A comparative assessment of water quality of two reservoirs in an ephemeral river in the desert climate of Namibia: Measures to control, and factors affecting growth of toxic cyanobacteria

By

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DECLARATION

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ABSTRACT

Globally, there is an increasing demand for freshwater of suitable quality. Satisfying the water requirement for ecosystem services and anthropogenic activities is challenging and may prove difficult in the context of climate change, growing population, and an increase in pollution from land use activities. With the rapid increase in human population in catchment areas, sustaining the associated human needs will often result in the generation of waste which will end up in water sources downstream causing eutrophication resulting in the proliferation of toxin-producing phytoplankton.

The overall aim of the research was to carry out a comparative assessment of the water quality deterioration of ephemeral river connected dams with desert climate conditions, found in the Omaruru-Swakop catchment associated with a mixture of land use activities. The Swakoppoort (SWP) and Von Bach (VB) dams as the major dams in the Omaruru-Swakop catchment found on the Swakop River were selected and assessed seasonally for a number of years. Water samples were analyzed for various constituents, firstly to assess the water quality status of the two dams receiving a mixture of pollutants over space and time using the combination Water Quality Index (WQI). Secondly, to describe the vertical and temporal dynamics of phytoplankton communities in the two dams. Thirdly to investigate the effect of prolonged drought on phytoplankton biomass measured as Chlorophyll *a* (Chl *a)* and cyanobacteria. Lastly, a case study to compare the effectiveness of two phytoplankton control measures employed in the SWP Dam was conducted.

The two dams were found to be impacted by nutrient, salinity, and particulate matter pollution. The WQIs showed poor water quality conditions in both dams for 17 years. Given the poor water quality of the two dams, *Microcystis* dominated the vertical and temporal dynamics, followed by *Dolichospermum*. In the dry seasons, higher cyanobacteria cell numbers were observed in comparison to the rainy season in both dams. In the SWP Dam, the preferable depth ranges for toxic cyanobacteria species were 5 to 10 m while in the VB Dam at 0 to 5 m range.

Higher frequencies of prolonged drought years were recorded in the VB Dam in comparison to SWP Dam. The influence of a decrease in vol % on the phytoplankton biomass was observed

in the SWP Dam but not in the VB Dam. However, the pattern and magnitude of the statistically significant responses (t-test, $p < 0.05$) varied among the drought and rainy years. Furthermore, the results showed that, the Solar Powered Circulation (SPC) and Phoslock® as the two phytoplankton control measures employed in the SWP Dam had no effect on cyanobacteria cells. It was evident that the two control measures were ineffective in reducing cyanobacterial cells. Incorporation of the current study research outputs in the utility surface water quality assessment and control program, water withdrawal for treatment and transfer plans, wastewater discharge regulations, and the revision of the integrated water resources management plan in Namibia, may ultimately safeguard the scarce water resources to improve resilience to climate change.

OPSOMMING

Wêreldwyd is daar 'n toenemende vraag na varswater van geskikte gehalte. Die bevrediging van die waterbehoefte vir ekosisteemdienste en antropogeniese aktiwiteite is uitdagend en kan moeilik wees in die konteks van klimaatsverandering, groeiende bevolking en 'n toename in besoedeling deur grondgebruik aktiwiteite. Met die vinnige toename in menslike bevolking in opvanggebiede, sal die instandhouding van die gepaardgaande menslike behoeftes dikwels lei tot die genereer van afval wat stroomaf in waterbronne beland, wat eutrofikasie veroorsaak en lei tot die verspreiding van fitoplankton wat toksiese stowwe produseer.

Die oorkoepelende doel van die navorsing was om 'n vergelykende assessering uit te voer oor die verswakking van die waterkwaliteit van kortstondige rivierverbonde damme met woestynklimaattoestande, wat gevind word in die Omaruru-Swakop-opvanggebied wat verband hou met 'n kombinasie van grondgebruik aktiwiteite. Die Swakoppoort (SWP) en Von Bach (VB) damme wat geag word as die belangrikste damme in die Omaruru-Swakop-opvanggebied en wat op die Swakoprivier gevind word, is geselekteer en vir 'n aantal jare seisoenaal waargeneem. Watermonsters is vir verskeie komponente ontleed, eerstens om die waterkwaliteitstatus van die twee damme, wat 'n mengsel van besoedelingstowwe oor ruimte en tyd ontvang, te bepaal deur die kombinasie Waterkwaliteit-indeks (WQI) te gebruik. Tweedens, om die vertikale en tydelike dinamika van fitoplankton gemeenskappe in die twee damme te beskryf. Derdens om die effek van langdurige droogte op fitoplankton biomassa, gemeet as Chlorofil *a* (Chl *a*), en sianobakterieë te ondersoek. Laastens is 'n gevallestudie uitgevoer om die doeltreffendheid van twee fitoplankton beheermaatreëls wat in die SWP-dam aangewend is, te vergelyk.

Daar is gevind dat die twee damme deur nutriënte, soutgehalte en besoedeling van organiese partikels geraak word. Die WQI's het vir 17 jaar swak watergehaltetoestande in beide damme getoon. Gegewe die swak waterkwaliteit van die twee damme, het *Microcystis* die vertikale en temporele dinamika oorheers, gevolg deur *Dolichospermum*. In die droë seisoene, is hoër sianobakterie-selgetalle waargeneem in vergelyking met die reënseisoen in beide damme. In die SWP-dam was die voorkeurdieptes vir toksiese sianobakterie-spesies 5 tot 10 m terwyl dit in die VB-dam van 0 tot 5 m was.

Hoër frekwensies van langdurige droogtejare is in die VB-dam aangeteken in vergelyking met SWP-dam. Die invloed van 'n afname in vol % op die fitoplankton biomassa is in die SWP Dam waargeneem, maar nie in die VB Dam nie. Die patroon en omvang van die statisties beduidende response (t-toets, $p < 0.05$) het egter tussen die droogte en reënjare gewissel. Verder het die resultate getoon dat die sonkrag-aangedrewe sirkulasie (SPC) en Phoslock® as die twee fitoplankton beheermaatreëls wat in die SWP-dam aangewend is, geen effek op sianobakterieselle gehad het nie. Dit was duidelik dat die twee beheermaatreëls ondoeltreffend was om sianobakteriese selle te verminder. Inkorporering van die huidige studienavorsingsuitsette deur gebruik te maak van die instrument in oppervlakwater kwaliteit assessering en -beheerprogram, wateronttrekking vir behandeling en oordragplanne, afvalwaterafvoerregulasies, en die hersiening van die geïntegreerde waterhulpbronbestuursplan in Namibië, kan uiteindelik die skaars waterbronne beskerm om weerstand teen klimaatsverandering te verbeter.

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PUBLICATIONS

Sirunda, J., Oberholster, P., & Wolfaardt, G. (2022). Assessing the Adverse Effects of Land Use Activities on the Water Quality of Selected Sub-Saharan Africa Reservoirs Using a Combination of Water Quality Indices. *Water Air Soil Pollut,* 233- 267. https://doi.org/10.1007/s11270-022-05703-9

Sirunda, J., Oberholster, P., Wolfaardt, G., Botes, M., & Truter, C. (2021a). Comparison of phytoplankton control measures in reducing cyanobacteria assemblage of reservoirs found in the arid region of Southern Africa. *Water Environment Research*, *93*(9), 1762–1778. https://doi.org/10.1002/wer.1564

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LIST OF ABBREVIATIONS

ANOVA:	Analysis of Variance
ANZECC:	Australian and New Zealand Environment and Conservation Council
APHA:	American Public Health Association
AWWA:	American Water Works Association
BWQI:	Bascaron Index
CCME :	Canadian Council of Ministers of the Environment
CETESB:	Company of the State of Sao Paulo
Cl:	Chloride
CPI:	Comprehensive Pollution Index
CSO:	Combined Sewer Overflows
DCM:	Deep Chlorophyll Maximum
DWA:	Department of Water Affairs
DWAF:	Department of Water Affairs and Forestry
EC:	Electrical Conductivity
EI:	Eutrophication Index
ELISA:	Enzyme-Linked Immunosorbent Assay
ENWC:	Eastern National Water Carrier
EPA:	Environmental Protection Agency
FIS:	Fuzzy Interface System
FSC:	Full Supply Capacity
HPLC:	High-Performance Liquid Chromatography
IFPRI:	International Food Policy Research Institute
IPCC:	Intergovernmental Panel on Climate Change
IWRM:	Integrated Water Resources Management
LSD:	Least Significant Difference
MWQI:	Malaysian Water Quality Index
NSF-WQI:	National Sanitation Foundation
OECD:	Organisation for Economic Cooperation and Development
OP:	Orthophosphate
OPI:	Organic Pollution Index
PCA:	Principal Component Analysis
PCR:	Polymerase Chain Reaction

PILD:	Modified Pollution Index
PPi:	Protein Phosphate Inhibition
SD:	Secchi Depths
SPC:	Solar Powered Circulation
SRDD:	Scottish Research development Department
TN:	Total Nitrogen
TN:TP:	Total Nitrogen and Total Phosphorus ratio
TP:	Total Phosphate
TPI:	Trace Metal Pollution Index
TWQR:	Target Water Quality Range
UN:	United Nations
UNEP:	United Nation Environmental Programme
USA:	United Sates of America
VBWTP:	Von Bach Water Treatment Plant
Vol %:	Water Volume
WHO:	World Health Organisation
WINGOC:	Windhoek Goreangab Operating Company
WJ-WQI:	West Java Water Quality Index
WPCF:	Water Pollution Control Federation
WQI:	Water Quality Index
WRM:	Water Resources Management
WT:	Water Temperature
WTP:	Water Treatment Plant
WWAP:	World Water Assessment Programme
WWTP:	Waste Water Treatment Plant
WWTP:	Waste Water Treatment Plant
Zeu:	The Euphotic Depth
Zmix:	The Mixing Depth

CHAPTER 1: GENERAL INTRODUCTION

1.1. Background

Clean water is a fundamental requirement for human life, dignity, and development (Bain et al., 2014; Chirenda et al., 2015; United Nations World Water Assessment, 2017; WWAP /UN-Water, 2018). Access to clean drinking water is Goal 6 of the Sustainable Development Agenda for 2030 (UN, 2020). The target of this goal is to achieve universal and equitable access to safe and affordable drinking water for all and improve its water quality by reducing pollution. However, the deterioration in water quality of surface water sources located downstream of urban and agricultural areas is on an increase due to anthropogenic activities exacerbated by climate change (IFPRI and Veolia, 2015; WWAP /UN-Water, 2018). Currently, about 12% of the world population drinks water from unsafe sources and more than 30% of the world population lives without any form of sanitation which contributes to water pollution (Boretti & Rosa, 2019).

Globally, about 80% of all industries and municipalities release wastewater without any prior treatment which ends up in water sources causing water quality deterioration (WWAP/UN-Water, 2018). Boretti & Rosa, (2019) states that about 90% of sewage in developing countries is discharged into the water sources untreated. On a yearly basis, around 730 million tons of sewage and other contaminants are discharged into the water worldwide (Boretti & Rosa, 2019). To manage these water sources requires testing of physiochemical and biological parameters and this is challenging over time and space to understand the changes in water quality status of surface water bodies (Uddin et al., 2021, Yang et al. 2021). Therefore, tools such as Water Quality Index (WQI) models are being used in different water bodies (Mishra et al. 2016, Oni & Fasakin 2016, Matta et al. 2017, Kumar et al. 2020, Son et al. 2020, Uddin et al. 2021). However, the applications in reservoirs found under desert climate conditions are lacking.

The discharge of sewage and other contaminants in water sources located downstream urban areas is causing eutrophication, which affects the quality of water by exacerbating the occurrence of toxic cyanobacteria blooms, which sometimes produces toxins harmful to public health. Globally, in many eutrophic lakes, toxic cyanobacteria blooms have been reported to dominate the phytoplankton community and this phenomenon is becoming an increasing concern to many water utility managers (Dalu & Wasserman, 2018; Oberholster et al., 2009).

The most dominant toxic species are *Cylindrospermopsis* (in tropical lakes and now also reported in temperate lakes), *Microcystis*, and *Anabaena* (*Dolichospermum)* (in subtropical and temperate lakes) (Ballot et al., 2014; Fastner et al., 2003; Sinha et al., 2012). Some strains of these species produce toxins, which pose a serious health risk to humans and animals (Hamilton et al., 2016; Oberholster et al., 2009; Svirčev et al., 2017). The dominance of the toxic cyanobacteria genera and toxin production are reported to be dynamic at vertical and temporal scales in many lakes (Dantas et al., 2012; Touati et al., 2019; Westrick et al., 2010). The vertical and temporal dynamics of cyanobacteria in the SWP and VB dams have not been established yet.

It has been reported that, an increase in water temperature, due to climate change, salinity, and nutrients from anthropogenic activities, have resulted in cyanobacteria blooms expansion over other phytoplankton species (Mantzouki et al., 2018; Paerl et al., 2008; Rigosi et al., 2014; Soares et al., 2009). With the projected change in climate and global warming, the occurrence of cyanobacteria blooms will increase across the globe, and will be even more severe on the African continent, especially in subtropical regions (Harke et al., 2016b; Ndlela et al., 2016; Quiblier et al., 2013; Rigosi et al., 2014) with a desert and semi-arid climate, like Namibia.

Some studies have reported the impact of climate change on cyanobacteria growth caused by warming and extreme events such as hydrological droughts (Brasil et al., 2016; Chapra et al., 2017; Delpla et al., 2009; Havens & Jeppesen, 2018; Lehman et al., 2019; Paerl, 2018; Paerl & Otten, 2013; Rigosi et al., 2014). Even though there are multiple studies in semi-arid regions on the impact of drought events on phytoplankton growth in reservoirs (Leite & Becker, 2019; Rocha Junior et al., 2018), nothing is yet reported in regions with a desert climate such as Namibia

The climate change events modify physico-chemical and biological variables which affect the quality of surface water by causing cyanobacteria blooms. This calls for implementation of phytoplankton control measures which ensure the prevention of the proliferation of nuisance bloom forming toxic phytoplankton species (Burford et al., 2019; Lürling & Mucci, 2020; Lürling & de Senerpont, 2016; Pęczuła, 2012; Visser et al., 2016). It is reported that the proactive reduction of external nutrients is more effective for the control of cyanobacterial blooms, compared to reactive control methods (Burford et al., 2019; Lürling et al., 2016). However, the ecological restoration of eutrophied lakes remains a great challenge with more failure than success, and some measures such as phytoplankton control is an alternative to face the degradation of man-made reservoirs. Phytoplankton blooms in warmer arid environments are reported to be a greater challenge to control as warmer drought conditions in itself create symptoms similar to eutrophication.

While there has been much research on toxic cyanobacteria blooms in numerous eutrophic lakes worldwide (Brasil et al., 2016; Dantas et al., 2011; Lehman et al., 2019), the water quality status, the cyanobacteria dynamics, impact of drought on cyanobacteria growth, and control measures of toxic cyanobacteria in dams situated in desert climate areas, for instance the SWP and VB dams, are not known or reported on. Therefore, understanding the water quality status, the dynamics, impacts of drought on growth, control measures of the toxic phytoplankton species in the SWP and VB dams is critical for water utility managers to ensure that good water quality is abstracted at the correct depths ranges during different seasons of the year for treatment with the existing conventional treatment processes. Findings from this study could therefore assist utility managers to inform decision making of operating the two dams connected by an ephemeral river, and the utilization of the treatment process to effectively treat the water from the two water sources.

Namibia is the driest country in Sub-Saharan Africa (Van Rensburg & Tortajada, 2021), situated between the Namib and Kalahari deserts. Temperature observations indicate that Namibia has experienced a considerable increase in temperature over recent years. In addition, the country has increasingly been experiencing erratic rainfall patterns, extreme droughts, and wildfires (Schachtschneider, 2010). Mean annual rainfall ranges from 25mm in the southern and coastal areas to 600mm in the Northeast (Kapuka & Hlásny, 2020). Only 2% of the rainfall ends up as surface runoff, 1% becomes available to recharge groundwater, 83% lost through evaporation, and 14% through evapotranspiration (IWRM Plan Joint Venture Namibia, 2010). Rainfall in Namibia is typically in the form of severe local showers, resulting in significant geographical and temporal rainfall variability both within and between years (Shikangalah, 2020). Recurrent drought is another element of the Namibia's dry climate that affects most of the Namibian territory, leading to livestock deaths, crop failures, poverty, and food insecurity due to lack of water (Olivieri et al., 2022). Because of Namibia's arid climate, all rivers in the interior are ephemeral and dammed. They typically flow briefly during the rainy season. The amount of water that can be withdrawn from these dams is very modest because of irregular flows and considerable evaporation losses. As a result, generally only half of the water in can be used. Together with groundwater sources, water from the dams play a significant role to ensure water security in the country, especially the central areas of Namibia. However, some of these dams' water quality is deteriorating as evidenced by of cyanobacteria blooms and occurrence of pathogenic microorganisms due to changes in land-use patterns caused by the increased population, urbanization, and poor wastewater infrastructure in catchment areas (Sirunda et al., 2022).

The SWP and VB dams are part of the three dams system of the Eastern National Water Carrier (ENWC) together with the Omatako Dam supplying water to the central area of Namibia (Van Rensburg & Tortajada, 2021). The VB Dam is the nearest to VB Water Treatment Plant, the capital city of Windhoek and Okahandja town. The Grootfontein-Omatako canal is also part of the ENWC, which transports water from the Karst Aquifer, including Kombat and Berg Aukus mine to Omatako Dam over a 263km distance (Schwartz & Ploethner, 2000). Water from Omatako Dam is transferred through pipelines as early as possible to the VB Dam to minimise evaporation losses (Scott et al., 2018; Slabbert, 2007). The SWP Dam water is pumped straight into either VB Dam or the VB Treatment Plant (Sirunda et al., 2021a) (Figure 1.1). These dams are used in an integrated manner to ensure supply to central Namibia (Sirunda et al., 2021b). However due to the increased water demand of the central areas, the three dam system are been augmented with groundwater from the Karst Aquifer (Figure 1.1) with plans for additional augmentation from the Okavango River, north east of the country (Lewis et al., 2019). When dams are operated in an integrated manner, they deliver a 95% assurance of supply of 20Mm^3 per annual and the water is mainly supplied to the capital city of Windhoek, Okahandja Town, Karibib Town, Otjimbingwe Village, and Navahacb Mine (Scott et al., 2018; Slabbert, 2007). The SWP Dam, built in 1977, with a maximum depth of 30m, capacity of 63.5Mm^3 and VB Dam, built in 1970, with a maximum depth of 29m, and capacity of 48.96Mm^3 all are situated on the Swakop River, which is an ephemeral river (Figure 1.1).

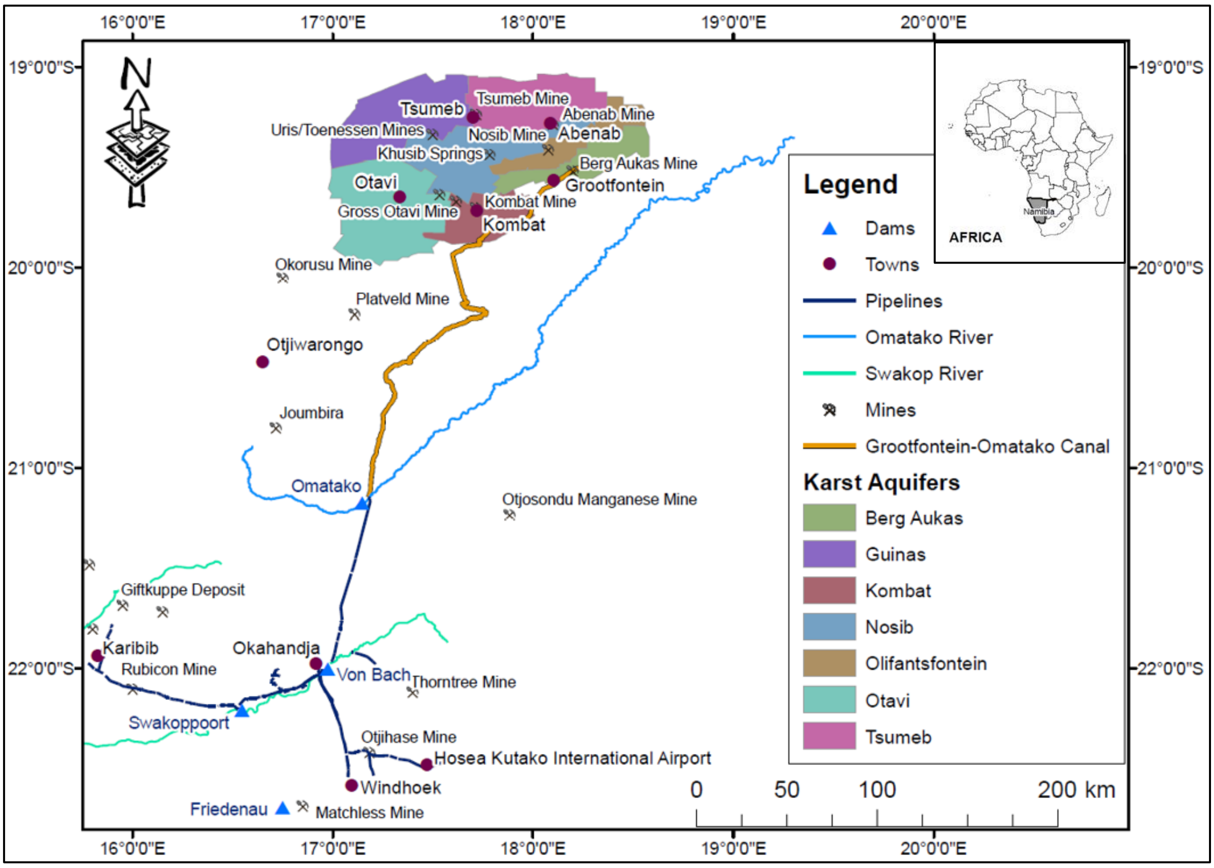

Figure 1.1. Map of the study area indicating the location of Namibia in Africa, and the linkages of the two dams being studied with other water sources in central Namibia.

The two dams of the study area are located in the Omaruru-Swakop catchment area on the Swakop River found (IWRM Plan Joint Venture Namibia, 2010). The dams were constructed to store water in a region characterized by prolonged droughts, and limited and erratic rainfall (Shikangalah, 2020). The land use activities in the upper catchment of the VB Dam comprises of commercial farms for livestock and game farming, settlements, mines, and lodges (Sirunda et al., 2022) (Figure 1.1). The diverse land use activities in the upper catchment of the SWP Dam include informal settlements (and resulting untreated sewage), a wastewater treatment plant, sewerage ponds and reclamation plant, a landfill site, beef feedlots, dairies and poultry farms, a tannery, breweries, lodges, mines, and a coal power plant (Sirunda et al., 2022) (Figure 1.1 and 1.2).

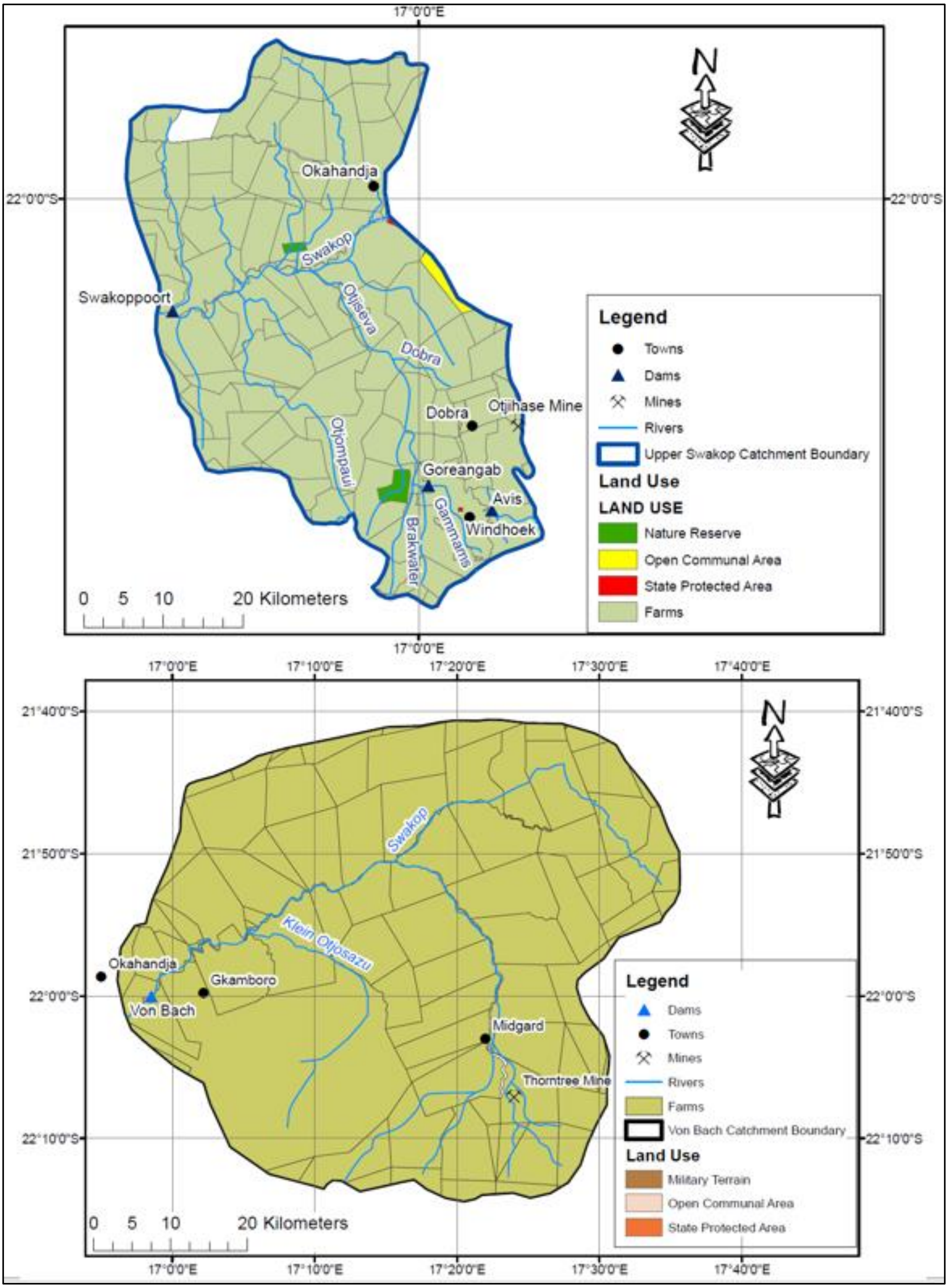

Figure 1.2. The sub-catchment area of the two dams with the associated land use activities: (a) SWP Dam sub-catchment and (b) the VB Dam sub-catchment.

1.2. Hypothesis, research aim and objectives

The central hypothesis of the research was that the quality of water sources found in desert climate, and downstream from an urban settlement, will deteriorate at different rates, depending on the mixture of organic and inorganic pollutants received. To test the hypothesis, the research was designed around an overall aim of carrying out a comparative assessment of the water quality status of the two dams connected by an ephemeral river, in relation to dynamics, impact of drought on growth, and control measures of toxic phytoplankton species in the source water for the VB Treatment Plant (VBWTP), namely VB(21° 59'59.27'' S 16° 58'54.76" E) and SWP (22° 12'44.31" S 16° 31'44.97" E) dams in central Namibia. The specific objectives were to:

i. Review the water quality status, toxic phytoplankton dynamics, in terms of occurrence and species diversity, factors affecting growth, and control measures of cyanobacteria of the central dams of Namibia – *Chapter 2;*
ii. Assess the adverse effects of land use activities on the water quality of selected Sub-Saharan Africa reservoirs using a combination of water quality indices - *Chapter 3;*
iii. Assess the phytoplankton dynamics in the two reservoirs with special reference to water abstraction for inter-basin transfers and potable water production – *Chapter 4;*
iv. Perform a long-term study on the impact of drought on the phytoplankton in two water supply reservoirs in the Sub-Saharan Africa – *Chapter 5*; and
v. Compare phytoplankton control measures in reducing cyanobacteria assemblage of reservoirs found in the arid region of Southern Africa – *Chapter 6*

CHAPTER 2: LITERATURE REVIEW: WATER QUALITY STATUS ASSESSMENT, TOXIC PHYTOPLANKTON DYNAMICS IN TERMS OF OCCURRENCE AND SPECIES DIVERSITY, FACTORS AFFECTING GROWTH, AND CONTROL MEASURES OF CYANOBACTERIA OF THE CENTRAL DAMS OF NAMIBIA

2.1. Introduction

Globally, the quality of water resources located downstream of urban and agricultural areas have undergone degradation due to natural processes and anthropogenic activities (Espejo et al., 2012; Son et al., 2020; Vadde et al., 2018). Anthropogenic activities such as mining, livestock farming, production, and disposal of waste (industrial, municipal, agricultural), increase in sediment runoff or soil erosion, and heavy metal pollution, are being reported to affect water quality in reservoirs (Effendi, 2016; Jian Guo Jiang & Shen, 2003; Tanjung et al., 2019; Uddin et al., 2021), potentially making water unsuitable for use (Castro-Roa et al., 2014; Dabrowski et al., 2013, 2014; Oberholster et al., 2021; Tanjung et al., 2019; Uddin et al., 2021). It is reported that many developing countries are challenged in protecting the water quality of reservoirs located downstream of urban and agricultural areas (Uddin et al., 2021). This is mainly due to malfunction of wastewater infrastructure, poor/lack of sanitation facilities in informal settlements, lack of knowledge of water pollution, lack of enforcement, financial constraint, and lack of compliance to discharge of untreated sewage and mining effluent into river tributaries emptying into reservoirs. Pollutants increase the concentration of physicochemical parameters to exceed the required standard guideline limit (Tanjung et al., 2019; Tirkey et al., 2013; Yang et al., 2021) and in turn influence the concentration or growth of aquatic microbiota. The main consequence of pollutants emanating from anthropogenic activities in catchment areas is reported to be eutrophication caused by excessive nutrient loads (Hamilton et al., 2016; Paerl et al., 2016; Paerl & Otten, 2013; Xu et al., 2015).

Eutrophication is defined as the nutrient enrichment of water bodies caused by human and natural activities, which affect the water quality. Vollenweider (1968) defined eutrophication as “an enrichment of nutrients and the progressive deterioration of water quality, particularly for lakes, due to excessive algal growth, and all its effects on the metabolism of the affected water”. The Organisation for Economic Cooperation and Development (OECD) (1982) defined eutrophication as “the enrichment of nutrients in a water body which causes a stimulation of large number of symptomatic changes among which are an augmentation of the production of cyanobacteria, the deterioration of the water quality and other changes which are negative and interfere with the usage of the affected water".

Eutrophication is a global problem (Chakrabarti, 2018). In sub-Saharan Africa, the ecological integrity of surface water has been compromised, as toxic cyanobacteria blooms are becoming abundant and dissolved oxygen levels are reduced in surface water bodies due to excessive nutrients loadings (Davies and Koop 2006; Nyenje et al., 2010). The water quality of surface water (i.e. lakes, reservoirs, rivers, oceans and wetlands) depends upon the quantity and quality of inflows they receive from their respective catchments (Xu et al., 2001). The main sources of nutrients affecting surface water bodies have been reported to be caused by increase in urbanisation, combined with poor town planning, which leads to discharges of wastewater from sewage, agriculture, urban, informal settlements, and industrial areas (Nyenje et al., 2014; Huang et al., 2014). The major sources of nutrients are classified as point and non-point sources (Luo et al., 2022). These sources of nutrients are mainly from land uses activities in catchment areas (Liu et al., 2011; Luo et al., 2022). In rural watersheds, non-point nutrient loading from agriculture directly affect lake water quality (Luo et al., 2022). Non-point sources, such as agriculture practices, contaminated sediments, urban runoff, and atmospheric deposition have been reported as the main cause of water quality deterioration in lakes and rivers (Luo et al., 2022). Point sources are municipal and industrial wastewater treatment plants (WWTP), combined sewer overflows (CSOs), and sanitary sewer overflows (Luo et al., 2022).

This chapter provides an overview of surface water quality deterioration due to the occurrence of cyanobacteria blooms caused by anthropogenic activities across the world, Africa and in particular, Namibia. Furthermore, this chapter reviews the characteristics of cyanobacteria, the vertical and temporal dynamics, factors controlling growths, and the control and management of cyanobacteria.

2.2. Water quality assessment

Water quality status assessment over time and space is challenging as it requires testing of physicochemical and biological parameters. However, to effectively understand the change in water quality status of surface water bodies, environmental tools such as WQI models are been used (Castro-Roa & Pinilla-Agudelo, 2014; Finotti et al., 2015; Jiang & Shen, 2006; Kumar et al., 2020; Matta et al., 2017; Mishra et al., 2016; Oberholster et al., 2021; Oni & Fasakin, 2016;

Son et al., 2020; Uddin et al., 2021). Globally, the most used WQIs in assessing the water quality status of surface water bodies are the Horton index, National Sanitation Foundation WQI (NSF-WQI), Scottish Research development Department (SRDD) Index, Canadian Council of Ministers of the Environment (CCME) WQI, Bascaron index (BWQI), Fuzzy Interface System (FIS), Malaysian WQI (MWQI), and West Java WQI (WJ-WQI) (Kachroud et al., 2019; Tirkey et al., 2013; Uddin et al., 2021), Eutrophication Index (EI), Trace Metal Pollution Index (TPI), Organic Pollution Index (OPI), Comprehensive Pollution Index (CPI) (Son et al., 2020), and the National Sanitation Foundation Environmental Sanitation Technology Company of the State of São Paulo (Cetesb) WQI (Finotti et al., 2015).

The CCME WQI is widely used, either as a standalone or with combination of other WQIs, like CPI. The CCME was developed by the Canadian Council of Ministers of the Environment (CCME) in 2001 (Bilgin, 2018) and its applications worldwide has been due to it applicability to different environmental conditions. The CCME enabales users from different countries to use site-specific standards for water quality parameters (Haider et al., 2021; Haile & Gabbiye, 2022; Khan et al., 2005). The CCME WQI combines all water quality parameters and provides a readily understood description of the water quality (Davies, 2006). The CCME WQI was developed based on the combination of factors, namely: scope, frequency, and amplitude of the water quality variables, all into one index (Bilgin, 2018; Davies, 2006). Adelagun, et al., (2021) mentioned that, over years many counties have accepted the use of the CCME as a WQI for surface and groundwater quality monitoring and assessment.

The comprehensive Pollution Index (CPI) is also widely used to assess the level of pollution of rivers, lakes, wetland, and groundwater (Son et al., 2020). For instance, Son et al. (2020) applied a combination of indices (WQI, CPI, Organic OPI, EI, and TPI) to assess the water quality of the Cau River, in Vietnam and reported serious organic pollution and eutrophic conditions downstream. Tang et al. (2011) applied the comprehensive pollution index to assess the water quality status of Qilu Lake, China. Matta et al. (2017) used the CPI to assess the water quality status of the Henwal River, India and reported moderate and severe pollution at different sampling sites. Mishra et al. (2016) applied a combination of indices (CPI, OPI, EI and TPI) to assess the water quality of the Surha Lake, in India, and reported that the lake is eutrophic and moderately polluted. Recently, Oberholster et al. (2021) used a modified pollution index (PILD) to determine the water quality status of the Loskop Dam, South Africa, and reported nutrient enrichment and heavy metal pollution. Catchment land use activities such as agriculture, dysfunctional wastewater infrastructure, mining etc., are being reported as the root causes of water quality deterioration in this dam. The application of these tools in reservoirs found in desert climates could be helpful for the water utility manager to monitor the changes in water quality for effective treatment and management.

2.3.Occurrence and species diversity of cyanobacteria

2.3.1. Cyanobacteria occurrence and species diversity a global perspective

The occurrence of cyanobacteria blooms, and their associated toxins, are a major concern to utility managers and plant operators (Harke et al., 2016a; Preece et al., 2017). At a global level, investigation of cyanobacteria blooms in freshwater ecosystems is increasing at a faster rate than those of brackish, estuarine, and marine environments (Preece et al., 2017). A review by Harke et al. (2016) reported that around 108 countries have records of cyanobacteria blooms, 79 records of which were associated with the potent hepatotoxin microcystin. Cyanobacteria is common and grows in the water column as planktonic single cells, aggregates on the water surface as metaphytic, attached to the algae, or macrophytes as epiphytic, or attached to the bottom substrate as benthic matter (Quiblier et al., 2013). Benthic cyanobacteria are not as widely studied as planktonic species at a global level, even though they do occur in freshwater systems and are known to produce toxins. Therefore, there is still a knowledge gap in understanding the species composition, toxin production, and distribution of benthic cyanobacteria (Quiblier et al., 2013). The below review will focus on cyanobacteria occurrence and species diversity at a global level.

Europe

In Europe, cyanobacteria blooms have been reported in coastal waters of Netherlands, Portugal and Spain, and in the Baltic Sea (Preece et al., 2017). The Baltic sea has recorded cyanobacteria blooms, which are on an increase occurring every summer due to high levels of nutrients and reduction in salinity (Preece et al., 2017), and dominated by *Nodularia*, *Spumigena*, *Aphanizomenon*, and *Dolichospermum* species (Preece et al., 2017). Furthetmore, in the lower salinity areas of the Baltic Sea, *Microcystis* species, and associated microcystin toxins are also found (Brutemark et al., 2015), at concentrations that may be a threat to human health (Preece et al., 2017). In Fehmarn Island, blooms of *Nodularia* spp. and associated microcystin toxins are mostly reported (Preece et al., 2017). Toxins of *Microcystis* were also recorded in the Curonian lagoon during 2006-2007, and also the Vistula Lagoon, Poland (Mazur-marzec, 2010). In addition, cyanobacteria blooms have been reported in the Guadiana Estuary, between Portugal and Spain (Rocha et al., 2002). In Lake Occhito of Italy, cyanobacteria blooms *of Planktonthrix rubescens* were reported in 2009, and after heavy rain, this water was discharged into River Fortore, which flows into the Adriatic Sea (De Pace et al., 2014). The occurrence of cyanobacterial blooms were, subsequently, reported in these areas, and were mainly attributed to the availability of nutrients. Various species of cyanobacteria were recorded and the associated microcystin toxins were detected.

In Turkey, blooms of *Microcystis* species were reported in the Horn Estuary and the Küçükçekmece Lagoon, with microcystin toxins that have been reported at a concentration harmful to human health (Preece et al., 2017). The high salinity concentration of the estuarine water was found to be associated with the release of dissolved microcystin toxins. The root causes of the cyanobacteria blooms were associated with nutrient loading and reduction in salinity, due to dilution caused by heavy rainfall during the rainy season (Preece et al., 2017). In the Baltic sea, *Microcystis* and associated microcystin toxins favour a lower salinity environment, and an increase in salinity concentration results in a stress factor and, thus, a release in microcystin toxins.

Mid-Atlantic Region, United States and Canada

Cyanobacteria blooms are on the upward trend in Chesapeake Bay, an estuary in the Mid-Atlantic region of the United States (Preece et al., 2017). Two rivers in the same region, the Sassafras and Potomac have reported cyanobacteria blooms (Preece et al., 2017). The dominant species in the above areas is mainly *Microcystis aeruginosa*. Cyanobacteria blooms have also been reported in the Neuse River (North Carolina), and the dominant species is mainly *Microcystis aeruginosa* and *Oscillatoria* sp. The main root causes of the occurrence of the above cyanobacteria blooms are reported to be nutrients from anthropogenic activities in the catchment area (Preece et al., 2017).

In Canada, investigation of the effect of toxins from cyanobacteria started through animal, livestock, wildlife poisoning in the 1950s (Kotak & Zurawell, 2007). Subsequently, several algal blooms have been reported, and appear to be mainly due to eutrophication and other factors such as climate change, food web alteration in invasive species and overfishing (Pick, 2016)., Microcystin producing cyanobacteria were reported in Lake Champlain's Missisquoic Bay in 2006 and 2007, with microcystin concentrations exceeding the Canadian drinking water guideline of 1.5µg/l during that time (Fortin et al., 2010). Between 1990 to 1995, around 96% of 133 samples collected from Driedmeat Lake were found to contain microcystin-LR, while 94% of the 104 samples collected from Little Beaver Lake in the same period also contained microcystin-LR as measured with HPLC (Kotak & Zurawell, 2007). Microcystin-LR, produced by *Microcystis*, is more prevalent in the Canadian water compared to anatoxin-a (also known as Very Fast Death Factor, which is a cyanotoxin causing acute neurotoxicity, and saxitoxin (responsible for red tide) (Kotak & Zurawell, 2007). The concentrations of these toxins are highly variable in Canadian water, both temporally and spatially.

South America

In South America, cyanobacteria blooms have been reported in Argentina, Columbia, Uruguay and Brazil. Most of the cyanobacteria blooms have been reported in the Brazilian lagoons and estuaries (Preece et al., 2017). In these estuaries, microcystins, at levels of up to 14.8ug/l has been reported, mainly in high salinity water (Preece et al., 2017). The dominant species in these blooms are found to be *Microcystis aeruginosa*, and other microcystin-producing toxins (Preece et al., 2017). The root causes of the occurrence of the above cyanobacteria blooms are reported to be due to the transported *Microcystis aeruginosa* from freshwater systems into the lagoons and estuary, along with the reduction in salinity of the water.

Apart from the occurrence of cyanobacteria blooms in the estuaries and lagoons, cyanobacteria blooms have also been reported in freshwater reservoirs. In Brazil, the Funil Reservoir, found in the southeastern part of the country, reported cyanobacteria blooms of *Microcystis aeruginosa* (Soares et al., 2009). The root cause was due to nutrient enrichment. Cyanobacteria comprised 97% of the algal species composition. Despite the dominance by cyanobacteria, a seasonal succession between cyanobacteria species of nitrogen-fixing and non-nitrogen fixing was observed (Soares et al., 2009). In a tropical lake of Lagoa Santa, Brazil, cyanobacterial blooms of *Cylindrospesmopsis raciborskii* were reported, and the root cause was associated with the increase in water temperature within the optimal growth range of *Cylindrospesmopsis raciborskii* (Figueredo & Giani, 2009).

In the Ingazeira Reservoir, North-eastern Brazilia, cyanobacterial blooms of *Cylindrospesmopsis raciborskii* were also reported, and an increase in water temperature was found to be the contributing environmental factor to the bloom (Bouvy et al., 1999). Similar findings were recorded in the Arcoverde Reservoir, found in north-eastern Brazilia, where the cyanobacteria blooms of *Cylindrospesmopsis. Raciborskii* were also dominant in both surface and bottom water. The locality of the reservoirs in a tropical climate, coupled with nutrient enrichment and climate change, could be the contributing factor for the *Cylindrospesmopsis raciborskii* dominancy in the above areas.

Australia and New Zealand

Australia is a semi-arid country with scarce water resources. Cyanobacteria blooms were reported after heavy rains, which transports the *Microcystis aeruginosa* into the Swan Estuary through the Swan River (Preece et al., 2017). As observed in many estuaries, the blooms were accelerated by the reduction in the water salinity during the rainy season as a result of dilution (Preece et al., 2017).

In the Murray Darling River, Australia, algal blooms of cyanobacteria have been reported since 2000 and are increasing in frequency (Biswas, 2017). For example, in the Lower Darling River, blooms of saxitoxins producing cyanobacteria genera have been reported (Biswas, 2017). In the Lower Darling River, *Dolichospermum circinalis* was found to be the dominant species followed by *Aphanizomenon*, *Planktonlyngbya*, and *Mersopodia* sp. (Biswas, 2017). Mitrovic et al. (2011) also reported the occurrence of cyanobacteria blooms in the Menindee Lake, one of the Murray Darling tributaries. In their study, flow release from the regulated Menindee Lake was investigated in suppressing bloom development and translocation of cyanobacteria cells (Mitrovic et al., 2011). The regulated flow was found to be effective in reducing the cyanobacterial cells from 100 000 to <1000cells/ml (Mitrovic et al., 2011). In the Murray Darling River, cyanobacteria genera such as *Dolichospermum circinalis, Cylindrospesmopsis raciborkii*, and *Microcystis flos-aquae* have been detected and the instream lakes such as Hume, Mulwala, Dartmouth, and Victoria were reported to be the areas of bloom formation (Biswas, 2017). These studies demonstrated the need to control the occurrence of cyanobacteria blooms with strains of toxic *Microcystis* to ensure the sustainable supply of water of good quality to consumers.

Unlike Australia, New Zealand largely has a temperate climate. Lake Rotoiti and Lake Okareka, found in the northern parts of the country, both have reports of benthic cyanobacteria mats, some of which contained toxins (Quiblier et al., 2013). *Microcystis* species appear to be dominant ones in these mats however, other species such as *Cylindrospesmopsis* also incorporated in the mats (Quiblier et al., 2013). Cyanobacteria blooms have also been on the increase since 2002 along the Waikato River, which is used for drinking water supply for City of Hamilton, and many other small towns (Kouzminov et al., 2007). Noteworthy is that these blooms were causing taste and odour problems in the drinking water (Kouzminov et al., 2007). Cyanobacteria genera were also found to dominate the phytoplankton species in the Waikato River (Kouzminov et al., 2007).

Asia

Lake Taihu, which is the third-largest freshwater lake in China, is currently eutrophied, due to pollution emanating from river discharges since 1981. This has resulted in an increase in phytoplankton diversity (Liu et al., 2011), with *Microcystis* dominating the phytoplankton composition (Qin et al., 2010). In May 2007, water supply to Wuxi residents was interrupted due to a cyanobacteria bloom, which recurs in the summertime in different years. The bloom occurred earlier than expected and the cause was linked to increases in nutrient concentrations and water temperature (Paerl et al., 2014; Qin, 2014; Qin et al., 2010). It was concluded that nitrogen and phosphorus reductions would be required until the nutrient storage in the sediment is depleted to bring this hypertrophic lake below the bloom threshold (Paerl et al., 2014; Qin, 2014).

Middle East region

In the Middle East region, algal blooms were also reported in estuaries and marine environments. In the Persian Gulf, Sea of Oman, and the Arabian Sea, annual algal blooms were frequently reported, and some of the reported blooms caused fish kills and impacted other marine life (Al-Azri et al., 2012). In those areas, diatoms, and dinoflagellates were found to dominate the phytoplankton biomass with *Noctiluca scintillasn* as the dominant species, followed by *Cochlodinium polykrikoides* (Al-Azri et al., 2012; Gholami et al., 2019). In the Red Sea, along the Saudi Arabian coast near Jeddah, Al-Shaoaiba, and Al-Quonfuduah, around 125 phytoplankton species have been recorded, and toxic species such as *Dinophysis miles*, and *Dolichospermum* were reported (Gomaa et al., 2018). The species diversity was low and this was linked to the high salinity and temperature of the desalinated water effluent that has been discharged along the coast (Gomaa et al., 2018).

2.3.2. Cyanobacteria occurrence and species diversity: an African perspective

At a global level, cyanobacteria bloom incidents have been well documented (Ndlela et al., 2016). An increase in water temperature, due to climate change, salinity, and nutrients from anthropogenic activities, have resulted in cyanobacteria expansion over other phytoplankton species (Mantzouki et al., 2018; Paerl et al., 2008; Rigosi et al., 2014; Soares et al., 2009). With the projected change in climate and global warming, the occurrence of cyanobacteria blooms will increase across the globe, and will be even more severe on the African continent, especially in subtropical regions (Harke et al., 2016b; Ndlela et al., 2016; Quiblier et al., 2013; Rigosi et al., 2014) with a desert and semi-arid climates, like Namibia.

The African continent is characterized by a high poverty rate, political instability, wars, poor water infrastructure, and minimal and poor sanitation facilities, all of which contribute to nutrient loading into lakes, estuaries, and rivers (Ndlela et al., 2016). Apart from these issues, availability of water in some parts of the continent is spatially and temporally variable under the impact of climate change (Conway et al., 2009; Collier et al., 2008; Urama et al., 2015) . About 85% of the major rivers in Africa are shared with several countries (Ashton et al., 2016). Population increases and changes in the pattern of water use suggest that more African countries will exceed the limits of their economically usable, land-based water resources (Ashton et al., 2016). The eastern part of Africa is projected to get much wetter, but the southern and northern parts of the continent will get drier and hotter, with less rainfall compared to the central, and western parts of the continent (Collier et al., 2008). If this happens, blooms of cyanobacteria will be prevalent in many of the continent's water supply sources (Ndlela et al., 2016). However, reports of cyanobacteria blooms are limited in Africa. Only 21 of the 54 countries have documented cyanobacteria blooms (Ndlela et al., 2016). Figure 2.1 shows some lakes in Africa that have reported cyanobacteria blooms and associated cyanotoxins. In Namibia, although cyanobacteria blooms were detected in the Freidenau region since 1983, no reports are available in the literature of these incidents for the SWP and VB dams in the region. This section below will provide an overview of incidents of cyanobacteria blooms in Africa.

