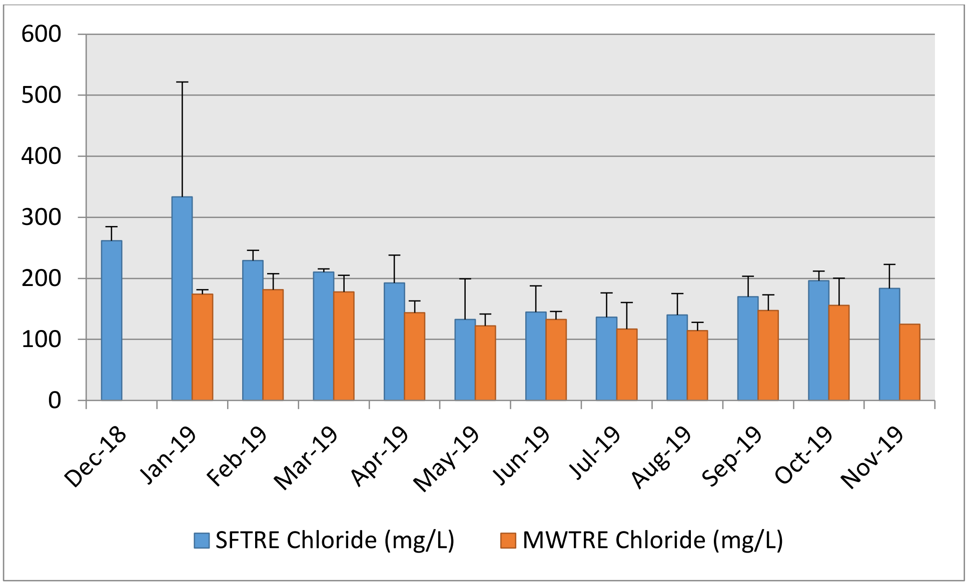

Figure 29: Chloride (mg/L) levels of the treated water from SF ad MW DWTPs.

3.11.2 Magnesium, sulphate, calcium and potassium in the treated water from Monwabisi and Strandfontein desalination plants

Magnesium, calcium, potassium and sulphates are all dissolved salts found in drinking water. The results from the treated water from Strandfontein desalination plant for magnesium, potassium and calcium ranged from 0.19 to 5.56mg/L; 0.82 to 23.16 mg/L; 1.4 to 233 mg/L respectively. The sulphate in the treated water from Strandfontein ranged from 0.8 to 225 mg/L. The results for the treated water from Monwabisi desalination plant for magnesium, potassium and calcium ranged from 0.2 to 6.23 mg/L; 1.24 to 10.97 mg/L; 1.6 to 38.7 mg/L. The sulphate in the drinking water from Monwabisi ranged from 1 to 8.9 mg/L.

Magnesium, calcium and potassium are not specified in the SANS 241:2015 standard and only sulphate is specified with a recommended limit of ≤ 250 mg/L for aesthetic and ≤ 500 mg/L for acute health. Currently there are no stipulated national guidelines for minerals such as Ca, Mg, K and Na specifically for drinking water produced through desalination. The sulphate in the treated water from both desalination plants was significantly lower than the limit of ≤ 250 mg/L, which points out to the efficient treatment process of reverse osmosis process. Ca, K and Mg were also relatively low in the drinking water from both plants as well.

Since the source water used was seawater, dissolved salts were assessed in this study. Seawater is distinguished from other water sources by its saline nature as a result of dissolved salts. It is important to assess for dissolved salts in the treated drinking water since water has the ability to dissolve numerous inorganic and some organic minerals or salts such as potassium, calcium, sodium, bicarbonates, chlorides, magnesium, sulphates etc. These minerals are known to be for producing an undesirable taste and diluted colour appearance of water when they exceed recommended limits, thus to screen for the quantities is done to ensure the aesthetic quality of water produced is efficient for drinking (Meride and Ayenew, 2016). The compliance of these dissolved salts in the treated water from both plants suggests that the water was aesthetically good. Although, all the salts detected in the treated water from Monwabisi and Strandfontein desalination plants were in low concentrations, a need exists to review the current standard guidelines to add a complete suite of salts such as calcium, potassium and magnesium which are not prescribed in the standard. This is particularly important for drinking water produced by seawater desalination as these salts are found in high concentrations in seawater. These salts play an important role in the maintenance of human health; however, at exceedingly high concentrations these salts can be detrimental to human health causing a number of ailments. Establishment of this guideline will ensure continuous monitoring of these salts for all desalination plants which are used in the country for production of potable water, which will ensure that these salts are measured against set criteria of the standard ensuring public protection of human health.

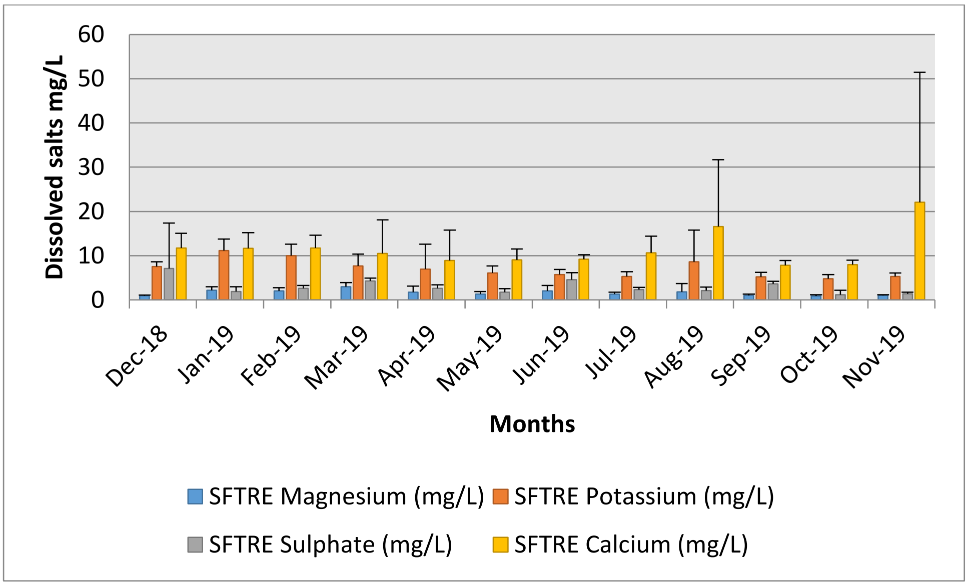

Figure 30: Sulphate, potassium, magnesium and calcium (mg/L) levels in the treated water from SF DWTP.

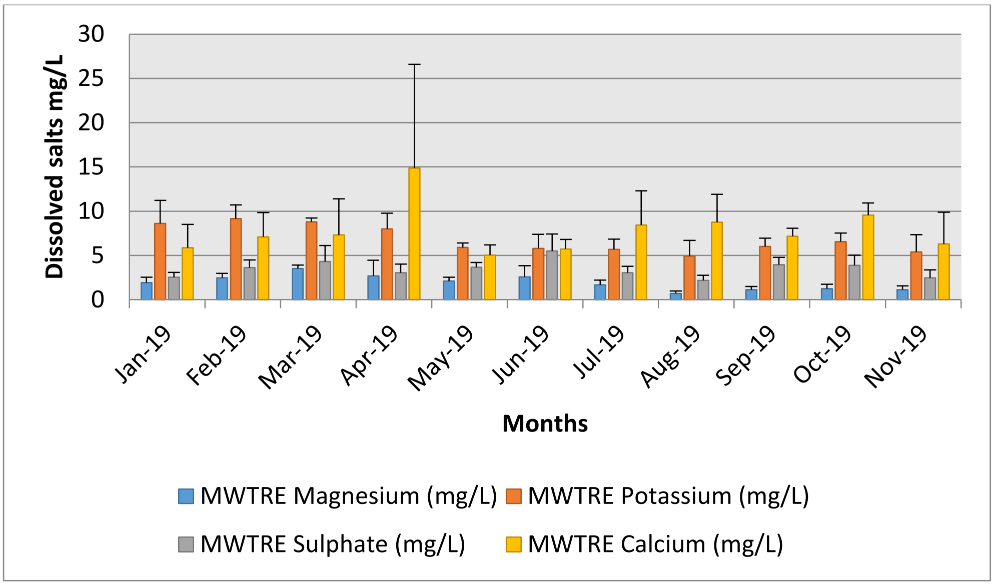

Figure 31: Sulphate, potassium, magnesium and calcium (mg/L) levels in the treated water from MW DWTP.

3.11.3 Zinc and fluoride in the treated water from Monwabisi and Strandfontein DWTPs

The Fluoride in the treated water from Strandfontein and Monwabisi ranged from 0.01 to 0.12 mg/L and 0.01 to 0.06 mg/L respectively. SANS 241 recommends limits of ≤ 1.5 mg/L for treated water. The results from both plants were within the SANS limit. In South Africa and other countries, fluoride is one of the chemical determinants of concern that occurs in high levels (Akuno *et al.*, 2019). Since fluoride is one of the major chemicals of concerns in South Africa associated with dental health risks, assessment for its presence in drinking water is important. The low fluoride levels found in the treated drinking water produced from these plants can be considered safe for consumption and may not be expected to cause any dental health risks.

The concentrations of Zinc from Strandfontein and Monwabisi ranged from 0.05 to 0.051 mg/L and 0.05 to 0.056 mg/L and were within SANS 241 limits of ≤ 5 mg/L. Excess zinc in drinking water is known to cause health effects such as gastrointestinal issues (stomach cramps, vomiting and diarrhoea) and thus monitoring of Zn in drinking water is important (Damielda and Kruse, 2019). The zinc concentration of the treated water from both plants was lower than the maximum permissible limit of SANS 241 (2015) and can be considered safe for consumption.

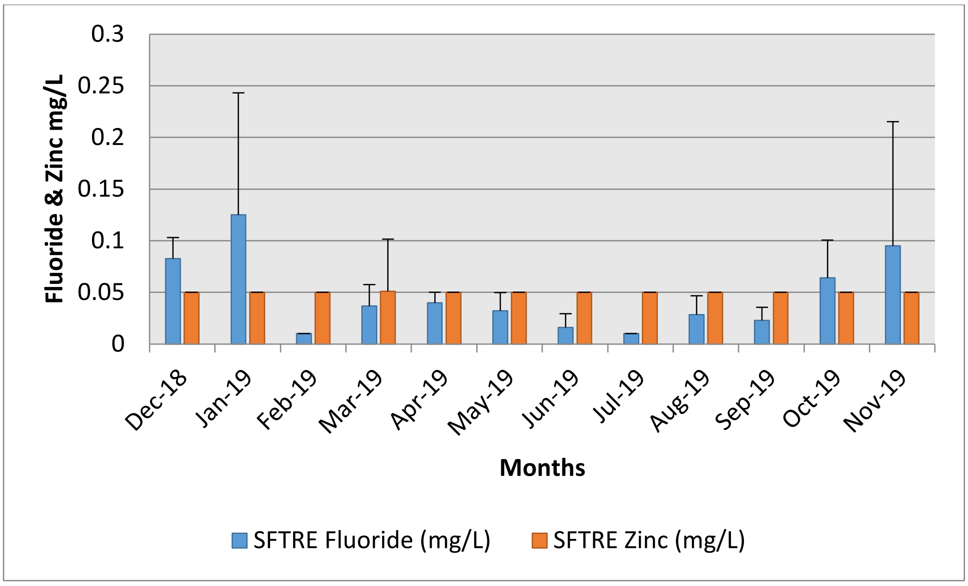

Figure 32: Zinc mg/L and fluoride mg/L levels for the treated water from SF DWTP.

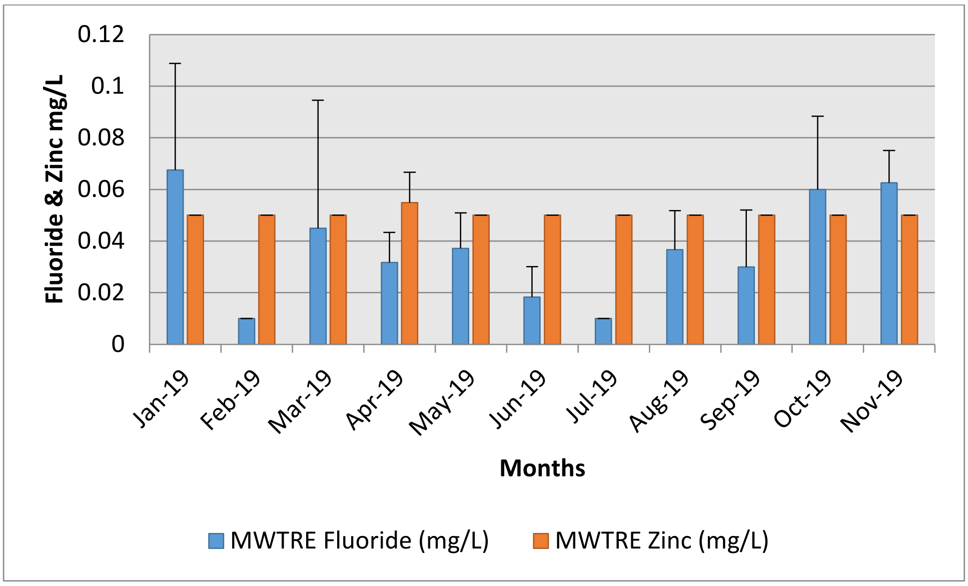

Figure 33: Zinc (mg/L) and fluoride (mg/L) for the treated water from MW DWTP.

3.11.4 Residual chlorine in the treated water from Monwabisi and Strandfontein DWTPs

The drinking water standard specifies that residual chlorine of ≤ 5 mg/ L must be maintained in order to suppress bacterial growth (WHO, 2008; SANS 241: 2015. The two desalination plants studied use chlorine to disinfect the water. The results of free chlorine in the final treated water all complied with SANS 241:2015 standard level of ≤ 5 mg/ L residual chlorine.

The residual chlorine levels in Strandfontein treated water ranged from 0.2 to 0.8 mg/L. The chlorine readings from Monwabisi DWTP for the treated water ranged from 0.4 to 1.1 mg/L. Thus, the sufficient amount of residual chlorine levels analysed in the treated water from Strandfontein and Monwabisi DWTP was mostly adequate in offering microbial protection to the treated water from both plants and was also within the stipulated guideline of ≤ 5 mg/ L residual chlorine. The presence of acceptable levels of HPC bacteria in treated water from both plants may be indicative of the effectiveness of the treatment and adequacy of disinfectant residuals chlorine levels since both plants complied with the HPC counts which were lower than the guideline limit of ≤ 1000 CFU/1 mL for HPC in drinking water.

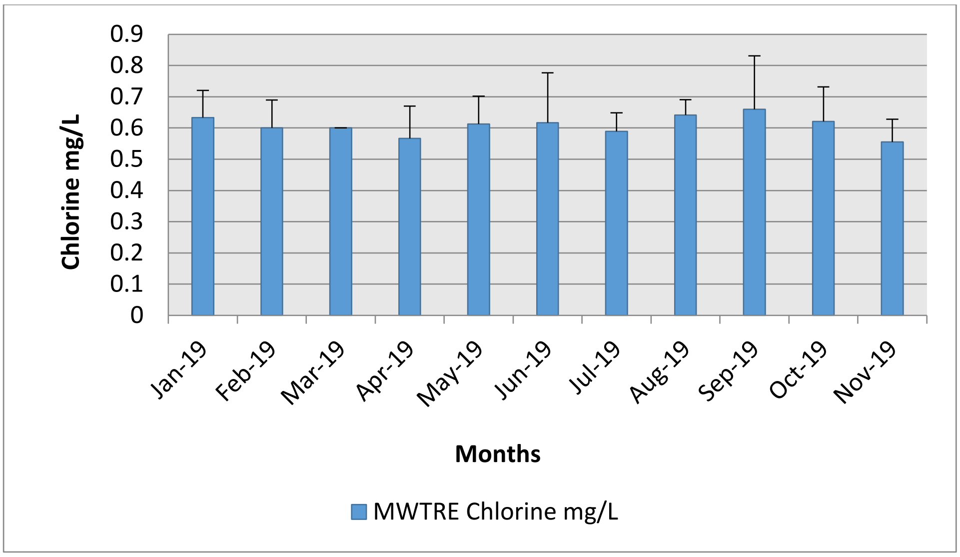

Figure 34: Residual free chlorine levels in the treated water from MW DWTP.

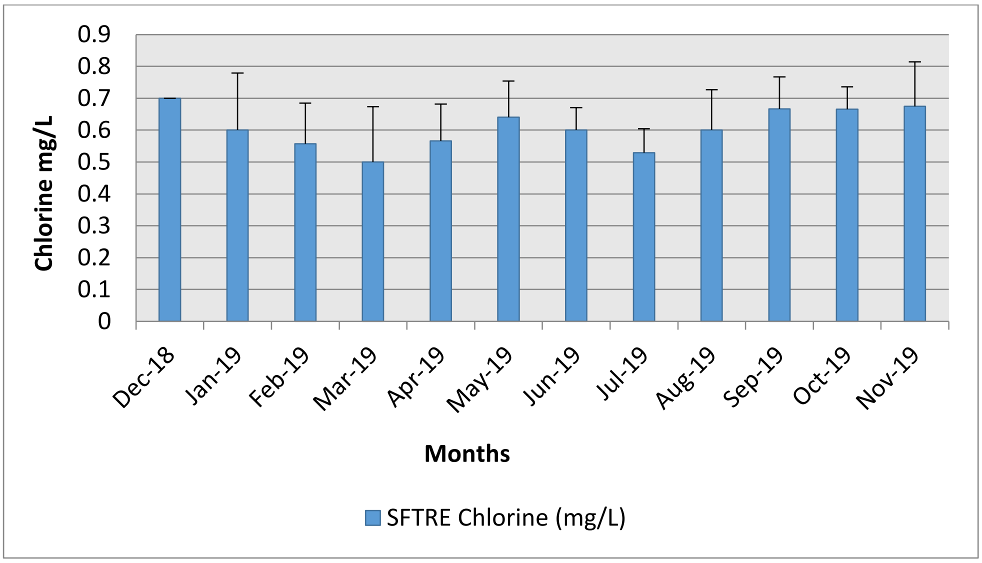

Figure 35: Residual free chlorine levels in the treated water from SF DWTP.

