Nutrient fluxes within the Berg River from 1976 to 2017, Western Cape, South Africa

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ABSTRACT

Deterioration of freshwater systems due to eutrophication is increasingly a global concern because it puts stress on the already limited freshwater systems. Eutrophication is caused by elevated levels of nutrients in river systems primarily from poor land management whereby excess nutrients are discharged into fresh water bodies. It is exacerbated by factors such as human population growth, economic growth and climate change. The main aim of this study was to analyze the changes in the nutrient status of the Berg River since the 1970s and tendencies toward hypertrophic conditions. The Berg River is an important source of bulk water supply for both domestic and industrial purposes in the Western Province, South Africa. The study examined water quality data from the Department of Water and Sanitation's Resource Quality Information Services for nine monitoring sites along the Berg River to determine the long-term trends. The data sets were analyzed using parametric statistics. The results show that nutrient levels in the Berg River are increasing at almost all the selected monitoring sites. The long term trend showed low mean values for both upstream and downstream with a peak nutrient levels observed along sections that were densely populated and extensively cultivated. Based on available total phosphorus data, all sites were classified as eutrophic except a monitoring site along the most populated and cultivated section of the Berg River which was permanently hypertrophic. Long-term mean values for total phosphorus exceed the recommended international guidelines for aquatic plant life. The long-term mean values of nitrate and nitrite also exceed the recommended guideline for aquatic plant life. Ortho-phospate mean values for the study showed that all sites experienced hypertrophic states at some stage during 1987 to 2017. Student t-test analyses confirm that nutrient concentration loads had increased in the past decade. The results confirm that anthrophogenic activities and climate change are two major drivers of change resulting in an increase in eutrophication. Therefore, serious attention should be paid to the role of anthrophogenic activites and climate change to mitigate the negative impact on freshwater systems.

DECLARATION

I, Lemogang Molebatsi, acknowledge that,

I am presenting this thesis in fulfillment of the requirements for my degree.

I know the meaning of plagiarism and declare that all the work in the thesis, save for that which is properly acknowledged is my own.

Signed by candidate

Signed: Date: October 2019

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CHAPTER 1: INTRODUCTION

1.1 Overview

Eutrophication is caused by elevated levels of nutrients in river systems, lakes and dams (Dodds & Smith 2016; Xia et al.,2016) It is a natural process, but anthropogenic activities are responsible for an increase in nutrient loading (Dodds & Smith, 2016). The deterioration of rivers due to eutrophication is a global concern (De Villiers, 2007; De Villiers & Thiart, 2007) because it leads to negative impacts such as a deterioration in water quality, human health, and a loss of biodiversity and aesthetics (DWAF 2002; Sullivan 2010). Typically phosphorus, as phosphate ions(PO_4^{3-}), and nitrogen, as nitrate(NO_3^-), nitrite(NO_2^-) and ammonium(NH_4^+) ions, contribute to nutrient enrichment of water bodies (De Villiers, 2007). In South Africa the most common anthropogenic point sources of inorganic nitrogen and phosphorus found in river systems originate from municipal sewage discharge, contaminated storm water and sanitary sewers, and combinations of waste water from livestock production, industrial effluent and runoff from waste disposal sites (De Villiers, 2007; Matthews & Bernard, 2015a; Griffin, 2017). By contrast, diffuse sources originate from agricultural activities including fertilizers, runoff of saturated nitrogen from forests and grasslands that have been burnt, urban runoff from non-sewered human settlements (typically informal), faulty sewer systems and dysfunctional municipal water treatment plants (De Villiers, 2007; De Villiers & Thiart, 2007).

Contamination of surface water bodies is serious because South Africa is a water scarce country with low rainfall and frequent droughts, and is heavily reliant on fresh surface water resources for its supply (Turpie et al., 2008). The main water supply is through the abatement of rivers. Most of the major rivers in South Africa are eutrophic due to a host of factors already mentioned including a lack of investment in infrastructure, lack of human capacity and skills to mitigate contaminated discharge, as well as limited capacity in the relevant government departments, poor governance in water resources and poor land use management practice (De Villiers & Thiart, 2007; van Ginkel, 2011; Harding, 2015). The trophic status of a water body refers to the level to which it is nutrient enriched. A water body may be classified as 'oligotrophic' (low in nutrients), 'mesotrophic' (intermediate levels of nutrients), 'eutrophic' (high in nutrients) or 'hypertrophic' (very high nutrients) (DWAF 2002; Matthews & Bernard 2015a). These categories are discussed in more detail in Chapter 2.

Studies that provide an overview of the trophic status of South Africa are limited. The most recent study by De Villiers & Thiart (2007) examined the nutrient status of 20 of South Africa's largest freshwater ecosystems and showed that nutrient levels were above the acceptable standard for plant life in all of these rivers except for one. In addition, these authors identified six rivers that were eutrophic throughout the year even when they were flushed during rainfall events. In another study De Villiers (2007) found that stretches along the Berg River, which is the site of interest in this research, that fell between densely populated urban areas, showed an increase in nutrient levels (nitrogen and phosphorus) and exceeded recommended international guidelines for water quality. De Villiers (2007) further states that all the DWS monitoring stations in the Berg River displayed episodic hypertrophic conditions and that the middle reaches of the Berg River were approaching a state of permanent hypertrophic conditions. De Villiers (2007) predicted that nutrient levels in the Berg River will continue to increase due to drought conditions and anthropogenic activities resulting in eutrophic and hypertrophic conditions. This study seeks to analyze the changes in the nutrient status of the Berg River by determining the trend in the nutrients level over time. The study will further determine if there are any significant differences in nutrient level over the last two decades from 1996 to 2006 prior to the commissioning and development of the Berg River Dam in the upper reaches of the Berg River, and from 2007 to 2017 when the dam became fully operationalised in 2009 following a construction period that commenced in 2004. The total capacity of the dam is 130 000 megalitres.

1.2 Hypothesis

Water quality in the Berg River has deteriorated since the 1970s from increasing nutrient loading.

1.3 Study aim and objectives.

The main aim of this study is to analyse the changes in the nutrient (nitrogen and phosphorus) status of the Berg River since the 1970s.

The aim will be achieved by examining three objectives:

1. Analysing changes in concentrations and trends in the nutrient loading of the Berg River.
2. Determining significant differences in nutrient loading by comparing conditions over last two decades.
3. Comparing the results with international guidelines and South African trophic status guidelines.

1.4 Research Questions

The objectives seek to answer the following questions:

1. How does the nutrient levels in the Berg River change over time?
2. Which sites show the most differences over time?
3. How do these sites differ over time and to what extent?
4. Which nutrients are increasing?
5. How have the nutrient levels changed over the last ten years

1.5 Study Design

The study is guided by a comprehensive literature review and statistical analysis of existing secondary time series data for the Berg River.The study examined historical water quality data from nine monitoring sites along the Berg River from 1976 to 2017 to determine downstream and long-term trends in nitrate and nitrite (NO_3^- and NO_2^-, or NO_x), ammonium (NH_4^+), total phosphorus (TP) and ortho-phosphate (PO_4^{3-}). Since 1967 the Department of Water and Sanitation (DWS) has established numerous monitoring stations at different sites along the Berg River but most of the monitoring stations were established in the late 1970s. This study only considered data from 1987 onwards because the length and completeness of time series data between stations was limited.

Data sets were extracted from the DWS Resource Quality Information Services (RQIS) website and statistically analyzed to determine changing eutrophic conditions between the different sites. Data were converted to monthly averages for ease of manipulation. The data sets were analysed using parametric statistical parameters such as mean, median and maximum values as an initial approach towards describing the variables and comparing the results of different years. The data sets were further analysed using the inferential t-test in statistical science to determine if there were any significant differences in the level of nutrients during the past two decades. The two decades were divided into Group 1 and Group 2. Group 1 represented data from 1996 to 2006 and Group 2 represented data from 2007 to 2017. The data were then compared with previous studies to determine whether the nutrient levels were increasing or decreasing. The nutrient levels in the Berg River were compared to both South African trophic status guideline values and international water quality guidelines. This study adopted a trophic status classification from De Villiers (2007) which is based on the South African Water Quality Guidelines for Aquatic Ecosystems of 1996 and National Eutrophication Monitoring Guidelines of 2002.

1.6 Study Region

The Berg River has its origins in the Franschhoek and Great Drakenstein Mountains situated approximately 80km east of Cape Town.The river is about 300km in length and drains an area of approximately 900km^2 flowing north westwards past the towns of Paarl, Wellington, Hermon and Gouda, where it is joined by the Klein Berg and Vier-en-Twintig Rivers (Mgese, 2010) as shown in Fiqure 1 below. It then flows westwards through Porterville, Piketberg, Hopefield and finally discharges into St Helena Bay on the west coast (River Health Programme, 2004; De Villiers, 2007). The geology of the upper catchment is characterized by sandstones and quartzites of the Cape Supergroup, while the middle catchment is characterized by the Cape granites and the lower catchment near the coast is characterized by recent sediments (De

Villiers 2007; Leaner et al., 2012). According to De Villiers (2007), the catchment is dominated by nutrient poor lithologies.

The climatic conditions of the catchment area are Mediterranean with warm dry summers and cool wet winters (Struyf et al., 2012). Rainfall occurs between the months of May and September receiving an average range of 500 to 1000mm, while the mean annual temperature is between 16°C in the east to 18°C along the west coast (Stuckenberg, 2012). Minimum temperatures occur in July with average daily minimum of 4.5°C and the maximum temperatures occur in January with average daily maximum of 29.4°C (Stuckenberg, 2012; Struyf et al., 2012).There are different land use activities along the Berg River catchment area. According to the River Health Programme (2004), dryland crops (mainly wheat) comprises 53% of the catchment area, natural vegetation 36%, irrigated crops (vineyards and fruits orchards) and forests plantation less than 1%. The economic activities along the catchment are mainly agricultural and these include wheat, grapes, deciduous fruits, wineries, canneries and other food processes factories (River Health Programme, 2004). Livestock farming is mainly practiced in the lower reaches of the river. The entire Berg River catchment area is heavily dependent on water for its economic activites and water is over utilised with an annual water demand of 690Mm^3/a shared as follows; household and business users (52%), irrigation (43%) and afforestation and alien vegetation (5%) (Pegram & Baleta, 2014).

The study area has experienced a substantial inward migration from the rural areas (DWS, 2017) resulting in the growth informal-settlements. Table 1 below shows the population of the key towns in the study area together with the percentage increase between the last two population census (Stats SA, 2011).

Table 1: Population of key towns in the study region

Town	Population (2011)	Population (2001)	% Increase
Franschhoek	17,556	1,463	1100
Paarl	112,045	82,713	35
Wellington	55,543	39,209	
Velddrif	11,017	7,338	

Note: the figures for Wellington and Velddrif are not comparable because of the changes in boundaries between the two census.

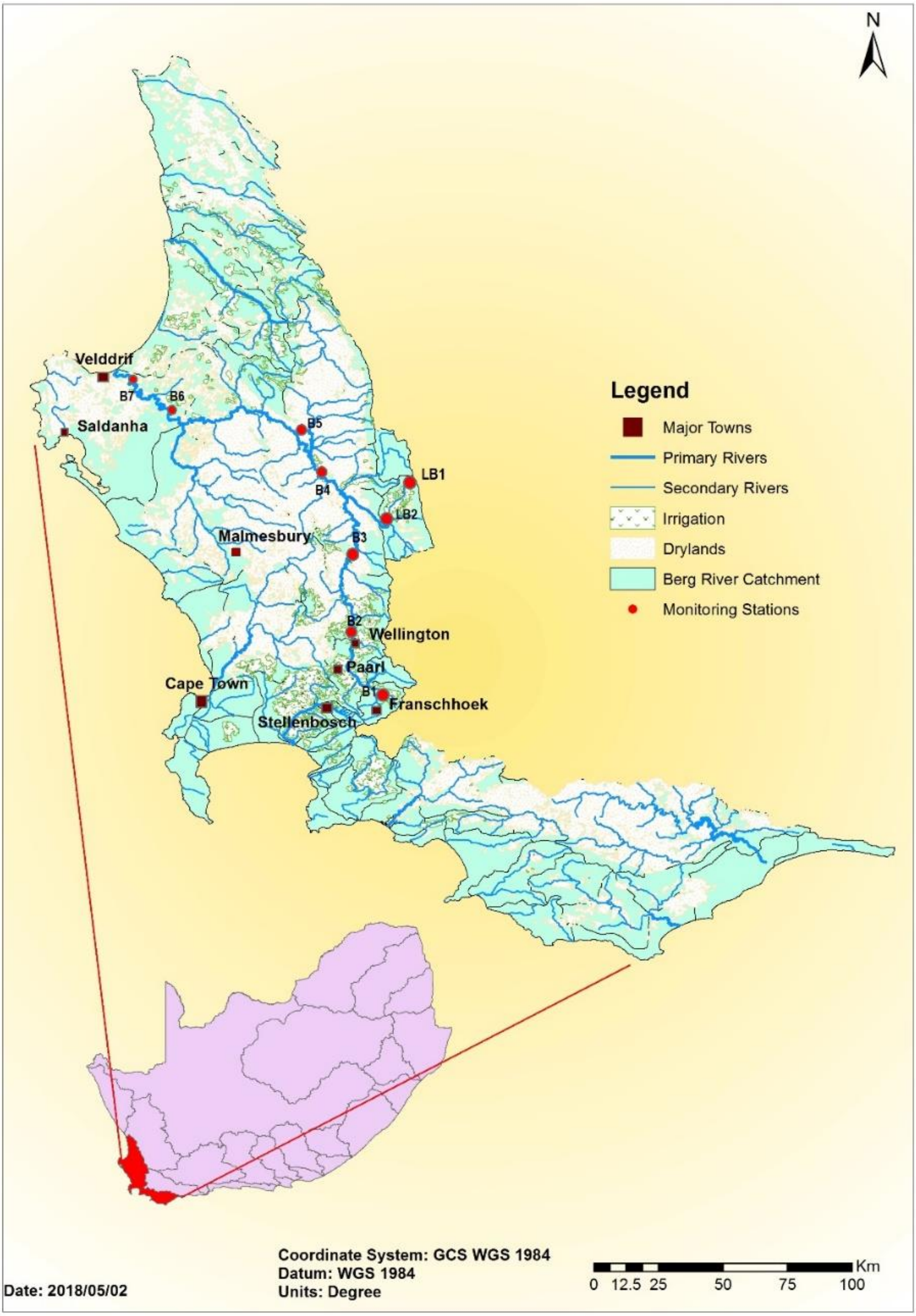

Figure 1 Map of the study area showing DWS monitoring stations

1.7 Study Limitations

The study relied on existing data extracted from the DWS RQWS database. The dataset has missing data resulting in gaps and inconsistencies. At some monitoring points data were collected weekly fortnightly or montlhy in others. The methods and protocols used to collect data were unknown. A further limitation is that chlorophyll-a was not available at all sites.

