INTEGRATED ECOLOGICAL INTEGRITY ASSESSMENT OF LAKE SIMBI, A DEEP ALKALINE-SALINE LAKE IN WESTERN KENYA.

BY

STEPHEN BALAKA OPIYO

BSC. NATURAL RESOURCES MANAGEMENT (KISII UNIVERSITY)

A THESIS SUBMITTED TO THE SCHOOL OF POST-GRADUATE STUDIES IN PARTIAL FULFILMENT OF THE REQUIREMENTS FOR THE AWARD OF DEGREE OF MASTER OF SCIENCE IN LIMNOLOGY OF THE SCHOOL OF AGRICULTURE AND NATURAL RESOURCES MANAGEMENT, DEPARTMENT OF FISHERIES AND AQUATIC SCIENCES, KISII UNIVERSITY.

NOVEMBER, 2019

DECLARATION

This thesis is my original work and has not, wholly or part, been submitted or presented for examination for an award of a degree in any other university.

Signature: ………………………………………………… Date 19/07/2019…………

Name: STEPHEN BALAKA OPIYO

Reg No: MAN19/50101/17

Department of Fisheries and Aquatic Sciences,

Faculty of Agriculture and Natural Resources Management,

Kisii University.

RECOMMENDATION/APPROVAL

This thesis has been submitted for examination with our recommendation and approval as University Supervisors.

SUPERVISORS:

Signature: ………………………………………………… Date 11/07/2019…………

Prof. Albert Mochache Getabu

Department of Aquatic Sciences and Fisheries, Kisii University.

Signature: ………………………………………………… Date 18/7/2019…………

Prof. Anakalo Shitandi

Department of Research and Extension (Registrar), Kisii University.

Signature: ………………………………………………… Date 08/07/2019…………

Dr. Lewis Sitoki

Department of Geo-Science and the Environment, Technical University of Kenya.

PLAGIARISM DECLARATION

PLAGIARISM DECLARATION

Definition of plagiarism

Is academic dishonesty which involves; taking and using the thoughts, writings, and inventions of another person as one's own.

DECLARATION BY STUDENT

i. I declare I have read and understood Kisii University Postgraduate Examination Rules and Regulations, and other documents concerning academic dishonesty.
ii. I do understand that ignorance of these rules and regulations is not an excuse for a violation of the said rules.
iii. If I have any questions or doubts, I realize that it is my responsibility to keep seeking an answer until I understand.
iv. I understand I must do my own work.
v. I also understand that if I commit any act of academic dishonesty like plagiarism, my thesis/project can be assigned a fail grade ("F")
vi. I further understand I may be suspended or expelled from the university for academic dishonesty.

Name: Stephen Balaka Opiyo Signature:________

Reg. No: MAN19/50101/17 Date: 19/07/2019

DECLARATION BY SUPERVISOR (S)

i. I/we declare that this thesis/project has been submitted to plagiarism detection service.
ii. The thesis/project contains less than 20% of plagiarized work.
iii. I/we hereby give consent for marking.

1. Name: Prof. Albert Mochache Getabu Signature:________

   Affiliation: Kisii University Date: 11/07/2019

2. Name: Prof. Anakalo Shitandi Signature:________

   Affiliation: Kisii University Date: 18/7/2019

3. Name: Dr. Lewis Sitoki Signature:________

   Affiliation: Technical University of University Date: 08/07/2019

DECLARATION OF NUMBER OF WORDS FOR MASTERS THESIS

DECLARATION OF NUMBER OF WORDS FOR MASTERS/PROJECT/ PHD THESES

This form should be signed by the candidate and the candidate's supervisor (s) and returned to Director of Postgraduate Studies at the same time as you copies of your thesis/project.

Please note at Kisii University Masters and PhD thesis shall comprise a piece of scholarly writing of not less than 20,000 words for the Masters degree and 50,000 words for the PhD degree. In both cases this length includes references, but excludes the bibliography and any appendices.

Where a candidate wishes to exceed or reduce the word limit for a thesis specified in the regulations, the candidate must enquire with the Director of Postgraduate about the procedures to be followed. Any such enquiries must be made at least 2 months before the submission of the thesis.

Please note in cases where students exceed/reduce the prescribed word limit set out, Director of Postgraduate may refer the thesis for resubmission requiring it to be shortened or lengthened.

Name of Candidate: Stephen Balaka Opiyo ADM NO: MAN19/50101/17

Faculty: FANRM Department: Fisheries and Aquatic Sciences

Thesis Title: Integrated Ecological Integrity Assessment Based on Phytoplankton Indices, Water Quality Parameters, Habitat Quality Survey and Trophic State Indices in Lake Simbi, a Deep Alkaline-Saline Lake in Kenya.

I confirm that the word length of:

1) the thesis, including footnotes, is ....43,828........... 2) the bibliography is ....6,113...........

and, if applicable, 3) the appendices are ....1,078.......

I also declare the electronic version is identical to the final, hard bound copy of the thesis and corresponds with those on which the examiners based their recommendation for the award of the degree.

Signed: .... Date: 19/07/2019

(Stephen Balaka Opiyo)

I confirm that the thesis submitted by the above-named candidate complies with the relevant word length specified in the School of Postgraduate and Commission of University Education regulations for the Masters and PhD Degrees.

Signed: .... Email: getabu@kisiiuniversity.ac.ke Tel: 0726544456 Date: 11/7/2019

(Prof. Albert Mochache Getabu)

Signed: .... Email: ashitandi@gmail.com Tel: 0722153494 Date: 18/7/2019

(Prof. Anakalo Shitandi)

Signed: .... Email: sitoki@hotmail.com Tel: 0723379868 Date: 08/07/2019

(Dr. Lewis Sitoki)

COPYRIGHT

DEDICATION

DEDICATED WITH EXTREME AFFECTION AND GRATITUDE TO

My parents Mr. Stanslaus Opiyo and Mrs. Sofia Opiyo

My siblings Elisha, Hanna, Tony,Naomi and Phoebe (Deceased)

My wife Mrs. Martha Nafula Opiyo

God bless you all!

ACKNOWLEDGEMENT

First, my special and sincere gratitude go to my supervisors Prof Albert Getabu, Prof. Anakalo Shitandi and Dr. Lewis Sitoki for their wise counsel, motivation and constant guidance. Their dedicated supervision and passion for research was not only a prerequisite for the successful development of my research study but also a lifetime inspiration. I would like to thank the Kisii University lecturers in the Department of Aquatic and Fisheries Sciences for the skills and knowledge gained. Special appreciation goes to my fellow AfDB graduate students for the commentaries and encouragement throughout my studies.

My special thanks also go to my family, parents (Sofia and Stanley Opiyo), siblings (Elisha, Hannah, Tonny, Naomi) and beloved wife (Martha Nafula), without whom this incredible experience would not have been possible. I acknowledge with much gratitude the role-played by KMFRI staff Mr. Ephraim Odada, Mr. Jared Babu, Mr. James Achiya and Mr. Moses Umani at Kisumu Station and many others who directly or indirectly supported me during my research proposal development period.

I am highly indebted to the African Development Bank for granting me a full scholarship to pursue a Master of Science degree in Limnology at Kisii University. Finally, I must express my sincere gratitude to Almighty God for giving me the ability, will and resources to get this far in my academic endeavors. Glory to God, Amen!

ABSTRACT

The resources and ecological services derived from lentic ecosystems are constantly threatened by climate change and unprecedented pressures from anthropogenic activities. This has led to alteration of their physical habitat, water quality, trophic state and aquatic biota creating a constant need to assess their integrity for management and conservation purposes. Lake Simbi National Bird Sanctuary is the least studied lake in Kenya despite being a bird haven with ecotourism value; gaps exist on the lake's biota and other key ecological aspects such as the water quality dynamics, phytoplankton and the trophic status. To bridge this knowledge gap, the current study was employed in assessing Lake Simbi's ecosystem integrity which has been brought into question by the recent decline in the numbers of the "near-threatened" lesser flamingos unique only to this lake. The objective was to assess the ecological integrity based on the water quality dynamics phytoplankton indices, habitat quality and trophic state index (TSI). Sampling was done on a monthly basis for 6 months from December 2018 to May 2019 at six fixed stations systematically selected. Standard methods stipulated in APHA (2012) were used in determining levels of selected water quality parameters and phytoplankton characteristics. The Carlson's and Nygaard's trophic state indices were used in determining the trophic status of the lake. Foot-based lake habitat survey based on the European Union Water Framework Directive standards was conducted to determine the physical habitat quality of the lake. The spatial and temporal trends of the parameters were analyzed using Excel and SPSS software. Phytoplankton diversity was estimated using Shannon-Wiener, Simpson, evenness and richness indices. Descriptive and correlation statistics was done for all variables. ANOVA and Tukey test for comparisons of means were used to screen for significant differences ($p < 0.05$) among the study variables. The results indicated that the water quality of Lake Simbi is heavily polluted and unsafe for any domestic usage since almost all the basic water quality variables (DO, pH, TDS, alkalinity, hardness and turbidity) measured exceeded the maximum permissible limits set by both National Environmental Management Authority and the World Health Organization. A total of 84 phytoplankton species were identified comprising of Cyanophyceae (36), Chlorophyceae (25), Bacillariophyceae (11), Zygnematophyceae (4), Dinophyceae (3) and Euglenophyceae (3). Reduced abundance of the cyanobacterial species, especially *Spirulina* species on which flamingos feed and the subsequent dominance of the toxin producing *Microcystis* sp. might have contributed to the decline of the flamingo population. The Carlson's and Nygaard's trophic state indices both characterized the waters as hypereutrophic. The Redfield's TN/TP ratios established that N was the limiting nutrient in Lake Simbi, also indicating its eutrophic nature. The Lake Habitat Survey indices of Lake Habitat Quality Assessment and Lake Habitat Modification Score collectively suggested that the Lake Simbi physical habitat is moderately pristine since its hydromorphology is moderately modified by the various pressures which are still operating at marginal scales. For most of the variables studied, significant temporal variations were realized while no significant spatial variations were observed. Generally, the poor ecosystem integrity of Lake Simbi can be attributed to the anthropogenic activities in its catchment and the changing climatic conditions. This pioneer study demonstrates that multivariate ecological indices can be effective in monitoring the ecological status of aquatic ecosystems in Kenya, hence should be adopted as a tool for informing decision-making for lake conservation and management purposes.

Key words: Ecological integrity, Lake Habitat Survey, nutrients, physico-chemical, phytoplankton, trophic state index and Lake Simbi.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

LIST OF ABBREVIATIONS

Chl-a:	Chlorophyll-a
DO:	Dissolved Oxygen
EC:	Electrical Conductivity
Mg m^{-3}:	Milligrams per Cubic meter
Mg L^{-1}:	Milligrams per Litre
Mm:	Millimetre
Sal.	Salinity
SD:	Secchi Depth
SE:	Standard Error of mean
SRP:	Soluble Reactive Phosphorous
TA:	Total Alkalinity
TH:	Total Hardness
TDS:	Total Dissolved Solids
TN:	Total Nitrogen
TP:	Total Phosphorous
Turb.:	Turbidity
TSS:	Total Suspended Solids
µg L^{-1}:	Micrograms per Litre
µS-cm^{-1}:	Micro-Siemens per centimetre
WT:	Water Temperature

LIST OF ACRONYMS

ANOVA:	Analysis of variance
APHA:	American Public Health Association
IUCN:	International Union for Conservation of Nature
KMFRI:	Kenya Marine and Fisheries Research Institute
KWS:	Kenya Wildlife Service
LHS:	Lake Habitat Survey
LHMS;	Lake Habitat Modification Score
LHQA:	Lake Habitat Quality Assessment
NEMA:	National Environmental Management Authority
SPSS:	Statistical Package for Social Sciences
TSI:	Trophic State Index
UNDP:	United Nations Development Program
UNEP:	United Nations Environmental Program
UNESCO:	United Nations Educational Scientific and Cultural Organization
USEPA:	United States Environmental Protection Agency
USGS:	United States Geological Society
WHO:	World Health Organization

CHAPTER ONE

INTRODUCTION

1.1 Background to the Study

The fauna and flora on earth depend on water for survival. Water is therefore an essential life-supporting component of the environment. The waters are contained in the inland and marine water bodies. The ecology of inland water bodies is studied in the scientific field of limnology. According to Gupta and Gupta (2006), limnology encompasses a study of the inland waters, both fresh and saline, and the factors that regulate life found in them.

Limnological studies have been globally prevalent because of the numerous ecological goods and services offered by aquatic ecosystems. According to Williams (1981), these studies focused more on freshwater resources as compared to saline water resources; even though the extent of both types of inland waters on the planet earth is nearly equal (Wetzel, 1983). Saline water resources in Africa have unique properties. According to McCulloch, Irvine, Eckardt and Bryant (2008), they are closed lakes with elevated levels of salinity and alkalinity, a combination of factors that generate drastic conditions that greatly influence the community structure of the living organisms found in them. The ecology of saline (soda) lake ecosystems such as Lake Simbi are of great importance since they support maximum levels of primary production (Talling, Wood, Prosser & Baxter, 1973). Alkaline-saline lakes of East Africa occur within the Intertropical Convergence Zone (ITCZ) where the erratic climatic patterns have substantial impacts on the lake's ecosystem and biodiversity (Smol & Stoermer, 2010). Since the unique ecology of these lakes is not well understood, detailed knowledge on lake ecosystems is necessary to formulate proper policies for management and conservation, which is why ecological assessment of these ecosystems has been receiving intense global interest over the years. Various aspects have been used to assess lake ecosystems.

The first and most common method is water quality assessment which involves measuring the physico-chemical characteristics of the water (Dhanya, 2017) including nutrients. This is because water quality governs the existence and the interactions of plants and animals in the water masses (Adoni *et al*., 1985). The nutrient dynamics defines the trophic state of aquatic ecosystems (Jekatierynczuk-Rudczyk *et al.*, 2014). According to Dodds and Cole (2007), the trophic state is indicative of the biological productivity in these environments. The changes in nutrient concentrations can lead to changes in the structure of the community in a particular trophic level, therefore numerous trophic state indices have been formulated for assessing these particular variations occurring in the ecosystem (Jeppesen, Jensen, Søndergaard, Lauridsen & Landkildehus, 2000). The assessment of the trophic state is crucial for the formulation of strategies for conservation and management (Sharma, Kumar & Rajvanshi, 2010).

The second method of ecological assessments is bio-monitoring which involves the use of living organisms as bio-indicators. Trishala, Deepak and Agrawal (2016) describes these bio-indicators which are also known as biomonitors as living organisms including plants, micro-organisms, animals and planktonic organisms that are used in evaluating ecological health of natural environments. Of all the plankton, the phytoplankton are more crucial since they play several roles in water ecosystems apart from the primary function of primary production which is crucial to the existence of other aquatic life forms since they form the basis for the food web which sustains the whole ecosystem. Phytoplankton dynamics such as their composition and abundance can reveal the nutritional status of aquatic ecosystem (Liu *et al.*, 2015), and the trophic state (Thakur, Jindal, Singh & Ahluwalia, 2013) and hence a central indicator of the conditions of an ecosystem (Domingues, Barbosa & Somme, 2012; Wang, Wei & Zhou, 2013). The zooplanktons are useful indicators of water pollution in lakes (Ramchandra, Rishiram & Karthik, 2006). These explain planktonic organisms are one of the most common bio-indicators used in ecological assessments.

The third method is habitat quality assessments which involve the use of geo morphological characteristics and some biological attributes that influence the habitat structure and energy input. One of the most common tools is Lake Habitat Survey, developed developed by an independent team of researchers to support the implementation of the European Union Water Framework Directive (WFD) (Rowan, 2005). The CEN (2011) listed LHS as a standard tool for use in assessing physical habitat of lakes in the European Union and ever since it has been widely applied across the world.

The last method is contained in the Water Framework Directive (WFD), established by the European Directive 2000/60/EC as guiding outline for water policy (European Commission, 2000). This method is an integrative framework involves the combination of physico-chemical, biological and hydro-morphological properties in the lake ecological assessments. This method is all-inclusive and therefore will be applied in establishing the ecological status of Lake Simbi.

Lake Simbi is an alkaline-saline lake and a vital ecosystem for numerous bird populations especially the lesser flamingoes which feeds on its phytoplankton assemblages. The ecosystem of this lake is at a greater risk of losing its quality and consequently its utility value because of the anthropogenic and climate-change related factors that seem to be altering its biological community as currently seen with the declining numbers of the bird populations and the deteriorating habitat conditions (Hayombe *et al.*, 2014; Oduor, 2018). Several human activities are carried out in the catchment of Lake Simbi including salt *(bala)* mining, waste disposal, agricultural and deforestation activities among others. The impacts of these activities pose a threat on the lake's ecosystem hence a need to assess its integrity. The present study is designed to provide information on the ecological status of Lake Simbi through an integrative WFD method involving the assessment of water quality, phytoplankton diversity, habitat quality and the trophic state indices. This is crucial since it will provide more insight about several of aspects of this ecosystem and this kind of knowledge as (Nowrouzi & Valavi, 2011) points out, may be a useful tool for the planning, restoration and conservation.

1.2 Statement of the Problem

Lake Simbi, an alkaline-saline lake adjacent to the Nyanzan Gulf in Homabay County of Kenya, is experiencing a deterioration of its ecological integrity as evidenced through the declining bird populations and the visibly degraded habitat quality (Hayombe *et al.,* 2014; Oduor, 2018). A declining ecological integrity could potentially translate to a diminished ecosystem, social, economic and aesthetic value of Lake Simbi. The Lake supports livelihoods through tourism and tourism-related activities since it provides spectacular scenery of large congregations of different birds but of major significance being the exotic lesser flamingo (*Phoeniconaias minor*) populations that migrate into it for feeding and breeding during certain seasons. The massive bird population is supported majorly by the

Arthrospira fusiformis, cyanobacterial algae that is prevalent in most alkaline-saline lakes. Recent reports (NMK & KWS, 2010; Oduor 2018) indicate that the populations of the lesser flamingo are declining in Lake Simbi as is the trend in other saline lakes around the world, and the changing phytoplankton community dynamics seems to be the main cause since it provides a food resource for the bird populations (Kyalo, 2012). The phytoplankton communities are at the core of any lake ecology since they play a significant role in primary production, sustenance of the food web and the substance turn over in aquatic environments, and therefore changing phytoplankton dynamics is an indicative of a changing biological community in the ecosystem. The changing biological community of the lake has been attributed to pollution loading (Kyalo, 2012), from nearby agricultural farms and communities, anthropogenic pressure from human encroachment through activities such as sand harvesting, and climate change impacts. A combined synergy of all these factors is causing the deterioration of the ecological health of the lake and chocking its ecosystem. There's little information about the present ecological integrity and status of the lake, since no known research on this has been conducted in the past one decade. Therefore, there is an urgent need to carry out an ecological integrity assessment of Lake Simbi based on the water quality, phytoplankton diversity, habitat quality survey and the trophic state index so as to gain insights that could help in its effective improvement, conservation and management by the relevant authorities, and hence the restoration and maintenance of its tourism, cultural and scientific value.