3.11.5 Nitrates and nitrites in the raw and treated water from Monwabisi and Strandfontein desalination plant

The nitrates and nitrites in the raw water Strandfontein desalination plant were mostly consistently low during the course of the study and ranged from 0.1 mg/L to 0.33 mg/L for nitrates and 0.05 mg/L to 0.1 mg/L for nitrites respectively. The treated water from Strandfontein was low and ranged from 0.1 to 0.2 mg/L for nitrates and 0.05 to 0.1 mg/L for nitrites. The nitrates and nitrites in the raw water Monwabisi desalination plant were mostly consistently low during the course of the study and ranged from 0.1 mg/L to 2.8 mg/L for nitrates and 0.05 mg/L to 0.1 mg/L for nitrites respectively. The observed values for the treated water from both desalination plants complied as the nitrates were mostly less than 0.2 mg/L which is significantly lower than the recommended value of ≤ 11 mg/L for nitrates in drinking water. Nitrates in the raw water from Monwabisi DWTP were consistently lower than 0.2 mg/L throughout the study period, except in May where nitrate levels of 2.8 mg/L were observed which caused high standard deviation as all other nitrate results throughout the study period were less than 0.2 mg/L. The SANS 241: 2015 standard for nitrite in drinking water is ≤0.9mg/L and all the treated samples from Monwabisi and Strandfontein were compliant as they were less than 0.9 mg/L for nitrites. Trends of low nitrates throughout the study period were observed which may suggest that these two desalination plants have low impact of organic pollution emanating from agricultural runoff, industrial wastewater discharges, urban domestic sewage, septic systems, human waste lagoons, amongst other sources of pollution (Batool *et al*., 2018).

In the raw and treated water it was observed that the nitrites were lower than the nitrates. Nitrates and nitrites are found in the environment and both are part of the oxidation of nitrogen as part of the cycle that is essential for all living organisms for the production of complex organic molecules such as proteins and enzymes (IARC, 2010). Since nitrates are a stable form of oxidized nitrogen, this may be the reason why there was high nitrates compared to nitrites. During anaerobic conditions and in the presence of a carbon source, microorganisms are able to reduce nitrates into nitrites which is relatively unstable and moderately reactive (WHO, 2016). The compliance of nitrates and nitrites for the treated water from both plants is therefore not expected to exert any taste, odour or health problems.

Combined nitrate plus nitrite measured for the treated water from both plants had a measured value of 0.1 (none) which was constant throughout the study period and complied with the specified limit of ≤ 1 (none) by SANS 241: 2015. Ammonia measured for the treated water from the same plants had a consistent measured value of 0.4 mg/L throughout the study period as well. SANS 241 recommends ammonia limits of ≤ 1.5 mg/L and both the plants were within limits, thus the aesthetic quality of the treated water from both plants may be regarded as good since the results throughout the study were compliant. Comparative analysis using independent sample t-test on statistica software was done to determine if significance of variation or differences in nitrate and nitrite concentrations between the raw and treated samples from both plants existed. T-test results revealed significant differences for nitrates in the raw and treated water ($p < 0.001$) from Strandfontein; however, there was no significant difference in the raw and treated water from Monwabisi for nitrates and nitrites for both plants.

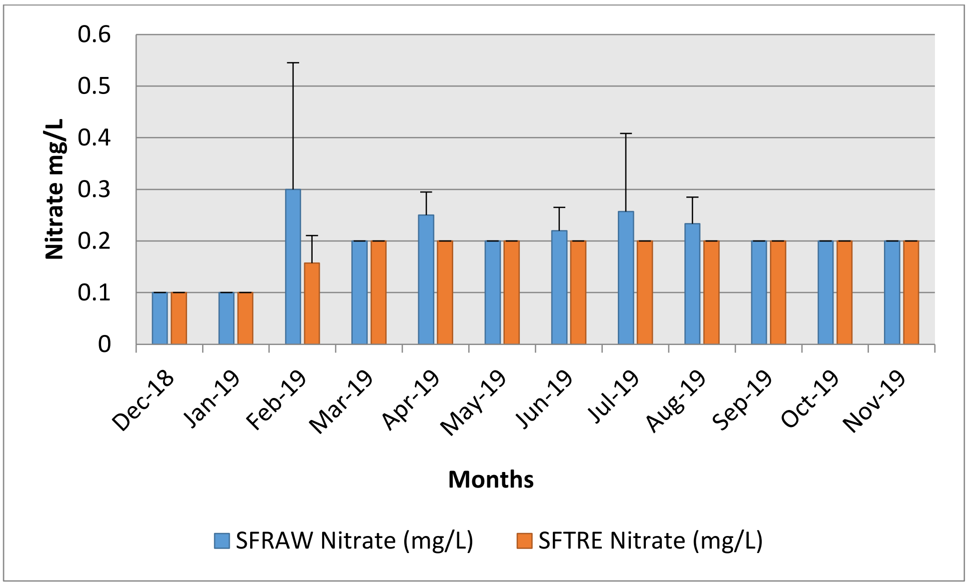

Figure 36: Nitrates (mg/L) levels on the raw water from SF DWTP.

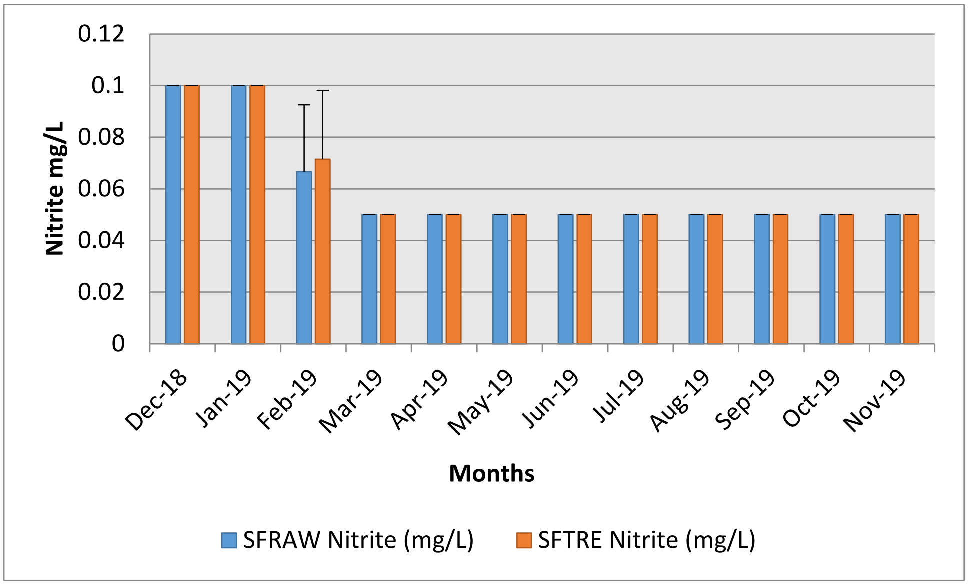

Figure 37: Nitrites (mg/L) levels in the raw water from SF DWTP.

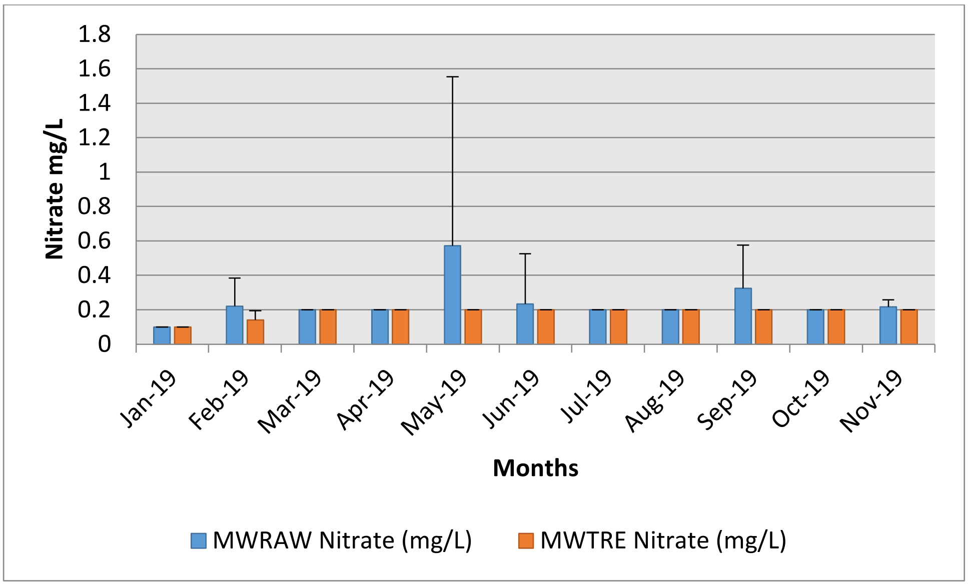

Figure 38: Nitrates (mg/L) levels in the treated water from MW DWTP.

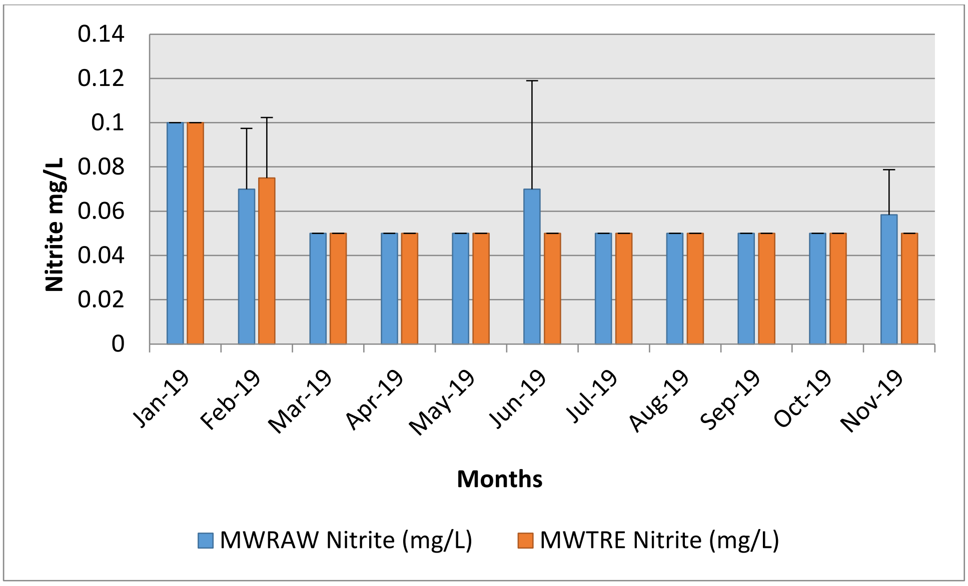

Figure 39: Nitrates (mg/L) levels in the treated water from SF DWTP.

3.11.6 Heavy metal determination in the treated water

Table 8: Heavy metal concentrations in the treated water from Monwabisi and Strandfontein DWTP

SANS 241:2015 limits for lifetime consumption					
Chemical determinants (Heavy metals)	**Risk**	**Standard limits**	**Units**	**Strandfontein desalination plant**	**Monwabisi desalination plant**
Aluminium	Operational	≤ 300	µg/L	51.68 ± 9.00	51.53 ± 6.87
Antimony	Chronic health	≤ 20	µg/L	1.11 ± 0.43	1.00 ± 0.24
Arsenic	Chronic health	≤ 10	µg/L	1.40 ± 0.64	1.47 ± 0.62
Barium	Chronic health	≤ 700	µg/L	50 ± 0	50 ± 0
Boron	Chronic health	≤ 2400	mg/L	1.344 ± 0.35	1.227 ± 0.33
Cadmium	Chronic health	≤ 3	µg/L	3 ± 0	3 ± 0
Chromium	Chronic health	≤ 50	µg/L	5.00 ± 0.05	5.09 ± 0.62
Copper	Chronic health	≤ 2000	µg/L	11.52 ± 6.14	10.85 ± 2.40
Cyanide	Acute health	≤ 200	µg/L	12.31 ± 10.78	14.89 ± 8.26
Iron	Aesthetic	≤ 300	µg/L	56.59 ± 22.32	53.04 ± 13.04

	Chronic health	≤ 2000	µg/L		
Lead	Chronic health	≤ 10	µg/L	10 ± 0	10 ± 0
Manganese	Chronic health	≤ 400	µg/L	5.23 ± 1.45	5.53 ± 3.06
	Aesthetic	≤ 100	µg/L		
Nickel	Chronic health	≤ 70	µg/L	5.09 ± 0.76	5.33 ± 2.12
Selenium	Chronic health	≤ 40	µg/L	1.06 ± 0.18	1.12 ± 0.37

*Mean values and standard deviation

Heavy metals in the drinking water produced from Monwabisi and Strandfontein were assessed to verify if concentrations in drinking water were not at detrimental levels to human health. Analysis of heavy metals in drinking water is an essential parameter, and most of studies on drinking water quality involve assessment of heavy metals.

Heavy metal concentrations in the drinking water shown in (Table 8) were generally low and all complied with SANS 241: 2015 standard. High concentrations of heavy metals are known to deteriorate water quality and pose significant health risks to the public due to their toxicity, persistence, and bio-accumulative nature (Obasi and Akudinobi, 2020). Other metals even at low concentrations are able to cause significant adverse effects and thus screening for heavy metals for drinking water is of the utmost importance.

The presence of a storm-water detention pond in Monwabisi and localized run-off in Strandfontein may constitute as a potential sources of heavy metal contamination in the desalinated water. Furthermore, research has showed an increase since 1985 of the concentration of metals such as cadmium, lead, and manganese in Western Cape marine ecosystems (Sparks *et al*., 2014). Thus it was important to test for the concentrations of these heavy metals in the treated water from Monwabisi and Strandfontein desalination plant to determine the presence of heavy metals, since the chemical pollutants found in the False Bay may seep through to the final treated water from both plants. The dominant heavy metal, Boron in seawater was assessed. Boron present in seawater is approximately 5 to 6 mg/L and mainly found as the mononuclear form of boric acid (B(OH)3) and borate ion (B(OH)4 (He *at al*., 2019). Seawater desalination by reverse osmosis (RO) membrane processes have been considered as a reliable technology for production of fresh water in many arid regions including Cape Town, South Africa. Studies have shown that the removal of boron in seawater can be influenced by many factors including the pH and the dissociation of constant boric acid (Kang *et al*., 2015). Boron can be toxic when the concentrations exceed the required amount (Cho *et al*., 2015). The treated water from Strandfontein and Monwabisi were assessed for the concentration of boron and they recorded mean values of 1.344 ± 0.35 and 1.227 ± 0.33 mg/L respectively. SANS 241: 2015 recommends values of ≤ 2.5 mg/L, the results from both plants were lower than the stipulated limit.

Another study by Gao *et al.* (2020) investigated the occurrence of heavy metals in desalinated water using reverse osmosis technology versus the multiple effect distillation (MED) technology. The findings of the study showed that RO method was efficient in removing numerous heavy metals including; As, Cd, Pb, Hg, Al, Fe, Mn, Cu, Ba, Be, B, Ni and Ti from the seawater and these values were compliant with the water quality standard limits in China. Furthermore, their findings showed that the RO method showed better treatment efficiency compared to the MED method. The findings of the current study also showed compliance of all the heavy metals in the treated drinking water from both plants suggesting that the treated water from Monwabisi and Strandfontein DWTP is not expected to exert any health risk to the consumers, heavy metals concentration were less than the specified SANS 241: 2015 limit. The use of the RO method by both plants suggests that this method is efficient as shown by the results of this present study.

The outcome of the present study supports the hypothesis of the study and showed that the water quality of the final treated water produced for drinking water from the desalination plants (Strandfontein and Monwabisi, Cape Town) complied with the SANS 241: 2015 which regulates the quality of acceptable drinking water and posed minimal toxicity effects to marine test organisms.

3.12 Statistical evaluation of water quality data against SANS 241:2015

Comparative statistical analyses were performed on the levels of some physical, chemical and microbiological determinants found in the raw water and treated drinking water from Monwabisi and Strandfontein desalination plant. Statistical analyses using statistica program were performed on the data to explore whether the treatment was efficient in removing some substances/materials in the raw water and to check the water quality of the treated drinking water. Significance of variation or differences in the determinants' concentrations were tested using independent sample t-test. Significance was accepted at a probability value *p* equal to or less than 0.05. The normality of the data was evaluated using the Shapiro-Wilk W test, whilst homogeneity of variance was tested using Levenes' test.

3.12.1 Microbiological statistical variance for the raw and treated water

3.12.1.1 Determination of the *E. coli* variance between the raw and treated drinking water samples from MW and SF DWTPs to check for treatment efficiency

The statistical results for comparison of the presence of *E. coli* in the raw and treated samples from MW (raw and treated) and SF (raw and treated) respectively had a p value of < 0.001 which is less than p value of 0.05 thus showing that there was a significant difference in *E. coli* counts (CFU/100 mL) in raw and treated water results. This is expected as the raw water often contains different organic pollutants and high microbial loads that may emanate from the discharge of untreated sewage marine outfalls into the ocean amongst other anthropogenic sources including industrial, agriculture, septic tanks and wastewater effluents (Petrik *et al*., 2017; Devane *et al*., 2020; McKee and Cruz, 2021). However, the treatment process is there to ensure that all those unwanted pathogens and chemicals are removed and the water is treated to comply with the set standards both nationally and internationally in order to ensure the protection of public health. The significant difference from the raw to the treated water for both plants shows the efficiency of the treatment process, *E. coli* was removed from the raw water, thus *E. coli* was not detected in the treated water and the desalinated water from the two plants can be regarded as safe for human consumption.