CHAPTER 2: LITERATURE REVIEW

2.1 Introduction

Cultural eutrophication is a concept that is used to emphasis the role of anthropogenic activities such as industrial pollution, domestic and agricultural activities, discharges from improper sanitation systems, inadequate infrastructure and lack of maintenance of existing infrastructure in formal and informal settlements (De Villiers, 2007; van Ginkel, 2011; Harding, 2015). Natural eutrophication is associated with deposits of nutrients from natural sources, such as rocks, soils and other features within the freshwater ecosystem (van Ginkel, 2011). Natural eutrophication occurs slowly according to natural processes while cultural eutrophication is controllable, can be avoided, but observed impacts are often rapid and enduring (van Ginkel, 2011). Eutrophication and the general deterioration of water quality in South Africa poses multiple risks to human and ecosystems health by reducing the availability of freshwater resources and destroying the ability of freshwater ecosystems to provide goods and services, including natural water purification (Matthews & Bernard, 2015a; UNESCO, 2018). It also poses an economic burden because of increased costs in water treatment, decreased value of adjacent properties and reduced income from water-based tourism activities (Matthews & Bernard, 2015a).

2.2 Land use and Eutrophication

Sources of nutrient pollution are diverse (Selma & Greenhalgh 2010). Nutrient enrichment in river systems is a result of both natural and anthropogenic sources. The most common inorganic nitrogen and phosphorus in freshwater systems in countries like South Africa are generated from sources that include dysfunctional wastewater treatment plants; overflowing storm and sanitary sewers; wastewater from livestock farming; runoff from waste disposal sites; and runoff from non-sewered human settlements (De Villiers, 2007). The anthropogenic diffuse sources are agricultural activities, mainly the use of manure and chemical fertilizers (Rizvi et al., 2015; Ayobahan et al.,2014; Dodds & Smith, 2016; De Villiers & Thiart, 2007).

The main urban and industrial sources of nutrient loads are wastewater treatment plants (WWTP) and industrial plants that discharge waste water directly into the river system. (Selma & Greenhalgh, 2010). Countries like South Africa are faced with rapid urbanization from rural areas (van Ginkel, 2011). Rapid rural-urban migration puts extreme pressure on existing infrastructure such as WWTPs which usually fail because they cannot cope with excessive load resulting in untreated sewage been discharged directly into river systems. At times WWTPs do not operate at acceptable standards resulting in the release of high concentrations of nutrients into river systems (Dabrowski & Klerk, 2013). Some of the WWTPs are not operating at acceptable standards because the municipalities fail to service the infrastructure either due to lack of capacity or mismanagement of resources.

Urban runoff is also a source of nutrients into the fresh water systems. Different studies around the world have observed that urban runoff contains nutrients such as nitrogen(N) and phosphorus(P) that are responsible for elevating the level of eutrophication in the receiving waters (Winter & Mgese, 2011; Capps et al., 2016; Fell 2017; Schwartz et al., 2017; Toor et al. 2017). A study conducted by Mgese (2010) in the Berg River, South Africa, shows that industrial effluent and storm water runoff from formal settlements contribute to the increase in nutrients in the Berg River, but the highest contributor was from surface water drainage from informal settlements. According to Mgese (2010), runoff from Oliver Tambo informal settlement, adjacent to the Berg River, with its poor drainage and sanitation facilities, showed an increase in nutrient load especially during rainfall events. In another study of runoff from an informal settlement in Franschhoek, Fell (2017) concluded that surface water quality was poorest in areas that were dominated by the informal settlement and attributed the poor surface water quality from the informal settlement to dysfunctional sanitation systems and poor drainage.

It must be noted that it is not only runoff from informal settlements that contribute nutrients to the urban water fresh systems. Formal settlements also contribute to the nutrient load in fresh water systems. According to Toor et al. (2017), cities and towns alter the natural hydrology through impermeable surfaces such as roads, pavements, channeled storm water drainage systems and large rooftops which are responsible for increasing the rate of runoff, and by reducing infiltration and the combined pathways for the transfer of elevated levels of nutrients. Poor waste management strategies in urban areas also increases the concentration of N and P in surface runoff into the fresh water systems. Some of these lifestyles activities and choices such as the application of fertilizers to household gardens (especially lawns), decomposition of unattended garden waste and discharge of water from vehicle washing (Ward & Winter 2016; Toor et al., 2017). In a study by Toor et al. (2017) at Orange County California, United States, it was found that the nutrient and sediment loads were high in the residential catchments which was attributed to using reclaimed water to irrigate the lawns.

Agriculture is one of the major contributors of eutrophication into river systems (Selma & Greenhalgh, 2010). According to Selma and Greenhalgh (2010), the use of synthetic nitrogen fertilizer increased more than sevenfold between 1960 and 1990 while the use of phosphorus tripled during the same period. Most of the increased application of fertilizers is excessive and this result in the fertilizer been lost through surface runoff during the rainy season and some is lost to volatilization during the dry season (Ulén et al. ,2007). Some of the volatilized ammonia is redeposited in fresh water systems through atmospheric deposition (Ulén et al., 2007).

Similarly intensive livestock production has also resulted in an increase of nutrient fluxes over the past decade (Ulén et al., 2007). In South Africa, as in many other parts of the world, livestock farming is adopting new production systems such as feedlots in order to meet the demand for meat and meat products (Meissner et al., 2013). These methods of production result in agricultural waste being concentrated in a confined area and this waste finds its way into river systems as runoff during the rainy season (Amis et al., 2011).

A final source of nutrient enrichment is from environmental deposition. This occurs with the burning of fossil fuel to produce energy releases nitrogen oxides (NOx) into the atmosphere (Selma & Greenhalgh, 2010). NOx is then re-deposited in river systems and land as fallout from rain and snow (wet deposition), or it can be re-deposited through dry deposition. Some of the primary sources of NOx from burning of fossil fuels are electric power generation, industries and exhaust from motor vehicles (Selma & Greenhalgh, 2010).

2.3 Drivers of Eutrophication

Eutrophication occurs as a result of complex interrelated socio-economic factors that increase nutrient pollution into fresh water systems (Selma & Greenhalgh, 2010). Indirect drivers include population growth, economic development and an increase in intensive agriculture. Direct drivers of eutrophication include higher energy demand, increased fertilizer use and land-use change (Selma & Greenhalgh, 2010).

Population growth is one of the indirect drivers of nutrient pollution. Most of the nutrients that enter river systems are associated with human activities and thus the growth in population is expected to have a positive correlation with nutrient pollution. Although there are uncertainties about future population projections, studies suggest that the world population is likely to grow by 30% between 2000 and 2025 and as much as 50% between 2000 and 2050 (United Nations Secretary-General's High-Level Panel on Global Sustainability, 2012). The global population is expected to reach 8.6 billion in 2030 and increase further to 9.8 billion in 2050 and 11.2 billion by 2100, with the majority of the population growth occurring in less developed countries (UNDESA, 2017). In South Africa, population is predicted to grow from 50,5 million currently to 56.8 million by 2050.

Population growth will lead to an increase in food demand resulting in intensified agriculture production and growth, and an increase in demand for energy for households, industries and for transport (Pauw, 2011; Selma & Greenhalgh, 2010). Population growth is also associated with urban migration. Urban migration puts pressure on existing infrastructure like waste water treatment works which results in infrastructure failure because the system is unable to perform optimally (Pauw, 2011; Dabrowski & Klerk, 2013). In a study by De Villiers (2007) on the status of nutrients in the Berg River, South Africa, the results showed that the most populated areas along the river coincided with high levels of nutrients. In another study by De Villiers & Thiart

(2007) on the nutrient status of the South African rivers showed that all the catchments with more than 20% agricultural land use area were high in N and P. The high N and P flux values, according to De Villiers & Thiart (2007), were due to the application of fertilizers in cultivated areas.

2.4 Climate Change and Eutrophication

The causes and drivers of eutrophication are well known but little research has been conducted on the impact of climate change on eutrophication and water quality in general (Whitehead., 2009; Charlton et al., 2018). Worldwide levels of nutrient loading in river systems are predicted to increase as a result of climate change (Xia et al., 2016, Charlton et al., 2018). However, assessing the impacts of climate change to eutrophication is difficult because eutrophication is a complex non-linear interplay between available nutrients, temperature, sunlight, residence time and flow conditions (Whitehead et al., 2009; Suikkanen et al., 2013). The general expectation is that eutrophication will increase due to climate change, due to increasing temperatures and decreased summer flows (Bowes et al., 2012; Bowes et al., 2016; Xia et al., 2016).

Different studies use climatic scenarios and hydrologic models to predict and understand the effect of climate change on eutrophication in freshwater systems (MacKellar et al., 2014). Due to the complexity of these processes, the results are often uncertain because some processes may be operating at different time scales (Fragoso et al., 2011). Currently widespread experiments, data analysis and process-based modelling have been conducted to identify the role of climate change on eutrophication but thus far have had no clear or defined success (Xia et al.,2016). Despite differences in the results of models and experiments on climate extremes, there is an expectation that climate extremes will affect freshwater systems negatively (Fischer et al., 2013).

2.4.1 Relationship between temperature and eutrophication

In 2018 the Intergovernmental Panel on Climate Change (IPCC) stated that anthropogenic activities were responsible for approximately 1.0°C of global warming above pre-industrial levels with possible ranges of 0.8°C to 1.2°C. The IPCC (2018) further stated that global warming at the current rate would reach 1.5°C between 2030 and 2052. According to the Climate Change Synthesis Report (2014), the global mean surface temperature is predicted to increase over the next 100 years from 2000 to 2100 as a result of two different emission scenarios being the Representative Concentration Pathways (RCP) 2.6 and RCP 8.5. The Climate Change Synthesis Report (2014) further revealed that surface temperature are predicted to rise from 2.6°C to 4.8°C under RCP 8.5, and 0.3°C to 1.7°C under RCP 2.6 by the end of the 21st century. The rise in temperature due to climate change will result in an increase in toxin-producing cyanobacterial algal blooms in the river systems as seen in the past few decades (Paerl & Otten, 2013). The most significant climate change factor is expected to be a change in water temperature due to the general raising temperatures. According to Xia et al.(2016), it is expected that water temperature in rivers will increase with increasing air temperature. When water temperature rises above 25°C it is likely to create conditions for increasing the growth rate of cyanobacteria (Suikkanen et al., 2013; Xia et al., 2016). It is also expected that warmer temperatures will stimulate other phytoplankton groups, e.g. green algae and diatoms (Xia et al., 2016).

Different case studies around the world show that there is a positive correlation between temperature rise and toxin-producing cyanobacterial algal blooms in river systems (Xia et al., 2016). Several controlled experiments and process-based models confirm the hypothesis that an increase in temperature will support the growth and distribution of cyanobacteria. A study conducted by Valdemarsen et al. (2015) in a shallow estuary (Odense Fjord, Denmark), showed that elevated temperatures increase microbial activity in the sediments and soils in benthic zones, thus increasing the release rate of internal phosphorus loading, which contributed to the total nutrient load in the water body. Other studies performed by Whitehead et al. (2009) and Xu et al. (2011) showed that annual temperature rise in the temperate zones resulted in algal growth. Another study conducted by O'Farrell et al. (2011) indicated that macrophytes play a key role in the interception and detention of nitrogen and phosphorus.

2.4.2 Precipitation extremes and eutrophication

Precipitation projections of global climate models show large uncertainties but indicators suggest an overall increase in the intensity of precipitation extremes (Fischer et al., 2013; O'Gorman 2015; Xia et al., 2016a). Model simulations from the Coupled Model Intercomparison Project Phase 5 (CMIP5) project predicts widespread variation in heavy precipitation events and successive number of dry days (Kharin et al., 2013; Fischer et al., 2013). Most studies on extreme rainfall focus on drought and only a few have concentrated on the impact of floods on eutrophication (Xia et al., 2015).

Intense precipitation is likely to increase the extent of eutrophication by mobilizing nutrients, pathogens and toxins on land from non-point sources and through the release of sediments through erosion thereby increasing the enrichment of receiving river systems (Whitehead et al., 2009; Xia et al., 2016a). Runoff from urban areas is known to transport pollutants such as chemicals, oils, solid waste, faecal and all other types of pollutants especially during the 'first flush' from rainfall events and these pollutants eventually end up in freshwater systems (Winter & Mgese, 2011). The highest concentration of nutrient runoff during the 'first flush' effect usually occurs after a long dry period when there has been time to allow pollutants to accumulate (Winter & Mgese, 2011).

Drought or low precipitation also have a negative impact on freshwater systems (Xia et al., 2016). Low or a lack of precipitation will result in low flow rates that will increase the residence time of water in the river system. An increase in residence time will in turn increase the growth potential of algae, increase the rate of sediment depositions and this eventually will reduce turbidity so that light penetration can boost algae growth (Whitehead., 2009). Low flow rates can also mean that the nutrients from agricultural activities or from WWTW are not diluted due to less water in the river system. Lack of dilution of nutrients entering freshwater systems will have an impact on the nutrient concentrations through an increase in biochemical oxygen (BOD) and lower dissolved oxygen (DO) potential to increase the risk of eutrophication (Xia et al., 2016).