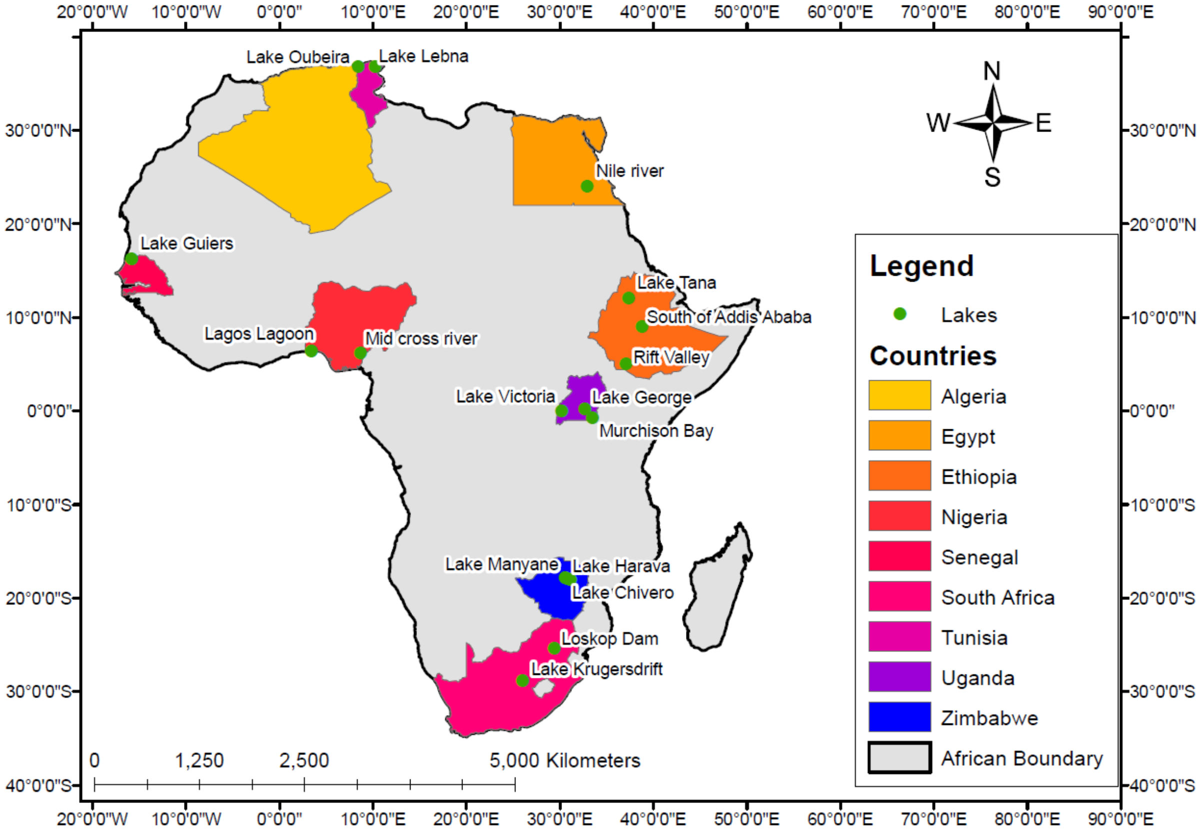

Figure 2.1. Lakes or rivers reported with cyanobacterial blooms and associated toxins in Africa.

North Africa

Cyanobacteria blooms, with the potential production of hepatotoxin and nodularin toxins, were identified and quantified in the Nile River Delta in Egypt (Amer, 2009). Among the identified cyanobacteria, the dominant genera were *Microcystis* (*wesenbergii* and *aeruginosa* sp.), followed by *Synechococcus sp.*, *Nodularia spumigena* and *Cylindrospermospis* sp. (Amer, 2009). The cause of the increase in growth was linked to the availability of phosphorus and nitrogen from anthropogenic activities in the catchment area. To understand the influence of the nutrients on cyanobacteria blooms, a multiple regression Analysis of Variance (ANOVA) test was carried out and the results showed a significant effect of total phosphorus on the abundance of hepatotoxic cyanobacteria bloom. The water was found to be limited by phosphorus. The hepatotoxin concentrations were however below the limit of 1µg/l set by World Health Organisation (WHO, 2021).

Blooms of *Cylindrospesmopsis raciborskii* have been reported in Lake Oubeira, in Algeria (Bouaıcha, 2004). A high count of this genera was observed in autumn, and *Microcystis* was also observed, although not dominant (Bouaıcha, 2004). Bouaicha, (2004) revealed the presence of microcystin which varied seasonally as dissolved (extracellular) and particulate (intracellular). The particulate concentration was the highest in late spring (April-May), and early spring (Feb-March), and increased gradually in late summer (August) to 28 346µg/l (Bouaıcha, 2004). In Lake Lebna, Tunisia, and the dominant genus was *Microcystis*. In the study by Herry et al. (2008), three species of *Microcystis* (*wesenbergii* and *aeruginosa*) were isolated from Lake Lebna and were compared morphologically, and showed varying cell size and colony morphology. These genera of *Microcystis* are commonly found in Northern African freshwater bodies and were, in most cases, associated with microcystin synthetase genes *mcy A*, *B,* and *C,* with the potential to produce toxins (Herry et al., 2008). Since the Northern part of African is characterized by a semi-arid climate, genera of cyanobacteria species found in those freshwater bodies could be similar to those found in freshwater bodies in Namibia.

Freshwater bodies, used for recreation and drinking water supply purposes, were observed to have toxic cyanobacteria blooms in Morocco. A study by Oudra et al. (2001) revealed the isolation of 19 cyanobacteria strains from reservoirs and ponds in Morocco. From the isolated strains, the dominant strains were *Microcystis*, *Dolichospermum*, and *Synechocystis* (Oudra et al., 2001). In the same water bodies, some of the strains were found to form blooms when the physicochemical and climatic conditions were favourable.

West Africa

In Lake Guiers Senegal, cyanobacteria were the second most dominant phytoplankton group, with a 56% value of the total phytoplankton biomass (Bouvy et al., 2006). There were 26 identified species of cyanobacteria, and *Cylindrospesmopsis* was the dominant species (Bergerl al., 2008). In the same lake, cyanobacteria were found to be dominant during the following weather conditions: (a) higher temperatures, (b) higher solar insolation, and (c) north-westerly winds (Berger et al., 2006). However, overall the dominant species was still *Cylindrospesmopsis raciborskii* , except during lower temperatures and low solar insolation when *Diatoms fragilaria* and *Dolichospermum miniata* were dominant.

In Nigeria, Zaira aquaculture ponds were screened for the presence of biotoxins hepatotoxic *Microcystis*. The study revealed four genera of cyanobacteria namely: *Microcystis*, *Nostoc*, *Planktothrix*, and *Dolichospermum* (Chia et al., 2009). Among the assessed ponds, some were found to contain microcystin toxins with concentrations ranging from 0.6 to 5.89 µg/l (Chia et al., 2009). It was suspected that bioaccumulation of the toxins in fish tissue was possible. In the Guinea Savanna area, excavation and mining ponds were found to contain cyanobacteria and the dominant species was also *Microcystis* (Chia & Bako, 2008).

East Africa

Lake George, which is a shallow lake in Uganda, has been dominated by *Microcystis* since 1962 (Green, 2011). The occurrence of blooms containing *Microcystis* are more permanent in tropical lakes found closer to the equator, such as Lake George. In temperate regions, *Microcystis* occurs in the summer season, and is mainly found in the epilimnion of eutrophic lakes (Green, 2011). In Murchison Bay, Lake Victoria, cyanobacteria blooms of the genera *Microcystis*, and *Dolichospermum* were found to be more dominant (Haande et al., 2011). However, diatoms were found to be dominant in the outer part of the bay (Haande et al., 2011). In the same bay, the occurrence of cyanobacteria blooms has become more frequent, causing odors, tastes, clogging of pumps and an increase in chlorine demand in the treated water supply to the capital, Kambala (Muyodi et al., 2009; Okello et al., 2009). In the subtropical climate of Namibia, the occurrence of *Microcystis* blooms follow a similar trend.

In Ethiopia, shallow small reservoirs are used for irrigation and livestock drinking water, which is causing the proliferation of cyanobacteria blooms due to euhrophication (Dejenie et al., 2008). In Lake Tana, the largest water body in Ethiopia, a study by Mankiewicz-boczek et al. (2014), revealed the presence of microcystin producing cyanobacteria. Microcystin was found to be occurring every post rainy season ranging from 0.58 to 2.65 µg/l. Detection of the toxin in the above concentrations resulted in the prohibition of water usage from the lake (Mankiewicz-boczek et al., 2014). Lakes in the Rift Valley, South of the Capital of Addis Ababa, were assessed for the presence of microcystin, due to the deaths and illnesses of wild and domestic animals. The study revealed the presence of microcystin with concentrations lethal to humans and animals (Willén et al., 2011). The dominant cyanobacteria genera causing the production of the toxins were found to be *Microcystis aeruginosa*, followed by *Microcystis panniformis, Anabaena spiroides and Cylindrospermopsis* (Willén et al., 2011).

Southern Africa

In sub-Saharan Africa the ecological integrity of surface water is compromised. The fish population is declining, toxic cyanobacteria algae blooms are becoming more abundant, and dissolved oxygen levels are declining in surface water bodies, due to excessive nutrient loadings (Davies & Koop 2006; Nyenje et al., 2010).

In South Africa, the most abundant species of cyanobacteria found in eutrophic water systems is *Microcystis aeruginosa* (Oberholster et al., 2005). This species has resulted in the death of livestock and wild animals in the country. In Hartbeespoort Dam, toxic strains of cyanobacteria, and microcystins were confirmed using the Enzyme-Linked Immunosorbent Assay (ELISA) (Mokoena & Mukhola, 2019). In Lake Krugersdrift, toxic strains of *Microcystis aeruginosa* were confirmed using ELISA and the same molecular maker techniques that detected the presence of microcystin (Oberholster et al., 2009). In the Kruger National Park, incidents of megaherbivores mortality were investigated, and it was discovered that the causes were due to intracellular microcystin toxins at a level of 23.718 µg/l, which was produced by *Microcystis aeruginosa*. The microcystin LR variant is the most dominant cyanobacteria toxin in southern African freshwaters. Harding et al. (2009) mentioned that conditions of raw water sources in South Africa have worsened, and about 35% of the total reservoirs were classified as eutrophic to hypertrophic. The causes were due to failing wastewater infrastructure in the catchment areas, and an increase in pollutants from urban runoff that comprises of a significant fraction of inflows to inland reservoirs. This is a common causes of eutrophication for most African lakes.

In the Limpopo province, South Africa, a study was carried out to assess the presence of toxic cyanobacteria in containers that store water for households (Fosso-Kankeu et al., 2008). The results revealed the presence of cyanobacteria, which has formed a biofilm inside the vessels (Fosso-Kankeu et al., 2008). Three cyanobacteria genera identified from the study were *Microcystis*, *Oscillatoria*, and *Dolichospermum* (Fosso-Kankeu et al., 2008). In the Limpopo river basin, rivers such as Limpopo, Crocodile, Mokolo, Magalakwena, Nzhelele, Lephalale, Sand, Notwane, and Shade Rivers all reported the presence of cyanobacteria toxic species. A study by Magonono et al. (2018) revealed that *Microcystis* species was dominant species followed by *Raphidiopsis raciborkii*, *Phormidium* and *Planktothrix* species in river bottom sediments in the same river basin.

Loskop Dam, South Africa, which supplies water for irrigation, is undergoing eutrophication due to anthropogenic activities in its catchment area such as acid mine drainage, and effluent from wastewater facilities (Dabrowski et al., 2013). The water quality is deteriorating due to the increase in blooms of *Microcystis* because of eutrophication (Dabrowski et al., 2013). In the Theewaterskloof Dam in the Western Cape Province of South Africa, a strain of

Dolichospermum with the production of microcystin-*LR* was also found (Oberholster et al., 2015). This is not common in African Lakes for *Dolichospermum* to produce cyanotoxin variants LR as opposed to its known variant Anatoxin-a and Anatoxin-a (s) (Oberholster et al., 2015).

On the Manyane River, in Zimbabwe, lakes found upstream were assessed for their trophic status. Nitrogen was found to be a limiting nutrient in Harava, Seke, and Manyane Lakes, while phosphorus was found to be a limiting nutrient in Bhiri Lake, and no nutrient limitation was observed in Lake Chivero (Tendaupenyu, 2012). The dominant phytoplankton in all the lakes was cyanobacteria with the genera *Microcystis* dominating (Tendaupenyu, 2012). However, *Dolichospermum* was also found to be dominant in Lake Chivero, and *Ceratium* in Lake Manyane (Tendaupenyu, 2012) .

Lake Chivero is one of the eutrophic reservoirs due to its locality downstream of the capital city of Harare, where industrial effluent is received, resulting in the development of cyanobacteria blooms over many years (Ndebele & Magadza, 2006). In their study of the occurrence of microcystin-LR in Lake Chivero, it was discovered that the dominant genera of cyanobacteria was *Microcystis* (Ndebele & Magadza, 2006). Average concentrations of 19.86µg/l for microcystin was detected. The concentration was a concern to humans and wildlife and was a call for eutrophication control. Nhapi et al. (2004) discovered that sewage effluent from the capital city of Harare, was the major source of nutrients in Lake Chivero, causing eutrophication. This could be similar to the SWP Dam located downstream of the capital city of Windhoek in Namibia.

2.3.3. Cyanobacteria occurrence and species diversity a Namibia perspective

There are 15 dams constructed in Namibia’s ephemeral rivers (Figure 2.2). The dams are used for water supply for humans and animals, and groundwater recharge (Botes et al., 2003). An ecological survey of the Friedenau Dam in 1983, which is relatively shallow with a capacity of 6Mm^3/a, revealed the presence of cyanobacteria and *Microcystis aeruginosa* as the dominant species in all the samples (DWA, 1983). During that survey, it was reported that the water turbidity was high due to the inflow of water from the catchment area (DWA, 1983). The frequency of cyanobacterial blooms as shown in (Figure 2.3), may increase in frequency due to upstream anthropogenic activities that generate nutrient inputs. However, no major bloom incidents have been reported since the survey, except for fish kills during the dam overturn period.

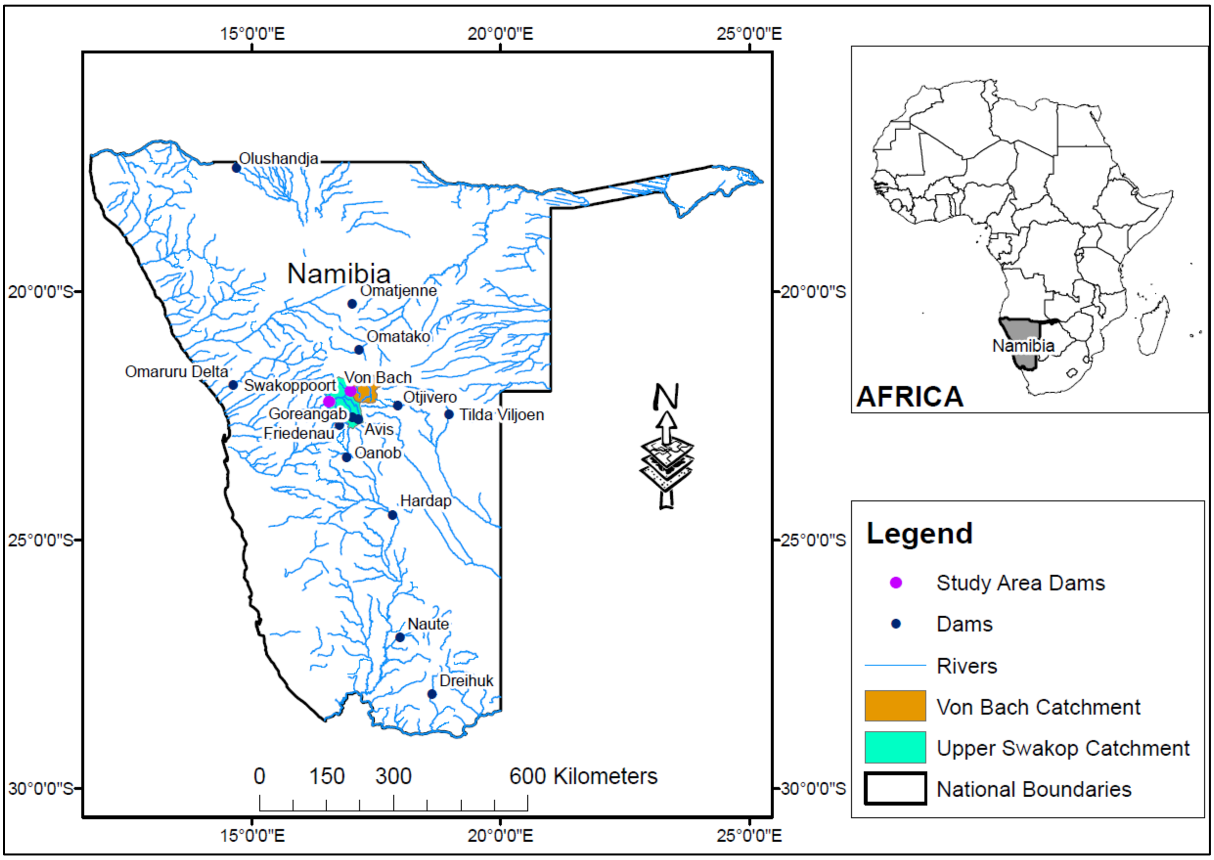

Figure 2.2. Man-made dams on the ephemeral rivers of Namibia.

A study of temperature and oxygen patterns in the Oanob Dam in 1997 indicated that oxygen closely followed the thermal pattern stratification in summer, with a gradual decrease in autumn, and complete mixing in winter (Schachtschneider, 1997). The epilimnion was supersaturated with oxygen, due to algal photosynthesis (Schachtschneider, 1997). No incidents of cyanobacteria blooms were previously reported from this dam; however, proposed housing developments, which will consist of human waste collection facilities around the dam, could create a suitable environment for cyanobacteria blooms to occur in the future.

In the VB Dam the occurrence of parasites such as *Giardia* and *Cryptosporidium* were reported in 1996 (MAWF, 1998). Blooms of cyanobacteria have been observed in the VB Dam as shown in Figure 2.3b below. There are speculations that the VB Dam will soon be eutrophied by nutrients emanating from the SWP Dam upstream. A study by Sirunda et al. (2021b), revealed that nitrate concentrations were higher in VB Dam than in the SWP Dam. Cyanobacteria were also reported, and the dominant species was *Dolichospermum*, with counts similar to those in SWP Dam (Sirunda et al., 2021b). *Microcystis* spp. were also recorded in lower concentrations than that of SWP Dam (Sirunda et al., 2021b).

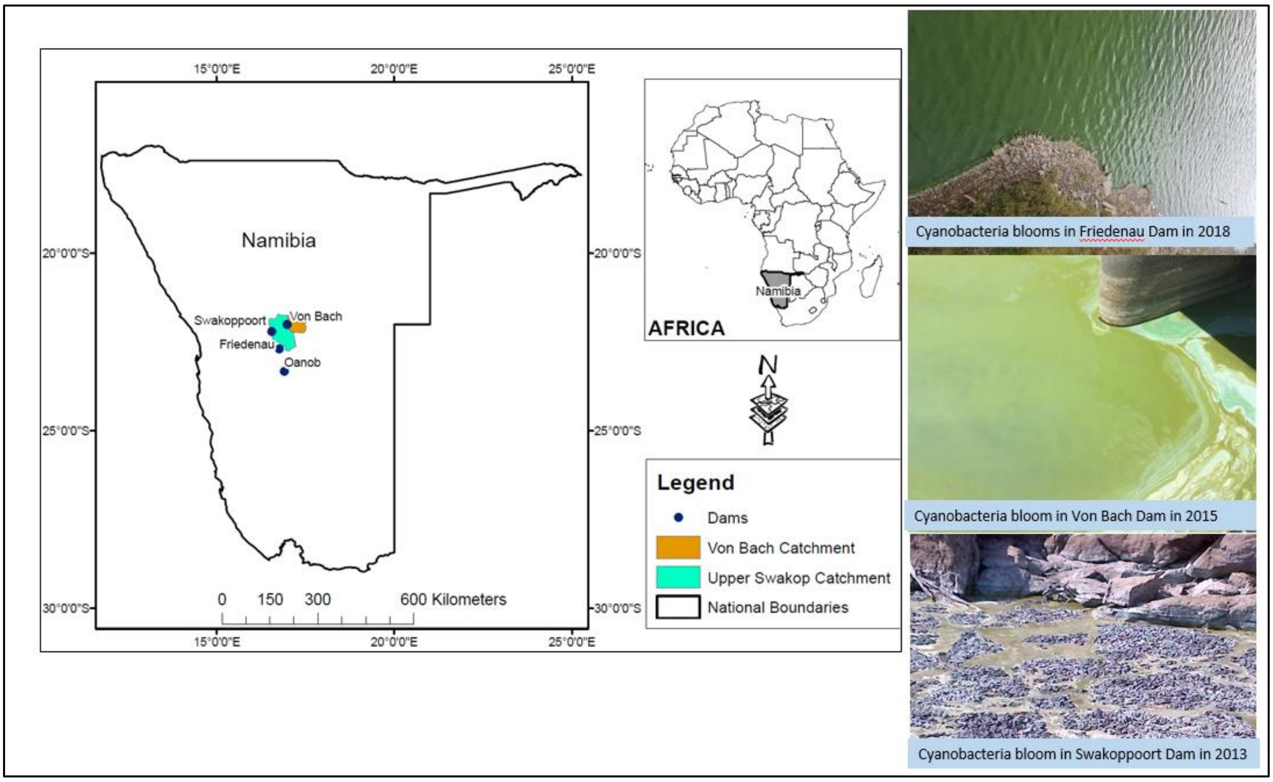

Figure 2.3. Cyanobacteria blooms: Friedenau Dam in 2018, VB Dam in 2015, and SWP Dam in 2013.

Among the dams in central Namibia, the SWP Dam could be the most polluted due to its locality downstream of the capital city, Windhoek, and Okahandja Town (Sirunda et al., 2022). Since 2010, the water quality of the SWP Dam started to deteriorate as evidenced by the frequent eruptions of cyanobacteria blooms with the potential to produce microcystin. These cyanobacteria blooms have posed a threat to the VBWTP which employ conventional treatment processes when treating SWP Dam water (for potable use), without blending, and the VB Dam, when the water is transferred to the latter (Sirunda and Dominic, 2014). Wastewater overflow from Municipal, agricultural and industrial facilities are reported to be the cause of cyanobacteria blooms in the SWP Dam (Figures 2.4, 2.5 & 2.6) (Lehmann, 2010). Quantification of the amount, path, fate, and transport of nutrients from these facilities remains limited. At the VB Treatment Plant, the main challenge is the removal of the associated toxins produced by cyanobacteria. It is widely known that cyanobacteria blooms sometimes produce toxins and these toxins could pose health implications to consumers if they are not removed (Westrick et al., 2010). It is not known how effective the processes are at the VBWTP at removing the toxins produced by cyanobacteria, which could compromise the quality of water produced from this plant. In addition, there are no studies conducted to understand the factors controlling growth, the dynamics, and control measures of the cyanobacteria and associated toxins of the SWP Dam on the treatment process of the VB Dam. A study carried out to understand the effect of discharges from the SWP Dam on the quality of the VB Dam concluded that the effect on water quality is more at the discharge point located at the abstraction point of water for treatment (Sirunda and Dominic, 2014). This gives the polluted water minimal time to affect the receiving water body water quality.

A study done by Cashman et al. (2014) revealed insufficient maintenance and monitoring of wastewater infrastructure in the city of Windhoek and the catchment area of the SWP Dam. As a result, numerous blockages were experienced, which cause wastewater spills into nearby tributaries, ultimately ending up in the SWP Dam downstream. Industries located in the city of Windhoek were challenged by financial shortfalls to maintain their wastewater treatment facilities (Cashman et al., 2014). Despite the above, the lack of awareness of the effect of pollution on surface water quality was found to be the cause of the lack of compliance to discharge untreated effluent in tributaries (Figure 2.5 & 2.6). Kgabi and Joseph, (2012) reported pollution from domestic waste, open defecation, and municipal sewers, which affected the water quality of the Gamammas River situated in the western part of Windhoek.

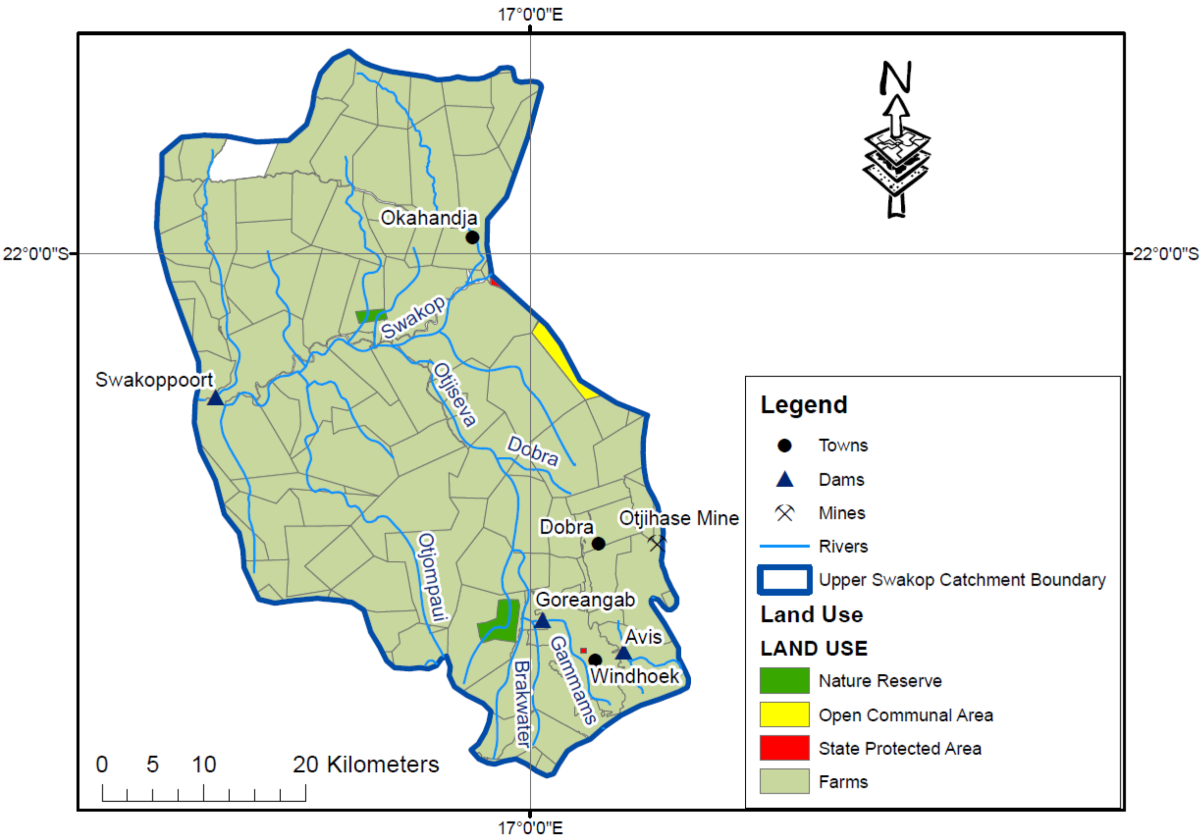

Figure 2.4. The upper SWP Dam catchment area with associated land use activities.

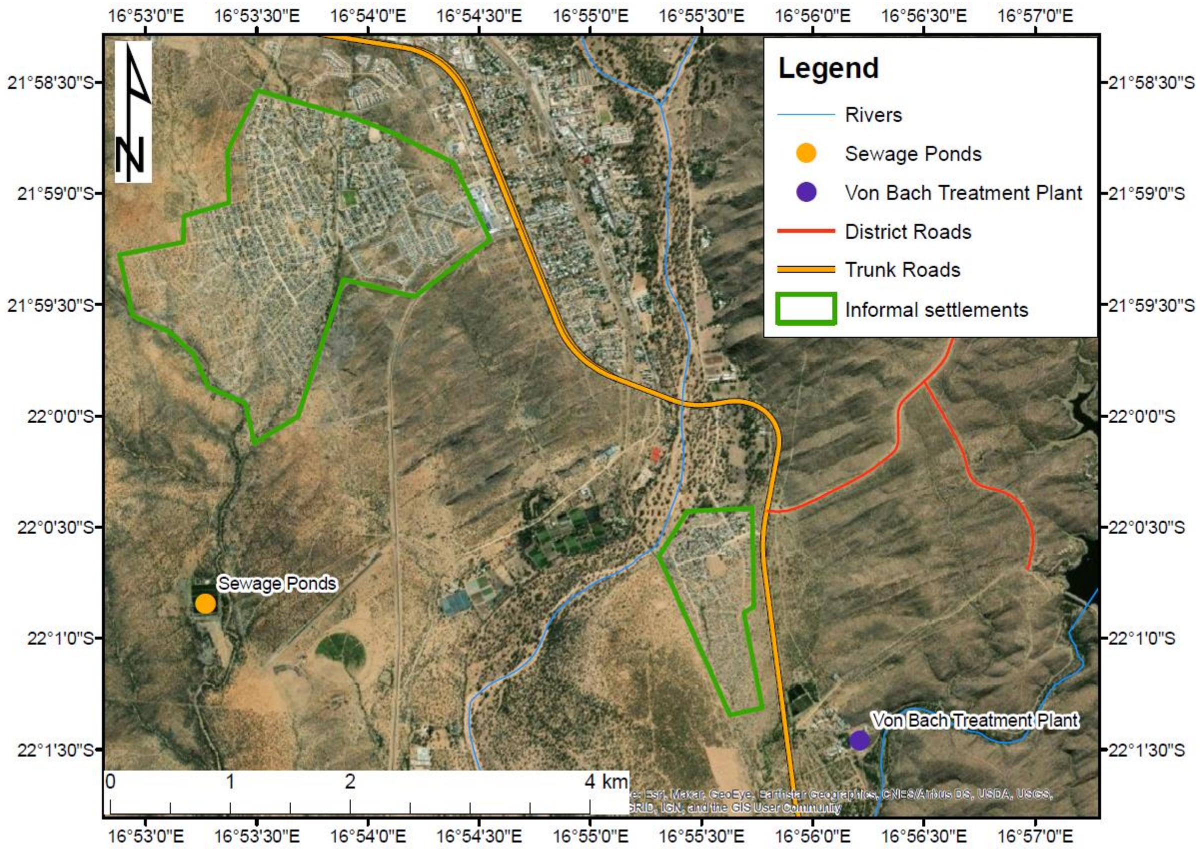

Figure 2.5. Okahandja town in the upper catchment of the SWP Dam with the informal settlement, VB Water Treatment Plant, and Sewage facility.

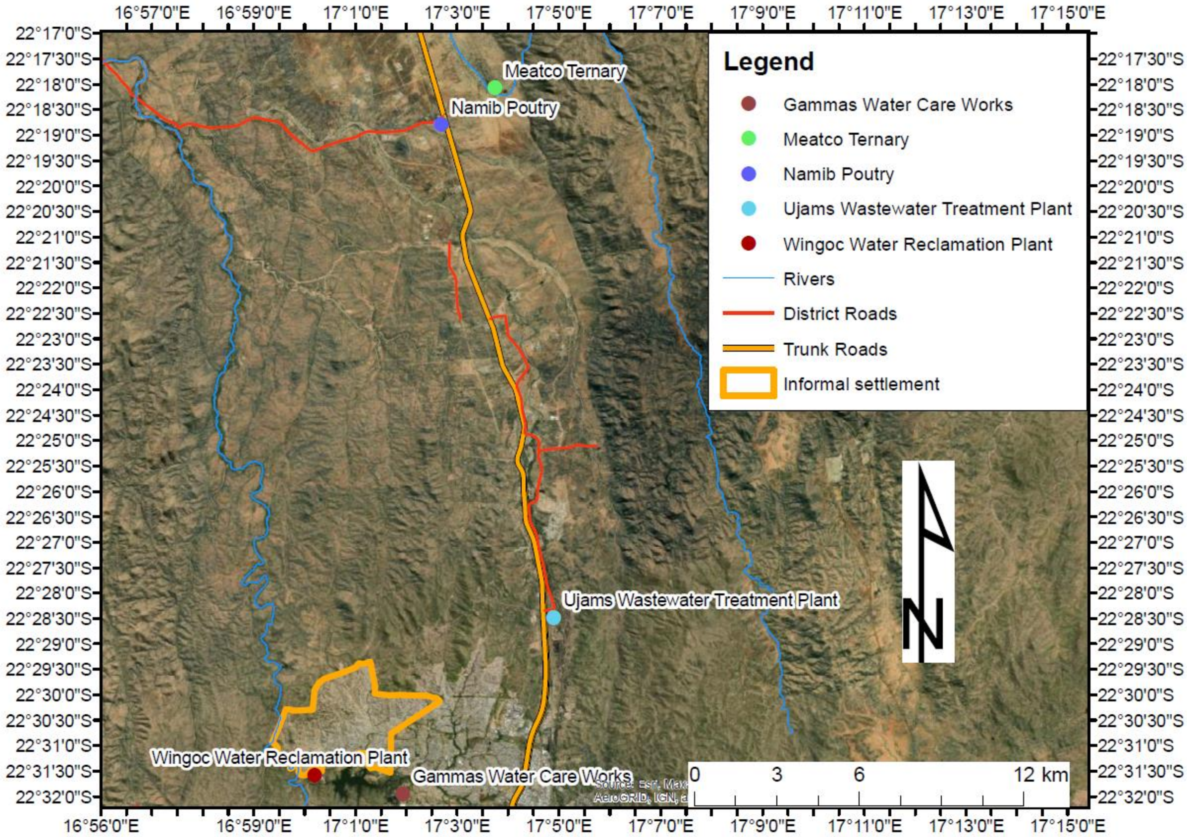

Figure 2.6. Land use activities in the upper catchment of the SWP Dam near the capital city of Windhoek, with the Ujams WWTP, Meatco Tannery and Namib Poultry, which may add extra burden on water quality.

From a governance point of view, the Department of Water Affairs (DWA) in the Ministry of Agriculture Water, and Land Reform is responsible for the control of water pollution in Namibia, and methods of control are stipulated in the Water Resources Management Act no.11 of 2013. The Integrated Water Resources Management (IWRM) Plan of 2010, recognizes the threat of increasing pollution to water security (Olivieri et al., 2022). The plan calls for sectors such as mining, tourism, and agriculture, to manage and control pollutants, which could lead to water quality improvement (Olivieri et al., 2022). Noted in the IWRM that most WWTPs in rural and urban areas are operated at or above capacity (IWRM Plan Joint Venture Namibia, 2010). A high rate of urbanization has created a major strain on the capacity to treat wastewater to unacceptable standards (IWRM Plan Joint Venture Namibia, 2010). Furthermore, the Environmental Management Act no 7 2007, also advocates for water pollution control, by listing activities which causes pollution not to be undertaken without an environmental impact assessment.

2.4. Cyanobacteria characteristics, toxins, health implications, and detection techniques

2.4.1. Cyanobacteria characteristics

Cyanobacteria are photosynthetic prokaryotes that capture sunlight for energy using Chl *a* and phycobilin pigments, absorb carbon dioxide, and emit oxygen (Chorus & Bartram, 1999; Jaiswal et al., 2008; Vincent, 2009). During the photosynthesis process, cyanobacteria cells absorb carbon dioxide and convert it into sugar through carbon fixation, while water acts as an electron donor, leading to the release of oxygen (Vincent, 2009). Cyanobacteria require water, carbon dioxide, nutrients, minerals, and light to grow (Chorus & Bartram, 1999). Photosynthesis is their principal mode of energy metabolism.

Cyanobacteria are found in both freshwater and marine environments and are sometimes called blue-green algae but their structure, genetic and physiology clearly identify them as bacteria (Khiari, 2019). They were the first organism to carry out oxygenic photosynthesis on earth and played a crucial role in the oxygenation of the earth's atmosphere. Cyanobacteria tolerate a wide range of climatic and environmental conditions, which lead to their proliferation as excessive masses in freshwater and marine ecosystems (Chorus & Bartram, 1999). Under favourable climatic and environmental conditions, cyanobacteria form dense and sometimes toxic blooms in freshwater and marine environments, which threaten ecosystem functioning and degrade water quality for recreation and drinking purposes (Berger et al., 2006; Ho et al., 2019; Huisman et al., 2018). However, they have the ability to store essential nutrients and metabolites within their cytoplasm under favourable conditions to survive in extreme conditions, making them good competitors over other phytoplankton.

Cyanobacteria lack a membrane-bound nucleus, chloroplasts and other organelles found in plants and algae. They have a double outer cell membrane and folded thylakoids membrane used in photosynthesis (Van Vuuren et al., 2006;). Cyanobacteria may exist in unicellular, colonial or multicellular filamentous forms and are identified morphologically by cell dimension, shape, colour, type of branching, sheath and cell contents (Chorus & Bartram, 1999). Canobacteria reproduce asexually via binary fusion and other reproductive strategies such as the production of baeoxytes exocytes, and motile filaments, called hormogonia. To inhabit different niches of aquatic ecosystems, cyanobacteria form ecological strategies such as scumming, homogeneously, stratifying, and nitrogen-fixing from the atmosphere (Chorus & Bartram, 1999). Despite their significant roles in aquatic ecosystems of producing oxygen, cyanobacteria produce various compounds that negatively and positively affect water quality

(Chorus & Bartram, 1999). These compounds are: geosmin, which affect the taste and colour of the water; 2-methylisoborneol, which give an earthy and musty odour from the water; and cyclocitrals, which gives off a grassy odour from the water. Of greater concern is the production of three classes of toxins: hepatotoxins, which attack the liver; neurotoxins, which attack the nervous system; and dermatotoxins, which cause skin irritations. These toxins are produced by certain strains of cyanobacteria species, which form blooms (Chorus & Bartram, 1999; Van Vuuren et al., 2006).

The most common bloom forming cyanobacteria species are *Cylindrospermopsis*, *Merismopedia*, *Microcystis* sp., *D. Oscillatoria and D. Anabaena (Dolichospermum)*. *Dolichospermum, Cylindrospermopsis and Oscillatoria are* multicellular filamentous genera of cyanobacteria (Dvorak et al., 2017). Unlike *Oscillatoria, Dolichospermum and Cylindrospermopsis* produce akinetes and heterocysts (Van Vuuren et al., 2006). Heterocytes main function is to fix atmospheric nitrogen using the enzyme nitrogenase. *Dolichospermum* is the mostly found in freshwater bodies of temperate and subtropical regions. *Cylindrospermopsis* is a common bloom forming genera found in lakes in tropical regions, with some reports in temperate regions (Fastner et al., 2003; Sinha et al., 2017). *Dolichospermum and Cylindrospermopsis* produce toxins, which are associated with adverse health effects in human and animals (Chorus & Bartram, 1999). *Dolichospermum* toxin is 95% intracellular and only released when the cells rupture and become extracellular. About 50% of *Cylindrospermopsis* toxins are extracellular and the remaining, intracellular (Environmental Protection Agency, 2014) .

Microcystis and *Merismopedia* are unicellular genera of cyanobacteria (Chorus & Bartram, 1999). Unlike *Merismopedia, Microcystis* is one of the most toxic bloom forming genera of cyanobacteria in freshwater systems found in temperate and subtropical regions (Environmental Protection Agency, 2014; Van Vuuren et al., 2006). Under the favourable environmental conditions, some strains of *Microcystis* produce microcystin, the most toxic of which is microcystin-LR (Oberholster et al., 2004; Oberholster et al., 2009). In 95% of cases, this toxin is retained intracellularly and only released when the cells rupture and become extracellular (Environmental Protection Agency, 2014) . Microcystin is reported to be lethal to humans and animals (Oberholster et al., 2005).

Cyanobacteria species possess gas vesicles as cytoplasmic inclusions, which enable them to regulate their buoyancy ability to move up and down the water column to search for food and light (Chorus & Bartram, 1999). Their gas vesicle is similar in molecular structure but different in shape, yield, and critical pressures (Walsby & Bleything, 1988; Walsby et al., 1995). *Microcystis* gas vesicles have a high critical pressure, i.e. stable when held under sustained pressures (Thomas & Walsby, 1985; Walsby et al., 1995). Given the types of gas vesicles and the associated sustaining pressures during the rainy season, species with high critical pressure gas vesicles could be dominant in turbid water as their gas vesicles do not collapse due to inflow water pressure (Li et al., 2014; Wang et al., 2011). Factors such as temperature and light affect the production of gas vesicles and control the flowing and sinking ability of cyanobacteria species in the water column (Li et al., 2014).

2.4.2. Cyanobacterial toxins production

Cyanobacteria produce various secondary metabolites such as cyanotoxins, geosmin, and 2mib (2-methylisorboneol), all of which have different biochemical activities (Bláha et al., 2009). Cyanotoxins are of two types, namely biotoxins and cytotoxins. Biotoxins are highly lethal, more poisonous to animals and human beings than cytotoxins (Jaiswal et al., 2008). Biotoxins are divided into three groups based on their chemical structure: cyclic peptides, alkaloids, and lipopolysaccharides. The most known cyclic peptides are microcystin and nodularin. Microcystin is a group of cyclic heptapeptides produced by cyanobacteria genera *Microcystis*. It consists of seven amino acids, which are made up of five non-protein amino acids and two protein amino acids. It is these two amino acids that distinguish microcystin from the other toxins. The amino acid single letter code nomenclature gives a designated name for each microcystin depending on the variable amino acid that completes their name. For example, the common and most toxic microcystin contains the amino acid leucine (L) and arginine (R) which is microcystin-LR (Bláha et al., 2009). Nodularins, which are also highly lethal, are found mainly in marine and brackish water environments, which are not used for drinking purposes.

Alkaloid toxins vary in their chemical structure and stabilities depending on the type. They can affect the nervous systems, skin, liver, or gastrointestinal tract. Examples of alkaloid toxins are anatoxin, saxitoxin, and cylindrospemopsin. Anatoxin is produced by *Dolichospermum*, with toxicity levels ranging between 20µg/kg to 250µg/kg making them more toxic than many microcystins (Jaiswal et al., 2008). Anatoxin structure resembles the structural analogue of cocaine and neurotransmitters (Jaiswal et al., 2008). Despite the different toxins produced, some cyanobacteria are genetically capable of producing toxins but do not do so under all conditions, and some do not produce toxins at all (Oberholster et al., 2006). Sigitas et al. (2017) stressed that toxicity testing has to become a standard routine in water quality monitoring for water bodies that are highly dynamic spatially and temporally and are vulnerable to environmental changes.

The toxicity levels of the cyanotoxins are affected by: (a) cellular concentration, (b) mode of exposure, (c) susceptibility of the victim, which are influenced by (d) age, and (e) weight (Carmichael, 1994). Despite the effect of the above-mentioned factors on toxicity levels, the growth phase, nutritional factors, and environmental factors also influence the toxicity levels of cyanobacteria (Jaiswal et al., 2008). Microcystin and anatoxin are released at the mid-exponential phase of the growth phase and when conditions are not favourable they are retained within the cells (Jaiswal et al., 2008).

From a nutritional point of view, various factors affect the production of toxins including nitrogen and phosphorus concentration in the water column. Some studies have reported less toxin production from *Microcystis aerugonosa* when the nitrogen source was removed or reduced (Codd and Poon, 1988). However, some reported toxin production by a lower level of phosphorus concentration (Sivonen, 1990). Phosphorus levels are needed for toxin production at a maximum saturation level of 0.4mg/l (Sivonen, 1990). Phosphorus was found to have a significant influence on the toxin production by cyanobacteria, but different responses among cyanobacteria strains and genera (Jaiswal et al., 2008). Light intensity also influences the production of the toxin. Toxin synthesis increased faster than protein synthesis at the light intensity between 20 and 40μEm^{-2}/s (Jaiswal et al., 2008). Changes in light quality have minor effects on toxicity but may affect growth. In addition to light, toxin production is also influenced by water temperature. Temperature ranges of 20-25ºC are best suited for toxin production in most cyanobacteria strains (Codd and Poon, 1988). Temperature has a varying effect on toxin production by different strains, e.g., *Microcystis aeruginosa* produces fewer toxins at the upper and lower temperature limits (Jaiswal et al., 2008). In addition to light and temperature, the pH of the water also influences the production of toxins by cyanobacteria. *Microcystis aeruginosa* is more toxic at a higher pH, and grow slower at lower pH values (Van der Westethuize and Eloff., 1983).

2.4.3. Cyanobacterial toxins: health implications

Cyanotoxins are associated with various negative health effects on both humans and animals. There is a high level of awareness about the health effects of cyanotoxins in developed countries; however, this is of relatively little concern in developing countries like Namibia. As a result, fewer reports are available of cyanotoxins in such countries. Cyanotoxins enters human and animals through various exposure routes, including skin (i.e. dermal route), inhalation, hemodialysis, and ingestion (Codd et al., 2017). These sometimes occur simultaneously (Codd et al., 2017).

Cyanotoxins entering humans and animals through these routes cause various reactions or infections. For example, for dermal routes exposure, the symptoms include (but are not limited to): rashes, blisters, allergic reaction, asthma, conjunctivitis, and ear or eye irritation. This route of entry is mainly for swimmers after contact with toxic blooms of cyanobacteria. Carmichael et al. (2001) mentioned that cyanotoxins are well known to cause poisoning in wild animals and fish. But little is known of this poisoning in human because, in most cases humans use treated water and avoid contact with water contaminated with toxic blooms.

2.4.4. Cyanobacterial toxins: incidents and detection techniques

Historically, the effects of cyanotoxins were never confirmed because of a lack of information about the vector and appropriate detection techniques (Carmichael et al., 2001). However, poisoning by cyanotoxins in humans has been reported at increased frequency. For example, in Brazil in 1996, an outbreak of toxin poisoning was reported to cause deaths to about 76 hemodialysis patients (Carmichael et al., 2001). To establish the cause of the deaths, phytoplankton samples were collected from the water supply reservoir and preserved in Lugol's iodine solution for analysis. Samples were also collected at the clinics' taps and filters. Liver samples were also collected from the deceased for chemical analysis using ELISA and HPLC (Carmichael et al., 2001). It was found that toxigenic cyanobacteria was the dominating phytoplankton, and the chemical evidence from liver tissues confirmed the presence of microcystin in the water used at the clinic (Carmichael et al., 2001). Falconer et al. (2005) stated that the reason why little is been reported on the toxins poisoning in humans is that the symptoms overlap with a gastrointestinal illness, which is caused by pathogens. Because of this, pathogens are investigated first then the toxins. Therefore, there is a need to understand how human cells react to toxin poisoning and its associated symptoms.

Apart from human infections by cyanotoxins, incidents have been reported in wild animals. In the Nhlanganwane Dam, Kruger National Park, South Africa, incidents of wild animal mortalities were reported in 2005 between the months of February and July (Oberholster et al., 2009). During that period a total of 52 carcasses were detected, comprising seven white rhinoceros, two lions, two cheetahs, nine zebras, 23 wildebeest, one hippopotamus, one giraffe, five buffalo, one warthog, and one kudu (Oberholster et al., 2009). The cause was identified to be microcystin –LR, which was measured at 23 718µg/l, while the extracellular level was at 2.1µg/l in the Nhlanganwane Dam (Oberholster et al., 2009). The extracellular microcystin –LR, detected in the Nhlanganwane Dam, was higher compared to other dams (e.g. Mpanama Dam at 0.581µg/l and Makhohlolo Dam at 0.317µg/l), where incidents of animal mortalities were also reported (Oberholster et al., 2009). During the investigation period, cyanobacteria dominated the phytoplankton biomass, with the genera of *Microcystis aeruginosa* reported in much higher concentrations than other phytoplankton species (Oberholster et al., 2009).

To detect cyanotoxins, various methods or techniques are used, with varying levels of accuracy. Detection methods include Mouse Bioassay, Enzyme-Linked Immunosorbent Assay (ELISA), Polymerase Chain Reaction (PCR), HPLC, and Protein Phosphate Inhibition (PPi) (Westrick et al., 2010). These methods differ in detection limits, cost, and technical complexity (Hawkins et al., 2005).

The mouse bioassay involves the injection of the cyanotoxins into an adult mouse, which is kept under observation. The symptoms of the toxins appear within minutes. This method has the ability to detect low levels of toxins, below the WHO limit of 1µg/l (Jaiswal et al., 2008). The method became less used due to the availability of more advanced methods, and due to the increased opposition in many countries to the use of animals for toxicity testing (Jaiswal et al., 2008).

ELISA is the most promising method for the rapid assessment of microcystin. This method is based on polyclonal or monoclonal antibodies used in combination with enzyme-labeled detection antibodies and chromogenic substrate (Hawkins et al., 2005). This method is used mainly for the screening of cyanotoxins such as microcystin and nodularin. It has a low detection limit and samples for this method and therefore do not require pre-concentration. The method detects microcystin in the range of 0.2-10µg/l (Westrick, 2008). Furthermore, the method has a quick turnaround time and is easy to use (Hawkins et al., 2005). De Almeida et al. (2016) used an ELISA to analyse water samples collected at Maestra Reservoir and Celeste Gobatto Water Treatment Plant in Brazil to determine the efficacy of the conventional treatment plant in removing cyanotoxin. The study revealed that microcystin in the Celeste Treatment Plant persisted up to the last treatment step, before it was completely removed. Studies by Szlag et al. (2015) and Hoeger et al. (2008) have also used ELISA methods to screen for microcystin-LR found in raw water reservoirs and conventional treatment plants.