Table 9: T- test results for microbiological properties of the raw and treated water

Microbiological determinant	**df**	**t-value**	**P value**
SF: *E. coli* (Raw vs Treated)	170	-9.07981	<0.001
MW: *E. coli* (Raw vs Treated)	137	1.382581	<0.001

*Df = degrees of freedom; T= t value; *p* value = probability; red colour = $p < 0.05$

3.12.1.2 Comparison of the *E. coli* and enterococcus of the raw water from MW and SF DWTPs

T- test results for the raw water for *E. coli* between SF raw and MW raw represented in Table 10 had a p value 0.274 and thus were greater than $p > 0.05$ indicating that the difference in *E. coli* for the raw water samples from the two plants were statistically not significant. The results for enterococcus in the raw water for SF Raw and MW Raw represented in Table 10 had a p value of = 0 .224 and thus were greater than $p > 0.05$ indicating that the enterococcus levels in Monwabisi and Strandfontein were statistically not significant. This was done to determine whether the *E. coli* and enterococcus levels in the raw water from these two plants had varied water quality. Since there was no significant difference in the raw water quality from both plants this suggests that the raw water used at the two plants is of similar composition as they lie off the same coastal region, under the False Bay catchment. These plants are influenced by similar climatic conditions as well as similar anthropogenic contamination as they are adjacent to one another and are in close proximity in distance; hence the statistical difference between enterococcus and *E. coli* in the raw water from both plants was not significant.

Table 10: T- test results for microbiological properties of the raw water

Microbiological determinant	**df**	**t-value**	***P* value**
E. coli (raw) SF vs MW	158	1.096970	0.274
Enterococcus (raw) SF vs MW	156	1.220696	0.224

*Df = degrees of freedom; T = t value; p value = probability; red colour = $p < 0.05$

3.12.1.3 Determination of statistical variance for HPC, *E. coli* and TC for the treated water from MW and SF DWTPs

Statistical results using t-test for HPC for the treated water from MW and SF DWTPs had a p = 0.305 which is greater than $p > 0.05$ shown in Table 11, thus showing that the two plants were not statistically significantly different in HPC CFU/mL. The HPC results from both plants were all compliant with the set limit by in the SANS 241:2015 drinking water standard which states that the HPC bacteria counts must not exceed 1 000 CFU/mL. The compliance of the treated desalinated drinking water from both plants shows the treatment efficiency of the plants and also the adequacy of the applied chlorine disinfectant residuals. The t-test results for the total coliforms CFU/100 mL in the treated water from Monwabisi and Strandfontein DWTPs had a p value= 0.504, represented in Table 11, which is greater than $p > 0.05$, thus showing that the two plants were not statistically significantly different in total coliforms for the two desalination plants. Statistical results using t-test for the *E. coli* in treated water for SF treated and MW treated water gave a $p = 0.305$ and thus were greater than $p > 0.05$ indicating that the results were statistically not significantly different. This indicates both plants treat the water that conform and meet the standard guideline which specifies that in treated water, *E. coli* must not be detected hence they are statistically not significant. The results showing no statistical significant difference may be due to the fact that these two desalination plants use the same treatment technology for treating the seawater into potable water.

Table 11: T- test results for microbiological properties of the treated water

Microbiological determinant	**df**	**t-value**	***P* value**
E. coli (Treated) SF vs MW	149	-1.02726	0.305
HPC (Treated) SF vs MW	147	1.028210	0.305
TC (Treated) SF vs MW	149	-0.668552	0.504

*Df = degrees of freedom; T = t value; p value = probability; red colour = p < 0.05

3.12.2 T- test for physical and chemical properties of the raw and treated water from MW and SF DWTPs.

Independent sample t- tests were done to determine the efficiency of the treatment at the desalination plants by comparing the raw and treated water using TDS, EC, alkalinity, pH, nitrates, nitrites and chlorides. These parameters were selected as they are known to indicate the concentration of dissolved salts in seawater. Since the source water used for the production of treated water was seawater, these parameters were selected. Other chemical parameters were excluded for statistical analysis due to the limited monitoring of the raw water from both plants; therefore, treatment efficiency from raw water to treated water was determined using the aforementioned parameters.

Table 12: T-test results of the raw and treated water from Monwabisi DWTP using physical and chemical properties

Chemical/physical determinant	dF	t-value	*P* value
		Monwabisi	
TDS	100	129.9851	<0.001
EC	119	152.0691	<0.001
pH	120	-22.1651	<0.001
Alkalinity	111	12.38147	<0.001
Chloride	128	36.35096	<0.001
Nitrate	122	1.699072	0.091
Nitrite	121	-0.123513	0.090

*Df = degrees of freedom; t = t value; *p* value = probability; red colour = $p < 0.05$

Table 13: T-test results of the raw and treated water from Strandfontein DWTP using physical and chemical properties

Chemical/physical determinant	dF	t-value	*P* value
		Strandfontein	
TDS	112	141.3244	<0.001
EC	112	141.4249	<0.001
pH	112	-20.9286	<0.001
Alkalinity	116	14.69813	<0.001
Chloride	120	78.92534	<0.001
Nitrate	113	2.265501	<0.001
Nitrite	114	-0.318217	0.750

*Df = degrees of freedom; T = t value; *p* value = probability; red colour = $p < 0.05$

The t-test results revealed significant statistical difference $p < 0.001$ for TDS (mg/L), EC (mS/m), pH (none), chlorides (mg/L) and alkalinity (mg/L) in the raw and treated water from Monwabisi and Strandfontein DWTPs. The significant reduction of these parameters from the initially high concentration in the raw water compared to the treated drinking water showed the efficiency of the reverse treatment technology in removing the dissolved salts previously found in the raw water and production of drinking water of low TDS, EC, pH, alkalinity and chlorides of acceptable water quality. Since drinking water with high TDS, chloride, alkalinity and EC is associated with high dissolved salts and decreased palatability, the reduction of these parameters shows that the drinking water produced by these two plants can be considered aesthetically good with increased palatability. EC, TDS, alkalinity and chlorides in this study were assessed as they are an important water quality parameter for determining the salt content of water which is essential particularly in desalinated drinking water due to its brackish nature. The pH of the raw and treated water remained basic throughout the study period due to the buffering nature of seawater. An increase in pH from the raw to the treated water was observed due to the addition of lime as part of the pH adjustment treatment process used in desalination which is known for increasing pH.

Statistical results using t-tests for nitrates in Strandfontein revealed $p < 0.001$ showing a significant difference. Reverse osmosis used at Strandfontein desalination plant is one of the techniques that are used in water quality to remove nitrates and some salts from water through a semi-permeable membrane by means of a pressure gradient. In drinking water, the presence of nitrate may indicate organic pollution which may emanate from agriculture, industrial and domestic waste amongst other sources (Batool *et al*., 2018). The significant change and reduction of the nitrates in the raw and treated water shows that the reverse osmosis technique was effective in removing of nitrates. T- tests results for nitrate levels in the raw and treated water from Monwabisi had p value of 0.091 which is greater than 0.05 thus showing no significant difference. Since there were low nitrate levels in the raw water from Monwabisi DWTP, this may explain why there was no significant difference between the raw and treated water.

Nitrite levels from Monwabisi and Strandfontein DWTPs had p value of 0.090 and 0.750 respectively which is greater than 0.05 showing that the raw and treated water for nitrite levels was statistically not significant. The nitrites found in the raw water and treated from Strandfontein and Monwabisi DWTPs was relatively low with mean value of 0.05 mg/L nitrites for both plants and within acceptable levels as SANS 241:2015 recommends limits of ≤ 0.9 mg/L for nitrites. Since there were low nitrite levels in the raw water from Monwabisi and Strandfontein DWTP, this may explain why there was no statistical significant difference between the raw and treated water. Water with reduced nitrite levels is ideal as elevated nitrite levels in drinking water may cause some health risks, particularly for infants and pregnant women (El Baba *et al*., 2020).

3.13 Toxicity testing results using three test organisms on the raw water, treated water and brine effluent

The insufficient elimination of chemical pollutants following treatment and subsequent risks/implications for the environment and humans due to chemical exposure necessitates the use of toxicity testing as a supplementary tool to traditional water quality assessments. In the present study toxicity tests using three test organisms; *P. tricornutum*, *A. franciscana* and *V. fischeri* were used to test for potential toxicity effects of the raw water, treated drinking water and the brine effluent from Monwabisi and Strandfontein DWTPs on the test organisms. Each toxicity test was measured as a unique single sample and mean values were not applicable, thus all the toxicity test graphs had no error bars to show standard deviation.

3.13.1 Algal growth inhibition test for Strandfontein and Monwabisi

P. tricornutum, a marine diatom, was used in this study to assess the potential toxicity effects of the raw water, treated drinking water and brine effluent from the two desalination plants. The single concentration screening test, toxicity was measured as percentage algal growth inhibition or stimulation. The endpoint measured the algal growth rate, which was measured in terms of cell density as recommended by numerous guidelines (ASTM, 2004; ISO, 2012; OECD, 2011). Growth measurements after 72-hour incubation for both test and control sets and the results are shown in Figure 40 and 41.

The SF raw water samples indicated growth inhibition less than 20%, indicating low toxicity in samples. The three toxicity tests for each test organisms were not done in April for Strandfontein desalination plant due to non-sampling. MW raw water samples indicated growth ranging from 39% inhibition in April to 15% growth stimulation in July. The raw water results showed less toxicity as most results were significantly lower than the other two matrices which is the treated water and the brine effluent for both desalination plants. This is because the raw samples resemble the natural system where this diatom is mostly found ubiquitously. The raw sample is not altered in any way as it is water before any pre-treatment or post-treatment processes, thus it does not contain any chemicals which are often added during treatment process of desalination. The negative results shown in Figure 41 for the raw water from Monwabisi in the month of July shows that there was growth stimulation observed relative to the control. The minimal growth inhibition and in some instances the growth stimulation shown by the increase in algal growth in the raw water from both plants may be due to the absence of toxic chemicals often used in the desalination process. Also the availability of trace nutrients and minerals in the raw water may have stimulated the growth of algae, as these are known to influence algal growth (Harbi *et al*., 2017).

All SF treated samples indicated growth inhibition more than 80%, ranging from 96 to 111 % algal growth inhibition indicating high toxicity in samples. MW treated samples indicated extremely high toxicity with growth inhibition ranging from 98 % in May to 113 % in August. The treated samples show that there was inhibition of the algae as the shown by the lack of growth in all the samples depicting the absence of the algae or rather the inhibition. The free chlorine in the treated water was tested and for samples that were above 0.2 mg/L sodium thiosulphate was added to the samples to neutralize the free chlorine to non-toxic levels of (<0.2 mg/L), which was done to eliminate potential toxicity from increased chlorine levels. The treated samples for both Monwabisi and Strandfontein showed the most toxicity by inhibition of algal growth compared to the unexposed control. This may be due to some chemicals used in the treatment process of desalination which may have inhibited the growth of *P. tricornutum*.

Previous research by Patterton (2013) showed that water produced from seawater desalination may be still contaminated with trace amounts of complex pollutants even after the reverse osmosis treatment process. Although most of the chemical determinants that were checked for compliance during the study period were mostly compliant with SANS 241: 2015 numerical limits, toxicity assays have the ability to incorporate the effect of the complex mixture of pollutants even at low concentration on test organisms including effects of unidentified pollutants (Žaltauskait *et al*., 2014; Berenguel-Felices *et al*., 2020). Toxicity assays, unlike traditional water quality assessments have the ability to show overall response from multiple contaminants such as heavy metals, including all the possible effects; possible additive, synergistic and antagonistic effects to be determined (Whadhia and Thomson, 2007; Žaltauskait *et al.,* 2014; Kocbus, 2015). However, numerous efforts are being carried out to interpret toxicological and kinetic data from in vitro bioassays to determine human health based threshold values from these assays (Allen *et al*., 2014; Bell *et al*., 2018). Since the translation of human based threshold levels using bioassays remains a gap in knowledge, the compliance of individual water quality parameters stipulated by SANS 241: 2015 for the treated water from both plants may be considered safe for human consumption.

Seawater reverse osmosis desalination plants abstract water from the sea and use it as a feed-water for the desalination process and generate concentrated brine effluent which is normally diluted and discharged back into the coastal environment (Peterson *et al*., 2019). The results for the brine effluent samples from Strandfontein ranged from 7 % to 75 % growth inhibition shown in Figure 40. The results for the brine effluent samples from Monwabisi ranged from -23 % growth stimulation to 66 % growth inhibition are shown in Figure 41. The algal growth inhibition results for the brine effluent from both plants showed some toxicity in algal growth inhibition which was lower than that of the treated samples. Brine effluent contains high salt concentrations which could have also led to the inhibition of the algae as a result of accumulation of toxic by-products from the increased salt concentration. Furthermore, brine effluent samples are known to comprise of several chemicals such as bio-fouling control additives, anti-scalants, biocides, neutralized acids and bases used for cleaning the membranes, coagulants and chlorine by-products which are added during desalination pre-treatment and post-treatment processes which may be toxic to algae (Missimer and Maliva, 2018; Kress, 2019). Thus, this may also constitute as a source of contamination which lead to algal growth inhibition observed in the brine effluent samples. The toxicity of the brine effluent from both effluents on the algae may represent a risk to the receiving marine environment where these plants are discharging.

Several studies have highlighted the marine environmental impact of the brine effluent discharged produced by seawater desalination plants (Jones *et al*., 2019; Kress *et al*., 2019). Researchers have suggested that environmental authorities from different countries should implement environmental impact studies and establish compliance limits and monitoring programmes which would inspect whether the measures are being adopted adequately to minimize the impacts on the receiving environment (Berenguel-Felices *et al*., 2020). Thus, the toxicity of the brine effluent to the marine algae may need further investigation to determine the extent of the impact of these desalination plants on the receiving water ecosystem and the establishment of compliance limits and monitoring of the discharged effluent in order to protect the marine environment and its aquatic organisms.

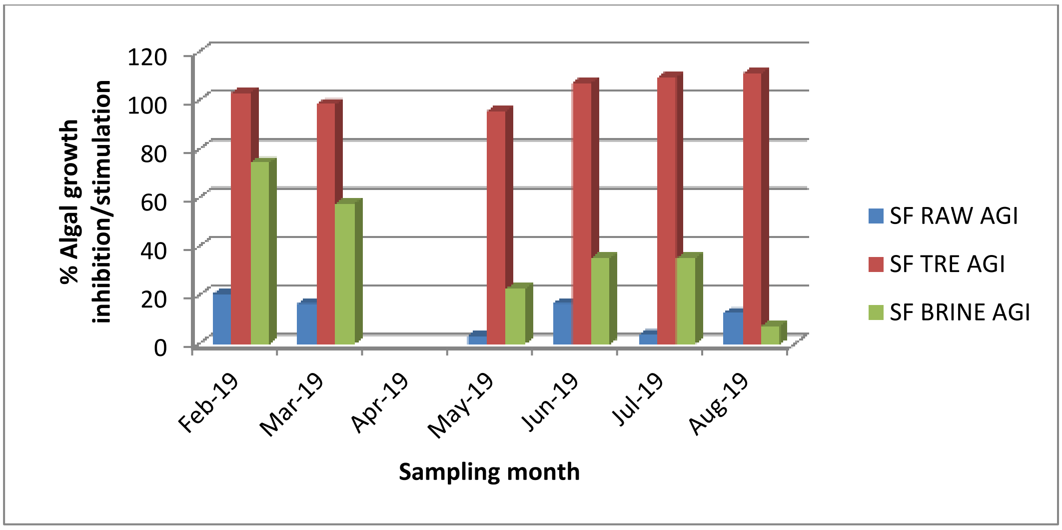

Figure 40: Percentage algal growth inhibition or stimulation results for samples from Strandfontein (raw water, treated water & brine effluent).

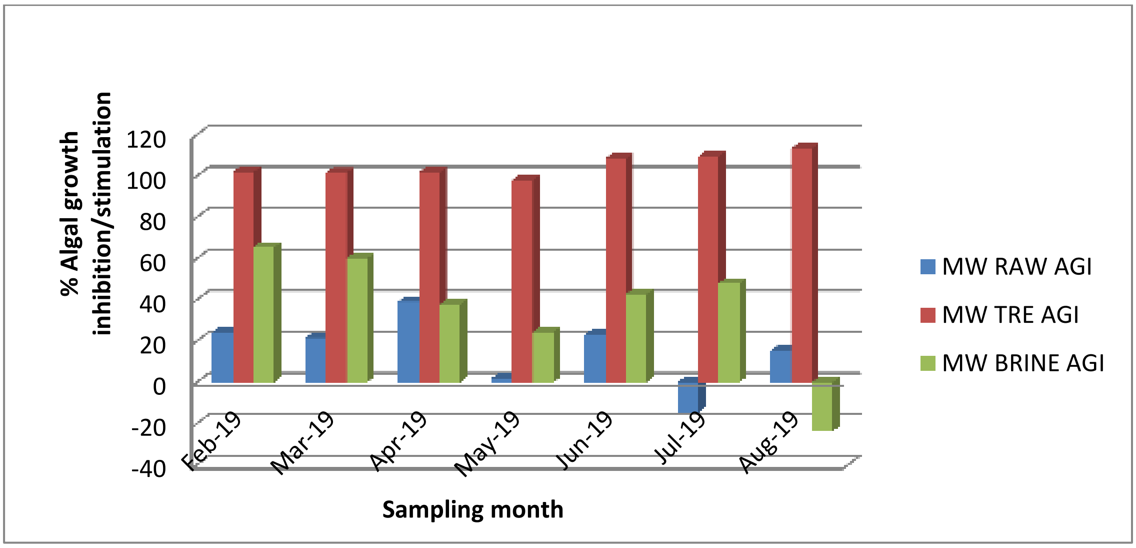

Figure 41: Percentage algal growth inhibition or stimulation for samples from Monwabisi (raw water, treated water & brine effluent).

3.13.2 *A. franciscana* mortality test results for Monwabisi and Strandfontein

Strandfontein and Monwabisi raw water sample toxicity testing results using *A. franciscana* ranged from 0 % to 3 % mortality respectively depicted in Figures 42 and 43. A control sample containing only the *A. franciscana* nauplii were also used for all the matrices tested (raw water, treated water & brine effluent). No mortality was observed in the larvae in the control groups. The results showed that the raw water samples for both Monwabisi and Strandfontein were not toxic to the *A. franciscana nauplii*, these crustaceans are known to survive in environments with high salinities. Less than 10% mortality indicate that samples are not toxic same as control.