2.4.2 Wind and solar radiation effect

The effect of wind and solar radiation on eutrophication has received limited attention when compared to studies on the effects of temperature and precipitation within the climate change research field (Xia et al., 2016a). Wind is a key mechanism transferring moisture and temperature, and solar radiation is the source of most energy transfers in the global system (Fant et al., 2016). There are predictions for long-term change in wind and solar radiation (Fant et al., 2016). According to Xia et al. (2016) there are indications of long-term changes in the large-scale atmospheric circulation and due to the poleward shift and strengthening of the westerly winds which are predicted to continue changing in future.

A study by Fant et al. (2016) applied Atmospheric General Circulation models (GCM) to predict wind speed changes for southern Africa from the present to 2050. The results showed a relatively small change with modes close to zero and ranges from -1.5 to +1.5 m/s. Though the changes are minimal they tend to suggest a general increase in wind speed by 2050 in most parts of southern Africa. In a similar study, Eichelberger et al. (2008) used GCMs to predict wind speed values for 2050. Here the results show that there will be stronger wind speeds in the northern hemisphere including Canada, northern Europe, central and South America, tropical and subtropical regions while weaker winds will be experienced in the west coast of South America. Increased wind speed alters the flow rate and circulation of water, thereby disturbing the water stability and supporting the mixing of nutrients (Xia et al., 2016). Another indirect way that increased wind speed affect eutrophication is when warm air is circulated through water it warms the top layers of the river water and through wind circulation these top layers are mixed with the lower layers resulting in an increase in river water temperature and warm water promotes the growth of algal blooms thus promoting eutrophication (Rui et al., 2012).

Just like other climate change factors research studies show that solar radiation is increasing. In a study by Williamson et al. (2014) on the observation and projection of changes in annual mean UV-B radiation from 1960 to 2100, show that the radiation change varies significantly at different latitudes with a more than 20% increase at 60º to 90º southern latitude, and around a 10% decrease at 60º to 90º northern latitude. Despite the different changes at different latitudes the study found that the overall trend of the solar radiation is increasing at the earth surface. According to Bais et al. (2011) the Chemistry Climatic Models (CCM) show that by the year 2100 UV radiation will have increased in the tropics and will have decreased at polar latitudes.

The modelling and prediction of solar radiation is complex because it depends on other various atmospheric factors such as clouds, total ozone, aerosols, temperature in the stratosphere and specific locations thus the models will contain high uncertainties in particular to so far as clouds are concerned (Bais et al., 2011). Normal solar radiation is essential for photosynthesis and is also important for the growth of aquatic organisms. Increasing or above normal solar radiation is not ideal for freshwater systems because an increase in temperature promotes the growth of phytoplankton which in turn increase the risk of eutrophication (Xia et al., 2016).

2.4.3 Climate change and eutrophication in South Africa

South Africa has a network of weather recording stations from which to develop models for future climate change (MacKellar et al., 2014). Much of South Africa's information on climate change is consolidated in the National Department of Environmental Affairs Long Term Adaptation Scenarios (LTAS) Project (DEA, 2013). The primary tool used to make projections under the LTAS is in downscaling two global circulation models (GCMs) under different scenarios separated into two main groups, being unmitigated Special Report on Emissions Scenarios (SRES) (A2 and B1) and Representative Concentration Pathway (A2 and RCP 8.5) and mitigated (B1 and RCP 4.5) future energy pathways (DEA, 2013). The GCM models show that long-term trends in temperature indices appear easier to model as compared to the changes in the rainfall indices with increasing certainty in temperature projections than rainfall projections (DEA 2013; Samuel et al., 2014; MacKellar et al., 2014).

GMC models in South Africa revealed that mean annual temperatures have increased by at least 1.5 times the observed global average of 0.65ºC over the past decades and it also shows that erratic rainfall and longer dry events have increased in frequency (DEA 2013; MacKellar et al., 2014). According to DEA (2013), climate projections for South Africa up to 2050 and beyond show significant warming of between 5–8°C over the interior and reduced warming over the coastal zones. The study further reveals drier conditions to the west and south of the country with wetter conditions over the eastern part of the country. In another study by Kruger & Sekele (2013) on trends in daily maximum and minimum extreme temperatures from 1962 – 2009, results indicate that maximum temperature indices are generally decreasing in warm extremes while the minimum temperature indices are generally decreasing in cold extremes in the western, north eastern and eastern parts of the country. A study by MacKellar et al. (2014)

also found a significant warming trend in maximum temperatures from 1960 to 2010 with the strongest average reaching close to 2 °C in the central interior during autumn and the weakest increase of 0.35 °C happening in summer within the same region. Interestingly, unlike many other authors, MacKellar et al. (2014) reported a decline in rainfall extremes which is different from other studies.

Climate change is a major concern for already stressed water resources in the Western Cape South Africa. According to MacKellar et al. (2014) trends in rainfall in the Western Cape province of South Africa show drier conditions along the coastal region with increased rain days near the west coast while the maximum and minimum temperatures increased significantly in all seasons. As already mentioned climate change will have a negative impact on fresh water systems particularly when there is high temperatures and less precipitation which is the predicted climatic condition for the Western Cape, South Africa (DEA, 2013). An increase in temperature has the potential to promote an increase in the growth of many cyanobacteria leading to eutrophication of freshwater systems (Xia et al., 2016). Low precipitation or reduced amount of rainfall could increase the residence time of water in river systems and a longer residence time will increase the chance of algal growth and a reduction turbidity thus exacerbating the process of eutrophication (Whitehead., 2009). In summary, climate change factors are likely to increase eutrophication because of higher temperatures, less rainfall, unstable wind speeds and stronger solar radiation. These factors coupled with land use changes such as intensified agricultural production, population growth and economic growth will make eutrophication difficult to manage.

2.5 Monitoring of Eutrophication

Accurate information is needed to make informed decisions and develop strategies and formulate policies to manage water resources in a sustainable manner. Hence, management of eutrophication should be informed by well-established monitoring programmes that are able to collect accurate and consistent data for proper assessment of eutrophication (DWA 2013). Data collection, analyses and interpretation of information on water quality are required to inform improved water resource management (DWA 2013). Monitoring of freshwater systems is limited in developing countries due to limitations of human capacity and technological resources. Most information on the current nutrient loads and fluxes is derived from a manual approach to monitoring and grab sampling (Pellerin et al., 2016).

The manual approach of water quality monitoring has several disadvantages such as human errors during sampling and analysis, errors during transportation and storage of samples and errors due to malfunctioning of instruments used during sampling procedures (Sarpong Adu-manu et al., 2017). The frequency and consistence of data collection is also crucial to the quality of information derived from the monitoring operations. There is always doubt in the precision and accuracy of data collected manually due to human error resulting in inconsistencies in data (Murphy et al., 2015). Sometimes data will be collected once week, fortnightly, once a month or not collected at all. While the manual water quality monitoring coupled with mathematical modelling and statistical techniques has been useful and produced important information, advancement in new technologies offers an opportunity to introduce water quality monitoring approaches with better precision and consistency (Murphy et al., 2015; Pellerin et al., 2016; Sarpong Adu-manu et al., 2017).

There is a gradual move from the manual water quality monitoring to a more internet and technology based approach especially in the developed countries (Chen & Han, 2018). The last ten years have seen an increase in the amount of water quality data collected through the use of smart sensors, remote sensing and earth observation systems (Sarpong Adu-manu et al., 2017; Chen & Han 2018). Wireless sensor networks (WSNs) are gaining popularity in water quality monitoring as technology and the internet improves (Sarpong Adu-manu et al., 2017). WSNs have demonstrated the ability to capture and analyze data from different sampling points and transmit the data in a cost effective manner using cheaper wireless communication platforms (Chen & Han, 2018). According to Pule et al. (2017), these low-cost WSNs are able to monitor water quality remotely, disseminate information in real time and with very little human involvement. WSNs are able to eliminate human error from both collection and analyses of data, reduce the cost of monitoring by eliminating transportation of samples and expensive laboratory analysis and the ability to monitor sites where human beings will not be able to get access (Sarpong Adu-manu et al., 2017).

Remote sensing application is another technique capable of measuring the water quality of inland water bodies like reservoirs, lakes streams and rivers (Gholizadeh et al., 2016; Chen & Han 2018). Although remote sensing is useful for monitoring water quality, it suffers from relatively low spatial resolution which makes it unsuitable to monitor water quality of freshwater systems like rivers and channels (Chen & Han, 2018). Even in lakes and reservoirs remote sensing is not recommended to be used as a stand-alone technique but rather to be used to compliment the manual water monitoring technique (Gholizadeh, Melesse & Reddi, 2016). Remote sensing focuses mainly on optical variables such as chlorophyll-a, total suspended solids and turbidity making it useful for monitoring and managing eutrophication (Chang et al., 2015; Gholizadeh et al., 2016). The current ongoing improvement in space technology and research on the use of remote sensing in monitoring eutrophication and cyanobacterial blooms in water bodies will make remote sensing a useful tool in monitoring eutrophication in the near future (Matthews & Bernard, 2015b). The future of eutrophication monitoring and/or water quality monitoring is dependent on low-cost, robust and easy to operate WSNs and an integrated application of remote sensing, on site sampling and computer aided water quality modelling (Murphy et al., 2015; Chen & Han 2018).

2.6 Management of Eutrophication

The challenge of managing eutrophication due to the competing demand of water between humans and ecosystems results in a complex interaction between freshwater systems and social systems. Furthermore, as these interactions change they create new feedbacks that need to be managed (Liu et al., 2015). Many countries struggle with the management of eutrophication due to lack of resources. In South Africa water quality is managed through the South African Water Quality Guidelines which stipulate the Target Water Quality Range (TWQR) (DWAF, 1996). Specifically, eutrophication is managed through the National Eutrophication Monitoring Programme (NEMP) implementation manual which is underpinned by the South Africa National Water Act, No. 36 of 1998 (DWAF, 2002). Despite all these efforts eutrophication continues to be a problem in South Africa and the deterioration of freshwater systems continues to be an overwhelming challenge for water resource managers.

There is no single management approach known to resolve cases of eutrophication and thus water resource managers are constantly exploring new approaches where different sets or combinations of management strategies support an improved understanding of eutrophication that interconnects with the broader problem of environmental degradation (Van Dolah et al.,2016; Voulvoulis et al.,2017). Eutrophication phenomenon cannot be managed in isolation and treated as separate from the whole environmental system (Voulvoulis et al.,2017). The traditional approach of managing nutrient enrichment of freshwater system in many countries including South Africa is through the reduction of the nutrient load into fresh water systems (van Ginkel, 2011; Harding, 2015; Voulvoulis, Arpon & Giakoumis, 2017). Under this approach, nutrient parameters are regulated and monitored at the point of source to try and limit nutrients entering the freshwater systems based on threshold standards (van Ginkel 2011; Voulvoulis et al.,2017). While this management approach had been effective in the past and assisted many countries to reduce high levels of nutrient load in the freshwater systems, it needs to be complemented with other existing and emerging management strategies that considers the complexity of the ecosystem as a whole (van Ginkel 2011; Pahl-Wostl et al., 2013; Van Dolah et al., 2016; Voulvoulis et al., 2017).

Several management initiatives have emerged that deal with eutrophication around the world. One of these initiatives is what van Ginkel (2011) terms adaptive eutrophication management. The concept of adaptive management has been widely deliberated in water resources management and ecosystem for some time (Folke et al., 2005; Folke et al., 2011; Pahl-Wostl et al. 2013). Adaptive management is based on the principle that management strategies should be flexible and be able to change as new information and results become available

(Pahl-Wostl 2007; Pahl-Wostl et al., 2013). Adaptive management is thus experimental by nature and is based on the concept of learning through doing and continually improving the management strategies based on the outcomes of the implementation process (Pahl-Wostl et al., 2013). Adaptive management is pro-active, flexible and allows changes to be made to suit different scenarios. According to van Ginkel (2011) there are six steps that are crucial to the success of adaptive eutrophication management.

The first step of adaptive eutrophication management is monitoring to determine which sites need intervention and monitoring leading to the second step which is to direct research that will identify site specific management solutions; the third step is to develop plans that are implementable and adequately budgeted for by the implementing institutions; the fourth step is to implement the plans; the fifth step is to monitor the success of the plans in order to measure their effectiveness; the last is to adapt management plans based on their success (van Ginkel, 2011).

Another management approach in eutrophication management is the Socio-Ecological System (SES) approach (Van Dolah et al., 2016). SES is an interdisciplinary approach based on the principle that human beings and the environment are intertwined and that you cannot study one in isolation of the other (Chaffin & Gunderson 2016; Van Dolah et al., 2016). SES combines the social and ecosystems sciences to investigate complex environmental problems like eutrophication and acknowledge that complex environmental problems like eutrophication *are nonlinear interactions between social, cultural, political, ecological, and biophysical processes* (Van Dolah et al., 2016). The use of SES in eutrophication research and management can identify the main drivers of the problem and can facilitate the correct remedial actions by bringing together the social and ecosystem processes (Van Dolah et al., 2016).

Finally a systems management approach involves an understanding of the interplay between land and water under different scenarios (Voulvoulis et al., 2017). One key example of systems approach in water resources management was the introduction of the European Union Water Framework Directive 2000/60/EC (WFD), which aims to end water management in isolation and rather to establish European freshwater management systems that include all aspects of the water environment (Vlachopoulou et al., 2014). The WFD demands a 'catchment-based approach' and 'integrated river basin management' in which water and land should be managed as a system (Voulvoulis et al., 2017). The implementation of the WFD is built on the foundation of proper monitoring to ensure that interventions are site specific and cost effective, and considers the complex interplay between human and natural systems requiring interdisciplinary research to address the complex issue of water resource management (Voulvoulis, Arpon & Giakoumis, 2017).

CHAPTER 3: METHODOLOGY

3.1 Introduction

The study examines the water quality data from nine monitoring sites distributed along the Berg River during the period from 1976 to 2016. The aim is to determine downstream and long-term trends from a selection of nutrient parameters: nitrate and nitrite (NO_3^- and NO_2^-, or NO_x), ammonium (NH_4^+), total phosphorus (TP) and ortho- phosphate (PO_4^{3-}). DWS water quality monitoring data are available for these stations and some stations have data from as far back as 1967 while the majority of the data were obtained from the late 1970s.