PCR is another method used to differentiate between toxic genera or strains and non-toxic genera or strains of microcystins (Jaiswal et al., 2008). It involves the use of a gene probe to distinguish the toxic and non-toxic strains. The method tests for 16S RNA using primers. This method is very sensitive, low cost, and has rapid turnaround time.

HPLC is an analytical method based on physicochemical properties of cyanotoxins, such as molecular weight, chromospheres, and reactivity. A detection limit of 0.2µg/l is achievable by this method. Unlike ELISA, this method is used for the identification and quantification of independent cyanotoxins. In many studies, these two methods are used together for screening, identification, and quantification of microcystin. A study by Hoeger et al. (2008) also determined the occurrence and removal of microcystin and cylindrospemopsin in conventional treatment plants using both ELISA and HPLC to detect the toxins in the water.

PPi is a sensitive screening method with a quick turnaround time for microcystin and nodularin, which uses biochemical activities of enzymes (Jaiswal et al., 2008). This method uses radioactive chemicals and requires specialized equipment. The detection range is good, 0.3-1µg/l, but not specific for *Microcystin*.

2.4.5. Cyanobacteria treatment in water supply system

The occurrence of toxic cyanobacteria in water supply systems that make use of conventional treatment plants could cause a water crisis if not removed (Qin, 2014). Qin et al. (2010) reported that the water crisis experienced in Waxi, Jiangsu Province in China, was due to the occurrence of a cyanobacteria bloom. Extra monitoring and detection techniques were required from the water utilities. However, various treatment processes exist for the removal cyanobacteria cells and its associated intracellular and extracellular toxins.

Conventional Water Treatment Plants (WTP) are defined as those treatment methods which use processes for removal of suspended and colloidal materials from surface water, such as dissolved organic or extracellular toxin (Koivunen, 2007). In conventional WTP, the removal of cyanobacteria cells, and their associated extracellular toxins, is of high treatment priority. However, the processes in conventional treatment plants have been reported as ineffective in removing intracellular toxins.

Conventional treatment plant processes such as coagulation, flocculation, sedimentation, filtration, and disinfection, are used to remove suspended and colloidal materials such as clay, silt, algae, intracellular toxins, and microorganisms, but not dissolved solids. In the study by Knappe et al. (2004) it was found that, coagulation, flocculation, and dissolved air floatation were effective in the removal of algal cells. Dissolved air floatation was also reported to be more effective in the removal of *Microcystis aeruginosa* than sedimentation (Teixeira & Rosa, 2006). In their study, natural organic material influences cyanobacteria cell removal by coagulation, filtration, and sedimentation processes. However, less influence was observed with coagulation, filtration, and dissolved air flotation (Teixeira & Rosa, 2006).

The above-mentioned processes remove intracellular toxins and are not effective for extracellular toxins. Removal of extracellular toxins involves physical removal by adsorption by activated carbon, exclusion by membrane filtration, and chemical or biological removal by biological activity, ultraviolet light unit and oxidants (Westrick et al., 2010). Although these processes are effective in the removal of extracellular toxins, they are very costly to implement and could cause water prices to increase. Some utilities use them as auxiliary barriers designed to minimize the risk of toxins (Westrick et al., 2010).

Activated carbon is an effective method for the removal of microcystin and other contaminants, but the removal rate is highly dependent on the mesoporous capacity (Westrick et al., 2010). Two types of activated carbon are used in water treatment, namely: powdered activated carbon and granular activated carbon. These two types of activated carbon are characterized by their physical porous structure. A study by Ho et al. (2011) investigated the application of powdered activated carbon for the removal of cylindrospemopsin and microcystin variants from drinking water, found a reduction in both toxins in the drinking water. Many studies are available on activated carbon adsorption of microcystin, but little or few on anatoxin-a (Westrick et al., 2010). Activated carbon can be used as auxiliary treatment barriers in the conventional water treatment process (Westrick et al., 2010).

Reverse osmosis, nanofiltration, and ultrafiltration process separate contaminants by size and charges, which depend on the physical and chemical characteristics of the membrane. Reverse osmosis is an effective process and capable of removing microcystin LR and RR in the range of 10 and 30µg/l (Westrick et al., 2010). Teixeira and Rosa (2006) investigated the effectiveness of nanofiltration in removing microcystin and anatoxin. In their study, the results showed that the nanofiltration membranes were effective barriers against anatoxin-a and microcystin. It was further mentioned that the removal of anatoxin-a was facilitated by electrostatic interactions and steric hindrance, whereas microcystin was a mainly steric hindrance. These methods can also be used as an auxiliary treatment together with conventional processes.

Another auxiliary method is UV disinfection. This method involves the use of a low to medium pressure lamp with doses between 10 to 40 mJ/cm^2 for disinfection. When adsorbed, the energy break the molecular bonds of the organism. Furthermore, oxidants are also widely used to remove cyanotoxins. Commonly used oxidants in drinking water are chlorine, ozone, hydroxyl, chloramines, potassium permanganate, and carbon dioxide. In the conventional treatment plant, these oxidants are used before filtration or after filtration processes. In a WTP, oxidation demand is highly dependent on source water characteristics (Qin et al., 2010; Westrick et al., 2010).

Among the oxidants, chlorine is widely used as a disinfectant in conventional WTPs. The inactivation of organic contaminants by chlorine is pH-dependent because the pH of hypochlorous acid is 7.6. Various studies have been conducted to test the effectiveness of chlorine in removing cyanotoxins. Ho et al. (2006) reported that chlorination is effective in the removal of microcystin variants in the order of YR>RR>LR>LA. The same was also reported by Acero et al. (2005), where chlorination was effective in removing microcystin, but the reaction rate of the variants LR, RR, and YR, with chlorine, were the same. Xagoraraki et al. (2006) reported that the inactivation rate increased with an increase in chlorine dose and decreased with a decrease in pH. Even though chlorination was found to be effective to inactivate microcystin, the same was not observed with anatoxin-a (Westrick et al., 2010). The inactivation rate of anatoxin with chlorine has been reported as a very slow process (Westrick et al., 2010). Chlorine is therefore not a suitable oxidant for the inactivation of anatoxin-a (Newcombe and Nitchoston, 2004). Chloramine and chlorine dioxide, which are weak oxidants, are also not effective treatment barriers for microcystin and anatoxin-a. Subsequently, they are not good barriers for cyanotoxin in conventional treatment plants. Therefore, they are only used to provide residual disinfectants to reduce the formation of by-products by chlorine (Ho et al., 2006; Westrick et al., 2010).

Potassium permanganate, on the other hand, inactivates microcystin variants from the water, but is not reactive with cylindrospemopsin (Westrick et al., 2010). The reactivity rate with microcystin variants also is not pH-dependent. Furthermore, potassium permanganate is also reactive with anatoxin-a; however, this reactivity is pH-dependent. Ozone inactivates microcystin, anatoxin, and cylindrospemopsin. The same as potassium permanganate, its reactivity with microcystin is not pH-dependent generally except when reacting with anatoxin and *Cylindrospemopsin* (Westrick et al., 2010).

2.5.Cyanobacterial dynamics and factors influencing growths

2.5.1. Cyanobacterial dynamics

Cyanobacterial production and composition are reported to have vertical (spatial) and temporal dynamics in aquatic ecosystems (Moura et al., 2011; Touati et al., 2019). One factor that causes the dominance of cyanobacterial species is that they use gas vesicles, which allows them to travel; between the surface layers of a waterbody, where light is abundant, and lower, more nutirnet rich water (Chorus & Bartram, 1999; Reynolds et al., 1987; Overman, &Wells, 2022). It is reported that the optimal habitat for phytoplankton growth is within the top 3 m to 5 m of the water column (Ndebele-Murisa et al., 2010). Phytoplankton growth is higher near the surface and tends to decrease with depths until the light becomes the limiting factor (Ndebele-Murisa et al., 2010; Zhang et al., 2008).

The depth variations of cyanobacteria and toxins could lead to an abstraction of contaminated water by a WTP. This is a concern to utilities using the conventional treatment process. Depending on the concentration of the toxins and treatment processes, the toxins could end up in drinking water and cause public health problems (Qin et al., 2010; Westrick et al., 2010). Even though the monitoring program calls for samples to be taken at the intake tower, very few studies has been done on the vertical and temporal dynamics of cyanobacteria during toxic blooms (Westrick et al., 2010; Moura et al., 2011). This has not been studied in the SWP and VB Dams. Determining the dynamics of cyanobacteria blooms in different water sources is important to determine the risk associated with poorly mixed water when subjected to treatment (Westrick et al., 2010).

The dynamics of cyanobacteria in the water column are driven by anthropogenic and climatic factors. Anthropogenic nutrient enrichment and hydrologic modifications, and water residence time, are major drivers of cyanobacteria dynamics and, subsequently, bloom formation in water bodies (Paerl, 2018). Climatic factors, such as warming, changes in precipitation runoff patterns and amount, and changes in light intensity, act synergistically with anthropogenic factors to exacerbate the dynamic and bloom formation of cyanobacteria (Paerl, 2018; Paerl et al., 2008; Paerl & Otten, 2013).

Furthermore, phytoplankton also varies with seasons governed mainly by water temperature, light and nutrients. Cyanobacteria has been reported to dominate the seasonal dynamics in many subtropical lakes causing seasonal blooms (Ndebele-Murisa et al., 2010). Some lakes have seasonal blooms that start in summer and last into autumn, others experience cyanobacterial blooms throughout the seasons, or experience episodic blooms which last for weeks or days. For example, Lake Ogelube, a tropical lake in Nigeria, reportedly had higher phytoplankton biomass in the rainy season, dominated by *Chlorophyceae*, followed by cyanobacteria (Ndebele-Murisa et al., 2010). Furthermore, tropical and subtropical African lakes, including Lake Kariba (Zimbabwe), Lake Malawi, Lake Tanganyika, (Tanzania), and Lake Victoria (Uganda), are reportedly dominated by cyanobacteria in summer and diatoms in winter (Hecky & Kling, 1981; Ndebele-Murisa et al., 2010).

Globally, studies of seasonal dynamics of cyanobacteria have been carried out in many lakes to ensure effective management and control of cyanobacteria blooms (Ballot et al., 2014; Bouvy et al., 2006; Chinyama et al., 2016; Lira et al., 2011; Ndebele-Murisa et al., 2010). In the Ingazeira Reservoir, Brazil, cyanobacteria were reported to dominate the phytoplankton community, with *Cylindrospermopsis* as the dominant spp. (Bouvy et al., 1999). In Arcoverde Lake, Brazil, cyanobacteria were also reported to dominate the phytoplankton community with *Cylindrospermopsis* as the dominant spp. at all depths (Bittencourt-Oliveira et al., 2012). In the Vaal Dam, South Africa, *Microcystis* and *Dolichospermum* were found to dominate the phytoplankton community in most of the months, except for February and March (Chinyama et al., 2016). Cyanobacteria was reported to dominate the Lake and their dominance could be related to seasonal dynamics and water column depths.

2.5.2. Hydrological and environmental factors which influence cyanobacterial growth

The global patterns of temperature and precipitation have significantly changed over the last century (Junior et al., 2018; Ziervogel, 2018). In Africa, the observed and projected rate of surface temperature has increased more rapidly than the global temperature, with human induced climate change being the main driver (IPCC, 2021; Ziervogel, 2018). An ambient temperature increases of at least 2°C is predicted in south-western Africa where Namibia is found, in combination with a decrease in mean annual precipitation, increased intensity and frequency of heavy precipitation events and pluvial flooding, observed and projected increase in aridity, agricultural and ecological/hydrological droughts, increase in dryness, and increase in wind speed (Arias et al., 2021).

An increase in global mean surface temperature will affect processes involved in water scarcity in semi-arid regions, like Namibia (IPCC, 2019; Matchaya et al., 2019; Ziervogel, 2018).. The projected increases in population and income, combined with changes in consumption patterns, will result in increased water demand by 2050 (IPCC, 2019). These changes have implications on water scarcity (Haghighi et al., 2020; IPCC, 2019, 2021; Kousari et al., 2014; Matchaya et al., 2019). The main factors explaining the impact of climate change on water quality are strongly related to warming and extreme events, such as hydrological droughts (Delpla et al., 2009; Xia et al., 2015). It is projected that surface temperature will rise, mean precipitation will decrease, and drought risks will increase in subtropical regions (IPCC, 2014). Therefore, these climate related events could affect environmental and hydrological variables that affect the quality of surface water (Delpla et al., 2009).

Water temperature

The rise in water temperature is known to affect surface water quality by modifying physicochemical and biological variables (Delpla et al., 2009). Overall increases in water temperatures, due to climate change, is causing lakes to stratify earlier in spring and di-stratify later in autumn (Funari et al., 2017). With extended stratification the optimal growth rate of cyanobacteria is lengthened and they become more competitive than other phytoplankton (Paerl et al., 2008; Kozak et al., 2017)because cyanobacteria dominate at higher temperatures(Chapra et al., 2017; Liu et al., 2011; Paerl, 2018). In many eutrophic water bodies, the occurrence of cyanobacterial blooms has been reported at water temperatures of greater than15℃ (Chen et al., 2003; Liu et al., 2011; Nalewajko & Murphy, 2001; Robarts & Zohary, 1987). Changes in water temperature influence the patterns of stratification, which greatly affect the seasonal growth of phytoplankton (Robarts & Zohary, 1987). The growth rate of cyanobacteria is temperature dependent, with an optimum above 25℃ (Konopka & Brock, 1978; Robarts & Zohary, 1987). In Lake Taihu, China, *Microcystis* dominated during periods of warm water temperatures ranging from 18.2℃ to 32.5℃ (Chen et al., 2003). In tropical Lake George, Uganda, *Microcystis* optimal growth was found in water temperatures ranging from 24℃ to 34℃ (Robarts & Zohary, 2010). Similarly, in the Hartbeespoort Dam, South Africa, optimal growth for cyanobacteria was found in water temperatures ranging from 12.3℃ to 28.9℃ (Robarts & Zohary, 2010). Lake Biwa, Japan, water temperature in the range of 28℃ to 32℃ was found to be suitable for *Microcystis* and *Dolichospermum* growth (Nalewajko & Murphy, 2001).

Cyanobacteria responses to changing water temperature vary. However, among the dominant genera of cyanobacteria, *Microcystis* growth occurs between 15℃ and 37℃ (Konopka & Brock, 1978; Robarts & Zohary, 2010). *Dolichospermum* grows very slowly between 10℃ and 15℃, but growth is optimal up to 30℃, while *Oscillatoria*, tend to prefer low temperatures of approximately 10℃ (Robarts & Zohary, 2010). An increase in water temperatures (due to increased air temperature due to global warming) will, therefore, have an effect on the growth rates of cyanobacteria (Chapra et al., 2017; Costa, 2012; Iglesias et al., 2007; Kimambo et al., 2019; Oberholster, Botha, et al., 2009; Paerl et al., 2008; Whitehead et al., 2009; Winder & Sommer, 2012; Yan et al., 2017). In Lake Tanganyika, Tanzania, cyanobacteria were found to dominate other phytoplankton and climate change (warming) was mentioned as the driving force (Kimambo et al., 2019). Apart from the influence on the euphotic depth (Zeu) thickness and depths, water temperature also causes thermal water stratification in lakes, forming a temperature gradient (Dantas et al., 2011). The temperature gradient is mostly used to define the mixing depth (Zmix) which affect cyanobacteria growth (Gray et al., 2020).

Light availability

Cyanobacteria have a better tolerance to higher light intensities compared to other phytoplankton, which can reach 700 -1000 µE m^2s^{-1}. The availability of light in water determines the extent to which the physiological properties of cyanobacteria will be at an advantage compared to other phytoplankton. Light is needed for cyanobacteria to undergo photosynthesis, and the depth at which this happens in the water body is called the Zeu (Chorus & Bartram, 1999). This depth zone extends from the water surface until the depth at which 1% of sunlight is detected (Chorus & Bartram, 1999). The Zeu is estimated by measuring water transparency with the Secchi disk and multiplying the Secchi depth reading by a factor of 2-3 (Chorus & Bartram, 1999). This depth may be deeper or shallower depending on the turbidity of the water. In highly turbid water the Zeu is shallower than the epilimnion (Zeu/Zmix is <1). The epilimnion is the upper depth in a thermally stratified lake, and the lower zone is called the hypolimnion. The ratio between Zeu and Zmix is widely used to measure the availability of sunlight in the water column for photosynthesis by cyanobacteria phytoplankton (Dantas et al., 2011; Khanna et al., 2009; Yang et al., 2019). In the turbulent water column, the irradiance encountered by the phytoplankton cells is the function of the Zeu and Zmix.

Light intensity decreases with an increase in the Zmix. Cyanobacteria genera respond differently to the variation in light intensity. The growth of some genera of cyanobacteria, such as *Oscillatoria*, is inhibited when exposed to an extended period of light intensity, but tend to dominate under mixed water with a low irradiance with a Zeu/Zmix ratio of less than 0.20 ratio (Chorus & Bartram, 1999; Havens, n.d). *Microcystis* and *Dolichospermum* differ to *Oscillatoria* as they are less sensitive to high light intensities (high Zeu/Zmix ratio), because their ability to regulate their buoyancy enables them to find light conditions that are optimal for growth. In contrast, *Cylindrospermopsis*, dominate under relatively low or high irradiance conditions in tropical lakes (Havens, n.d). The dominance of cyanobacteria may persist for years if the ratio Zeu/Zmix is above the limit (Havens, n.d). This could be the case for many dams; however, light is not the only limiting variables affecting the dominance of cyanobacteria. In the Suswa River in India, the mean light limitation was found to be at a Zeu/Zmix ratio of 0.35, above the limit minimum Zeu/Zmix ratio of 0.20 needed for phytoplankton growth, indicating that light was sufficient for phytoplankton growth (Khanna et al., 2009).

Nutrients

In many lakes, the phytoplankton biomass attained is determined primarily by the pool of nutrients available for growth at the beginning of the each season (Thomas et al., 1996). The nitrogen (N) and phosphorus (P) in eutrophic lakes are used until growth is completed by the phytoplankton, and the exhaustion of either one of them places a final limit to cyanobacteria growth. The exhausted nutrient is the limiting nutrient in a lake. The ratio between TN:TP is used to determine the limiting nutrient. The widely used ratio is the Redfield ratio with TN:TP of 16:1. The Redfield ratio is often used as a benchmark to differentiate between N and P limitation (Guildford & Hecky, 2000; Kim et al., 2007; Redfield, 1958). The ratio assumes that cyanobacteria is N limited at TN:TP less than16:1, and that P limited at TN:TP greater than16:1 (Kim et al., 2007; Redfield, 1934, 1958). Nitrogen fixing (N-fixing) cyanobacteria (e.g. *Dolichospermum*, *Aphanizomenon*, and *Cylindrospermopsis*) are known to dominate in nitrogen limited water and non- nitrogen fixing cyanobacteria (e.g. *Microcystis*) dominate in phosphate limited water (Diehl, 2002; Dolman et al., 2012; Guildford & Hecky, 2000; Kim et al., 2007; Redfield, 1958).

Water salinity

Increased water salinity can also affect cyanobacteria growth in some lakes. A study by Barinova et al. (2009) on 34 lakes along the Karasu River in Kazakhstan revealed low diversity in phytoplankton communities due to high salinity. Costa et al. (2016) also reported the high dominance of cyanobacteria in shallow lakes of the Pantanal wetland in Brazil during the dry period due to an increase in salinity, where *Dolichospermum* was the major bloom forming genus.

Water level

Hydrological variables, such as changes in the water level due to drought related conditions, also influence the growth of cyanobacteria species (Paerl, 2018; Paerl et al., 2008; Paerl & Otten, 2013). For example, Maggiore Lake, Italy, Eymir Lake, Turkey, and Xeresa Lake, Spain were all found to experience cyanobacteria blooms due to drought-induced decreases in water levels (Bakker & Hilt, 2016). In the Three Gorges Reservoir, China, the water column stability was found to influence surface water Chl *a* concentration (Zhang et al., 2019). A study by Huang et al. (2020) reported that Chl *a* was decreasing with the increase in water level fluctuations. It has also been observed that the magnitude, spatial, and temporal distribution of *Microcystis* increased during dry periods and decreased during wet periods (Lehman et al., 2019). A study by Brasil et al. (2016) of 40 man-made lakes in a warm semi-arid region of Brazil, revealed that cyanobacteria communities increased in the dry season compared to the wet season.

The above environmental variables work in combination with each other influencing cyanobacterial growth, different variables impacting primary or secondary growth influence factors (Amblard, 1988; Ashton, 1985; Reynolds et al., 1984; Reynolds, 1980; Sommer, 1985; Sommer et al., 1986; Xiao et al., 2018).

2.6. Control and management of toxic cyanobacteria

Globally, many freshwater ecosystems are highly threatened by pollution and eutrophication causing cyanobacterial blooms (Jeppesen et al., 2017). Bozelli, (2019) described strategies and restoration measures aimed at improving water quality. Restoration should involve the use of ecologically friendly sound principles to return a lake or reservoir to the closest approximation of its original condition (Bozelli, 2019). Management of cyanobacteria involves monitoring, prediction, and mitigations to improve the lake and reservoir to enhance usage for water supply and recreational activities such as swimming and fishing. Mitigations are considered the most effective if nutrients are managed at the catchment level; however, this is typically complex due to factors such as the scale, multiple stakeholders / interest groups, and typically requires long-term investment.

Although ecological restoration remains a great challenge with more failure than success reported, some measures, such as phytoplankton control show notable promise as a strategy to manage the degradation of dams (Beklİoğlu, 1999; Bozelli, 2019; Gulati et al., 2008). Over the last four decades, different phytoplankton control measures were implemented, mainly in the developed world, to improve water quality by effective manipulation of nuisance phytoplankton blooms (Bozelli, 2019; Jeppesen et al., 2017). It is widely known that it is difficult to control phytoplankton blooms in warmer arid environments, as warmer drought conditions create symptoms conducive to eutrophication (Jeppesen et al., 2017; Jeppesen et al., 2011; Søndergaard et al., 2007; Yu et al., 2016). Bozelli (2019) stated that, unlike in the developed world, restoration is limited in the developing world, due to the costs involved and lack of experience. Although various restoration measures have been successfully being implemented in many regions, they are not simple and directly transferable due to differences in climatic conditions despite similar problems (Bozelli, 2019). Therefore, there is a need for modifications of existing methods and development of new alternative methods for reservoirs (Jeppesen et al., 2017).

The main purpose of phytoplankton control is to ensure the prevention of the proliferation of nuisance bloom forming phytoplankton species (Burford et al., 2019; Lürling et al., 2016; Lürling & Mucci, 2020; Pęczuła, 2012; Visser et al., 2016). According to Pęczuła (2012), phytoplankton control measures includes the following: (1) physical (screens/barriers, dredging, aeration, hydrologic manipulations, circulation, reservoir drawdown, ultrasound, etc.) (2) biological (plant extracts, floating islands and wetlands, biomanipulation, bacteria, natural predators, etc.), and (3) chemical controls (algaecides, hydrogen peroxide, copper sulphate, geochemical compound, coagulation, nutrient-binding clay, etc.). Pęczuła, (2012) stated that the success of any phytoplankton control measure is dependent on external nutrient control. The latter was verified in a study on Lake Washington, USA, where external nutrient reduction resulted in the reduction of toxic cyanobacterial blooms (Edmondson & Lehman, 1981). However, due to internal nutrient loading, the reduction of external nutrients alone did not lead to long-term water quality improvement, as observed in Alderfen Broad in Great Britain (Pęczuła, 2012). Nevertheless, external nutrient reduction is considered to be the most promising approach for control of cyanobacterial blooms, which should be considered as a first strategy (Burford et al., 2019; Lürling et al., 2016).

To suppress cyanobacterial blooms, epilimnion Solar Powered Circulation (SPC) technology has been used in some reservoirs (Kirke, 2000; Lürling & Mucci, 2020). Installations in Crystal Lake, Illinois (2 SPCs), East Gravel Lake, Colorado (3SPCs), and Lake Palmdale, California (6 SPCs), were all found to suppress cyanobacteria within the treatment zones, which resulted in the increase in density of green algae and diatoms following SPC initiation (Hudnell et al., 2010). These are, however, shallow reservoirs with depths less than 10 m, and surface areas greater than one square kilometer (Hudnell et al., 2010) which could be the reason for success of the SPCs installed. The SPC method is based on the principles of bio-manipulation, which treat the symptoms of eutrophication, not the cause of eutrophication (Hudnell et al., 2010). SPCs suppress cyanobacteria by creating mixing in the water column that inhibits the growth of the cyanobacterial cells and, consequently, improves water quality (Hudnell et al., 2010). While inhibiting cyanobacteria growth, SPCs create an environment that allows edible algae to grow and get consumed by zooplankton (Visser et al., 2016). The habitat disturbance of the cyanobacterial colonies are within the epilimnion zone of the dam, where the SPCs intake hose is set above the thermocline zone of the water column. One unit of SPC is designed to circulate about 40 000 L of water per minute.

2.7. Conclusion

Water quality of dams located downstream of populated human activities are deteriorating. To understand the water quality status of these essential sources requires assessing the different water quality parameters which is challenging over time and space without using water quality indices. The main concern is pollution caused by nutrients, which lead to eutrophication, often followed by the proliferation of toxic phytoplankton. The occurrence of toxic cyanobacteria in freshwater systems used as sources for drinking water is a major concern for many water utilities. Cyanobacteria are typically highly dynamic, both temporally and vertically in the water column, while numerous environmental and hydrological variables control their growth. Controlling their growth is thus complex and requires a range of strategies, especially in desert regions such as Namibia. To meet this challenge, it is necessary to develop a better understanding of the overall survival strategies employed by cyanobacteria, as well as their interaction with their environment. Filling the knowledge gaps in terms of water quality assessment using indexs, understanding the dynamic of phytoplankton, how there are affected by drought and how to control their growth will make valuable contribution in this regard.

CHAPTER 3: ASSESSING THE ADVERSE EFFECTS OF LAND USE ACTIVITIES ON THE WATER QUALITY OF SELECTED SUB-SAHARAN AFRICA RESERVOIRS USING A COMBINATION OF WATER QUALITY INDICES

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Declaration by the candidate

With regard to Chapter 3, the nature and scope of my contribution were as follows:

Nature of contribution	**Extent of contribution**
Conceptual design, fieldwork, experimental work, manuscript writing, data curation, formal analysis, methodology	80%

The following co-authors have contributed to Chapter 3:

Name	**Email address and institutional affiliation**	**Nature of contribution**	**Extent of contribution**
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Abstract

The present study aimed to employ different modified indices CCME WQI and CPI to determine the water quality status of the SWP and VB dams based on physicochemical and biological parameters. The data generated shows that the CCME WQI effectively determined nutrient, salinity, and particulate matter pollution in the two dams. Variables that exceeded the limits of the South African, Canadian, Australian, and New Zealand guideline levels were Cl, NTU, TP, OP, TN, NH_3, EC, Cu^{2+}, pH, and Chl *a* in the CCME WQIs. The CCME WQIs showed poor water quality conditions in both dams for 17 years. CPI also showed that the dams were polluted mostly by the selected variables, except for F, SO_4, and Mn. The occurrence of Cl concentration above the Target Water Quality Range (TWQR) for aquatic ecosystems (5µg/l) throughout the study period in both dams could be related to point source sewage pollution. It was evident from this study that the two applied indices showed similar results and, therefore, should be considered for water quality assessment in reservoirs found in the desert climate conditions.

3.1. Introduction

Globally, the quality of water resources located downstream of urban and agricultural areas has undergone degradation due to natural and anthropogenic activities (Espejo et al., 2012; Son et al., 2020; Vadde et al., 2018). Anthropogenic activities such as mining, livestock farming, production, and disposal of waste (industrial, municipal, agricultural); increased sediment runoff or soil erosion; and heavy metal pollution are being reported to affect water quality in reservoirs (Effendi, 2016; Jiang & Shen, 2003; Tanjung et al., 2019; Uddin et al., 2021) making the water not suitable for various users (Dabrowski et al., 2013, 2014; Castro-Roa & Agudelo 2014; Tanjung et al., 2019; Oberholster et al., 2021; Uddin et al., 2021). It is reported that many developing countries are challenged in protecting the water quality of reservoirs located downstream of urban and agricultural areas (Uddin et al., 2021). This is mainly due to malfunction wastewater infrastructure, poor/lack of sanitation facilities in informal settlements, lack of knowledge of water pollution, lack of enforcement, financial constraint, and lack of compliance to discharge of untreated sewage and mining effluent into river tributaries emptying into reservoirs.

Tributaries flowing into reservoirs bring along industrial, agricultural, and human waste, which compromises the reservoir's assimilation capacity to deal with pollutants. Once in the water bodies, the organic and inorganic pollutants increase the concentration of physicochemical parameters to exceed the required standard guideline limit (Tanjung et al., 2019; Tirkey et al., 2013; Yang et al., 2021) and in turn influence the concentration or growth of biological variables. The main consequence of pollutants emanating from anthropogenic activities in catchment areas is reported to be eutrophication caused by excessive nutrient loads (Hamilton et al., 2016; Paerl & Otten, 2013; Paerl et al., 2016; Xu et al., 2015). This is a major concern to many water utility managers, as the outcome affects water quality and increases costs of treatment.

The water quality management in the surface water bodies requires testing physicochemical and biological parameters. However, understanding the water quality status of a particular reservoir by assessing the different measured water quality parameters over time and space is challenging (Uddin et al., 2021; Yang et al., 2021). Therefore, to effectively understand the change in water quality status of surface water bodies, environmental tools such as WQI models have been used in different water bodies (Castro- Roa & Agudelo, 2014; Finotti et al., 2015; Jiang & Shen, 2006; Kumar et al., 2020; Matta et al., 2017; Mishra et al., 2016; Oberholster et al., 2021; Oni & Fasakin, 2016; Son et al., 2020; Uddin et al., 2021). The WQI is reported to effectively determine the changes in the water quality status of rivers, groundwater, lakes, reservoirs, wetlands, and marine. These tools use aggregation to convert multiple water quality data into one value (Kachroud et al., 2019; Tirkey et al., 2013; Uddin et al., 2021). Since its development in the 1960s, it has become a popular tool due to its generalized structure and ease of use (Kachroud et al., 2019; Uddin et al., 2021).

The WQIs are associated with the following benefits: (a) reduction in the number of parameters required to compare water quality for use, (b) provision of a single number that represents overall water quality, (c) identification of space and time dynamics in the quality of water, (d) provision of assurance on the safety of a water body to users, and (e) are one of the most simplified methods of communicating water quality information to the large public and to legislative decision-makers (Adelagun et al., 2021). Despite the benefits of WQI, it is however associated with challenges such as (a) being not an absolute measure of degree of pollution or the actual water quality, (b) lack of precision and accuracy in classification technique of importance of evaluation of parameters, (c) inefficiency in dealing with uncertainty and subjectivity in a complex environmental issue, (d) lack of a uniform method for measuring water pollution involving biological parameter, and (e) inadequate to transfer complex environmental data into information (Etim et al., 2013).

However, some WQIs are usually developed based on site-specific guidelines for a particular region and are therefore not generic. Although they produce uncertainty in converting large amounts of water quality data into a single index (Kachroud et al., 2019; Uddin et al., 2021), they are widely used to determine the water quality suitability for intended use. The first index was developed by Horton in 1965 which presents a mathematical method of calculating a single value to represent water quality from 10 multiple water quality parameters, such as dissolved oxygen, pH, coliforms, EC, alkalinity, and chloride (Bilgin, 2018). This was followed with numerous WQIs which were proposed by different scientists and organizations for assessment of water quality (Edwin & Murtala, 2013).

The most used WQI in assessing the water quality status of surface water bodies are Horton index, National sanitation Foundation WQI (NSF-WQI), Scottish Research development Department (SRDD) index, Canadian Council of Ministers of the Environment (CCME) WQI, Bascaron index (BWQI), Fuzzy interface system (FIS), Malaysian water quality index (MWQI) and West Java WQI (WJ-WQI) (Kachroud et al., 2019; Tirkey et al., 2013; Uddin et al., 2021), eutrophication index (EI), trace metal pollution index (TPI), organic pollution index (OPI), and CPI (Son et al., 2020), National Sanitation Foundation Environmental Sanitation Technology Company of the State of Sao Paulo (CETESB) WQI (Finotti et al., 2015).

Globally, among the WQIs, the CCME WQI is widely used in Canada either as standalone or combination of other WQIs like CPI. The CCME was developed for the Canadian water by the CCME in 2001 (Bilgin, 2018) and its applications worldwide have been due to its flexibility to suit different conditions. The CCME is very flexible, enabling user from different countries to use site-specific standards and concerning water quality parameters (Haider et al., 2021; Haile & Gabbiye, 2022; Khan et al., 2005). The CCME WQI combines all water quality parameters and provides a readily understood description of the water quality (Davies, 2006). The CCME WQI was developed based on a combination of factors, namely, scope, frequency, and amplitude of water quality variables, into one index (Bilgin, 2018; Davies, 2006). Adelagun et al. (2021) mentioned that over the years, many countries have accepted the use of the CCME as a WQI for surface and groundwater quality monitoring and assessment.

For example, the River Asa, in Kwara State, Nigeria, water quality status was assessed using the CCME WQI (Adelagun et al., 2021). The CCME WQI ranked the river to have marginal to poor water quality (Edwin & Murtala, 2013). Bilgin (2018) evaluated the water quality of the Coruh River Basin in Turkey using the CCME WQI and reported poor, marginal to fair classifications of water quality at different sites. The river basin water quality was reported to be degraded and threatened by wastewater from anthropogenic activities (Bilgin, 2018). The Tafna Basin, located in the north-western Algeria, water quality status was evaluated using the CCME and other 10 WQIs (Hamlat et al., 2017). The results of the CCME classified the basin in marginal water quality indicating threatened water quality, and the results were similar to that of the British Columbia WQI (Hamlat et al., 2017).

Davies (2006) used the CCME WQI to evaluate the water quality status of the Qu'Appelle River Valley, Saskatchewan, in Canada, before and after the removal of phosphorus from the wastewater. The CCME reported a slight improvement in the water quality along the river after the installation of tertiary clarifiers for the removal of phosphorus from wastewater effluent (Davies, 2006). Davies (2006) reported that urban activitie and nonpoint pollutants are the causes for the water quality deterioration of the Qu'Appelle River Valley. Lumb et al. (2006)

applied the CCME WQI to assess the water quality status of the Mackenzie-Great Bear sub-basin in Canada and reported turbidity and trace metal pollution. Espejo et al. (2012) also applied the CCME WQI to assess the water quality status of four watersheds (Huasco basin, Elqui basin, Choapa basin, and Limari basin) in Norte Chico zone, North-Central Chile, and reported fairly good water quality overall more especially in the Limari basin. In these applications, the suspended Chl *a* was not considered as a water quality parameter to represent the phytoplankton biomass.

CPI is one of the widely used indices to assess the level of pollution of rivers, lakes, wetland, and groundwater (Son et al., 2020). Son et al. (2020) applied a combination of indices (WQI, CPI, OPI, EI, and TPI) to assess the water quality of the Cau River and reported serious organic pollution and eutrophic conditions downstream. Tang et al. (2011) applied the comprehensive pollution index to assess the water quality status of Qilu Lake, China. Matta et al. (2017) used the CPI to assess the water quality status of the Henwal River, India, and reported moderate and severe pollution at different sampling sites. Mishra et al. (2016) applied a combination of indices (CPI, OPI, EI, and TPI) to assess the water quality of the Surha Lake in India and reported that the lake is eutrophic and moderately polluted. Recently, Oberholster et al. (2021) used a modified pollution index (PILD) to determine the water quality status of the Loskop Dam, South Africa, and reported nutrient enrichment and heavy metal pollution. Catchment land use activities such as agriculture, dysfunctional wastewater infrastructure, and mining are being reported as the root causes of water quality deterioration. The application of these tools in reservoirs found in desert climates could be helpful for the water utility manager to monitor the changes in water quality for effective treatment and management.

Although these tools are being modified and applied in various reservoirs, the applications in reservoirs found under desert climate conditions are lacking. These tools are reported to be meaningful in understanding changes in water quality of surface water bodies. However, the majority of the reported applications of the water quality and pollution indices have not considered suspended Chl *a* as a parameter (Castro-Roa & Agudelo, 2014; Matta et al., 2017; Mishra et al., 2016; Oberholster et al., 2021; Son et al., 2020; Tanjung et al., 2019). Many studies have used Chl *a* as an indicator of algal biomass and trophic status (Huang et al., 2020; Zhang et al., 2019). The study aimed to assess the water quality status of the ephemeral river-connected reservoirs receiving a mixture of pollutants over space and time using the combination of CCME WQI and CPI. The objectives of the study are (a) to determine the status of the water quality of the two ephemeral river-connected reservoirs using two pollution indices and (b) to establish if these tools can be used by utility managers as a decision support tool in monitoring changes in the water quality of ephemeral river reservoirs found under desert climate conditions. The two reservoirs only received inflows during the rainy seasons. The reservoirs are designed without any environmental flow consideration, and also to keep the water for two rainy seasons. Overflow in the two reservoirs is only experienced when excess inflows are received. The SWP Dam water is transferred to VB Dam to minimize evaporation losses due to its smaller surface area, and also to bring water closer to the VB Water Treatment Plant. Sirunda et al. (2021b) reported that both dams experience cyanobacteria blooms mainly during autumn and spring seasons.

3.2. Materials and methods

3.2.1. Study area and land use activities

The two man-made dams are located in a warm semiarid region in central Namibia. The VB(21°59′59.27″ S, 16°58′54.76″ E) and SWP Dams (22°12′44.31″ S, 16°31′44.97″ E) are monomictic man-made lakes that are stratified throughout the year and only experience overturn or mixing during the winter season (Sirunda et al., 2021b) (Figure 3.1). The dams are situated in the Omaruru-Swakop Basin, in the Upper-Swakop sub-basin with a desert climate, which is characterized by large differences in day and night temperature, erratic rainfall, frequent droughts, and high evapotranspiration (Table 3.1).

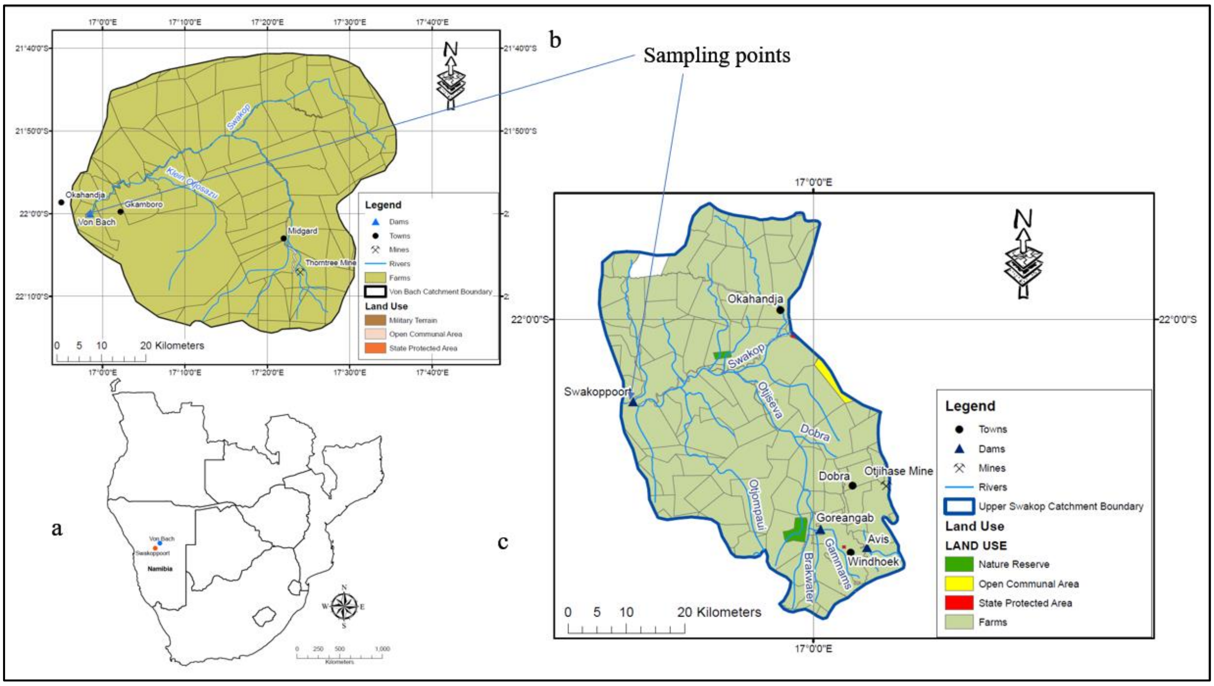


Figure 3.1. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset, indicating a location of Namibia within Africa; b the location of the VB Dam in its sub-catchment area with available land use activities; c the location of the SWP Dam in its sub catchment area with available land-use activities.

Table 3.1. General features of the two dams found on the Swakop River.

Dam features	**Dams**	
	VB	SWP
Basin	Omaruru – Swakop	Omaruru – Swakop
Sub-basin	Upper-Swakop	Upper-Swakop
Capacity (Mm^3)	48.56	63.48
Max. Depth (m)	29	30
Evapo. Losses (mm/a)	2254	2275
Ann.rainfall (mm/a)	370	350
Surface area (FSC) (km^2)	4.89	7.81
Catchment area size (km^2)	3.1	5.5

The land use activities in the VB Dam's upper catchment comprise commercial farms for livestock and game farming, settlements, mines, and lodges (Figure 3.1). The land use activities in the upper catchment of the SWP Dam are mainly open defecations from informal settlements, Namib Poultry, Okapuka Tannery, Meatco Feedlot, Ujams Ponds, Goreangab Dam, Gammams Wastewater Treatment Plant, New Goreangab Reclamation Plant, industries (Namibia Diaries, Nambia Breweries), Kupferberg Landfill Site, lodges, mines (Otjihase Copper Mine), coal power plant (Van Eck Power Station), and sewerage ponds (Figures 3.1 and 3.2).

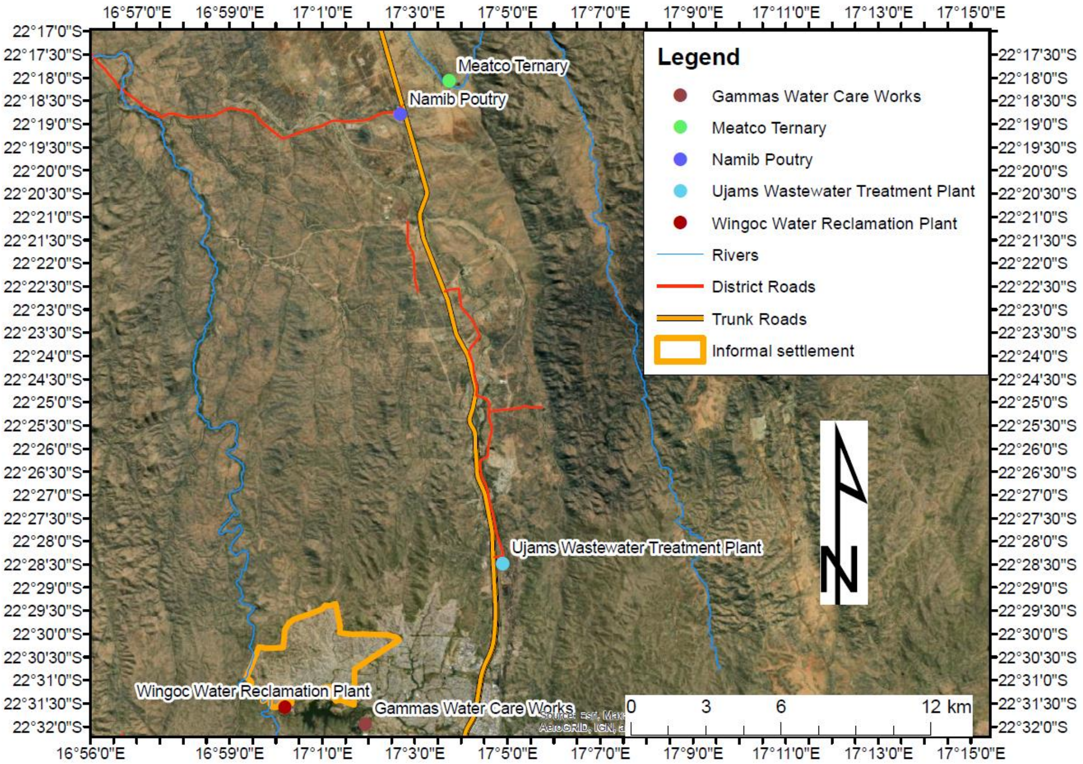
3.2.2. Field sampling and analysis

Historical data were obtained from an established sampling site located near the dam wall in each dam for a period of 17 years from 2003 to 2019 on a monthly interval using the dip sampling method (Figure 3.1). A dip sampling method involves the dipping of a Von Dorn 5L water sampler into the water to retrieve samples at different depths, which are then transferred to the appropriate acid-washed sample containers (Burns et al., 2000). The water sampling was always performed between 09:30 and 14:00 using a designated boat at different depths, which changes due to a decrease in water volume in both dams. Water samples were collected at selected different water depths. In the VB Dam: total phosphate (TP) n = 525, ortho-phosphate (OP), total nitrogen (TN) n = 525, chlorophyll *a* (Chl *a*) n = 525, turbidity n = 525, chloride (Cl) n = 139, electrical conductivity (EC) n = 139, fluoride (F) n = 139, sulfate (SO4) n = 139, manganese (Mn) n = 139, copper (Cu^{2+}) n = 139, and ammonia (NH_3) n = 525, while in the SWP Dam: total phosphate (TP) n = 383, total nitrogen (TN) n = 383, chlorophyll *a* (Chl *a*) n = 383, turbidity n = 383, chloride (Cl) n = 157, electrical conductivity (EC) n = 157, fluoride (F) n = 157, sulfate (SO_4) n = 157, manganese (Mn) n = 157, copper (Cu^{2+}) n = 157, and ammonia (NH_3) n = 383. All field samples were stored in plastic and glass bottles kept in a mobile icebox and analyzed in the laboratory within 24 h. The cooler boxes were transported to the laboratory and kept in a refrigerator at 4 °C.

3.2.3. Water quality analyses

The turbidity of the water samples was analyzed using the turbidity meter and the nephelometric method (Part 2130 B) in the laboratory. The pH of the water samples was analyzed using a pH meter method (Part 4500 H B). The ortho and total phosphate in the water samples were analyzed using the ascorbic acid method (Part 4500 P E), and a spectrophotometer with infrared phototubes was used as colorimetric equipment. The total nitrogen in the water samples was analyzed using the cadmium reduction method (Part 4500 NO3-E). In the analysis of total nitrogen, a reduction column was used as an apparatus in combination with a spectrophotometer. The ammonia in the water samples was analyzed using a titrimetric method (Part 4500-NH_3 C). The electrical conductivity of the water samples was analyzed using the conductivity method (Part 2510 B). Chloride was analyzed using the automated ferricyanide method (Part 4500-Cl- E). Suspended Chl *a* contained in the water samples was analyzed using the spectrophotometric determination method (Part 10,200 H). The fluoride of the water samples was analyzed using the method (Part 4500 F). The sulfate of the water samples was analyzed using gravimetric methods (4500-SO42 C and D) while the manganese of the water samples was analyzed using the method (Part 3500 Mn). Copper of the water samples was analyzed using the method Part 3500-Cu. All the water quality parameters were analyzed according to the methods described in the American Public Health Association (APHA, 1998).

a.



b.