1.3 Justification of the Study

Lake Simbi is a national bird sanctuary managed by the Kenya Wildlife Service (KWS). It is an important ecosystem in the western Kenya region supporting ecotourism through its significant unique bird populations especially the lesser flamingoes which provide popular attraction for a substantial amount of bird watchers yearly, and through the cultural legendary stories that have been published in literature. In addition, the Lake also supports a unique biodiversity conservation, recreation and salt (*bala*) mining activities for the locals. All these services improve the quality of life of the local communities as well as earning revenue for the Homabay County government hence spurring economic development in the region. Because of these reasons, a report by Deloitte (2014) commissioned by the Lake Region Economic Bloc (LREB) formed by counties in the Western Kenya listed it as an ecotourism site of great potential that should be tapped. It's noteworthy that tourism sector is the greatest revenue generating sector in the Kenyan economy.

But the deteriorating health of the lake's ecosystem has become an issue of concern for NEMA, KWS, the counties that make up the Lake Regional Economic bloc and community-based environmental groups such as the Lake Simbi Nyaima Conservancy who are advocating for its restoration and improvement. However, restoration efforts would require a comprehensive assessment of the overall ecosystem health in terms of the biota, the water quality, habitat quality and the trophic state to provide profound insights on the ecological integrity of the lake's ecosystem.

Since, there's limited limnological and ecological data on Lake Simbi, this present study was carried out with the aim of assessing the ecological integrity of Lake Simbi to bridge the existing information lacunae on the ecological integrity status of the lake by documenting the phytoplankton dynamics, water quality status, habitat quality status and the trophic state of the lake. This baseline information is crucial for the designing of evidence-based holistic sustainable management strategies by the relevant agencies to maintain Lake Simbi's significance as an ecotourism, cultural, educational and scientific site, and preventing errors which commonly occur to the blanket transfer of management approaches effective in different ecosystems. The knowledge generated is a crucial reference for future researchers, educationists and scientists. Conservation and management of Lake Simbi is likely to be beneficial in the conservation of the exotic "near threatened" lesser flamingos that sojourn there seasonally. These birds are of great scientific and ecological significance.

1.4 Objectives of the Study

1.4.1 General Objective

To assess the ecological integrity of Lake Simbi based on water quality (physico-chemical), phytoplankton diversity, trophic state indices and habitat quality to generate baseline information useful for conservation and management purposes.

1.4.2 Specific Objectives

(i) To assess the spatial and temporal variation of selected physico-chemical parameters of Lake Simbi;

(ii) To determine the spatial and temporal variation of phytoplankton abundance, species composition and diversity indices in Lake Simbi;

(iii) To determine the spatial and temporal variation of the trophic state indices (TSI) in Lake Simbi;

(iv) To evaluate the spatial variation of habitat quality indices in Lake Simbi;

(v) To establish the relationship between the physico-chemical parameters, the phytoplankton community structure and the trophic state indices of Lake Simbi.

1.5 Null Hypotheses

(i) There are no significant differences in the spatial and temporal distribution of the selected physico-chemical parameters in Lake Simbi.

(ii) There are no significant differences in the spatial and temporal distributions of phytoplankton abundance, species composition and diversity indices in Lake Simbi.

(iii) There are no significant differences in the spatial and temporal variation of trophic state indices (STI) in Lake Simbi.

(iv) There are no significant differences in the spatial variation of habitat quality indices in Lake Simbi.

(v) There are no significant differences in the relationship between the physico-chemical parameters, the phytoplankton community structure and the trophic state indices.

1.6 Limitations of the Study

The information generated from this research study can only be applicable to Lake Simbi and might not be generalized to the other saline-alkaline lakes in Kenya. The results attained from this study only represent the research site during the time of the study with the prevailing environmental settings and therefore their extrapolation may not represent a true picture of the time to come. Presence of a few previous studies on Lake Simbi was also a limiting factor.

1.7 Assumptions of the Study

The basic assumption during the sampling time frame is that the ecological conditions in Lake Simbi were relatively stable. In addition, the study assumed that the dynamic equilibrium from year to year of the Lake Simbi's water quality variables assumed a constant state. The study also assumed that the selected sampling stations were representative of the whole lake ecosystem.

1.8 The Scope of the Study

This study was restricted to the Lake Simbi, a small-sized saline-alkaline lake in Kenya. It concentrated on assessing the ecological integrity of the lake based on the water quality (physico-chemical), phytoplankton diversity, habitat quality survey and trophic state index (TSI) throughout the sampling time frame of December 2018 to May 2019. Profiling of the physico-chemical parameters was limited to: -Temperature (ºC), Dissolved Oxygen concentration (mgL^{-1}), pH, conductivity (μScm^{-1}), salinity, hardness, Turbidity (NTU), Total Dissolved Solids (TDS), alkalinity and transparency.

Nutrient profiling was limited to: - Total Nitrogen (TN), Soluble Reactive Nitrates (NO_3-N), Nitrites (NO_2-N), Ammonia (NH_3-N), Total Phosphorous (TP), Soluble Reactive Phosphates (SRP) and Silicates (SiO_2). The phytoplankton analysis was limited to the species composition, occurrence and abundance. The trophic state indices were limited to the Carlson's and Nygaard's trophic state indices. Lake Habitat Survey (LHS) was limited to assessing the vegetation cover, the substrate properties, morphology and the human pressures which were recorded, analyzed and scored on the two LHS metrics of Lake Habitat Quality Assessment (LHQA) and Lake Habitat Modification Score (LHMS).

CHAPTER TWO

LITERATURE REVIEW

2.1 Introduction

Limnological and ecological research in African lakes have been evolving and gaining momentum among the scientific community since the 19^{th} century. These studies focus on inland waters both saline and fresh. Water drives life on earth and that explains why about three-quarters of the earth is covered by it. The total volume of water on earth is estimated to be about 1.3 billion cubic kilometers of which an estimated 97.5% is saline distributed in lakes and oceans (National Geographic, 2016). A lake is a closed body of water surrounded by land (Otterbine, 2003). Lakes are formed either through the processes of natural catastrophes such as tectonics or artificially by human beings (Jørgensen, 1980). Alkaline lakes are no exception, they originate from the activities of both tectonics and vulcanicity and according to Thomas, Meybeck and Beim (1996), and this involves the processes of folding, tilting or faulting.

Lakes age through eutrophication which results in changes in their water quality (Gilbert, 1991). The lake features are defined by the lake's origin, water balance, nutrient concentration levels and the physico-chemical parameters. Despite lakes providing numerous services to humans in terms of provision of drinking water, irrigation, hydroelectric power generation and recreation among others, the society have usually underestimated the value of these resources. Therefore, they have been neglected to the extent that some lake ecosystems are collapsing out of eutrophication and continuous anthropogenic encroachments. Efforts should be intensified towards developing a vivid understanding of these fragile ecosystems in order to alleviate further destruction (Stevenson & Pan, 1999). Moss (1994), laments that the current research in limnology and hydrobiology has mostly ignored saline lakes. These are lakes characterized by high salinity and alkalinity levels in their ecosystems, for instance the Kenyan alkaline-saline lakes such as Lake Simbi and Magadi. Their simple environments usually experience changes induced by rainfall and other abiotic factors which consequently can modify their phytoplankton ecology. Saline-alkaline lakes such as Lake Simbi are part of the world's "flamingo lakes" which hosts massive bird population all throughout the year or during some seasons and therefore conservation of these lakes is vital for sustaining the vast bird population. These birds are known to migrate to these lakes in large numbers when the phytoplankton structure of the lakes becomes majorly comprised of *Arthrospira fusiformis* which forms their main source of food. The International Union for Conservation of Nature (2010) listed the lesser flamingo (*Phoeniconaias minor*) in their red list indicating that the population of these birds loved by tourists all round the world are diminishing and therefore considered "near threatened" species.

This chapter reviews related literature available on the ecological integrity aspects such as water quality, phytoplankton dynamics and their interactions with physico-chemical variables and nutrients in alkaline-saline lake ecosystems; trophic state index and the habitat quality index. It also explores the features of alkaline-saline lakes and their connection to the flamingo bird populations adored by tourists. This is all aimed at portraying the significance of this study while positioning it within the present state of knowledge in the field and identifying the research gap that needs to be bridged.

2.2 The nature of East African Alkaline-Saline Lakes

2.2.1 Formation and characteristics

The Great Rift Valley in Eastern Africa can be seen as a system comprising of chains of alkaline-saline lakes, hosting more than ten lakes in the Kenyan portion alone. The Great Rift Valley spans from Ethiopia through Kenya all the way to Tanzania. It is a phenomenon of wonder which as Ward (2015) describes it, a beautiful semi-desert scenery consisting of shallow lakes on the plains lined by the ridges. The Rift Valley is said to have been formed during the Cenozoic era due to the continuous action of tectonic and volcanic forces in the region. According to Pecoraino, D'Alessandro and Inguaggiato (2015) the worldwide distribution of these lakes indicates their connection to volcanism. These lakes are known as alkaline-saline lakes or simply as soda lakes since they are characterized by high salinity and high alkalinity levels and going by the definition of Deocampo and Jones (2014), they are lakes having dissolved sodium and carbonate ions as the main ions originating from three processes; rainwater, weathering and hydrothermal fluids. Salinization in these lakes is a consequence of natural variation in water balance which is basically sustained by the evaporation rates (Meybeck, 1995).

As for the formation of the numerous alkaline lakes, Richardson & Richardson (1972) hypothesized that about 9,200 years ago in the Rift Valley, there existed a great lake which receded and shrank to a smaller one by about 3,000 years ago and therefore the current small-sized alkaline-saline lakes might have resulted from the remains of this lake. According to Grant (2004) the alkaline-saline lakes in Kenyan are Simbi, Nakuru, Magadi, Elementaita, Bogoria. Grant, Jones and Mwetha (1990) describe these lakes as environments of high alkalinity and stable pH values greater than 11.5 caused by carbonate minerals. He further stated that the high alkalinity results when Na+ trachyte is weathered from the nearby volcanic highlands and carried by the run-off water into these lakes. This is reinforced by the study by Mills and Sims (1995) that suggest that this alkalinity is caused by the weathering of basaltic minerals releasing Na^{+} and Ca^{2+} ions.

Since alkaline-saline lakes are closed, meaning lacking water outlets, (Rosen, 1994) they utilize evaporation as their outlets, and therefore during dry seasons with high evaporation rates, the soda ash remains after evaporation. This is then mined and used as a raw material in the manufacturing of toothpaste, glass and sometimes used as food additive in some communities (Nielsen & Dahi, 1995). Soda ash, chemically expressed as Na2CO3·NaHCO3·2H2O and known as trona is referred to as “magadi” by the local communities is formed when there exists a condition characterized by low levels of Mg2+ and Ca^{2+} and high levels of Na^{+} (Grant, 2004).

The status of a lake at a given time is responsible for the various processes that lead to the formation of carbonates of Mg^{2+} and Ca^{2+} (Deocampo, 2010; Hargrave, Hicks & Scholz, 2014). Previous studies (e.g Eugster & Hardie 1978; Eugster & Jones, 1979) didn’t recognize hydrothermal as one of the processed contributing the solutes in soda lakes but recent studies (e.g. Earman, Phillips & McPherson, 2005; Pecoraino *et al.*, 2015) have documented that thermal fluids can possibly cause the occurrence of solutes in these lakes. Lugonzo (2014) documents that the Rift Valley has an arid and semi-arid climate characterized by Na^{+} trachyte produced from the lava resulting from volcanism. According to Demenocal (1995), about 2.5 million years ago, the commencement of glacial cycles in the Northern Hemisphere made African climate to become drier. And Trauth (2005) argues that it is from this period that this region began experiencing interchanging periods of humidness and drought.

The numerous saline lakes found in the Rift Valley have pH values of above 10.5 with different salinity levels with some lakes such as Lake Magadi described as hypersaline. The deeper alkaline-saline lakes are characterized by high stable conductivity of about 70,000 µS cm–1 while shallow ones the conductivity levels varying from 11,000 – 160,000 µS cm–1 attributed to by the fluctuation of the water levels (Harper *et al.,* 2003; Verschuren *et al.*, 2004; Schagerl & Oduor, 2008). The overall geochemistry of soda lakes is mainly influenced by the groundwater conditions, atmosphere and chemical weathering (Deocampo and Jones 2014). Alkaline-saline lakes represent a significant unique kind of lakes since they are offer exceptional biological environments that have social, economic effects as well as impacts on the water resources (Deocampo & Renault 2016).

2.2.2 Biological productivity and diversity of alkaline-saline lakes

According to Jones and Grant (1999), biodiversity of alkaline-saline lakes varies depending on the salinity levels. Vareschi and Jacobs (1985) while asserting the significance of phytoplankton studies in alkaline-saline water bodies, notes that these water bodies not only support massive bird population but also large algal community upon which the birds feed on. The rate of primary production of alkaline-saline lakes in East Africa is one of the highest recorded among three continents of North America, Africa and Australia (Hammer, 1981). This indicates the biological richness of these ecosystems. Earlier studies (Talling *et al.*, 1973; Melack & Kilham, 1974) had also established the prominence of primary production in these ecosystems and further stated that it is mainly composed of *A. fusiformis*. The high primary production in alkaline-saline lakes can also be attributed to the presence of decreased heterotroph density which means the organic substances remain locked in the sediment layers of the lake waters (Renaut & Tiercelin, 1994; Deocampo & Jones, 2014).

Recent studies (Schagerl & Oduor, 2008; Kaggwa, Burian, Oduor & Schagerl, 2012), indicate that the diversity in alkaline-saline lakes comprises of cyanobacterial (Synechococcus, A. fusiformis and Anabaenopsis) and diatoms (Anomoeoneis, Nitzschia and Navicula) species. This confirmed the earlier studies by Vareschi (1982) who discovered cyanobacteria species and study by De Deckker (1988) who found diatoms in these alkaline-saline lakes.

2.2.3 Alkaline-Saline Lakes and the Flamingo Connection

Flamingos are birds which are known for their distinct thin legs and long necks. Studies by Bildstein *et al* (1993) established that flamingos' ancestry could be traced to microphagous, bird colonies that lived in hypersaline lakes within the tropics. The Phoenicopteridae family of birds around the globe is currently composed of six species of flamingos with two species inhabiting East Africa, namely lesser flamingo (*Phoeniconaias minor*) and greater flamingo (*Phoenicopterus ruber roseus*). Of the six species currently in the world, the lesser flamingos are the most abundant in terms of population with an approximated size ranging from two to three million (Childress, Nagy & Hughes, 2008; Birdlife International, 2012). They are the also the most loved flamingo birds by tourists, because of the characteristic pink feathers with beaks dark red in color (Stevenson & Fanshawe, 2002). Despite being the most abundant in the world, in East Africa the IUCN (2010) listed as these flamingos as "near-threatened" after studies by Nature Kenya, NMK & KWS (2010) indicated that their population had declined to about 1,200,000 from the previous 2,000,000 observed by Brown (1973) in 1950s.

The lesser flamingos always flock in multitudes in the saline-alkaline lakes in Kenya where they feed on cyanobacteria known as *Arthrospira fusiformis* which forms their major food supply (Kyalo, 2012). This agrees with Tuite (1979) observation that whenever there is massive biomass of *A. fusiformis* in the saline lakes, there is always a high flamingo population too. Vareschi (1978) found that in the Lakes Nakuru and Elementaita, the lesser flamingos also consume other species of cyanobacteria (Tuite, 1981) such as the diatoms as alternative sources of food when the levels of *Arthrospira fusiformis* are totally depleted. Further studies by Kyalo (2012) found that since the lesser flamingos are filter feeders, they can also feed on micro-organisms of filterable size including the protozoans but not the ones with a diameter range of two to six microns because they are too minute hence unfilterable.

Saline-alkaline lakes are havens for these migratory birds, and Tuite (1981) explains that these saline lakes would sometimes experience unexpected crush in terms of the availability of *A. fusiformis* which makes the flamingos to migrate from one saline lake to another looking for better environments with sufficient food. According to Ridley, Moss and Percy (1955) and Brown (1973), the lesser flamingo gathers and feed on the filterable phytoplankton found within the water surface using their bills by swaying their heads sideways. They possess web like feet that help them shake up the phytoplankton trapped on the bed of less deep waters (Jenkin, 1957).

2.3 Water Quality Parameters

2.3.1 Physico-Chemical Parameters

The physico-chemical parameters are key features of water quality of any water body and are crucial in determining the capability of a water body to maintain environmental values (ANZECC & ARMCANZ, 1994b). Therefore, they form vital tools for the assessment of the water quality, and hence planning and conservation (Ogendi, 2017). This is because they are indicative of the prevailing conditions of the lake in terms of nutrients levels and the state of the trophic structure. ANZECC AND ARMCANZ (2000) showed that alterations of the water quality may consequently alter the ecological functioning and structure of an ecosystem. This alteration may interfere with the health of the water ecosystems resulting into alterations in species biodiversity. Besides water quality, water levels also affect the ecological dynamics of a water body and this is echoed by the USEPA (1995) which found that the density and productivity of the phytoplankton declines when the level of water rises in water body because it opens more spaces which are taken up by the zooplankton that feeds on the phytoplankton. Adhoni, Shivasharan and Kaliwal (2015) shows that the production, abundance and species richness of phytoplankton is usually influenced by the interaction of the various physico-chemical variables such as temperature, dissolved oxygen, light, conductivity, turbidity, alkalinity and Total Dissolved Solids.