3.2 Sample Site Selection

Nine monitoring stations were chosen because they were adjacent to certain land use activities along the Berg River that were dominant and easy to characterise. The monitoring points are labelled LB1 and LB2, and B1 to B7. The labelling was adopted from the DWS RQWS for easy reference. LB1 refers to the Little Berg River monitoring point 1 and 2 while B1 means Berg River monitoring Site 1 to site 7. B1 is deliberately excluded from the analysis because data from this point was only available until 2002. This is the area where the Berg River dam was built in 2002 and was completed in 2007. LB1 and LB2 are located in the upper reaches of the study area where there is minimal human development or disturbances in the vicinity of these two sites. B2 and B3 are situated downstream of the towns of Paarl and Wellington respectively. These are relatively densely populated areas and where vineyards and fruit orchards are irrigated using water from the Berg River. B4 and B5 are located in a predominantly dryland farming area of wheat and livestock production. B6 and B7 are located towards the lower reaches of the catchment near the coast in an area where irrigation farming takes place. Table 2 below shows the geo-referenced position and description of each monitoring stations. DWS ID refers to reference code that form part of the DWS database.

Table 2: Location of monitoring sites along the Berg River

Monitoring Point	DWS ID	Location	Lat °S	Long °E
LB1	G1H021	Mountainview	33.185	19.155
LB2	G1H008	Niewkloof	33.311	19.075
B1	G1H004	Bergriviershoek	33.927	19.061
B2	G1H020	Dal Josafat	33.708	18.911
B3	G1H036	Hermon	33.435	18.957
B4	G1H013	Drieheuvels	33.133	18.862
B5	G1H031	Misverstand	32.997	18.779
B6	G1H023	Jantjiesfontein	32.925	18.329
B7	G1H024	Kliphoek	32.817	18.194

Different land use activities along the Berg River include dryland crops, irrigated fields, livestock farming, forests plantation, residential (formal and informal), industrial and natural vegetation as shown below in Figure 2. The sites were also chosen after due consideration of differences in length and completeness of time series data between monitoring stations.

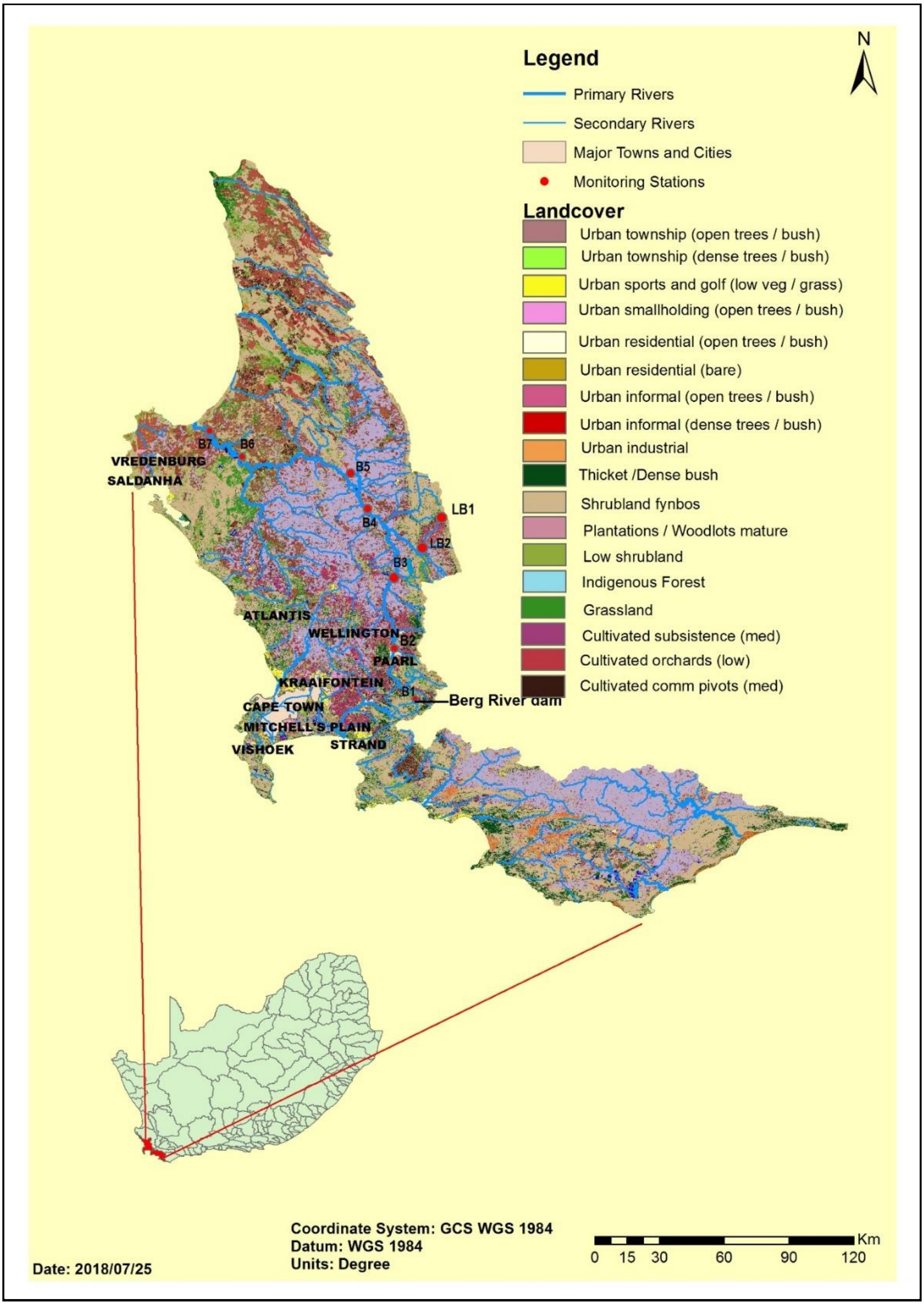

Figure 2: Map showing different land use activities along the Berg River

3.3 Data Collection

Data were extracted from the DWS Resource Quality Information Services (RQIS) website (DWS, 2017c) (http://www.dwa.gov.za). In some cases these monitoring stations were sampled weekly while others were sampled monthly. In order to maximise accuracy an average monthly value was used to provide a time-series data at a monthly resolution. The study only focused on the following nutrient parameters: nitrate and nitrite (NO_3^- + NO_2^-, or NO_x) and NH_4^+ expressed as µg N/ℓ, dissolved total phosphorus (TP) expressed as µg P/ℓ and soluble reactive phosphate (SRB) measured as (PO_4^3), expressed as µg P/ℓ. Ideally chlorophyll-a should have been included but this data was un-available for most monitoring stations.

3.4 Statistical Analysis

The monthly data sets were statistically analyzed to determine if there was any significant differences between the different sites using parametric statistical parameters such as mean, median, and maximum values to describe the variables and compare the results of different years. The data were further analyzed using student t-tests to determine if there were any significant differences in the nutrient levels between monitoring sites during the last two decades (1996-2006 and 2007-2017). The last two decades are important because that is when the population and economic activities in the study region increased (Refer to population data in Table 1). Another reason is that the Berg River dam construction started officially in 2002 and was in finally put into operation in 2009. The introduction of the dam could influence the nutrient concentration in the river. De Villiers (2007) had predicted in an early study that nutrient levels are likely to increase with the abatement of the Berg River in the upper catchment. The data were also compared with previous studies to determine whether the nutrients were increasing or decreasing.

3.5 Eutrophication Guidelines

This study compared the nutrient levels in the Berg River to both South African standards and International standards in order to determine the status of the Big River. The South African Trophic status classification of freshwater ecosystems were developed based on (a) average summer levels of inorganic nitrogen (NO_3^- + NO_2^- + NH_4^+ + NH_3) and inorganic phosphorus (measured as PO_4^{3-}) from the South African Water Quality guidelines for Aquatic Ecosystems and (b) mean annual chlorophyll-a and mean annual total phosphorus levels, from the National Eutrophication Monitoring guidelines (De Villiers, 2007). Table 3 shows the trophic status classification as adopted from De Villiers (2007).

Table 3: Trophic status classification of freshwater ecosystems

Trophic status classification of freshwater ecosystems				
Parameter	**Oligotrophic**	**Mesotrophic**	**Eutrophic**	**Hypertrophic**
Inorganic Nitrogen(ugN/l) – SAWQ guidelines for Aquatic Ecosystems	< 500	500 – 2 500	2500 -10 00	> 10 000
Inorganic Phosphorus (ugP/l) -SAWQ guidelines for Aquatic Ecosystems	< 5	5 – 25	25 -250	> 250
Mean annual total phosphorus (ug P/l) - NEMP	< 15	15 - 47	47 - 130	> 130
Mean annual chlorophyll a (ug/l) - NEMP	< 10	10 - 20	20 - 30	> 30

For purposes of international comparison, the study referred to the Australian and New Zealand Guidelines for fresh and Marine Water Quality (NWQMS) ANZECC & ARMCANZ (2000) and the International Water Quality Guidelines for Ecosystems (IWQGES) developed by the United Nations Environmental Programme (UNEP) (UNEP/UNU-EHS, 2016). The two guidelines were chosen because they both recognise that the best approach to water quality management is based on high-status guidelines that determine the national objective while flexible and adaptive to different conditions at regional and local levels. Both guidelines promote an iterative, adaptive management framework to deal with nutrients in freshwater systems while maintaining flexibility in order to achieve the goal of protecting freshwater systems (UNEP/UNU-EHS, 2016).

The Australian and New-Zealand general environmental regulation and management are relevent because they offer a holistic and cohesive pollution prevention approach to environmental protection. Table 4 below shows the water quality guidelines for Australia and New Zealand. The Australia and New Zealand NWQMS are divided into different regions and these regions are selected on the different climatic conditions that prevail in those countries. The table is adopted to show only the parameters that are relevant to the study. Table 5 below shows the water quality guidelines from the IWQGES and shows only those parameters that are relevent to this study.

Table 4: Australian and New Zealand Guidelines for fresh & Marine Water Quality

South East Australia					
Ecosystem Type	**Chl-a (ugL-1)**	**TP (ugPL-1)**	**TN (ugNL-1)**	**NOx (ugNL-1)**	**NH4+ (ugNL-1)**
Upperland river	N/A	20b	250[c]	15[h]	13[i]
Lowerland river	5	50	500	40o	20
South West Australia					
Ecosystem Type	**Chl-a (ugL-1)**	**TP (ugPL-1)**	**TN (ugNL-1)**	**NOx (ugNL-1)**	**NH4+ (ugNL-1)**
Upperland river	N/A	20	450	200	60
Lowerland river	3 - 5	65	1200	150	80
Tropical Australia					
Ecosystem Type	**Chl-a (ugL-1)**	**TP (ugPL-1)**	**TN (ugNL-1)**	**NOx (ugNL-1)**	**NH4+ (ugNL-1)**
Upperland river	N/A	20	450	200	60
Lowerland river	3 -5	65	1200	150	80

South Central Australia					
Ecosystem Type	**Chl-a (ugL-1)**	**TP (ugPL-1)**	**TN (ugNL-1)**	**NOx (ugNL-1)**	**NH4+ (ugNL-1)**
Upperland river	na	NO data	1000	100	100
Lowerland river	na	100	1000	100	25

na = not applicable;
a = monitoring of periphyton and not phytoplankton biomass is recommended in upland rivers — values for periphyton biomass (mg Chl a m-2) to be developed;
b = values are 30 µgL-1 for Queensland rivers, 10 µgL-1 for Vic. alpine streams and 13 µgL-1 for Tas. rivers;
c = values are 100 µgL-1 for Victorian. alpine streams and 480 µgL-1 for Tasmanian rivers;
c = values are 100 µgL-1 for Vic. alpine streams and 480 µgL-1 for Tas. rivers;
d = values are 3 µgL-1 for Chl a, 25 µgL-1 for TP and 350 µgL-1 for TN for NSW & Vic. east flowing coastal rivers;
e = values are 3 µgL-1 for Tasmanian. lakes;
f = value is 5 µgL-1 for Queensland estuaries;
g = value is 5 µgL-1 for Victorian. alpine streams and Tasmanian rivers;
h = value is 190 µgL-1 for Tasmanian. rivers;
i = value is 10 µgL-1 for Queensland. rivers;
o = value is 60 µgL-1 for Queensland rivers;

Table 5: International Water Quality Guidelines for Ecosystems

Physico-chemical	**Category 1 (High Integrity)**	**Category 4 (Extreme impairment)**
Total Phosphorus (TP) (µg/ℓ)	≤ 20	≥ 190
Total Nitrogen (µg/ℓ)	≤ 700	≥ 2500
Chlorophyll-a (µg/ℓ)	≤ 5.0	≥ 125
Un-ionized ammonia (µg NH_3/ℓ)	15[5]	100[5]

The general purpose of water quality guidelines are to set a water quality criteria that is used to protect and manage the freshwater ecosystems (UNEP/UNU-EHS, 2016). These guidelines stipulate the threshold values of the level of concentrations of chemicals and physical parameters (UNEP/UNU-EHS, 2016). The purpose of the guidelines is to offer stakeholders a set of tools for assessment and management of water quality (ANZECC & ARMCANZ, 2000). Most of the eutrophication guidelines use numeric water quality criteria for nitrogen and phosphorus as the parameters to measure eutrophication while a few guidelines include chlorophyll-a as a measure of eutrophication (DWAF, 2002). Countries and regions have different water quality criteria guidelines and use different approaches and methodologies. Some specify the maximum concentrations permissible while others specify a range of values that are allowed within a specific site (DWAF, 1996).

CHAPTER 4: RESULTS AND DISCUSSIONS

4.1 Introduction

Statistical analyses were conducted for the purpose of determining the concentration and trends in nutrient loading of the Berg River. The assumption is that anthrophogenic activites in the catchment area have impacted on the quality of water resources of the Berg River. A t-test analysis was conducted for three selected sites to determine the change in the nutrient load between the last two decades. The first decade is represented by data from 1996 to 2006 and the second decade is represented by data from 2007 to 2017. For purposes of t-test analyses, one site was selected in the upper section of the river, another one in the middle section of the river and the last one was selected in the lower section of the river. LB1 represent the upper section of the river, B3 represented the middle section of the river and B7 represent the lower section of the river. The monitoring sites are shown in Figure 3 below.