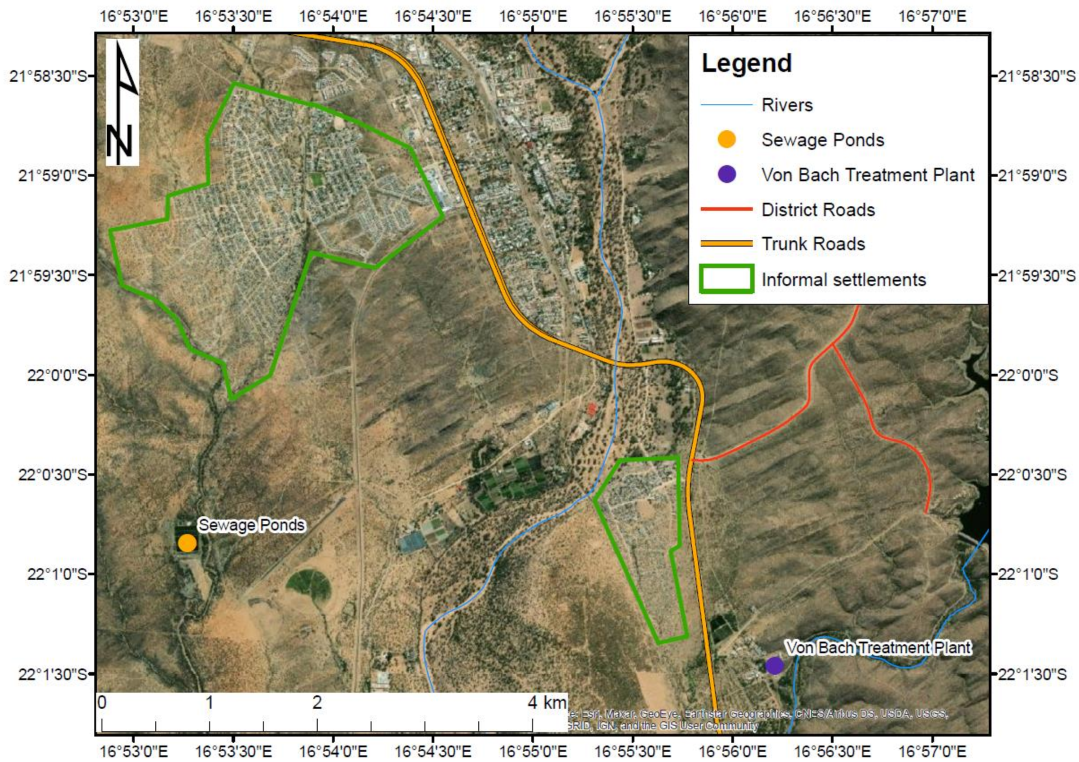


Figure 3.2. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset, indicating a location of land use activities in the upper catchment of the

SWP Dam, below the Capital, Windhoek; b the location of land use activities in the upper catchment of the SWP Dam, below the Town of Okahandja.

3.2.4. Pollution Index

A combination of two water quality indices was applied in this study, and three water quality standards guidelines (ANZECC, 2000; DWAF, 1996; CCME, 2012) were used to determine if the selected water quality parameters exceed the limit or not. The first was the (a) CCME WQI. The CCME WQI was originally developed as the Canadian Water Quality Index (CWQI) (Lumb et al., 2006; Uddin et al., 2021). The CCME WQI, although it was developed in Canada, has been applied to a wide range of surface water bodies due to its ease of application and because it provides flexibility in choosing the water quality parameters to be included in the model (Uddin et al., 2021). In the review by Uddin et al. (2021), a range of CCME model applications for the assessment of surface (river or marine) water quality in various regions of the world was reported. Canadian Council of Ministers of the Environment, (2017) reported that the CCME WQI has been adopted for use in the United Nations Environment Programme and has been used to rate water quality in Morocco, Argentina, Japan, Republic of Korea, Belgium, Poland, Switzerland, South Africa, India, Pakistan, and the Russian Federation. Furthermore, the CCME WQI was also formed as the basis for an Egyptian WQI (Canadian Council of Ministers of the Environment, 2017; Khan et al., 2005).

The CCME WQI comprises three factors and is well documented (Lumb et al., 2006; Uddin et al., 2021). The CCME WQI was developed in 1990 for use in Canadian and Australian water. The index was developed as a communication tool to transform complex water quality data into understandable descriptions, maximum usage of monitoring data, consistent use, applies to all beneficial uses, and amendable to multiple reporting scales. It takes a minimum of 4 water quality parameters and a maximum of 47 parameters (Kachroud et al., 2019; Tirkey et al., 2013). The number of water quality parameters selected is a function of what is to be assessed. It is recommended that a set of one (core set of parameters) (e.g. metals, nutrients) can possibly be used when there is only one use to be assessed (Tri-Star Environmental Consulting, 2012). However, the selected parameter needs to be relevant to the stressor on the system for the CCME WQI calculated to be relevant and accurate (Canadian Council of Ministers of the Environment, 2017). Using too many parameters in the CCME WQI calculation will reduce the importance of any one parameter, while too few parameters will increase the importance

(Tri-Star Environmental Consulting, 2012). It is for this reason that a minimum number and maximum numbers of water quality parameters are considered when applying the CCME WQI. In the current study, the index was applied with the inclusion of suspended chl a as the indicator of phytoplankton biomass. The index is based on three steps. The first step was to calculate the scope (F1) which represents the percentage of water quality variables that do not meet their guidelines at least once during the time period under consideration (for each month for the data collected for 17 years) ("failed variables"), relative to the total number of variables measured using Eq. 1. The selected water quality variables used in the calculation were TP, TN, EC, Cl, NTU, OP, F, SO_4, Mn, Cu^{2+}, Chl *a*, and pH. The surface water variables were selected based on their association with toxic phytoplankton blooms and the land use activities in the catchment areas of the two dams:

$$F_1 = \left(\frac{Number\ of\ failed\ variables}{Total\ number\ of\ variables}\right) \times 100 \qquad (1)$$

Step 2 was to compute the frequency (F2). F2 represents the percentage of individual tests that do not meet the guidelines ("failed tests"). F3 assesses the frequency with which the guidelines are not met:

$$F_2 = \left(\frac{Number\ of\ failed\ tests}{Total\ number\ of\ tests}\right) \times 100 \qquad (2)$$

Step 3 is the final step for calculating the amplitude (F3). F3 is calculated in three steps. Amplitude assesses the amount by which the guidelines are not met. F3 indicates the amount by which the failed test values do not meet their guidelines and is calculated in 3 steps.

1. The number of times by which an individual concentration is greater than (or less than, when the objective is a minimum) the objective is termed an "excursion" and is expressed as follows. When the test value must not exceed the objective:

$$excursion_i = \left(\frac{Failed\ Test\ Value_i}{Objective_j}\right) - 1 \qquad (3a)$$

When the test value must not fall below the guideline:

$$excursion_i = \left(\frac{Objective_j}{FailedTestValue_i}\right) - 1 \qquad (3b)$$

2. The collective amount by which individual tests are out of compliance is calculated by summing the excursions of individual tests from their objectives and dividing by the

total number of tests (both those meeting objectives and those not meeting objectives). This variable, referred to as the normalized sum of excursions, or nse, is calculated as

$$nse = \frac{\sum_{i=1}^{n} excursion_i}{\#\ of\ tests} \qquad (4)$$

3. F3 is then calculated by an asymptotic function that scales the normalized sum of the excursions from objectives (nse) to yield a range between 0 and 100:

$$F_3 = \left(\frac{nse}{0.01nse + 0.01}\right) \qquad (5)$$

Once the factors have been obtained, the index itself can be calculated by summing the three factors as if they were vectors. The sum of the squares of each factor is therefore equal to the square of the index. This approach treats the index as a three dimensional space defined by each factor along one axis. With this model, the index changes in direct proportion to changes in all three factors.

The CCME Water Quality Index (CCME WQI):

$$CCMEWQI = 100 - \left(\frac{\sqrt{{F_1}^2 + {F_2}^2 + {F_3}^2}}{1.732}\right)$$

The divisor 1.732 normalizes the resultant values to range between 0 and 100, where 0 represents the “worst” water quality and 100 represents the “best” water quality.

The assignment of CCME WQI values to different categories is a somewhat subjective process and demands expert judgment and public’s expectations of water quality. The water quality is ranked in the following 5 categories:

1. Excellent: (CCME WQI values 95–100)
2. Good: (CCME WQI values 80–94)
3. Fair: (CCME WQI values 60–79)
4. Marginal: (CCME WQI values 45–59).
5. Poor: (CCME WQI values 0–44)

The second WQI which was applied was the (b) CPI method to evaluate water quality qualitatively. Many authors have applied the CPI to assess the water quality status of surface water bodies (Matta et al., 2017; Mishra et al., 2016; Son et al., 2020; Tang et al., 2011). Similar water quality parameters were selected as those used in the CCME WQI, with the inclusion of Chl *a* as a phytoplankton indicator. The CPI is evaluated by the following equation:

$$CI = \frac{1}{n}\sum_{i=1}^{n} PI$$

where CPI = comprehensive pollution index for each parameter measured, n = number of monitoring parameters, and PI = Pollution Index number for each
parameter. The PI for each parameter is calculated according to the following equation:

$$PI = \frac{Ci}{Si}$$

where C*i* is the measured concentration of the i^{th} parameter. S*i* is the water quality standard of the i^{th} parameter, and n is the number of parameters considered.
The CPI is classified into five categories as follows (Son et al., 2020):

i. Category 1: CPI from 0 to 0.20 (clean);
ii. Category 2: CPI from 0.21 to 0.40 (sub clean);
iii. Category 3: CPI from 0.41 to 1.00 (slightly polluted);
iv. Category 4: CPI from 1.01–2.00 (moderately polluted); and
v. Category 5: CPI >2.01 (heavily polluted).

3.2.5. Statistical analysis

To determine the influences of the changes in physicochemical parameters on Chl *a* as an indicator of algal biomass, the Pearson correlation analysis was used to determine the relationship between Chl *a* and the physicochemical variables.

3.3. Results

3.3.1. Physicochemical and biological parameters

During the 12-month period over 17 years, the average turbidity value ranges between 6.51 and 60NTU in VB and 12.86 and 25.46NTU in SWP Dam (Tables 3.2 and 3.3). TP varied between 0.08 and 0.34mg/l in VB Dam, while in the SWP Dam, it was 0.16 to 0.74mg/l (Tables 3.2 and 3.3). The months of February and March recorded higher concentrations of TP in both dams over 17 years (Tables 3.2 and 3.3). The average OP was between 0.02 and 0.07mg/l in the VB Dam, while in the SWP Dam, it was between 0.09 and 0.18mg/l throughout the study period. The NH_3 concentration varied through the study period in both dams, with higher values in the summer months (Nov, Dec, Jan, Feb, March) and autumn months (April and May). The electrical conductivity concentration varied through the study period, with higher concentrations in the SWP Dam ranging from 73.54mS/m in March to 182mS/m in December, while in VB Dam from 31.17mS/m in May to 43.08mS/m November.

The Cl concentration ranged from 17.87 to 35.83mg/l in VB Dam, while in SWP Dam, it was from 88.56 to 142.27mg/l. The F, pH, SO_4, Mn, and Cu^{2+} concentrations varied through the study period in both dams. The suspended Chl *a* concentration was higher in SWP Dam compared to VB Dam. Based on Tables 1 and 2, the Chl *a* in VB Dam was higher in spring months (Sept. and Oct.), followed by autumn months (April). Similarly, in the SWP Dam, the autumn months (April) followed by winter months (Jul and Aug) recorded a higher concentration of Chl *a*.

Table 3.2. The average physicochemical and biological parameters measured from January to December during 2003-2019 in the VB Dam.

		Months											
Parameter	***N***	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Turbidity (NTU)	525	60.19±94.28	22.22±30.33	22.44±41.25	14.65±15.24	12.15±18.50	8.53±4.32	7.19±7.72	7.64±11.43	12.17±23.54	10.18±11.68	5.95±3.44	6.51±4.36
TP (mg/l)	525	0.16±0.12	0.29±0.77	0.32±0.62	0.23±0.28	0.12±0.09	0.18±0.32	0.19±0.25	0.21±0.30	0.16±0.18	0.34±0.56	0.13±0.13	0.08±0.05
OP (mg/l)	525	0.05±0.06	0.07±0.12	0.06±0.12	0.05±0.05	0.04±0.03	0.02±0.02	0.03±0.02	0.02±0.02	0.03±0.03	0.06±0.03	0.03±0.03	0.03±0.05
TN (mg/l)	525	1.57±1.13	1.65±0.79	1.49±0.84	1.41±0.87	1.56±1.01	1.554±0.86	1.19±0.76	1.23±0.76	1.60±0.81	1.51±0.88	1.64±1.26	1.39±0.73
NH3 (mg/l)	525	0.48±0.58	0.44±0.46	0.53±0.64	0.55±0.66	0.36±0.36	0.27±0.23	0.17±0.19	0.09±0.10	0.08±0.17	0.11±0.15	0.18±0.23	0.22±0.26
EC (mS/m)	139	36.52±15.51	38.47±12.40	38.05±12.74	32.28±10.12	33.46±10.41	31.17±10.41	34.69±13.95	32.37±13.96	40.77±15.01	40.71±18.56	39.33±19.07	43.08±22.24
Cl (mg/l)	139	28.00±23.74	31.50±19.11	31.17±17.46	18.32±15.40	22.75±14.43	17.87±16.44	19.86±17.64	19.83±19.59	31.47±21.54	31.33±27.23	29.14±29.85	35.83±32.78
F (mg/l)	139	0.26±0.09	0.26±0.07	0.23±0.08	0.22±0.07	0.21±0.07	0.24±0.07	0.24±0.13	0.25±0.10	0.24±0.07	0.27±0.07	0.35±0.09	0.28±0.04
pH	139	7.97±0.23	7.99±0.50	7.91±0.40	7.91±0.36	7.78±0.26	7.69±0.31	7.38±0.34	8.10±0.40	8.17±0.67	8.18±0.68	8.05±0.50	8.14±0.23
SO4 (mg/l)	139	18.82±17.84	22.83±11.69	25.44±12.24	14.19±9.90	17.50±7.06	13.22±9.80	15.29±12.51	14.06±12.88	23.07±13.21	23.73±16.11	20.21±19.14	23.67±19.88
Mn (mg/l)	139	0.02±0.01	0.05±0.14	0.02±0.01	0.07±0.10	0.03±0.01	0.03±0.03	0.01±0.00	0.02±0.01	0.02±0.02	0.02±0.01	0.03±0.03	0.03±0.00
Cu^{2+} (mg/l)	139	0.03±0.01	0.02±0.01	0.02±0.01	0.02±0.01	0.03±0.03	0.02±0.01	0.03±0.01	0.02±0.01	0.02±0.01	0.03±0.02	0.02±0.01	0.03±0.01
Chl *a* (µg/l)	525	10.05±15.13	13.86±24.11	14.16±20.22	32.02±85.01	13.16±11.21	13.06±10.09	18.39±4059	24.18±59.24	50.29±163.82	29.89±66.96	19.47±34.36	16.72±36.39

Table 3.3. The average physicochemical and biological parameters measured from January to December during 2003-2019 in the SWP Dam.

		Months											
Parameter	**N**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Turbidity (NTU)	383	25.46±21.92	15.41±8.89	16.47±20.06	25.06±61.10	23.76±55.94	12.86±15.80	17.25±31.32	20.31±29.88	23.03±35.97	22.71±22.55	20.86±19.73	14.07±7.87
TP (mg/l)	383	0.23±0.17	0.74±2.70	0.34±0.28	0.33±0.44	0.26±0.11	0.29±0.37	0.38±0.38	0.44±0.66	0.26±0.22	0.28±0.27	0.16±0.09	0.19±0.05
OP (mg/l)	383	0.09±0.10	0.16±0.16	0.14±0.13	0.15±0.12	0.18±0.12	0.11±0.09	0.09±0.08	0.08±0.07	0.11±0.12	0.15±0.14	0.09±0.08	0.11±0.06
TN (mg/l)	383	2.07±0.76	2.33±1.27	2.30±1.27	1.88±0.74	2.31±1.01	2±0.77	2.25±1.22	1.85±0.87	1.91±0.92	1.62±0.84	2.15±0.91	2.21±1.42
NH3 (mg/l)	383	0.22±0.41	0.53±0.76	0.45±0.67	0.38±0.49	0.28±0.30	0.22±0.24	0.17±0.21	0.22±0.24	0.11±0.16	0.20±0.26	1.65±9.15	0.22±0.43
EC (mS/m)	157	176.47±91.80	146.31±169.22	73.54±48.45	97.67±51.37	95.9±54.70	83.98±44.65	78.28±51.77	132.29±162.91	89.26±41.76	111.67±53.28	110.46±62.33	182.07±74.52
Cl (mg/l)	157	78.5±94.31	83.86±69.14	87.07±63.47	123.42±80.78	113.91±74.34	100.71±62.98	89.5±77.41	107.14±83.52	105.38±62.58	142.27±78.98	129.31±82.15	88.55±99.30
F (mg/l)	157	0.2±0.20	0.25±0.16	0.28±0.12	0.35±0.11	0.33±0.13	0.3± 0.12	0.31±0.16	0.33±0.16	0.3±0.09	0.44±0.23	0.39±0.26	0.24±0.26
pH	157	8.73±0.50	8.46±0.54	8.58±0.71	8.59±0.57	8.33±0.48	7.01±2.78	8.11±0.32	7.69±2.17	8.57±0.40	8.46±1.46	8.58±0.57	8.74±0.31
SO4 (mg/l)	157	58.64±74.51	66.57±61.13	63.29±58.01	95.33±71.87	90.54±67.64	70.64±57.31	63.14±59.42	83.88±76.61	82.54±57.50	109.27±62.54	93.15±69.15	68.44±79.26
Mn (mg/l)	157	0.03±0.04	0.04±0.06	0.02±0.01	0.03±0.02	0.03±0.02	0.03±0.03	0.02±0.02	0.01±0.01	0.01±0.01	0.02±0.03	0.02±0.02	0.02±0.01
Cu^{2+}(mg/l)	157	0.02±0.03	0.02±0.01	0.04±0.07	0.03±0.03	0.03±0.01	0.02±0.02	0.03±0.02	0.04±0.03	0.02±0.01	0.03±0.02	0.02±0.01	0.02±0.04
Chl *a* (µg/l)	383	45.76±40.36	30.04±22.80	44.35±41.38	94.59±132.17	69.46±106.34	43.60±49.35	64.14±119.82	82.06±101.68	48.72±52.03	65.22±62.48	62.12±41.74	67.14±51.61

3.3.2. Water quality index analyses of the two dams

From the number of water quality parameters selected to compute the WQI of the two dams, only 10 out of 13 parameters in both dams were above the selected standards of the Department of Water Affairs and Forestry (DWAF) (1996) legislation and CCME (2012) and ANZECC (2000) guidelines (Tables 3.4 and 3.5). These variables were not meeting the water quality objectives. NTU, TP, OP, TN, NH_3, EC, pH, Cu^{2+}, and Chl *a* were found to exceed the standards set by the TWQR for aquatic ecosystems according to DWAF (1996) and CCME (2012) for aquatic ecosystems and the ANZECC (2000) guidelines for the conservation of aquatic areas throughout the study period. Only F, SO_4, and Mn were found to be within the standard guideline limits in the two dams (Tables 3.4 and 3.5).

Table 3.4. The selected water quality parameters in the VB Dam against the surface water quality standards limits from 2003-2019.

Parameters	NTU**	TP***	OP***	TN***	NH_3 *	EC***	Cl	F**	pH**	SO_4	Mn**	Cu^{2+}*	Chl*a****
Standard	**5**	**0.025**	**0.01**	**1**	**0.007**	**30**	**0.005**	**0.75**	**6-8**	**200**	**0.18**	**0.0003**	**5**
Jan	60	0.16	0.05	1.57	0.48	36.52	28.00	0.26	7.97	18.82	0.02	0.03	10.05
Feb	22	0.29	0.07	1.65	0.44	38.47	31.50	0.26	8.00	22.83	0.05	0.02	13.86
Mar	22	0.32	0.06	1.49	0.53	38.04	31.17	0.23	7.91	25.44	0.02	0.02	14.16
Apr	15	0.23	0.05	1.41	0.54	32.28	18.32	0.22	7.91	14.19	0.07	0.02	32.02
May	12	0.11	0.04	1.56	0.36	33.46	22.75	0.21	7.78	17.50	0.03	0.03	13.16
Jun	9	0.18	0.02	1.55	0.27	31.17	17.87	0.24	7.69	13.22	0.03	0.02	13.06
Jul	7	0.19	0.03	1.19	0.17	34.69	19.86	0.24	7.38	15.29	0.01	0.03	18.39
Aug	8	0.21	0.02	1.23	0.09	32.37	19.83	0.25	8.10	14.06	0.02	0.02	24.18
Sep	12	0.16	0.03	1.61	0.08	40.77	31.47	0.24	8.17	23.07	0.02	0.02	50.29
Oct	10	0.34	0.03	1.51	0.11	40.71	31.33	0.27	8.18	23.73	0.02	0.02	29.89
Nov	6	0.13	0.03	1.64	0.18	39.33	29.14	0.35	8.05	20.21	0.03	0.02	19.47
Dec	7	0.08	0.03	1.39	0.22	43.08	35.83	0.28	8.14	23.67	0.01	0.03	16.72

TWQR Target water quality range for aquatic ecosystems according to DWAF (1996)* or CCME (2012)** for aquatic ecosystems and the ANZECC (2000) guidelines for conservation aquatic areas ***

Table 3.5. The selected water quality parameters in the SWP Dam against the surface standards limits from 2003-2019.

Parameters	NTU**	TP***	OP***	TN***	NH_3*	EC***	Cl	F**	pH**	SO_4	Mn**	Cu^{2+}*	Chl*a****
Standard	**5**	**0.025**	**0.01**	**1**	**0.007**	**30**	**0.005**	**0.75**	**8**	**200**	**0.18**	**0.0003**	**5**
Jan	25.5	0.23	0.09	2.07	0.22	176.47	78.50	0.20	8.73	58.64	0.03	0.02	45.76
Feb	15.4	0.74	0.16	2.34	0.53	146.31	83.86	0.25	8.46	66.57	0.04	0.02	30.04
Mar	16.5	0.34	0.14	2.30	0.45	73.54	87.07	0.28	8.58	63.29	0.02	0.04	44.35
Apr	25.1	0.33	0.15	1.88	0.38	97.67	123.42	0.35	8.59	95.33	0.03	0.03	94.60
May	23.8	0.26	0.18	2.31	0.29	95.90	113.91	0.33	8.33	90.55	0.02	0.03	69.46
Jun	12.9	0.30	0.11	2.00	0.22	83.98	100.71	0.30	7.01	70.64	0.03	0.02	43.60
Jul	17.3	0.37	0.09	2.25	0.17	78.29	89.50	0.31	8.12	63.14	0.02	0.03	64.14
Aug	20.3	0.44	0.09	1.85	0.22	132.29	107.14	0.33	7.69	83.86	0.01	0.04	82.06

Sep	23.0	0.26	0.11	1.91	0.11	89.26	105.38	0.30	8.57	82.54	0.01	0.02	48.72
Oct	22.7	0.28	0.15	1.62	0.20	111.67	142.27	0.44	8.46	109.27	0.02	0.03	65.23
Nov	20.9	0.16	0.09	2.15	1.65	110.46	129.31	0.39	8.58	93.15	0.02	0.02	62.12
Dec	14.1	0.19	0.11	2.21	0.22	182.08	88.56	0.24	8.74	68.44	0.02	0.02	67.14

TWQR Target water quality range for aquatic ecosystems according to DWAF (1996)* or CCME (2012)** for aquatic ecosystems and the ANZECC (2000) guidelines for conservation aquatic areas ***

3.3.3. CCME WQI

The scope (F1) for variables not meeting the objectives calculated was 77% in both dams (Table 3.6). The scope assesses the extent of compliance with water quality guidelines over the period of the study. The frequency (F2) of the individual variables which do not meet the guidelines was 72% in VB Dam and 76% in the SWP Dam (Table 3.6). More water quality parameters were not meeting the limited guideline concentrations in the SWP Dam compared to the VB Dam. The amount by which the guidelines were not met by the selected variable (F3) was found higher in the SWP Dam (99.94%) than in the VB Dam (99.97%). The calculated CCME WQI for the VB Dam was 16.10 ranking the water quality status in the categories of poor water quality. At the same time, the CCME WQI of SWP Dam was 15.09, ranking the water quality status in the same category as that of the VB Dam. The scope (F1) for variables not meeting the objectives calculated was 77% in both dams (Table 3.6). The scope assesses the extent of compliance with water quality guidelines over the period of the study. The frequency (F2) of the individual variables which do not meet the guidelines was 72% in VB Dam and 76% in the SWP Dam (Table 3.6). More water quality parameters were not meeting the limited guideline concentrations in the SWP Dam compared to the VB Dam. The amount by which the guidelines were not met by the selected variable (F3) was found higher in the SWP Dam (99.94%) than in the VB Dam (99.97%). The calculated CCME WQI for the VB Dam was 16.10 ranking the water quality status in the categories of poor water quality. While the CCME WQI of SWP Dam was 15.09 ranking the water quality status in the same categories of that of the VB Dam.

Table 3.6. The CCME WQI results of the two dams for a 12 month period of over 17 years.

Dams	Years	F1	F2	F3	CCME QI	WQI ranking
VB	17	76.92	72.44	99.76	16.10	poor water quality
SWP	17	76.92	75.64	99.94	15.09	poor water quality

3.3.4. A Comprehensive Pollution Index

All measured water quality variables were used to calculate the CPI of the two dams. The results are shown in Table 3.7. In the VB Dam, TP, OP, NTU, and Cl were found to be heavily polluted except for Fe and SO_4, which were classified as clean. However, TN, EC, pH, and Mn indicated slight to medium polluted conditions in the VB Dam (Table 3.7). Similarly, in the SWP Dam, NTU, TP, OP, TN, EC, Cl, Cu^{2+}, and Chl *a* indicated heavily polluted conditions (Table 3.7). However, Mn, pH, F, and SO_4 were found to be between clean and sub-clean in the SWP Dam (Table 3.7).

Table 3.7. CPI of the VB and SWP dams over a period of 12 months.

Parameters	Dams					
	VB			SWP		
	PI	CPI	Polluted	PI	CPI	Polluted
NTU	37.96706	2.92054	Heavily	47.4548	3.65	Heavily
TP	95.2782	7.32909	Heavily	155.993	12.00	Heavily
OP	45.49549	3.49965	Heavily	147.204	11.32	Heavily
TN	17.81664	1.37051	Medium polluted	24.8935	1.91	Heavily
NH_3	497.7063	38.2851	Heavily	666.231	51.25	Heavily
EC	14.69596	1.13046	Medium polluted	45.9306	3.53	Heavily
Cl	63415.1	4878.08	Heavily	249925	19225.00	Heavily
F	4.072073	0.31324	Sub clean	4.95775	0.38	Sub clean
pH	11.90671	0.9159	Slightly polluted	12.4834	0.96	Slightly polluted
SO_4	1.160146	0.08924	Clean	4.72713	0.36	Sub clean
Mn	1.774314	0.13649	Medium polluted	1.49674	0.12	Clean
Cu^{2+}	993.9587	76.4584	Heavily	1057.34	81.33	Heavily
Chl*a*	51.04639	3.92665	Heavily	143.446	11.03	Heavily

3.3.5. Comparison of the two indices

The CCME WQI was found to be in the range of 0–44, i.e., showing the polluted water quality of the two dams. The SWP Dam CCME WQI was the lowest (15.09) compared to that of the VB Dam (16.10). The CPI was found to be > 2.01 in both dams depicting a heavily polluted lake. Similar to the CCME WQI, CPI of SWP Dam was highest compared to that of VB Dam.

3.3.6. Physicochemical and biological parameters relationship

Some selected variables in both dams were found to correlate significantly positive and negative with others. In the VB Dam, the NTU, NH_3, and EC correlate with Chl *a* (Figure 3.3).

In the SWP Dam, the NTU, TP, and EC were found to correlate with Chl *a* (Figure 3.4). Chl *a* as a phytoplankton indicator was related to NTU, EC, TP, and NH_3 in the two dams.

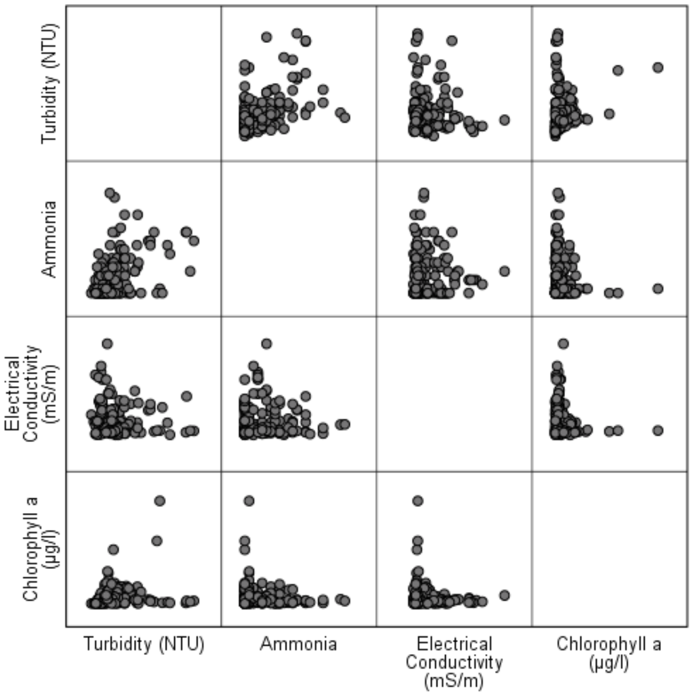


Figure 3.3. The pearson correlation of the water quality variables of the VB Dam.

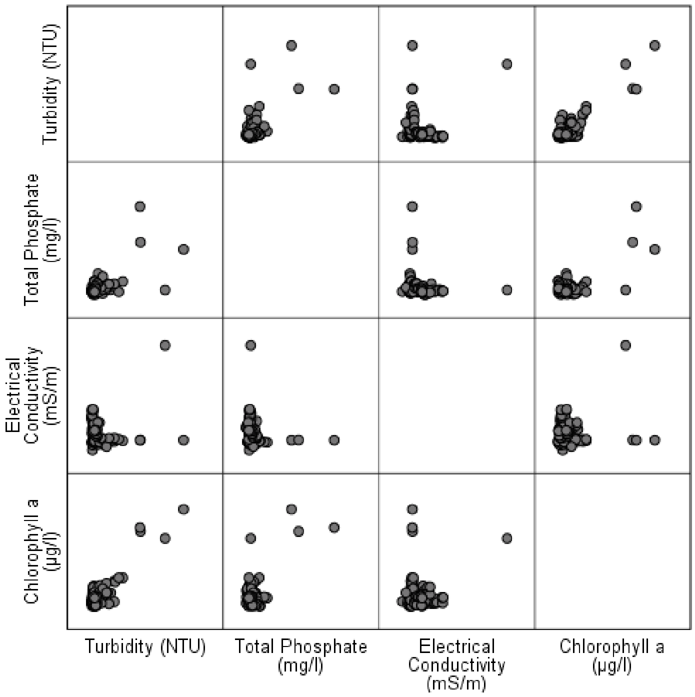


Figure 3.4. The Pearson correlation of the water quality variables of the SWP Dam.

3.4. Discussion

Pollutants emanating from anthropogenic activities in catchment areas are being reported to cause deterioration in surface water quality. This could be more severe for reservoirs located downstream catchment areas associated with dysfunctional wastewater infrastructure, discharges of waste from mines, agriculture and industries, and open defecation from informal settlements. In the current study, it was evident that certain water quality parameters sampled from 2003 to 2019 exceeded the standards set by the TWQR for aquatic ecosystems according to DWAF (1996) and CCME (2012) for aquatic ecosystems and the ANZECC (2000) guidelines for aquatic conservation areas. Water quality status of the two dams located under desert climate conditions was assessed using the combined indices (CCME WQI and CPI). The results show that the two dams are polluted mainly by nutrients, salinity, and particulate matter pollution. Among the two dams, SWP Dam was found by both indices to be heavily polluted compared to VB Dam, which could be due to the land use activities found in the catchment areas of the SWP Dam. Surprisingly, the water quality status of the VB Dam was found to be between moderate to heavily polluted despite the minimal pollutants generating human activities in its catchment area.

The selected two indices were useful in assessing the water quality status of the two dams. The findings of this study are similar to that of Hamlat et al. (2017), who applied 10 water quality indices, including the CCME WQI, in the assessment of the water quality of the Tafna Catchment (including Beni Bahdel bassin, Mouillah wadi bassin, Tafna wadi bassin, the Sikkak and Hamman Boughrara reservoirs) in northwestern Algeria. They reported poor water quality in the Hamman Boughrara reservoirs and Mouillah wadi bassin with CCME WQI ranges of 0–44 (Hamlat et al., 2017). The CCME WQI of the SWP Dam was 15.09 and that of the VB Dam was 16.10, which was lower in comparison to the Hamman Boughrara reservoirs at 31.03 while the Mouillah wadi bassin at 21.06 respectively, although in the same range. Among the tested indices, the CCMEWQI was found to provide good results for the indexation of the general water quality (Hamlat et al., 2017). Domestic and industrial waste in the catchment area was found to be the root causes of water deterioration (Hamlat et al., 2017), which could be similar to the SWP Dam situation. Finotti et al. (2015) used the two indices (CCME WQI and the CETESB) to assess the water quality of the water resources in the urban areas of the Caxias do Sul, Brazil. The sampled stations were found to show different levels of contamination by domestic sewage and industrial effluent (Finotti et al., 2015). All the sampled twelve stations were found to have poor water quality with CCME WQI range of 0–44 (Finotti et al., 2015) which was similar to that of the two dams. The CCME WQI was found to provide reliable results of the degree of contamination by domestic and industrial wastewater (Finotti et al., 2015).

In Surha Lake, India, Mishra et al. (2016) used ten indices, including the CPI, to assess the water quality status during the wet and dry seasons of 2014–2015. The CPI ranged between 1and 4, indicating moderate to heavy pollution, classifying the lake as not suitable for drinking water without treatment (Mishra et al., 2016). The CPI of the two dams in the current study was > 2.01, indicating heavily polluted water not suitable for human consumption prior to treatment. Matta et al. (2017) also applied the CPI in the Henwal River and reported the CPI to be 2.07–8.83 (CPI > 2), which is an indication of the severely polluted water of Henwal River. However, the case studies presented above were not related to desert climate conditions. The water turbidity in both dams was found to be above the standard limit of South African, Canadian, Australian and New Zealand guidelines. The sources could be due to sediments in the inflow of water from the catchment areas caused by soil erosion from urban development and the proliferation of algal blooms. Lumb et al. (2006) reported that the water quality of the Mackenzie-Great Bear sub-basin is impacted by high turbidity due to high suspended loads from the catchment area. In the two dams, the nutrients (TP, OP, TN, and NH_3) were all found to be above the standard limits of TWQR target water quality range for aquatic ecosystems according to DWAF (1996) or CCME (2012) for aquatic ecosystems and the ANZECC (2000) guidelines for conservation of aquatic areas, with higher concentrations in the SWP Dam. Similar to the two dams, Oberholster et al. (2021) reported TP in Loskop Dam, South Africa, to be above the South African, Canadian, Australian, and New Zealand guideline, causing nutrient enrichment. The increase in nutrients in the two dams could be due to point sources of pollution from WWTPs in the catchment area. Cashman et al. (2014) reported malfunctioning wastewater infrastructure in the city of Windhoek due to a lack of maintenance and monitoring, leading to the introduction of waste into tributaries flowing into the SWP Dam. Sirunda et al. (2021a) also reported malfunction of the Ujams Wastewater Treatment Plant between 2012 and 2015 and continuous overflowing of the Goreangab Dam, which holds domestic wastewater. Given the minimal human activities in the VB Dam catchment area, the deterioration in water quality in that dam could be linked to the water transferred from the SWP Dam. Sirunda and Mazvimavi (2014) reported the negative effect of water transfers from the SWP Dam on the water quality of the VB Dam.

Furthermore, the salinity (Cl and EC) concentration in both dams was above the standard guideline limits in both dams. However, the concentration in SWP Dam was higher than that of the VB Dam. The salinity concentrations in the SWP Dam could be due to the industries such as Namib Poultry, Okapuka Meatco Tannery, and Nakara Tannery which use salts for preservations by controlling spoilage through inhibiting the growth of microorganism. Unfortunately, these types of industries which use salts are not found in the VB Dam catchment area. Pazvakawambwa (2018) reported higher concentrations of sodium, TDS, Cl, and SO_4 from wastewater emanating from Okapuka Meatco Tannery, Nakara Tannery, and Namib Poultry above the effluent standard of the Ministry of Agriculture Water and Land Reform (MAWLR) of 2013. Oberholster et al. (2021) reported a higher concentration of Cl in the Loskop Dam above the South African, Canadian, Australian, and New Zealand guidelines. Alobaidy et al. (2010) reported a decline in the water quality of the Dokan Lake, Iraq, due to anthropogenic activities associated with an increase in EC. The transfer of water from the SWP Dam could be the contributing source to the increase in salinity of the VB Dam above the standard limits. However, the increase in Cl could be emanating from sewage facilities of the hospitality infrastructure around the VB Dam. Copper concentration was found to be above the standard guideline limit in both dams, which could be due to the copper mining activities in both catchment areas. Pazvakawambwa (2018) reported spatial variation in copper concentration within the permissible effluent limits in the wastewater emanating from the upper catchment of SWP Dam, in Namibia. However, in the current study, the concentration of copper was above the surface water quality guideline standard of South Africa.

3.5. Conclusion and recommendations

Based on the calculation of the CCME WQI and CPI related to aquatic ecosystems, it has been found that both dams were polluted with nutrients, salinity, and particulate matter in all the 12 months over a period of 17 years. The autumn months were reported with the highest pollution in SWP Dam, while the summer and spring months in the VB Dam. The concentration of nutrients was higher in the spring months in the VB Dam, which could be attributed to water transfers from SWP Dam, and in the autumn months in SWP Dam, which could be due to inflows from the catchment area. Salinity concentration was highest in all spring's months in the two dams, which could be associated with more water evaporation given the desert climate condition during that time of the season. The particulate matter was found to be in a higher concentration during the summer months in VB Dam, which could be attributed to water inflows from the catchment area, and autumn months in SWP Dam, which could be due to phytoplankton blooms. The phytoplankton biomass (Chl *a*) dominates in the spring months in VB Dam and in the autumn months in the SWP Dam. The VB Dam pollution ranges from moderate to heavily polluted in some assessed water quality parameters, except for F, SO_4, and Mn. In the SWP Dam, the majority of the assessed water quality parameters were heavily polluted, except for F, SO_4, and Mn. The impact of the quality water transfer from the SWP Dam to the water quality of the VB Dam was evident, given the lack of pollutant generating anthropogenic activities in the VB catchment. The applied two indices were found to generate similar results in evaluating the two dams' water quality status and could be considered by the water utility manager in assessing the water quality status of dams found in the desert conditions. Nutrient pollution is a major concern in the two dams causing toxic cyanobacteria blooms which increase public health risks.

To improve the water quality of the two dams while ensuring water transfers for water security in central Namibia, several measures must be put in place in the upper catchment of the SWP Dam: (a) fixing or upgrading dysfunctional wastewater treatment plants, (b) enforcement of compliance of industries to effluent discharge standards, (c) annual stakeholder engagement on pollution issues in the upper catchment of the Swakop River, (d) mapping of land use activities and pollution monitoring in the catchment areas of the two dams, (e) monitoring of the accumulative impact of the polluted water transferred into the VB Dam on parameters of concerned, and (f) utilizations of the water quality indices to aid pollution monitoring in the catchment areas for water quality parameters exceeding the standard guideline limits. Further studies are required to further research (a) the development or modification of water quality indices to include toxic phytoplankton species and (b) the treatment of mining effluent from the copper mine Otjihase.

CHAPTER 4: THE ASSESSMENT OF PHYTOPLANKTON DYNAMICS IN TWO RESERVOIRS IN SOUTHERN AFRICA WITH SPECIAL REFERENCE TO WATER ABSTRACTION FOR INTER-BASIN TRANSFERS AND POTABLE WATER PRODUCTION

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Declaration by the candidate

With regard to Chapter 4, the nature and scope of my contribution were as follows:

Nature of contribution	**Extent of contribution**
Conceptual design, fieldwork, experimental work, manuscript writing, data curation, formal analysis, methodology	80%

The following co-authors have contributed to Chapter 4:

Name	**Email address and institutional affiliation**	**Nature of contribution**	**Extent of contribution**
Gideon Wolfaardt	Stellenbosch University Water Institute, Faculty of Science, Stellenbosch University, Matieland 7600, South Africa	Conceptual design, manuscript writing-review & editing.	20%
Paul Oberholster	Centre for Environmental Management, Faculty of Natural and Agricultural Sciences, University of the Free State, Bloemfontein 9300, Republic of South Africa	Conceptual design, manuscript writing-review & editing	
Marelize Botes	Stellenbosch University Water Institute, Faculty of Science, Stellenbosch University, Matieland 7600, South Africa	Conceptual design, manuscript writing-review & editing	
Christoff Truter	University of Pretoria, Department of Paraclinical Sciences, Faculty of	Conceptual design, manuscript writing-	

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Abstract

Toxic phytoplankton in the aquatic ecosystems are dynamic, affecting water quality. It remains unclear as to how possible toxic phytoplankton assemblages vary vertically and temporally in SWP and VB dams, located in a dry subtropical desert region in central Namibia. The following variables were analyzed: pH, Secchi depths, NTU, water temperature (WT), TP , OP, Chl *a*, phytoplankton cells, and water depths. Cyanobacteria dominated the phytoplankton community in the autumn, winter and spring (dry) and summer (wet) seasons, at all the depth ranges in both dams. *Microcystis* dominated the vertical and temporal dynamics, followed by *Dolichospermum*. In the dry seasons, higher cyanobacteria cell numbers were observed in comparison to the rainy season in both dams. Spring blooms of cyanobacteria were evident in the VB Dam while autumn and spring cyanobacteria blooms were observed in the SWP Dam. In the SWP Dam, the preferable depth ranges for toxic cyanobacteria species were at 5 to 10m while in the VB Dam at 0 to 5m range. The findings of the current study indicate that the traditional selective withdrawal of water in the two dams should be performed with vertical and temporal dynamics of possible toxic cyanobacteria accounted for to aid the abstraction of water with the lowest possible toxic phytoplankton numbers, which could lower the public health risk.

4.1. Introduction

Phytoplankton production and composition have been reported to be dynamic, vertically, and seasonally in aquatic ecosystems (Moura et al., 2011; Touati et al., 2019). The dynamics could be exacerbated by elevation in water temperature, and nutrients due to changes in climatic conditions, plus anthropogenic activities such as agricultural runoff, and dysfunctional WWTPs (Chapra et al., 2017; Paerl et al., 2008; Rigosi et al., 2014). In many subtropical eutrophic reservoirs used for portable water, toxic cyanobacteria have been reported to dominate the phytoplankton community and this phenomenon is becoming an increasing concern to many water utility managers (Dalu & Wasserman, 2018; Dantas et al., 2011; Dantas et al., 2012; Kim et al., 2020; Lu et al., 2012; Oberholster et al., 2004; Oberholster & Botha, 2011; Oberholster et al., 2005; Piccin-Santos & Bittencourt-Oliveira, 2012). The occurrence of potentially toxic cyanobacteria species in drinking water reservoirs leads to the abstraction of poor quality raw water for inter-basin and potable water provision, especially when the vertical and temporal dynamics are not known. The impact could be more severe in reservoirs found in a subtropical desert climate characterized by low rainfall, high evaporation, and spatially distributed populations, where water is transferred between dams to minimize evaporation losses.

Cyanobacteria with a tendency to dominate phytoplankton communities are known to tolerate a wide range of climatic and anthropogenic conditions, which lead to their proliferation as excessive masses in freshwater and marine ecosystems (Chapra et al., 2017; Chorus & Bartram, 1999; Paerl & Paul, 2011). Under favorable climatic and anthropogenic induced environmental conditions, cyanobacteria form dense and sometimes toxic blooms in the freshwater and marine environment which threaten ecosystem functioning and reduce water quality for recreation and drinking (Berger et al., 2006; Ho et al., 2019; Huisman et al., 2018). However, when conditions are not favorable, cyanobacteria have the ability to store essential nutrients and metabolites within the cytoplasm to survive such conditions making them good competitors with other phytoplankton species. Previous studies reported that in many tropical and subtropical reservoirs phytoplankton composition shifts from diatom dominant systems toward high temperature tolerant species of cyanobacteria dominated by mainly *Microcystis* sp. (Fadel et al., 2014; Jeppesen et al., 2017; Ndebele-Murisa et al., 2010; Ndebele & Magadza, 2006b) due to climate and anthropogenic induced changes in water temperature and nutrient availability.

Cyanobacteria species may gain dominance by using gas vesicles, which enable them to regulate their buoyancy, for vertical movement in the water column to track for light, nutrients, and favorable water temperature (Zhang et al., 2008). These gas vesicles have notable similarities in molecular structure amongst cyanobacterial genera, but they differ in shape, yield, and critical pressures, with *Microcystis* having a more stable gas vesicle compared to others (Tonetta et al., 2015). It is reported that the optimal habitat for phytoplankton growth is at a depth of 3 m to 5 m within the water column (Ndebele-Murisa et al., 2010). Phytoplankton growth is lower near the water surface due to photo-inhibition and tends to increase with depth until the light becomes the limiting factor (Ndebele-Murisa et al., 2010; Zhang et al., 2008). Zhang et al., (2008) observed that maximum growth occurred at 0.3 m of the water column for diatoms, chlorophytes, and cyanobacteria which decreased as irradiance increased in Lake Taihu, China. Similarly, in Peri Lake, Brazil, a higher primary production rate of the toxic cyanobacteria species *Cylindrospermopsis* was found at the surface and decreased as a function of water column depth (Tonetta et al., 2015). In addition, the thermal gradient and mixing which causes stratification in the water column is also reported to influence the seasonal composition of phytoplankton by controlling the nutrient distribution in the water column(Ndebele & Magadza, 2006a). Chl *a* is reported to vary with depths in many aquatic ecosystems, causing a shift in deep chlorophyll maximum (DCM) within the water column which is driven by light availability and an increase in cellular Chl *a* concentration (Hamilton et al., 2010; Reinl et al., 2020).

In addition to vertical variations in the water column, phytoplankton assemblages also vary with seasons, governed mainly by associated changes in water, light, temperature, and nutrients. Cyanobacteria has been reported to dominate in all seasons in many subtropical lakes causing seasonal blooms (Ndebele-Murisa et al., 2010). While seasonal blooms may start in summer and last until autumn, some blooms occur throughout the seasons, whereas, episodic blooms which last for weeks or days, may also occur. For example, tropical Lake Ogelube, in Nigeria, reportedly had higher phytoplankton biomass in the rainy season, dominated by chlorophyceae followed by cyanobacteria (Ndebele-Murisa et al., 2010). Furthermore, tropical and subtropical African lakes including Kariba, in Zimbabwe, Malawi, in Malawi, Tanganyika, in Tanzania, and Victoria in Uganda are reportedly dominated by cyanobacteria in summer and diatoms in winter (Hecky & Kling, 1981; Ndebele-Murisa et al., 2010). Mignot et al., (2014) and Reinl et al., (2020) reported an increase in Chl *a* in winter causing DCM to be shallower and a decrease in Chl *a* in summer resulting in an increased DCM depth.

Subtropical lakes found outside the desert climate region such as the Hartbeespoort and Vaal dams, in South Africa, are reportedly dominated by cyanobacteria (i.e., *Dolichospermum* and *Microcystis* sp.) during the warmer summer season (Ballot et al., 2014; Chinyama et al., 2016; Ndebele-Murisa et al., 2010; Ndlela et al., 2016). Subtropical reservoirs, found in a desert climate like Lake Kinneret, Israel, were subjected to spring blooms of *Peridium gatunense* (Zohary, 2004). Lake Nasser, in southern Egypt and northern Sudan, situated in a desert climate, was found to be dominated by *Microcystis* in the spring months (Jeppesen et al., 2017; Kassem et al., 2020). However, suspended Chl*a* concentrations were higher in both the autumn and spring seasons (Kassem et al., 2020). Karaoun Reservoir, Lebanon which is also located in a desert climate was found to be dominated by the bloom forming cyanobacteria *Aphanizomenon* in spring and autumn, and *Microcystis* in summer (Fadel et al., 2014).

The SWP and VB dams are important water sources in central Namibia (Sirunda & Mazvimavi, 2014). They are located in a subtropical desert climate characterized by large differences in day and night-time temperature, low rainfall, low humidity, and high evapotranspiration (Kottek et al., 2006). The two dams are within an ephemeral river, which only flows sporadically during the rainy season (Sirunda & Mazvimavi, 2014). Since the SWP Dam water is transferred to VB Dam before treatment (Lewis et al., 2019; Sirunda & Mazvimavi, 2014), the mixing of this water could cause treatment challenges in the future if the vertical and temporal dynamics of cyanobacteria as a function of water column depths and season are not well understood. Furthermore, the water in the two dams under study is abstracted selectively at different intake points (Sirunda & Mazvimavi, 2014), and the abstraction at these points is informed by water quality results of more than seven days on average from the time the water samples are collected to analyses in the laboratory. The water quality results do not take the vertical and temporal dynamics of possible toxic phytoplankton into consideration. Therefore, the combination of climate change and anthropogenic conditions could affect the water quality abstracted from dams, and if not properly treated could pose public health risks.