Temperature

Temperature is an environmental variable that refers to the degree of hotness or the coldness, measured in degree Celsius (ºC). In aquatic ecosystems, the variation of temperature is caused by the regular variation of temperature of the moving wind, the intensity of solar radiation and diurnal length (David, Lam, William & Schertzer, 1999). Jørgensen (1980) study showed how solar intensity causes thermal stratification where the top water surface becomes heated up as the bottom is left in its cold state. Temperature is one of the pivotal parameters in marine environments since according to Ibelings *et al.* (2011), it controls several chemical, biological and physical processes. James and Vallarino (1969) observe that temperature affects the concentration of oxygen and other dissolved gases in water hence controlling its chemistry. This is in line with the Vant Hoff rule that states that as temperature increases it leads to a corresponding increase in the rate of chemical processes taking place. The USEPA (2013) indicates that the toxicity of ammonia increases with increase in temperature.

Since temperature affects the photosynthetic rates, it influences the cycle of oxygen and carbon dioxide in the water column and hence the algal reproduction (Robarts & Zohary, 1987; Salmaso, Buzzi, Garibaldi, Morabito and Simona, 2012). In addition, temperature directly or indirectly affects the behavior and interaction of other living organisms in the water ecosystems in terms of growth, development and reproduction (Kingsolver, 2009). Studies conducted on river systems (Wałkuska & Wilczek, 2010; Forster, Hirst & Atkinson, 2012) indicate that some organism can't survive in temperatures above 30ºC.

Dissolved Oxygen (DO)

Dissolved oxygen refers to the amount of free oxygen concentration in the water, measured in mgL^{-1} (Biddle, 2008). It's limnologically essential as it is needed for respiration by both plants and animals and for decomposition of organic substances by anaerobes which aids in nutrient cycling in the water column (Indabawa, 2010). Medudhula and Samath (2012) reports that the amount of dissolved oxygen becomes limiting in the aquatic ecosystems when there exists a huge accumulation of organic matter at the bottom of the lake since the available oxygen is depleted by the micro-organism undertaking decomposition.

DO concentrations in aquatic ecosystems depend on the rate of decomposition, photosynthesis and respiration (Manuel *et al.,* 2007). Organism desire optimum DO concentration of about 5 – 9 mgL^{-1} hence concentration out of this range could be unfavorable to their survival (USEPA, 1992). During the day, the oxygen levels are high because of photosynthesis but declines to low levels during the night because of the respiration (USEPA, 2000b; Hunt & Christiansen, 2000). The oxygen level required by microbes for decomposition is called Biological oxygen demand (BOD). This BOD affects DO by the relationship indicated by Thirupathacah (2012), that when one variable (BOD) rises in the water column, the other variable (DO) falls and vice versa. DO concentration in aquatic ecosystems is a consequence of the biological processes taking place in the water column and therefore one of the most important parameter for water quality monitoring.

pH

The pH describes the alkalinity or acidity of a solution. It’s expressed as a negative logarithm of the concentration of hydrogen ions (H^+) in water (Ogendi, 2017). The pH scale runs from 0-14, where pH value of below 7 is considered acidic, above 7 is considered alkaline/basic and value at exactly 7 is considered neutral (water) (WordWeb dictionaries online, 2018). The optimum range of pH recommended for all surface waters should be between 6.5 and 8.5 (USEPA, 1992). The pH as a water quality variable is significant because, it governs the solubility and accessibility of nutrients (Rao, 1989).The process of photosynthesis affects the pH variation in the water because the carbon dioxide dissolved in the water has acidizing effect when it reacts with water leading to the formation of carbonic acid but as plants absorb dissolved carbon dioxide from the water during photosynthesis the water becomes more basic. This explains why Abubakar (2017) study observed high pH values during the day and low pH values during the night, because during the day the photosynthetic process rids the water of carbon hence reducing acidity while in the night the accumulation of carbon dioxide in absence of photosynthesis makes pH values decline with the water leaning towards higher acidity. Apart from photosynthesis, WHO (1996) list the other factors that may affect the variation of pH in the water as mining, industrial pollution, municipal waste dumping and indirectly through agricultural run-off. Marine life is sensitive to pH changes and this is pointed by the USEPA (2005) that waters of pH values below 6 and above 9 are beyond the optimum range required by aquatic life thus toxic. This is further reinforced by the USEPA (1992) study emphasizing that the survival of fish requires biodiversity with optimum pH of between 6 and 8. According to the study by Guinotte & Fabry (2008), extreme acidity of water masses causes mortality of living organisms. This finding is enhanced by Abubakar (2017) study that indicated that pH alterations in the water amplify the toxicity of trace minerals which in turn have detrimental effects on the biological diversity.

Conductivity and salinity

Conductivity is defined by George and Schroeder (1987) as the capacity of a solution to conduct electrical current, while salinity measures the concentration of salts in the water. It is measured in micro-Siemens per centimeter denoted as (μScm^{-1}). These two water quality parameters are associated with the ions in TDS in the water such that an increase in TDS corresponds to increasing ionic concentration in the water and hence conductivity. The transmission of electric current in water occurs through the ions contained in it and so when the concentration of ions rises, the water conductivity also rises (Fatoki & Gogivane, 2003). Their study asserts that the major significance of conductivity in water quality assessments and monitoring is that it can help in quantifying the amount of dissolved solids in the water of interest. This is because conductivity originates from dissolved solids such as from run-off water, waste and minerals.

Measuring water conductivity of natural water can provide a clear picture of the concentration of nutrients that dissolved in water such as nitrates and phosphates (Fatoki & Gogivane, 2003). In their assessment they state that the conductivity of natural water ranges between 20- 1500 µs/cm. The connection between salinity and conductivity arises from the fact that, salinity describes the measure of dissolved salts in the water and these dissolved salts are made up of ions which transmit electrical current. Some organisms are tolerant to saline environments but increased salinity beyond the optimum may have deleterious effect on them.

Total Dissolved Solids (TDS)

This is a parameter of water quality as a measure of the content of solids suspended in the water column and it's measured in mgL^{-1}. These solids constitute both organic and inorganic solid substances liable upon its level of volatility. TDS ends up in the water bodies from both natural and human sources, and include organic material and mineral salts (WHO, 1996). In aquatic ecosystems, decreased solubility of gases is one of consequences of high TDS in water (Garg, Saksena & Rao (2006). Their study further determined that osmoregulation is also affected by such water as is the usefulness of the water too. In explaining the significance of this parameter, Rao and Kumar (2003) implicates TDS in regulation of the biological and mechanical processes involved during the treatment of waste water. Several studies (WHO, 1996; USEPA, 2007) have established that TDS level beyond 1200 mgL^{-1} is related to the presence of toxic contaminants in the water body.

Alkalinity

Alkalinity is basically the condition describing a solution with a pH higher than 7. As a water quality parameter, it is a measure of the buffering ability. As a buffer, it shields the water body from the variations in pH by removing the extra hydrogen ions (H^+) (Abubakar, 2017) Alkalinity originates from limestone, chemically known as Calcium Carbonate ($CaCO_3$) which neutralizes the acidity released from other deposits. Manahan (1993) lists bicarbonate ion, carbonate ion and hydroxyl ion as the key contributors of alkalinity in water.

Abubakar (2017) states that some aquatic organisms such as the *Chlophyta spp* require protection from erratic pH variations and so they would thrive well in alkaline waters. Water hardness is associated with alkalinity when it has carbonates of alkali metals such as sodium. Soft water has low alkalinity hence poor buffering capability.

Turbidity

Turbidity is a water quality parameter that measures the cloudiness of water and is expressed in terms of Nephelometric Turbidity Units (NTU). It is a measure of the quantity of floating matter in the water (Rasolofomanana, 2009). According to Rasolofomanana (2009), the sources of turbidity in the water are excessive algal growth, soil deposits from run-off and decomposition of organic substances. He further explained that the high content of floating substances on the water increases the quantity of light scattered hence large values of NTU. Phytoplankton require light for photosynthesis and therefore turbidity affects their composition and species richness (Flöder, Urabe & Kawabata, 2002), because high turbidity prevents light penetration in the water column.

2.3.2 Nutrients Characteristics

Nutrients refer to substances required by living organisms for growth and development. In aquatic ecosystems, the two most vital nutrients are the derivatives of nitrogen and phosphorus. Adhoni *et al.* (2015) concurs through his conclusion that the plant life in waters requires soluble reactive phosphate and nitrogen derivatives (NO_2^-, NO_3^- and NH_4^+) for their metabolic processes. However, they are only required at optimum levels since, according to (Adhoni *et al.,* 2015) excess concentration of these nutrients could result in excessive water enrichment and hence blooms. This makes the biogeochemical cycles of nitrogen and phosphorus crucial for the aquatic ecosystems. Naturally, nitrogen is the most abundant gas in the planet as opposed to phosphorus, and the ratio of N to P is 1:16.

Nutrients in water masses originate from the natural processes or through anthropogenic activities. The mechanism through which the nutrients end up in aquatic ecosystems is by leaching or through water run-off (Idu, 2015). This concurs with a previous study by Clark, Haverkamp and Chapman (1985) that established that nutrients in water masses originates from within these ecosystems from the different natural processes or outside these ecosystems from the human-influenced sources and the air. Guldin (1989) opines that apart from the sources emanating from human activities such as mining, industrial operations and poor agricultural practices, nutrients ending up in water masses can occur from the natural processes of the atmosphere and fixation by micro-organisms and lightning. The combination of these processes causes excessive nutrient enrichment of the water bodies which consequently alters the aquatic community (Dupont, 1992).

Nitrates are produced naturally from the nitrogen cycle when the organic materials are broken down by bacteria (Adakole, Abolude & Balarabe, 2001). Of all forms of nitrogen, nitrates are the most readily available in aquatic ecosystems. Kigamba (2005) stated that the lakes in Africa contain high concentration of nitrates percolating from high agricultural use of nitrogenous fertilizers. Other potential sources of nitrates in aquatic ecosystems according to the Department of Environmental Quality-Oregon (2015) include the waste stream emanating from industries and the sewage systems from the cities. Since eutrophication and death of aquatic life has been largely attributed to the presence of high nitrate concentration in water bodies, and continuous monitoring of its level in the water bodies is pivotal for effective control of water pollution (Jaji, Bamgbose & Arowlo, 2007). Study by Boesch, Brinsfield, and Magnien, (2001) observes that the possible impacts of nitrates in saline waters are globally attracting increased interest.

According to the USEPA (1995) massive plant growth occurring in the water bodies due to nutrient enrichment lowers the availability of dissolved oxygen in the water leading to the suffocation and even death of other aquatic life. Ogendi (2017) reported that the breakdown of organic materials of dead plant in the water also could reduce the level of dissolved oxygen making it unfavorable for other living organisms.

Bade (2004) mentioned that even though phosphorus is needed in little concentration by aquatic plants, it is equally significant nutrient for growth and development. Apart from nitrogen which enters water bodies through leaching in solution form, phosphorus enters the water bodies by sticking on the surface of sediments which are then transported by floods (Pionke, Gburek, Sharpley & Schnabel, 1995). It is also important to note that phosphorus forms part of the biogeochemical cycles that support life in the planet. Dinnes *et al*. (2002) postulated that it takes a combination of both human and environmental influences for nutrients (phosphates and nitrates) to end up in aquatic environments from their original sources. Phosphorus has always been associated to eutrophication but conflicting findings by George and Schroeder (1987) indicates that restrict the growth of aquatic plants. Rani, Gupta and Srivastava (2004) asserts that analysis of phosphates in the water is of significance when studying aquatic environments.

In aquatic ecosystems, the concentration of phosphates and nitrates in the water has a profound impact on the biodiversity and therefore it's important to analyze them together with other nutrients such as silica and chlorides for a comprehensive picture of the status of the water quality.

2.3.3 Relationship between Physico-Chemical Parameters and Phytoplankton Community Structure

Roger (2010) describes phytoplankton as microscopic organisms in aquatic ecosystems that appear as colored maculation because of their characteristics chlorophyll. And Reynolds (2005) shows that life in the water bodies is supported by the phytoplankton. The phytoplankton provides a pathway for energy flow in aquatic ecosystems through primary production which constitute the foundation for the food chain (Dhanam, Sathya and Elayaraj, 2016). The primary productivity according to Grace and Michael (2010) drives the secondary productivity. Wetzel (2001) established that the primary producers (phytoplankton) sustain the balance between biotic and abiotic components of the environment.

According to the Schagerl and Odour (2008), the physico-chemical parameters that are significant in studying phytoplankton ecology are light, conductivity and the nutrients (N, P). Lake environments like any other ecosystems depend on different abiotic variables such as pH, nutrients, temperature and light intensity and duration. The lake ecosystems usually adapt to the changes in these variables and this adaptation results in changes to the phytoplankton community. This concurs with assorted studies (Defew, Perkins & Paterson, 2004; Murrel & Lores, 2004; Marshall, Lacouture, Buchanan & Johnson , 2006) that indicated that the phytoplankton dynamics are governed by physico-chemical parameters; salinity, nutrients, temperature and light. Investigating the dynamics between physico-chemical variables and the phytoplankton structure is therefore vital for establishing the workings of lake ecosystems (Rajdeep, Pratihary, Gauns & Naqvi, 2006). In addition, he emphasizes the significant role of phytoplankton in nutrient flow in lake ecosystems.

A study by Vareschi (1978) observed that species diversity in alkaline-saline lakes is caused by the wide variations in the physico-chemical variables. He further discovered a link between decreasing conductivity and dominance of *A. fusiformis* in these waters. Alkaline-saline lakes possess high conductivity resulting from their high salinity and Ballot *et al*. (2005) confirmed previous studies that found a connection between the dominance of cyanobacteria species and greater conductivity. Later study by Krienitz, DaDheech and Kotut (2013) in Lake Oloidien also indicated an enormous increase in cyanobacteria as the waters become more saline. Oduor and Schagerl (2007) while investigating the relationship between the water quality parameters and the phytoplankton of three Kenyan lakes of Nakuru, Elementaita and Bogoria concluded that species variety goes down when the variables of temperature and conductivity rises. This is similar to an assessment by Fogg

(1975) that found species richness is seasonally influenced by temperature and nutrient levels. According to Margalef (1978), the density of phytoplankton has a direct correlation with the nutrient levels in the water column. And Cohen (2003) opines that the alkaline-saline lakes' nutrients levels govern the mechanisms of primary production. Different studies (Melack and Kilham 1974; Vareschi 1982; Hecky and Kilham 1988; Ballot *et al*. 2005) on saline-alkaline lakes in East Africa have identified phytoplankton species from the algal classes of Chlorophyceae, Cyanobacteria and Bacillariophyceae. They point that this diversity is a consequence of the interactions between the phytoplankton and the environmental conditions. From the above studies, it's clearly evident that the environmental dynamics have significant control on phytoplankton with temperature required for metabolism; light for photosynthetic productivity; salinity governs the osmoregulation function; and nutrient levels governing growth.

2.4 Trophic State Index

Ivanković, Velagić and Knezović (2018) define trophic state to be the total biomass presence in water ecosystems in a particular place and at a particular time. The trophic state is a response to the elevated nutrient concentration levels in a water body (Naumann, 1929). Nutrient levels variations in a water body may be caused by planktonic grazing, depth of mixing and seasons (Prasad, 2012). Trophic state index therefore measures the biological productivity in aquatic ecosystems (Florida Lakewatch, 2018), and therefore a significant indicator of the ecological integrity. There are several methods for measuring the trophic status of aquatic ecosystems, but the standard most commonly utilized model is the one developed by (Carlson 1977) which was initially tested in the lakes in North America. The Carlson's TSI has been applied worldwide as a useful tool in assessing the general ecological health of a lake (Prasad, 2012). The Florida Lakewatch (2018) indicates that the TSI values are crucial in classification of water bodies which then enables the management authorities to prioritize them for preservation, conservation and/or restoration efforts in order to maintain their ecological integrity. Through TSI, impaired water bodies can easily be identified, and appropriate conservatory strategies developed for its conservation and management.

The trophic state index introduced by Carlson (1977) has globally been accepted to assess the biological health of aquatic ecosystems, because it is simple and uses only a few parameters of as opposed to other complex multi-parameter models (Melcher, 2013). The Carlson's TSI is calculated based on the Secchi depth, chlorophyll-a level, and total phosphorus level. Ivanković *et al.* (2018) shows that since these three parameters affect the phytoplankton biomass, they are therefore crucial in the establishing the trophic status of a water body. Their study states that the use of concentration of chlorophyll is the most ideal because it could correctly and precisely predict the phytoplankton biomass. Carlson (1977) found that when investigating the trophic state for the summer, the total phosphorus is favored since it is an accurate index of chlorophyll. He further states that the Secchi depth should only be employed if the use of other methods is not viable. According to Ivanković *et al.* (2018), the benefit of TSI is its ability to work out the relationship between the parameters which then enables the identification of particular variables in aquatic ecosystem that limit phytoplankton growth. According to the Florida Lakewatch (2018), oligotrophic aquatic ecosystems are characterized by lowest ecological productivity, mesotrophic possess moderate ecological productivity, eutrophic have high level and hypereutrophic highest level of productivity.

2.5 Lake Habitat Quality

An ecosystem is a self-governing system involving the interplay of both abiotic and biotic elements. The integrity of an ecosystem is achieved when the natural composition, structure and functioning of the ecosystem is not impaired. It compromises of various organisms and their habitats although habitats and ecosystems are sometimes used interchangeably. According to the National Geographic (2018), habitat is described as a locality in where fauna or flora resides and grows in. The capability of a habitat to offer appropriate environments for single species and populations as well is described as habitat quality (Hall, Krausman & Morrison, 1997). This implies that habitat is the primary source of biodiversity and therefore should be protected.