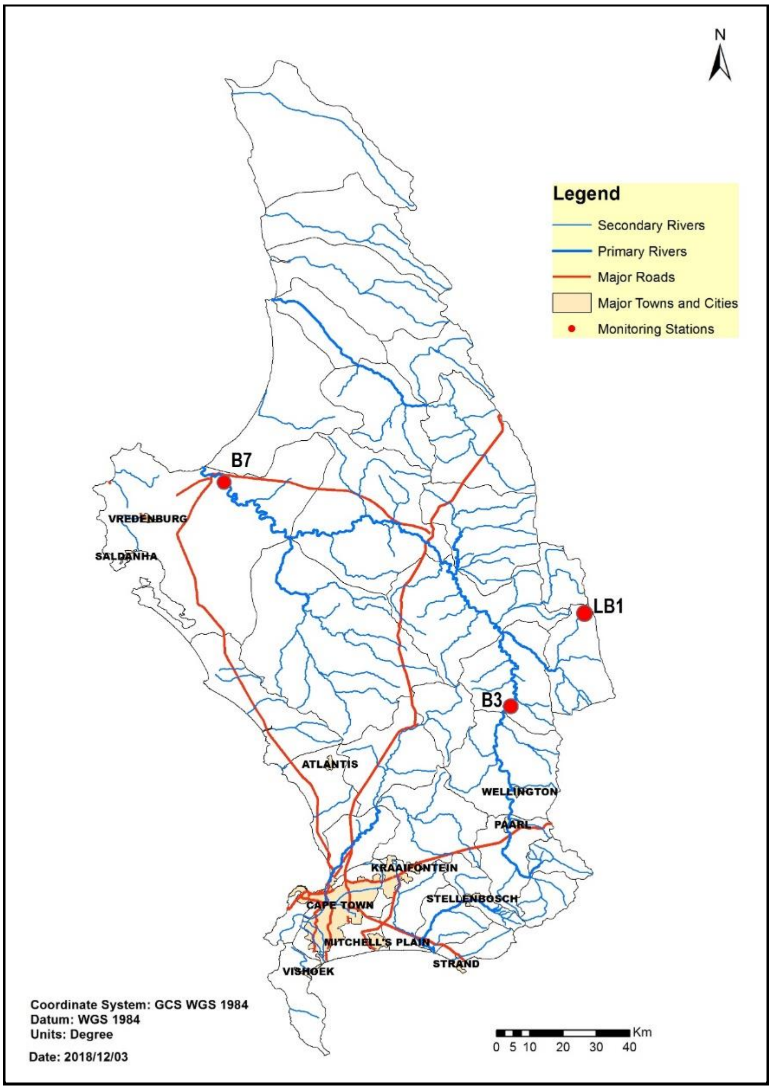

Figure 3: Map showing the monitoring stations along the Berg River

4.2 Concentration and long-term trends in river water nutrient levels

4.2.1 General trend in nutrient levels

Table 6 below shows the long-term monthly median, mean and maximum values for points LB1, LB2, B2 to B7. As already mentioned site B1 was omitted because data were only available until 2005.

Table 6: Long-term monthly median, mean and maximum river nutrient values

Berg River water quality monitoring station detail, time-series data median, mean and maximum dissolved (NOx), NH_4^+, TP and PO_4^{3-} values from 1987 to 2017. (N/A denotes stations were no data is available).												
Monitoring Site	NOx (μgN/ℓ)			NH_4^+ (μgN/ℓ)			TP (μgP/ℓ)			PO_4^{3-} (μgP/ℓ)		
	med	mean	max	med	mean	max	med	mean	max	med	mean	max
LB1 - Mountainview	68	73	401	25	37	335	N/A	N/A	N/A	10	17	304
LB2 - Niewkloof	139	272	2390	25	41	725	51	66	1590	17	26	1090
B2 - Dal Josafat	587	593	2858	50	67	1900	90	107	559	26	33	780
B3 - Hermon	988	1030	5353	50	73	1798	263	278	880	91	116	915
B4 - Drieheuvels	392	594	3774	37	51	561	107	115	610	29	42	489
B5 - Misverstand	329	544	7107	42	53	534	98	116	992	26	33	487
B6 - Jantjiesfontein	144	482	6346	50	53	357	N/A	N/A	N/A	37	37	495
B7 - Kliphoek	118	380	3270	80	160	1509	N/A	N/A	N/A	42	49	499

According to Table 6 above, the upper sections of the river (LB1 and LB2) recorded the lowest values for all nutrient parameters. Low concentrations were expected as there is minimal human activites in the upper catchment of the river. The long term mean values show a downstream increase in NOx, NH_4^+ ,TP and PO_4^{3-} with a peak at site B3 along the middle sections of the river. All parameters were observed to have increased at monitoring Site B3 making it the most polluted point throughout the whole catchment. Though the values are still high there is a decrease towards the lower sections of the river at monitoring site B7 when compared to the middle section of the river at B3. The general observation is that of low values for the upper section of the river (LB1 and LB2) with a steady increase at B2 followed by a sharp increase at B3. The lower values in the lower section of the river can be attributed to the river's assimilative capacity (Hashemi Monfared et al., 2017).

4.2.2 Comparisons

The long term mean, median and maximum nutrient values that are presented in Table 6 were compared with South African trophic status classification of freshwater ecosystems (De Villiers 2007), the Australian and New Zealand Guidelines for fresh and Marine Water Quality (NWQMS) ANZECC & ARMCANZ (2000) and the International Water Quality Guidelines for Ecosystems (IWQGES) developed by the United Nations Environmental Programme (UNEP) (UNEP/UNU-EHS, 2016). De Villiers (2007) used the trophic classification of freshwater ecosystems that was based on (a) average summer levels of inorganic nitrogen (NO_3^- + NO_2^-

+ NH_4^+ + NH_3) and inorganic phosphorus (measured as PO_4^{3-}) from the South African Water Quality guidelines for Aquatic Ecosystems (DWAF,1996a) and (b) mean annual chlorophyll-a and mean annual total phosphorus levels, from the National Eutrophication Monitoring guideline (DWAF,2002).

According to the trophic status classification of fresh ecosystems (Table 3), almost all the monitoring sites on the Berg River were classified as eutrophic. Total phosphorous (TP) data were not available for all sites but were used in this study because TP levels are considered to act as a more reliable indicator of emerging eutrophication (van Ginkel, 2011; Harding, 2015; Griffin, 2017). TP at all sites was eutrophic with the exception of B3 which was hypertrophic as shown in Figure 4 below.

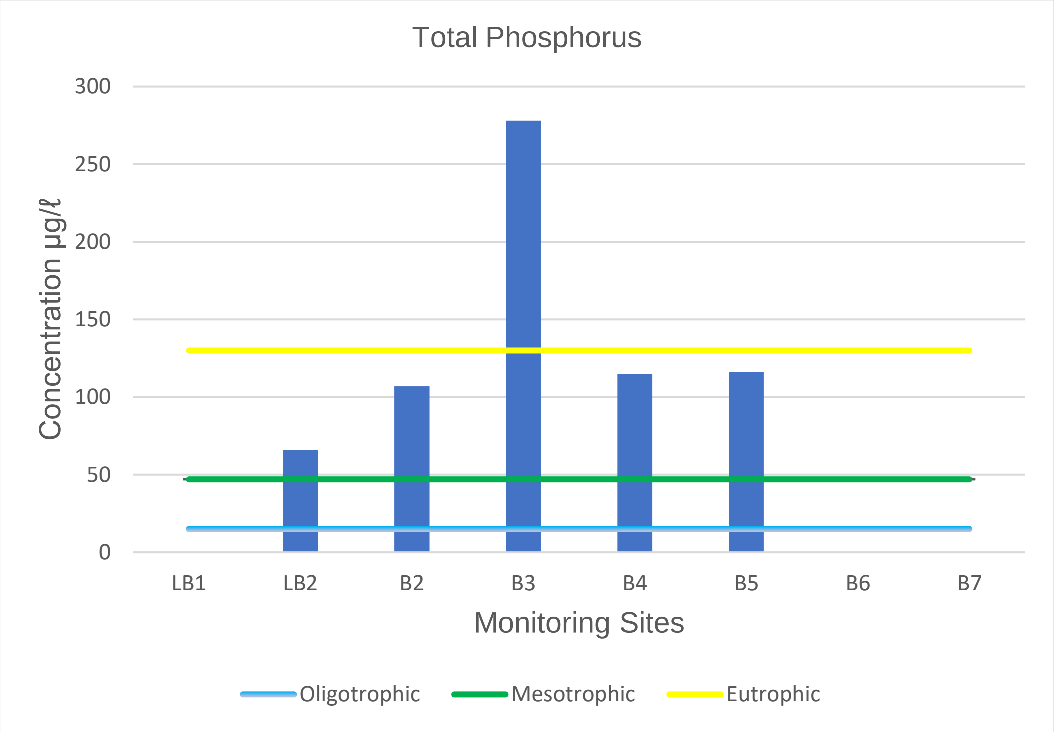

Figure 4: Trophic status of the Berg River for the period 1987 – 2017 (Note: no TP data available for Sites LB1, B6 and B7)

All the monitoring sites where TP data was available exceeded the 20 µg/ℓ which is the recommended international quality guideline for ecosystems. The highest recorded TP mean value was 278 µg/ℓ and it was recorded at monitoring Site B3. In the 10 year period from 1997 to 2006, Site B3 observations were hypertrophic for 92% of the time and between from 2007 to 2017 it was hypertrophic 93% of the time. The long-term mean values of NOx in the Berg River for B2 to B6 show results exceed the recommended guideline for ecosystems of 450 µg/ℓ for Australian and New Zealand guidelines for freshwater quality. Monitoring Site B3 shows the highest mean value of NOx recorded at 1030 µg/ℓ. This is more than double the recommended guideline for freshwater ecosystems. The highest mean value of PO_4^{3-} recorded was 116 µg/ℓ observed at monitoring Site B3. The PO_4^{3-} maximum values for all monitoring sites show that all sites experienced the hypertrophic state at some point between 1987 to 2017. High phosphate levels are a major concern as phosphate is deemed to be a key determinant of eutrophication (Griffin, 2017).

The results show that all the nutrients are increasing when compared to a similar study by De Villiers (2007) in the Berg River using the same monitoring stations as the current study. During De Villiers' (2007) study, which investigated the nutrient status of the Berg River from 1985 – 2007, the long term mean value for NOx at monitoring site B3 was 829 µg/ℓ while the long term mean value for the same monitoring site (B3) in the current study is 1030 µg/ℓ. The long term mean value for NH_4^+ for monitoring site B3 during De Villiers (2007) was 58 µg/ℓ while in the current study it is 73 µg/ℓ. Furthermore De Villiers (2007) revealed that long term mean values at monitoring site B3 for TP was 271 µg/ℓ and for PO_4^{3-} the long term mean value was 91 µg/ℓ while the current study found that the long term mean values for the same monitoring site were 278 µg/ℓ for TP and PO_4^3 was 116 µg/ℓ. The comparison between the De Villiers (2007) studies and the current study show that all parameters have increased in all monitoring sites. The results also confirms that there was an increase in the nutrient load after the construction and operation of the Berg River Dam as predicted by De Villiers (2007).

4.3 Analysis of Nitrates and Nitrites

4.3.1 Trends in Nitrates and Nitrites

The lowest mean value of NOx recorded was 73 µg/ℓ at Site LB1. Site LB1 is in the upper most section of the river and there was minimal human disturbance or activity at this point. Figure 2 below shows a sharp increase of NOx at monitoring Site B3. The monitoring site is located in the middle section of the river and is located after two major towns, Paarl and Wellington. The peak increase in NOx of 1030 µg/ℓ at monitoring site B3 is probably due to anthrophogenic activities at both Paarl and Wellington. Paarl and Wellington are major towns with ever increasing population and the increase in NOx might be due to discharges from waste water treatment works or/and surface runoff from the townships. According to DWS (2017), there is an increase of informal settlements both in Paarl and Wellington and these informal settlements are not sewered and do not have formal drainage systems. The other likely reason associated with the eleveted NOx values at monitoring site at B3 might be agricultural runoff from the farms in the area. The predominate land use activity between B2, B3 and B4 is agriculture and elevated NOx levels could be attributed to the use of fertilizers in the area (DWAF, 1996). This section of the river is also the most populated area in the whole length of the river.

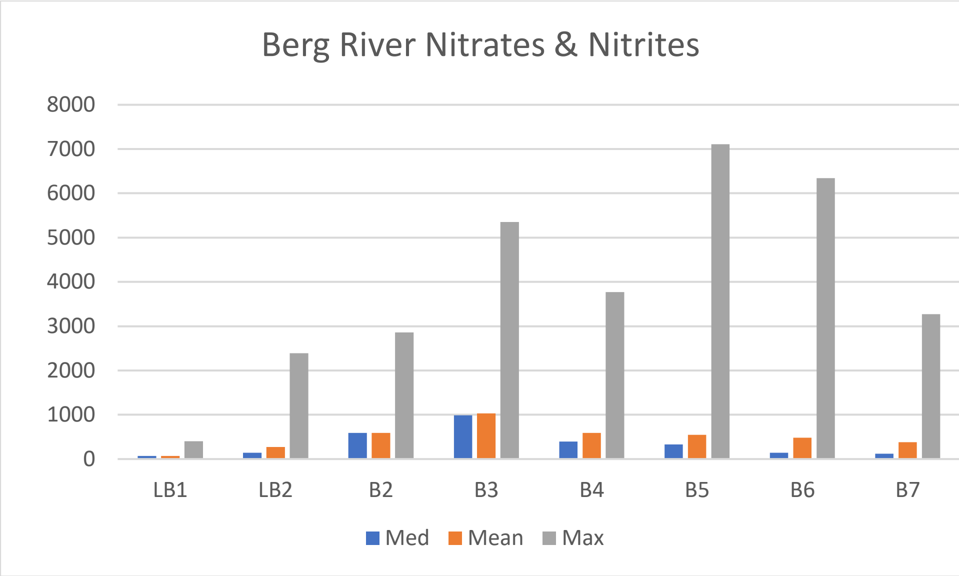

Figure 5 Mean, median and maximum values of NOx in the Berg River

4.3.2 Statistical analysis of Nitrates and Nitrites

Results for NOx for the monitoring site LB1 are shown below in Table 7 and Figure 6. Group 1 represent data from 1996 to 2006 and group 2 represent data from 2007 to 2017. Group 1(N=88, M=0.08, SD=0.05) and Group 2 (N=100, M= 0.06, SD =0.04) t=0.01 and p= 0.04. p is less than 0.05 therefore the null hypothesis is rejected and showed a significant difference between the two decades. Surprisingly the analysis showed a decrease in NOx load at monitoring site LB1 over the past decade (Figure 6), but there is no obvious reason to explain the decline.