The vertical and temporal dynamics of cyanobacteria in the SWP and VB dams have not been established yet. However, cyanobacteria cells were reported not to be reduced by phytoplankton control measures in the SWP Dam (Sirunda et al., 2021a). While the climate in which the dams are located is reported to be favorable for cyanobacteria growth, it remains unclear as to which depth ranges these species prefer to inhabit seasonally. The aim of the current study was to describe the vertical and temporal dynamics of phytoplankton communities during the wet (summer) and dry (autumn, winter and spring) seasons in the SWP and VB dams and the resultant implications for water abstraction and inter-basin water transfers between the two dams as well as potable water provision. The objectives of the current study were (a) to define the vertical and temporal dynamics of potentially toxic phytoplankton species to prevent abstraction of those species for bulk water supply, and (b) to determine the depth ranges related to subtropical desert conditions for the inhabitation of possible toxic phytoplankton species in the SWP and VB dams.

4.2. Materials and methods

4.2.1. Characteristics of the study area and field sampling sites

The VB(21°59′59.27″ S 16°58′54.76″ E) and SWP dams (22°12′44.31″ S 16°31′44.97″ E) are located in a dry subtropical desert region in central Namibia. These impoundments are warm monomictic man-made dams that, are stratified throughout the year and only experiences overturn or mixing during the winter season due to cold conditions (Figure 4.1) (Sirunda & Mazvimavi, 2014). The dams are used to supply water to the city of Windhoek, Okahandja Town, Karibib Town, Otjimbingwe Village, and Navahacb Mine (Scott et al., 2018; Slabbert, 2007).

Water from the SWP Dam is transferred through a pipeline to the VB Dam due to its proximity to the treatment plant and the reduced water loss from evaporation due to its smaller surface area (Sirunda & Mazvimavi, 2014). Water from the VB Dam is then abstracted and transported via a pipeline to the VB Treatment Plant for potable water production (Rensburg, 2006; Scott et al., 2018). SWP Dam has the largest capacity (63.5 Mm^3), with a catchment area of 5480 km^2 and low annual rainfall of 350 mm/a (Table 4.1).

A single sampling site with existing historical datasets was selected per dam in the proximity of the dam wall (Figure 4.1b), as this is the deepest point in the dam, where water is abstracted for transfers and treatment. The selected sampling sites are standard sites established by the utility for continuous monitoring of the dams’ water quality. Depending on the water level in the selected dams, water samples were collected at different intake points. The current study is based on data collected at these sites from 2003 to 2019, on monthly basis in each season.

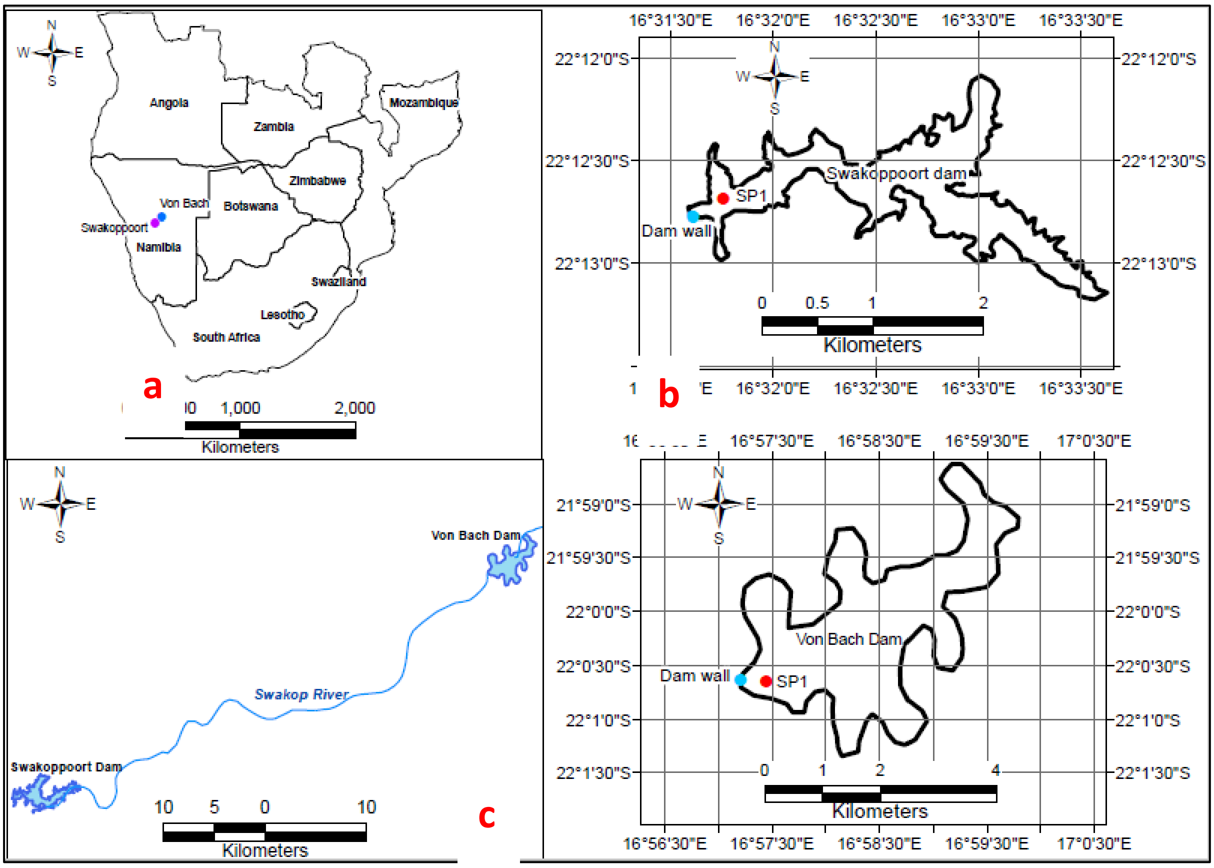

Figure 4.1. Maps indicating the (a) location of Namibia within Africa; (b) SWP and VB dams located in central of Namibia, with sampling sites nearby the dam wall indicated; and (c) location of the SWP and VB dams in the Swakop River. The maps were created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset.

Table 4.1. The main features of the SWP and VB dams.

Feature	VB Dam	SWP Dam
River	Swakop River	Swakop River
Capacity (Mm^3)	48.56	63.48
Max. depth (m)	29	30
Potential evaporation losses (mm/a)	2254	2275
Annual rainfall (mm/a)	370	350
Surface area (FSC) (km^2)	4.89	7.81
Catchment area size (km^2)	2920	5480
Geology of the areas	Schist and granite	Schists and granite
Year completed	1970	1977
Mean depth (m)	10	8.1

FSC: Full supply capacity.

4.2.2. Analyses of abiotic and biotic variables

The sampling was always performed between 09:30 am and 2:00 pm using a designated boat in both dams. Water temperature (WT) was measured on-site using a digital thermometer (Yellow Springs Instrument, Model 54A, Yellow Springs, OH, USA) with a range of −5 °C to +45 °C, accuracy of ±0.7 °C, at 0.5 m intervals under the water column. Secchi depths (SD) were measured using the standard black and white Secchi disc, 20 cm diameter in size, at each sampling site. Water samples for chemical, phytoplankton identification, and suspended Chl*a* analyses were collected from the two sampling sites on a monthly basis from 2003–2019. To collect a representative sample at each sampling site, a dip sampling method was followed using a Von Dorn 5 L water sampler (Burns et al., 2000). Water samples were collected at the different depths ranging from the surface to the bottom at the sampling sites found nearby the dam wall, in both the SWP and VB dams (Figure 1b). Three depth ranges were defined as (a) 0 to 5 m, (b) 5 to 10 m, and (c) 10 to 20 m in the two dams taking into consideration the different intake points where water is abstracted. In the SWP Dam, the different intake points 1, 2, 3, and 4 were classified under the depth range of 0 to 5 m, intake points 5, and 6 under 5 to 10 m, and intake 7, and 8 under 10 to 20 m. In VB Dam, intake 1 was classified under depth range 0 to 5 m, intake 2, under 5 to 10 m, and intake 3 and 4 under 10 to 20 m respectively. The vertical variation of the biochemical and phytoplankton cells samples were established using the defined depth ranges. Taking into consideration the light penetration gradient of the two dams, the depth ranges of 0 to 5 m was in the photic zone, and both 5 to 10 m and 10 to 20 m were in the aphotic zone.

The collected water samples for biochemical and phytoplankton cells were transferred into 1 L labelled acid washed plastic and glass containers, which were kept in the dark in cooler boxes. The cooler boxes were transported to the laboratory where the phytoplankton were preserved and analyzed within 24 h.

To identify phytoplankton, samples were sedimented in a Sedgewich–Rafter counting chamber and analyzed under an inverted microscope at 400× magnification using the strip-count method (American Public Health Association (APHA); American Water Works Association (AWWA) and WPCF (WPCF), 1992). All phytoplankton were identified according to (Taylor et al., 2007; Truter, 1987; Vuuren et al., 2006; Wehr et al., 2015). The OP and TP of the water samples were measured using the ascorbic acid method as described in American Public Health

Association (APHA) (1998: part 4500 P E) using a spectrophotometer (American Public Health Association (APHA), 1998). The total nitrogen of the water samples was measured using the cadmium reduction method as described in APHA (1998: part 4500 NO3-E) (American Public Health Association (APHA), 1998. In the analysis of total nitrogen, a reduction column was used together with a spectrophotometer. The turbidity (NTU) of the water samples was measured using a turbidity meter following the nephelometric method (APHA, 1998: part 2130 B) in the laboratory [48]. Suspended Chl*a* contained in the water samples was measured using the spectrophotometric determination method as described in APHA (1998: part 10200 H) (American Public Health Association (APHA), 1998).

A blank sample without the analyte was used for quality control and assurance of the analytical processes for each analyzed biochemical sample to correct for potential background signals. The water samples were analysed in triplicate and the results of analysis were arranged in chronological order and sorted according to sampling sites, dates, seasons, and depths. A visual scan was performed to identify possible outliers that were re-analyzed, if necessary.

4.2.3. Statistical analysis

The data from 2003–2019 did contain periods where data was missing in some months of the year due to issues of not sampling caused by the unavailability of the boat plus human resources and the timing of the sampling were also not always similar in both the dams. However, the long-term dataset analyses were sufficient to provide insights into the vertical and temporal dynamics of phytoplankton in the two dams. In the study, the summer season was defined as November, December, January, February, and March. Autumn was defined as the months April, and May, winter was defined as June, July, and August, and spring was defined as September and October. The wet season was summer, and the dry seasons were autumn, winter, and spring.

Variation in cyanobacteria abundance and water chemistry data within and among the two dams investigated were assessed using Repeated Measures Mixed Models (Variance Estimation and Precision Module, Statistica v13, Tibco Software, Palo Alto, CA, USA) (Richardson et al., 2019; Wu, 2009). Seasonal means per depth range per dam per year were used for the analyses. “Season” and “Year” were specified as random factors and applied as repeated measures, whereas, “Depth range” and “Dam” represented fixed effects. Pairwise differences were assessed through Fisher’s least significant difference (LSD) post hoc test. The normality of the datasets applied in mixed models was evaluated using normal probability plots of residuals. Cyanobacteria data that were not normally distributed were rank transformed before analysis, whilst, non-parametric chemistry data were boxcox transformed.

The phytoplankton species diversity of the SWP and VB dams was calculated using Shannon's diversity index (H) (Motwani et al., 2014). The TN: P molar ratio was used for the classification of the trophic status of the two dams during the study period using the nutrients index criteria (Dodds, 2006; Dodds & Smith, 2016; Jones et al., 2003; Modabberi et al., 2019).

Furthermore, a multivariate statistical analysis of surface water quality based on correlations and variations in the data set, i.e., principal component analysis (PCA) was performed. A PCA triplot was constructed using seasonal average cyanobacteria cell count and water chemistry data representing the 0 to 5m, 5 to 10m, and 10 to 20m depth ranges for both dams investigated. The cyanobacteria data form the focal plot of the PCA triplot, whereas water quality parameters were treated as supplementary variables. Log transformed cyanobacteria data were centered and standardized before use in the PCA (Ter Braak,& Smilauer, 2002). Canoco v5 (Microcomputer Power, Ithaca, NY, USA) was used for multivariate statistical analyses.

4.3. Results

4.3.1. Physiochemical characteristic

During the study period from 2003–2019, the vertical and temporal dynamics of the water temperature was assessed. The summer months (November, December, January, February, and March) were warmer followed by autumn months (April, and May) and spring months (September and October) in both dams, which could be associated with thermal stratification (Table 4.2). However, winter months (June, July, and August) were found to be colder in both dams for the duration of the study, which could be associated with complete mixing (Table 4.2).

Table 4.2. The average ± standard deviation of water column temperature in the SWP and VB dams during the study period (2003–2019) at (a) different depth ranges and (b) among seasons.

(a)				
Dam	**Depth Ranges**	**Average Temp °C**	**Minimum Temp °C**	**Maximum Temp °C**
SWP Dam	0 to 5 m	20.7 ± 3.8	14.0	27.0
	5 to 10 m	19.8 ± 3.1	14.0	25.0

	10 to 20 m	15.9 ± 2.1	13.0	23.0
VB Dam	0 to 5 m	20.4 ± 4.1	12.0	27.0
	5 to 10 m	18.6 ± 4.5	8.0	25.0
	10 to 20 m	16.0 ± 2.4	12.0	21.0

(b)

Dam	Season	Average Temp °C	Minimum Temp °C	Maximum Temp °C
SWP Dam	Summer	23.2 ± 2.4	12.0	28.0
	Autum	21.5 ± 2.3	17.0	28.0
	Winter	15.9 ± 1.6	13.0	21.0
	Spring	18.5 ± 2.6	14.0	28.0
VB Dam	Summer	21.2 ± 5.5	7.0	25.0
	Autum	20.2 ± 2.3	14.5	26.0
	Winter	14.9 ± 1.4	4.5	21.0
	Spring	17.2 ± 2.6	0.5	26.5

Vertically, both the SWP and VB dams exhibited a thermal difference between the depth ranges, and water temperature decreased as a function of depth (Table 4.2a). Seasonally in the SWP Dam, the summer mean water temperature was at 23.2°C reaching a maximum of 28.0°C, 21.5°C reaching a maximum of 28.0°C in autumn, and 18.5 °C reaching a maximum of 28.0°C in the spring season. Seasonally, the VB Dam, summer mean water temperature was 21.2°C reaching a maximum of 30.0°C, 20.2°C reaching a maximum of 26.0°C in autumn, and 17.2°C reaching a maximum of 26.5°C in spring (Table 4.2b). Winter water temperatures in the SWP and VB dams were 15.9°C and 15.0°C respectively, reaching a maximum of 21.0°C, which is lower compared to the other seasons, but favorable for the growth of toxic cyanobacteria. The temperature depth profiles indicate that thermal stratification was experienced in all seasons except for the winter season in both dams. The favorable water temperature at all the depth ranges and seasons could be associated with the desert climate in which the dams are located (Table 4.2b).

The Secchi depth (SD) was on average 0.6m reaching a maximum of 4.1 m in the SWP Dam and was 0.9m reaching a maximum of 3.6m in the VB Dam. In the SWP Dam, the average water turbidity was higher at the depth ranges of 0 to 5m being 32.62NTU, compared to deeper depth ranges of 5 to 10m and 10 to 20m, although only significantly so relative to the 10 to 20m range (p = 0.04) (Table 4.3a). Seasonally, the water turbidity was higher during the dry season of autumn (24.50 NTU), spring (22.87NTU), and rainy season of summer (18.48NTU) (Table 4.3b) which could be related to cyanobacteria blooms during the dry seasons and inflow of suspended solids during the rainy season in the SWP Dam.

Table 4.3. The average ± standard deviation of the physiochemical characteristics of the SWP and VB dams at (a) the different depth ranges; and (b) among seasons.

(a)

Dams	Depth ranges (m)	Total phosphorus (mg/l)	Total nitrogen (mg/l)	Ortho-phosphate (mg/l)	Turbidity (NTU)	Chl *a* (µg/l)
SWP	0 to 5m	0.35 ± 0.45	2.01 ± 0.99	0.10 ± 0.10	32.62 ± 49.76	102.35 ± 117.20
	5 to 10m	0.38 ± 1.38	1.98 ± 0.96	0.11 ± 0.11	15.01 ± 13.38	51.88 ± 38.85
	10 to 20m	0.29 ± 0.22	2.18 ± 1.10	0.15 ± 0.12	12.27 ± 8.96	24.71 ± 23.17
VB	0 to 5m	0.28 ± 0.62	1.43 ± 1.02	0.03 ± 0.06	14.08 ± 35.39	45.19 ± 114.14
	5 to 10m	0.18 ± 0.28	1.34 ± 0.83	0.03 ± 0.07	13.75 ± 33.64	20.04 ± 29.20
	10 to 20m	0.18 ± 0.22	1.61 ± 0.84	0.05 ± 0.05	17.57 ± 32.91	7.18 ± 10.02

(b)

Dams	Seasons	Total phosphorus (mg/l)	Total Nitrogen (mg/l)	Ortho-phosphate (mg/l)	Turbidity (NTU)	Chlorophyll *a* (µg/l)
SWP	Summer	0.37 ± 1.34	2.23 ± 1.12	0.12 ± 0.12	18.48 ± 17.92	46.42 ± 39.53
	Autum	0.30 ± 0.34	2.07 ± 0.88	0.16 ± 0.12	24.50 ± 58.35	83.98 ± 121.28
	Winter	0.37 ± 0.50	2.03 ± 0.99	0.10 ± 0.08	17.21 ± 27.22	65.94 ± 98.90
	Spring	0.27 ± 0.24	1.77 ± 0.89	0.13 ± 0.13	22.87 ± 29.82	57.10 ± 57.73
VB	Summer	0.22 ± 0.50	1.56 ± 0.97	0.05 ± 0.09	24.45 ± 51.43	14.67 ± 25.65
	Autum	0.18 ± 0.23	1.48 ± 0.93	0.04 ± 0.04	13.60 ± 16.62	24.10 ± 65.53
	Winter	0.19 ± 0.29	1.32 ± 0.80	0.03 ± 0.02	7.80 ± 8.65	19.06 ± 43.75
	Spring	0.25 ± 0.44	1.55 ± 0.85	0.03 ± 0.03	11.10 ±18.08	39.32 ± 121.51

However, the opposite was observed in the VB Dam, where the average turbidity was generally higher at the deeper depth ranges of 10 to 20m (17.57NTU) relative to 0 to 5m and 5 to 10m depth ranges (Table 4.3a). This could be attributed to the inflow of rainfall water carrying suspended solids from the catchment area, as the turbidity was higher in the wet season of summer compared to the dry season of autumn, spring, and winter (Table 4.3b). The only significant difference in turbidity was, however, between the 5 to 10m and 10 to 20m categories (p = 0.02) in the VB Dam. When the seasons were considered collectively, turbidity was significantly higher in the SWP Dam than the VB Dam ($F_{1.290}$ = 147.71, p < 0.001).

Throughout the study period from 2003 to 2019, the SWP Dam was eutrophic with a low N:P ratio of 17.2, while the VB Dam was mesotrophic with a high N:P ratio of 26.7. In the SWP Dam, there was a high concentration of TP at the upper depth ranges of 0 to 5 m (0.35mg/L) and 5 to 10 m (0.38mg/L) compared to the deeper depth range of 10 to 20m (0.29mg/L) (Table 4.3a). Seasonally, summer and winter seasons were observed with a higher concentration of TP followed by autumn and spring (Table 4.3b). However, there was no significant difference in TP between the different depth ranges ($F_{2.111} = 0.79$, $p = 0.46$). In addition, pH varied significantly among depth ranges ($F_{2.111} = 24.91$, $p < 001$), although, the only significant pairwise difference was among the 0 to 5m and the 10 to 20m ranges ($p < 0.001$). Unlike TP, OP concentration was, however, higher at the depth range of 10 to 20m (0.15mg/L) compared to the upper depth ranges (Table 4.3a). Seasonally, OP concentration was higher in autumn followed by spring and summer (Table 4.3b). However, there was no significant difference in OP between the different depth ranges ($F_{2.108} = 0.21$, $p = 0.81$).

A higher concentration of TP was observed at the upper depth ranges of 0 to 5m (0.28mg/L) compared to the two lower depth ranges in the VB Dam (Table 4.3a). The TP concentration was higher in the spring (0.25mg/L), and summer (0.22mg/L) compared to the other seasons (Table 4.3b). However, there was no significant difference in TP among the depth ranges ($p > 0.05$). Moreover, TP varied significantly among the two dams when the seasons were considered collectively, being significantly higher in the SWP Dam ($F_{1.285} = 41.96$, $p < 0.001$). Unlike TP, OP concentration was higher at the deeper depth range (0.05 mg/L) compared to the upper depth ranges (Table 4.3a). Seasonally, summer (0.05mg/L) and autumn (0.04mg/L) recorded a higher concentration of OP compared to the other seasons (Table 4.3b). The concentration of OP was, however, not significantly different between the depth ranges in VB Dam ($F_{2.144} = 0.61$, $p = 0.55$) or among the two dams ($F_{1.278} = 3.47$, $p = 0.06$).

4.3.2. Phytoplankton community

Throughout the study period from 2003 to 2019, in the SWP Dam, the phytoplankton community consisted of seven taxonomic groups and 62 species. The taxonomic groups were cyanobacteria (blue-green algae), chrysophyceae (brown algae), bacillariophyceae (diatoms), cryptophyceae (crytomonads), dinophyceae (dinophyta), euglenophyceae (euglenoids), and chlorophyceae (green algae). Chlorophyceae were the most diverse group, with 39 species, which comprised 63% of the total species richness, followed by bacillariophyceae with 10 species, which were 16% of the total species richness. The latter was followed by cyanobacteria with five species, which were 8% of the total species richness. Chlorophyceae was the most diverse group but only accounted for 13% of the total cell counts. Among the phytoplankton, cyanobacteria dominated the community at 85% of the total cell counts.

In the SWP Dam, the phytoplankton community was found to vary vertically among depth ranges. Cyanobacteria cell numbers were dominating followed by chlorophyceae at all the depth ranges, but more concentrated at the 5 to 10m depth range (Table 4.4a, Figure 4.2a). Cyanobacteria cell numbers were significantly different between the depth ranges ($F_{2.392}$ = 10.69, $p < 0.001$) in the SWP Dam. Similarly, Chl*a* varied significantly as function of depth ($F_{2.101}$ = 30.89, $p < 0.001$), being lower at the 5 to 10m compared to both the 0 to 5m and 10 to 20m depth ranges ($p < 0.001$) (Table 4.3a).

Table 4.4. The average ± standard deviation of the phytoplankton communities of the SWP and VB Dams at (a) different depths and (b) among seasons.

(a)

Dam	**Depth ranges (m)**	**Cyanobacteria (cells/ml)**	**Chrysophyceae (cells/ml)**	**Bacillariophyceae (cells/ml)**	**Euglenophyceae (cells/ml)**	**Dinophyceae (cells/ml)**	**Cryptophyceae (cells/ml)**	**Chlorophyceae (cells/ml)**
SWP Dam	0 to 5m	99 270 ± 173 371	4 ± 38	1 047 ± 1 380	37 ± 218	17 ± 156	599 ± 1 742	14 885 ± 40 087
	5 to 10m	123 823 ± 136 905	6 ± 56	1 684 ± 3 599	48 ± 279	24 ± 177	603 ± 999	21 900 ± 61 762
	10 to 20m	69 661 ± 106 762	1 ± 6	1 688 ± 2 731	60 ± 347	41 ± 323	283 ± 459	9 506 ± 20 820
VB Dam	0 to 5m	26 691 ± 146 972	10 ± 45	379 ± 683	40 ± 116	314 ± 802	240 ± 602	3 661 ± 6 760
	5 to 10m	6 702 ± 16 018	3 ± 10	391 ± 846	36 ± 101	107 ± 222	140 ± 409	2 403 ± 3 541
	10 to 20m	4 406 ± 14 861	48 ± 484	195 ± 458	15 ± 26	27 ± 61	32 ± 73	1 818 ± 4 613

(b)

Dam	**Seasons**	**Cyanobacteria (cells/ml)**	**Chrysophyceae (cells/ml)**	**Bacillariophyceae (cells/ml)**	**Euglenophyceae (cells/ml)**	**Dinophyceae (cells/ml)**	**Cryptophyceae (cells/ml)**	**Chlorophyceae (cells/ml)**
SWP Dam	Autum	146 069 ± 166 630	13 ± 86	1 962 ± 2 972	9 ± 26	12 ± 101	634 ± 1 945	39 175 ± 90 125
	Spring	124 262 ± 174 443	-	595 ± 746	19 ± 62	24 ± 210	344 ± 658	6 151 ± 9 787
	Summer	69 364 ± 78 962	3 ± 19	1 752 ± 3 630	105 ± 469	9 ± 66	677 ± 1 287	13 701 ± 24 106
	Winter	86 462 ± 16 697	-	1 280 ± 1772	21 ± 55	61 ± 376	308 ± 504	7 423 ± 15 910
VB Dam	Autum	8 555 ± 11 778	52 ± 476	398 ± 524	30 ± 38	42 ± 190	153 ± 240	3 827 ± 6 210
	Spring	25 416 ± 183 529	2 ± 8	147 ± 361	9 ± 18	409 ± 963	75 225	2 381 ± 5 821
	summer	9 714 ± 28 595	32 ± 371	262 ± 698	39 ± 134	94 ± 134	151 599	2 633 ± 5 783

Winter	9 493 ± 58 952	5 ± 15	425 ± 820	28 ± 43	102 ± 326	118 270	1 761 ± 2 169

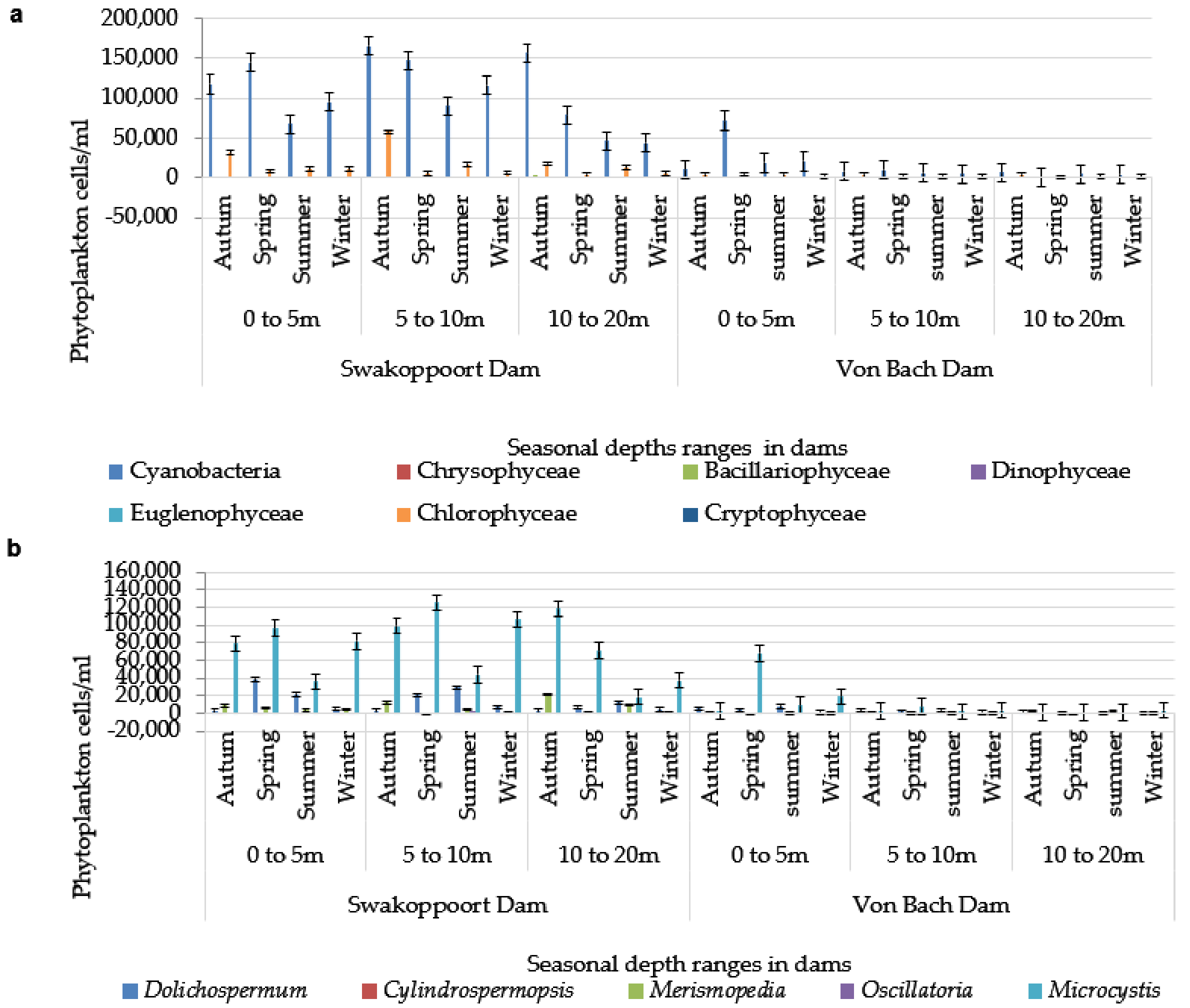

Figure 4.2. (a), Seasonal mean phytoplankton counts in the SWP and VB dams at different depth ranges and seasons from 2003–2019 and (b), the dominant cyanobacteria species in the SWP and VB dams at different depth ranges and seasons from 2003–2019. Error bars indicate standard deviations.

Temporal variation with seasons were also observed and cyanobacteria dominated the phytoplankton community at the depth ranges of 5 to 10m during the dry autumn (165,897cells/mL) and, spring (147,299cells/mL), and winter (116,338cells/mL) seasons compared to the wet summer season (90,401cells/mL) in the SWP Dam (Table 4.4b, Figure 4.2a). In the aforementioned depth ranges and seasons, *Microcystis* dominated the cyanobacteria followed by *Dolichospermum* (Table 4.4a, Figure 4.2b). Significant variation associated with water depth was observed in the abundance of *Dolichospermum* ($F_{2.392}$ = 7.29 p < 0.001) and *Microcystis* ($F_{2.392}$ = 8.74, p < 0.001), whereas *Merismopedia* abundance did not vary among the depth ranges ($F_{2.392}$ = 0.50, p = 0.61). Applying the seasonal means, cyanobacteria were found to be dominant at all the depth ranges in SWP Dam but were more concentrated at the 5 to 10 m depth range over the 17-year study period (Figure 4.3a).

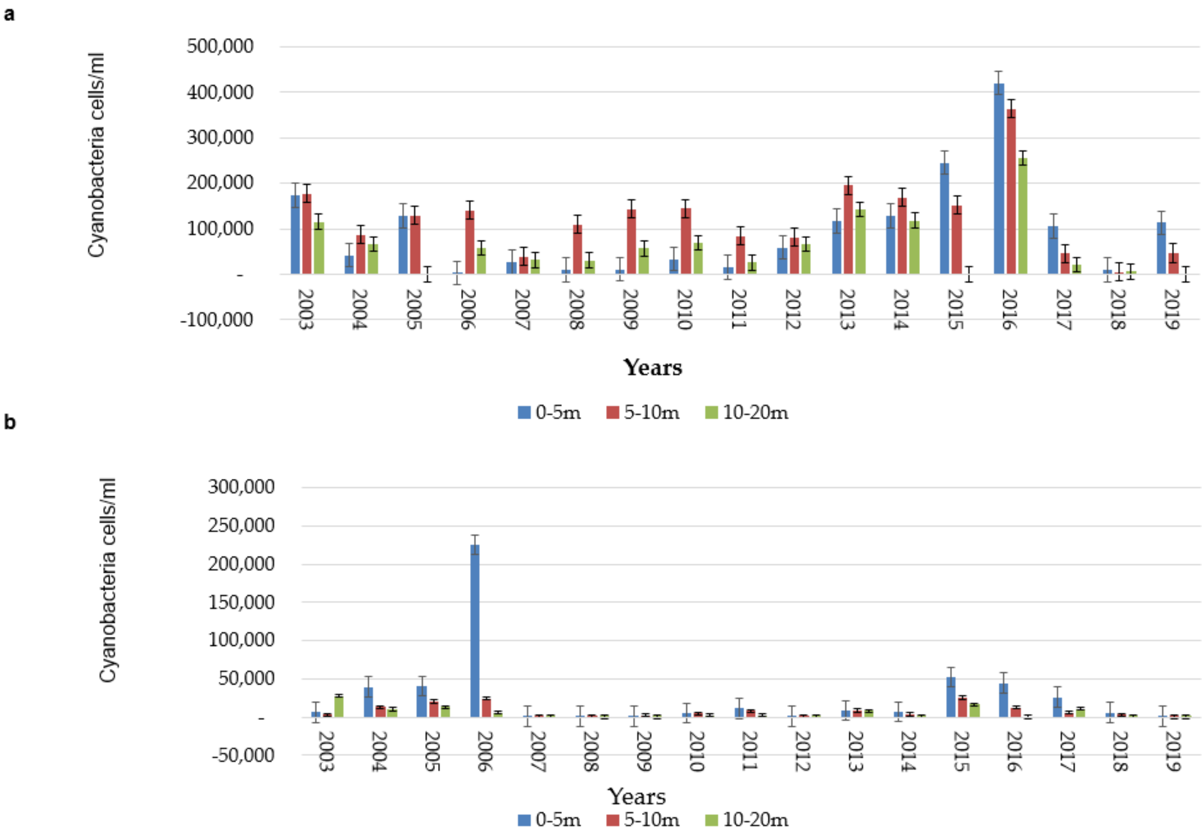

Figure 4.3. Yearly mean of cyanobacteria species cell counts in, (a) the SWP Dam and (b) the VB Dam at different depths ranges from 2003–2019. Error bars indicate standard deviations.

In the VB Dam, the phytoplankton community did consist of seven taxonomic groups and 78 species. The taxonomic groups were cyanobacteria (blue-green algae), chrysophyceae (brown algae), bacillariophyceae (diatoms), cryptophyceae (cryptomonads), dinophyceae (dinophyta), euglenophyceae (euglenoids), and chlorophyceae (green algae). Chlorophyceae was the most diverse group, with 48 species (62% of the total species richness), followed by bacillariophyceae with 13 species (17%), followed by cyanobacteria with 5 species (6%). Chlorophyceae was the most diverse group, but they only accounted for 17% of the total cell counts during the study period. Cyanobacteria dominated the phytoplankton community in most of the study period representing 79% of the total cell counts.

The phytoplankton community was found to vary vertically, being dominated by cyanobacteria at the depth ranges of 0 to 5 m in VB Dam (Table 4.4a, Figure 4.2a). Total cyanobacteria abundance varied significantly among the different depth ranges ($F_{2.538} = 20.83$, $p < 0.001$).

Chl*a* also varied significantly as a function of depth ($F_{2.147} = 37.06$, $p < 0.001$), and was found to be higher at the 0 to 5m depth range compared to the 5 to 10m ($p = 0.002$) and 10 to 20 m ($p < 0.001$) depth ranges (Table 4.3a).

Temporal variations with seasons were also found and cyanobacteria dominated the phytoplankton community during the dry season of spring (25,416cells/mL) compared to the summer rainy season (9,714cells/mL) in the VB Dam (Table 4.4b, Figure 4.). In the depth ranges of 0 to 5m and spring season, *Microcystis* dominated the cyanobacteria followed by *Dolichospermum* (Figure 2b). Significant variation associated with water depth was observed in the abundance of both *Dolichospermum* ($F_{2.538} = 27.11$ $p < 0.001$) and *Microcystis* ($F_{2.538} = 11.43$, $p < 0.001$), whereas *Merismopedia* abundance did not vary among the depth ranges ($F_{2.538} = 1.35$, $p = 0.26$). Applying the seasonal means, yearly cyanobacteria cells in the VB Dam dominated at all the depth categories but were more concentrated at the 0 to 5m depth range throughout the 17 year study period (Figure 4.3b).

Cyanobacteria cell numbers were significantly higher in the SWP Dam relative to VB Dam ($F_{1.949} = 875.43$, $p < 0.001$) (Table 4.4a,b, Figure 4.2a). Moreover, significantly higher cell counts of *Dolichospermum* ($F_{1.949} = 155.84$, $p < 0.001$), *Microcystis* ($F_{1.949} = 829.64$, $p < 0.001$) and *Merismopedia* ($F_{1.949} = 75.10$, $p < 0.001$) were observed in the SWP Dam relative to the VB Dam (Table 4.5a,b, Figure 4.2b).

Table 4.5. The average ± standard deviation of the abundance of cyanobacteria species observed in SWP and VB dams at (a) different depth ranges and (b) among seasons.

(a)

Dams	**Depth ranges**	***Dolichospermum*** **(cells/ml)**	***Cylindrospermopsis*** **(cell/ml)**	***Merismopedia*** **(cell/ml)**	***Microcystis*** **(cell/ml)**	***Oscillatoria*** **(cell/ml)**
SWP Dam	0 to 5m	16 655 ± 67 699	321 ± 1 615	5 356 ± 18 437	67 921 ± 145 498	12 ± 140
	5 to 10m	15 916 ± 39 976	520 ± 2 150	4 347 ± 15 279	86 578 ± 126 867	702 ± 8 515
	10 to 20m	7 442 ± 12 149	308 ± 1 457	7 109 ± 18 199	50 546 ± 105 331	2 ± 21
VB Dam	0 to 5m	5 104 ± 13 612	127 ± 931	372 ± 1 514	20 631 ± 146 475	19 ± 181
	5 to 10m	2 437 ± 6 986	97 ± 981	258 ± 1 122	3 524 ± 13 388	124 ± 1 376
	10 to 20m	756 ± 2 674	81 ± 1 106	1 599 ± 12 118	1 528 ± 6 809	3 ± 46

(b)

Dams	**Seasons**	***Dolichospermum*** **(cells/ml)**	***Cylindrospermopsis*** **(cell/ml)**	***Merismopedia*** **(cell/ml)**	***Microcystis*** **(cell/ml)**	***Oscillatoria*** **(cell/ml)**
SWP Dam	Autum	2 743 ± 4 266	442 ± 2 235	12 996 ± 2 235	96 267 ± 156 646	-
	Spring	22 890 ± 85 142	333 ± 1 484	2 608 ± 18 559	97 370 ± 133 018	-
	Summer	21 681 ± 49 748	390 ± 1 492	5 502 ± 11 827	33 945 ± 57 724	719 ± 8 603
	Winter	5 683 ± 9 229	380 ± 1 937	1 855 ± 9 152	76 527 ± 156 657	15 ± 160
VB Dam	Autum	3 458 ± 7 363	396 ± 2 249	1 389 ± 2 800	2 049 ± 4 796	20 ± 145
	Spring	1 817 ± 6 484	-	11 ± 63	23 535 ± 183 298	10 ± 100
	Summer	3 819 ± 12 634	62 ± 476	1 407 ± 12 384	3 993 ± 22 366	100 ± 1 214
	Winter	934 ± 1 939	15 ± 113	145 ± 581	8 297 ± 58 085	-

Using the Shannon's Diversity Index (H), the species richness of the phytoplankton community considering the dominant taxonomic group, which is cyanobacteria, was calculated for the two dams. The VB Dam was characterized by low species diversity as indicated by Shannon's Diversity Index H = 0.4. Similarly, SWP Dam did also have a low species diversity of H = 0.3.

The PCA and associated triplot indicates that phytoplankton abundance and water quality parameters varied among the two dams investigated – based on the segregation in ordinal space (Figure 4.4). The chemistry parameters Chl *a*, pH, TN, and OP were positively correlated with phytoplankton indicators including total phytoplankton cells, total cyanobacteria, and the abundance scores of various phytoplankton groups investigated (Figure 4.4). In addition, TN, OP, and suspended Chl *a* as well as the majority of phytoplankton groups studied associated positively with the SWP data points representing both seasons and depth classes except for the wet season 5–10 m class (Figure 4.4).

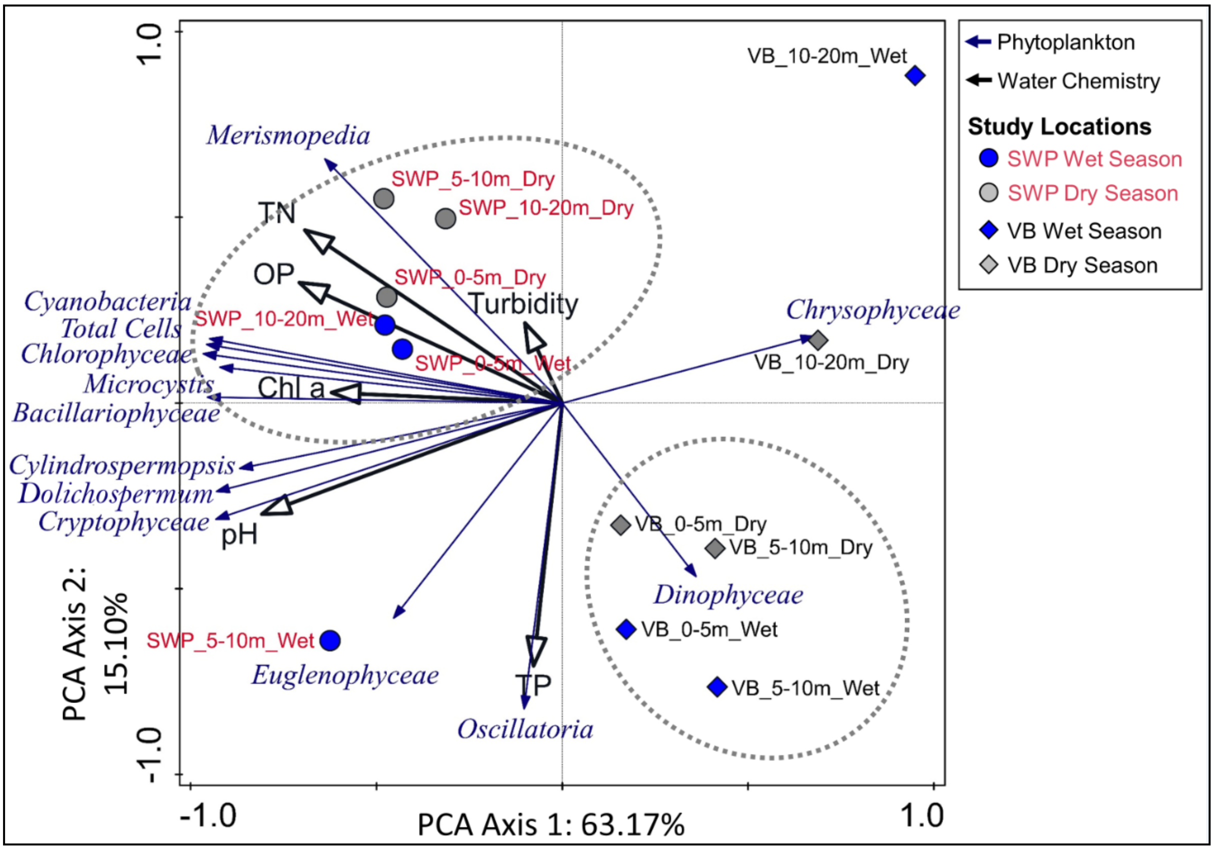

Figure 4.4. Principal component analysis (PCA) triplot indicating the associations between selected phytoplankton classes and genera, total phytoplankton cell number, and selected water quality parameters at different depth range in the SWP and VB dams.

4.4. Discussion

Aquatic ecosystems exhibit spatial and temporal variability in the phytoplankton community. The dominance of a particular species is related to changing environmental conditions. In the current study, the phytoplankton community was dominated by cyanobacteria and, in particular, mainly the species of *Microcystis* followed by *Dolichospermum* in both dams, at all the studied depth ranges and seasons. Vertical variation as a function of depth was found in both dams, and cyanobacteria was the dominant phytoplankton species at all depth ranges. However, temporal variations with dry and wet seasons was only found in the SWP Dam, and not in the VB Dam. The high cell numbers of cyanobacteria in the SWP Dam could be due to its eutrophic status as compared to the mesotrophic status of the VB Dam during the study period.

The results of the current study are similar to other reports in the literature which justify the dominance of *Microcystis* in subtropical lakes with favorable water temperature and elevated nutrient levels caused by partially treated sewage and industrial effluent (Jeppesen et al., 2017; Kassem et al., 2020; Mbukwa et al., 2012; Oberholster & Botha, 2010; T. Wu et al., 2013). The dominance of cyanobacteria in the subtropical Lake Chivero, in Zimbabwe and Hartbeespoort Dam in South Africa, has previously been reported (Ballot et al., 2014; Bittencourt-Oliveira et al., 2012; Davis et al., 2009; Mbukwa et al., 2012), corresponding to the results of the present study featuring two water bodies which are also located in a subtropical region although also semi-arid. A study by Kassem et al., (2020) on the subtropical semi-arid Lake Nasser in Egypt revealed similar findings of the dominance of the phytoplankton community by cyanobacteria species of *Microcystis*. In addition, Ballot et al., (2014), reported that cyanobacteria species of *Microcystis* dominates the Hartbeespoort Dam in South African, surface water. In fact, the dominance of the phytoplankton community by cyanobacteria species of *Microcystis* is common in subtropical lakes, but less common in tropical lakes where other species like *Cylindrospermopsis* are present (Davis et al., 2009; Haakonsson et al., 2017; Mbukwa et al., 2012; T. Wu et al., 2013). Throughout the study, both dams exhibited thermal variation in the water temperature at the three depth ranges and seasons, which could be due to the desert climate conditions in the region. This could have contributed toward the depth profiles and seasonal dynamics of the phytoplankton communities, and the concentration of phytoplankton cell numbers at favorable depth ranges.

Cyanobacteria species are reported to dominate other phytoplankton species due to elevated water temperature of >15 °C (Noori et al., 2018, 2021). The two dams' water temperatures were above 15 °C at all the depth ranges, and across seasons. Ndebele-Murisa et al., (2010) states that the increase in water temperature due to climate warming will shift the phytoplankton species composition from Chlorophyceae to Cyanophyceae. Haakonsson et al., (2017) reported the dominance by *Microcystis*, which was caused by the increase in water temperature in subtropical lentic systems. The increase in water temperature causes thermal stratification in lakes, which influences the quality of water in the water column. Noori et al., (2021) reported the strongest thermal stratification occurred during summer in Karkheh Dam Reservoir, when the water temperature difference between the surface and bottom in the reservoirs exceed 18 °C. The thermal stratifications were reported to cause the spatial temporal variation in ammonium and nitrate in Karkheh Dam Reservoir (Reynolds et al., 1987).

At the preferred depth ranges for the habitation of toxic phytoplankton, the water turbidity was slightly lowered compared to the other depth ranges in both dams. These findings support the observations by Jeppesen et al., (2017), on the occurrence of major toxic cyanobacteria blooms of *Microcystis* during the decline in turbidity in subtropical and tropical lakes. The nutrient concentrations in the two dams were suitable for cyanobacteria growths at all the depth ranges and seasons. This was similar to conditions of the nutrient-enriched subtropical lakes of Chivero, in Zimbabwe, and the South African dams of Hartbeespoort, Erfenis, and Allemanskraal (Mbukwa et al., 2012). Noori et al., (2021) reported a rapid transition from oligotrophic to eutrophic causing water deterioration of the Sabalan Reservoir, Iran, due to external pollution loads (natural and anthropogenic activities), internal pollutant cycling from the sediments, reduced inflows, and reservoir operations strategy. Similar causes may have resulted in water quality deterioration in the two dams presently studied, which are also used for domestic water supply.

The preferred depth range of cyanobacteria in the VB Dam was in the photic zone, and that of the SWP Dam was in the aphotic zone. The findings of the occurrence of increased phytoplankton abundance in the photic and aphotic zones of the two study areas correspond to the observations by Moura et al., (2011), in the Caprina Reservoir, Brazil. However, the assemblages were dominated by multi-species of cyanobacteria unlike in the SWP and VB dams with one dominant species of cyanobacteria. The findings of Moura et al., (2011) revealed that vertical variation was less pronounced than seasonal variation in the cyanobacteria population, while in the current study both vertical and seasonal variations were pronounced in the SWP Dam and only vertical variation in the VB Dam. A limitation of the study by Moura et al., (2011), is however duration as it features only two years of observations.

Bittencourt-Oliveira et al., (2012) reported the dominance of cyanobacteria by *Cylindrospermopsis* during the dry and rainy seasons due to stratification and de-stratification in the Arcoverde Reservoir in Brazil. In the current study, *Microcystis* was found to dominate during both the dry and rainy seasons. These findings support the observations by Kassem et al., (2020), in the Khor Ramla and Khor Abu-Simbel of Lake Nasser where more phytoplankton was reported in the dry season, with spring blooms of *Microcystis*. The observations of the current study corresponds to those of Kassem et al., (2020) under desert climate conditions, with spring blooms observed in both dams with higher numbers of cyanobacteria cells, compared to the other seasons. Blooms were also observed during the dry autumn and winter season in the two dams. In Lake George in the United States of America, dry periods are characterized by dense cyanobacteria blooms (Jeppesen et al., 2017). Lake

Nasser in southern Egypt and northern Sudan, and Lake George in the USA are also subtropical and tropical lakes with similar desert climates like that of the SWP and VB dams.