The integrity of an ecosystem (or habitat) can be compromised by the natural processes of the environment and anthropogenic interventions. A study by Nellemann, Kullerud, Vistnes and Forbes (2001) observes that the closeness of various land uses to any habitat significantly influences its quality. Lakes are lentic ecosystems that provide essential resources and ecological services to man. The quantity and quality of these benefits are currently facing constant threat from climate change and unprecedented pressures from anthropogenic exploitation. The risk of losses that could potentially result from habitat modification and destruction has created the need for constant ecological monitoring of the quality of the habitats of these systems as well as the degree of anthropogenic perturbations in them. For this reason, Lake Ecosystem integrity assessments have become globally prevalent. Several regions around the globe have come up with various tools for assessing the lake habitat quality. One such tool is Lake Habitat Survey, developed by the University of Dundee (Rowan, 2005) based on the Water Framework Directive (WFD), established by the

European Directive 2000/60/EC as guiding outline for water policy (European Commission, 2000). In introducing the concept of ecological status, WFD recognized that hydro-morphological alterations have potential impacts on the composition and abundance of biotic communities in surface waters. The LHS is considered a standard tool by the CEN (2011) for use in assessing physical habitat of lakes in the European Union. This tool has been widely used in countries found in European Union (Radulović, Laketić, Popović & Teodorović, 2010) and Australia (EPA Victoria, 2010), but has had limited use in Africa with recent trials conducted in few reservoirs in Zimbabwe (Dalu, Clegg & Nhiwatiwa, 2013). LHS method was designed to describe the shoreline of the lake habitat in terms of the vegetation cover, macrophytes assemblages, littoral substrate and the human pressures occurring along it. These elements are important drivers responsible for regulating the ecological characteristics of an aquatic ecosystem (Stella, Rodríguez-González, Dufour & Bendix, 2013). Kaufmann *et al*. (2014a) observes that the methodology of LHS was formulated based on the methods from US Environmental Protection Agency's Environmental Monitoring Assessment Program (EMAP) and the River Habitat Survey (RHS) which had been earlier developed in the UK.

The detailed description of the LHS as outlined in Kaufmann *et al.* (2014a,c) states that the geomorphology is investigated by having ten predetermined Hab-plots set up around the entire length of the lake and using them to examine the littoral, shore and riparian zones of the lake. In addition, the information is collected on the human pressures and development modifications happening throughout the lake. Data is mainly collected through observation of the different characteristics occurring in the Hab-plots. Since LHS provides such robust and integrative protocols for scanning the habitat quality and scale of human alterations in water bodies of conservation value, it can be a valuable tool for watershed management (Latinopoulos, Ntislidou & Kagalou, 2016). LHS can help in monitoring ecological status of an ecosystem with the aim of informing an evidence-based decision-making for appropriate and effective conservation and management by relevant stakeholders.

2.6 Existing Limnological Studies in Lake Simbi and the Research Gap

From the review of the related literature it suffices to conclude that the Lake Simbi just like the other African saline lakes is ones of the least studied ecosystems (Williams, 1981). This is notwithstanding their recognition as the most productive environments around the globe (Vareschi, Melack & Kilham, 1981).The initial limnological investigation of Lake Simbi was through a study by Melack (1976) which investigated the photosynthetic capacity of a single phytoplankton species, *Spirulina platensis* (Cyanophyta). His study revealed high photosynthesis rates in the upper water surface of the lake. The studies that followed this investigated the environmental dynamics of the planktonic communities in the Lake Simbi ecosystem (Melack, 1979; Tuite, 1981; Finlay *et al.,* 1987). The study by Ochumba and Kibaara (1988) on the thermal stratification aspects of Lake Simbi revealed that a robust chemocline exists at the four meters depth of the upper water column while a cool layer exists at a depth of 2.5 meters on top of a little warmer layer. Their study concluded that the decreased temperature levels observed on the higher depths of the water are due to the water flowing into the lake form rainfall and rivers, and possibly the cold winds of the night.

The latest study carried out by Ballot *et al.* (2005) focused on the algal cyanobacteria and its toxins. Their study discovered that the cyanobacteria toxins of microcystins and anatoxin-a comes from the *Arthrospira fusiformis* which is the most abundant algal species in lake

Simbi. They attributed the dominance of the cyanobacteria in Lake Simbi to the properties of salinity and alkalinity which rises from the lake's geochemistry.

From the foregoing, it's evident that the limnological and ecological information on Lake Simbi is limited declining populations of the lesser flamingos and other bird species. This constitutes a research gap that requires bridging because according to studies by Schagerl and Oduor (2008) and McCulloch *et al.* (2008) the saline-alkaline lakes in the tropics (Lake Simbi included) are a "unique habitat" characterized by an ever-changing algal community and constant varying environmental variables. Therefore my study, while building on the previous studies, is aimed at bridging this fundamental gap by investigating the ecological integrity of the Lake Simbi based on the prevailing phytoplankton dynamics, water quality, lake habitat quality and finally establish the trophic state of the lake.

2.7 Conceptual Framework

The ecological integrity of water bodies depend on the water quality, the planktonic dynamics, habitat quality and the trophic state, and the interplay between these variables with the physical environment.

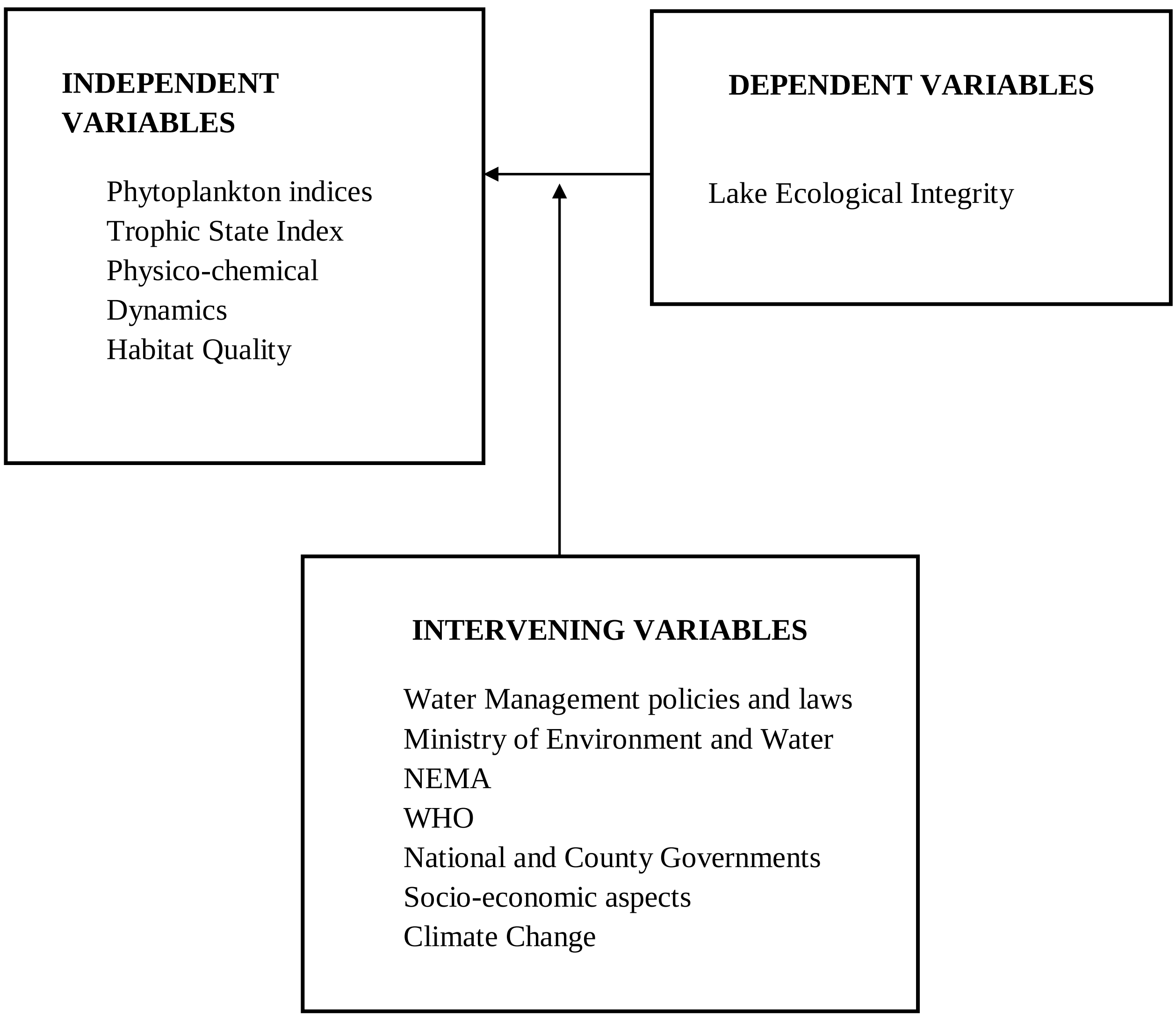

Figure 1: Conceptual framework showing relationships between variables that influence the ecological health and integrity of an aquatic ecosystem.

CHAPTER THREE

MATERIALS AND METHODS

3.1 Introduction

This chapter highlights the study area and the sampling sites, the research design, the sampling protocol, the mechanism for maintaining the integrity of the samples for the study, the validity of instruments used in profiling of the physico-chemical variables. In addition, it highlights the analytical procedures for the different nutrients of interest in Lake Simbi (TN, (NO_3-N), (NO_2-N), (NH_3-N), TP, SRP and SiO_2), the phytoplankton composition and abundance, the analysis of the trophic state and the Lake Habitat Survey procedure. Lastly, it describes the data analysis procedures and tools.

3.2 Description of the Study Area

Lake Simbi Nyaima (Fig. 2) below is small alkaline-saline lake in Kenya that supports massive bird populations especially the lesser flamingos which makes it one of the valuable tourist attraction sites in Western Kenya. Due to this, it is gazetted as a national bird sanctuary placed under the management of Kenya Wildlife Service (KWS, 2018). The lake is situated at an altitude of 1142 meters above sea level and lies between 0°22'5"N and 34°37'47"E on the Nyanzan Gulf about 1000 meters from the shores of L. Victoria. The morphometric characteristics of Lake Simbi are summarized in Table 1. The lake is located close to Kendu Bay, a town center in Homabay County of Kenya, in a semi-arid region receiving an average rainfall of between 500 and 1700 millimeters per year with temperature range between 18 degrees Celsius and 31 degrees Celsius. As a volcanic endorheic lake, it lacks any known inlet and outlet. The water level in the lake is replenished majorly by the rainfall and underground flow from its hydrology. Agriculture is the main economic activity of the locals besides sand and salt *(bala)* mining from the lake (KWS, 2018). The lake doesn't support fishing since the extreme alkaline-saline conditions coupled with hypoxic conditions of its waters are not favorable for the survival of fish populations. Nonetheless, this lake is important for ecotourism, and provides scientific, cultural, religious and educational benefits.

Table 1: Morphometric Characteristics of Lake Simbi

Morphometric Characteristics of Lake Simbi	
GPS coordinates	0°22'5''N and 34°37'47''E
Altitude (m)	1142
Maximum depth (m)	27.7
Average depth (m)	17
Surface area (km^2)	0.301
Shoreline perimeter (km)	2.097
Volume ($\times 10^6$ m^3)	5.124
Water body type	Volcanic (Endorheic) Lake

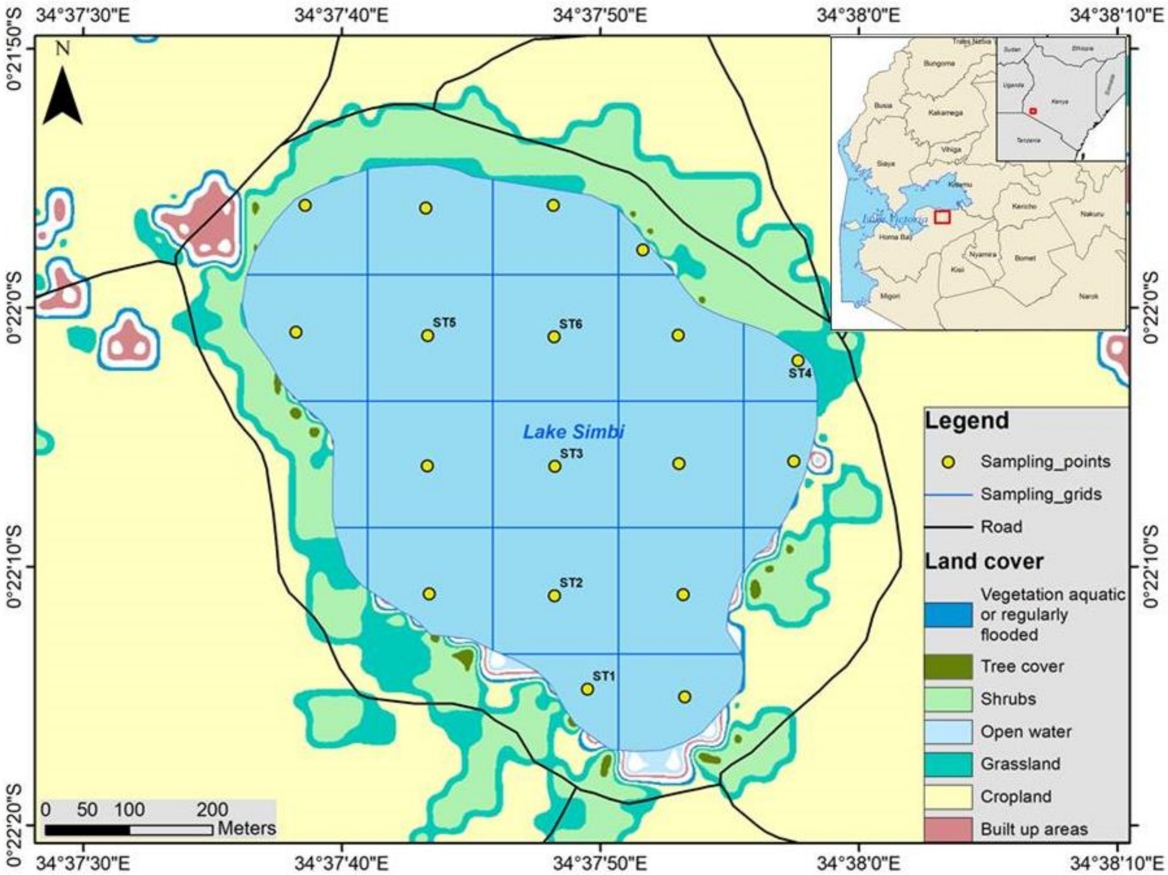

Figure 2: Lake Simbi showing the sampling stations and the land cover of the area (Author, 2019)

3.3 Research Design

The systematic transect sampling design was adopted in this study. This design follows the United States Geological Survey (2018) guidelines which involves randomly choosing the initial sampling point in the Lake Simbi and then the placing the rest sampling points equidistant from each other (Averett & Schroder, 1994). From the foregoing, imaginary grids were drawn over the map of the lake and each of these grids sampled in a linearly fashion whereby six stations in two transects were chosen, that is three sites on vertical transect ST1 (0.37014 S, 34.63204 E), ST2 (0.36973 S, 34.63080 E) and ST3 (0.36820 S, 34.63016 E), stratified according to nearness to the shore or depth, and three sites on horizontal transect ST4 (0.36756 S, 34.63289 E), ST5 (0.36773 S, 34.62727 E) and ST6 (0.36709 S, 34.62967 E) as shown in Fig. 2. These stations were stratified in a way that three stations would appear on shores and the other three offshores. The first sampling station (ST1) was placed near the shore close to the point where the rainfall run-off enters the lake while the rest were placed equidistant from each other. The sampling stations were identified by the Magellan Global Positioning System (GPS) 315 meridian. From each sampling station, apart from the physico-chemical measurements, the information on the weather, day, the GPS details were also be recorded from the Magellan Global Positioning System (GPS) 315 meridian.

3.4 Sampling Protocol

The physico-chemical variables together with the composition and abundance of phytoplankton were investigated on a monthly basis for six months (December 2018- May 2019) at 6 fixed sampling stations in Lake Simbi. The sampling period encompassed both the dry season months (December, January and February) and wet season months (March, April and May). Sampling was always carried out in the morning between 8am to 12 noon.

Sampling protocols followed the standardized procedures as spelt out in APHA (2012). Three measurements of the physico-chemical parameters were taken *in situ* at each sampling station; three integrated water samples taken from depths 1m, 2m and 3m of the water column were taken in 500 ml bottles from each of the three sampling stations to the laboratory for nutrients analysis; and one water sample was collected in 500 ml bottle from each of the three sampling stations to the laboratory for the examination of phytoplankton diversity indices. The following water quality parameters were measured with Hanna Instruments® Multiparameter Probe (YSI Professional Plus model): Temperature (ºC), Dissolved Oxygen concentration (mgL^{-1}), pH, Conductivity (μScm^{-1}), Turbidity (NTU), TDS (mgL^{-1}) and Salinity (ppt). This equipment was calibrated to have a measurement accuracy of 0.0. Hardness and Alkalinity measurements were obtained through laboratory analysis according to the procedure described in APHA (2012) while the Transparency depth was measured at each sampling station with a standard Secchi disc.

Immediately after collection, the three samples for laboratory nutrient analyses were preserved with mercuric oxide while the sample for phytoplankton diversity indices preserved with 2 drops of Lugol's solution. All the water samples were accurately labeled and then stored in a safe cooler box with temperatures maintained at 4°C before they were transferred to Kenya Marine and Fisheries Research Institute (KMFRI) laboratory in Kisumu within one day of collection where they underwent further examination. The Carlson's trophic state index was computed from the Secchi depth, chlorophyll-a level, and total phosphorus parameters of water quality. The Lake Habitat Survey was done in the field through a two-day site surveys of 10 Hab-plots stationed around the perimeter of the lake in late March 2019 based on the procedure described by Rowan *et al.* (2008).

3.5 Validity of the Sampling Equipment and Samples Integrity

The validity of this study was achieved by employing the globally recognized standard protocols (APHA, 2012) which outlines the standard methods and guidelines for the sampling and analysis of water and waste water. For nutrient analysis in the lab, high accuracy level was ensured through strict adherence to the described analytical procedures and reagents in APHA (2012). Before conducting the real sampling, a pilot test was initially conducted with the samples obtained from the Kisii University ponds to enable the supervisors ascertain that the all the sampling equipment were in good condition, correctly calibrated and functional, and that the sampling protocols described in this study are viable and valid.