Table 7: The results of t-test for NOx at Site LB1

T-test for Independent Samples (Sheet1 in LB1 NOX)
Note: Variables were treated as independent samples

Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
LB1_NOX 1ST vs. LB1_NOx 2ND	0.077723	0.064765	2.122853	186	0.035089	88	100	0.041679	0.041838	1.007647	0.974410

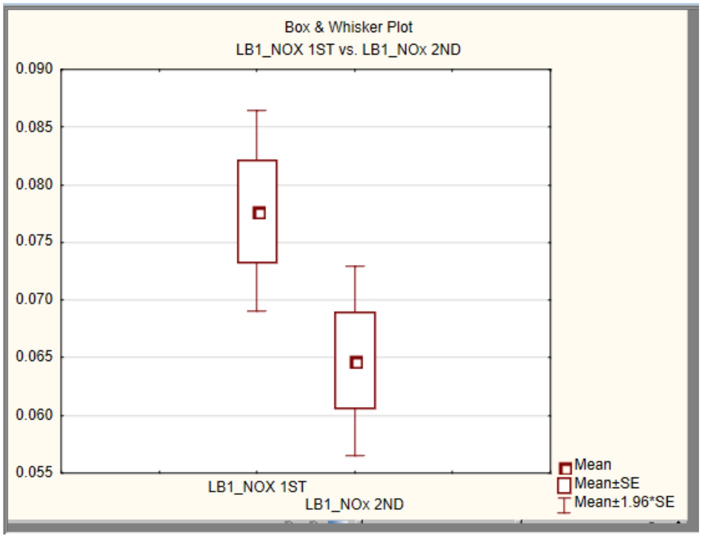

Figure 6 Whisker & Box graph at Site LB1

Table 8 and Figure 7 below show t-test results for Nitrate and Nitrite (NOx) for monitoring site B3. The first set, Group 1 represent data from 1996 to 2006 and the second Group 2 represent data from 2007 to 2017. Group 1(N=130, M=0.038, SD=0.026) and Group 2 (N=110, M=0.091, SD=0.23) t=-2.5 and p= 0.012. p is less than 0.05. The null hypothesis is rejected and there is a significant difference between the Group 1 and Group 2. The difference between the two means shows that there was a 30% increase in NOx in the past two decades at monitoring station B3.

Table 8: The t-test results for NOx at Site B3

	T-test for Independent Samples (B3 NOx in B3 NOx) Note: Variables were treated as independent samples										
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
B3_NOx 1ST vs. B3_NOX 2ND	1.088781	1.310031	-2.15393	229	0.032289	130	101	0.689954	0.871373	1.595030	0.012643

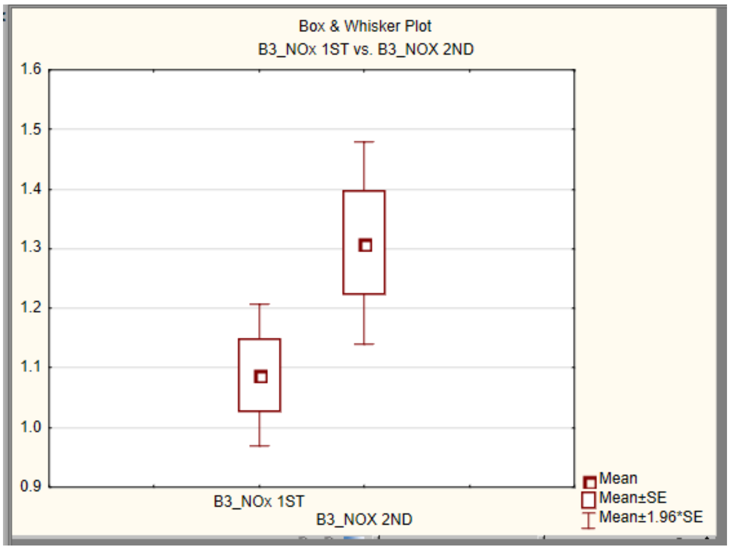

Figure 7 Box & Whisker graph for NOx at Site B3

A t-test results for Nitrate and Nitrate (NOx) for monitoring site B7 are shown below in Table 9 and Figure 8. The first set, Group 1 represent data from 1996 to 2006 and the second Group 2 represent data from 2007 to 2017. Group 1(N=254, M=0.379, SD=0.54) and Group 2 (N=254, M= 0.379, SD =0.54) t=0.01 and p= 1. p is greater than 0.05 and there is no significant difference between the two decades. The results demonstrate that the NOx load did not increase at monitoring station B7 for the past decade.

Table 9: The t-test results for NOx at Site B7

	T-test for Independent Samples (Sheet1 in B7 Kliphoek) Note: Variables were treated as independent samples										
Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	**p**	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
MonAv_NOx vs. MonAv_NOx	0.379970	0.379970	0.00	506	1.000000	254	254	0.543468	0.543468	1.000000	1.000000

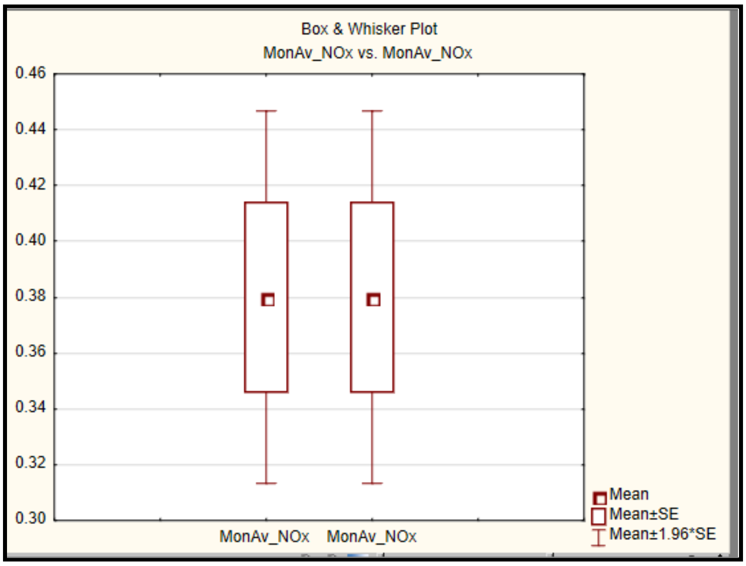

Figure 8: Box & Whisker graph for NOx at Site B7

4.4 Analysis of Ammonium

4.4.1 Trends in Ammonium

Ammonium values are relatively low when compared with other parameters along the Berg river. Figure 9 below shows the mean, median and maximum values of NH_4^+ along the Berg river. LB1 and LB2 which is the upper section of the river shows the lowest mean values(37 µg/ℓ) of NH_4^+ recorded during the study period. The highest mean value recorded is at monitoring site B3. From B3 the mean values are ≤50 µg/ℓ. Surprisingly a sharp increase was found at monitoring site B7. The maximum value of NH_4^+ recorded during the study period was 1900 µg/ℓ at monitoring site B2 followed by 1798 µg/ℓ at monitoring site B3. These two sites are near two major towns of Paarl and Wellington respective. The high values can be associated with anthrophogenic activites in around the two towns. The increasing concentration levels of NH_4^+ can be attributed to discharge from WWTP, industrial effluents and agricultural runoff. Ammonium is a nitrogenous fertilizer for aquatic plants and if in excess it can cause eutrophication and indirectly reduce the dissolved oxygen due to increased BOD.

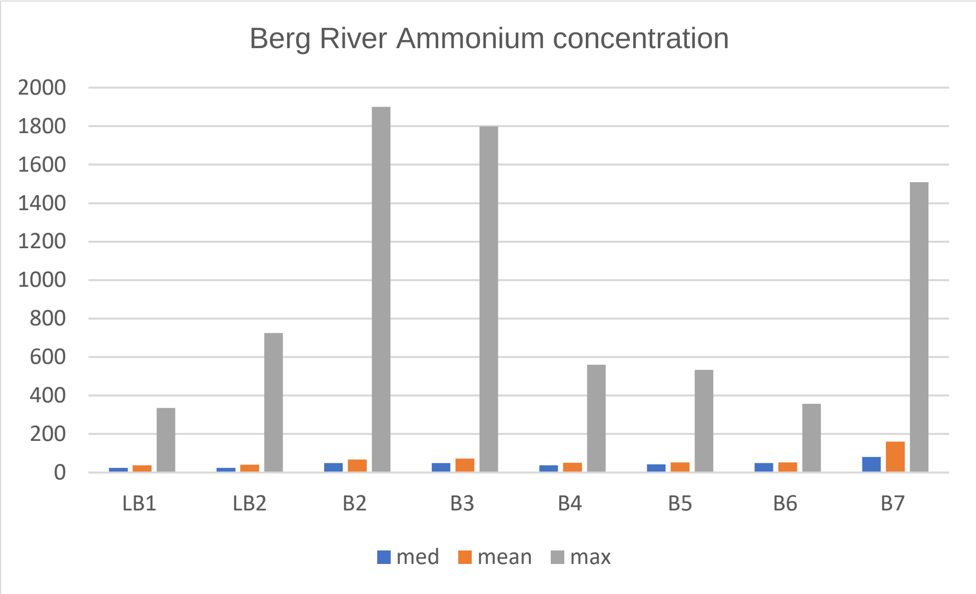

Figure 9 Mean median and maximum values of NH_4^+ in the Berg River

4.4.2 Statistical analysis of Ammonium

Table 10 and Figure 10 below present the t-test result for NH4+. Group 1 (N=88, M=0.03, *SD* =0.01) and Group 2 (N=105, M=0.05, *SD* 0.04) t=-5.10 and P=0.001. P is less than 0.05 therefore the null hypothesis is accepted and there is a significant difference between the two groups over the last two decades. The results confirm that there has been an increase in the amount of NH4+ in the Berg River at monitoring site LB1 in the last decade.

Table 10: The t-test results for NH_4^+ at Site LB1

	T-test for Independent Samples (Sheet1 in LB1 NH4) Note: Variables were treated as independent samples									
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
LB1_NH4 1ST vs. LB1_NH4 2ND	0.025985	0.049343	-5.10143	191	0.000001	88	105	0.014969	0.040693	7.389891	0.000000

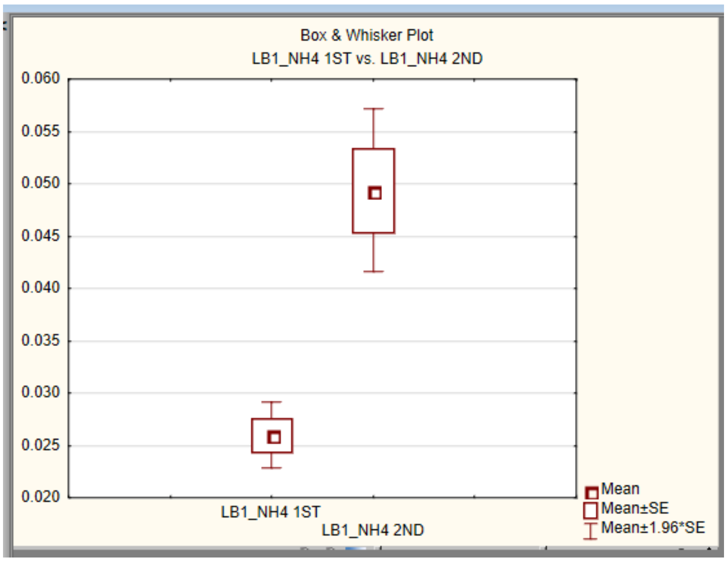

Figure 10: Box & Whisker graph for NH_4^+ at Site LB1

Results for NH_4^+ for monitoring Site B3 are shown below in Table 11 and figure 11. Group 1 represents the data set between 1996 to 2006 and Group 2 represents the data set from 2007 to 2017. Group 1 (N=130, M=0.038, SD=0.026) and Group 2 (N=110, M=0.091, SD=0.235) t=-2.5 and p=0.01. The p value is less than 0.05. The null hypothesis is rejected and there is a significant difference in between the two data sets. The difference between the two means shows a 200% increase of NH_4^+ in the past two decades.

Table 11: The t-test results for NH_4^+ at Site B3

T-test for Independent Samples (B3 NH4 in B3 NH4)
Note: Variables were treated as independent samples

Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
B3_NH4 1ST vs. B3_NH4 2ND	0.038426	0.090847	-2.51975	238	0.012400	130	110	0.026155	0.235582	81.12963	0.00

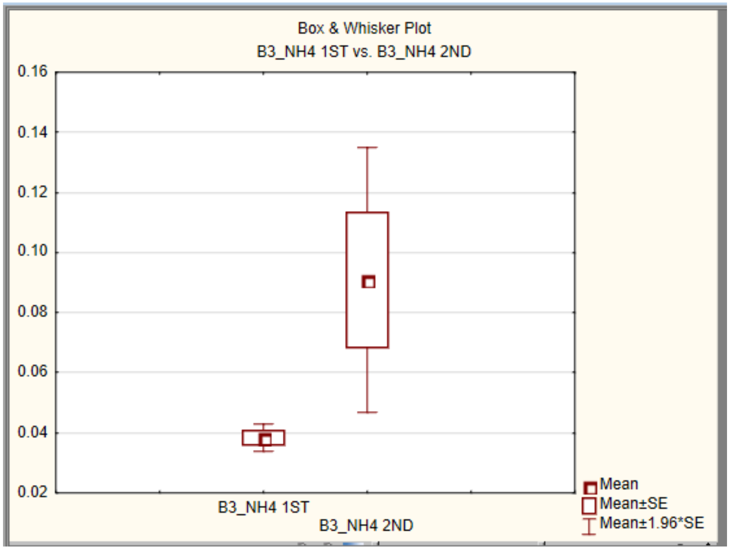

Figure 11: Box & Whisker graph for NH_4^+ at Site B3

A T-test result for NH_4^+ for monitoring site B7 are shown below in Table 12 and Figure 12. The first set, NH4 1st (group 1) represented data from 1996 to 2006 and the second set NH4 2nd (group 2) represent data from 2007 to 2017. Group 1(N=92, M=0.298, *SD* =0.093) and Group 2(N=92, M=0.298, *SD* 0.28) t=-7.65 and p=0.001. p is less than 0.05 therefore the null hypothesis is accepted and there is a significant difference between the two groups over the last two decades. The results confirm that there has been an increase in NH_4^+ in the Berg River at monitoring site B7 in the last decade.