The SF treated water samples had mortalities that ranged from 20 % to 73 % mortality shown in Figure 42. MW treated samples had mortalities that ranged from 13 % to 57 % shown in Figure 43. This showed that the treated samples were toxic to *A. franciscana*, which could be attributed to the lack of salt in the treated water as the salts are removed during the desalination process. These organisms are known to survive short periods of time in freshwater, however they cannot reproduce in it (Kumar and Babu, 2015). The mortality of *A. franciscana* may also be due to the presence of chemicals used in the desalination treatment technology which may seep through into the final treated water. Toxicity of treated water may therefore need to be evaluated against non-marine organisms including freshwater test organisms.

The brine effluent that is discharged by reverse osmosis desalination plants frequently contains high quantities of chemicals which may impair the coastal water quality as well as the normal functioning of marine ecosystems (Lattemann & Höpner, 2008; Elsaid *et al*., 2020). The acute toxicity assay using *A. franciscana* showed that the Strandfontein brine effluent samples results ranged from 0 % to 3 % mortality. Monwabisi brine effluent samples results ranged from 0 % to 13 % mortality. This shows that the brine effluent was not toxic to the *A. franciscana*. These organisms are known to be adapted to extreme hypersaline conditions, which may suggest why the brine effluent was not toxic to this test organism.

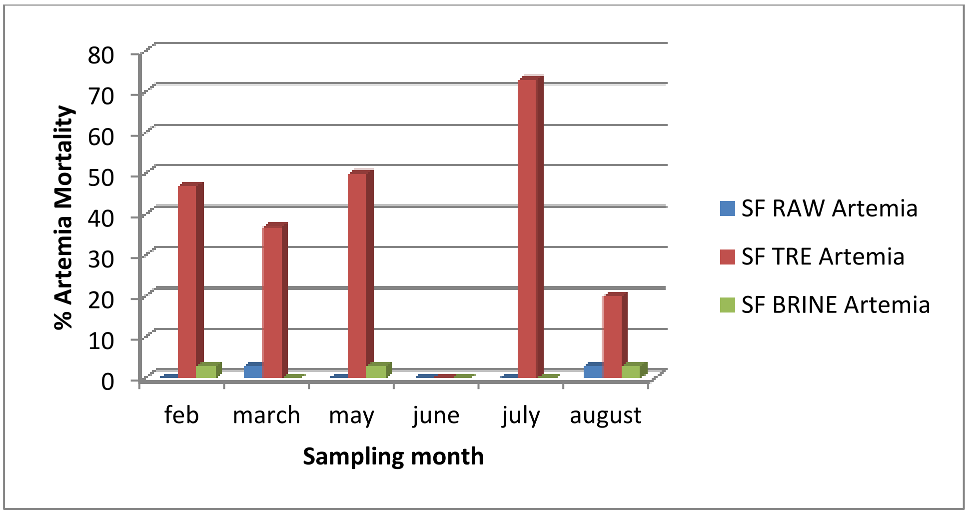

Figure 42: Percentage *A. franciscana* mortality results for samples from Strandfontein (raw water, treated water & brine effluent).

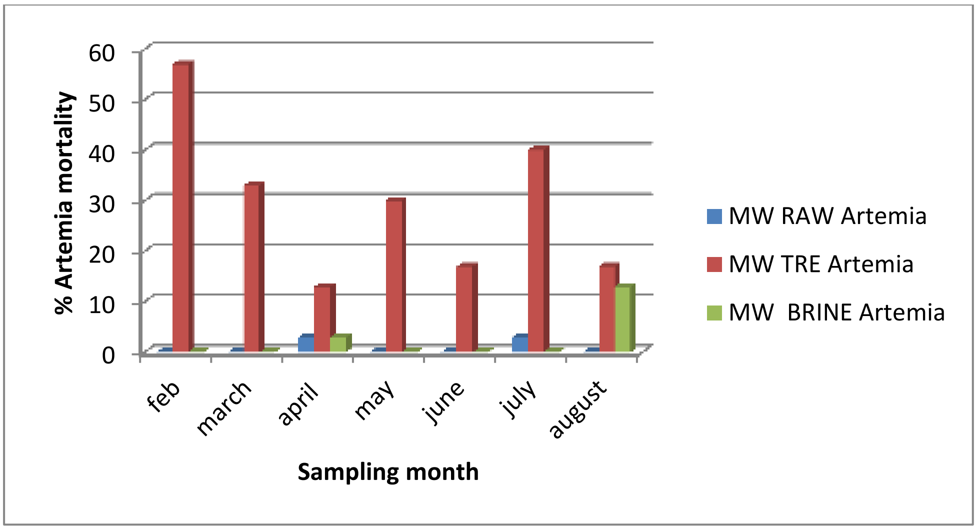

Figure 43: ***A. franciscana*** **% mortality results for samples from Monwabisi (raw water, treated water & brine effluent).**

3.13.3 *V. fischeri* bioluminescence toxicity test

The principle of the *V. fischeri* bioluminescence inhibition test is that when this marine bacterium is exposed to toxic substances the natural luminescence of the bacteria is inhibited by the presence of toxic substances, thus the inhibition of the luminescence of the bacterium is correlated to the degree of the organism's stress level (Rotini *et al.*, 2017). After 15 and 30 min of exposure to the desalinated raw water, treated water and brine effluent sample, the final luminescence was measured. Toxicity results in terms of bioluminescence inhibition or stimulation after 30 min exposure of the control and water samples are presented in Figure 44 and 45 respectively. *V. fischeri* results for MW raw water ranged from -45.86 % to -0.19 % for the raw water; the treated water samples ranged from -48.47 % to -0.52 % and lastly for the brine effluent the results ranged from -30.37 % to 2.08 %. The *V. fischeri* results for the raw water from SF ranged from -6.58 % to 45.29 %, the treated water results ranged from -19.9 % to 0.09 % and lastly for the brine effluent the results ranged from -16.55 % to 3.23 %. The % bioluminescence inhibition detected in SFRAW in the month of June may be associated with contaminants which were present in the raw water, since the treated water from the same month showed bacterial stimulation. The *V. fischeri* bioluminescence test results from both plants for the three matrices (raw water and treated water and the brine) mostly showed minimal inhibition of bioluminescence and in some instances showed some bacterial stimulation. Since the bioluminescence of this bacterium is known to be reduced by the presence of toxic contaminants, the negative results shown in Figures 44 and 45 showing stimulation of the bacterium's natural bioluminescence suggests the absence of toxic contaminants in the raw water, treated water and brine effluent.

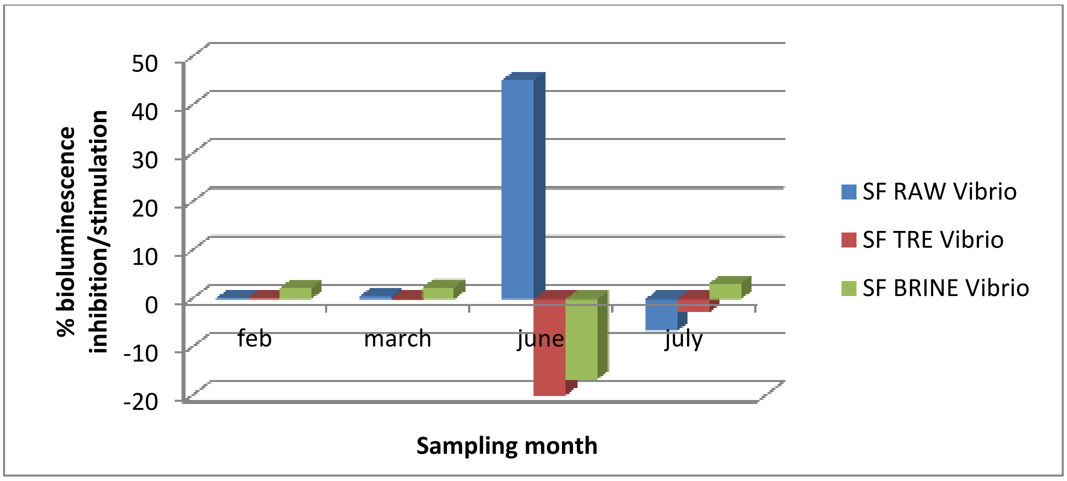

Figure 44: ***V. fischeri*** **% bioluminescence inhibition or stimulation results for SF samples (raw, treated & brine) after 30-minute exposure.**

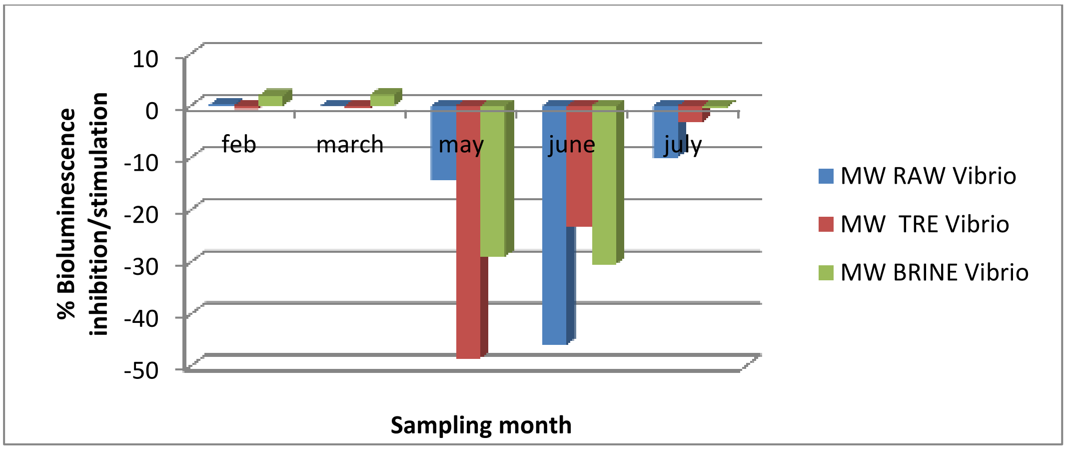

Figure 45: ***V. fischeri*** **bioluminescence inhibition or stimulation results for MW samples (raw, treated & brine) after 30-minute exposure.**

CHAPTER 5 SUMMARY, CONCLUSION AND RECOMMENDATIONS

5.1 Summary

Water is an essential element required by most organisms. Water quality monitoring plays a crucial role in terms of evaluating the acceptability of drinking water which is defined in terms of microbiological, physical and chemical determinants. Toxicity assays are also often carried out to supplement the aforementioned traditional ways of monitoring drinking water quality. Water quality monitoring is essential since there are numerous conditions and activities which influence water quality including natural and anthropogenic sources. The treated drinking water from Monwabisi and Strandfontein DWTPs was compared to the national guideline SANS 241:2015 which specifies numerical limits for drinking water.

The microbiological quality of the treated drinking water from the two plants was relatively of good quality. The microbiological results for comparison of the raw and treated water showed a statistical significant difference which points out to the efficacy of the treatment due to the subsequent reduction in microbial counts in the final treated water. The physical and chemical parameters used for determining treatment efficiency using TDS, nitrates, EC, alkalinity, pH and chloride for the raw and treated water from Strandfontein and Monwabisi showed that there was a marked decrease in concentration of these parameters. The concentration of these parameters for the treated water were within SANS 241 (2015) limits and would not pose a risk to both the environment (raw water) and the consumers (drinking water). Furthermore, t-tests for the same parameters revealed a statistical significant difference from the raw water to the treated drinking water, indicative of treatment efficiency. This was due to the reduction in concentration of these parameters following the treatment process using a membrane thus making the water from Monwabisi and Strandfontein of good quality for drinking.

The toxicity tests using three test organisms (*P. tricornutum*, *V. fischeri* and *A. franciscana*) to test for potential effects caused by the raw water, treated water and the brine effluent were determined. The results showed that the raw water from both plants was the least toxic to marine algae *P. tricornutum* and marine crustacean *A. franciscana*. The minimal toxicity of the raw water on algal growth inhibition of *P. tricornutum* and least mortality of *A. franciscana* may be due to the absence of toxic chemicals in the raw water since the raw water is before the addition of any chemicals used in pre-treatment or post-treatment processes which may affect *P. tricornutum* and *A. franciscana*. The growth stimulation of *P. tricornutum* is normally associated with the presence of nutrients and minerals in trace amounts (Harbi *et al*., 2017), thus this may be associated with the depicted increase in growth results of *P. tricornutum* when exposed to the raw samples from Monwabisi and Strandfontein DWTP.

The study results revealed that the treated water from the two desalination plants showed the most toxicity to *P. tricornutum* and *A. franciscana*. *A. franciscana* is known to be highly adapted to saline environments and the removal of salt during desalination may have influenced the recorded mortality since these organisms are known to survive short periods of time in freshwater and cannot reproduce in it (Kumar and Babu, 2015). The addition of chemicals in the treatment process may have had an influence on toxicity of the treated water towards *P. tricornutum* and *A. franciscana*. The brine effluent also showed some toxicity for *P. tricornutum.* Since there is a gap in the translation of human based threshold levels using toxicity tests, the compliance of individual water quality parameters (microbiological, physical and chemical) stipulated by SANS 241 (2015) for the treated water from both plants showed that the drinking water may be considered safe for human consumption.

The inhibition of algal growth and mortality of *A. franciscana* when exposed to the treated water and brine effluent may be a result of the presence of several chemicals such as coagulants, bio-fouling control additives, anti-scalants, biocides, neutralized acids and chlorine by-products amongst other chemical which are used desalination pre-treatment and post-treatment processes and thus this may constitute as a source of contamination which led to the algal growth inhibition and mortality of *A. franciscana*. The brine effluent is also known to have up to double the strength of salinity compared to the original seawater (Del-Pilar-Ruso *et al*., 2015) which may have contributed to the accumulation of high concentration of salt and subsequent toxicity of the brine to *P. tricornutum. V. fischeri* results for the treated matrices (raw and treated water and the brine) showed bacterial stimulation. The bioluminescence of this bacterium is known to be reduced by the presence of toxic contaminants. The bacterial stimulation may indicate the absence of detrimental chemicals in the raw and treated water and tolerance to brine effluent; hence there was minimal inhibition of the natural bioluminescence of *V. fischeri*.

Whilst desalination offers the benefit of alleviating water scarcity and safeguarding water resources for human use, there are environmental concerns related to the brine effluent that is discharged into the environment and the chemicals used in the treatment process for production of drinking water. The toxicity of the brine effluent showed no toxicity effects to the marine crustacean and bacterium; however, it was toxic to the marine algae. The toxicity of the brine effluent on the algae may need further investigation to determine the extent of the impact of these desalination plants on the receiving water ecosystem, since any changes in aquatic food-chain may be detrimental to overall aquatic ecosystem. Thus, desalination showed minimal impacts on the aquatic organisms and can be used as a good drinking water supply alternative.

5.2. Conclusions

Since the City of Cape Town is progressively becoming threatened as a result of water shortages caused by the recent drought, this compelled the need of alternative water supply such as seawater desalination to meet water demand. With the key findings of the study showing that the water quality of the treated water produced by these two plants was relatively good, desalination can be adopted at a large scale as an alternative from Monwabisi and Strandfontein. Negligibly, there were a few non-compliances for some of the determinants to national standard guideline (SANS 241:2015). The water from these plants seemed to be influenced by natural influences including weather and climate conditions (rainfall, sea tides, and temperature), neighbouring conditions and land use activities. Based on the results of the study, the traditional water quality assessments using chemical, physical, aesthetic and microbiological parameters were effective for monitoring overall water quality of drinking water produced by desalination. The production of treated water with low concentrations of determinants initially found in the raw water showed the efficiency of the reverse osmosis treatment technology, proving its effectiveness as the most used treatment technology of choice world-wide, however it should be acknowledge the economics of the process were not a consideration in this study. Desalinated water produced by the RO plants is not expected to exert any health risks to the consumers. Toxicological assessments showed that the brine produced by desalination plants may have an impact on the neighbouring environment where these plants discharge. Continuous investigations on the impacts of desalination plants need regular monitoring to ensure protection of aquatic ecosystem and the environment. Lastly, the toxicological results on the treated water showed that there is a need to develop alternative toxicity- based methods to determine the potential toxicity of chemicals used in the desalination process which may seep into final water. These chemicals may remain undetected using traditional water quality assessments, posing a health risk to the consumers over long-term. The overall findings of the study showed final water product from both plants was of high quality complying to SANS 241: 2015 and with limited toxicity against test organisms and thus can be considered fit for human consumption. Findings suggest that regular water quality monitoring of the desalination plant is an essential component. In conclusion, the desalination technology offers a great benefit to augment water supplies and narrowing the gap between freshwater availability and water demand particularly to areas of water scarcity to assist the constrained freshwater resources for producing safe drinking water of high quality.

5.3. Recommendations

Although the reverse osmosis technology is the most used desalination technology of choice in South Africa and around the world, desalinated water from seawater may contain trace contaminants as previously shown by previous studies (Patterton, 2013; Petrik *et al*., 2017) even after the RO process. Thus toxicity testing of drinking water produced from desalination, as done here is beneficial in order to check for chronic toxicity of the treated water over a longer period. Monitoring of the treated water is essential in order to ensure that the treatment process functions optimally to ensure the complete breakdown and removal of the lethal contaminants that may be found in the water, thus ensuring that the water distributed by the City is not toxic and safe for consumption.

The inability of chemical characterization in water quality to show specific biological information about potential hazards and toxicity of unidentified complex mixtures necessitates the need to integrate monitoring strategies to include toxicity testing. This will provide a holistic overview on the potential toxicity effects of desalination plants on the receiving environment and the bioavailability of the chemicals determined during routine traditional water quality assessments. Additionally, since previous research has shown that desalinated water may be still contaminated with trace contaminants after the desalination process, toxicity testing as done here is important as it will provide information on the potential toxicity of the product water to determine whether the drinking water produced by reverse osmosis desalination poses no threat to the consumer. Mutagenicity testing is also recommended to determine the long-term effects of consuming this new water resource in the

City of Cape Town which is desalinated water. Also it recommended that legislative authorities in the country should also look into developing guidelines which are specific to governing water produced by desalination. This can include the addition of parameters such dissolved salts; calcium, potassium and magnesium which are major components of seawater, however when present in high concentrations may be associated with some human health risks.