During the first day of actual sampling in the lake, three identical water samples from first station were randomly picked and examined to demonstrate the duplicability of results. Extra samples for laboratory analyzes were frozen and stored for the entire length of the sampling period to help with the recovery in case of any mishap during analyzes. The integrity of this study samples was achieved by the following measures: instantly preserving the accurately labelled water samples collected for laboratory nutrients analysis using mercuric oxide and the accurately labelled water sample for phytoplankton diversity indices using Lugol's solution, analysis of freshly prepared samples, safe storage in cooler box with temperatures maintained at 4ºC in a dark room. Once transferred to the laboratory, they were safely stored in a deep freezer before they were immediately analysed within 72 hours.

3.6 Nutrients Analyses in the laboratory

3.6.1 Total Phosphorus

The total phosphorus (TP) was determined by first digesting and reducing the forms of phosphorus present in the water into the free orthophosphate form (SRP) using persulphate digestion method described in APHA (2012). After the digestion, the total reduced forms into the SRP formed were analysed using the ascorbic acid method. A standard calibration curve for total phosphorus was made in which the standard solutions underwent the persulphate digestion procedure followed with the soluble reactive phosphorus test procedure explained below.

3.6.2 Soluble Reactive Phosphorus (SRP)

Soluble reactive phosphorus (SRP) was determined using the ascorbic acid method (APHA, 2012) on the filtered samples. All the used glassware were acid-washed with 10% H_2SO_4 at least one day before the commencement of the exercise and double rinsed with distilled water. The Standard solution for the calibration of the standard curve was made by dissolving 5.623 g of potassium hydrogen phosphate (K_2HPO_4) pre-dried salt in the oven for 24 hours at approximately 70°C in 1 litre of distilled water to make a stock solution with a concentration of 1g of PO_4-P L^{-1}. From this stock solution, 10 ml was diluted up to 1 litre using distilled water to give an intermediate solution with a concentration of 10 µg L^{-1}. The intermediate solution was further diluted in a ratio of 1:20 (25 ml in 500 ml-distilled water) to give a working solution with a concentration of 500 µg L^{-1}. The standard series was prepared by taking the following respective volumes of the working solution and diluting to 25 ml with distilled water as shown on the Table 2. Triplicate samples were taken for each concentration.

Table 2: Standard series dilutions for the calibration curve for orthophosphate determinations

Conc. ($\mu g\ L^{-1}$)	0	10	20	50	100	200	400	500
Working solution (ml)	0	0.5	1.0	2.5	5.0	10.0	20.0	25.0
Vol. dist. water (ml)	25.0	24.5	24.0	22.5	20.0	15.0	5.0	0.0

The Ammonium molybdate solution, Sulphuric acid, Ascorbic acid and Potassium-Antimonyltartrate Solution reagents were mixed in 2:5:2:1, ratios respectively. The resulting solution was added to the sample in a ratio of 1:10, e.g. 2.5 ml reagent added to 25 ml of the sample. The prepared sample's absorbance was measured after 15 minutes of adding reagents to the samples at a wavelength of 885 nm with distilled water as a blank. Where the samples were highly turbid, a natural reference sample for turbidity correction was made by using 25 ml of the sample in which only reagents Ammonium molybdate solution and Sulphuric acid were mixed in the same ratio and added to the sample, and its absorbance measured. The concentrations of all the samples were analysed and determined from the standard curve using their respective absorbencies.

3.6.3 Total Nitrogen

Total nitrogen was determined by first carrying out persulphate digestion to convert the nitrogen forms into the nitrate which was further converted to nitrite using cadmium reduction through a column (APHA, 2012). The resulting solution was then tested for nitrite concentration. The total nitrogen was estimated by adding up the concentration of ammonium-nitrogen to the value calculated from this process.

3.6.4 Nitrate-nitrogen (NO_3-N)

Nitrate-nitrogen was determined using the sodium-salycilate method (APHA, 2012) with standard solutions of the nitrate prepared for the standard calibration curve. Fresh sodium salicylate solution was always used in each determination because it spoils quickly. A stock solution was prepared by dissolving 6.067 g of sodium nitrate ($NaNO_3$) in 1 litre of distilled water to make a stock solution of 1000 µg L^{-1}. From this stock solution, a working solution with a concentration of 5 mg NO_3-N L^{-1} was made by taking 5 ml of the stock solution and diluting to 1 litre. A standard series with different concentrations was then made by taking the given concentrations of the working solution and diluting with distilled water as shown in Table 3. Triplicate samples for each concentration were made to make the calibration curve.

Table 3: Standard series dilutions for the calibration curve for nitrate determinations

Conc. (µg L^{-1})	0	0.25	0.5	1	2.5	5
Working soln (ml)	0	1.0	2.0	4.0	10.0	20.0
Vol. dist. Water (ml)	20	19.0	18.0	16.0	10.0	0.0

Four hundred grammes (400 g) of NaOH were dissolved in 1 litre distilled water, together with 50 g K-Na-Tartrate to make this solution. 20 ml of filtered water sample was placed in an evaporation bottle and to this as well as to the standard series; 1 ml of sodium salycilate solution freshly prepared was added. The bottles were then put into the oven and the samples dried at a temperature of 95°C. The resulting residue was dissolved quantitatively by adding 1 ml of conc. H_2SO_4 and the bottles swirled carefully while still warm. Next, 40 ml of distilled water was added and mixed. Finally 7 ml of potassium-sodium hydroxide-tartarate solution was added mixed and the absorbance determined at a wavelength of 420 nm.

Correction turbidity was made (where samples were highly turbid) by taking 20 ml of sample, oven-drying without adding sodium salicylate. The Concentrated H_2SO_4 was added to this dried sample to dissolve the materials after oven drying. 30 ml of water added as well as the 7 ml of the sodium hydroxide-tartarate and the volume made up to 50 ml with distilled water. Absorbance of this sample was read and the value subtracted from the sample values for the colour correction. The blank was double distilled water. The standards were treated in the same way and a standard curve of concentration against the absorbencies was plotted, from which the concentration of analysed samples were determined using their respective absorbencies.

3.6.5 Nitrite-Nitrogen (NO_2-N)

The nitrite-nitrogen determination was carried out using the reaction between sulfanilamide and N-Naphthyl-(1)-ethylendiamine-dihydrochloride which gives an intense pink colour with the nitrite. $NaNO_2$ salt with a molar weight of 69 g was used to make the standard calibration curve for the nitrite determination. A stock solution with a concentration of 1 g NO_2-N L^{-1} was made by dissolving 1.2322 g of $NaNO_2$ in 250 ml distilled water. 5 ml of this stock solution was then diluted to 500 ml with distilled water to make an intermediate solution having a concentration 10 µg L^{-1}. 5 ml of this intermediate solution was diluted into 1 litre with distilled water to give a working solution with a concentration of 50 µg L^{-1}.

A series of standards with determined concentrations were made to determine the calibration curve by diluting this working solution as given in Table 4.

Table 4: Standard series dilutions for the calibration curve for nitrite determinations

Conc. ($\mu g\ L^{-1}$)	0	2	5	10	20	50
Working solution (ml)	0	1	2.5	5	10	25
Vol. dist. Water (ml)	25	24	22.5	20	15	0

Twenty five millilitres (25 ml) of the respective concentrations was taken in triplicate, the test reagents added and the absorbance was read from the spectrophotometer to develop the calibration curve. To 25 ml of the filtered sample, 1 ml of Sulfanilamide solution was added. After 2-8 minutes, 1 ml of N-Naphthyl-(1)-ethylendiamine-dihydrochloride solution was added to this mixture and gently mixed. The solution was then be left standing for 10 minutes after which its absorbance was read from the spectrophotometer at a wavelength of 543 nm against the distilled water blank. The colour complex formed was normally stable for two hours.

3.6.6 Ammonium-Nitrogen (NH_3-N)

This was made by using NH_4Cl solids with a molecular weight of 53.492 g in which the nitrogen proportion of NH_4–N is 1:14 of the weight. 0.955 g NH_4Cl was dissolved in a 250 ml volumetric flask to prepare 250 ml of this solution, giving a stock solution with a concentration of 1g NH_4-N L^{-1}. 10 ml of this stock solution was diluted further to 1 litre with distilled water, giving an intermediate solution with a concentration of 10 $\mu g\ L^{-1}$. From this solution, a working solution was made with a concentration of 250 $\mu g\ L^{-1}$ by taking 25 ml of the intermediate solution and diluting it up to 1 litre. The working volume for calibration curve used is 25 ml. Different concentrations of this standard working solution made to determine the calibration curve by taking the following respective volumes of the working solution and diluting to 25 ml with distilled water as given in Table 5.

Table 5: Standard series dilutions for the calibration curve for ammonia determinations

Conc. ($\mu g\ l^{-1}$)	0	10	20	50	100	250
Working solution (ml)	0	1.0	2.0	5.0	10.0	25.0
Vol. dist. water (ml)	25	24.0	23.0	20.0	15.0	0.0

Triplicate samples for each of the concentrations were used to determine the calibration curve. To 25 ml of the sample, 2.5 ml of Sodium salicylate solution was added, followed immediately by the addition of 2.5 ml of Hypochloride solution. The sample was then placed in a water bath at a temperature of 25 °C in the dark for 90 minutes. Absorbance was then determined at a wavelength of 655 nm with distilled water blank.

3.6.7 Silicates (SiO_2)

Plastic flasks were used in the determination of silicate concentration. Soluble reactive silicates were determined using the ammonium molybdate method on filtered samples (APHA, 2012). Prior to this, a series of standards for SiO_2 concentration were prepared using a 1.00 g SiO_2/ml stock solution (Titrisol, Merck) for the calibration curve.

A working solution with a concentration of 20 µg L^{-1} was made by diluting 20 ml of the stock solution having a concentration of 1.00 g SiO_2 L^{-1} (Titrisol, Merck standard) bought ready-made into 1 litre flask. The following volumes of this working solution were taken and diluted with the respective volumes of distilled water as given in Table 5.

Table 6: Standard series dilutions for the calibration curve for silicate determinations

Conc. SiO_2 (µg L^{-1})	0	0.8	2.0	4.0	8.0	20.0
Working solution.	0	1	2.5	5	10	25.0
Vol. dist. water	25	24	22.5	20	15	0

Using plastic flasks, 2 ml of mixed solution (Ammonium heptamolybdate solution and Sulphuric acid) was added to 25 ml filtered water sample (diluted appropriately to fit the calibration curve). After 10 min 1 ml of Citrate solution was added. The treated samples were left standing for 6 minutes for reactions to be complete and then absorbance measured at 434 nm against distilled water blank. The spectrophotometer used in all these exercises was of Model IR UV spectrophotometer.

3.7 Phytoplankton Analysis

Phytoplankton species composition and abundance were evaluated from microscopic examination - identification, counting/enumeration and spectroscopic Chlorophyll-a determination.

3.7.1 Chlorophyll-a Determination

According to the USGS (2018), chlorophyll-a concentration is a useful tool for determining the density (biomass) of the phytoplankton population. It indicates the phytoplankton abundance (HELCOM, 2017). For determination of the chlorophyll-a concentration, a 500 ml water sample drawn from each sampling station was prepared. Using a hand vacuum filter pump, the water sample was filtered through Whatman GFC filter paper having 0.45 micrometers pore size. The filter paper was introduced into a test tube holding 15 ml of ethanol. The test tube was covered with an aluminum foil and allowed to stand for one night in a deep freezer, so that the chlorophyll-a is removed and concentrated into the ethanol solution. The filter paper then underwent further pressing to release the chlorophyll-a material still trapped in it. The chlorophyll-a material collected in the test tube was then transferred into centrifuge cuvettes and centrifuged at 2500 rpm for ten minutes.

The supernatant produced from the centrifuged chlorophyll-a solution, underwent decantation into 1cm pathway spectrophotometer cuvettes and then absorbance measurements carried out at wavelengths of 750 nm and 665 nm. The absorbance of chlorophyll-a concentration was established from the difference between the two absorbencies.

The chlorophyll-a concentration was calculated using the Talling and Driver (1961), formulae as follows:

$$\text{Chl-a, } (\mu g\ L^{-1}) = (11.40\ (E665 - E750) \times V_1) / (V_2 \times L) \ ....(\text{Eq. } 1)$$

Where:

11.40 is the absorption coefficient for Chl-a,

V_1 = volume of extract in ml;

V_2 = volume of the filtered water sample in litres;

L = light path length of cuvette in cm;

E665, E750 = optical densities of the sample.

3.7.2 Phytoplankton Cell enumeration

From each sampling station, 500 ml sample of water was collected for the algal cell enumeration. These samples were initially fixed with acidic Lugol's solution for preservation. From the sample, 1 ml was taken and put in Utermohl sedimentation chamber and left to settle for 3 hours and then immediately taken for microscopic examinination and counting using Zeiss Axioinvert 35 inverted microscope at 400x magnification. All the observed single cells, filaments and colonies were counted. Species identification was carried out by reference to the Algae identification field guide and algae identification lab guide by Huynh & Serediak (2006) and the methods described by Cocquyt, Vyverman, & Compére, (1993) from Huber-Pestalozzi (1968) publications on East African Lakes.

The phytoplankton abundance expressed in individuals per litre ($Ind.L^{-1}$) was then determined based on the formula provided by (HELCOM 2017):

Phytoplankton abundance ($Ind.L^{-1}$) = No. of units counted x Coefficient, C.

But $C (L) = A*1000 / (N*a_1*V)$ or $C (L) = A*1000 / (a_2*V)$

where:

A = cross-section area of the top cylinder of the combined sedimentation chamber; the usual inner diameter is 25.0 mm, giving A = 491 mm^2

N = number of counted fields or transects

a_1 = area of single field or transect

a_2 = total counted area

V = volume (cm^3) of sedimented aliquot

3.7.3 Phytoplankton Diversity Indices

After identification of the phytoplankton to the lowest possible taxonomic level (genus/species), four diversity indices (Shannon-Wiener Diversity, Simpson's Diversity, Pielou's, Pielou's species evenness and Margalef's species richness) were computed using the following formulas;

Shannon-Weiner's Diversity Index formula (Shannon & Wiener, 1949);

$$\mathrm{H} = -\sum_{i=I}^{S} p_i \mathrm{In}(p_i)$$

Pielou's Evenness Index (Pielou, 1966);

$$E_H = {}^{\mathrm{H}}/_{\mathrm{InS}}$$

Simpson's Index of Diversity;

Simpson index of diversity = 1-D

$$D = \sum_{i=I}^{s} Pi^2$$

Margalef's Richness Index (d) (Margalef, 1958);

$$\mathrm{d}= \frac{(\mathrm{S}-1)}{In\ N}$$

pi - the proportion of individuals calculated as abundance of individual species divided by total number of individuals in the community sampled

In - the natural log

Σ - The sum of all calculation

S - The total number of species

N- The total number of individuals in the sample

H - Shannon -Wiener index of diversity

D – Simpson's diversity index.

d – Maglef's richness index.

3.7.4 Nygaard's Phytoplankton Quotients

The number of phytoplankton species identified from the families of Cyanophyceae (Myxophyceae), Chlorophyceae, Bacillariophyceae, Euglenophyceae and Zygnematophyceae (Desimidaceae) were used to calculate the Nygaard's phytoplankton quotients of trophic state indices (Nygaard, 1949) by applying the formulae below:

Myxophycean index = (Myxophyceae) / Desimidaceae …………………………….. (Eq. 2)

Chlorophycean index = (Chlorophyceae) / Desimidaceae ………………………….. (Eq. 3)

Euglenophycean index = Euglenophyceae / (Myxophyceae + Chlorophyceae)……... (Eq. 4)

Compound Coefficient = (Myxophyceae + Chlorophyceae +Bacillariophyceae +

Euglenophyceae) / Desmidaceae ……………………..(Eq. 5)

3.8 Trophic State Index

The Carlson Trophic State Index (TSI) was computed mathematically via equations for each of the three parameters (Secchi depth, chlorophyll-a level, and total phosphorus) and then finally finding the average of the values obtained from those equations (Melcher, 2013). The equations were as follows:

$TSI_{SD} = 60 - 14.41(SD)$ …………………………………………… Eq. 6.

$TSI_{Chla} = 9.81(Chla) + 30.6$ …………………………………………Eq. 7.

$TSI_{TP} = 14.42(TP) + 4.15$……………………………………………Eq. 8.

$$TSI_{tot} = \frac{(TSISD + TSIChla + TSITP)}{3}$$

where: Chl-a = Chlorophyll-a concentration (µg/L)

SD = Secchi disk depth (meters)

TP = Total phosphorus concentration (µg/L)

LN = Natural logarithm

TSI = Trophic State Index

The final value obtained as the Carlson's TSI was subsequently used to determine and classify the trophic status of the lake based on the criteria with an index ranging from 0-100 (Table 7).

Table 7: Lake Classification according to Carlson's trophic state index (Carlson, Simpson, 1996).

TSI	Trophic Status	Secchi Depth (SD)	Total Phosphorus (TP)	Chlorophyll-a (Chl-a)
0 – 40	Oligotrophic	>8 – 4	0 – 12	0 – 2.6
40 – 50	Mesotrophic	4 – 2	12 – 24	2.6 – 7.3
50 – 70	Eutrophic	2 – 0.5	24 – 96	7.3 – 56
70 – 100+	Hypereutrophic	0.5 - < 0.25	96 – 384 +	56 – 155 +

3.9 Lake Habitat Survey Protocol

According to the procedure for Lake Habitat Survey described by Rowan *et al.* (2008) and validated by EPA Victoria (2010), LHS was done by a combination of both site surveys and desk-based data collection. Before the LHS field surveys, and as basic prerequisites for LHS study, the vertical profiles of dissolved oxygen and temperature were taken *in-situ* at depths of 0.5m, 1m, 1.5m, and then 2m through 10m using Hanna Instruments® Multiparameter Probe (YSI Professional Plus model) at the "Index Site" which represents the position in the lake with has the greatest depth (Fig. 3a).