Table 12: The t-test results for Site NH_4^+ at Site B7

T-test for Independent Samples (B7 NH4 in B7 NH4 t-test)
Note: Variables were treated as independent samples

Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
NH4_ 1ST vs. NH4_2ND	0.071835	0.298402	-7.64634	190	0.000000	100	92	0.093004	0.280051	9.067099	0.000000

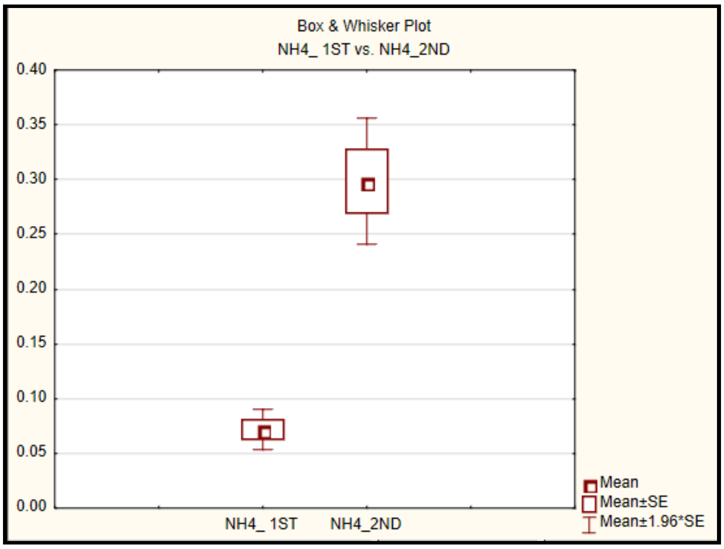

Figure 12: Box & Whisker graph for Site NH_4^+ at Site B7

4.5 Analysis of Total Phosphorus

4.5.1 Trends in Total Phosphorus

Monitoring data for TP was not available for all sites. Some of the monitoring sites were missing TP data and these sites are LB1, B6 and B7. The lowest mean value recorded during the study was 66 µg/ℓ found at site LB2. The low values of TP are due to less human activity in this area. The mean values increase at Site B2 with the highest mean value of 278 µg/ℓ recorded at monitoring site B3. After B3 the mean values decline by half with B5 recording 115 µg/ℓ and B6 recording 116 µg/ℓ. Surprisingly the maximum TP value recorded during the study period was at site LB2 which also recorded the lowest TP mean value. The common known sources of TP in fresh water systems in South Africa include agricultural effluent and sewage (De Villiers, 2007; Matthews, 2014; Griffin, 2017). According to Griffin (2017), a significant phosphate load is likely due to the use of detergents especially washing powders. In the case of the Berg River, the agricultural effluent is made up of stock-intensive operations and nutrient-enriched run-off from cultivated fields. TP is important because the levels are generally considered to be a more reliable indicator of emerging eutrophication.

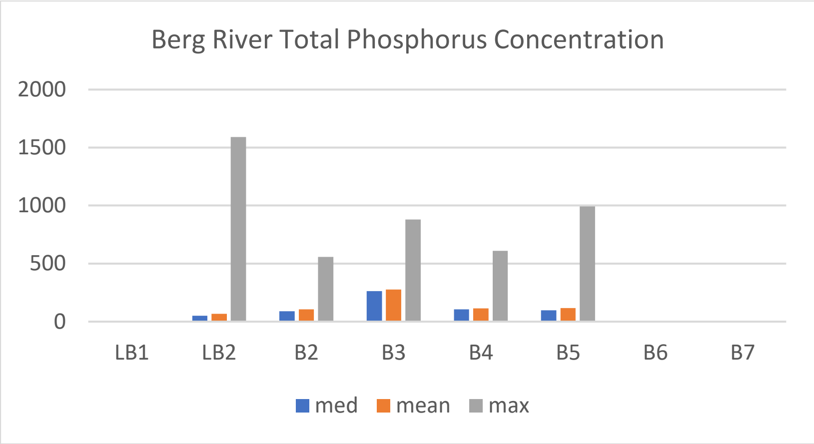

Figure 13 Mean, median and maximum values of TP in the Berg River

4.5.2 Statistical analysis of Total Phosphorus

As already mentioned, TP data for LB1, B6 and B7 were not available. The literature confirms the importance of TP as a strong indicator of eutrophication. The following sites were selected for TP statistical analyses: LB2, B3 and B5. The t-test results for monitoring site LB2 shown in Table 13 below confirms that there is a significant difference in the two data sets. Group 1 (N=84, M=0.07, SD=0.03) Group 2 (N=93, M= 0.07, SD=0.08) t = -0.07 and p= 0.43. Figure 14 confirms that there was an increase in TP load at monitoring site LB2 in the past decade.

Table 13: The t-test results for TP at Site LB2

	T-test for Independent Samples (Sheet1 in LB2 TP) Note: Variables were treated as independent samples									
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
LB2_TP 1ST vs. LB2_TP 2ND	0.065488	0.072444	-0.777760	175	0.437761	84	93	0.030710	0.076584	6.218906	0.000000

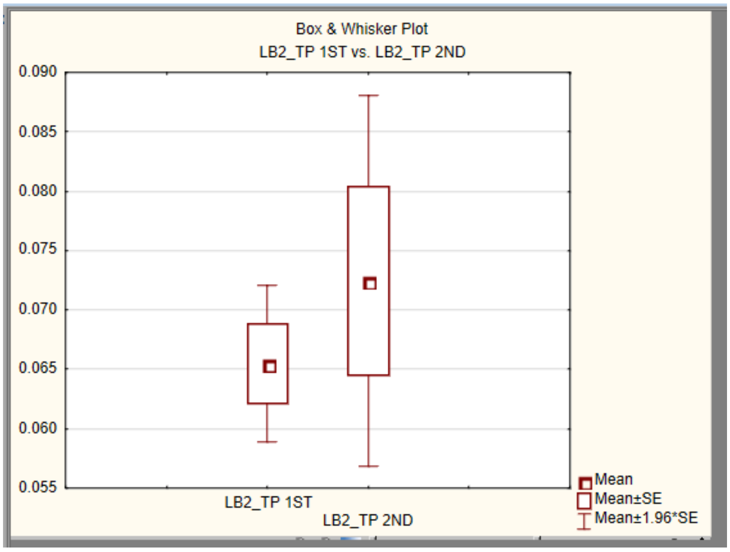

Figure 14: Box & Whisker graph for TP at Site LB2

A t-test was conducted for TP values at monitoring site B3 to determine if there is any significant difference between the two data sets over the past two decades. Group 1 represent data set from 1996-2006 and Group 2 represent 2006-2017. The results are shown in Table 14 and Figure 15 below. Group 1 (N=85, M=0.27, SD=0.10) and Group 2 (N=91, M= 0.29, SD=0.12) with t = -1.22 and p= 0.23. p value is > 0.05 therefore the null hypothesis is rejected and there is a significant difference between the two data sets. These results confirm that there has been an increase in TP load at monitoring station B3 as shown by the Box and Whisker graphs (Figure 15).

Table 14: The t-test results for TP at Site B3

T-test for Independent Samples (Sheet1 in B3 TP)
Note: Variables were treated as independent samples

Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
B3_TP 1ST vs. B3-TP 2ND	0.267025	0.287819	-1.21681	174	0.225324	85	91	0.103850	0.121431	1.367251	0.148242

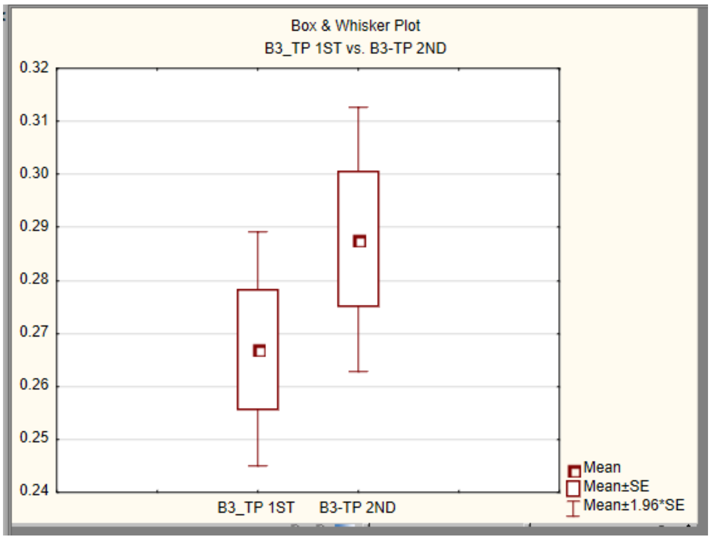

Figure 15 Box & Whisker graph for TP at Site B3

The t-test results for TP at site B5 are shown below in Table 15 which reveals that there was a significant difference between the two data sets in the last two decades. Group 1 (N=129, M= 0.12, *SD* =0.06) and Group 2 (N=90, M=0.12, *SD* = 0.06) t= -0.23 and p= 0.82. p value is greater than 0.05 and the null hypothesis is rejected, and thus confirms that there is a difference between the two data sets. Figure 16 below shows that there was a slight increase in TP concentration in the last decade.

Table 15: The t-test results for TP at Site B5

T-test for Independent Samples (Sheet1 in B5 TP)
Note: Variables were treated as independent samples

Group 1 vs. Group 2	Mean Group 1	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
B5_TP 1ST vs. B5_TP 2ND	0.121700	0.123613	-0.232245	217	0.816567	129	90	0.059321	0.060898	1.053869	0.779244

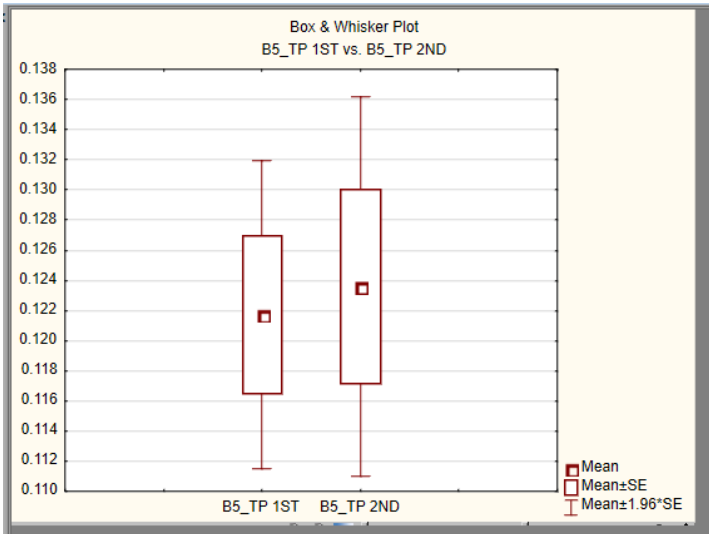

Figure 16: Box & Whisker graph for TP at Site B5

4.6 Analysis of Ortho-phosphates

4.6.1 Trends in Ortho-phosphates

The lowest mean value of PO_4^{3-} during the study was 17 µg/ℓ and was recorded at site LB1. This monitoring site is located at the upper section of the river where there is minimal disturbance due to anthropogenic activites. The highest mean value of PO_4^{3-} was recorded at B3 at Hermon. This site is downstream of the towns of Paarl and Wellington. The mean values show a sharp decrease after monitoring point B3 and there is a slight increase towards the ocean at monitoring station B7. The maximum value is recorded at monitoring site LB2 at 1090 µg/ℓ. As already mentioned the major sources of phosphate in fresh water systems in South Africa is due to sewage and agricultural effluent (Griffin, 2017). Another source of orthophosphate is observed in greywater runoff from informal settlements. Most informal settlements are unsewered and thus grey water from the households flows through the settlement only to be discharged into a nearbly fresh river systems.The site that shows the greatest increase in PO_4^{3-} is an area that is most populated and heavily cultivated.

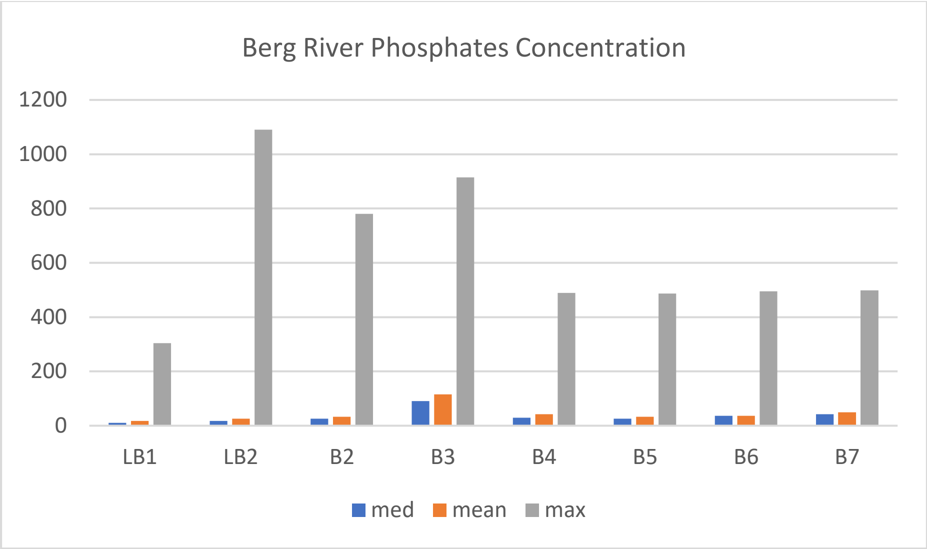

Figure 17: Mean, median and maximum values of PO_4^{3-} in the Berg River

4.6.2 Statistical analysis of Ortho-phosphates

A t-test was conducted for $PO4^{3-}$ at site LB1 to determine if there was any significant change in the past two decades. The results are shown below in Table 16 and Figure 18. The first set, Group 1 represent data from 1996 to 2006 and the second set, Group 2 represents data from 2007 to 2017. Group 1 (N=88, M= 0.02, *SD* =0.03) and Group 2 (N=105, M=0.01, *SD* = 0.01) t=3.15 and p= 0.01. p is less than 0.05 therefore the null hypothesis is rejected and there is a significant difference between the two groups. The Box & Whisker plot (Figure 18) shows that the amount of PO_4^{3-} reduced in the past decade.