Thus, it is recommended for the municipality to establish compliance discharge limits for the brine effluents by desalination plants and implementation of regular monitoring programmes for drinking water produced by desalination which include toxicity testing as supplementary approach to ensure the delivery of safe drinking water and protection of the marine environment. Lastly, although determination of individual chemical contaminants provides reasonable public health protection, the advances in toxicity assessments and improvement of alternative water supplies show suitable promise for future use of South Africa's alternative water resources.

CHAPTER 6 REFERENCES

Abbas, M., Adil, M., Ehtisham-ul-Haque, S., Munir, B., Yameen, M., Ghaffar, A., and Iqbal, M. (2018). Vibrio fischeri bioluminescence inhibition assay for ecotoxicity assessment: a review. *Science of the Total Environment, 626, 1295-1309*.

Abberton, C. L., Bereschenko, L., van der Wielen, P. W., and Smith, C. J. (2016). Survival, biofilm formation, and growth potential of environmental and enteric Escherichia coli strains in drinking water microcosms. *Applied and environmental microbiology, 82(17), 5320-5331*.

Agoro, M. A., Adeniji, A. O., Adefisoye, M. A., and Okoh, O. O. (2020). Heavy Metals in Wastewater and Sewage Sludge from Selected Municipal Treatment Plants in Eastern Cape Province, South Africa. *Water, 12(10), 2746*.

Alipour, M., Hajiesmaili, R., Talebjannat, M., and Yahyapour, Y. (2014). Identification and antimicrobial resistance of Enterococcus spp. isolated from the river and coastal waters in northern Iran. *The Scientific World Journal, Article ID: 287458*.

Aloke, C., Uzuegbu, I. E., Ogbu, P. N., Ugwuja, E. I., Orinya, O. F., and Obasi, I. O. (2019). Comparative assessment of heavy metals in drinking water sources from Enyigba Community in Abakaliki Local Government Area, Ebonyi State, Nigeria. *African Journal of Environmental Science and Technology, 13(4), 149-154*.

Al-Karaghouli, A., and Kazmerski, L. L. (2013). Energy consumption and water production cost of conventional and renewable-energy-powered desalination processes. *Renewable and Sustainable Energy Reviews, 24, 343-356.*

Akoto, O., Gyamfi, O., Darko, G., and Barnes, V. R. (2017). Changes in water quality in the Owabi water treatment plant in Ghana. *Applied water science, 7(1), 175-186*.

Akter, T., Jhohura, F. T., Akter, F., Chowdhury, T. R., Mistry, S. K., Dey, D., and Rahman, M. (2016). Water Quality Index for measuring drinking water quality in rural Bangladesh: a cross-sectional study. *Journal of Health, Population and Nutrition, 35(1), 1-12*.

Akuno, M. H., Nocella, G., Milia, E. P., and Gutierrez, L. (2019). Factors influencing the relationship between fluoride in drinking water and dental fluorosis: a ten-year systematic review and meta-analysis. *Journal of water and health, 17(6), 845-862*.

Amanidaz, N., Zafarzadeh, A. and Mahvi, A. H. 2015. The Interaction between heterotrophic bacteria and coliform, fecal coliform, fecal Streptococci bacteria in the water supply networks. *Iranian Journal of Public Health, 44(12):1685-169.*

Angelova, I., Ivanov, I., and Venelinov, T. (2020). Origin of Aluminium in the Raw Drinking Water of Sofia City, Bulgaria. *Water, Air and Soil Pollution, 231(9), 1-10.*

Anticó, E., Cot, S., Ribó, A., Rodríguez-Roda, I., and Fontàs, C. (2017). Survey of Heavy Metal Contamination in Water Sources in the Municipality of Torola, El Salvador, through In Situ Sorbent Extraction. *Water, 9(11), 877*.

Arias, C. A., and Murray, B. E. (2012). The rise of the Enterococcus: beyond vancomycin resistance. *Nature Reviews Microbiology, 10(4), 266-278.*

Aris, A. Z., and Looi, L. J. (2015). Estuaries ecosystems health status–profiling the advancements in metal analysis. In Environmental Management and Governance (pp. 429-453). Springer, Cham.

Arome, D., and Chinedu, E. (2013). The importance of toxicity testing. *Journal of Pharmaceutical and BioSciences, 4, 146-148.*

Arroyo, M. G., Pérez-Herranz, V., Montanes, M. T., García-Antón, J., and Guinon, J. L. (2009). Effect of pH and chloride concentration on the removal of hexavalent chromium in a batch electrocoagulation reactor. *Journal of Hazardous Materials, 169(1-3), 1127-1133.*

ASTM (American Society for Testing and Materials). (2012). Standard Guide for Acute Toxicity Test with the Rotifer Brachionus. Method E1440-91.

Aydemir, S, Sönmez, O., and Sakin, E. (2005). The effects of commonly used chemical substances on water quality in Geliş, Tarihi. *Soil Science, 9(2), 1-10.*

Bagatin, R., Klemeš, J. J., Reverberi, A. P., and Huisingh, D. (2014). Conservation and improvements in water resource management: a global challenge. *Journal of Cleaner Production, 77, 1-9.*

Barceló, D., Žonja, B., and Ginebreda, A. (2020). Toxicity tests in wastewater and drinking water treatment processes: A complementary assessment tool to be on your radar. *Journal of Environmental Chemical Engineering, 8(5), 104262.*

Batayneh, A. T. (2012). Toxic (aluminum, beryllium, boron, chromium and zinc) in groundwater: health risk assessment. International journal of environmental science and technology, 9(1), 153-162.

Batool, A., Samad, N., Kazmi, S. S., Ghufran, M. A., Imad, S., Shafqat, M., and Mahmood, T. (2018). Spring water quality and human health: an assessment of natural springs of margalla hills Islamabad zone–III. *International Journal of Hydrology, 2, 41-46.*

Bedada, T.L., Mezemir, W.D., Dera, F.A., Sima, W.G., Gebre, S.G., Edicho, R.M., Biegna, A.G., Teklu, D.S., and Tullu K.D. 2018. Virological and bacteriological quality of drinking water in Ethiopia. *Applied Water Science, 8(70):1-6.*

Belkin, N., Rahav, E., Elifantz, H., Kress, N., and Berman-Frank, I. (2017). The effect of coagulants and antiscalants discharged with seawater desalination brines on coastal microbial communities: a laboratory and in situ study from the southeastern Mediterranean. *Water research, 110, 321-331.*

Benoiston, A. S., Ibarbalz, F. M., Bittner, L., Guidi, L., Jahn, O., Dutkiewicz, S. and Bowler, C. (2017). The evolution of diatoms and their biogeochemical functions. Philosophical Transactions of the Royal Society B. *Biological Sciences, 372(1728), 20160397.*

Berenguel-Felices, F., Lara-Galera, A., and Muñoz-Medina, M. B. (2020). Requirements for the construction of new desalination plants into a framework of sustainability. *Sustainability, 12(12), 5124.*

Bhattacharya, A., and Sharma, S., (2017). Drinking water contamination and treatment techniques. *Applied Water Science, 7(3), 1043-1067.*

Blersch, C. L. (2014). Planning for seawater desalination in the context of the Western Cape water supply system (Doctoral dissertation, Stellenbosch: Stellenbosch University).

Blersch, C. L., and Du Plessis, J. A. (2017). Planning for desalination in the context of the Western Cape water supply system. *Journal of the South African Institution of Civil Engineering, 59(1), 11-21.*

Blokker, E. J., Furnass, W. R., Machell, J., Mounce, S. R., Schaap, P. G. and Boxall, J. B. (2016). Relating water quality and age in drinking water distribution systems using self-organising maps. *Environments, 3(2), 10.*

Birnhack, L., Voutchkov, N., and Lahav, O. (2011). Fundamental chemistry and engineering aspects of post-treatment processes for desalinated water—a review. *Desalination, 273(1), 6-22.*

Blue Flag beach criteria and explanatory notes [document on the Internet]. No date [cited 2020/11/30]. Available from https://static1.squarespace.com/static/55371ebde4b0e49a1e2ee9f6/t/56cc2a59859fd03dbee43223/1456220762132/Beach+Criteria+and+Explanatory+Notes.pdf .

Boehm, A. B., and Sassoubre, L. M. (2014). Enterococci as indicators of environmental fecal contamination. Enterococci: from commensals to leading causes of drug resistant infection Massachusetts Eye and Ear Infirmary, Boston.

Bohulu, E., Ntombela, N., Low, M., Ming, D., and Harding, K. (2019). Drinking seawater: Investigations into desalination. *Procedia Manufacturing, 35, 743-748.*

Botai, C. M., Botai, J. O., De Wit, J. P., Ncongwane, K. P., and Adeola, A. M. (2017). Drought characteristics over the western cape province, South Africa. *Water, 9(11), 876.*

Bowler, C., Allen, A. E., Badger, J. H., Grimwood, J., Jabbari, K., Kuo, A., and Grigoriev, I. V. (2008). The Phaeodactylum genome reveals the evolutionary history of diatom genomes. *Nature, 456(7219), 239-244.*

Boyd, C. E., Tucker, C. S., and Somridhivej, B. (2016). Alkalinity and hardness: critical but elusive concepts in aquaculture. *Journal of the World Aquaculture Society, 47(1), 6-41.*

Butler, T., Kapoore, R. V.and Vaidyanathan, S. (2020). Phaeodactylum tricornutum: a diatom cell factory. Trends in biotechnology, 38(6), 606-622.

Cabral, J. P. (2010). Water microbiology. Bacterial pathogens and water. *International journal of environmental research and public health, 7(10), 3657-3703.*

Caldera, U., and Breyer, C. (2017). Learning curve for seawater reverse osmosis desalination plants: Capital cost trend of the past, present, and future. *Water Resources Research, 53(12), 10523-10538.*

Carbonell, G., Fernández, C., and Tarazona, J. V. (2010). A cost/effective screening method for assessing the toxicity of nutrient rich effluents to algae. *Bulletin of environmental contamination and toxicology, 85(1), 72-78.*

Carrillo-Gómez, J., Durán-Acevedo, C., and García-Rico, R. (2019). Concentration Detection of the E. coli Bacteria in Drinking Water Treatment Plants through an E-Nose and a Volatiles Extraction System (VES). Water, 11(4), 774.

Carvalho, R. N., Arukwe, A., Ait-Aissa, S., Bado-Nilles, A., Balzamo, S., Baun, A., and Lettieri, T. (2014). Mixtures of chemical pollutants at European legislation safety concentrations: how safe are they?. Toxicological sciences, 141(1), 218-233.

Chidya, R. C. G., Sajidu, S. M. I., Mwatseteza, J. F., and Masamba, W. R. L. (2011). Evaluation and assessment of water quality in Likangala River and its catchment area. Physics and *Chemistry of the Earth, Parts A/B/C, 36(14-15), 865-871*

Cho, B. Y., Kim, H. W. and Shin, Y. S. (2015). A Study on boron removal for seawater desalination using the combination process of mineral cluster and RO membrane system. Environmental Engineering Research, 20(3), 285-289.

Cho, S., Jackson, C. R. and Frye, J. G. (2020). The prevalence and antimicrobial resistance phenotypes of Salmonella, Escherichia coli and Enterococcus sp. in surface water. *Letters in applied Microbiology, 71(1), 3-25.*

Choi, C. J., Berges, J. A., and Young, E. B. (2012). Rapid effects of diverse toxic water pollutants on chlorophyll a fluorescence: variable responses among freshwater microalgae. *water research, 46(8), 2615-2626.*

Chowdhury, S., Mazumder, M. J., Al-Attas, O., and Husain, T. (2016). Heavy metals in drinking water: occurrences, implications, and future needs in developing countries. *Science of the total Environment*, *569*, 476-488.

Chughtai, M., Mustafa, S., and Mumtaz, M. (2014). Study of physicochemical parameters of rainwater: A case study of Karachi, Pakistan. *American Journal of Analytical Chemistry, 5, 235-242.*

City of Cape Town (CoCT). (2018). City of cape town state of the environment report. Available from

https://resource.capetown.gov.za/documentcentre/Documents/City%20research%20reports%20and%20review/State%20of%20the%20Environment%202018%20Report.pdf

City of Cape Town (CoCT). (2019). Know your Coast, 2019: Key findings from over 10 000 sample bacterial tests at 90 sites along 307 km of coastline. No date [cited 2020/11/30]. Available from: http://resource.capetown.gov.za/documentcentre/Documents/City%20research%20reports%20and%20review/Know%20Your%20Coast%202019.pdf.

Cooley, H., Ajam, N. and Heberger, M. (2013). Key Issues in Seawater Desalination in California: Marine Impacts, Pacific Institute Oakland, CA.

Collivignarelli, M. C., Abbà, A., Benigna, I., Sorlini, S., and Torretta, V. (2018). Overview of the main disinfection processes for wastewater and drinking water treatment plants. *Sustainability, 10(1), 86*.

Copat, C., Arena, G., Fiore, M., Ledda, C., Fallico, R., Sciacca, S., and Ferrante, M. (2013). Heavy metals concentrations in fish and shellfish from eastern Mediterranean Sea: consumption advisories. *Food and chemical toxicology, 53, 33-37.*

Cortés, A. A., Sánchez-Fortún, S., García, M., Martínez, H., and Bartolomé, M. C. (2018). Toxicological assessment of binary mixtures and individually of chemical compounds used in reverse osmosis desalination on Artemia franciscana nauplii. *Latin american journal of aquatic research, 46(4), 673-682.*

Cosgrove, W. J., & Loucks, D. P. (2015). Water management: Current and future challenges and research directions. *Water Resources Research, 51(6), 4823-4839*.

Costas-Gil, A., Díaz-Tielas, C., Reigosa, M. J., & Sánchez-Moreiras, A. M. (2015). Algicide activity of rosmarinic acid on Phaeodactylum tricornutum. *Journal of Allelochemical Interactions, 1(1), 39-47.*

Cruzeiro, C., Amaral, S., Rocha, E., and Rocha, M. J. (2017). Determination of 54 pesticides in waters of the Iberian Douro River estuary and risk assessment of environmentally relevant mixtures using theoretical approaches and Artemia salina and Daphnia magna bioassays. *Ecotoxicology and environmental safety, 145, 126-134*.

CSIR (Council for Scientific and Industrial Research). (2014). Cape Town Outfalls Monitoring Programme: Surveys made in 2015/2016. CSIR Report CSIR/NRE/ECOS/IR/2017/0035/B.

Dalmieda, J., and Kruse, P. (2019). Metal cation detection in drinking water. *Sensors, 19(23), 5134.*

Damania, R., Desbureaux, S., Hyland, M., Islam, A., Moore, S., Rodella, A.S., Russ, J. and Zaveri, E. (2017). Uncharted Waters: The New Economics of Water Scarcity and Variability.World Bank, Washington, DC © World Bank.

Damkjaer, S., and Taylor, R. (2017). The measurement of water scarcity: Defining a meaningful indicator. *A Journal of the Human Environment, 46(5), 513-531.*

Darre, N. C., and Toor, G. S. (2018). Desalination of water: a review. *Current Pollution Reports, 4(2), 104-111*.

Darling, E. S., and Côté, I. M. (2008). Quantifying the evidence for ecological synergies. *Ecology letters, 11(12), 1278-1286*.

Darwish, M., Hassabou, A. H., and Shomar, B. (2013). Using Seawater Reverse Osmosis (SWRO) desalting system for less environmental impacts in Qatar. *Desalination, 309, 113-124*.

Dawoud, M.A. and Al Mulla, M.M. (2012).Environmental impacts of seawater desalination: Arabian Gulf case study, International Journal of Environment and Sustainable development. 1, 22-37.

Del-Pilar-Ruso, Y., Martinez-Garcia, E., Giménez-Casalduero, F., Loya-Fernández, A., Ferrero-Vicente, L. M., Marco-Méndez, C., and Sánchez-Lizaso, J. L. (2015). Benthic community recovery from brine impact after the implementation of mitigation measures. *Water research, 70, 325-336.*

Devane, M., Moriarty, E., Weaver, L., Cookson, A., & Gilpin, B. (2020). Fecal indicator bacteria from environmental sources; strategies for identification to improve water quality monitoring. *Water Research, 116204*.

Devi M, Kirubagaran RL. (2016).Geographic information systems and multivariate analysis to evaluate fecal bacterial pollution in coastal waters of Andaman, India. *Environmental Pollution, 214, 45-53.*

Dingemans, M. M., Baken, K. A., van der Oost, R., Schriks, M., and van Wezel, A. P. (2019). Risk-based approach in the revised European Union drinking water legislation: Opportunities for bioanalytical tools. *Integrated Environmental Assessment and Management, 15(1), 126-134.*

Djuma, H., Bruggeman, A., Eliades, M., and Lange, M. A. (2016). Non-conventional water resources research in semi-arid countries of the Middle East. *Desalination and Water Treatment, 57(5), 2290-2303.*

Drzymała, J. and Kalka, J. (2020). Elimination of the hormesis phenomenon by the use of synthetic sea water in a toxicity test towards Aliivibrio fischeri. Chemosphere, 248, 126085.DWAF (Department of Water Affairs and Forestry, South Africa). (1996). South African Water Quality Guidelines. Volume 1: Domestic Water Use Second Edition. Department of Water Affairs and Forestry, Pretoria.

Duffus, J. H. (2002). "Heavy metals" a meaningless term? (IUPAC Technical Report). *Pure and applied chemistry, 74(5), 793-807*.

D'Ugo, E., Marcheggiani, Z., D'Angelo, A. M., Caciolli, S., Puccinelli, C., Giuseppetti, R., Marcoaldi, R., Romanelli, C., and Mancini, L. (2018). Microbiological water quality in the medical device industry in Italy. *Microchemical Journal, 136, 293-299*.

Duressa, G., Assefa, F., & Jida, M. (2019). Assessment of bacteriological and physicochemical quality of drinking water from source to household tap connection in Nekemte, Oromia, Ethiopia. *Journal of environmental and public health*, 2129792.

DWAF. (Department of Water Affairs and Forestry, South Africa). (1996).South African Water Quality Guidelines.1: Domestic Water Use (2nd edn.) Department of Water Affairs & Forestry,Pretoria, South Africa.

DWAF (Department of Water Affairs and Forestry, South Africa). (1998) Quality of domestic water supplies assessment guide, 2nd edn. Department of Water Affairs of Forestry, Department of Health and Water Research Commission, Pretoria, South Africa.