In the field, a foot-based approach was employed to conduct a two-day site surveys in early April 2019 since this is a small lake. 10 habitat observation plots (Hab-plot) named from A-J were set up around the lake perimeter as shown in the Fig. 3 (picture a) below. The position of the first Hab-plot was selected randomly, with the rest being distributed evenly around the entire length of the lake. Each Hab-plot consists of three zones; the riparian zone, exposed shore zone and the littoral zone (Fig. 3b). For each plot, a detailed intensive data collection process was carried out by filling a standard LHS questionnaire provided by (Rowan *et al.*, 2008). In these zones, a more detailed visual examination of the vegetation cover, the substrate properties, morphology and the human pressures were recorded and analyzed together with the more generalized visual examination of the entire perimeter of the lake.

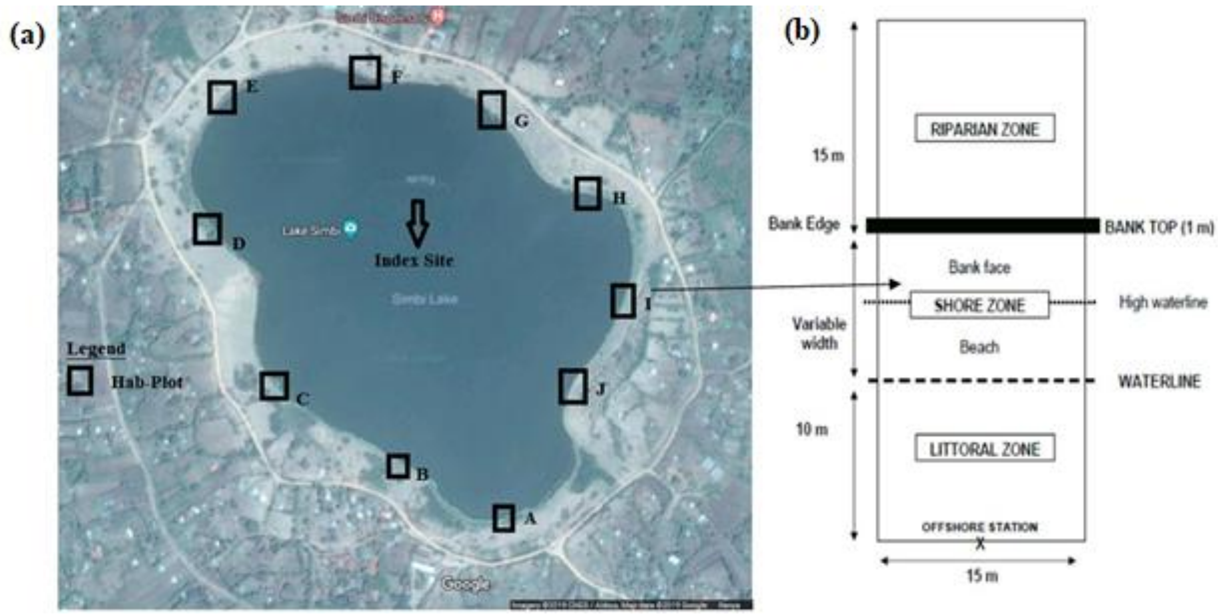

Figure 3: Lake Simbi map showing the positions of the Habitat Plots (Hab-plots) and the Index Site used for LHS study (picture a). The three Hab-plot Zones are shown in picture b. (Author, 2019)

Based on this information, the two LHS metrics of Lake Habitat Quality Assessment (LHQA) and Lake Habitat Modification Score (LHMS) were estimated and scored for Lake Simbi based on the criteria in Tables 9 and 10 as described by Rowan *et al.* (2008). The rated scores were summed up to find the total values for LHQA and LHMS indices for the entire lake. The LHS rating system provided in EPA Victoria (2010) shown in Table 8 below was then adopted in characterizing the ecological condition of the lake measured by each index as; very poor, poor, moderate, and or good depending on the indices values. LHMS is an LHS index that is designed to classify the habitat quality in terms of the degree of hydro-morphological alterations in the lake habitat resulting from the various anthropogenic activities (Rowan *et al.*, 2006). It has a scale running from 0-42, with 0 representing least modified habitat (high ecological status) and 42 representing highly modified habitat (low ecological status). LHQA is an LHS index that is designed to quantify the degree of diversity and naturalness of the lake habitat (Rowan *et al.*, 2006; Dalu *et al.*, 2013, 2016). It has a scale running from 0-112, with 0 representing a habitat that is highly degraded and low diversity, while a 112 representing a habitat that is not degraded at all and has high diversity.

Table 8: LHS Ratings of Habitat Quality and Potential Threats (Victoria EPA, 2010)

Rating	**Lake Habitat Quality Assessment (LHQA)**	**Lake Habitat Modification Score (LHMS)**
Good	>45	0 - 15
Moderate	30 - 45	15 - 30
Poor	15 - 30	30 - 45
Very poor	0 - 15	> 45

Lake Habitat Quality Assessment (LHQA) Scoring Criteria

Table 9: Lake Habitat Quality Assessment (LHQA) Scoring Criteria (Rowan *et al.*, 2008)

LAKE ZONE	Characteristic measured	Measurable feature	Scores	Max
RIPARIAN	Vegetation structural complexity	Proportion of Hab-Plots with complex or simple riparian vegetation structure	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Vegetation longevity/stability	Proportion of Hab-Plots with >10% cover of trees with DBH > 0.3m	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Extent of natural landcover types	Proportion of Hab-Plots with either natural/seminatural woodland, wetland, moorland heath or rock, scree and dunes	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Diversity of natural landcover types	Number of natural cover types recorded	1 for each type, maximum score of 4	4
	Diversity of bank-top features	Number of bank-top features recorded	1 for each type, maximum score of 4	4
SHORE	Shore structural habitat diversity	Proportion of Hab-Plots with an earth or sand bank >1m	1 for 2-4 2 for 5-6 3 for 7-8 4 for 9-10	4
		Proportion of Hab-Plots with trash-line	1 for 2-4 2 for 5-6 3 for 7-8 4 for 9-10	4
	Bank naturalness	Proportion of Hab-Plots with natural bank material	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Diversity of natural bank habitat	Number of natural bank materials recorded	1 for each type, maximum score of 4	4
	Beach naturalness	Proportion of Hab-Plots with natural beach material	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Diversity of natural beach habitats	Number of natural beach materials recorded	1 for each type, maximum score of 4	4
LITTORAL	Hypsographic variation	Coefficient of variation for depth at 10 m from shore over all plots	1 for >25 2 for >50 4 for >75	4
	Extent of natural littoral zones	Proportion of Hab-Plots with natural littoral substrate	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Diversity of natural littoral zone types	Number of natural littoral substrate types recorded	1 for each type, maximum score of 4	4
	Extent of macrophyte cover	Average of total macrophyte cover over all plots	1 for a ‘1’ 2 for a ‘2’ 3 for a ‘3’ 4 for a ‘4’	4
		Number of Hab-Plots where macrophyte cover extends lakewards	1 for 1-3 2 for 4-6 3 for 7-8 4 for 9-10	4
	Diversity of macrophyte structural types	Number of macrophyte cover types recorded (not including filamentous algae)	1 for each type, maximum score of 4	4
	Extent of littoral habitat features	Average of total cover for fish over all plots	1 for a ‘1’ 2 for a ‘2’ 3 for a ‘3’ 4 for a ‘4’	4
	Diversity of littoral habitat features	Number of littoral habitat feature types recorded	1 for each type, maximum score of 4	4
WHOLE LAKE	Diversity of special habitat features	Number of special habitat features (excl. diseased alders)	5 for each type, maximum score of 20	20
		Number of islands	2 for 1 5 for 2-4 10 for 5 or more	10
		Number of deltaic depositional features recorded (excl. unvegetated sand and silt deposits)	2 each type	6

Lake Habitat Modification Score (LHMS) Scoring Criteria

Table 10: Lake Habitat Modification Score (LHMS) Scoring Criteria (Rowan *et al.*, 2008)

Pressure	Scores 0	Scores 2	Scores 4	Scores 6	Scores 8
Shore zone modification	>10% shoreline affected by hard engineering AND Shore reinforcement recorded at 0-1 Hab-Plots	≥10%, <30% shoreline affected by hard engineering OR Shore reinforcement recorded at 2 Hab-Plots OR Poaching recorded at 3 or more Hab-Plots	≥30%, <50% shoreline affected by hard engineering OR Shore reinforcement recorded at 3–4 Hab-Plots	≥50%, <75% shoreline affected by hard engineering OR Shore reinforcement recorded at 5–7 Hab-Plots	≥75% shoreline affected by hard engineering OR Shore reinforcement recorded at 8 or more Hab-Plots
Shore zone intensive use	<10% shoreline non-natural land-cover AND Non-natural land-cover recorded at 0–1 HabPlots	≥10%, <30% shoreline nonnatural land-cover OR Nonnatural land-cover recorded at 2 Hab-Plots	≥30%, <50% shoreline nonnatural land-cover OR Non-natural land-cover recorded at 3–4 Hab-Plots	≥50%, <75% shoreline nonnatural land-cover OR Non-natural land-cover recorded at 5–7 Hab-Plots	≥75% shoreline non-natural land-cover OR Non-natural land-cover recorded at 8 or more Hab-Plots
In-lake use	No in-lake pressures (excluding litter or odour)	1 in-lake pressure (excluding litter or odour)	2 in-lake pressures (excluding litter or odour)	3 in-lake pressures	> 3 in-lake pressures
Hydrology	0–1 hydrological structures	2 hydrological structures OR Presence of an upstream impoundment	3 or more hydrological structures	Principal use hydropower, flood control, water supply OR Raised or lowasd by > ± 1m	1 dam (no fish pass) OR Principal use hydropower, flood control, water supply AND Annual fluctuation > 5m or 50.5 m
Sediment regime	<25% shore affected by erosion AND <25% in-lake area affected by deposition (excluding vegetated islands)	≥25%, <50% affected by erosion OR ≥25%, <50% lake area affected by deposition (excluding vegetated islands)	≥50%, <70% shore affected by erosion OR ≥50%, <70% lake area affected by deposition (excluding vegetated islands)	≥70% shore affect ed by erosi on OR ≥70% lake area affected by deposition (excluding vegetated islands)	

3.10 Statistical Analyses

After sampling and laboratory analysis, the data was managed using the Microsoft Excel 2016 and analysed by both Statistical Package for the Social Science (SPSS) software version 20 and Minitab. The data was subjected to descriptive statistical analyses by Excel spread sheet program. Differences in parameter variability at different sampling sites and sampling months were tested using the ANOVA method at a predetermined *p-value* of $\leq$ 0.05 while those at sampling seasons were tested using Independent t-test at a predetermined *p-value* of $\leq$ 0.05. Whenever significant differences were found using ANOVA, post hoc Tukey pairwise comparisons of Minitab was done in order to bring clarity of sampling sites or seasons which were significantly different. Pearson's correlation analysis was used in establishing relationships between various variables ($p < 0.05$). The analyzed results were presented in the form of tables, bar and line graphs. Indices of ecosystem integrity were evaluated based on some selected physico-chemical and phytoplankton parameters. Nutrients and phytoplankton diversity indices were calculated to form references for future ecosystem perturbations due to anthropogenic and climate change factors.

CHAPTER FOUR

RESULTS

4.1 Introduction

This chapter presents the findings for the spatial and temporal variations of selected water quality (physico-chemical) parameters, phytoplankton community structure, trophic state index (TSI) and habitat quality survey of Lake Simbi. The results for the ANOVA, Tukey test and the independent t-test used to determine whether there were statistical spatial and temporal variation in the means of all the variables assessed ($p < 0.05$) are also presented. Lastly, a correlation analysis computed using Pearson's correlation to determine relationship between different variables ($p < 0.05$) is also presented in this chapter.

4.2 Spatial and temporal variation of the physico-chemical parameters of Lake Simbi

4.2.1 Secchi Depth (Transparency)

On the spatial scale, the Secchi depth registered an overall mean of 0.59 ± 0.01 m in Lake Simbi (Appendix 2). The lowest mean value (0.55 ± 0.01 m) was recorded at ST4 whereas the highest (0.63 ± 0.01 m) was found at both ST1 and ST2. The mean values exhibited a general decreasing trend over time among the stations sampled (Fig. 4).

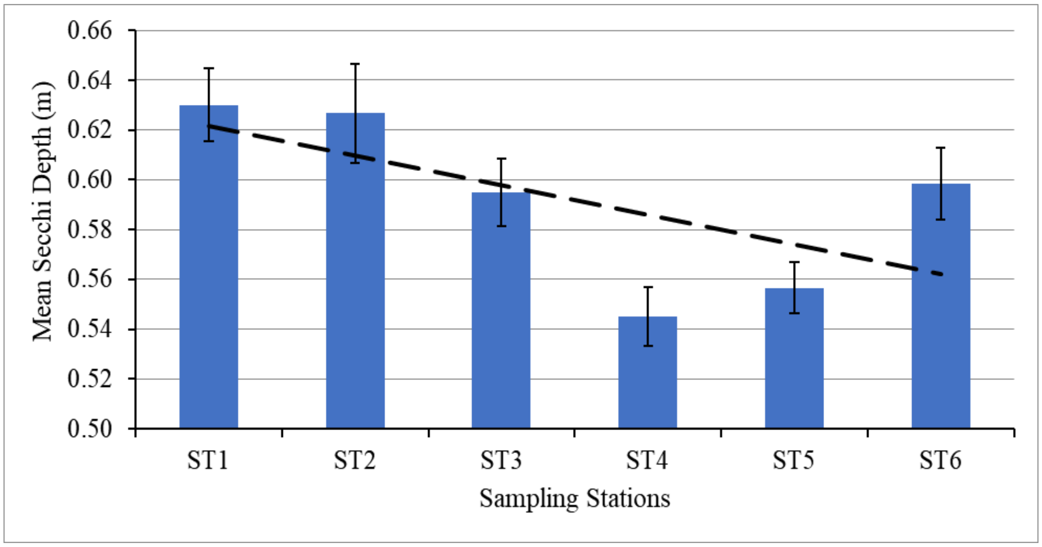

Figure 4: Mean Secchi depth (± SE) at different sampling stations

ANOVA revealed that there were significant differences ($p<0.05$) in the mean Secchi depth values between the stations sampled ($F (5, 30) = 5.914$, $p = 0.001$). Further analysis by Tukey test revealed that the mean Secchi depth of ST1 was significantly higher than the mean Secchi depth values of both ST4 (by between 0.14 and 0.02) and ST5 (by between 0.13 and 0.01); but not significantly different from the mean Secchi depth of ST2, ST3 and ST6. The mean Secchi depth of ST2 was significantly higher than the mean Secchi depth values of both ST4 (by between 0.14 and 0.01) and ST5 (by between 0.13 and 0.008); but not significantly different from the mean Secchi depth of ST3 and ST6.

On the temporal scale, the Secchi depth registered an overall mean of 0.59 ± 0.01 m in the lake (Appendix 3). The lowest mean Secchi depth of 0.56 ± 0.01 m was found in March 2019 while December 2018 ascribed the highest mean value of 0.63 ± 0.01 m. The mean generally had a decreasing trend over time during the sampled months (Fig. 5).

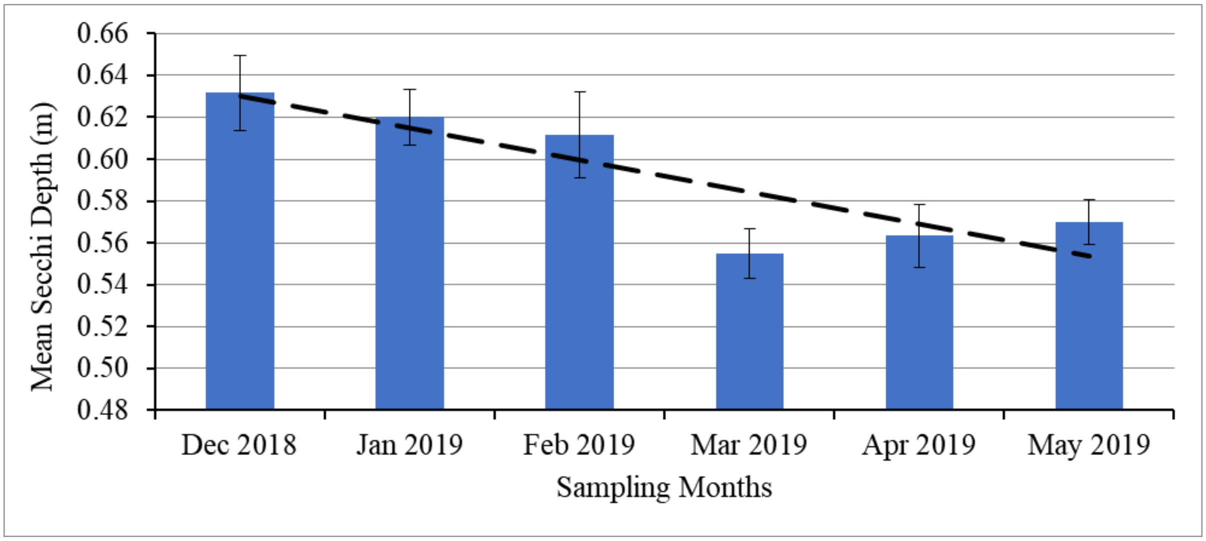

Figure 5: Mean Secchi depth (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean Secchi depth values between the sampled months (F (5, 30) = 4.661, p = 0.003). Further analysis by Tukey test revealed that the mean Secchi depth of December was significantly higher than the mean Secchi depth of April 2019 by between 0.002 and 0.133; but was not significantly different from the mean Secchi depth values of January, February, March and May. Also, the mean Secchi depth of March was significantly lower than the mean Secchi depth of December by between 0.14 and 0.01; but not significantly different from the mean Secchi depth values of January, February and May.

Seasonally, the mean Secchi depth (0.62 ± 0.01 m) of dry season (Dec 2018-Feb 2019) was higher than the mean Secchi depth (0.56 ± 0.01 m) of the wet season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean Secchi depth values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = 4.798, p = 0.000).

4.2.2 Temperature

On the spatial scale, the temperature registered an overall mean of 29.39 ± 0.23 °C in Lake Simbi (Appendix 2). The lowest mean value (29.91 ± 0.51 °C) was recorded at ST1 whereas the highest (29.94 ± 0.59 °C) was found at ST2. There was no clear trend exhibited over time by the mean values among the stations sampled, however, ST3, ST5 and ST6 had relatively similar means (Fig. 6).