Table 16: The t-test results for TP at Site LB1

	T-test for Independent Samples (Sheet1 in LB1 PO4) Note: Variables were treated as independent samples									
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
LB1_PO4 1ST vs. LB1_PO4 2ND	0.024621	0.012419	3.146692	191	0.001916	88	105	0.033509	0.019566	2.932930	0.000000

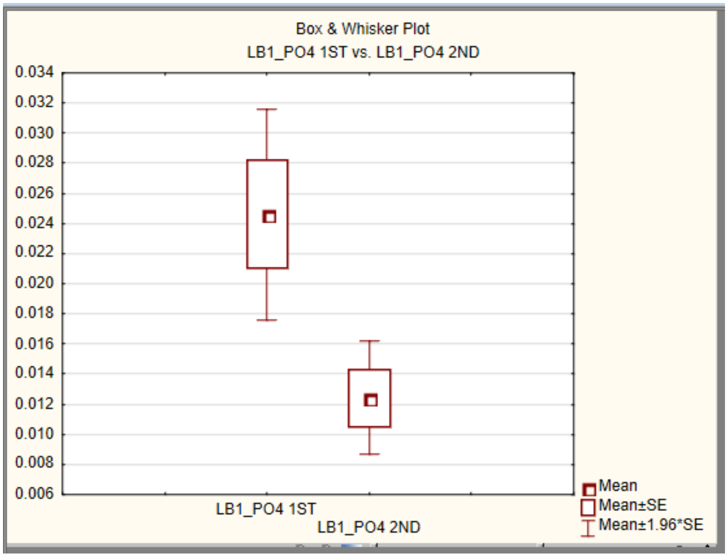

Figure 18: Box & Whisker graph for PO_4^{3-} at Site LB1

To test if there were any significant differences between the past two decades for the values of PO_4^{3-} at monitoring site B3, a t-test analyses was conducted, and the results are shown below in Table 17 and Figure 19. Group 1 (N= 130, M=0.14, SD=0.10) and Group 2 (N=110, M=0.15, SD=0.12) *t* =-0.71 and p=0.48. p value is greater than 0.05 and the null hypothesis is rejected indicating a significant difference between the two data sets. The Box and Whisker plot indicate that there was an increase in PO_4^{3-} in the past ten years.

Table 17: The t-test results for PO_4^{3-} at Site B3

	T-test for Independent Samples (Sheet1 in B3 PO4) Note: Variables were treated as independent samples										
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
B3_PO4 1ST vs. B3_PO4 2ND	0.136291	0.146583	-0.710801	238	0.477904	130	110	0.102435	0.121896	1.416072	0.057903

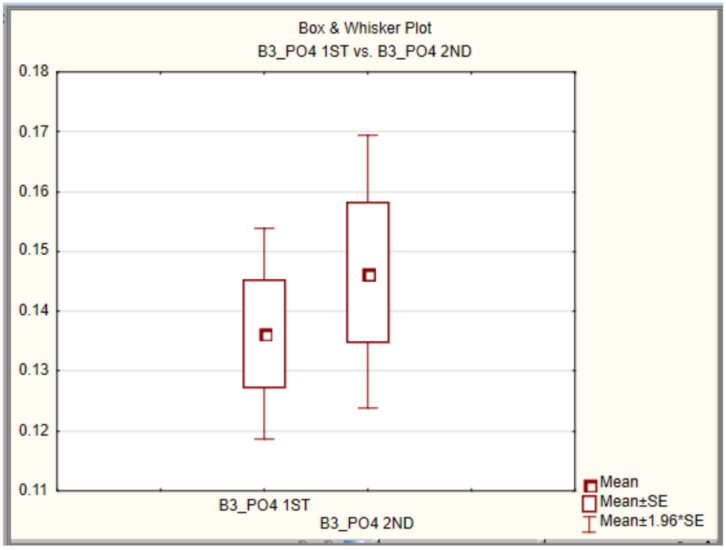

Figure 19: Box & Whisker graph for PO_4^{3-} at Site B3

The results are shown below in Table 18 and Figure 20. The first set, Group 1 represent data from 1996 to 2006 and the second set, Group 2 represent data from 2007 to 2017. Group 1 (N=100, M= 0.061, *SD* =0.038) and Group 2 (N=92, M=0.054, *SD* = 0.057) t=1.027 and p= 0.04. p is less than 0.005 and the null hypothesis was rejected. The results show that there is a significant difference between the two groups.

Table 18: The t-test results for PO_4^{3-} at Site B7

	T-test for Independent Samples (Sheet1 in B7 PO4) Note: Variables were treated as independent samples										
Group 1 vs. Group 2	**Mean Group 1**	Mean Group 2	t-value	df	p	Valid N Group 1	Valid N Group 2	Std.Dev. Group 1	Std.Dev. Group 2	F-ratio Variances	p Variances
PO4_1ST vs. PO4_2ND	0.061905	0.054755	1.027265	190	0.305602	100	92	0.038006	0.057225	2.267029	0.000079

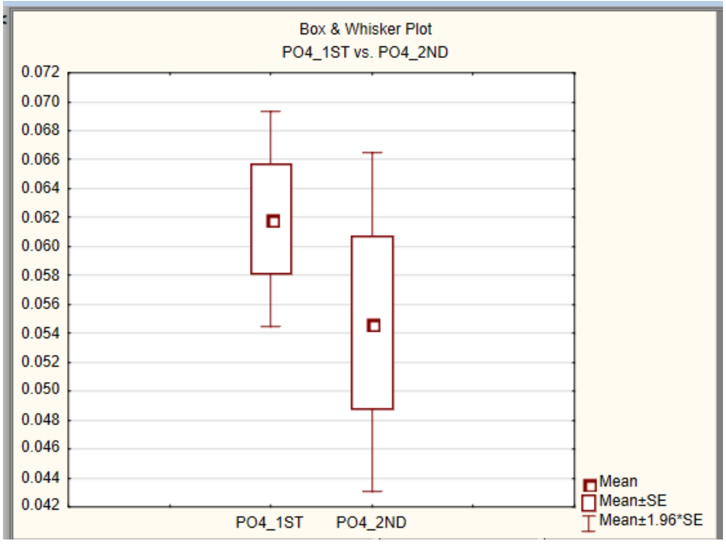

Figure 20: Box & Whisker graph for PO_4^{3-} at Site B7

4.7 Summary of Results

The study investigated the nutrient load in the Berg River by examining water quality data from 9 monitoring sites along the Berg River from 1976 to 2016 to determine downstream and long-term trends in nutrients as indicators of eutrophication conditions. The parameters were compared with both the South African and international water quality guidelines for fresh water systems. The study also determined which parameters were increasing and any significant differences between them over the last two decades (1996-2006 and 2007-2017). The results show a mixed trend but generally a stable trend in nutrient loadings in the upper sections of the river with a sharp increase in the middle section of the river followed by a slight decrease in the lower sections of the river. The t-test analysis revealed that there is a significant difference between the last two decades.

The trend for NOx shows that the nutrient load was stable in the upper section of the river at monitoring site LB1 with the lowest mean value recorded in this particular site. There is a sharp increase in NOx load at the middle section of the river particulary at monitoring site B3. The site is located down stream of the most populated areas in the Berg River catchment area. The population of the two towns had increased in the last decade (DWS 2017) and it is also the most cultivated region in the catchment area. Thus, the sharp increase in NOx at B3 could be associated with anthrophogenic activites such as the extensive use of fertilizers, dysfuntional WWTP and non-sewered settlements. The t-test results for NOx shows that concentrations had increased in all sites except for LB1 and B7.

NH_4^+ shows similar temporal trends to NOx. The upper sections of the river (LB1 and LB2) show relatively low mean values with an increase in the middle section of the river. The highest mean value recorded is at B3 in the middle section of the river. The mean values drop after monitoring site B3 but surprisingly increased again at B7. The expectation is that the mean value at B7 would decrease due to the natural assimilation of pollutants in the river. The t-test results confirms that there is significant differences between the past two decades in NH_4^+ concentration load in the Berg River. According to the Box & Whiskers graphs NH_4^+ concentration has been increasing in the past decade and all monitoring sites. The increase in concentration load of NH_4^+ is attributed to discharge from WWTP, industrial effluents and agricultural runoff (Griffin, 2017).

The trend for TP is similar to other nutrients. The lowest TP mean values are recorded at the upper section of the the river with the TP mean values increasing in the middle sections of the river with a sharp increase at monitoring site B3. The TP values decrease after B3 with the lower sections of the river experience mean values lower than the middle section of the river. The highest maximum value was recorded at LB2 which is a surprise as there is very little human activity in this area. The TP concentration load is attributed to agricultural effluent and sewage. The t-test results for TP reveals that there is a significant difference between the last two decades. The Box & Whisker Graphs shows that TP concentration load in the Berg river have been increasing in the past ten years. The trend in PO_4^{3-} concentration load is that the upper reaches of the river experience stable mean values while the mean values increase in the middle section of the river. The mean values decrease in the lower sections of the river. The highest peak in mean value is recorderd at monitoring site B3. The t-test results for PO_4^{3-} shows that there is differences between the last two decades. The Box & Whisker graphs shows that PO_4^{3-} has been reducing in the last ten years in the majority of the monitoring sites. However PO_4^{3-} had increased in the past decade at sites LB2, B3, B4 and B6 while it was decreasing at sites LB1 and B7.

CHAPTER 5: CONCLUSIONS AND RECOMMENDATIONS

Different studies in South Africa have alluded to an increasing trend in the level of nutrient concentration in freshwater ecosystems (De Villiers, 2007; Matthews & Bernard, 2015b; Griffin, 2017). Similar to the findings of De Villiers (2007), this study has established that there was an increase in the level of nutrients in the Berg River. Increasing nutrient levels in freshwater systems can have devastating and undesirable economic and ecological consequences that include loss of biodiversity, loss of asthetics and negative health impacts from poor water quality. Eutrophication can completely alter the ecosystems such that the goods and services derived from that ecosystem are limited or totally lost (Matthews & Bernard, 2015a; UNESCO, 2018).

The results from this study reveal a pattern showing a lower level of nutrient concentrations in the upper sections of the river, with an increase in the middle section of the river, and a slight decrease in the lower section towards the estuary. Lower concentrations in the upper section of the river can be attributed to sparse land use activities. By contrast, in the area in middle section of river there are higher population densities, urban infrastructure, industries and business activities that are likely causal factors including failed infrastructure, lack of maintenance of WWTP and agricultural runoff.

The results showed that most parts of the Berg River are consistently eutrophic throughout the year. Based on available TP data, all sites were eutrophic except for conditions at site B3 which was classified as permanently hypertrophic. Long-term mean values of TP exceed the recommended international guideline for ecosystems of 20 µg/ℓ. The long-term mean values of NOx also exceed the recommended guideline for ecosystems of 450 µg/ℓ and monitoring site B3 recorded the highest mean value of 1030 µg/ℓ which is double the recommended guideline for aquatic plant life. According to Griffin (2017), phosphate levels are meaningful determinants of eutrophication and PO_4^{3-} mean values from this study show that all sites were hypertrophic at some time during the period 1987 to 2017. The highest mean value PO_4^{3-} recorded during the study was 116 µg/ℓ and this was at monitoring station B3.

The t-test analysis showed that nutrient concentrations varied over the last decade. The study confirmed that nutrient load has increased in the last decade. The t-test results showed that the nutrient load was increasing in almost all the sites. Site B3 indicate a reason for concern as all parameters that were analysed showed an increase over the last decade. The increase at Site B3 can be attributted to population growth and its associated impacts in the towns of Paarl and Wellington. StatsSA (2011) shows that the population of Paarl increased by 35% in between the last two population census (i.e. 1996 to 2011). The lower section of the river, particularly monitoring Sites B6 and B7 also raised concern as there are periodic elevations in nutrient loading.

5.1 Recommendations

The increasing level of nutrients in the the Berg River is due to a variety of different factors at play along the river catchement. Based on this study and the the literature review, the most likely anthropogenic sources of nutrients in the Berg River are dysfunctional WWTP, non-sewered communities (mostly informal settlements) and agricultural runoff (De Villiers, 2007; van Ginkel, 2011). These variety of sources need to be managed on the land to avoid the further deteriorition of the Berg river. The recommendations below are suggested as part of an effort to remedy and revise the current situation in the Berg river.

- Investigate and identify specific sources of nutrients flowing into the the Berg River especially at Sites B3, B6 and B7.
- Increase the number of monitoring stations and continously monitor the level of nutrients.
- Develop an adaptive management strategy in order to reduce the nutrient load entering the Berg River. These management strategies should include good land use management practices along the river catchement.
- Construct and maintain wetlands in hotspot areas inorder to reduce the level of nutrients reaching the Berg river from in the land sources.
- Establish multi level public awareness strategies inorder to promote environmental stewardship along the Berg river.
- RQIS programme to include chlorophyll-a as one of its parameters and include TP data collection in those sites that TP data is currently not been collected. This is very important because chlorophyll-a and TP are key indicators of eutrophication.
- DWS to invest and upgrade the monitoring system to include low-cost, robust easy to operate WSNs in order to improve data collection within the Berg River.
- Little is known on the impact of climate change on eutrophication of freshwater system and thus, there is need to investigate and understand the impact of climate change on eutrophication.

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