DWAF (Department of Water Affairs and Forestry, South Africa). (2005). Drinking water quality in South Africa :A consumer guide. Department of Water Affairs and Forestry, Pretoria.

DWAF (Department of Water Affairs and Forestry). (2008). Water for Growth and Development in South Africa. Pretoria: Department of Water Affairs and Forestry, South Africa.

DWA (Department of Water Affairs). (2010). Integrated Water Resource Planning for South Africa: A Situation Analysis 2010. Pretoria: Department of Water Affairs, South Africa.

DWA (Department of Water Affairs). (2013). National Water Resource Strategy: Water for an Equitable and Sustainable Future. Pretoria: Department of Water Affairs, South Africa.

Edberg, S. C. L., Rice, E. W., Karlin, R. J., and Allen, M. J. (2000). Escherichia coli: the best biological drinking water indicator for public health protection. *Journal of applied microbiology, 88(S1), 106S-116S.*

Edokpayi, J. N., Rogawski, E. T., Kahler, D. M., Hill, C. L., Reynolds, C., Nyathi, E., and Dillingham, R. (2018). Challenges to sustainable safe drinking water: a case study of water quality and use across seasons in rural communities in Limpopo province, South Africa. *Water*, *10*(2), 159.

Efe, S. I., Ogban, F. E., Horsfall, M. J., and Akporhonor, E. E. (2005). Seasonal variations of physico-chemical characteristics in water resources quality in western Niger Delta region, Nigeria. *Journal of Applied Sciences and Environmental Management, 9(1), 191-195*.

El Baba, M., Kayastha, P., Huysmans, M., and De Smedt, F. (2020). Evaluation of the groundwater quality using the water quality index and geostatistical analysis in the Dier al-Balah Governorate, Gaza Strip, Palestine. *Water, 12(1), 262*.

Elimelech, M., and Phillip, W. A. (2011). The future of seawater desalination: energy, technology, and the environment. *Science, 333(6043), 712-717.*

Ellis, K., Gowdy, C., Jakomis, N., Ryan, B., Thom, C., Biggs, C. A., and Speight, V. (2018). Understanding the costs of investigating coliform and E. coli detections during routine drinking water quality monitoring. Urban Water Journal, 15(2), 101-108.

Elsaid, K., Kamil, M., Sayed, E. T., Abdelkareem, M. A., Wilberforce, T., and Olabi, A. (2020). Environmental impact of desalination technologies: A review. Science of The Total Environment, 748, 141528.

El Wahab, M., and Hamoda, A. Z. (2012). Effect of desalination plants on the marine environment along the Red sea, Egypt (case study). *International Journal of Marine Science, 28(2), 27-36*.

Enqvist, J. P., and Ziervogel, G. (2019). Water governance and justice in Cape Town: An overview. Wiley Interdisciplinary Reviews: *Water, 6(4), 1354*.

Fakayode, I. B., and Ogunjobi, A. A. (2018). Quality assessment and prevalence of antibiotic resistant bacteria in government approved mini-water schemes in Southwest, Nigeria. *International Biodeterioration & Biodegradation, 133, 151-158.*

Faria, E. C., Brown, B. J. T., and Snook, R. D. (2004). Water toxicity monitoring using Vibrio fischeri: a method free of interferences from colour and turbidity. *Journal of Environmental Monitoring, 6(2), 97-102.*

Feijão, E., Cruz de Carvalho, R., Duarte, I. A., Matos, A. R., Cabrita, M. T., Novais, S. C. and Duarte, B. (2020). Fluoxetine Arrests Growth of the Model Diatom Phaeodactylum tricornutum by Increasing Oxidative Stress and Altering Energetic and Lipid Metabolism. *Frontiers in Microbiology, 11, 1803.*

Fernández-Torquemada, Y., Gónzalez-Correa, J. M., Loya, A., Ferrero, L. M., Díaz-Valdés, M., and Sánchez-Lizaso, J. L. (2009). Dispersion of brine discharge from seawater reverse osmosis desalination plants. *Desalination and Water Treatment, 5(1-3), 137-145*.

Fichet, D., and Miramand, P. (1998). Vanadium toxicity to three marine invertebrates larvae: Crassostrea gigas, Paracentrotus lividus and Artemia salina. *Chemosphere, 37(7), 1363-1368*.

Fritzmann, C., Löwenberg, J., Wintgens, T., and Melin, T. (2007). State-of-the-art of reverse osmosis desalination. *Desalination, 216(1-3), 1-76*.

Fu, Q., Zheng, B., Zhao, X., Wang, L., and Liu, C. (2012). Ammonia pollution characteristics of centralized drinking water sources in China. *Journal of Environmental Sciences, 24(10), 1739-1743*.

Gacem, Y., Taleb, S., Ramdani ,A., Senadjki, S., and Ghaffour N (2012) Physical and chemical assessment of MSF distillate and SWRO product for drinking purpose. *Desalination 290,107–114*.

Gajardo, G. M., and Beardmore, J. A. (2012). The brine shrimp Artemia: adapted to critical life conditions. *Frontiers in physiology, 3, 185*.

Gao, B., Yu, H., Gao, J., Liu, L., Wang, Q., Zhu, C., and Wu, W. (2020). Occurrence and health risk assessment of trace metals in desalinated seawater using two desalination technologies. *PolishJournal of Environmental Studies, 29(5)*.

Ghaffour, E. Mino. (2012). Water scarcity and drought in WANA countries. *Procedia Engineering, 33, 14–29*.

Ghaffour, N., Missimer, T. M., and Amy, G. L. (2013). Technical review and evaluation of the economics of water desalination: current and future challenges for better water supply sustainability. *Desalination, 309, 197-207*.

Ghernaout, D., Elboughdiri, N., and Al Arni, S. (2019). Water Reuse (WR): Dares, restrictions, and trends. *Applied Engineering, 3, 159-170*.

Gil, M.I., Selma, M. V., Suslow, T., Jacxsens, L., Uyttendaele, M., and Allende, A. (2015). Pre-and postharvest preventive measures and intervention strategies to control microbial food safety hazards of fresh leafy vegetables. Critical reviews in food *science and nutrition, 55(4), 453-468*.

Gizaw, Z., Adane, T., Azanaw, J., Addisu, A., and Haile, D. (2018). Childhood intestinal parasitic infection and sanitation predictors in rural Dembiya, northwest Ethiopia. *Environmental health and preventive medicine, 23(1), 1-10.*

Gong, S., Wang, H., Zhu, Z., Bai, Q., & Wang, C. (2019). Comprehensive utilization of seawater in China: a description of the present situation, restrictive factors and potential countermeasures. *Water, 11(2), 397*.

Gopi, R.A., Ayyappan, S., Chandrasehar, G., Krishna Varma, K., and Goparaju, A. (2012). Effect of Potassium Dichromate on the Survival and Reproduction of Daphnia magna. *Bulletin of Environment, Pharmacology and Life Sciences, 1(7), 98 – 94*.

Gosling, S. N., and Arnell, N. W. (2016). A global assessment of the impact of climate change on water scarcity. *Climatic Change, 134(3), 371-385*.

Greve, P., Kahil, T., Mochizuki, J., Schinko, T., Satoh, Y., Burek, P., Fischer, G., and Tramberend, S. (2018). Global assessment of water challenges under uncertainty in water scarcity projections. *Nature Sustainability, 1, 486–494*.

Gude, V. G. (2016). Desalination and sustainability–an appraisal and current perspective. *Water research, 89, 87-106*.

Gude, V.G. (2017). Desalination and water reuse to address global water Scarcity. *Reviews in Environmental Science and Biotechnology, 16, 591–609*.

Gutiérrez-Lucas,L.R., Chávez-Ramírez, L.M., and Arregui, L.(2017). Physico-Chemical and Microbiological Analysis of Water of the “Presa De Los Patos” in the Desierto De Los Leones National Park, Mexico. *Advances in Biological Chemistry, 7, 122-138*.

Gwimbi, P., George, M., and Ramphalile, M. (2019). Bacterial contamination of drinking water sources in rural villages of Mohale Basin, Lesotho: exposures through neighbourhood sanitation and hygiene practices. *Environmental health and preventive medicine, 24(1), 1-7.*

Hale, S.E., Skulcov, L., Pípal, M., Cornelissen, G., Oen, A.M.P, Eek, E and Bielsk, L. (2019). Monitoring wastewater discharge from the oil and gas industry using passive sampling and Danio rerio bioassay as complimentary tools. *Chemosphere, 216, 404-412*.

Hall, LW. And Anderson, R.D. (1995). The influence of salinity on the toxicity of various classes of chemicals to aquatic biota. *Critical Reviews in Toxicology, 25, 281-346*.

Hamidi, M.R., Jovanova, B. and Panovska, T.K. (2014). Toxicological evaluation of the plant products using Brine Shrimp (Artemia salina L.) model. *Macedonian pharmaceutical bulletin, 60, 9-18.*

Harbi, K., Makridis, P., Koukoumis, C., Papadionysiou, M., Vgenis, T., Kornaros, M., and Dailianis, S. (2017). Evaluation of a battery of marine species-based bioassays against raw and treated municipal wastewaters. *Journal of hazardous materials, 321, 537-546*.

Harwood, V. J., Levine, A. D., Scott, T. M., Chivukula, V., Lukasik, J., Farrah, S. R., and Rose, J. B. (2005). Validity of the indicator organism paradigm for pathogen reduction in reclaimed water and public health protection. *Applied and environmental microbiology, 71(6), 3163-3170*.

He, R., Yu, T., Du, J., and Qu, C. (2019, March). Study on the Existence Form and Removal of Boron Acid. In IOP Conference Series, *Materials* Science and Engineering,484, 1,012059.

Hill, I. R. (1993). Guidance Document on Sediment Toxicity Tests and Bioassays for Freshwater and Marine Environments; from the" Workshop on Sediment Toxicity Assessment", Held at Slot Moermond Congrescentrum, Renesse, the Netherlands, 8-10 November 1993. Society of Environmental Toxicology and Chemistry, Europe, SETAC.

Holloway, A., Fortune, G., Zweig, P., Barret, I., Benjamin, A., Chasi, V., and de Waal, J. (2012). Eden & Central Karoo Drought disaster 2009-2011 "the scramble for water", Report,DisasterMitigation for Sustainable Livelihoods Programme, Stellenbosch University.

Holinger, E. P., Ross, K. A., Robertson, C. E., Stevens, M. J., Harris, J. K., and Pace, N. R. (2014). Molecular analysis of point-of-use municipal drinking water microbiology. *Water research, 49, 225-235*.

Hu, H., Li, X., Huang, P., Zhang, Q., and Yuan, W. (2017). Efficient removal of copper from wastewater by using mechanically activated calcium carbonate. *Journal of environmental management, 203, 1-7*.

Hung, D. T., Thi Cuc, V., Thi Bich Phuong, V., Thi Thanh Diu, D., Thi Huyen Trang, N., Phuong Thoa, N., and Van Long, N. (2020). Evaluation of Drinking Water Quality in Schools in a District Area in Hanoi, Vietnam. *Environmental Health Insights, 14, 1-8*.

IARC. (2010). Ingested nitrate and nitrite and cyanobacterial peptide toxins. Lyon: International Agency for Research on Cancer; 1–464 (IARC Monographs on the Evaluation of Carcinogenic Risks to Humans, Volume 94).

Ibrahim, A.G.M., Rashad, A.M. and Dincer, I. (2017). Exergoeconomic analysis for cost optimization of a solar distillation system. *Solar Energy, 151, 22–32*.

Ibrahim, M. N. (2019). Assessing groundwater quality for drinking purpose in Jordan: application of water quality index. *Journal of Ecological Engineering, 20(3)*.

Iloms, E., Ololade, O. O., Ogola, H. J., and Selvarajan, R. (2020). Investigating industrial effluent impact on municipal wastewater treatment plant in Vaal, South Africa. *International journal of environmental research and public health, 17(3), 1096*.

ISO 10253. (2016). Water quality – Marine algal growth inhibition test with skeletonema costatum and Phaeodactylum tricornutum.

ISO, (1998), Water quality — Determination of the inhibitory effect of water samples on the light emission of Vibrio fischeri (Luminescent bacteria test) (first edition)

Intanai, I., Taylor, E. W., and Whiteley, N. M. (2009). Effects of salinity on rates of protein synthesis and oxygen uptake in the post-larvae and juveniles of the tropical prawn Macrobrachium rosenbergii (de Man). Comparative Biochemistry and Physiology Part A. *Molecular and Integrative Physiology, 152(3), 372-378*.

Januário, A. P., Afonso, C. N., Mendes, S. and Rodrigues, M. J. (2020). Faecal indicator bacteria and Pseudomonas aeruginosa in marine coastal waters: is there a relationship?. *Pathogens, 9(1), 13*.

Jenkins, S., Paduan, J., Roberts, P., Schlenk, D. and Weis, J. (2012). Management of Brine Discharges to Coastal Waters Recommendations of a Science Advisory Panel; Southern California Coastal Water Research Project: Costa Mesa, CA, USA.

Jones, E., Qadir, M., van Vliet, M. T., Smakhtin, V., and Kang, S. M. (2019). The state of desalination and brine production: A global outlook. *Science of the Total Environment, 657, 1343-1356.*

Kang, J. S., Eusebio, R. C., and Kim, H. S. (2011). Boron removal by activated carbon and microfiltration for pre-treatment of seawater desalination. *Water Science and Technology: Water Supply, 11(5), 560-567*.

Kavitha, J., Rajalakshmi, M., Phani, A. R., & Padaki, M. (2019). Pretreatment processes for seawater reverse osmosis desalination systems—A review. *Journal of Water Process Engineering, 32, 100926*.

Khan, S., Shahnaz, M., Jehan, N., Rehman, S., Shah, M. T., and Din, I. (2013). Drinking water quality and human health risk in Charsadda district, Pakistan. Journal of *cleaner production, 60, 93-101*.

Kim, E. J., Herrera, J. E., Huggins, D., Braam, J., and Koshowski, S. (2011). Effect of pH on the concentrations of lead and trace contaminants in drinking water: a combined batch, pipe loop and sentinel home study. *Water research, 45(9), 2763-2774*.

Kinuthia, G. K., Ngure, V., Beti, D., Lugalia, R., Wangila, A., and Kamau, L. (2020). Levels of heavy metals in wastewater and soil samples from open drainage channels in Nairobi, Kenya: Community health implication. *Scientific Reports, 10(1), 1-13*.

Koçbaş, F., and Oral, R. (2015). Daphnia magna as a test species for toxicity evaluation of municipal wastewater treatment plant effluents on freshwater cladoceran in Turkey. *Turkish Journal of Fisheries and Aquatic Sciences, 15(3), 619-624*.

Kolm, H. E., Gomes, K. V., Ishii, F. K., and Martins, C. C. (2018). An integrated appraisement of multiple faecal indicator bacteria and sterols in the detection of sewage contamination in subtropical tidal creeks. *International journal of hygiene and environmental health, 221(7), 1032-1039*.

Korajkic, A., Wanjugi, P., Brooks, L., Cao, Y., & Harwood, V. J. (2019). Persistence and decay of fecal microbiota in aquatic habitats. Microbiology *and Molecular Biology Reviews, 83(4)*.

Kress, N., Shoham-Frider, E., Lubinevski, H. (2019). Marine Monitoring at the Brine Outfalls of the

Palmachim and Soreq Desalination Plants. Final Report for the 2018 Surveys. IOLR Report H22/2019.

Kültz, D. (2015). Physiological mechanisms used by fish to cope with salinity stress. *Journal of Experimental Biology, 218(12), 1907-1914*.

Kungolos, A., Samaras, P., and Koutseris, E. (2003). Using bioassays for testing seawater quality in Greece. *Journal of Environmental Science and Health, Part A, 38(3), 533-544*.

Kurihara, M., and Takeuchi, H. (2018). SWRO-PRO system in “Mega-ton Water System” for energy reduction and low environmental impact. *Water, 10(1), 48*.

Laird, M.C., Wright, A.G., Massie, V. and Clark, B.M. (2017). Marine Environmental Impact Assessment for the Proposed Desalination Plants around the Cape Peninsula, South Africa. Report no. 1768/3 prepared by Anchor Environmental Consultants for Advisian: WorleyParsons Group.

Lamine, I., Alla, A. A., Bourouache, M., and Moukrim, A. (2019). Monitoring of Physicochemical and Microbiological Quality of Taghazout Seawater (Southwest of Morocco): Impact of the New Tourist Resort" Taghazout Bay". *Journal of Ecological Engineering, 20(7).*

Lapointe, B. E., Thacker, K., Hanson, C., and Getten, L. (2011). Sewage pollution in Negril, Jamaica: effects on nutrition and ecology of coral reef macroalgae. Chinese Journal of Oceanology and Limnology, 29(4), 775-789.

Latteman, S. and Höpner, T. (2008). Seawater Desalination. In: Impacts of Brine and Chemical Discharges on the Marine Environment. Desalination Publications, L'Aquila, Italy. Lattemann, S. and Höpner, T. Environmental impact and impact assessment of seawater desalination. *Desalination, 220, 1–15*.

Lattemann, S. (2009). Protecting the marine environment. In Cipollina, A., Micale, G., & Rizzuti, L. (eds.) Seawater Desalination: Conventional and Renewable Energy Processes. *Springer, Berlin Heidelberg, 273- 299.*

Laureano-Rosario, A. E., Symonds, E. M., Rueda-Roa, D., Otis, D., and Muller-Karger, F. E. (2017). Environmental factors correlated with culturable enterococci concentrations in tropical recreational waters: a case study in Escambron Beach, San Juan, Puerto Rico. *International journal of environmental research and public health, 14(12), 1602*.

Lawson, E. O. (2011). Physico-chemical parameters and heavy metal contents of water from the mangrove swamps of Lagos Lagoon, Lagos, Nigeria. Advances in biological research, 5(1), 8-21.

Leedy, P.D. (2005). Practical Research: Planning and Design. 8th Edition. Pearson Merrill Prentice Hall.

Levi, A., Bar-Zeev, E., Elifantz, H., Berman, T., and Berman-Frank, I., (2016). Characterization of microbial communities in water and biofilms along a large scale SWRO desalination facility: site-specific prerequisite for biofouling treatments. *Desalination, 378, 44-52*.