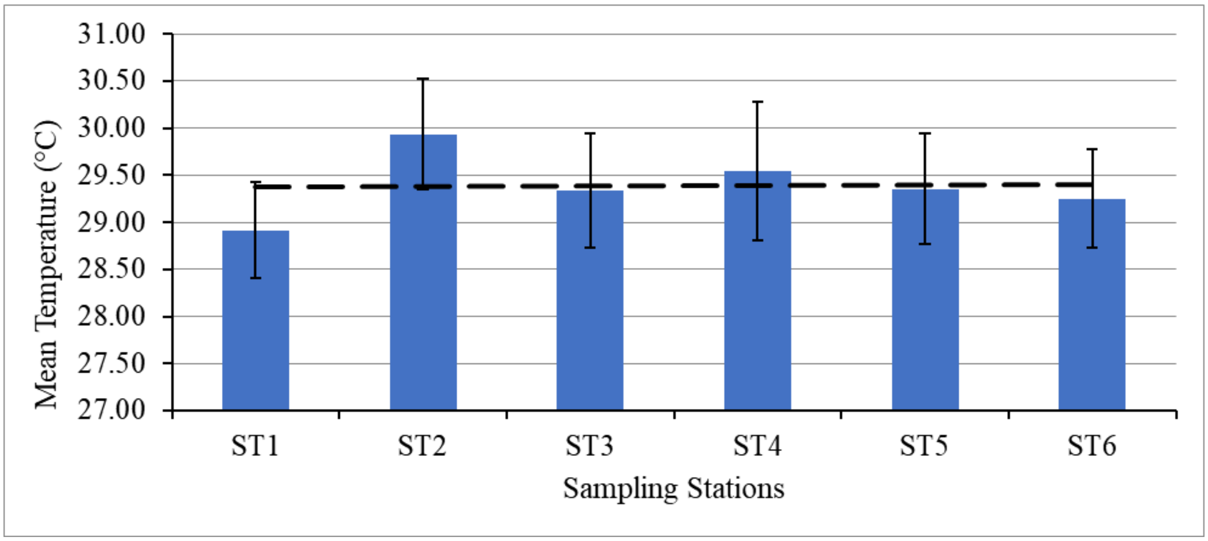

Figure 6: Mean Temperature (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean temperature values between the stations sampled (F (5, 30) = 0.321, p = 0.896).

On the temporal scale, the temperature registered an overall mean of 29.39 ± 0.23 °C in the lake (Appendix 3). The lowest mean Temperature of 26.91 ± 0.10 °C was found in December 2018 while May 2019 ascribed the highest mean value of 30.60 ± 0.24 °C. The mean generally had an increasing trend over time during the sampled months (Fig. 7).

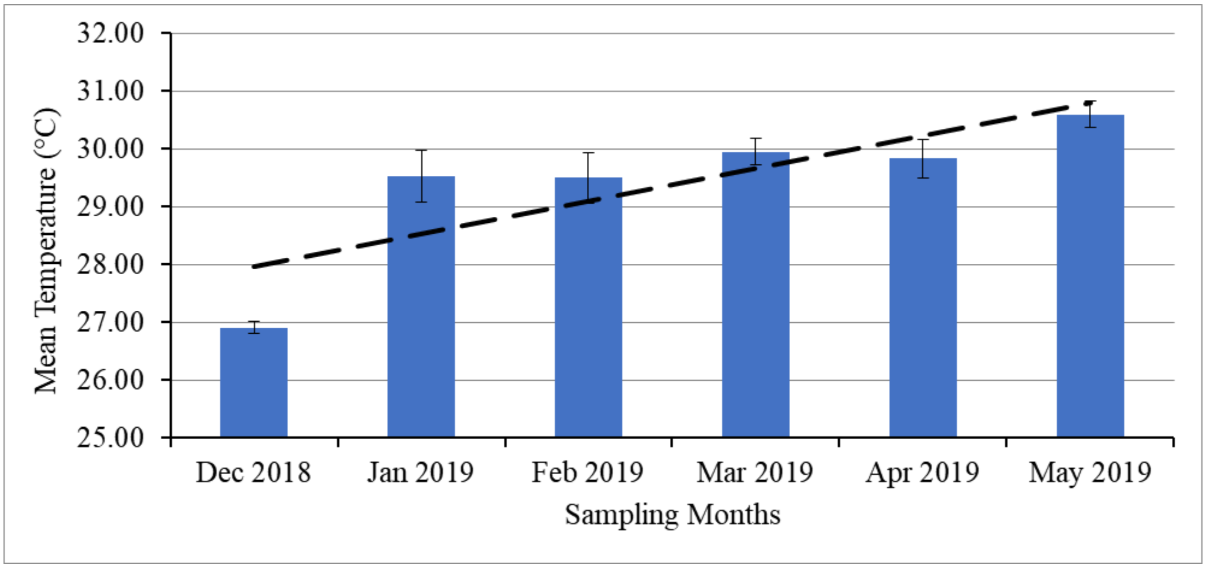

Figure 7: Mean Temperature (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean temperature values between the sampled months (F (5, 30) = 15.788, p = 0.000). Further analysis by Tukey test revealed that the mean temperature of January 2019 was significantly higher than the mean temperature of December by between 1.24 and 4.00; but was not significantly different from the mean temperature values of February, March, April and May. Further, mean temperature of February was significantly higher than the mean temperature of December by between 1.20 and 3.96; but was not significantly different from the mean temperature values of January, March, April and May. The mean temperature of March was significantly higher from the mean temperature of December by between 1.66 and 4.42; but was not significantly different from the mean temperature values of January, February, April and May. The mean temperature of April was significantly higher from the mean temperature of December by between 1.54 and 4.30; but was not significantly different from the mean temperature values of January, February, March and May. Finally, the mean temperature of

May was significantly higher than the mean temperature of December by between 2.31 and 5.07; but not significantly different from the mean temperature values of January, February, April and May.

Seasonally, the mean Temperature (30.13 ± 0.17 °C) of wet season (Dec 2018-Feb 2019) was higher than the mean Temperature (28.65 ± (0.36 °C) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean Temperature values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -3.746$, $p = 0.001$).

4.2.3 Dissolved Oxygen Concentration

On the spatial scale, the Dissolved Oxygen registered an overall mean of 5.25 ± 0.22 mgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (4.73 ± 0.30 mgL^{-1}) was recorded at ST1 whereas the highest (5.54 ± 0.70 mgL^{-1}) was found at ST6. The mean values exhibited a general increasing trend over time among the stations sampled (Fig. 8).

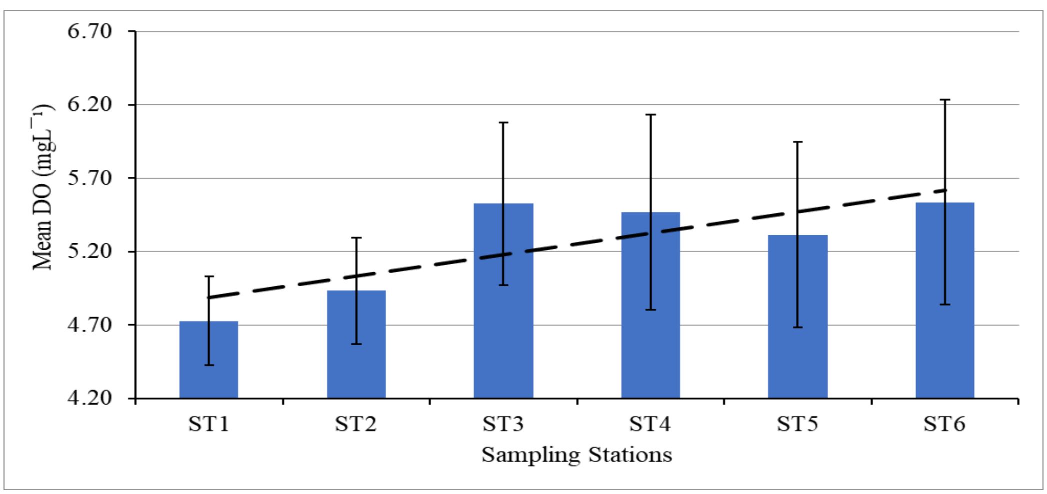

Figure 8: Figure 6: Mean DO (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Dissolved Oxygen concentration values between the stations sampled ($F(5, 30) = 0.376$, $p = 0.861$).

On the temporal scale, the Dissolved Oxygen registered an overall mean of 5.25 ± 0.22 mgL^{-1} in the lake (Appendix 3). The lowest mean Dissolved Oxygen of 4.20 ± 0.06 mgL^{-1} was found in December 2018 while April 2019 ascribed the highest mean value of 7.24 ± 0.69 mgL^{-1}. The mean generally had an increasing trend over time during the sampled months (Fig. 9).

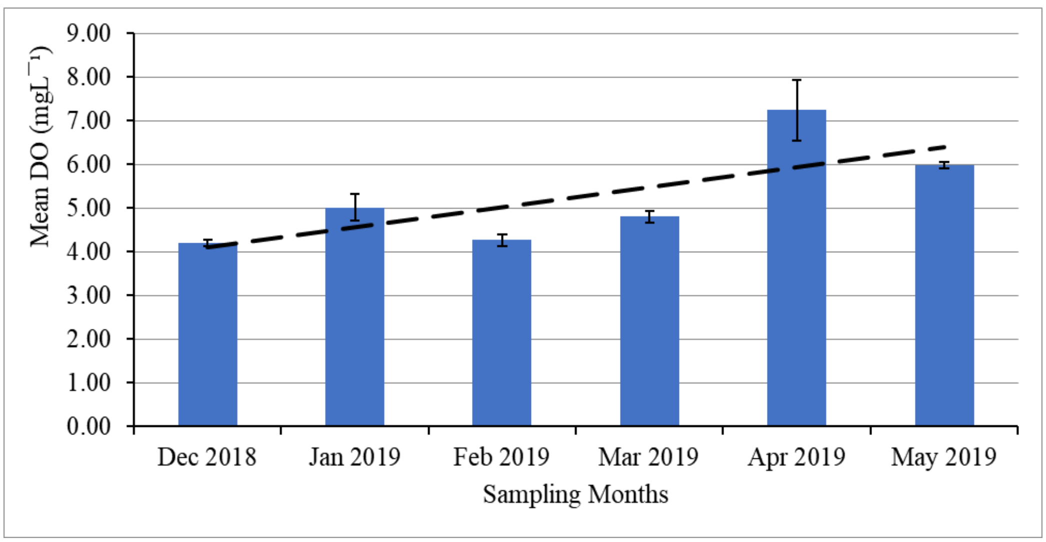

Figure 9: Mean DO (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean Dissolved Oxygen values between the sampled months ($F(5, 30) = 13.280$, $p = 0.000$). Further analysis by Tukey test revealed that the mean DO of December was significantly lower than the mean DO values of the months of April 2019 (by between 4.41 and 1.65), February (by between 4.35 and 1.59), January (by between 3.60 and 0.84), March (by between 3.80 and 1.05); but not significantly different from the mean DO value of May. Also, the mean DO of May was significantly higher than the mean DO of December (by between 0.40 and 3.15); but not significantly different from the mean DO values of January, February and March. The mean DO of May was significantly higher than the mean DO of February (by between 0.34 and 3.09); but not significantly different from the mean DO values of January and March.

Seasonally, the mean Dissolved Oxygen concentration (6.01 ± 0.33 mgL^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean Dissolved Oxygen concentration (4.49 ± 0.14) mgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean Dissolved Oxygen concentration values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = -4.265, p = 0.000).

4.2.4 pH

On the spatial scale, the pH registered an overall mean of 10.23 ± 0.11 in Lake Simbi (Appendix 2). The lowest mean value (10.02 ± 0.32) was recorded at ST3 whereas the highest (10.57 ± 0.30) was found at ST2. There was no clear trend exhibited over time by the mean values among the stations sampled since most stations had relatively similar means except ST2 and ST3 (Fig. 10).

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean pH values between the stations sampled (F (5, 30) = 0.455, p = 0.806).

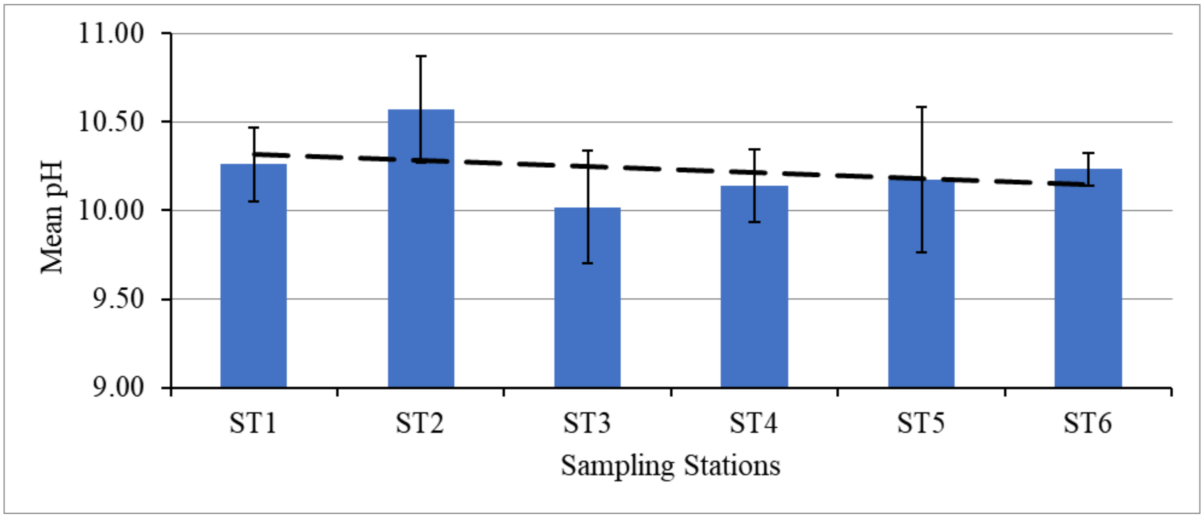

Figure 10: Mean pH (± SE) at different sampling stations

On the temporal scale, the pH registered an overall mean of 10.23 ± 0.11 in the lake (Appendix 3). The lowest mean pH of 9.77 ± 0.56 was found in April 2019 while March 2019 ascribed the highest mean value of 10.59 ± 0.12. There was no clear trend exhibited over time by the mean values among the sampled months since most months had relatively similar means except for March 2019 and April 2019 (Fig. 11).

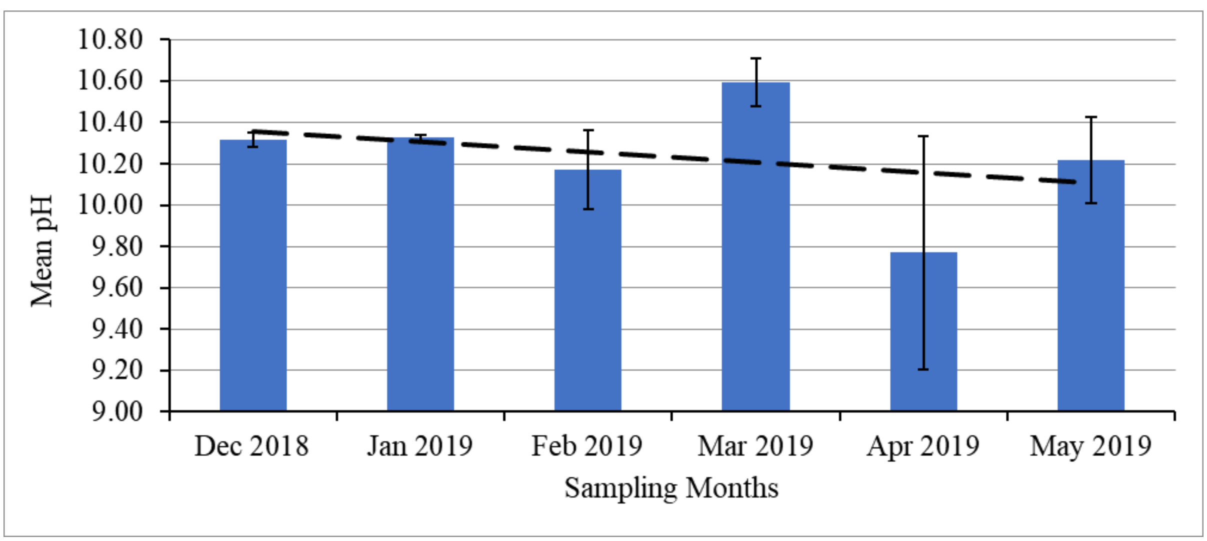

Figure 11: Mean pH (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean pH values between the sampled months ($F (5, 30) = 1.060$, $p = 0.402$).

Seasonally, the mean pH (10.27 ± 0.06) of dry season (Dec 2018-Feb 2019) was higher than the mean pH (10.19 ± 0.21) of the wet season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean pH values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t (34) = 0.360$, $p = 0.721$).

4.2.5 Electrical Conductivity

On the spatial scale, the electrical conductivity registered an overall mean of 17874.75 ± 200.88 μScm^{-1} in Lake Simbi (Appendix 2). The lowest mean value (17658.83 ± 566.71 μScm^{-1}) was recorded at ST4 whereas the highest (18017.33 ± 591.58 μScm^{-1}) was found at ST5. There was no clear trend exhibited over time by the mean values among the stations sampled since most stations recorded relatively similar means (Fig. 12).