Graham et al., (2008) state that cyanobacteria maintain the position in the photic zone despite the mixing, and may migrate to different locations within the same zone using buoyancy regulation. The authors further mentioned that the cyanobacteria population distribution was in the proximity of the surface at night, and early morning, followed by movement to deeper water later in the day. This movement is dominated mainly by *Microcystis* (Graham et al., 2008). The findings of the present study support the theory of Graham et al., (2008) as *Microcystis* was found to dominate in the upper water column, although some depths were aphotic. In Hartbeespoort Dam, South Africa, *Microcystis* was reported to dominate at the depth of 5 m, which was a layer with minimal light due to hyperscum covering the water surface (Robarts & Zohary, 2010). Oberholster & Botha, (2011) further state that the concentration of *Microcystis* at the low light strata results in no reverse buoyancy. Reynolds et al., (1987) stated that *Microcystis* generally dominate subtropical systems. This is due to diel alteration of thermal stratification and mixing conditions (Reynolds et al., 1987, 2002). Therefore, the abstraction of water with potentially higher toxic cyanobacteria abundance could pose challenges to water transfers between dams and potable water production due to the buoyancy regulation of *Microcystis*.

Chemical pollution is a further factor to account for as part of water extraction management practices. Aradpour et al., (2021) reported concentrations of arsenic in the water column and sediment of the Sabalan Dam Reservoir, Iran. A scenario-based risk assessment indicated that water extracted below 10m can pose a human health risk due to higher concentrations of arsenic at depths exceeding 10m (Aradpour et al., 2021). The results of Aradpour et al., (2021) and the present findings suggest that water extraction strategies should account for both hazardous chemical and potentially harmful phytoplankton depth profiles in waterbodies known to be polluted.

The findings in this study on the vertical and temporal dynamics of possible toxic phytoplankton could enhance the traditional selective withdrawal of water from the two dams to ensure that good water quality is abstracted for treatment and transfers at preferred depth ranges in different seasons.

Preliminary investigation has shown microcystin-LR levels exceeding 1µg/L (World Health Organisation drinking water guideline) at certain times of the year in both SWP and VB dams (data not presented). However, a more in depth investigation is needed to assess variation in cyanotoxin levels among different depth ranges in the aforementioned reservoirs, as well as associations with cyanobacteria assemblages

4.5. Conclusion and recommendations

In conclusion, the phytoplankton communities in the two dams were dominated by toxic cyanobacteria, mainly by *Microcystis*, followed by *Dolichospermum* at all the depth ranges and seasons. The dry seasons of autumn, spring, and winter were characterized by increased phytoplankton cell numbers compared to the wet summer season. Spring and autumn cyanobacterial blooms were observed in the two dams with more phytoplankton cell numbers compared to the dry winter and autum. In the SWP Dam, the preferred depth ranges by the potentially toxic phytoplankton was at 5 to 10m, and in VB Dam the preferred depth was shallower, being 0 to 5m. The depth ranges preferred by cyanobacteria corresponded to the depths at which favorable water temperature and nutrient concentrations for phytoplankton growth was observed.

Given the vertical and temporal dynamics in the phytoplankton community of the dams, we recommend the following: firstly, the traditional selective withdrawal at varied depths to be enhanced with consideration of vertical and temporal dynamics of possible toxic cyanobacteria to ensure the abstraction of good quality water with minimal numbers of potentially toxic phytoplankton species from both dams. The 5 to 10m depth range should be avoided in the SWP Dam during the dry seasons. The depth range 0 to 5m should be considered in all the seasons and if possible also the depth range of 10 to 20m. While in the VB Dam, the depth range of 0 to 5m can be considered for abstraction in all the seasons except in the spring season. The other VB Dam depth ranges of 5 to 10m and 10 to 20m should be considered throughout the year.

Secondly, the utility manager may consider preparing for a possible eruption of toxins from the dominant toxic cyanobacteria species such as *Microcystis* and *Dolichospermum*. Because the water extracted from SWP Dam is mixed with VB Dam water before treatment, consideration for continuous cyanotoxin testing during the dry seasons of autumn, spring, and winter could ensure the safety of water produced at the VB Dam treatment plant for the capital city of Windhoek, Namibia.

Thirdly, the transfer of nutrient rich and salt water from SWP to VB dams needs to be monitored over time.

Fourthly, during water transfer from SWP to VB dams, the operation strategy should consider for sufficient water for dilutions of pollutants emanating from internal cycling from the sediment bottom of the SWP Dam.

Fifthly, future studies are required to further research (a) the variation of cyanobacteria toxin at the different depth ranges of the two dams, (b) water withdrawal influence on internal nutrients cycling from the bottom of the sediments.

CHAPTER 5: LONG-TERM STUDY OF THE DROUGHT IMPACT ON THE PHYTOPLANKTON IN TWO WATER SUPPLY RESERVOIRS IN THE SUB-SAHARAN AFRICA

This research chapter is under review for publication by the African Journal of Aquatic Science

Declaration by the candidate

With regard to Chapter 5, the nature and scope of my contribution were as follows:

Nature of contribution	Extent of contribution
Conceptual design, fieldwork, experimental work, manuscript writing, data curation, formal analysis, methodology	80%

The following co-authors have contributed to Chapter 5:

Name	Email address and institutional affiliation	Nature of contribution	Extent of contribution
Gideon Wolfaardt	Stellenbosch University Water Institute, Faculty of Science, Stellenbosch University, Matieland 7600, South Africa	Conceptual design, manuscript writing-review & editing.	20%
Paul Oberholster	Centre for Environmental Management, Faculty of Natural and Agricultural Sciences, University of the Free State, Bloemfontein 9300, Republic of South Africa	Conceptual design, manuscript writing-review & editing	
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Abstract

Changes in global and regional climate patterns, especially higher temperatures and associated greater variability in dry and wet season maxima, are expected to have notable influence on dams and other surface waters in terms of volumes and timing of water availability. We hypothesized that the decrease in reservoir volumes due to drought will favour the growth of phytoplankton biomass measured as Chl *a* in dams with desert climate conditions. Using the threshold levels method of Q50, 10 drought years were recorded in the VB Dam in comparison to 7 in the SWP Dam. Both dams had significant reduction in vol % (percentage of full capacity), total phosphate (TP), and orthophosphate (OP) concentrations, but higher water temperature, light availability, pH, electrical conductivity, chloride, ammonia, and total nitrogen during the drought years. In both drought and rainy years, using TP concentration as criterion, SWP Dam was classified as hypertrophic (>0.4mg/l), while VB Dam as eutrophic (>0,2mg/l). The Chl *a* in SWP was 81µg/l in drought and 48µg/l in rainy years, compared to VB 21µg/l in drought and 24µg/l in rainy years, respectively. Average cyanobacterial cell counts in SWP were 129 704 and 77 838 cells/ml in drought and rainy years, and 5 925 cells/ml and 20 070 cells/ml in drought and rainy years, respectively in VB. The cell numbers of the *Microcystis* numbers were notably higher in SWP (85 285 and 58 262cells/ml in drought and rainy years) compared to VB (1 097 and 17 153cells/ml in drought and rainy years, respectively). Decreases in phytoplankton biomass, total cyanobacteria, and *Microcystis* were observed in the SWP but not in the VB. The pattern and magnitude of the statistically significant responses (t-test, $p < 0.05$) varied among the drought and rainy years , therefore, we can conclude that data regarding other environmental drivers, such as land use, will be needed to further elucidate the response of water quality to droughts.

5.1. Introduction

Global patterns of temperature and precipitation have significantly changed over the last century (Rocha Junior et al., 2018; Ziervogel, 2018). In Africa, the observed and projected rate of surface temperature has increased more rapidly than the global averages, with human-induced climate change being the main driver (Ziervogel, 2018; IPCC, 2021). An ambient temperature increase of at least 2 °C is predicted in southwestern Africa where Namibia is found, in combination with a decrease in mean precipitation, but accompanied by heavy precipitation and pluvial flooding, interspersed with more intense drought periods (IPCC, 2021).

Increases in global mean surface temperature will exacerbate water scarcity in semi-arid regions like Namibia (Ziervogel, 2018; IPCC, 2019; Matchaya et al., 2019). The vulnerability of populations in the semi-arid regions in southwestern Africa to water scarcity is projected to impact 178 million people by 2050 at average increase of 1.5°C, 220 million people at a 2°C increase, and 277 million people at a 3 °C increase, respectively (IPCC, 2019). Concurrent increases in human population numbers and income, and changes in consumption patterns, will further result in increased water demand in 2050 (IPCC, 2019). These changes will pose new demands to water providers, and more broadly, water security (Kousari et al., 2014; IPCC 2019, 2021; Matchaya et al., 2019; Haghighi et al., 2020).

Water quality also has an impact on water scarcity when considered from the perspective of the availability of water of sufficient standard to meet humanity's needs (Ziervogel, 2018; Matchaya et al., 2019; IPCC 2021), with the Intergovernmental Panel on Climate Change fifth assessment report also pointing out that climate change will reduce raw water quality that will pose risks to humans if conventional treatment strategies are not adapted to these new realities (Jiménez Cisneros et al., 2014)

The impact of climate change on water quality can be related to extreme phenomena such as an increased frequency of hydrological droughts, rising in temperatures, and intense rainfall (Delpla et al., 2009; Nosrati 2011). These events play an important role in regulating nutrients in lakes (Rocha Junior et al., 2018) for example decreases in reservoirs' water levels affect physiological, chemical, and biological variables that may favour toxic phytoplankton growth

(Delpla et al., 2009), and intense rainfall and associated increased soil erosion accelerate the inputs of phosphorus and nitrogen in reservoirs, causing eutrophication (Rocha Junior et al., 2018).

Decrease in reservoir water levels is known to influence the growth of cyanobacteria (Paerl et al., 2008; Paerl and Otten, 2013; Paerl 2018). For example, Maggiore Lake, Italy, Eymir Lake, Turkey, and Xeresa Lake, Spain were found to experienced cyanobacteria blooms due to drought-induced decreases in water levels (Bakker and Hilt, 2016). A study by Huang et al. (2020) reported that Chl *a* was decreasing with the increase in water level fluctuations. In another study, it was observed that the magnitude, spatial, and temporal distribution of the cyanobacteria *Microcystis* increased during the drought period and decreased during the rainy period (Lehman et al., 2019). Rocha Junior et al. (2018) reported that long periods of drought in semi-arid regions and associated reduced water levels in reservoirs and lakes, and higher nutrient concentrations in the water column led to increases in *Microcystis* cell numbers. Similar findings were made by others, e.g., the study by Brasil et al. (2016) on 40 man-made lakes in a warm semi-arid region of Brazil, that showed increases in toxic phytoplankton in the dry season associated with reduced water levels.

Limnological variables such as water temperature, light intensity and salinity are reported to favour of the growth of toxic cyanobacteria during periods of drought. In lake Taihu, China, *Microcystis* species were dominant during a warm period with water temperature ranging from 18.2℃ to 32.5℃ (Chen et al., 2003). Some genera of cyanobacteria such as *Oscillatoria* can be inhibited when exposed to an extended period of high light intensity, but these microorganisms tend to dominate under mixed water conditions with low irradiance <0.20 ratio Zeu/Zmix (Havens, n.d.; Chorus and Bartram, 1999). Unlike *Microcystis* and *Dolichospermum*, *Oscillatoria* were less sensitive to high light intensities (high ratio Zeu/Zmix) since their enhanced buoyancy regulation enable them to locate light conditions that are optimal for their growth (Chorus and Bartram,1999).

A study by Barinova et al. (2009) on 34 lakes in the Karasu River in Kazakhstan revealed low diversity in phytoplankton communities due to high salinity. A previous study by Costa et al. (2016) also reported the dominance of cyanobacteria in shallow lakes of the Pantanal region of Brazil during the drought periods due to an increase in salinity, the cyanobacterium *Dolichospermum* was the major-forming genus.

Even though multiple studies were done in semi-arid regions on the impact of drought events on phytoplankton growth in reservoirs (Rocha Junior et al., 2018; Leite and Becker, 2019), there is a paucity of reports in regions with a desert climate such as Namibia, especially in terms of the extent that drought conditions have on the growth of algae biomass, and more specifically, cyanobacterial blooms during dry and wet years. We have recently reported a higher occurrence of toxic cyanobacteria blooms in the SWP Dam compared to the VB Dam, even though the two dams are on the same ephemeral river (Sirunda et al., 2021b). Sirunda et al. (2021a) further reported on toxic cyanobacterial dynamics with season and varying depth in the two dams. The current study aimed to build on these earlier reports by investigating the effect of prolonged drought on phytoplankton biomass (Chl *a)* and cyanobacteria in these two dams.

5.2. Materials and methods

5.2.1. Study area

The two man-made dams are located in a warm semiarid region, in central Namibia. The VB (21° 59'59.27'' S 16° 58'54.76" E) and SWP dams (22° 12'44.31" S 16° 31'44.97" E) are monomictic man-made lakes that are stratified throughout the year and experience overturn or mixing during winter (Sirunda et al., 2021b) (Figure 5.1). The dams are situated in the Omaruru-Swakop Basin, in the Upper-Swakop sub-basin with a desert climate, which is characterized by large differences in day and night temperature, erratic rainfall, frequent droughts, and high evapotranspiration (Table 5.1). In the Upper – Swakop catchment area, land use including various industries, mining, commercial farms for livestock, game farming and lodges, informal settlements, Wastewater Treatment Plants, and a coal-fired power station.

Highest rainfall in the basin occurs in the months of February and March, and again from September to December (Mendelsohn et al. 2002) (Figure 5.2). The SWP dam receives more inflow compared to the VB Dam, due to the difference in catchment size (Table 5.1: Figure 5.2). From the rainfall received, it is estimated that only 2% of the water ends up as surface run-off and 1% becomes available to recharge groundwater, 83% is lost through evaporation, and 14% is lost through evapotranspiration (IWRM Plan Joint Venture Namibia 2010). The average annual temperature of the study areas is 18°C to 20°C, with average maximum temperature during the hottest months (October to February) being 30℃ and 32°C, respectively (Mendelsohn et al., 2002).

VB and SWP dams are the main water sources of the three dam system, which supply water to central Namibia. The three dam system consists of the SWP Dam situated at 90 km, VB Dam situated at 70 km, and Omatako situated at 160 km, from the capital Windhoek (Scott et al., 2018). The system is owned, operated, and maintained by NamWater as a bulk water supplier (Rensburg 2006, Scott et al., 2018). Within the three dam system, VB Dam is used as a recipient dam due to its location near the major drinking water treatment plant and lowest relative water loss due to evaporation (Sirunda and Mazvimavi, 2014). VB Dam is augmented with water from SWP Omatako Dams when its water volume is below 60%. The transfer of water from SWP Dam to VB Dam is only when required and only if 10% of the available water is permitted for transfer, whereas the remaining is for other consumers. The Omatako Dam receives river water from the Omatako River and groundwater from the Karst Aquifer, transported through a canal (Grootfontein-Omatako Eastern National Water Carrier) (Slabbert, 2007; Sirunda and Mazvimavi, 2014; Lewis et al., 2019). The total safe yield water capacity of the three dams is about 20 Mm^3/a based on the 95% assurance of supply to the capital Windhoek (Biggs and Williams, n.d.; Lewis et al., 2019). Under typical conditions, 70% of the city of Windhoek's water demand is acquired from these three dams, while the remaining demand is supplied from ground- and reclaimed water (Scott et al., 2018).

Besides the main water supply to the city of Windhoek, others such as Okahandja Town, Karibib Town, Otjihase Copper Mine, Otjimbingwe Village, and Navahacb Gold Mine are also supplied from these dams (Slabbert, 2007; Scott et al. 2018). The SWP Dam was built in 1977, with a maximum depth of 30m, and VB Dam, built-in 1970, with a maximum depth of 29m (Table 5.1). The two dams are situated on the Swakop River, which is an ephemeral river (Table 5.1). SWP Dam has the largest capacity (63.5Mm^3), with a catchment area of 5 480 km^2 and low annual rainfall of 350 mm per annum (Table 5.1). VB Dam has a lower evaporative loss per surface area of 2 254 mm per annum compared to the SWP Dam (Table 5.1).

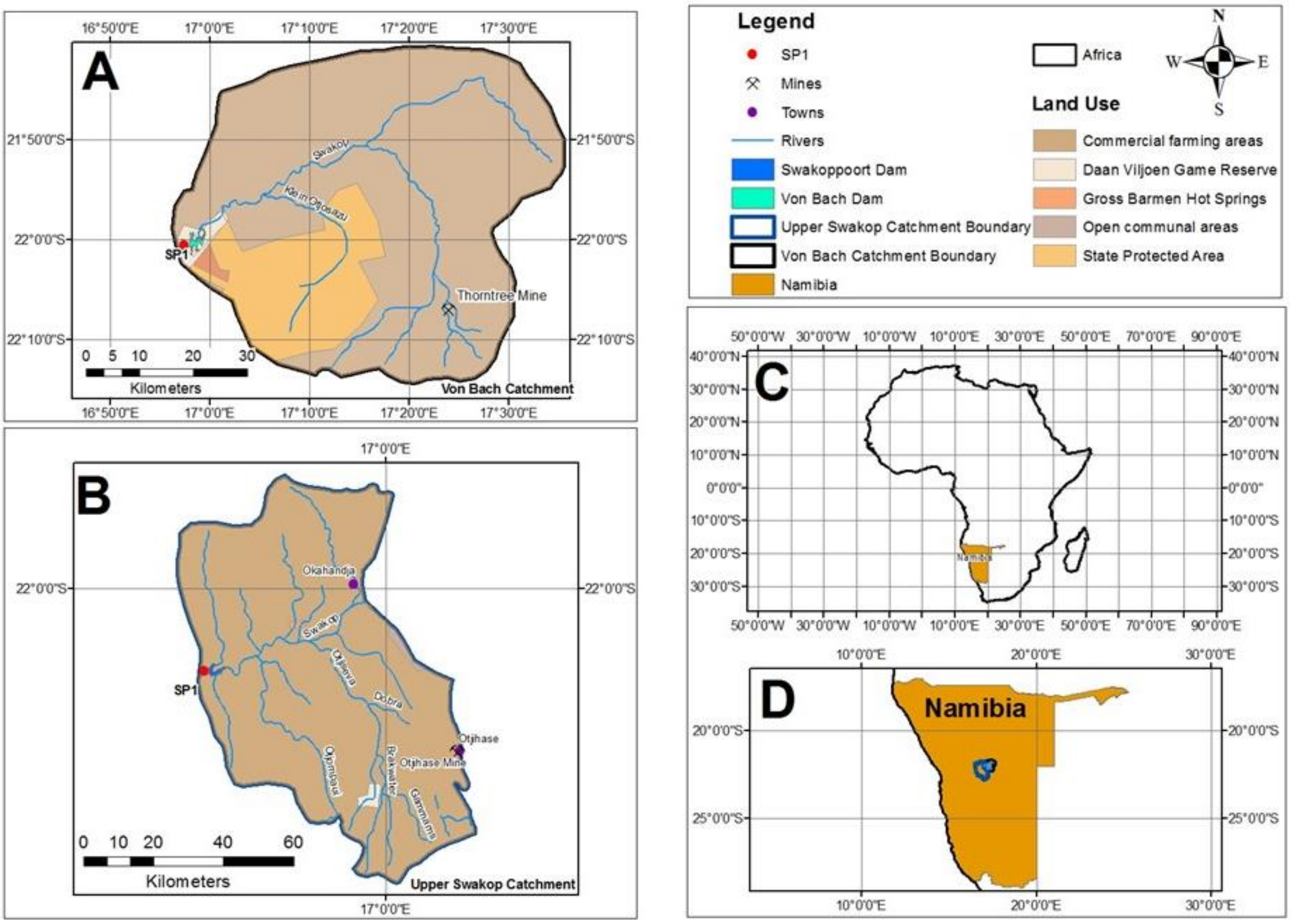

Figure 5.1. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset indicating, a) the location of the VB Dam in its sub-catchment area with available land use activities; b) the location of the SWP Dam in its sub-catchment area with available land-use activities; c) location of Namibia within Africa, and d) location of the two dams in Namibia,

Table 5.1. General features of the two dams found on the Swakop River.

Dams	**Basin**	**Subbasin**	**Capacity (Mm^3)**	**Max. Depth (m)**	**Evapo.Losses (mm/a)**	**Ann.rainfall (mm/a)**	**Surface area (FSC) (km^2)**	**Catchment area (km^2)**
VB Dam	Omaruru - Swakop	Upper-Swakop	48.56	29	2254	370	4.89	3.1
SWP Dam	Omaruru - Swakop	Upper-Swakop	63.48	30	2275	350	7.81	5.5

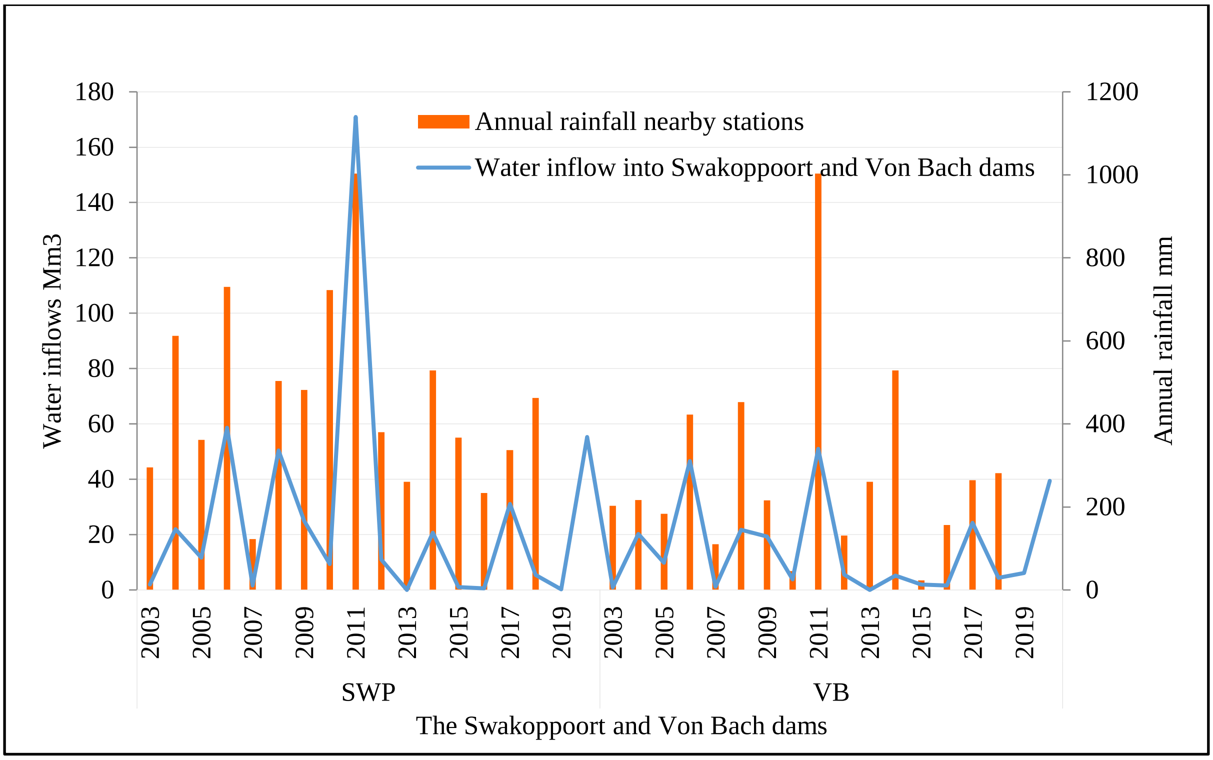

Figure 5.2. Streamflow of the Swakop River into SWP and VB dams over the study period of 2003-2019, in relation to annual rainfall at nearby meteorological services station around the two dams.

5.2.2. Field sampling and analysis

Historical data were obtained from an established sampling site located nearby the dam wall in each dam between 2003 and 2019 on a monthly interval using the dip sampling method, using a Von Dorn 5L water sampler and in acid washed sample containers (Burns et al. 2000). The water sampling was always performed between 09:30 and 14:00 at the respective depths, taking depth at the time of sampling into consideration. Water temperature (WT) was measured on-site using a digital thermometer (Yellow Springs Instrument, Model 54A, Yellow Springs, OH, USA) with a range of −5°C to +45°C, accuracy of ±0.7°C, at 0.5m intervals under the water column. Secchi depth (SD) was measured using a standard 20 cm diameter black and white Secchi disk at each sampling site in each dam. Water analyses included total phosphate (TP) n=908, total nitrogen (TN) n=908, chlorophyll a (Chl *a)* n=885, turbidity n=908, chloride (Cl) n=337, electrical conductivity (EC) n=350, and ammonia (NH_3) n=908. Sampling also included Copper (Cu^{2+} n = 352) in view of copper mining activities upstream in the respective catchment areas. Copper concentration was compared with the Target Water Quality Range (TWQR) for aquatic ecosystems according to the Department of Water Affairs and Forestry (DWAF, 1996).

The water samples (n =975) for phytoplankton analyses were collected in 1L containers at different water depths (surface to bottom), in 1L plastic bottles for chemical analyses, and 100 ml dark brown glass bottles covered with aluminium foil for suspended Chl a. All field were transported on ice and analysed within 24 hours.

Monthly data on both dams' volumes and historical streamflow data of the Swakop River were obtained for 2003-2019 the study period from the NamWater hydrology department, and precipitation data from the Okahandja meteorological station (Namibian Meteorological Office, Windhoek).

5.2.3. Water quality analyses

All the water quality parameters were analyzed according to the methods described in the American Public Health Association (APHA, 1998) (Table 5.2).

Table 5.2. Water quality parameters analysed and APHA methods

Water Quality Parameters	Methods
turbidity	APHA, 1998: part 2130 B
pH	APHA, 1998: part 4500 H B
OP and TP	APHA, 1998: part 4500 P E
total nitrogen	APHA,1998: part 4500 NO3-E
The ammonia	APHA,1998: part 4500 – NH_3 C
Electrical conductivity	APHA,1998: 2510 B
Suspended Chl *a*	APHA,1998: part 10200 H
Chloride	APHA,1998: 4500-Cl^- E
Copper	APHA,1998:3500-Cu

To identify phytoplankton (cyanobacteria and *Microcystis*), samples were sedimented in a Sedgewich–Rafter counting chamber and analysed under an inverted microscope at 400× magnification using the strip-count method (APHA 1992). All phytoplankton were identified according to Truter, (1987), Wehr & Sheath, (2003), Van Vuuren et al. (2006), and Taylor et al. (2007).

5.2.4. Data analysis

Designation of hydrological drought years, when the dams received reduced inflow than normal due to precipitation shortfalls was established based on the selected threshold level method of Q50 (Rivera et al., 2017; Tallaksen et al., 2004). This method entails defining a threshold below which the flow of a river is considered as a drought (Tallaksen et al., 2004). This method is used to compare hydrological, physiological, chemical and biological variables during drought and rainy years (Kousari et al., 2014). Various studies indicate that a threshold is selected in different ways and most often based on the type of water deficit in the area of study. Hisdal et al. (2004) and Meigh et al. (2002) indicate that threshold values of Q70 and Q90 are considered reasonable for perennial rivers but not for an ephemeral river like the Swakop River. Historical streamflow data for inflow into the VB and SWP dams was collected at the Ministry of Agriculture Water and Land Reform (MAWLR). The severities and durations for drought events were then calculated at the selected threshold level of Q50. The streamflow duration curve was then developed. The Q50 in VB was at 7.6Mm^3 and 9.4Mm^3 in SWP Dam. There was notable variation in duration and severity of drought years.

Summer (November till March) is the wet season in the region, and the dry seasons were autumn, winter, and spring. The euphotic depth (Zeu) was calculated as 2.7 times the measured Secchi depth at each sampling site (Chorus and Bartram 1999, Dantas et al., 2011; Yang et al., 2019). The mixing depth (Zmix) was estimated from measured temperature profiles at a gradient of 1°C for both dams as explained in (Chorus and Bartram, 1999, Coloso et al., 2008; Dantas et al., 2011; Yang et al., 2019; Gray et al., 2020). The temperature gradient for the estimation of the Zmix is more widely used than the density gradient (Gray et al., 2020b). In the winter season, the Zmix was taken as the maximum depth of the sampling sites due to the absence of a temperature gradient. The ratio between the euphotic depth (Zeu) and Zmix was used as a measure of the availability of light in the water column for phytoplankton photosynthesis (Havens, n.d.; Khanna et al., 2009; Dantas et al., 2011; Yang et al., 2019). The mean value of 0.20 limit ratio of light availability was used as minimum condition indicating that sufficient light is available for phytoplankton to grow / survive(Havens, n.d.; Dantas et al., 2011; Yang et al., 2019).

The TN:TP ratio, TP, and suspended Chl *a* concentrations (Jones et al., 2003; Dodds, 2006; Dodds and Smith, 2016) were used for the classification of the trophic status of the two dams during the rainy and drought years. Chl *a* is commonly used to represent algal biomass as many researchers have done in other studies (Zhang et al., 2019; Huang et al., 2020). The limiting nutrients of the two dams were established using the Redfield ratio of Total Nitrogen: Total Phosphate (TN:TP) of 16:1, where TN:TP >16 is designated as phosphate limited and TN:TP

<16 is designated as nitrogen limited (Redfield, 1958; Guildford and Hecky, 2000; Kim et al., 2019).

5.2.5. Statistical analysis

The open-source software package R (R Core Team, 2021) was used for statistical analysis. Box plots were constructed to visualise the distribution of values for each measurement. Measurements with distributions that are heavily skewed to the right were transformed before including them in the t-tests or regressions (R Core Team, 2021). The following variables were transformed: conductivity (mS/m), TN:TP ratio, turbidity, NH_3, total phosphate (mg/l), orthophosphate (mg/l), suspended Chl *a* (µg/l), *Microcystis* (cells/ml), cyanobacteria (cells/ml). The transformation applied was the natural logarithm (ln), except in cases where the values in a variable started at 0, 1 was first added to all the values of that variable (preserving the minimum value at 0). The difference between the drought and rainy years over the study period of 2003-2019 in relation to water quality variables were assessed using t-tests (α =0.05). For t-test analysis, adjusted p-values were reported along with the regular p-values, where the adjustment was made according to the Holm-Bonferroni method. To explore the relationship between target variables and explanatory water quality variables, multiple linear regression was used. The explanatory variables were depth, drought and rainy years, water volume, conductivity, pH, total nitrogen, turbidity, orthophosphate, and TN:TP ratio. For each target variable, the data from different source data sets were matched. Only values within 60 days of observation were considered as candidates for matching. Furthermore, plots showing the relationships between the target variables and sets of explanatory variables over time were constructed.

We report the outcomes of ordinary linear regression analyses on each target variable. Note that each target variable is first transformed using the natural logarithm before the regression is performed, and the estimated coefficients are then exponentiated. The format is: significant explanatory variable (effect size; p-value), where the effect size is a multiplicative factor (1=no change, <1 denotes a decrease, >1 denotes an increase) (R Core Team, 2021). Numeric explanatory variables are standardised before being entered in the regression model, so that effect sizes can be interpreted in terms of standard deviations (R Core Team, 2021). For example, an effect size of 1.1 indicates that a 1 standard deviation increase in the explanatory variable results in a 10% increase in the target variable, and an effect size of 0.9 indicates a

10% decrease. Note that a drought autumn was used as the baseline and significant period and season differences can be interpreted relative to that (R Core Team, 2021).

5.3. Results

5.3.1. Abiotic environment

Using the threshold levels during the study period from 2003-2019, ten (10) hydrologically drought years (2003, 2007, 2010, 2012, 2013, 2014, 2015, 2016, 2018, and 2019) were recorded in the VB Dam in comparison to seven (7) years (2003, 2007, 2013, 2015, 2016, 2018, and 2019) in the SWP Dam. The dams received no inflow in some of the drought years, SWP Dam received inflow of an average 26Mm^3 in comparison to VB Dam's 14Mm^3 during the study period, with average auunual rainfall of 462 and 281mm in the SWP and VB Dam catchments, respectively.

The t-test results show that the vol % in the two dams was significantly higher in the rainy years compared to the drought years (Table 5.3 and 5.4). The mean vol % in SWP was 36.16% in drough and 69.19% in rain years, in comparison to 49.57% in drought and 78.13% in rainy years in the VB (Table 5.5). The mean WT during the drought years was significantly higher compared to the rainy years in both dams (Figure 5.3a, Table 5.3, 5. and 5.5). In the SWP Dam, the mean WT in the drought years was 21.23°C, while in the VB Dam, the mean WT during the drought years was 18.58°C (Table 5.5). The water pH was found to be significantly higher during the drought years compared to the rainy years in both dams (Figure 5.3b; Table 5.3, 5.4, and 5.5). The was a significant different in the water turbidity in the VB Dam compared to the SWP Dam in both drought and rainy years (Figure 5.3c; Table 5.3 and 5.4). The mean light availability ratio (Zeu/Zmix) for phytoplankton growth was above the limit of 0.20 ratio which is required as sufficient light for phytoplankton growth in both dams (Table 5.5). The SWP mean light availability ratio was higher than that of VB during the both drought and rainy years (Table 5.3 ,5.4 and 5.5). The results also showed that there was no significant difference in the Zmix between the drought and rainy years in both dams (Table 5.3 and 5.4).

SWP had a higher concentration of mean TP, TN, OP, Chl *a*, EC, Cl than VB in both drought and rainy years (Table 5.6). There was a significant difference in TP, OP, Chl *a*, TN: TP, EC, and Cl between the drought and rainy years in SWP Dam (Table 5.3, Figure 5.4a-b and Figure 5.5a,b and d). The mean TP concentration in SWP Dam was 0.23mg/l in the drought years and

0.40mg/l in the rainy years, while the OP concentration was 0.07mg/l in the drought years and 0.15mg/l in the rainy years in the SWP Dam (Table 5.6). In the VB Dam, the t-tests showed that there was a significant difference in EC, Cl, TN, and NH_3 between the drought and rainy years (Table 5.4, Figure 5.4a and b, and Figure 5.5c). The mean TP was 0.19mg/l in the drought years and 0.25mg/l in the rainy years in the VB Dam (Table 5.6). The OP was 0.036mg/l in the drought years and 0.04mg/l in the rainy years in the VB Dam (Table 5.6). The suspended mean Chl *a* was 81µg/l in drought years and 48µg/l in rainy years in SWP, while 21µg/l in drought years and 24µg/l in rainy years in VB Dam (Table 5.7).

Based on the mean TP and Chl *a* concentration, the trophic state of the reservoirs varied between hypertrophic and eutrophic, whereas SWP Dam was hypertrophic and VB Dam as eutrophic. The Chl *a* concentration was above 10µg/l, signalling blooms in both dams during the rainy and drought period according to the World Health Organization (WHO) 1999 Guideline Values for Cyanobacteria in Freshwater and Cyanobacteria Incident Management Framework (CIMFs) for applications by drinking water suppliers (Du Preez et al. 2007, WHO 2021). The TN:TP ratio in both dams varied, indicating P limitation in VB Dam, while in SWP Dam P limitation in the drought years and N limitation in the rainy years (Figure 5.5d). Based on the TN:TP ratio, the reservoirs' trophic state varied between eutrophic and mesotrophic between drought and rainy years.

SWP Dam recorded higher counts of cyanobacterial cells than VB Dam in both drought and rainy years (Table 5.7: Figure 5.6a and b). During the drought years, cyanobacterial average cell numbers were 129 704cells/ml in SWP Dam and 5 959cells/ml in VB Dam over a period of 17 years (Table 5.6, Figure 5.7). While during the rainy years, the average cyanobacteria cell numbers were 77 838 in SWP Dam and 20 070 in VB Dam (Table 5.7, Figure 5.7). Conversely, the average *Microcystis* cell numbers were 85 285 in drought years and 58 265 in rainy years in SWP Dam, while VB Dam 10 98 during drought years and 17 154 in rainy years (Table 5.7, Figure5.6 and 5.7). The reported cyanobacteria cell numbers in the current study were > 2000cells/ml and represented blooms with possible adverse health effects in both dams according to the World Health Organization (WHO) 1999 Guideline Values for Cyanobacteria in Freshwater and Cyanobacteria Incident Management Framework (CIMFs) for applications by drinking water suppliers (Preez et al., 2007; WHO 2021). The t-tests showed that there was no significant difference in the cyanobacteria cell numbers between the drought and rainy years in both dams (Table 5.3 and 5.4). However, there was a significant difference in *Microcystis* cells in the VB Dam compared to SWP Dam (Table 5.3 and 5.4). The heavy metal (Cu^{2+}) measured during the study period was above the South African TWQR (>0.0003mg/l) for aquatic ecosystems during the rainy years.

Table 5.3. t-test for differences in the mean hydrological and physio-chemical water quality variables in the SWP Dam between the dry and wet years.

Variables	**Drought years**		**Rainy years**		**t-statistic**	**P-value**
	Mean	SD	Mean	SD		
Vol %	36.159	21.210	69.191	26.180	-22.302	0.000***
Zeu (m)	0.767	0.380	1.917	3.010	-1.443	0.911
Zmix (m)	3.400	3.730	4.933	4.480	-0.755	1.000
Zeu: Zmix (ratio)	0.420	0.310	0.364	0.460	0.304	1.000
Temperature (°C)	21.234	3.490	18.660	3.790	11.207	0.000***
Log Conductivity (log mS/m)	4.763	0.530	4.354	0.630	4.386	0.000***
Chloride (mg/l)	160.016	79.700	79.038	41.760	7.304	0.000***
Depth (m)	5.590	3.760	8.142	4.410	-6.332	0.000***
Log *Microcystis* (log cells/ml)	8.973	3.600	9.436	2.640	-1.437	0.911
Log Cyanobacteria (log cells/ml)	10.559	2.400	10.153	2.170	1.762	0.553
pH	8.612	0.590	8.383	0.500	3.714	0.002**
Log Turbidity (log NTU)	2.797	0.850	2.567	0.740	2.562	0.089
Log Ammonia (log mg/l)	-2.590	1.650	-2.354	1.600	-1.311	0.911
Total nitrogen (mg/l)	2.074	0.880	2.113	1.030	-0.379	1.000
Log Total phosphate (log mg/l)	-1.821	0.710	-1.276	0.720	-6.822	0.000***
Log Orthophosphate (log mg/l	-2.912	0.940	-2.210	1.000	-6.344	0.000***
Log Chl *a* (log µg/l)	3.958	1.060	3.306	1.180	5.317	0.000***
Log TN:TP (log ratio)	2.9448	0.75	2.4364	0.78	5.979	0.000***

** µ = mean, SD = standard deviation, p.adj = Holm-Bonferroni adjusted p. P-value <0.05 indicates a significant difference between the drought- and rainy years for that specific variable*

Table 5.4. t-test for differences in the mean hydrological and Physiological and Chemical water quality variables in the VB Dam between the dry and wet years.

	Drought years		Rainy years		t-	P-value
Variable	Mean	SD	Mean	SD	**statistic**	
Vol %	49.565	20.205	78.134	24.338	-23.575	0.000***
Zeu (m)	2.983	2.124	1.286	1.347	3.984	0.003
Zmix (m)	9.625	3.643	12.129	6.112	-2.244	0.139
Zeu: Zmix (ratio)	0.326	0.254	0.206	0.482	1.424	0.525
Temperature (°C)	18.579	3.943	17.840	3.885	5.755	0.000***
Log Conductivity (log mS/m)	3.590	0.488	3.231	0.240	6.881	0.000***
Chloride (mg/l)	32.365	37.245	11.739	7.680	5.988	0.000***
Depth (m)	7.749	4.663	9.233	5.576	-3.361	0.008**
Log *Microcystis* (log cells/ml)	2.927	3.288	4.075	4.296	-3.475	0.006**
Log Cyanobacteria (log cells/ml)	5.678	3.503	5.606	4.154	0.220	0.826
pH	8.084	0.459	7.868	0.435	5.490	0.000***
Log Turbidity (log NTU)	1.967	0.660	2.493	1.076	-6.470	0.000***
Log Ammonia (log mg/l)	-2.067	1.589	-2.598	1.647	3.638	0.004**
Total nitrogen (mg/l)	1.408	0.869	1.709	0.833	-3.963	0.001**
Log Total phosphate (log mg/l)	-2.112	0.940	-2.004	1.023	-1.207	0.525
Log Orthophosphate (log mg/l	-3.674	0.884	-3.472	0.879	-2.424	0.102
Log Chl *a* (log µg/l)	2.265	1.259	1.970	1.439	2.451	0.102
Log TN:TP (log ratio)	2.799	0.958	2.931	0.961	-1.512	0.525

** µ = mean, SD = standard deviation, p.adj = Holm-Bonferroni adjusted p. P-value <0.05 indicates a significant difference between the drought- and rainy years for that specific variable*

Table 5.5. Physical variables of SWP and VB dams

		Drought Years			Rainy Years		
Dams	Variables	Mean	Minimum	Maximum	Mean	Minimum	Maximum
SWP	vol %	36.16	5.37	84.41	69.19	5.22	104.36
VB		49.57	9.68	90.35	78.13	10.48	101.93

SWP	Zeu (m)	0.77	0.32	1.30	1.92	0.27	11.02
VB		2.98	0.27	7.13	1.29	0.27	6.21
SWP	Zmix (m)	3.40	1.00	9.50	4.93	1.00	13.50
VB		9.63	4.00	16.50	12.13	1.50	20.50
SWP	Zeu: Zmix	0.42	0.09	0.92	0.36	0.01	1.55
VB	(ratio)	0.33	0.03	0.84	0.21	0.01	3.06
SWP	Temperature	21.23	14.00	28.30	18.66	13.00	30.80
VB	(°C)	18.58	7.00	28.00	17.84	4.50	30.00
SWP	pH	8.61	7.30	9.80	8.38	7.10	9.40
VB		8.08	7.00	9.30	7.87	6.70	9.10
SWP	Turbidity	26.58	4.26	341.00	16.64	0.00	216.00
VB	(NTU)	8.85	0.00	244.00	24.40	1.44	402.00

Table 5.6. Chemical variables of SWP and VB dams

		Drought Years			**Rainy Years**		
Dams	Variables	Mean	Minimum	Maximum	Mean	Minimum	Maximum
SWP	Conductivity (mS/m)	132.57	46.60	263.00	100.91	22.70	693.00
VB		41.26	19.70	144.30	26.07	18.50	53.10
SWP	Chloride (mg/l)	160.02	44.00	340.00	79.04	9.00	340.00
VB		32.37	1.00	215.00	11.74	3.00	47.00
SWP	Ammonia (mg/l)	0.67	0.01	55.00	0.29	0.01	3.43
VB		0.33	0.01	2.25	0.26	0.01	2.95
SWP	Total nitrogen (mg/l)	2.07	0.40	5.00	2.11	0.40	6.30
VB		1.41	0.10	6.40	1.71	0.20	4.60
SWP	Total phosphate (mg/l)	0.23	0.03	3.22	0.42	0.03	16.00
VB		0.21	0.01	5.20	0.25	0.01	4.22
SWP	Orthophosphate (mg/l)	0.08	0.01	0.27	0.16	0.01	0.65
VB		0.04	0.01	0.75	0.05	0.01	0.84
SWP	TN:TP (ratio)	23.05	1.19	95.00	14.35	0.00	120.67
VB		24.99	0.27	306.45	28.71	0.71	222.18

Table 5.7. Biological variables of SWP and VB dams

		Drought Years			**Rainy Years**		
Dams	Variables	Mean	Minimum	Maximum	Mean	Minimum	Maximum
SWP	*Microcystis* (cells/ml)	85285.16	0.00	714171.00	58265.11	0.00	1077240.00
VB		1097.67	0.00	63639.00	17153.54	0.00	1810176.00
SWP	Cyanobacteria (cells/ml)	129703.81	0.00	718414.00	77838.19	0.00	1128846.00
VB		5925.14	0.00	178514.00	20070.41	0.00	1810176.00

SWP	Chlorophyll *a* (µg/l)	81.13	0.12	709.00	47.67	0.00	502.00
VB		20.85	0.00	563.46	24.28	0.00	1126.00

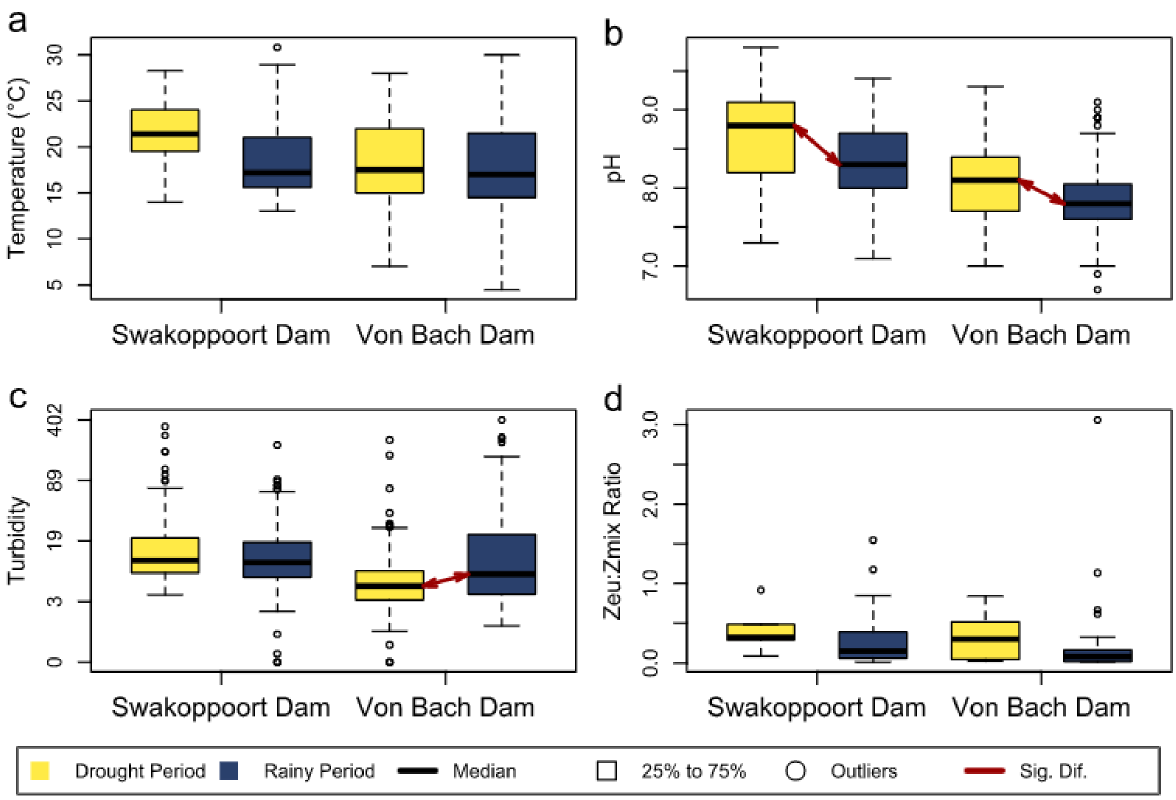

Figure 5.3. Box plot comparisons of the physical variables recorded during the study period

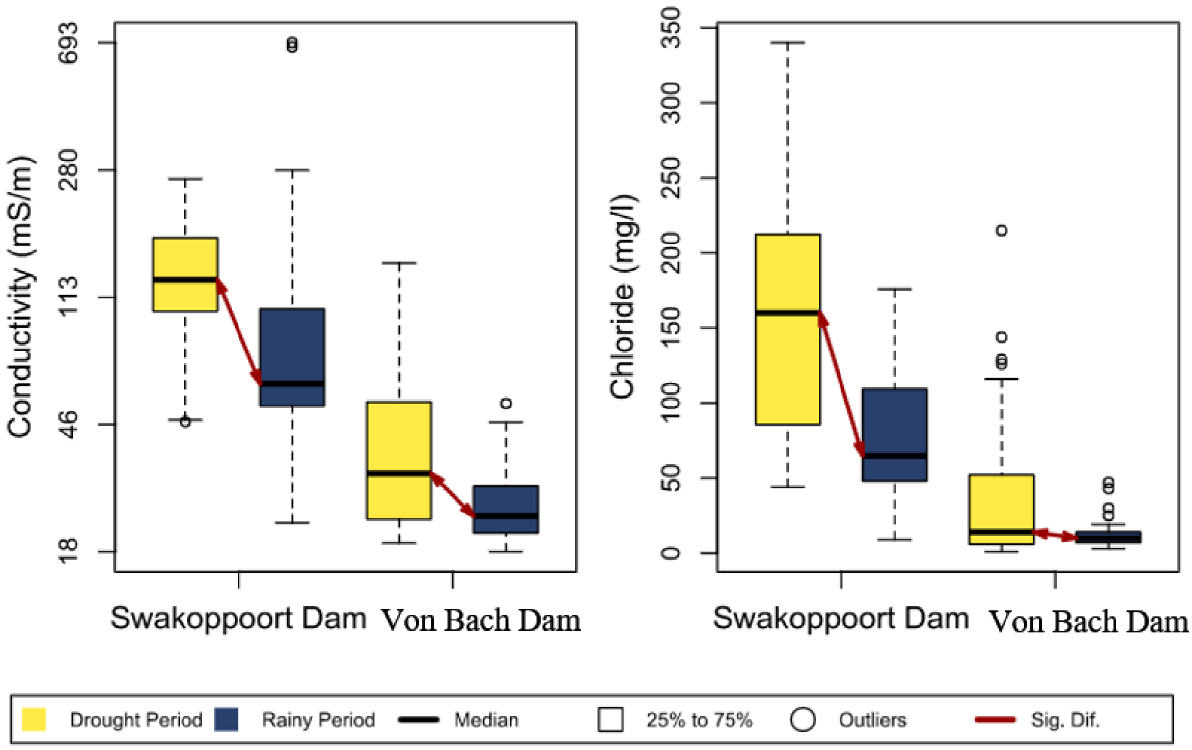

Figure 5.4. Box plot comparisons of the chemical variables recorded during the study period

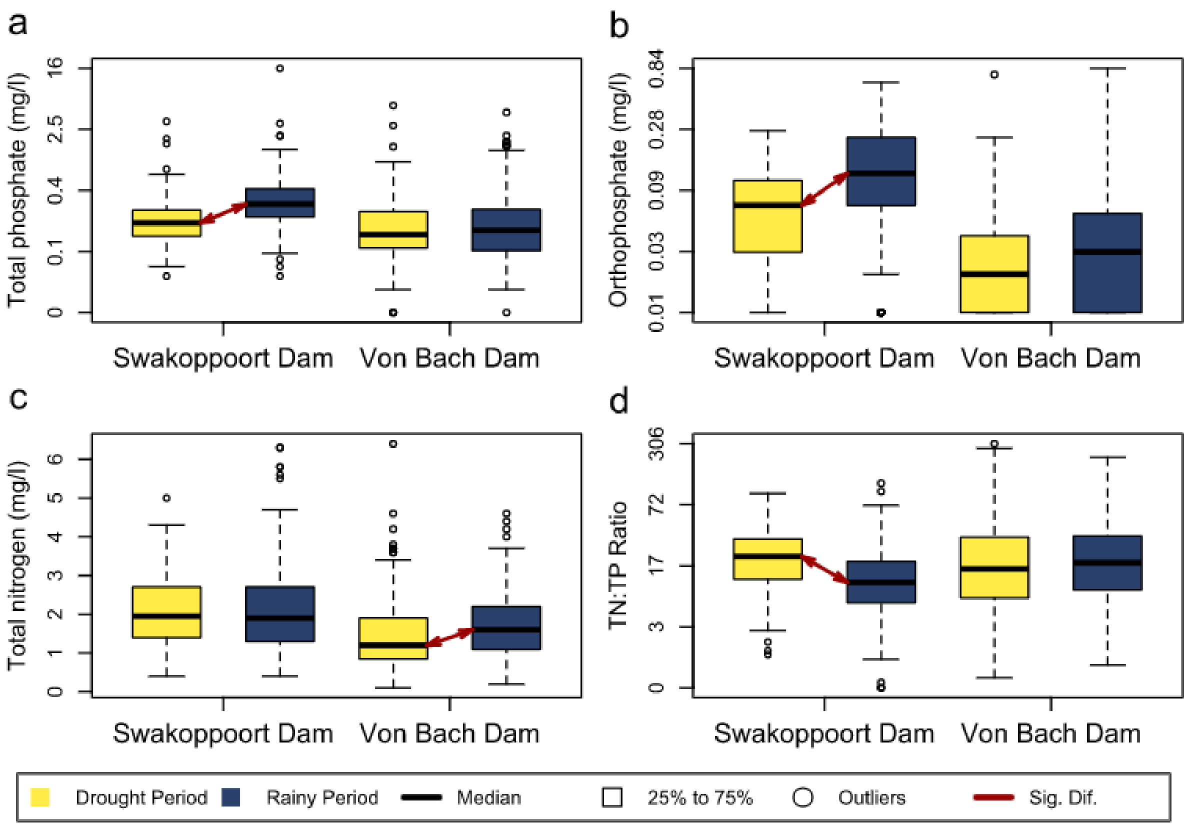

Figure 5.5. Box plots of (a) total phosphate, (b) orthophosphate, (c) total nitrogen, and (d) the ratio of nitrogen to phosphate in the two dams between rainy and drought years.