Levy, K., Nelson, K. L., Hubbard, A., and Eisenberg, J. N. (2012). Rethinking indicators of microbial drinking water quality for health studies in tropical developing countries: case study in northern coastal Ecuador. *The American journal of tropical medicine and hygiene, 86(3), 499-507*.

Libralato, G., Ghirardini, A. V., and Avezzù, F. (2010). Seawater ecotoxicity of monoethanolamine, diethanolamine and triethanolamine. *Journal of hazardous materials, 176(1-3), 535-539*.

Ligaraya, M., Kimc, N.H., Parka, S., Parkc, J.S., Parkb, J., Kimc,Y., and Kyung, K.H. (2020).

Energy projection of the seawater battery desalination system using the reverse osmosis system analysis model. *Chemical Engineering Journal, 395, 125082*.

Lish, R. A. D., Johari, S. A., Sarkheil, M., and Yu, I. J. (2019). On how environmental and experimental conditions affect the results of aquatic nanotoxicology on brine shrimp (Artemia salina): A case of silver nanoparticles toxicity. *Environmental Pollution, 255, 113358*.

Liu, J., Mwamulima, T., Wang, Y., Fang, Y., Song, S., and Peng, C. (2017). Removal of Pb (II) and Cr (VI) from aqueous solutions using the fly ash-based adsorbent material-supported zero-valent iron. *Journal of Molecular Liquids, 243, 205-211*.

Liu, Q., Tang, X., Wang, Y., Yang, Y., Zhang, W., Zhao, Y. and Zhang, X. (2019). ROS changes are responsible for tributyl phosphate (TBP)-induced toxicity in the alga Phaeodactylum tricornutum. *Aquatic Toxicology, 208, 168-178*.

Loganathan, P., Naidu, G., and Vigneswaran, S. (2017). Mining valuable minerals from seawater: a critical review. *Environmental Science: Water Research & Technology*, 3(1), 37-53.

Maguvu, T. E., Bezuidenhout, C. C., Kritzinger, R., Tsholo, K., Plaatjie, M., Molale-Tom, L. G., and Coertze, R. D. (2020). Combining physicochemical properties and microbiome data to evaluate the water quality of South African drinking water production plants. *Primary library of science one, 15(8), e0237335*.

Mahmud, Z. H., Islam, M. S., Imran, K. M., Hakim, S. A. I., Worth, M., Ahmed, A., and Ahmed, N. (2019). Occurrence of Escherichia coli and faecal coliforms in drinking water at source and household point-of-use in Rohingya camps, Bangladesh. *Gut pathogens, 11(1), 1-11*.

Maiti, S.K. (2004). Handbook of methods in environmental studies. Vol. 1, Water and Waste Water Analysis, ABD Publishers, Jaipur.

Malaeb, L., and Ayoub, G. M. (2011). Reverse osmosis technology for water treatment: State of the art review. *Desalination, 267(1), 1-8*.

Manes, C. D. O., Barbe, C., West, N. J., Rapenne, S., and Lebaron, P. (2011). Impact of seawater-quality and water treatment procedures on the active bacterial assemblages at two desalination sites. *Environmental science & technology, 45(14), 5943-5951*.

Marigómez, I., Soto, M., Orbea, A., Cancio, I., and Cajaraville, M. P. (2004). Biomonitoring of environmental pollution along the Basque coast, using molecular, cellular and tissue-level biomarkers: an integrative approach. *Oceanography and Marine Environment of the Basque Country, Elsevier Oceanography Series, Elsevier, Amsterdam, 70, 335-364*.

Masante, D., McCormick, N., Vogt, J., Carmona-Moreno, C., Cordano, E. and Ameztoy, I. (2018). Drought and Water Crisis in Southern Africa, European Commission, Ispra,

Mauck, B. A. (2017). The capacity of the Cape Flats aquifer and its role in water sensitive urban design in Cape Town.

McKee, A. M. and Cruz, M. A. (2021). Microbial and Viral Indicators of Pathogens and Human Health Risks from Recreational Exposure to Waters Impaired by Fecal Contamination. *Journal of Sustainable Water in the Built Environment, 7(2), 03121001*.

McKnight, U. S., Rasmussen, J. J., Kronvang, B., Bjerg, P. L., and Binning, P. J. (2012). Integrated assessment of the impact of chemical stressors on surface water ecosystems. *Science of the total Environment, 427, 319-331*.

Mdzeke, N. P. (2004). Contamination levels in and cellular responses of intertidal invertebrates as biomarkers of toxic stress caused by heavy metal contamination in False Bay (Doctoral dissertation, Stellenbosch: Stellenbosch University)

Mekki, A., and Sayadi, S. (2017). Study of heavy metal accumulation and residual toxicity in soil saturated with phosphate processing wastewater. *Water, Air, & Soil Pollution, 228(6), 215*.

Mekonnen, M. M., and Hoekstra, A. Y. (2016). Four billion people facing severe water scarcity. *Science advances, 2(2), e1500323*.

Meque, A., and Abiodun, B. J. (2015). Simulating the link between ENSO and summer drought in Southern Africa using regional climate models. *Climate Dynamics, 44(7), 1881-1900*.

Meride, Y., and Ayenew, B. (2016). Drinking water quality assessment and its effects on residents health in Wondo genet campus, Ethiopia. *Environmental Systems Research, 5(1), 1-7*.

Metcalf, G., Pillay, L., Murutu, L. C., Chiburi, S., Gumede, N., and Gaydon, P. (2014). Wastewater reclamation for potable reuse Volume 2: Integration of MBR technology with advanced treatment processes. Research Report no. TT611/14, Water Research Commission, Pretoria, South Africa, 2014 Search PubMed.

Miri, R., and Chouikhi, A. (2005). Ecotoxicological marine impacts from seawater desalination plants. *Desalination, 182(1-3), 403-410*.

Missimer, T. M., and Maliva, R. G. (2018). Environmental issues in seawater reverse osmosis desalination: Intakes and outfalls. *Desalination, 434, 198-215*.

Mohood, C. V., and Dhote, J. (2013). Review of heavy metals in drinking water and their effect on human health. *International Journal of Innovative Research in Science, Engineering and Technology, 2(7), 2992-2996*.

Mokhosi, A., and Dzwairo, B. (2015). Microbiological water quality along Vaal Gamagara's potable water distribution system. *Environmental Economics. 6 (1),152-158*

Mokhtari, S.A., Fazlzadeh, Davil, M.and Dorraji, B. (2011). Survey of bacteriological quality of the drinking water in rural areas of Ardabil city. *Journal of Health, 2(1), 66-73*.

Muller, R., Tang J.Y.M., Their, R. and Mueller, J.F. (2007). Combining passive sampling and toxicity testing for evaluation of mixtures of polar organic chemicals in sewage treatment plant effluents. *Journal of Environmental Monitoring, 9 (1), 104-109*.

Msimang, Z. (2017). A study of the negative impacts of informal settlement on the environment: a case study of Jika Joe, Pietermaritzburg (Doctoral dissertation).

Nakada, N., Nyunoya, H., Nakamura, M., Hara, A., Iguchi, T., and Takada, H. (2004). Identification of estrogenic compounds in wastewater effluent. Environmental *Toxicology and Chemistry: An International Journal, 23(12), 2807-2815*.

Naubi, I., Zardari, N. H., Shirazi, S. M., Ibrahim, N. F. B., and Baloo, L. (2016). Effectiveness of Water Quality Index for Monitoring Malaysian River Water Quality. *Polish Journal of Environmental Studies, 25(1), 231-239*.

Nayar, R. (2020). Assessment of water quality index and monitoring of pollutants by physico-chemical analysis in water bodies: a review. *International Journal of Engineering Research and Technology, 9(1)*.

Niyoyitungiye, L., Giri, A., and Ndayisenga, M. (2020). Assessment of Coliforms Bacteria Contamination in Lake Tanganyika as Bioindicators of Recreational and Drinking Water Quality. *South Asian Journal of Research in Microbiology, 9-16*.

Nowicki, S., deLaurent, Z. R., de Villiers, E. P., Githinji, G. and Charles, K. J. (2021). The utility of Escherichia coli as a contamination indicator for rural drinking water: Evidence from whole genome sequencing. *Plos one, 16(1), e0245910*.

Nriagu, J., Darroudi, F., and Shomar, B. (2016). Health effects of desalinated water: Role of electrolyte disturbance in cancer development. *Environmental research, 150, 191-204*.

Nsor, C.A., Acquah, E. and Braimah, C.A. (2016). Seasonal Dynamics of Physicochemical Characteristics in Wetlands of Northern Region (Ghana): Implications on the Functional Status. *International Journal of Aquatic Science, 7(1), 39-49.*

Nthunya, L. N., Maifadi, S., Mamba, B. B., Verliefde, A. R., and Mhlanga, S. D. (2018). Spectroscopic determination of water salinity in brackish surface water in Nandoni Dam, at Vhembe District, Limpopo Province, South Africa. *Water, 10(8), 990.*

Nyamukamba, P., Moloto, M. J., Tavengwa, N., and Ejidike, I. P. (2019). Evaluating physicochemical parameters, heavy metals, and antibiotics in the influents and final effluents of South African wastewater treatment plants. *Polish Journal of Environmental Studies, 28(3), 1305-1312*.

Obasi, P. N., and Akudinobi, B. B. (2020). Potential health risk and levels of heavy metals in water resources of lead–zinc mining communities of Abakaliki, southeast Nigeria. *Applied Water Science, 10(7), 1-23*.

Offenbaume, K. L., Bertone, E., and Stewart, R. A. (2020). Monitoring Approaches for Faecal Indicator Bacteria in Water: Visioning a Remote Real-Time Sensor for *E. coli* and Enterococci. *Water, 12(9), 2591*.

Okay, O. S., Tolun, L., Tüfekçi, V., Karacik, B., Kungolos, A., Samaras, P., and Tsiridis, V. (2005). Comparison of several toxicity tests applied to complex wastewaters and mussel biomarkers in receiving waters. *Journal of Environmental Science and Health, Part A, 40(8), 1525-1541*.

Olsson, G. (2012). Water and Energy: Threats and Opportunities. IWA Publishing, London.

Oshiro, R. K. (2002). Method 1603: *Escherichia coli* (*E. coli*) in Water by Membrane Filtration Using Modified Membrane Thermotolerant Escherichia coli Agar (Modified mTEC). United States Environmental Protection Agency: Washington, DC, USA.

Panagopoulos, A. and Haralambous, K. J. (2020). Environmental impacts of desalination and brine treatment-Challenges and mitigation measures. *Marine Pollution Bulletin, 161, 111773*.

Pangarkar, B. L., Sane, M. G., and Guddad, M. (2011). Reverse osmosis and membrane distillation for desalination of groundwater: a review. *International Scholarly Research Notices, 1-9.*

Parks, R., Mclaren, M., Toumi, R., Rivett, U., 2019. Experiences and lessons in managing water from Cape Town. London, England.

Passerat, J., Ouattara, N. K., Mouchel, J. M., Rocher, V., and Servais, P. (2011). Impact of an intense combined sewer overflow event on the microbiological water quality of the Seine River. *Water research, 45(2), 893-903*.

Patterton, H.G. (2013). Scoping study and research strategy development on currently known and emerging contaminants influencing drinking water quality. WRC report no. 2093/1/13. Pretoria: WRC.

Peters, C., Becker, S., Noack, U., Pfitzner, S., Bülow, W., Barz, K., and Berghahn, R. (2002). A marine bioassay test set to assess marine water and sediment quality–its need, the approach and first results. *Ecotoxicology, 11(5), 379-383*.

Petersen, F., and Hubbart, J. A. (2020). Physical Factors Impacting the Survival and Occurrence of Escherichia coli in Secondary Habitats. *Water, 12(6), 1796*.

Petrik, L., Green, L., Abegunde, A. P., Zackon, M., Sanusi, C. Y., and Barnes, J. (2017). Desalination and seawater quality at Green Point, Cape Town: A study on the effects of marine sewage outfalls. *South African Journal of Science, 113(11), 1-10*.

Pfaff, M. C., Logston, R. C., Raemaekers, S. J., Hermes, J. C., Blamey, L. K., Cawthra, H. C., and Kirkman, S. P. (2019). A synthesis of three decades of socio-ecological change in False Bay, South Africa: setting the scene for multidisciplinary research and management. *Elementa: Science of the Anthropocene, 7, 32*.

Pistocchi, A., Bleninger, T., Breyer, C., Caldera, U., Dorati, C., Ganora, D., and Zaragoza, G. (2020). Can seawater desalination be a win-win fix to our water cycle? *Water research, 182, 115906*.

Pontius, F. (2018). Treatability of a highly-impaired, saline surface water for potential Urban water use. *Water, 10(3), 324*.

Public Health Laboratory Services (PHLS). (1999). Enumeration of Coliforms and Escherichia by IDEXX Quanti - Tray™ Standard Method. London.

Pule, M., Yahya, A., & Chuma, J. (2017). Wireless sensor networks: A survey on monitoring water quality. *Journal of applied research and technology, 15(6), 562-570*.

Prinsloo, R. M. P. (2014). Potential pathogenicity of heterotrophic plate count bacteria isolated from untreated drinking water (Doctoral dissertation).

Qar, H., and Abdel-Monem, U. M. (2014). Effect of drinking natural sea saline water on growth performance, some blood parameters and carcass traits on New Zealand white rabbits. *Journal of American Science, 10(11), 55-59*.

Quattara, N.K., Passerat, J. and Servais, P. (2009). Microbiological water quality in rivers of the

Scheldt drainage network (Belgium): impact of urban wastewater release. 8th World Wide Workshop for Young Environmental Scientists WWW-YES 2009: *Urban waters: resource or risks? 1,1-10*.

Quick, A. J. R., and Pistorius, P. A. (1994, June). Environmental issues and management strategies in Metropolitan Cape Town. In Urban Forum, 5, (2). 45-68. Springer Netherlands.

Ramanathan, N., Padmavathy, P., Francis, T., Athithian, S., and Selvaranjitham, N. (2005). Manual on polyculture of tiger shrimp and carps in freshwater. Tamil Nadu Veterinary and Animal Sciences University, Fisheries College and Research Institute, *Thothukudi, 1, 161*.

Ray, S., and Ray, M. K. (2009). Bioremediation of heavy metal toxicity-with special reference to chromium. *Al Ameen Journal of Medical Science 2(2), 57-63*.

Roberts, D. A., Johnston, E. L., and Knott, N. A. (2010). Impacts of desalination plant discharges on the marine environment: A critical review of published studies. *Water research, 44(18), 5117-5128*.

Rock, C. M., Brassill, N., Dery, J. L., Carr, D., McLain, J. E., Bright, K. R. and Gerba, C. P. (2019). Review of water quality criteria for water reuse and risk-based implications for irrigated produce under the FDA Food Safety Modernization Act, produce safety rule. *Environmental research, 172, 616-629.*

Rodrigues, A. P., Oliveira, P. C., Guilhermino, L., and Guimaraes, L. (2012). Effects of salinity stress on neurotransmission, energy metabolism, and anti-oxidant biomarkers of Carcinus maenas from two estuaries of the NW Iberian Peninsula. *Marine biology, 159(9), 2061-2074*.

Rodrigues, C., and Cunha, M. Â. (2017). Assessment of the microbiological quality of recreational waters: indicators and methods. *Euro-Mediterranean Journal for Environmental Integration, 2(1), 1-18*.

Rothenheber, D., and Jones, S. (2018). Enterococcal concentrations in a coastal ecosystem are a function of fecal source input, environmental conditions, and environmental sources. *Applied and environmental microbiology, 84(17)*.

Rotini, A., Manfra, L., Spanu, F., Pisapia, M., Cicero, A. M., and Migliore, L. (2017). Ecotoxicological method with marine bacteria Vibrio anguillarum to evaluate the acute toxicity of environmental contaminants. *Journal of visualized experiments,123*.

Rouault, M., and Richard, Y. (2003). Intensity and spatial extension of drought in South Africa at different time scales.

Ruiz-Beviá, F., and Fernández-Torres, M. J. (2019). Effective catalytic removal of nitrates from drinking water: An unresolved problem? *Journal of Cleaner Production, 217, 398-408.Water Sa, 29(4), 489-500*.

Rygala, A., Berlowska, J., and Kregiel, D. (2020). Heterotrophic Plate Count for Bottled Water Safety Management. *Processes, 8(6), 739*.

Sadhwani, J. J., Veza, J. M., and Santana, C. (2005). Case studies on environmental impact of seawater desalination. *Desalination, 185(1-3), 1-8*.

Saeed, M. O., Al-Nomazi, M. A., and Al-Amoudi, A. S. (2019). Evaluating suitability of source water for a proposed SWRO plant location. *Heliyon, 5(1), e01119*.

Saingam, P., Li, B. and Yan, T. (2020). Fecal indicator bacteria, direct pathogen detection, and microbial community analysis provide different microbiological water quality assessment of a tropical urban marine estuary. *Water Research, 185, 116280*.

Sankhla, M. S., and Kumar, R. (2018). Fluoride contamination of water in India and its impact on public health. *ARC Journal Forensic Science, 3(2), 10-15*.

SANS (South African National Standard) 7899-2. (2004). Water quality- Detection and Enumeration of intestinal enterococci. Part 2: Membrane filtration method.

SANS (South African National Standard) 5221. (2007). Microbiological Analysis of Water General Test Method.

Sarker, S., Mahmud, S., Sultana, R., Biswas, R., Sarkar, P. P., Munayem, M. A., and Evamoni, F. Z. (2019). Quality Assessment of Surface and Drinking Water of Nakla Paurosova, Sherpur, Bangladesh. *Advances in Microbiology, 9(08), 703*.

Schewe, J., Heinke, J., Gerten, D., Haddeland, I., Arnell, N. W., Clark, D. B., and Kabat, P. (2014). Multimodel assessment of water scarcity under climate change. *Proceedings of the National Academy of Sciences, 111(9), 3245-3250*.

Semiat, R., and Hasson, D. (2012). Water desalination. *Reviews in Chemical Engineering, 28(1), 43-60*.