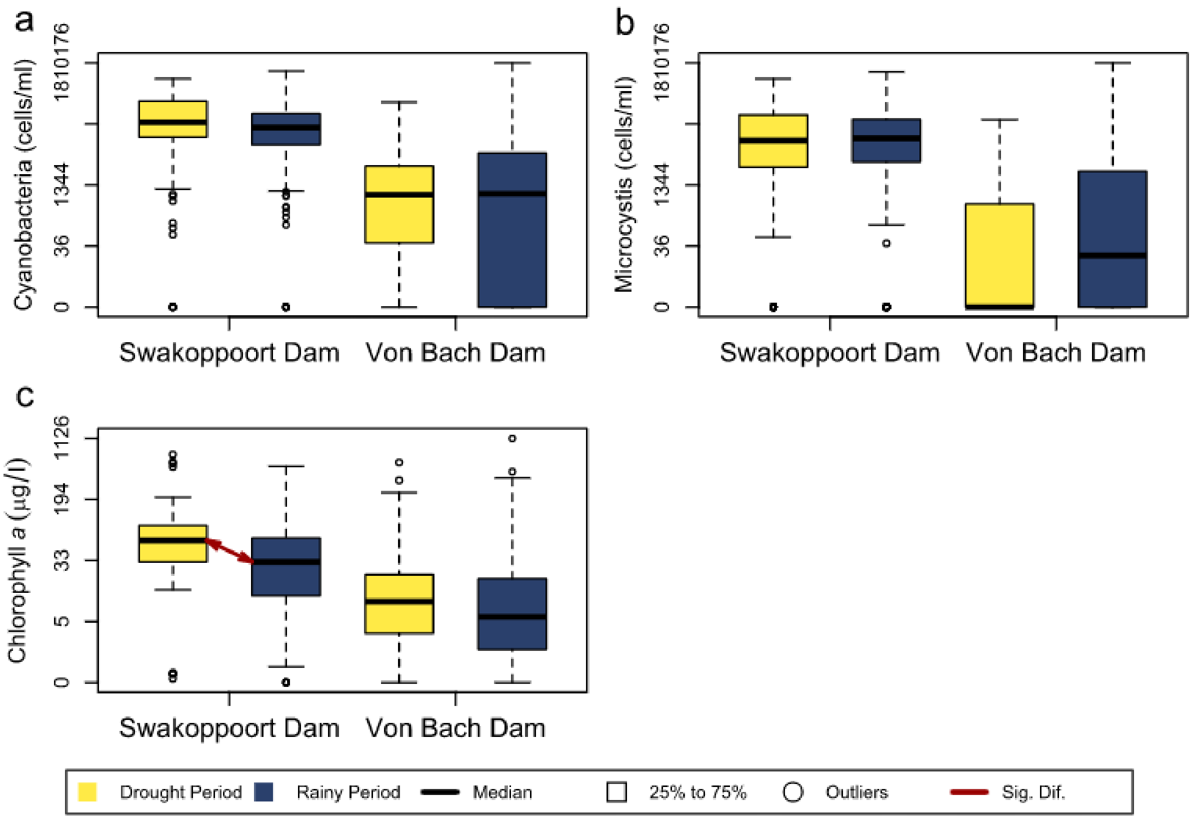

Figure 5.6. Box plot comparisons of the biological variables recorded during the study period

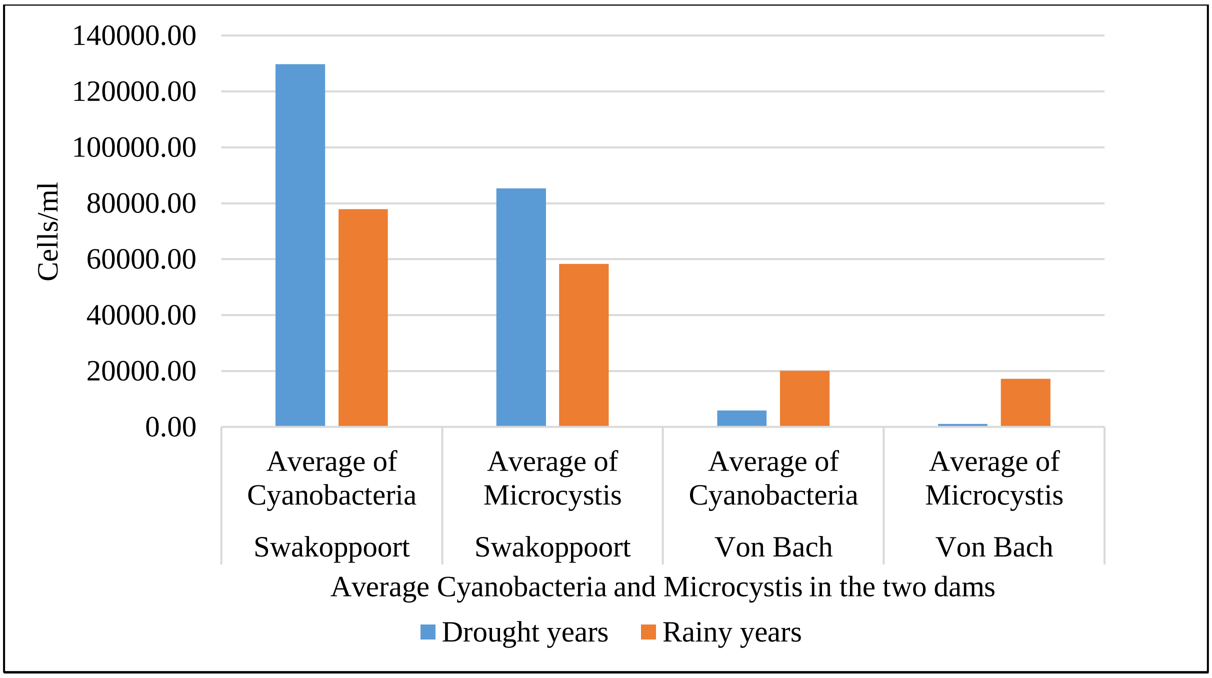

Figure 5.7. Average Cyanobacteria and Microcystis in the two dams during drought and rainy years

5.3.2. Regression analysis

The linear regression results show that the variables found to best relate to the variation in Chl *a* in SWP were lnNTU (1.436; 0.000), Depth (0.714; 0.000), Summer (0.573; 0.000), pH (1.200; 0.011), Vol % (0.823; 0.012), Cl (0.782; 0.016), lnEC (1.197; 0.029), and lnOP (0.883; 0.040). The variables found to best explain the variation in Chl *a* in the VB are lnNTU (1.509; 0.000), Depth (0.650; 0.000), Rainy years (0.440; 0.000), Summer (0.491; 0.000), lnNH3 (0.721; 0.000), Vol% (1.363; 0.000), TKN (1.208; 0.001), lnNPRatio (1.164; 0.009), Cl (1.380; 0.021), and lnOP (0.883; 0.038).

The variables found to best explain the variation in *Microcystis* in cell numbers in the SWP were Spring (3.322; 0.001), Cl (10.440; 0.001), Rainy years (3.419; 0.002), Depth (1.399; 0.038), and Winter (2.046; 0.050). The variables found to best explain the variation in *Microcystis* cell numbers in then VB are Rainy years (36.750; 0.000), Vol % (0.253; 0.000), pH (2.637; 0.000), and Summer (0.336; 0.019).

The variables found to best explain the variation in Cyanobacteria in SWP were Depth (1.448; 0.001), Vol % (0.659; 0.005), lnNTU (1.392; 0.015), $lnNH_3$ (0.776; 0.039), and pH (1.355; 0.049). The variables found to best explain the variation in Cyanobacteria in theVB were Spring (0.055; 0.000), Rainy years (6.181; 0.000), Depth (0.439; 0.000), pH (2.322; 0.000), Winter (0.277; 0.003), $lnNH_3$ (1.929; 0.007), Summer (0.308; 0.014), and Vol % (0.511; 0.016).

5.4. Discussion

The application of water augmentation was evident by the higher average water volume (% of full supply capacity) in the VB Dam compared to SWP Dam. Statistically significant differences ($p < 0.05$) were noted for most physicochemical and phytoplankton parameters in the two dams, interestingly, decreases in average phytoplankton biomass, total cyanobacteria, and *Microcystis* in the SWP but not in the VB in the drought years. Other studies showed similar trends, for instance in the warm semi-arid North-eastern Brazil where seasonal variations in water volume are followed by nutrient and Chl *a* concentration increases during drought periods (Brasil et al., 2016; Rocha Junior et al., 2018), which is similar to what Nosrati (2011) reported in Behshahr, Neka and Sari in the Mazandaran Province in northern Iran.

Globally, drought periods have led to an increase in phytoplankton biomass and cyanobacteria cells in reservoirs (Brasil et al., 2016; Rocha Junior et al., 2018; Leite and Becker, 2019). In this study, SWP Dam recorded a higher concentration of mean suspended Chl *a* and cyanobacteria cells in the prolonged drought years while VB Dam in the rainy years. A study by Brasil et al. (2016) revealed that the phytoplankton biomass and cyanobacterial cell numbers in 40 man-made lakes in the state of Rio Grande do Norte in Northeastern Brazil were higher in the drought season compared to the rainy season. Furthermore, Huang et al. (2020) reported that the suspended Chl *a* was decreasing with an increase in water level in the shallow Yangtze-connected lakes (Lake Dongting, Poyang, Chao, and Tai). Leite and Becker (2019) reported low algal biomass post-reflooding. This was similar to the finding of the present study in the SWP Dam, but was not comparable with the findings of the VB Dam.

Lehman et al. (2017) in their study on the impact of drought on *Microcystis* blooms in the San Francisco Estuary reported that severe drought conditions lead to an increase in the abundance of *Microcystis* and other cyanobacteria species. In a follow-up study in the same study area, they reported that conditions associated with rainy periods was found not to eliminate *Microcystis* cell numbers, although the blooms were smaller with a low abundance of *Microcystis* cell numbers and a short duration (Lehman et al., 2019). *Microcystis* is clearly adapted to varying climatic conditions and water quality (Lehman et al., 2019). It was further concluded that the increase in magnitude, duration, and toxicity of *Microcystis* cell numbers in San Francisco Estuary was associated with elevated temperatures and low streamflow (Lehman et al., 2019).

Various reports showed that prolonged drought events that lead to decreased reservoir water volume can magnify the concentration limnological variables which favour the growth of toxic phytoplankton (Bouvy et al., 2000; Mosley,2015; Costa et al., 2018; Lehman et al., 2019; Peña-Guerrero et al., 2020). One of the critical limnological variables is nutrients (TP&TN), which is a major concern in excessive amount as it causes eutrophication leading to water quality deterioration. TP is reported to be the main nutrient influencing toxic phytoplankton growth. Based on TP concentration in the study, SWP Dam was classified as hypertrophic (>0.4mg/l), while VB Dam as eutrophic (>0,2mg/l) in both drought and rainy years. This can be explained by the difference in vol% between the two dams, however, VB Dam showed little variability in water volume, which could be due to water augmentation. The result supports the finding by (Rocha Junior et al., 2018), where most of the 16 reservoirs found in the Piranhas-Açu River watershed in the semi-arid region of Northeastern Brazil were considered eutrophic during the drought period.

Findings from the current study differ from those by Rocha Junior et al. (2018) of an increase in the concentration of nutrients during drought years. Furthermore, the findings do not support the report by Mosley (2015) that nutrients often increase in lakes due to reduced flushing and resuspension in some shallow lakes during periods of drought. It was observed in the current study that the TP, and OP in the two dams increased in the rainy years compared to the drought years. This could be due to higher rainfall-runoff transporting nutrients from dam catchment areas with unsustainable agricultural practices (e.g., over-grazing) or internal nutrients load resuspension from the sediments as described by Lee et al. (2019). Notably, the nutrient (TP) concentration was >0.1mg/l, which can cause toxic phytoplankton blooms (WHO, 2021).

Leite and Becker (2019) reported that TP and OP concentrations that showed little change after recurring flood events in the Dourado reservoir located in Currais Novos city, in the northeast of Brazil. It was mentioned that TP could be in an organic matter already present in the recurring floods, and that OP was not consumed by phytoplankton as there was not sufficient time for the establishment of algal biomass (Leite and Becker, 2019). The reduction in the concentration of nutrients during the drought years in the current study could be due to limited internal nutrient loading from the sediment, prevented by a stratified water column, due to the deep depths of the two dams.

In addition to nutrients, other limnological variables such as the WT, pH, Zeu/Zmix, EC, and Cl concentrations can influence phytoplankton growth and concurrent increases in Chl *a* concentration. In the current study, these variables increased in concentrations in the drought years relative to the rainy years in both dams. Peña-Guerrero et al. (2020) reported that pH, EC and Cl concentrations increased during the drought periods in the Maipo River Basin, Central Chile. Similarly, 40 man-made lakes found in the state of Rio Grande do Norte in Northeastern Brazil were also reported to have an increase in WT, pH, WT, and EC during the drought seasons compared to rainy seasons (Brasil et al., 2016).

Although the water temperature was not related to variations in Chl *a* and cyanobacterial cell numbers, it is reported that rising temperatures favour the growth of toxic cyanobacterial blooms in temperate and sub-tropical reservoirs (Nalewajko and Murphy, 2001; Davis et al., 2009; Yamamoto, 2009; Lürling et al., 2013; Mantzouki et al., 2018). In the two dams the average WT was >15℃ in both drought and rainy years which is reported as sufficient to support cyanobacteria growth in eutrophic water bodies (Nalewajko and Murphy, 2001; Liu et al., 2011; Lürling et al., 2013; Thomas and Litchman, 2016). Changes in WT due to air temperature increases and longer hydraulic time influence the patterns of stratification, which greatly affect the seasonal growth of phytoplankton (Mosley, 2015; Yang et al., 2019). In lake Taihu, China, *Microcystis* were detected and dominated during a period of warm WT ranging from 18.2℃ to 32.5℃ (Chen et al., 2003). Lake Biwa, Japan, WT in the range of 28℃ to 32℃ was found suitable for *Microcystis* and *Dolichospermum* growth (Nalewajko and Murphy 2001).

The water turbidity was significantly different in the SWP Dam between drought and rainy years. This could be explained by the increase in phytoplankton biomass. Leite and Becker (2019) reported an improvement in water turbidity in the Dourado reservoir located in Currais Novos city in north-east Brazil after recurring flood, with very low algal biomass. The VB Dam turbidity supports the findings of the study by Leite and Becker (2019). Brasil et al. (2016) reported lower water transparency in the 40 man-made lakes found in the state of Rio Grande do Norte in Northeastern Brazil. Mosley, (2015) indicated that turbidity increases in drought years due to resuspension in shallow lakes. Light availability did not correlate with the variation in Chl *a* and cyanobacterial cells, with no significantly difference between the drought and rainy years in both dams. The SWP Dam's salinity (EC) was higher than that of the VB Dam, this could be due to the high evaporation rate, given the dam's larger surface area and different land use activities in the two catchment areas. However, the salinity of the SWP Dam (114 mg/l) was low compared to other global lakes like Lake Eyre (40 000 mg/l) and Lake Corangamite (105 000 mg/l) in Australia (Langbein,1961), which is in agreement with the view that reduction on dam volumes during drought leads to increased salinity (Mosley, 2015).

Our hypothesis that decreased dam levels during periods of drought favour the growth of phytoplankton biomass was only confirmed with findings from the SWP. The different result in VB may be explained by inhibition of phytoplankton by free ionic copper (>0.0003mg/l of TWQR) emanating from Otjihase Mine in the upper catchment area, which has been in operation for over 40 years. Thus, despite the favourable water temperature, the availability of sufficient nutrients and light in both dams, the free ionic copper which was above the South African Target Water Quality Ranges of 0.0003 mg/l in the VB could contribute to the inhibition of growth during the rainy season. Cyanobacteria have relatively high metal requirement in the form of metal cofactors in oxygenic photosynthetic electron transfer (Facey et al., 2019; Yadav et al., 2021). Copper is essential to cyanobacteria as a micronutrient as it is a component of cytochrome oxidase and plastocymin in the electron transport chain. However, at higher concentrations, copper inhibit growth of cyanobacteria, causing a heperoxidative state, and chlorosis. Mohy El.Din, (2017) reported that lower doses of copper (Cu^{2+}) had a stimulatory effect on biomass of *Spirulina platensis*, whereas the higher dose were found to be inhibitory to growth. Lehman et al. (2004) reported that an addition of as little as 1 µg/l copper suppressed algal biomass measured as particulate Chl *a* in Saginaw Bay, Lake Huron, North America. It thus seems reasonable to expect that the copper levels found in VB are due to copper inhibition.

5.5. Conclusion and recommendations

In conclusion although drought years are being reported in some studies to have a profound water quality effect in dams, catchment-specific characteristics also play a role. In the current study, the volume decrease during drought was only observed to have a greater effect on phytoplankton biomass in SWP Dam than in VB Dam, and although the physiochemical parameters were favourable for phytoplankton proliferation during drought years, the pattern and magnitude of varied between the drought and rainy years. Therefore, to enhance water security, deeper elucidation of environmental drivers of water quality dynamics, such as land use must be incorporated further in the context of recurring droughts.

In terms of the study area, the following should be considered: (a) increased water quality monitoring of the three dams should be carried out during the period of water transfer to VB Dam, (b) the impact of water abstraction during periods of drought on phytoplankton proliferation, especially against the backdrop of climate change, and (c) the correlation between depth of extraction, drought phytoplankton proliferation.

CHAPTER 6: COMPARISON OF PHYTOPLANKTON CONTROL MEASURES IN REDUCING CYANOBACTERIA ASSEMBLAGE OF RESERVOIRS FOUND IN THE ARID REGION OF SOUTHERN AFRICA

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Declaration by the candidate

With regard to Chapter 6, the nature and scope of my contribution were as follows:

Nature of contribution	Extent of contribution
Conceptual design, fieldwork, experimental work, manuscript writing, data curation, formal analysis, methodology	80%

The following co-authors have contributed to Chapter 6:

Name	Email address and institutional affiliation	Nature of contribution	Extent of contribution
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Abstract

Ecological restorations of reservoirs are implemented worldwide, however, minimal successes are reported and understood for warmer African lakes like SWP Dam, Namibia. The objectives of the study were (a), to establish the effectiveness of the two control measures in reducing cyanobacteria growths in comparison to untreated control areas, (b) to compare the results generated before and after control measures with the reference VB Dam. During Phoslock® treatment, the average cyanobacteria cells and total phosphate (TP) were 90 521cells/ml and 0.3mg/l in the treated area and 55 338cells/ml and 0.1mg/l in the control area. During Solar Powered Circulation (SPC) treatment, the average cyanobacteria cells were on average 906 420 cells/ml in the treated areas and 121 891cells/ml in the control area. The TP on average was 0.3mg/l during SPC treatment. While during the combined treatment the average cyanobacteria cells, TP, and total nitrogen (TN) were 18 387 226cells/ml, 0.27mg/l and 2.41mg/l before and 22 836 511cells/ml, 0.42mg/l and 1.50mg/l after treatment. This was higher compared to the reference site. PCA triplot indicates no grouping pattern and the repeated measures mixed model analyses indicate that treatment had no significant effect on cyanobacteria cells. It was evident that, the two control measures were ineffective in reducing cyanobacterial cells.

6.1. Introduction

Freshwater ecosystems such as man-made dams are constructed to provide a wide range of ecosystem services to human beings. However, since most of man-made dams are constructed adjacent to populated cities or towns, they can become eutrophic due to poor environmental protection structures upstream (Bozelli, 2019). The eutrophication state, is causing the formation of toxic phytoplankton blooms, which are limiting the ecosystem services from man-made dams. Although ecological restoration remains a great challenge with more failure than success, some measures such as phytoplankton control is an alternative to face the degradation of man-made dams (Beklİoğlu et al., 2008).

Over the last four decades, different phytoplankton control measures have been implemented globally mainly in the developed world, to improve water quality by effective manipulation of nuisance phytoplankton blooms (Bozelli, 2019; Jeppesen, Søndergaard, & Liu, 2017). However, very little is known and reported about phytoplankton control measures in man-made dams situated in the warmer arid environment of Southern Africa. It is widely known that it is very difficult to control phytoplankton blooms in warmer arid environments as warmer drought conditions in itself create symptoms similar to eutrophication (Jeppesen & Meerhoff, 2011; Jeppesen et al., 2017; Søndergaard et al., 2007; Yu et al., 2016). Bozelli, (2019) states that, unlike in the developed world, restoration is limited in the developing world due to costs and knowledge. Although restoration measures are implemented in the developed world, they are not simple and directly transferable due to differences in climatic conditions despite similar problems (Bozelli, 2019), thus the need for modifications of existing and alternative methods for warmer reservoirs (Jeppesen et al., 2017).

The main purpose of phytoplankton control is to ensure the prevention of the proliferation of nuisance bloom forming phytoplankton species (Burford et al., 2019; Lürling & Mucci, 2020; Lürling et al., 2016; Pęczuła, 2012; Visser, Ibelings, Bormans, & Huisman, 2016). According to Pęczuła, (2012) phytoplankton control measures includes the following: (1) physical (aeration, hydrologic manipulations, circulation, reservoir drawdown, ultrasound, etc.) (2) biological (biomanipulation, bacteria, natural predators, etc.) and (3) chemical controls (algaecides, coagulation, nutrient-binding clay, etc.). (Pęczuła, 2012) stated that the success of any phytoplankton control measure is dependent on external nutrient control. The latter was verified in a study on Lake Washington, Washington, USA where external nutrient reduction resulted in the reduction of toxic cyanobacterial blooms ((Edmondson & Lehman, 1981). However, due to internal nutrient loading, the reduction of external nutrients alone did not lead to long term water quality improvement, as observed in Alderfen Broad in Great Britain (Pęczuła, 2012). External nutrients reduction has proven to be the most sustained effective approach for control of cyanobacterial blooms, even before considering any control methods or products (Burford et al., 2019; Lürling et al., 2016).

According to Søndergaard et al. (2007) the results from the various cyanobacterial control studies were not similar. For examples capping of phosphorus through the application of gypsum in Paldang Reservoir (Korea), Phoslock® in Swan-Canning River and Vasse estuaries (Australia) (Robb, Greenop, Goss, Douglas, & Adeney, 2003), aluminum sulphate in Green Lake, Seattle (USA), and iron (III) chloride in Lake Groot Vogelenzang (Netherlands) resulted in only a short term water quality improvements mainly in water transparency and Chl *a* (Pęczuła, 2012). Phoslock® is a modified clay designed to bind phosphorus in the water as it descends and traps it in the bottom sediment by forming a cap that prevents its release from the sediment into the water column (Burford et al., 2019; Robb et al., 2003).

To suppress cyanobacterial blooms in the epilimnion Solar Powered Circulation (SPC) technology has been used in some reservoirs (Kirke, 2000; Lürling & Mucci, 2020). Installation in Crystal Lake, Des Moines, Illinois, (2 SPCs), East Gravel Lake, Thornton, Colorado (3 SPCs), and Lake Palmdale, California (6 SPCs), USA, were found to suppress cyanobacteria within the treatment zones, which resulted in the increase in density of green algae and diatoms following SPC initiation (Hudnell et al., 2010). These are however shallow reservoirs of <10m, with surface area of >1km^2 (Hudnell et al., 2010). SPCs suppress cyanobacteria by creating mixing that inhibits the growth of the inedible cyanobacterial cells and thus improves the water quality (Hudnell et al., 2010). While inhibiting cyanobacteria growth, it creates an environment that allows edible algae to grow and get consumed by zooplankton (Visser et al., 2016a). The method is based on the principles of bio-manipulation and it only treats symptoms of eutrophication but not the cause of eutrophication (Hudnell et al., 2010). The habitat disturbance of the cyanobacterial colonies are within the epilimnion zone of the dam, where the SPCs intake hose is set above the thermocline zone of the water column. One unit of SPC is capable of circulating 40 000 liters of water per minute.

The current study compares the effectiveness of two phytoplankton control measures employed in the SWP Dam, in Namibia. The objectives of the study were (a), to establish the effectiveness of the two control measures in reducing cyanobacteria growths in comparison to untreated control areas in the SWP Dam, (b) to compare the results generated before and after phytoplankton control measures with the nearby reference VB Dam.

It should be noted that the SWP and VB dams are important water sources to central Namibia. They are found in the subtropical desert climate characterised with large differences in day and night-time temperature, low rainfall, low humidity, and high evapotranspiration. The two dams were constructed on an ephemeral river, which only flows during the rainy season. The SWP and VB dams are warm, monomictic man-made dams that stratify throughout the year, with only overturn or mixing during the winter season (Sirunda & Mazvimavi, 2014). The SWP Dam is reported with frequent cyanobacteria blooms (Garus-oas, 2017; Lehmann, 2010; Sirunda & Mazvimavi, 2014). Total phosphorus concentration in the SWP Dam is reported to be higher in magnitude in comparison to that of the VB Dam (Sirunda & Mazvimavi, 2014). The poor water quality of the SWP Dam is reported to be linked to the changes in land use activities in the catchment area (Cashman et al., 2014; Garus-oas, 2017; Sirunda & Mazvimavi, 2014). To the authors' knowledge, this is the first study on phytoplankton control measures on a selected dam in the desert region in Southern Africa.

6.2. Material and methods

6.2.1. Study area

Characteristics of the study area

The VB(21° 59'59.27'' S 16° 58'54.76" E) and SWP dams (22° 12'44.31" S 16° 31'44.97" E) are situated in central Namibia with a desert climate (Figure 6.1). The latter dams are used to supply water to the city of Windhoek, Okahandja Town, Karibib Town, Otjimbingwe Village, and Navahacb Mine (Scott et al., 2018; Slabbert & Grobbelaar, 2007). The application of the phytoplankton control measures was done in the SWP Dam, which is designed to hold water for two rainy seasons with no flow into the environment, except during periods of water overflow due excessive rainfall runoff (Figure 6.1). Although control sites were set out in the SWP Dam where treatment was applied, the nearby untreated VB Dam was also used as a reference Dam (Figure 6.1). The VB Dam was selected as a reference site because it is on the same river as the SWP Dam (Figure 6.1; Table 6.1) and experience the same environmental conditions. Furthermore, the phytoplankton assemblage of the VB Dam is dominated by the same cyanobacterial species of *Microcystis* followed by *Anabaena* as in the case of the SWP Dam. Nevertheless, any environmental conditions e.g. climate that may have changed during the employment of the two phytoplankton control measure period in the SWP Dam will also have the same effect on the environmental conditions and cyanobacterial assemblage in the VB Dam.

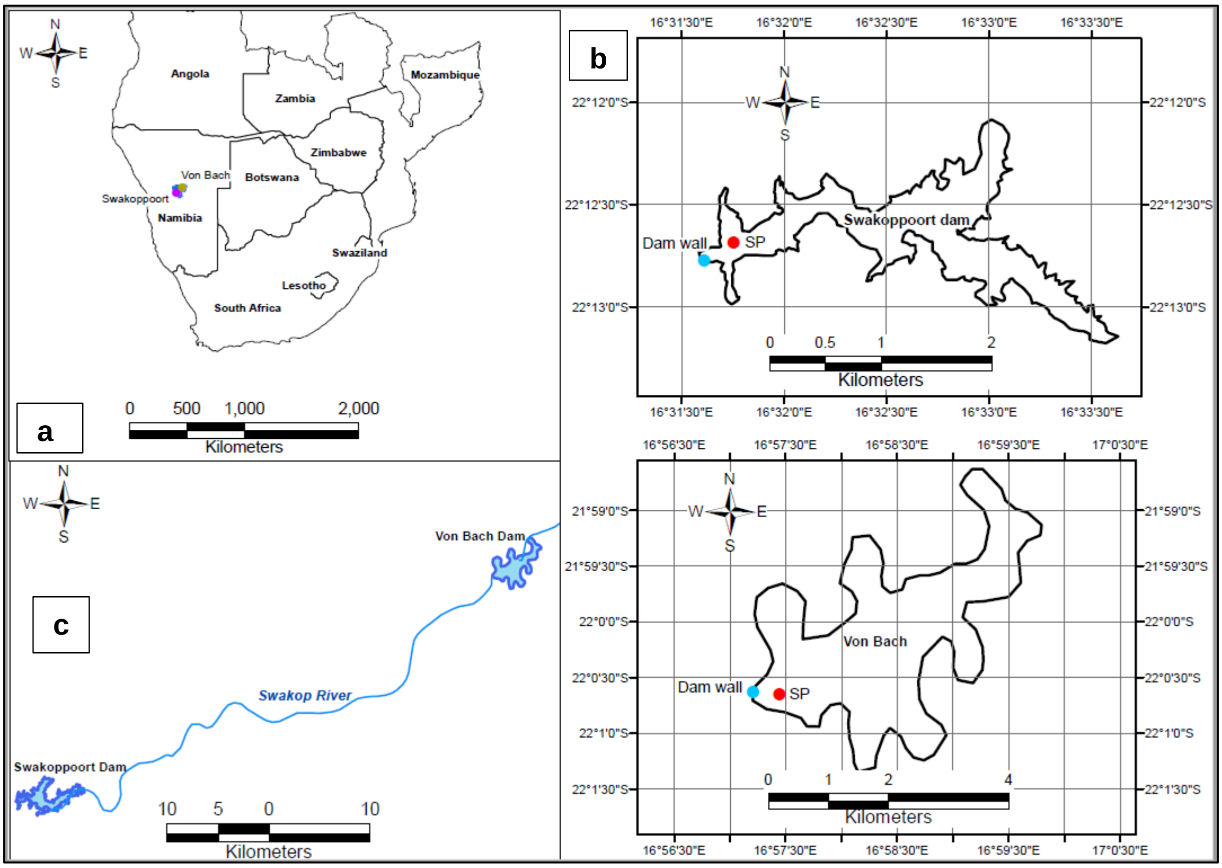

Figure 6.1. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset, indicating a) location of Namibia within Africa; b) the study site, treated SWP and the reference VB dams located in the central of Namibia, with a sampling point, and c) the location of the study areas on the Swakop River.

Table 6.1. The main morphometric features of the SWP, and VB dams situated on the Swakop River.

Features	VB Dam	SWP Dam
Capacity (Mm^3)	48.56	63.48
Max. Depth (m)	29	30
Evapo. Losses (mm/a)	2254	2275
Ann. Rainfall (mm/a)	370	350

Surface area (FSC) (km^2)	4.89	7.81
Catchment area size (km^2)	2 920	5 480
Geology of the areas	Schist and granite	Schists and granite
Year completed	1970	1977

FSC: full supply capacity

6.2.2. Application of phytoplankton control measures in the SWP Dam

To reduce the cyanobacterial blooms in the SWP Dam, two types of phytoplankton control measures namely Phoslock® and SPCs were implemented. The first control measure was with Phoslock® which was applied in a 0.245 hectare bay of the SWP Dam on the 14th of November 2012 and monitored until April 2013 (Figure 6.2). The bay was boomed off with floating curtains which spanned the length of the entire water column and from the surface to the bottom to ensure that the Phoslock® slurry does not move out of the bay (Figure 6.2). The floating curtains length for the treated area was 24.5m and with a depth of 2.5m. To compare the effectiveness of the Phoslock® treatment, another bay (Phoslock® Control Area) of a size of 0.162 hectare was selected as a control site with no treatment and boomed off with floating curtains which spanned the length of the entire water column and from the surface to the bottom to ensure similar conditions as the latter treated area (Figure 6.2). The floating curtains length for the control area was 27.0m and with a depth of 2.5m The distance between the treated (SP1) and the control (SP2) areas was 1.2km (Figure 6.4).

In general Phoslock® dosages are calculated for each water body using the quantity of phosphorus in the water column, and the quantity of releasable phosphorous in the sediment (Epe, Finsterle, & Yasseri, 2017; Finsterle, 2014; Pallí, 2015). However, in an event that the quantity of phosphorus in the water column and sediment is not known, the manufacturer's recommends a Phoslock® dosage of 2 tonnes per hectare (Finsterle, 2014). For the selected treatment area size of 0.245hectare, around 0.612 tonnes of Phoslock® was applied. The applied dosage of Phoslock® (0.6125tonnes) was increased by 25% over the theoretical requirement of (0.4900tonnes) to compensate for changes in surface area and increases in phosphorus concentration due to water inflow, since the field experiment was done during the rainy season. To prepare a dosage of 2 tonnes per hectare, two 25kg of Phoslock® were mixed in a 400l water tank and the mixed slurry was sprayed onto the water surface using a water pump and a hosepipe (Figures 6.2). The dissolved phosphorus was on average 8.3mg/kg in the sediment and 0.9mg/kg in the water column of the treated area before application of Phoslock®. The water column pH was on average 8.9, which was within the effective range of Phoslock® pH 4 to 11.

Figure 6.2. Pictures of the application of Phoslock as a phytoplankton control measure in the SWP Dam from 14 November 2012 to 16 November 2012 for a period of 3 days.

Biochemical and phytoplankton cell number analyses indicated that the Phoslock in comparison to the control site in the SWP Dam was not effective. Therefore it was decided to install SPCs in April 2013 to cover 0.78km^2 (10%) surface area of the SWP Dam. The area was then monitored until June 2015 (Figure 6.3). The covered area was toward the water abstraction area near the dam wall, where water is abstracted for treatment for human consumption and also for limnological sampling (Figure 6.5). The SPCs were deployed at densities of approximately 0.16km^2/unit (Figures 6.5). The SPCs were installed 400m apart, as their sphere of influencing the cyanobacteria growths is within the 200m from the SPC machine as per the manufacturer specifications (Figure 6.5). The cyanobacterial cells in the treated area were compared with that of the control site (Ctrl) which was established further away from the SPCs in the SWP Dam (Figure 6.5).

Figure 6.3. Pictures of Solar Powered Circulation installation as a phytoplankton.

6.2.3. Selection of sampling points and data collection

Phoslock® treatment

During the Phoslock® treatment, one sampling point was established inside the boomed areas of the treated and untreated control area (Figure 6.4). At both the treated and control area, seventeen (17) water samples for biochemical and phytoplankton cell collection were taken at a depth of 30cm below the water surface in the photic zone with a total of thirty-four (34) samples from November 2012 to April 2013. Water samples for biochemical and phytoplankton cell numbers were collected on an hourly basis for two days during the application, weekly after application, and monthly for a period of six (6) months at the treated and untreated control sites in the SWP Dam.

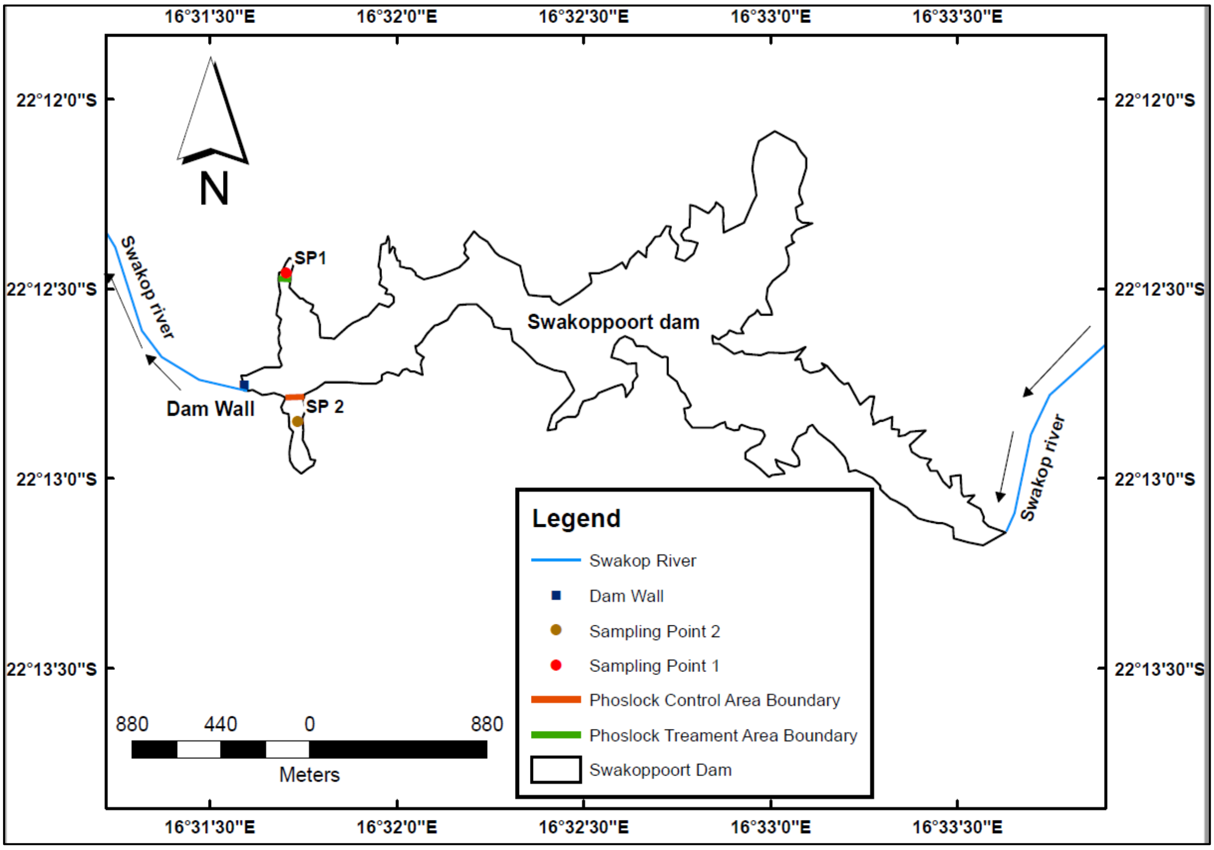

Figure 6.4. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset, indicating Phoslock treatment, with treated area SP1 and control area SP2.

Solar Powered Circulation treatment

During the SPCs treatment as part of the experimental design, firstly, six (6) sampling points were established at the SPC machines, and eight (8) water samples were collected at each SPC at a depth of 30cm with a total of forty-eight (48) samples from May 2013 to July 2013, without the untreated control sampling point (Ctrl) (Figure 6.5). This was done to establish the exact effect of the SPCs machine on cyanobacteria not further away from the machine. Secondly, after three months, in August 2013, seven (7) sampling points were established at 200m away from the SPCs machine where the water samples were collected at a depth of 30cm (Figure 6.5). One of the seven sampling points (Ctrl) was an untreated control sampling point located at 800m away from SPC 5 (Figure 6.5). Around forty-five (45) water samples were collected from these sampling points with a total of 315 samples from August 2013 to June 2015. This was done to establish the effect of the SPCs on cyanobacteria cells at a distance of 200m from the machine. Lastly, as part of the experimental design, ten (10) water samples were also collected at an interval of 50m, 100m, 150m, and 200m from SPC 2 at a depth of 30cm with a total of forty (40) water samples from July 2014 to February 2015 to establish changes in phytoplankton at spatial scale caused by the machine. During the SPC treatment, the intake hose of the installed SPCs were set and adjusted to be above the thermocline at all times as per the manufacturer requirement when targeting the treatment/control of cyanobacteria growths. Water samples for phytoplankton cell numbers were collected twice a month after installation for a period of 24 months, at all the selected sampling points and an untreated control site in the SWP Dam. During this period, biochemical water samples (chemistry, turbidity, suspended chl-*a*) were collected at one sampling point nearby the dam wall.

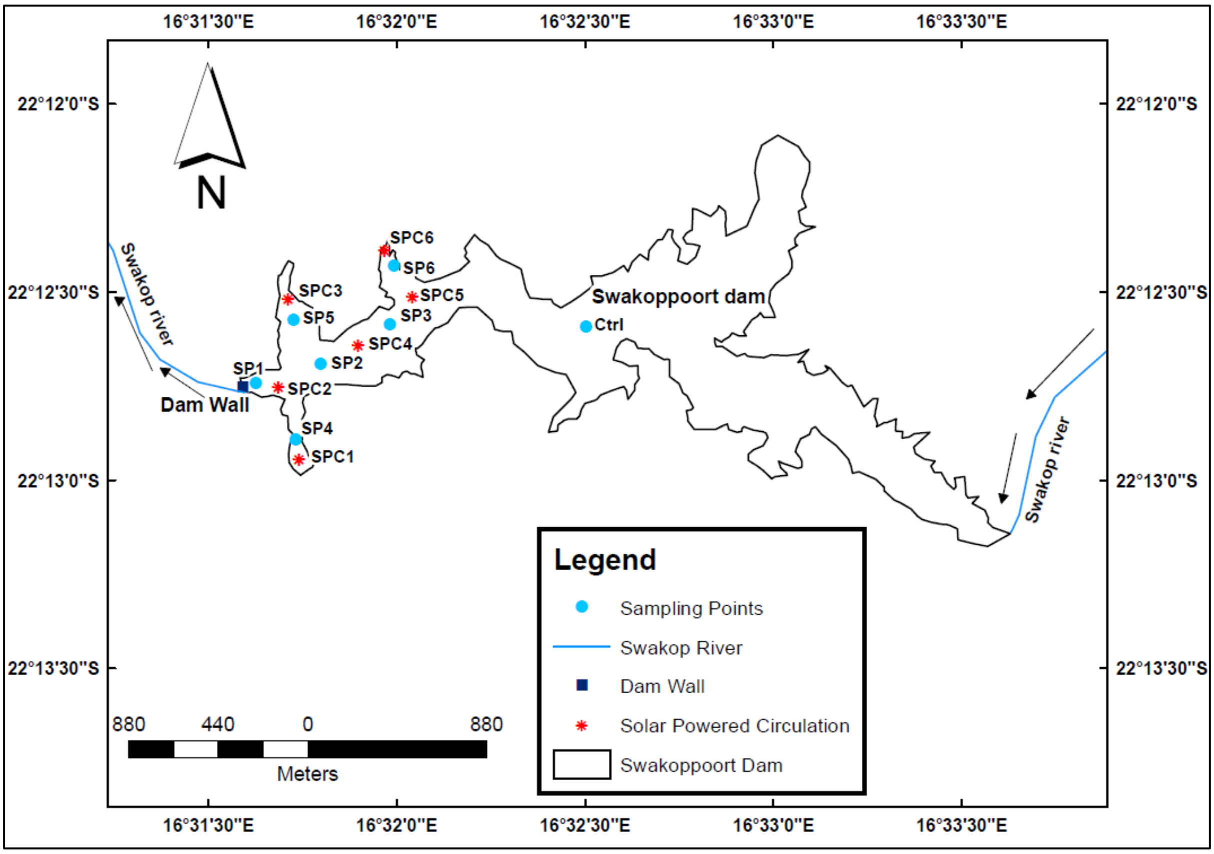

Figure 6.5. Maps created with QGIS v 3.14 pi (Open Source Geospatial Foundation Project) using the Namibia Water Corporation dataset, indicating Solar Powered Circulation treatment, with the locations of the SPC machine in the dam, sampling points (SP) in the treated area (SP1-6) and control sampling point (Ctrl).

6.2.4. The combined effect of phytoplankton control measures

For the combined effect of the phytoplankton control measures water samples for biochemical and phytoplankton cells were collected on monthly basis. Water samples were collected at the different depths ranging from the surface to the bottom at the sampling point located at the dam wall, in both SWP Dam and the reference VB Dam (Figure 6.1). The collected data at the different depths on a monthly basis in both dams from 2003 to 2019 were used to determine the combined effect of the phytoplankton control measures in reducing phytoplankton growths in the SWP Dam, in comparison to untreated reference VB Dam.

For the current study, the set of data collected from 2003 to 2019 for the two phytoplankton control measures used, and the frequency of sampling method was sufficient as it provided more insight on the effectiveness of the combined control measures employed for phytoplankton growths.

To collect a representative sample at selected sampling points, a dip sampling method was employed. A dip sampling method involves the dipping of the *Van Dorn* 5L water sampler from a boat into the water to retrieve the water sample which was transferred to the appropriate sample container (Burns et al.,2000). The collected water samples were transferred into labelled acid wash plastic and glass containers, which were preserved in cooler boxes. The cooler boxes were transported to the laboratory where the algae and suspended chl-*a* were analysed within 24 hours.

6.2.5. Field and laboratory water samples analyses

In the laboratory, water samples were analysed in replicates for phytoplankton assemblage. To identify phytoplankton, samples were sedimented in a Sedgewich-Rafter counting chamber and analysed under an inverted microscope at 400 × magnification using the strip-count method (APHA 1992). All algae were identified according to (Truter, 1987), (Wehr, Sheath, & Kociolek, 2015), (Van Vuuren et al., 2006) and (Taylor, Harding, & Archibald, 2007). The total phosphate and nitrogen of the water samples were also measured in the laboratory. The ortho and total phosphate of the water samples was measured using the Ascorbic Acid method as described in APHA (1998: part 4500 P E). A spectrophotometer with infrared phototubes was used as colorimetric equipment. The total nitrogen of the water samples was measured using the Cadmium Reduction method as described in APHA (1998: part 4500 NO3-E). In the analysis of total nitrogen, the reduction column was used as an apparatus together with the spectrophotometer. The total phosphate and nitrogen of the water samples were reported in mg/l. The turbidity of the water samples was measured using the turbidity meter and the nephelometric method (APHA, 1998: part 2130 B) in the laboratory. Furthermore, the value for turbidity was reported in Nephelometric Turbidity Units (NTU). Suspended Chl *a* contained in the water samples was measured using the spectrophotometric determination method as described in APHA (1998: part 10200 H). A blank sample without the analyte was used for quality control of the analytical processes for each analysed phytoplankton. Calibration standards were checked to make sure that there were made up correctly. Laboratory generated data were recorded on a spreadsheet. The data were arranged in chronological order and sorted according to sampling points, treatment type, date, and depths.

6.2.6. Data analysis

The period during which SWP Dam was treated with Phoslock® was in November 2012, and the possible water quality changes were monitored until April 2013. Treatment with Solar Powered Circulation commenced in May 2013 and water quality improvement was monitored until June 2015. In this study, the effectiveness of each control measure in reducing the cyanobacteria cell numbers in the treated area of the SWP Dam was compared to the untreated control sites. The combined effect of the phytoplankton control measures was determined using the monthly data collected at the water abstraction tower near the dam wall before and after treatment periods from 2003-2019, while the nearby VB Dam was used as a reference untreated control site where water samples were also collected at the intake at different depths similar to that SWP Dam.

Descriptive statistics such as mean and standard deviation were estimated for each phytoplankton parameter measured in the two dams during Phoslock®, SPC and combined treatment before and after. Spearman correlation was used to assess the relationship between the dominant phytoplankton in the two dams and scatterplots were generated for the significant positive correlation variables using SPSS 26.0 statistical package software and the level of significance used was $p < 0.05$ for all tests. The limiting nutrients of the two dams were established using the molar mass Redfield ratio of Total Nitrogen: Total Phosphate (TN:TP) of 16:1, which implies that, TN:TP >16 is designated as phosphate limiting and TN:TP <16 is designated as nitrogen limiting (Guildford & Hecky, 2000; Redfield, 1958). The TN:TP ratio was used for classification of the trophic status of the two dams before and after phytoplankton control measures using the nutrients index criteria (Dodds, 2006; Dodds & Smith, 2016; Jones, Knowlton, & An, 2003).

A PCA triplot was constructed using annual average phytoplankton cell count data representing the 0–5m depth range in combination with water quality parameters for the same period before and after treatment in the SWP Dam. The phytoplankton data forms the focal plot of the PCA triplot, whereas, water quality parameters were treated as supplementary variables.

Phytoplankton data were centred and standardised prior to use in the PCA (Ter Braak and Smilauer 2002). Canoco v5 (Microcomputer Power, USA) was used for Multivariate statistical analyses.

Variation in phytoplankton abundance between periods before and after phytoplankton control measures in the SWP Dam was assessed using repeated measures mixed models (Variance Estimation and Precision Module, Statistica v13, Tibco Software, USA). The mixed models featured seasonal means of abundance estimates representing 2003 to 2019. “Season” and “Year” were applied as repeated measures and depth range of the intake tower in the SWP Dam and treatment as fixed effect. In addition, short term variation of phytoplankton abundance in response to SPC treatment was assessed using repeated measures mixed models with “sampling event” specified as repeated measure. Normality of the datasets applied in for mixed models was evaluated using normal probability plots. Data that were not normally distributed were rank transformed before analysis.

6.3. Results

6.3.1. Effectiveness of the phytoplankton control measures in reducing cyanobacteria growths in the SWP Dam

Phoslock® treatment

During Phoslock® treatment the cyanobacteria cells at the treated area at SP1 were higher compare to the cyanobacterial cells recorded at the untreated control area at SP2 (Figure 6.6). In the treated area at SP1, the average cyanobacteria cell numbers were 90 521cells/ml, and 55 338cells/ml at the control area at SP2 for the duration of the treatment period (Figure 6.6). At all the sampling points *Microcystis* was the dominating species followed by *Anabaena*, and *Merismopedia* during the treatment period. Suspended Chl *a* was variable but was reported to be higher at the untreated control area at SP2 with an average of 69.9ug/l compared to the treated area at SP1 with an average of 46.7ug/l (Table 6.2a &b). The turbidity of the water was higher at the treated area at SP1 (18.2NTU) compared to the untreated control area at SP2 (17.4NTU) during the Phoslock® application dates, thereafter, similar conditions were observed at both treated and the control areas (Table 6.2a &b). The average total phosphate concentration in the treated area at SP1 was 0.3mg/l in comparison to 0.1mg/l at the untreated control area (SP2) (Table 6.2a &b). The average Ortho-P at the treated area (SP1) was 0.1mg/l and the untreated control area (SP2) was 0.1mg/l (Table 6.2a &b). It was evident from the data that the suspended Chl *a* and turbidity of the treated area was affected by Phoslock® (Table 6.2a &b).

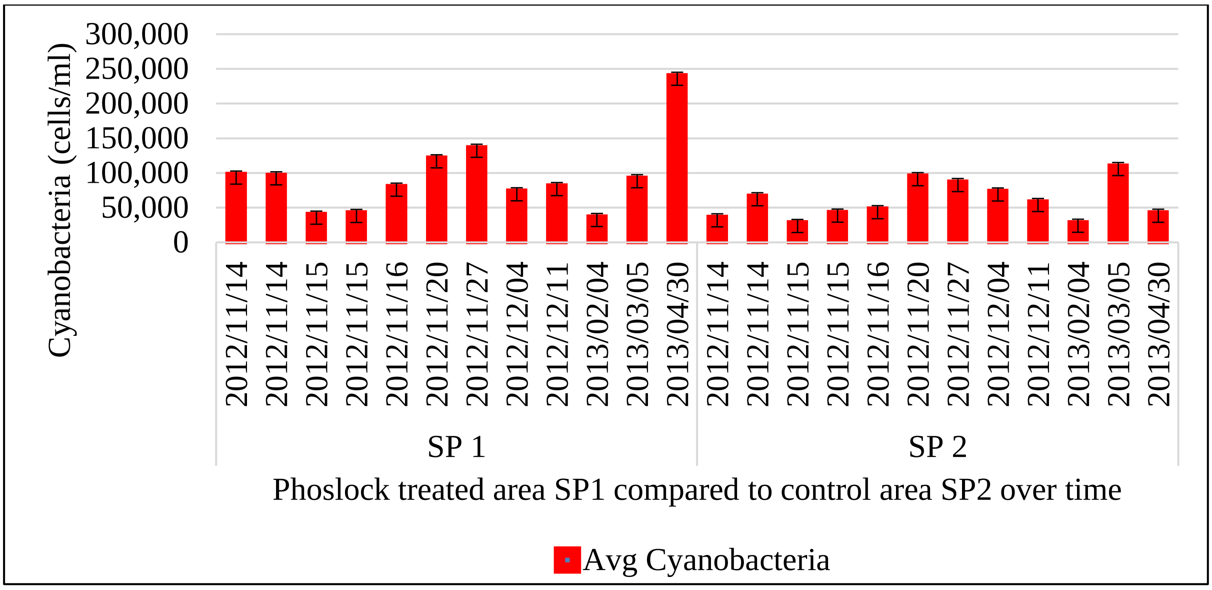

Figure 6.6. Cyanobacterial cell number during the Phoslock application, a comparison of the treatment area (SP1) and the control area (SP2). Error bars indicate standard error.

Table 6.2. a, Water quality parameters of the Phoslock® treated area (November 2012-April 2013); b, Water quality parameters of the Phoslock® untreated control area (November 2012-April 2013).

a

Treated area	14-Nov-12	14-Nov-12	15-Nov-12	15-Nov-12	16-Nov-12	20-Nov-12	27-Nov-12	04-Dec-12	11-Dec-12	04-Feb-13	05-Mar-13	30-Apr-13	Avg	Std
Turbidity (NTU)	15,9	36,3	12,8	28,7	19,5	22,8	22,7	17,1	14,9	8,44	8,69	10,2	18,2	8,4
Total Nitrogen (mg/l)	1,1	1,3	0,9	1	0,7	0,7	0,6	1,3	1,5	0,9	0,4	0,6	0,9	0,3
Total Phosphate (mg/l)	0,14	0,06	2,21	0,13	0,09	0,25	0,12	0,1	0,25	0,43	0,07	0,07	0,3	0,6
Ortho-Phosphate (mg/l)	0,08	0,04	0,1	0,06	0,07	0,04	0,03	0,07	0,06	0,02	0,03	0,02	0,1	0,0
Chlorophyll *a* (µg/l)	35,5	63,1	28,2	28,2	46,2	26,5	35,4	55,6	46,3	25,6	63,5	106,0	46,7	23,2
Cyanobacteria (cells/ml)	93 337	92 277	35 638	38 042	75 943	116 813	132 016	69 296	76 791	32 244	88 105	235 752	90521	55 215

b

Control area	14-Nov-12	14-Nov-12	15-Nov-12	15-Nov-12	16-Nov-12	20-Nov-12	27-Nov-12	04-Dec-12	11-Dec-12	04-Feb-13	05-Mar-13	30-Apr-13	Avg	Std
Turbidity (NTU)	16,5	16,1	16,4	16,4	17,6	24,6	28	19,1	17,5	11,1	14	11,5	17,4	4,8
Total Nitrogen (mg/l)	1,4	1	1,1	0,9	0,9	0,5	0,5	1,1	0,8	0,7	1,1	0,8	0,9	0,3
Total Phosphate (mg/l)	0,13	0,1	0,26	0,12	0,09	0,15	0,15	0,13	0,15	0,13	0,07	0,08	0,1	0,0
Ortho-Phosphate (mg/l)	0,06	0,04	0,08	0,06	0,05	0,06	0,03	0,08	0,07	0,03	0,02	0,03	0,1	0,0
Chlorophyll *a* (µg/l)	63,5	70,5	41,6	41,6	55,9	37,3	51,7	66,7	62,3	31,9	136,0	180,0	69,9	43,9
Cyanobacteria (cells/ml)	31 820	62 154	23 617	38 537	43 487	91 074	82 589	69 013	53 810	23 971	105 641	38 340	55338	27 037

Solar Powered Circulation treatment

During the SPC treatment, cyanobacteria abundance at sampling points in the proximity of the treated area (SP1, 2, 3, 4, 5, & 6) did not vary significantly from the untreated control area ($F_{6,300}$ = 1.32, P = 0.25) (Figure 6.7c). At all the sampling points, *Microcystis* was the dominating species followed by *Anabaena* during the treatment period (Figure 6.8). Furthermore, 50m interval collections away from SPC2 (i.e. 50m, 100m, 150m and 200m) revealed no significant spatial difference in cyanobacteria abundance due to the effect of the machine ($F_{3,9}$ = 1.04, P = 0.42) (Figure 6.7b). In addition, cyanobacteria cells collected at SPC machines did not vary among the different SPCs ($F_{5,35}$ = 1.91, P = 0.12) and also revealed no temporal significant change due to the treatment process ($F_{2,33}$ = 1.60, P = 0.22) (Figure 6.7a). During the SPC treatment period from May 2013 to June 2015, on average the total phosphate was 0.3mg/l, ortho-phosphate was 0.1mg/l, total nitrogen was 1.2mg/l, suspended chlorophyll-*a* was 92.6µg/l.

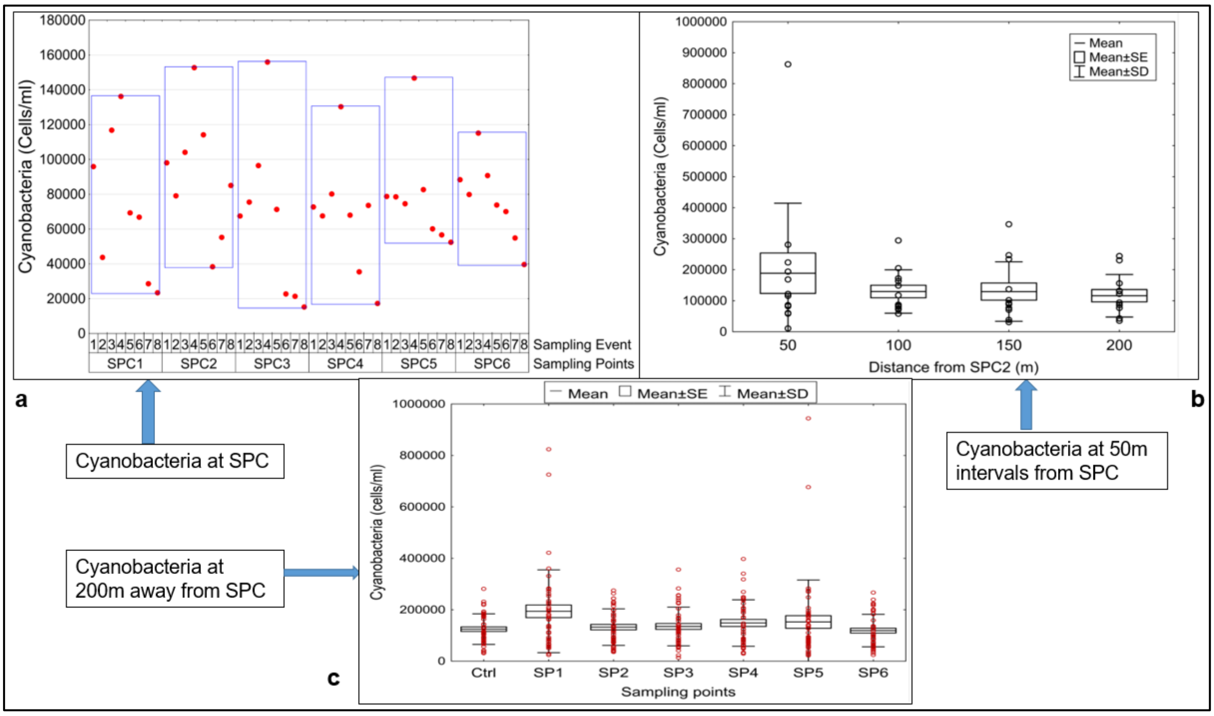

Figure 6.7. The figure depicts the result of the Solar Powered Circulation (SPC) treatment, indicating a) the average cyanobacteria cells monitored at the SPC machine from May 2013 to July 2013, b) the average cyanobacteria cell number monitored at 50m intervals from the Solar Powered Circulation from July 2014 to February 2015, and c) the average cyanobacteria cells monitored at a 200m away from the Solar Powered Circulation (in the treated area (Sampling Point 1-6) compared to the control area (Control Sampling Point Ctrl) from August 2013 to June 2015. Error bars indicate standard error.

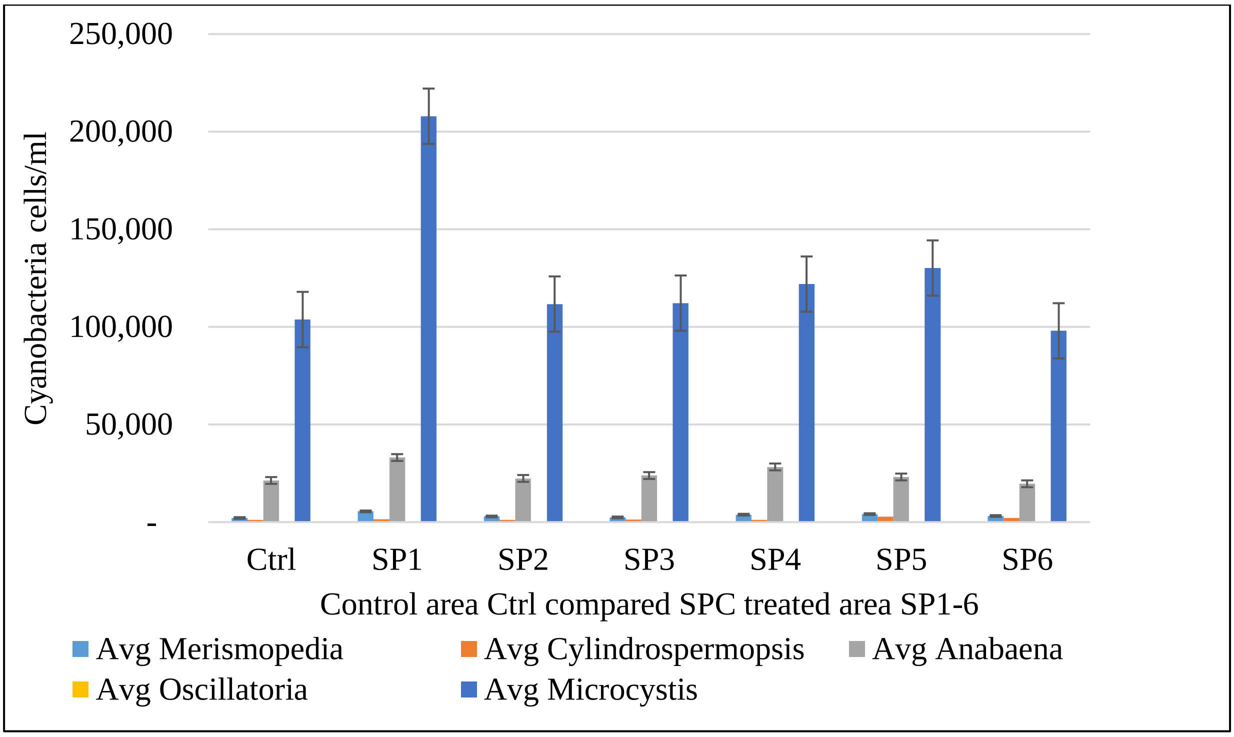

Figure 6.8. The dominant cyanobacterial species during the Solar Powered Circulation (SPC) treatment at 200m away from a SPC monitored in the treated area (Sampling Point 1-6) compared to the control area (Control Sampling Point Ctrl) from August 2013 to June 2015 Error bars indicate standard error.

6.3.2. The combined effect of the phytoplankton control measures

The combined effect of the control measures in the SWP Dam resulted in cyanobacteria which dominated the phytoplankton community by 85% of the total cell counts, before and after control measures. While in the untreated reference VB Dam, the cyanobacteria dominated the phytoplankton community by 79% of the total cell counts. The treated SWP Dam recorded more cyanobacterial cells compared to the untreated VB Dam during the study period (Figure 6.9c). In the SWP Dam, on average the cyanobacterial cell numbers before control measures were 18 387 226 cells/ml and after treatment, it increased to 22 836 511 cells/ml (Figure 6.9a). The average cyanobacterial count in the VB Dam (untreated reference) was reduced from an average of 14 020 cells/ml to 8 026 cells/ml after treatment. Among the cyanobacteria species, *Microcystis* and *Anabaena* dominated in the SWP Dam over the 17 years before and after control measures (Figure 6.9b), which was similar to the untreated control area. It was evident that there was a 58% increase in *Microcystis* and a 49% decrease in *Anabaena* after control measures in SWP Dam.

During the study period from 2003 to 2019, the average TP and TN was 0.27mg/l and 2.41mg/l before phytoplankton control measures and 0.42mg/l and 1.46mg/l after control measures in the SWP Dam. While in the untreated reference VB Dam, TP and TN were 0.19mg/l and 1.52mg/l during the study period. The average suspended Chl *a* was 41.8µg/l before control measures and 82.3µg/l after control measures in the SWP Dam. In the untreated reference VB Dam, on the other hand, the average Chl *a* was 22.3µg/l during the whole study period respectively. Before treatment, using the mean TN: TP ratio, the SWP Dam was eutrophic with a ratio of 20.0, however after treatment, the dam become hypertrophic with a ratio of 12.0. The reference VB Dam on the other hand was mesotrophic with a ratio of 32.0 before control measures and eutrophic with a ratio of 16.0 after control measures.

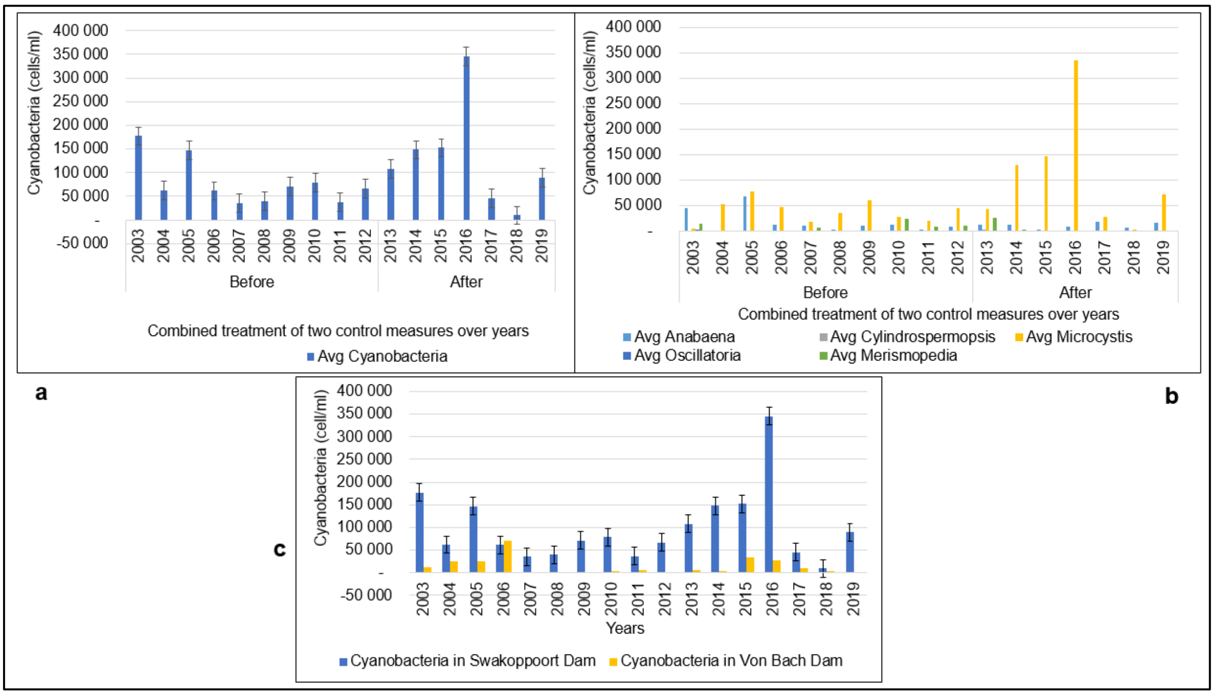

Figure 6.9. The figure depicts the results of the combined effect of the two phytoplankton control measures applied in the SWP Dam, before and after treatment, indicating, a) the effect of the two control measures on the cyanobacteria before and after treatment 2003-2019, b) the effect of the control measures on the dominance of the cyanobacteria species 2003 to 2019, and c) comparison of cyanobacterial cells in the treated SWP Dam to the reference VB Dam. Error bars indicate standard error.

Before control measures in the SWP Dam a significant positive correlation was observed between cyanobacteria and *Microcystis* (r=*0.57*; R^2=0.411*)*, between cyanobacteria and *Anabaena* (r=*0.47*; R^2=0.496), and cyanobacteria and total phytoplankton (r=*0.97* ; R^2=0.993). After control measures, a significant positive correlation *0.926* (r = *0.926*; R^2=0.936) existed between cyanobacteria and *Microcystis*, (r= *0.989*; R^2=0.999) as was the case between total algae cells and cyanobacteria in SWP Dam. In the VB Dam, a significant positive correlation of (r= *0.718*; R^2=0.979) was observed between cyanobacteria and *Microcystis*, (r= *0.705*; R^2=0.014) and between cyanobacteria and *Anabaena*, as well as (r= *0.878;* R^2=0.997*)* a between cyanobacteria and total phytoplankton.

A Principal Component Analysis triplot indicates no grouping pattern in ordinal space of the years before and after phytoplankton control in the SWP Dam (Figure 6.10). Suspended Chl *a*, turbidity, pH, and TP were positively correlated with total phytoplankton cells, total cyanobacteria, *Microcystis,* and *Anabaena* (Figure 6.10). Increased abundance of the cyanobacteria *Cylindrospermosis* and *Merismopedia* occurred in 2013 relative to the other years studied as indicated by the close association in ordinal space (Figure 6.10).

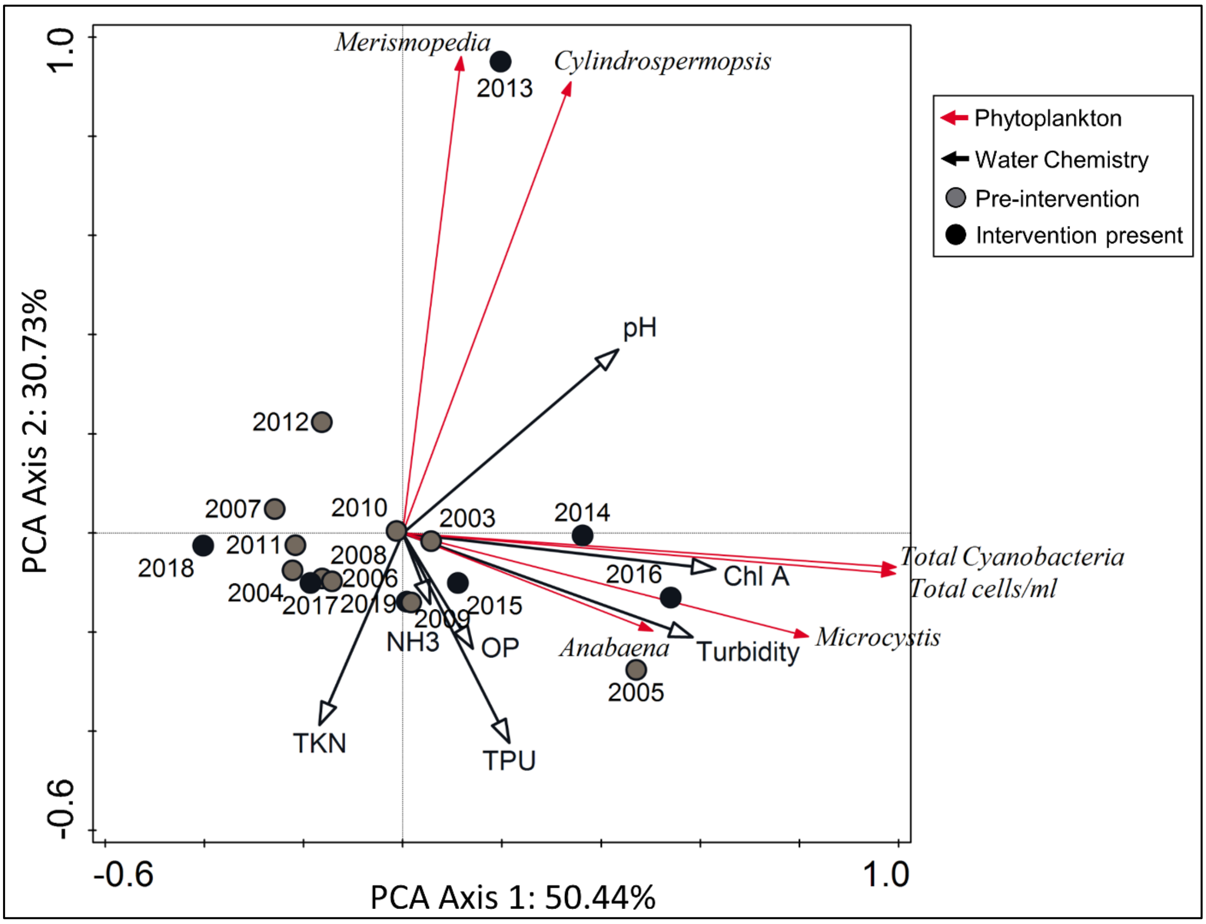

Figure 6.10. Principal component analysis (PCA) triplot indicating the associations between selected cyanobacteria genera, total phytoplankton cell number, and selected water quality parameters over time in the SWP Dam. Grey and black circles indicate the periods prior to intervention and during intervention respectively.

Chl *a*: Chlorophyll A; NH3: Ammonia; TKN: Total Nitrogen; OP: Orthophosphate; TPU: Total Phosphate - Unfiltered.

Cyanobacteria cells was not significantly altered in the SWP Dam when preventative measures were in place. In particular, treatment had no significant effect on cell counts for the cyanobacterial genera *Anabaena* and *Microcystis*, total cyanobacteria, or total phytoplankton (Table 6.3). Depth ranges however had a significant effect on the aforementioned cyanobacterial genera, total cyanobacteria, and total phytoplankton (Table 6.3).

Table 6.3. Repeated measures mixed model analyses indicating the influence of treatment, depth ranges, and the interaction between treatment and depth ranges on the abundance of (a) Anabaena, (b) Microcystis, (c) total cyanobacteria, and (d) total phytoplankton.

	(a) *Anabaena*			**(b) *Microcystis***		
Effect	*Df*	*F*	*P*	*df*	*F*	*P*
Treatment	1.126	0.07	0.79	1,126	3.69	0.06
Depth ranges	2.126	28.29	<0.001*	2,126	11.73	<0.001*
Treatm*Depth ranges	2.126	0.01	0.99	2,126	1.01	0.37
	(c) Total Cyanobacteria			(d) Total Phytoplankton		
Effect	*Df*	*F*	*P*	*df*	*F*	*P*
Treatment	1.126	2.51	0.12	1,126	2.23	0.14
Depth ranges	2.126	13.49	<0.001*	2,126	14.68	<0.001*
Treatm*Depth ranges	2.126	0.57	0.56	2,126	0.57	0.57

6.4. Discussion

6.4.1. Effectiveness of the phytoplankton control measures in reducing cyanobacteria growths in the SWP Dam

The Phoslock® treatment in the SWP Dam was found not to reduce TP as higher concentrations were recorded at the treatment area compared to the untreated (control) area. The lack of TP reduction seemed to indicate that the Phoslock active ingredient Lanthanum (La^{3+}) likely bound to compounds other than TP. It could be important to do jar tests in a laboratory setting before field treatment is planned to prevent high costs of such treatments. The higher TP concentration at the treated area could have led to the growth of more cyanobacterial cells in that area as compared to the untreated control area. The average cyanobacteria cell numbers were higher at the treated area (90 521cells/ml), compared to the untreated control area (55 338 cells/ml). Although there was a variation in abundance, TP and Orth-P on hourly sampling collected during treatment was still higher, and not different from what was recorded in the untreated (control) area (Table 2a &b).

The shortcoming of the Phoslock® treatment could have been due to the occurrence of higher concentrations of nutrients in the SWP Dam as a result of the malfunctioned WWTP upstream during this time, which was only fixed in 2014 after the treatment. However, the long-term effects of most restoration measures of this nature are also questionable as some reservoirs tend to return to normal states after trial periods (Jeppesen et al., 2017; Søndergaard et al., 2007). Materials such as zeolite, calcium compound, and clay are used to form a barrier that inactivates phosphorus (Pęczuła, 2012). For example, the application of bentonite clay (Phoslock®) in river Canning in Australia resulted in the reduction of phosphate concentration for a short-term. The decrease in phosphate due to Phoslock® treatment was also observed in the Swan-Canning River and Vesse in Australia (Robb et al., 2003). Phoslock® leads to short-term improvement in water quality immediately after treatment. This product is more useful in aquatic systems where phosphorus loads originates from the bottom sediment, rather than from external point sources or non-point such as catchment runoff (Burford et al., 2019).

The SPC treatment was also found not to reduce cyanobacterial cell numbers at the treatment area compared to the untreated control area. Surprisingly, even samples taken at the location of the SPC and those taken at the 50m distance interval from the machine reveal no temporal and spatial difference in cyanobacterial cell numbers. The poor performance of the SPCs in the SWP Dam could be related to the nature of the treatment (partial), which covered only 10% of the surface area, and the location of the SPCs toward the dam wall, which receives more cyanobacteria cell sometimes due to wind movement.

Successful mixing requires a high rate of movement in a large part of the lake and that light available to cyanobacteria is limited (Lürling & Mucci, 2020; Visser et al., 2016). (Burford et al., 2019) further mentioned that the effectiveness to prevent buoyant cyanobacteria from accumulating at the surface and to form blooms, is dependent on the severity and rate of mixing. They further mentioned that, mixing of surface water causes accumulation immediately outside the mixing zone (Burford et al., 2019). These preconditions could explain the increase in the cyanobacterial cell counts at the sampling points within the treatment area compared to the control area of the SWP Dam.

The successful treatment of SPCs as observed by (Hudnell et al., 2010) could be related to trials that were carried out in smaller and shallow reservoirs compared to the SWP Dam. Installation of SPCs in small shallow reservoirs with a surface area <1km^2 were observed to suppress cyanobacterial growths within the treatment area, which also showed an increase in diatoms (Hudnell et al., 2010). The findings from the two control measures revealed that, both treatments were not effective in reducing cyanobacterial cells, which was dominated mainly by *Microcystis* and *Anabaena*.

6.4.2. The combined effect of the phytoplankton control measures

The combined effect of the two control measures was observed with an increase in cyanobacterial cell numbers from 18 387 226 cells/ml before treatment to 22 836 511 cells/ml after treatment. The opposite was observed in the untreated reference VB Dam that could be due to low TP concentration, as this dam catchment area contains no towns and cities with potential activities to generate nutrients unlike the SWP Dam although on the same river. There was less TN available and more TP, which may have, promoted *Microcystis* dominance during the treatment period in the SWP Dam. The SWP Dam was found to be hypertrophic during the treatment period, while the VB Dam was eutrophic.

Nutrients coupled with other climatic factors such as temperature and sunlight play a critical role in the proliferation of cyanobacteria in many reservoirs. Over the past seventeen years, the concentration of nutrients in SWP Dam was sufficient in relationship with climate conditions for the nuisance cyanobacterial growth (United States Environmental Protection Agency, 2000) even before and after control measures. The nutrient concentrations in both the studied dams were found to be higher than that of Australian and Brazilian standard guideline for freshwater quality (Sharip & Suratman, 2012).

The high concentration of nutrients in the SWP Dam was reflected by the high concentration in suspended Chl *a* before and after control measures as compared to the nearby reference VB Dam. The concentration of the total phosphorus was lower before control measures in comparison to after control measures in the SWP Dam. While total nitrogen was high before treatment and decreased during the treatment period in the SWP Dam. The high concentration of total phosphorus compared to total nitrogen could be due to the change in the types of land use activities in the catchment of the SWP Dam during the study period. (Lehmann, 2010) states that wastewater overflow from Gorengab Dam, Ujam ponds, Chicken Farm, Okapuka

Tannery, Okahandja Sewage Ponds are reported to be the root cause of cyanobacteria blooms in the SWP Dam.

The high concentration of TP during treatment compared to before treatment could also be due to the accumulation in the sediment of the dam over time because of the increase in effluent discharge due to changes in land use activities in the SWP Dam catchment area. During the implementation of the phytoplankton control measures period, the Old Ujams Wastewater Treatment Plant (UWWTP), which treats industrial effluent, was malfunctioning due to overloads as result of the expansion of industries such as tannery, brewery, and abattoir effluent discharges. The UWWTP was replaced with a Membrane Bioreactor Plant in 2014. During the period of malfunction, the effluent was discharged in the Klein Windhoek River, which empties into the Swakop River. Besides, the overflowing of the Goreangab Dam, which holds domestic wastewater, could have also contributed to the increase in TP concentrations in the SWP Dam. The Goreangab Dam which is used to store domestic wastewater was overflowing during this period when the ammonia concentration was very high (i.e. max 2.3mg/l) and as a result, the water was not abstracted for treatment at the Windhoek Goreangab Operating Company (Pty) Ltd (WINGOC) Wastewater Treatment Plant. The overflow was release into the Otjiseva River, which drains into the Swakop River.

A study conducted by Cashman et al., (2014) showed that wastewater infrastructure in the city of Windhoek lacks maintenance and monitoring. As a result, numerous blockages are experienced, which cause wastewater to spill into nearby tributaries ending up in the SWP Dam, located downstream. Industries located in the city of Windhoek face financial challenges to maintain their wastewater treatment facilities (Cashman et al., 2014). Despite the above, lack of awareness of the effect of pollution on surface water quality was found to be the root cause of lack of compliance to discharge untreated effluent in tributaries (Kgabi & Joseph, 2012). Kgabi and Joseph, (2012), reported pollution from domestic waste, open defecation, and municipal sewer on the water quality of the Gamammas River situated in the western part of Windhoek. Nutrients emanating from the Windhoek area are resulting in an eruption of cyanobacteria blooms in the SWP Dam, affecting the quality of water to be subjected to treatment at the VB Treatment Plant. These studies support that, nutrient-rich effluent was emanating from the catchment area of the SWP Dam during the period of phytoplankton control measures.

Control of external nutrients minimise cyanobacterial growths of shallow and rapidly flushed reservoirs (Burford et al., 2019; Pęczuła, 2012). For example, in Denmark, the quality of shallow reservoirs have improved due to countermeasures such as modernization of sewage treatment plants, usage of phosphate-free detergents, increase in storage capacity of animal manure, 2m cultivation free riparian buffer zone, and increase afforestation (Pęczuła, 2012). With the implementation of the above measures, about 75% reduction in phosphorus has been achieved in Denmark shallow reservoirs. These measures could also be suitable for the SWP Dam catchment areas, although the reduction of 75% might not be achieved due to the difference in dam structure and climatic conditions (Bozelli, 2019; Jeppesen et al., 2017). For deeper dams of this type internal phosphorus in the sediments plays a critical role and this delays the reservoir recovery due to the reduction in external phosphorus by 10-15 years (Klapper, 2003; Pęczuła, 2012).

The period before the implementation of control measures and the period when control measures were in place in the SWP Dam did not segregate or group in ordinal space on a PCA triplot, providing evidence suggesting control measures did not affect cyanobacterial assemblages and the water quality parameters investigated.

In view of the ineffective control measures tested, it is proposed to improve the water quality of the SWP Dam, by repairing/renovating the failed WWTP to reduce the external nutrients from the watershed. Subsequently, internal nutrients reduction methods, such as sediment removal by dredging, could be implemented. Although dredging method is associated with environmental costs (Beklİoğlu, 1999; Gulati et al., 2008; Pęczuła, 2012) this was applied in Cockshoot Broad in Great Britain, and the Braband reservoir in Denmark (Pęczuła, 2012). A positive response such as a decline in Chl *a* and phosphorus were noted (Pęczuła, 2012). However, these are all shallow reservoirs of less than 1m deep compared to SWP and VB dams, which are 30m deep. (Bozelli, 2019), advocates for the focus on urgent restoration, which implies focusing on environmental protection and management of the catchment area, because the traditional restoration measures yield less convincing results and are very costly. This could be a solution to restore SWP Dam, although this might take longer to implement and yield results, it is more sustainable compare to the other costly and complex traditional restoration measures.

6.5. Conclusion and recommendations

In conclusion, the two phytoplankton control measures (Phoslock® and Solar Powered Circulation) were found ineffective to reduce the cyanobacterial cell numbers over the study period as high cell counts of cyanobacteria were recorded at the treatment areas compared to untreated control areas. The combined effect of the two control measures was also found ineffective as more cyanobacterial cell numbers were recorded during the treatment period. The concentration of the nutrients was found suitable to support cyanobacteria growths during both individual and combined assessment. During the application of the control measures, the SWP Dam was hypertrophic, with an increase in TP and OP, which could be related to changes in land-use activities. A Principal Component Analysis triplot indicates no grouping pattern in the ordinal space of the years prior to and during the implementation of phytoplankton control measures. Repeated measures mixed model analyses indicate that treatment had no significant effect on cyanobacteria cell counts. It was evident that, the two phytoplankton control measures were ineffective in reducing cyanobacterial cells during the individual and combined assessment, which could be due to the small treatment area or treatment period and higher TP and OP concentrations during the treatment period from the malfunctioned WWTP upstream. The outcome of the current study could assist water managers in the future on the selection of appropriate restoration measures related to the treatment area and for dams with high nutrient enrichment situated in warmer arid environments.

As a result, the following recommendations are proposed when considering restoration of the SWP Dam: Firstly, there is a need to understand the trophic relationships, climatic conditions, the concentration of the internal nutrients, and sources of external nutrients of the SWP Dam. Secondly, water managers need to focus on point and non-point sources of nutrient pollution in the upper catchment of the SWP Dam since these are the root causes of the degradation of the SWP Dam water.

CHAPTER 7: OVERALL CONCLUSIONS AND RECOMMENDATIONS

7.1. Summary of the findings

It is evident that surface water quality, including in reservoirs that are a critical part of water supply, will continue to deteriorate due to pollutants emanating from anthropogenic activities. A notable example of the deterioration in the quality of water sources is the increase in reports on the occurrence of toxic cyanobacteria blooms in reservoirs. There is strong evidence that this will be exacerbated by climate change. Most anthropogenic activities in the catchment area of dams are perceived as needed for human development to support industry and enhanced economic and agricultural development to improve livelihoods. However, waste generated by these activities needs to be regulated and better managed to mitigate the negative impact of such waste on overall water quality. This will ensure the sustainability of the water supply from these dams, contribute to water security and enhance the resilience of the community dependent on such water sources from climate change impacts.

Anthropogenic activities such as mine-tailing dams, hazardous waste landfill sites, pit-latrines, wastewater treatment plants, leaking sewer pipelines, wastewater ponds, industrial effluents and untreated sewage introduce pollutants that affect the quality of water. Of a major concern is the accompanying nutrients that cause eutrophication that leads to eruptions of toxic cyanobacteria blooms. Under the directorate of Namibia's Water Affairs, existing legislation in the Ministry of Agriculture Water and Land Reform mandates the regulations of nutrients and other chemical constituents and stipulates that effluents containing the pollutants as listed above may not to be discharged into surface waters without treatment. However, the enforcement of such regulations are often lacking and improvement in this regard is essential. Similar to Namibia, this is especially important for developing countries that experience rapid urbanization and industrial growth.

In Namibia urbanization occurred rapidly after independence. The total urban population of Namibia has increased from 28% at independence to 53% in 2021. The capital city of Windhoek has been a major focal point of urbanization, with its population nearly doubling from 233 529 in 2001 to 446 000 in 2021 due to urban population influx, population growth, rural-urban migration, and industrialization. This increase, in both formal and informal settlements has put tremendous pressure on service delivery like water supply and sewer networks. Access to services in the informal settlements of Windhoek and Okahandja, both in the catchment area of the SWP Dam is very poor leading to open defecation, and when coupled with effluent from mines, overflowing wastewater holding facilities, leakages, and blockages of wastewater pipelines result in pollutants ending up in the SWP and VB dams, which are critical water sources for the central region of Namibia, including Windhoek. Severe water shortages and excessively high treatment cost will follow if this trend continues, which highlights the urgency for introducing preventative initiatives that focus on reducing pollution in the upstream catchments.

This study attempted to describe factors, such as pollution caused by rapid urbanization and industrial growth, that affect two major Namibian dams to support decision-making towards improved water security. Analysing data such as rainfall, water inflow and other available information over a period of 17 years, the aim was to better understand factors impacting the proliferation of phytoplankton and related water quality status. The focus was on the dynamics and control measures currently implemented to mitigate the risk of toxic phytoplankton, as well as factors controlling growth during period of prolonged drought. Few studies have been conducted in Africa using the comparative assessment approach to understand the deterioration in water quality of ephemeral-river connected dams. Therefore, the information revealed by the study should be useful to utility managers at the two dams and the treatment plant to ensure the sustainable management of the water sources. Delineating the link between pollution status and effective management is needed to ensure abstraction of water with minimal toxic cyanobacteria to deliver safe and clean water to the consumers. The information should also be of value to other countries in the region.

Using water quality indices related to aquatic ecosystems, it was shown in *chapter 3* that both dams were polluted with nutrients, with high salinity and particulate matter over a period of 17 years. The autumn months were reported with the highest pollution in SWP Dam, compared to the summer and spring months in the case of VB Dam. The concentration of nutrients was higher in the spring in the VB Dam, which could be attributed to transfers of nutrient rich water from SWP Dam to augment supply, compared to autumn in the case of SWP Dam. Salinity was highest in spring in both two dams. The particulate matter was highest during the summer months in VB Dam, which is attributed to turbulent water inflow carrying silt after heavy rains, compared to autumn in SWP Dam which could be due to phytoplankton blooms since phytoplankton biomass (Chl *a*) was highest in the SWP Dam in autumn, compared to spring in VB Dam. Pollution levels in the VB Dam ranged between moderately and heavily polluted in most quality parameters, except for F, $S0_4$, and Mn. In the SWP Dam, the majority of the assessed water quality parameters were heavily polluted, also with the exception of F, $S0_4$, and Mn. The impact of the poor quality of water transferred from the SWP Dam to the VB Dam was evident, showing that better management of the SWP catchment is needed for better quality water in both dams.

In *chapter 4*, it was shown that the phytoplankton communities in the two dams were dominated by potentially toxic cyanobacteria, mainly *Microcystis*, followed by *Dolichospermum* at all the depth ranges and seasons. The dry seasons of autumn, spring, and winter were characterized by increased phytoplankton cell numbers compared to the wet summer season. Spring and autumn cyanobacterial blooms were observed in the two dams with more phytoplankton cell numbers compared to winter and summer. While the potentially toxic phytoplankton was at depths of 5 to 10m in the SWP Dam, they were positioned between 0 to 5m in VB Dam. The depth ranges preferred by cyanobacteria corresponded to the depths at which favourable water temperature (i.e. average temperature in SWP Dam 19.8 ± 3.1°C and in VB Dam 20.4 ± 4.1°C) and nutrient concentrations (i.e. average total phosphorus in SWP Dam 0.38 ± 1.38 and in VB Dam 0.28 ± 0.62**)** was observed.

The influence of prolonged drought periods on phytoplankton biomass and cyanobacterial blooms was investigated in *chapter 5*. The SWP Dam was hypertrophic, and VB Dam was eutrophic in both drought and rainy years over the 17 years, with higher average TP concentration in rainy years. VB Dam recorded more phytoplankton biomass and cyanobacterial cells during rainy years, while SWP Dam during drought years. Other water quality variables such as WT, pH, Zeu/Zmix, EC, and Cl increased in concentration during the drought years relative to the rainy years in both dams. Although there was a higher concentration of nutrients in the rainy years compared to the drought years, the variation in cyanobacterial cell numbers in both dams was best related to water column depths, season, Vol % (percentage of full capacity), turbidity, NH_3, and pH. Suspended Chl *a* was best related to water column depths, vol % (percentage of full capacity), EC, Cl, pH, turbidity, NH_3, TP:TN ratio, and OP. The reduction vol % (percentage of full capacity) during drought was observed to have an effect on phytoplankton biomass in SWP Dam, but not in VB Dam. Overall, despite physiochemical parameters favourable during drought years in both dams, the pattern and magnitude varied between them, indicating that other environmental drivers such as land use will be needed to further elucidate the response of water quality to droughts.

In *chapter 6*, the two phytoplankton control measures (Phoslock® and Solar Powered Circulation) were found ineffective to reduce the cyanobacterial cell numbers over the study period as high cell counts of cyanobacteria were recorded at the treatment areas compared to untreated control areas. The combined effect of the two control measures was also found ineffective as more cyanobacterial cell numbers were recorded during the treatment period. The concentration of the nutrients was found suitable to support cyanobacteria growths during both individual and combined assessment. During the application of the control measures, the SWP Dam was hypertrophic, with an increase in TP and OP, which could be related to changes in land-use activities. A Principal Component Analysis triplot did not reveal grouping pattern in the years prior to, and during the implementation of phytoplankton control measures. Repeated mixed model analyses indicated that treatment had no significant effect on cyanobacteria cell counts.

7.2. Recommendations and future research

To address the challenges described in this dissertation, and to improve the water quality of the two dams discussed, while ensuring water transfer for water security in central Namibia, several measures are needed in the upper catchment of the SWP Dam, as follows:

- fixing or upgrading of dysfunctional wastewater treatment plants;
- enforcement of compliance of industries to effluent discharge standards;
- annual stakeholders engagement on pollution issues in the upper catchment of the Swakop River;
- mapping of land use activities and pollution monitoring in the catchment areas of the two dams;
- monitoring of the accumulative impact of the polluted water transferred into the VB Dam on parameters of concerned; and
- utilisation of the newly developed water quality indices to aid pollution monitoring in the catchments, with further refinement of these indices to include toxic phytoplankton species.
- monitoring the toxicity of dominant cyanobacteria species in the two dams

These measures may need refinement and therefore warrant research that are focused on ensuring water security in the context of population growth and rapid urbanization, and climate change. For instance, the study demonstrated vertical and temporal dynamics in the phytoplankton communities of the dams, which need to be better incorporated in future selective withdrawal at varying depths to minimize abstraction of potentially toxic phytoplankton species.

In addition to evaluating nascent technologies, future research should also consider the provision of water in the context of growing financial constrains, which directly impact day-to-day operation of utility managers in aspects such as preparing for possible eruptions of toxic cyanobacteria. This is especially relevant in cases of water transfer between dams such as described in this dissertation. Growing financial constrains also justify objective assessment of the feasibility of water tariffs and development of appropriate polluter-pay priciples.

Overall, water provision cannot be achieved without considering some of the challenges in the sustainable revolution, namely a rapidly increasing world population in the context of dwindling resources, and less time to adapt. Lastly future research on risk models for predicting cyanobacterial blooms in the two dams in different seasons should be considered.

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