Seo, M., Lee, H. and Kim, Y. (2019). Relationship between coliform bacteria and water quality factors at weir stations in the Nakdong River, South Korea. *Water, 11(6), 1171*.

Sepehr, M., Fatemi, S. M. R., Danehkar, A., and Mashinchian Moradi, A. (2017). Application of Delphi method in site selection of desalination plants. *Global Journal of Environmental Science and Management, 3(1), 89-102.*

Shabankareh, F.E, Hayati, R. and Dobaradaran, S. (2014). Evaluation of physical, chemical and microbial quality of distribution network drinking water in Bushehr, Iran. *Iran South Medicine Journal, 17(6), 1223-35.*

Shahryari A, Mehdinejad M, Ahmadi N. Impact of human activities on the rate of microbial contamination of the Gorgan Ziarat River, Iran. *Ecology, Environment and Conservation, 23(3), 1291-1297.*

Shakoor, S., Ahmed, I., Mukhtiar, S., Ahmed, I., Hirani, F., Sultana, S., and Hasan, R. (2018). High heterotrophic counts in potable water and antimicrobial resistance among indicator organisms in two peri-urban communities of Karachi, Pakistan. *BMC research notes, 11(1), 1-6.*

Shatat, M., and Riffat, S. B. (2014). Water desalination technologies utilizing conventional and renewable energy sources. *International Journal of Low-Carbon Technologies, 9(1), 1-19.*

Sharmila, D and Darun, S.K. (2013). A New Idea on Removing the Salt in Sea Water to Make Fresh Water Production Easier Using Nano Graphene Pores. *International Journal of Materials Science and Engineering, 1, 1.*

Shifat-E-Raihan, K., Hasan, A., Hossain, M., Hassan, R., Juliana, F.M., Islam, M.J., Alam, J., , Hossain, M.N., Nesha, M. , Asma, R. and Asaduzzaman, M. (2017). Heterotrophic Plate Count (HPC) of the Commercially Available Bottled Water in Dhaka, Bangladesh. *Journal of Nursing and Health Science,6(5), 23–27.*

Shomar, B., and Hawari, J. (2017). Desalinated drinking water in the GCC countries–The need to address consumer perceptions. *Environmental research, 158, 203-211.*

Shrestha, A. K., Basnet, N., Bohora, C. K., and Khadka, P. (2017). Variation of electrical conductivity of the different sources of water with temperature and concentration of electrolyte solution NaCl. *International Journal of Recent Research and Review, 10(3), 24-26.*

Soto, K., Collantes, G., Zahr, M. and Kuznar, J. (2005). Simultaneous enumeration of Phaeodactylum tricornutum (MLB292) and bacteria growing in mixed communities. *Investigaciones marinas, 33, 143-149.*

Stevenson, J. (2014). Ecological assessments with algae: a review and synthesis. *Journal of Phycology, 50(3), 437-461.*

Subramani, A., and Jacangelo, J. G. (2015). Emerging desalination technologies for water treatment: a critical review. *Water research, 75, 164-187*.

Sun, Y., Chen, Z., Wu, G., Wu, Q., Zhang, F., Niu, Z., and Hu, H. Y. (2016). Characteristics of water quality of municipal wastewater treatment plants in China: implications for resources utilization and management. *Journal of Cleaner Production, 131, 1-9*.

Suski, C. D., Killen, S. S., Kieffer, J. D., and Tufts, B. L. (2006). The influence of environmental temperature and oxygen concentration on the recovery of largemouth bass from exercise: implications for live–release angling tournaments. *Journal of Fish Biology, 68(1), 120-136*.

Takabatake, H., Taniguchi, M., and Kurihara, M. (2021). Advanced Technologies for Stabilization and High Performance of Seawater RO Membrane Desalination Plants. *Membranes, 11(2), 138*.

Talavera, J. P., & Ruiz, J. Q. (2001). Identification of the mixing processes in brine discharges carried out in Barranco del Toro Beach, south of Gran Canaria (Canary Islands). *Desalination, 139(1-3), 277-286*.

Taljaard, S., Van Ballengooyen, R. C., and Morant, P. D. (2000). False Bay water quality review. CSIR Report Env-SC-2000-086, 1, 1-10.

Tan, C. W., Thishalini, A., Goh, E. G., and Edlic, S. (2017). Studies on turbidity in relation to suspended solid, velocity, temperature, pH, conductivity, colour and time. *ARPN Journal of Engineering and Applied Sciences, 12(19), 5626-5635*.

Taneja, P., Labhasetwar, P., and Nagarnaik, P. (2019). Nitrate in drinking water and vegetables: intake and risk assessment in rural and urban areas of Nagpur and Bhandara districts of India. *Environmental Science and Pollution Research, 26(3), 2026-2037*.

Titilawo, Y., Adeniji, A., Adeniyi, M., and Okoh, A. (2018). Determination of levels of some metal contaminants in the freshwater environments of Osun State, Southwest Nigeria: A risk assessment approach to predict health threat. *Chemosphere, 211, 834-843*.

Triantaphyllidis, G., Abatzopoulos, T., and Sorgeloos, P. (1998). Review of the biogeography of the genus Artemia (Crustacea, Anostraca). *Journal of biogeography, 25(2), 213-226*.

Tularam, G. A., and Ilahee, M. (2007). Environmental concerns of desalinating seawater using reverse osmosis. *Journal of Environmental monitoring, 9(8), 805-813*.

Tyson, P. D., and Preston-Whyte, R. A. (2000). Weather and climate of southern Africa. Oxford University Press.

Uddin, S. (2014). Environmental impacts of desalination activities in the Arabian Gulf. International *Journal of Environmental Science and Development, 5(2), 114-117*.

UNESCO (United Nations Educational, Scientific and Cultural Organization), 2015.

Water for a Sustainable World. The United Nations World Water Development

Report 2015 available from: http://unesdoc.unesco.org/images/0023/002318/231823E.pdf.

USEPA (United States Environmental Protection Agency). (1999). 25 years of the safe drinking

water act: History and trends. USEPA Report 816-R-99-007.

Van Ballegooyen, R. C., Steffani, C. N., and PD, M. (2004). Kudu Power Plant Environmental Impact Assessment: Assessment of a proposed cooling water discharge into the marine environment. CSIR Report ENV-SC, 86.

Van der Merwe, I.J., Vlok, A.C. and van der Merwe, J.H. (1991). Land use and population characteristics in the False Bay coastal frame. *Transactions of the Royal Society of South Africa, 47(4), 693-707.*

Van der Merwe, R., Hammes, F., Lattemann, S. and Amy, G. (2014). Flow cytometric assessment of microbial abundance in the near-field area of seawater reverse osmosis concentrate discharge. *Desalination, 343, 208–216*.

Vargas-Chacoff, L., Arjona, F. J., Polakof, S., del Río, M. P. M., Soengas, J. L., and Mancera, J. M. (2009). Interactive effects of environmental salinity and temperature on metabolic responses of gilthead sea bream Sparus aurata. *Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology, 154(3), 417-424.*

Veeramani, T., Santhanam, P., Manickam, N. and Rajthilak, C. (2019). Introduction to Artemia Culture. In Basic and Applied Zooplankton Biology (pp. 209-224). Springer, Singapore.Veni, T., and Pushpanathan, T. (2014). Comparison of the Artemia salina and Artemia fransiscana bioassays for toxicity of Indian medicinal plants. *Journal of Coastal Life Medicine, 2(6), 453-457*.

Verga, R. N., Tolosano, J. A., Cazzaniga, N. J., and Gil, D. G. (2020). Assessment of seawater quality and bacteriological pollution of rocky shores in the central coast of San Jorge Gulf (Patagonia, Argentina). *Marine pollution bulletin, 150, 110749*.

Virgili, F. (2016). Desalination market update. Third Quarter Assessment. Oxford, UK: Global Water Intelligence. Available from www.desaata.com/forecasts.

Voutchkov, N. (2018). Energy use for membrane seawater desalination–current status and trends. *Desalination, 431, 2-14*.

Wadhia, K., and Thompson, K. C. (2007). Low-cost ecotoxicity testing of environmental samples using microbiotests for potential implementation of the Water Framework Directive. *Trends in Analytical Chemistry, 26(4), 300-307*.

Waideman, M. A., Teixeira, V. P., Uemura, E. H., Stamford, T. M., Leal, D. A. G., Stangarlin-Fiori, L., and Beux, M. R. (2020). Enterococci used as complementary indicator of fecal contamination to assess water quality from public schools in the city of Curitiba, Paraná, Brazil. *Brazilian Journal of Food Technology, 23*.

Wang, Z. X., Chen, J. Q., Chai, L. Y., Yang, Z. H., Huang, S. H., and Zheng, Y. (2011). Environmental impact and site-specific human health risks of chromium in the vicinity of a ferro-alloy manufactory, China. *Journal of hazardous materials, 190(1), 980-985*.

Wang, X. N., Liu, Y., Pan, X. H., Han, J. X., and Hao, J. (2017). Parameters for seawater reverse osmosis product water: a review. *Exposure and Health, 9(3), 157-168*.

Wei, D., Tan, Z., and Du, Y. (2011). A biological safety evaluation on reclaimed water reused as scenic water using a bioassay battery. *Journal of Environmental Sciences, 23(10), 1611-1618*.

Wen, X., Chen, F., Lin, Y., Zhu, H., Yuan, F., Kuang, D. and Yuan, Z. (2020). Microbial Indicators and Their Use for Monitoring Drinking Water Quality—A Review. *Sustainability, 12(6), 2249*.

Wieczerzak, M., Namieśnik, J., and Kudłak, B. (2016). Bioassays as one of the Green Chemistry tools for assessing environmental quality: A review. *Environment international, 94, 341-361*.

Wilder, M. O., Aguilar-Barajas, I., Pineda-Pablos, N., Varady, R. G., Megdal, S. B., McEvoy, J., and Scott, C. A. (2016). Desalination and water security in the US–Mexico border region: assessing the social, environmental and political impacts. *Water International, 41(5), 756-775*.

Wilson, B. A., Smith, V. H., deNoyelles, F., and Larive, C. K. (2003). Effects of three pharmaceutical and personal care products on natural freshwater algal assemblages. *Environmental science and technology, 37(9), 1713-1719*.

Wimalawansa, S. J. (2013). Purification of contaminated water with reverse osmosis: effective solution of providing clean water for human needs in developing countries. *International journal of emerging technology and advanced engineering, 3(12), 75-89*.

Winter, T., Pannell, D. J., & McCann, L. M. (2002). The Economics of Desalination and It's Potential Application to Australia. University of Western Australia.

WHO (World Health Organization) (2005). Nutrients in Drinking Water. World Health Organization, Geneva.

WHO (World Health Organization) (2007). Desalination for Safe Water Supply, Guidance for the Health and Environmental Aspects Applicable to Desalination, Public Health and the Environment, World Health Organization, Geneva.

WHO (World Health Organization). (2008). Guidelines for Drinking Water Quality.3rd edn. Incooperating the first and second addenda. Volume 1, Recommendations. World Health Organization, Geneva.

WHO (World Health Organization). (2016). Barium in Drinking-Water; World Health Organization, Geneva.

WHO (World Health Organization). (2018). A global overview of national regulations and standards for drinking-water quality. World Health Organization, Geneva.

Wolski, P., Otto, F. E., Lehner, F., Tebaldi, C., Van Oldenborgh, G. J., Hogesteeger, S., and New, M. (2018). Anthropogenic influence on the drivers of the Western Cape drought 2015–2017. *Environmental Research Letters, 13(12), 124010*.

WRC (Water Research Commission). (2015). Investigation into the Cost and Operation of Southern African Desalination and Water Reuse Plants. WRC Report No. TT 636/15.

Xu, J., Wei, D., Wang, F., Bai, C., and Du, Y. (2020). Bioassay: A useful tool for evaluating reclaimed water safety. *Journal* of Environmental Sciences, 88, 165-176.

Yang, L. H., Ying, G.G., Su, H.C., Stauber, J. L., Adams, M. S., and Binet, M. T. (2008). Growth-inhibiting effects of 12 antibacterial agents and their mixtures on the freshwater microalga pseudokirchneriella subcapitata. *Environmental Toxicology and Chemistry: An International Journal, 27(5), 1201-1208*.

Yang, Q., Liu, Y., and Li, Y. (2010). Humic acid fouling mitigation by antiscalant in reverse osmosis system. *Environmental science & technology, 44(13), 5153-5158*.

Yi, J., Lee, J., Jung, H., Park, P. K. and Noh, S. H. (2019). Reduction of bacterial regrowth in treated water by minimizing water stagnation in the filtrate line of a gravity-driven membrane system. *Environmental Engineering Research, 24(1), 17-23*.

Yi, Y. J., and Zhang, S. H. (2012). The relationships between fish heavy metal concentrations and fish size in the upper and middle reach of Yangtze River. Procedia Environmental Sciences, 13, 1699-1707.

Youssef, P. G., Al-Dadah, R. K., and Mahmoud, S. M. (2014). Comparative analysis of desalination technologies. *Energy Procedia, 61, 2604-2607.*

Yousefi, Z., Motallebi, R., and Takdastan, A. (2014). Water desalination by membrane technology (RO) in southern Iran (Jask city). *Environmental Health Engineering and Management Journal, 1(1), 13-18.*

Yu, J., and Lu, Y. (2018). Artemia spp. model-a well-established method for rapidly assessing the toxicity on an environmental perspective. *Medical Research Archives, 6(2).*

Yunus, S., Hamzah, Z., Wood, A. K., and Ahmad. (2015, April). Assessment of heavy metals in seawater and fish tissues at Pulau Indah, Selangor, Malaysia. *In AIP Conference Proceedings 1659, 1-6.*

Žaltauskait, J., Sujetovien, G., Cypait. A., and Aužbikaviciut, A. (2014). Lemna minor as a tool for wastewater toxicity assessment and pollutants removal agent. *Environmental engineering. Proceedings of the international conference on environmental engineering, 9, 1.*

Zargar, A., Sadiq, R., Naser, B., & Khan, F. I. (2011). A review of drought indices. *Environmental Reviews, 19, 333-349.*

Zhang, Q., Li, Q., Singh, V. P., Shi, P., Huang, Q., and Sun, P. (2018). Nonparametric integrated agrometeorological drought monitoring: Model development and application. *Journal of Geophysical Research: Atmospheres, 123(1), 73-88.*

Zhang, H., and Xie, Y. (2019). Alleviating freshwater shortages with combined desert-based large-scale renewable energy and coastal desalination plants supported by Global Energy Interconnection. *Global Energy Interconnection, 2(3), 205-213.*

Zhao, J., Zhang, Q., Zhu, X., Shen, Z., and Yu, H. (2020). Drought risk assessment in China: evaluation framework and influencing factors. Geography and Sustainability, 1(3), 220-228.

Ziolkowska, J. R. (2015). Is desalination affordable? regional cost and price analysis. *Water Resources Management, 29(5), 1385-1397.*

APPENDICES

Appendix A

UNISA university of south africa

CAES HEALTH RESEARCH ETHICS COMMITTEE

Date: 08/10/2018

NHREC Registration # : REC-170616-051
REC Reference # : 2018/CAES/127
Name : Ms B Badiso
Student #: 60051094

Dear Ms Badiso

Decision: Ethics Approval from 04/10/2018 to 30/09/2019

Researcher(s): Ms B Badiso
60051094@mylife.unisa.ac.za

Supervisor (s): Prof M Tekere
tekerm@unisa.ac.za; 011-471-2270

Working title of research:

The use of toxicity-based method to evaluate the potential ecological hazard of desalinated seawater for drinking water purposes in a drought stricken city of Cape Town, South Africa

Qualification: MSc Environmental Science

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

Due date for progress report: 30 September 2019

Please note the points below for further action:

1. The researcher is requested to provide more information on the hazardous waste disposal process of the laboratory facility that will be used. Such facilities normally have a Standard Operating Procedure (SOP) in place, and the researcher can provide this SOP to the Committee.
2. The research proposal makes no mention of the collection of the sampling of control sites independent of the identified sampling points. How will the researcher determine what the baseline level of contamination is away from these sampling points?

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The ***minimal risk application*** *was* ***reviewed*** *by the CAES Health Research Ethics Committee on 04 October 2018 in compliance with the Unisa Policy on Research Ethics and the Standard Operating Procedure on Research Ethics Risk Assessment.*

The proposed research may now commence with the provisions that:

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

Note:

The reference number ***2018/CAES/127*** *should be clearly indicated on all forms of communication with the intended research participants, as well as with the Committee.*

Yours sincerely,

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Appendix B

CITY OF CAPE TOWN
ISIXEKO SASEKAPA
STAD KAAPSTAD

UTILITY SERVICE:
WATER AND SANITATION

Mjikisile Vulindlu
Head: Biological Sciences Laboratory: Scientific Services

T: 021 444 9158
E: Mjikisile.Vulindlu@capetown.gov.za

Date: **20 September 2018**
Purpose: **Permission to conduct research at Scientific Services**

This letter serves to confirm that Ms. Bulelwa Badiso, Staff number 10047442, has been granted permission to execute the necessary studies at the Scientific Services Branch, at the City of Cape Town.

Bulelwa Badiso is currently working in Scientific Services' Biological Sciences Laboratory as a permanent competent technician. The Scientific Services Branch undertakes compliance monitoring using toxicity testing kits and as part of expanding the City's scope. She has discussed her project with me and is hereby is granted permission to do her MSc degree titled "The use of toxicity-based method to evaluate the potential ecological hazard of desalinated seawater for drinking water purposes in a drought stricken City of Cape Town, South Africa".

The testing been done will add to the City Knowledge

Signed ____________ Date 2018.09.21

Mjikisile Vulindlu
Head: Biological Sciences: SCIENTIFIC SERVICES: WATER AND SANITATION

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Water quality and potential toxicity assessment of desalinated seawater for drinking purposes in the City of Cape Town, South Africa

By

Bulelwa Badiso

Submitted in accordance with the requirements

for the degree of

Master of Environmental Science

in the

COLLEGE OF AGRICULTURE AND ENVIRONMENTAL SCIENCES

DEPARTMENT OF ENVIRONMENTAL SCIENCES

at the

University of South Africa

FLORIDA CAMPUS

SUPERVISOR: PROF.

M. TEKERE

February 2021

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