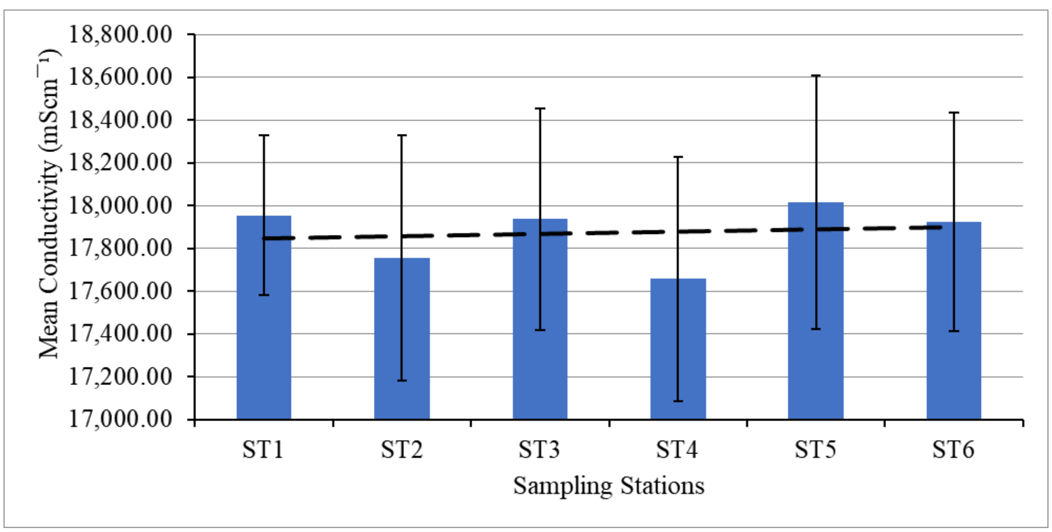

Figure 12: Mean EC (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean electrical conductivity values between the stations sampled ($F(5, 30) = 0.067$, $p = 0.997$).

On the temporal scale, the electrical conductivity registered an overall mean of 17874.75 ± 200.88 µScm^{-1} in the lake (Appendix 3). The lowest mean electrical conductivity of 16060.00 ± 151.60 µScm^{-1} was found in December 2018 while April 2019 ascribed the highest mean value of 19026.67 ± 153.87 µScm^{-1}. The mean generally had an increasing trend over time during the sampled months till April 2019 (Fig. 13).

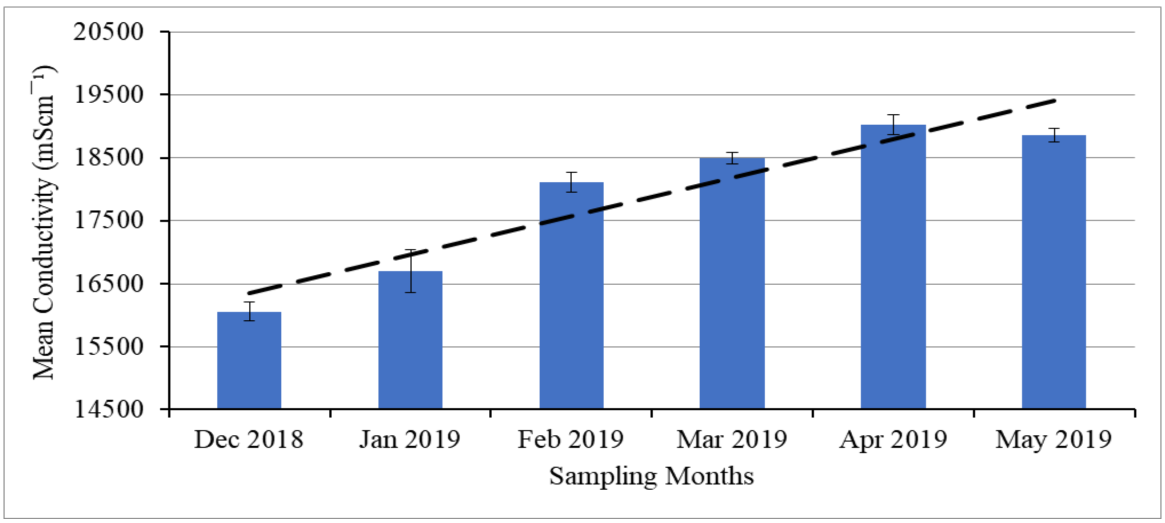

Figure 13: Mean EC (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean electrical conductivity values between the sampled months ($F(5, 30) = 41.799$, $p = 0.000$). Further analysis by Tukey test revealed that the mean electrical conductivity of April 2019 was significantly higher than the mean electrical conductivity values of the months of the months of December 2018 (by between 3776.4 and 2157), January 2019 (by between 3137.5 and 1518.1), February (by between 1725.2 and 105.8); but not significantly different from the mean electrical conductivity value of March 2019 and May 2019. Also, the mean electrical conductivity of December were significantly lower than the mean electrical conductivity values of the months of the months of February 2019 (by between 1241.5 and 2860.9), March 2019 (by between 1622 and 3241.4) and May 2019 (by between 1990.5 and 3609); but not significantly different from the mean electrical conductivity value of January 2019. The mean electrical conductivity of February 2019 was significantly higher than the mean electrical conductivity value of January 2019 (by between 2222.0 and 602.6), but not significantly different from the mean electrical conductivity value of March and May 2019. Finally, the mean electrical conductivity value of January 2019 was significantly lower than the mean electrical conductivity value of March 2019 (between 983 and 2602.5) and May 2019 (by between 1351.6 and 2971.0).

Seasonally, the mean electrical conductivity (18792.83 ± 84.42 μScm^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean electrical conductivity (16956.67 ± 244.69 μScm^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean electrical conductivity values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t (34) = -7.094$, $p = 0.000$).

4.2.6 Salinity

On the spatial scale, the salinity registered an overall mean of 9.81 ± 0.06 ppt in Lake Simbi (Appendix 2). The lowest mean value (9.76 ± 0.17 ppt) was recorded at ST6 whereas the highest (9.91 ± 0.16 ppt) was found at ST4. There was no clear trend exhibited over time by the mean values among the stations sampled since most them recorded relatively similar means (Fig. 14).

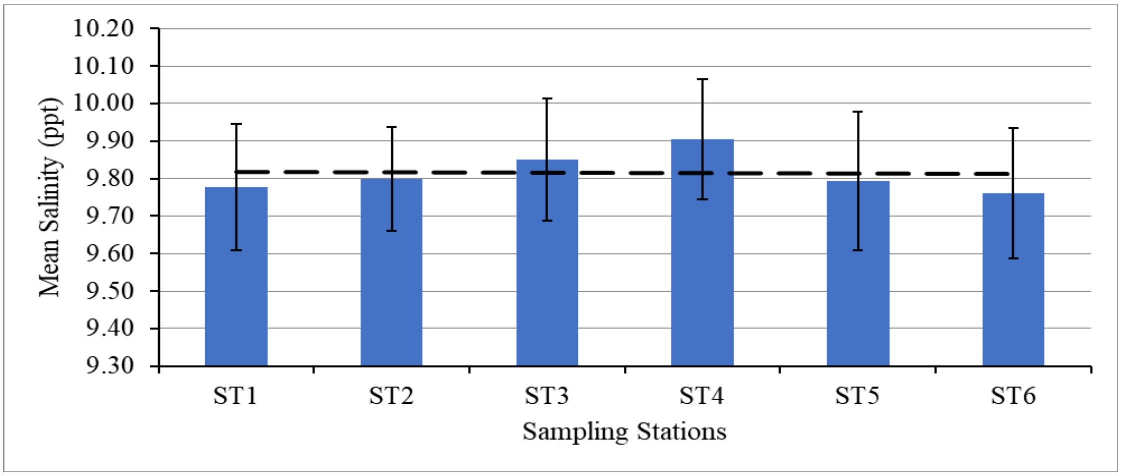

Figure 14: Mean Salinity (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean salinity values between the stations sampled (F (5, 30) = 0.105, p = 0.990).

On the temporal scale, the salinity registered an overall mean of 9.81 ± 0.06 ppt in the lake (Appendix 3). The lowest mean salinity of 9.26 ± 0.13 ppt was found in December 2018 while April 2019 ascribed the highest mean value of 10.21 ± 0.01 ppt. Just like in conductivity, the means of salinity generally had an increasing trend during the sampled months till April 2019 (Fig. 15).

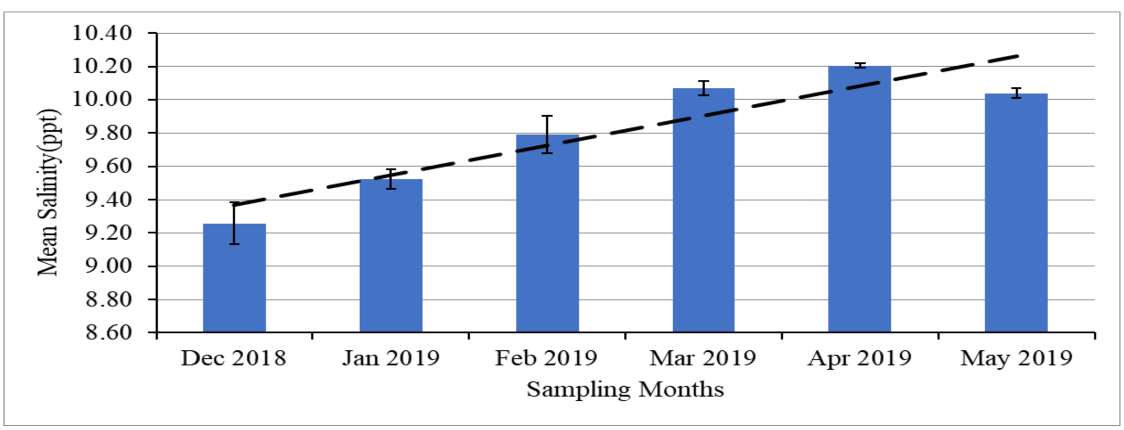

Figure 15: Mean Salinity (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean salinity values between the sampled months ($F(5, 30) = 22.762$, $p = 0.000$). Further analysis by Tukey test revealed that the mean salinity of April 2019 was significantly higher than the mean salinity values of the months of December 2018 (by 1.28 and 0.62), January 2019 (by between 1.01 and 0.35) and February (by between 0.74 and 0.08); but not significantly different from the mean salinity values of March 2019 and May 2019. Also, the mean salinity of December was significantly lower than the mean salinity of the months of February 2019 (by between 0.21 and 0.87), March 2019 (by between 0.48 and 1.14) and May 2019 (by between 0.45 and 1.11); but not significantly different from the mean salinity value of January 2019. Finally, the mean salinity of January 2019 was significantly lower than the mean salinity values of the months of March 2019 (by between 0.21 and 0.87) and May 2019 (by between 0.18 and 0.84).

Seasonally, the mean salinity (10.10 ± 0.03 ppt) of wet season (Dec 2018-Feb 2019) was higher than the mean salinity (9.52 ± 0.08 ppt) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean salinity values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -7.176$, $p = 0.000$).

4.2.7 Alkalinity

On the spatial scale, the alkalinity registered an overall mean of 8115.11 ± 123.66 mgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (7713.67 ± 439.81 mgL^{-1}) was recorded at ST2 whereas the highest (8873.67 ± 412.90 mgL^{-1}) was found at ST1. The mean values exhibited a slightly decreasing trend over time among the stations sampled (Fig. 16).

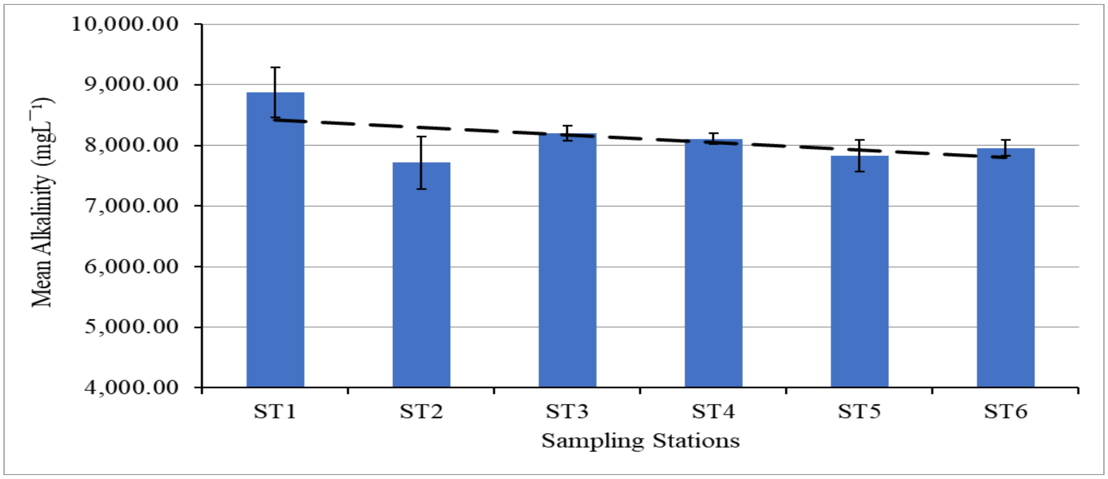

Figure 16: Mean Alkalinity (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean alkalinity values between the stations sampled (F (5, 30) = 2.160, p = 0.085).

On the temporal scale, the alkalinity registered an overall mean of 8115.11 ± 123.66 mgL^{-1} in the lake (Appendix 3). The lowest mean alkalinity of 7738.50 ± 122.34 mgL^{-1} was found in May 2019 while January 2019 ascribed the highest mean value of 9087.67 ± 307.74 mgL^{-1}. The mean values exhibited a general decreasing trend over time among the sampled months (Fig. 17).

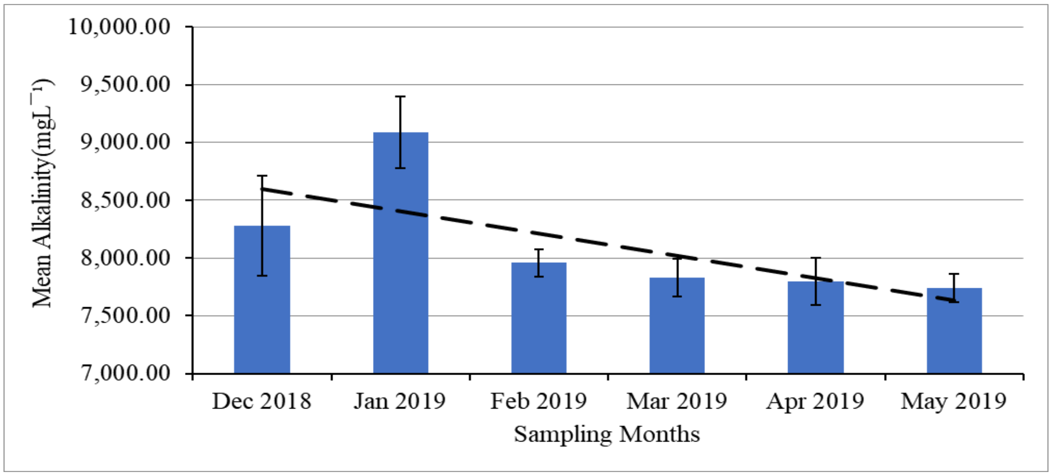

Figure 17: Mean Alkalinity (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean alkalinity values between the sampled months ($F (5, 30) = 4.195$, $p = 0.005$). Further analysis by Tukey test revealed that the mean alkalinity of April 2019 was significantly lower than the mean alkalinity of January 2019 (by between 213.0 and 2371.7); but not significantly different from the mean alkalinity values of December 2018, February 2019, March 2019 and May 2019. Also, the mean alkalinity of February 2019 was significantly lower than the mean alkalinity of January 2019 (by between 51.2 and 2209.7); but not significantly different from the mean alkalinity values of March 2019 and May 2019. The mean alkalinity of January 2019 was significantly higher than the mean alkalinity values of the months of March 2019 (by between 2334 and 175.3) and May (by between 2428.5 and 269.8).

Seasonally, the mean alkalinity (8441.28 ± 205.48 mgL^{-1}) of dry season (Dec 2018-Feb 2019) was higher than the mean alkalinity (7788.94 ± 90.73 mgL^{-1}) of the wet season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean alkalinity values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = 2.904$, $p = 0.006$).

4.2.8 Total Hardness

On the spatial scale, the total hardness registered an overall mean of 139.22 ± 8.75 mgL^{-1} in Lake Simbi (Appendix 2). The lowest mean hardness value (133.83± 22.84 mgL^{-1}) was recorded at ST3 whereas the highest (146.83 ± 24.95 mgL^{-1}) was found at ST6. There was no clear trend exhibited over time by the mean values among the stations sampled since most stations recorded relatively similar means (Fig. 18).

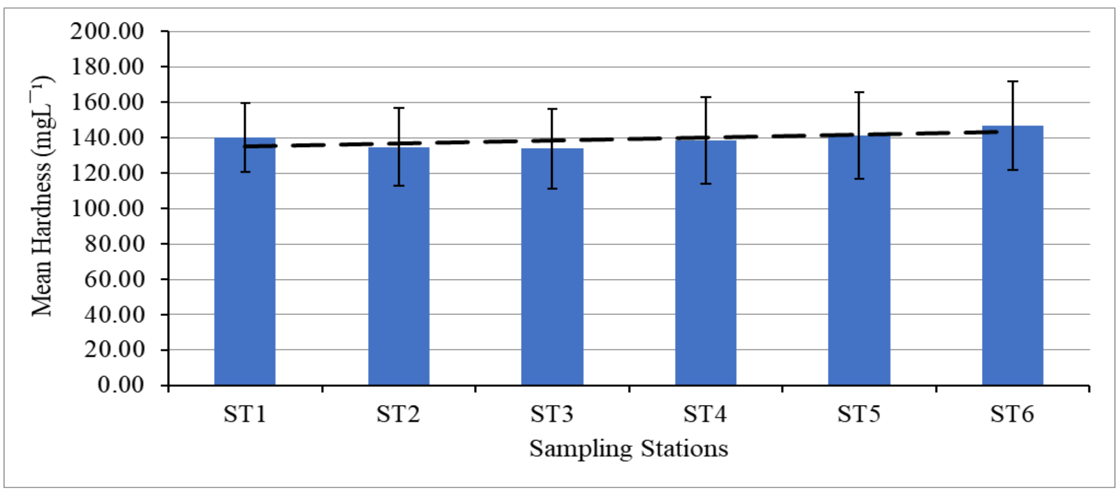

Figure 18: Mean Hardness (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean hardness values between the stations sampled ($F(5, 30) = 0.043$, $p = 0.999$).

On the temporal scale, the hardness registered an overall mean of 139.22 ± 8.75 mgL^{-1} in the lake (Appendix 2). The lowest mean hardness of 87.33 ± 1.43 mgL^{-1} was found in January 2019 while April 2019 ascribed the highest mean value of 195.83 ± 4.89 mgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 19).

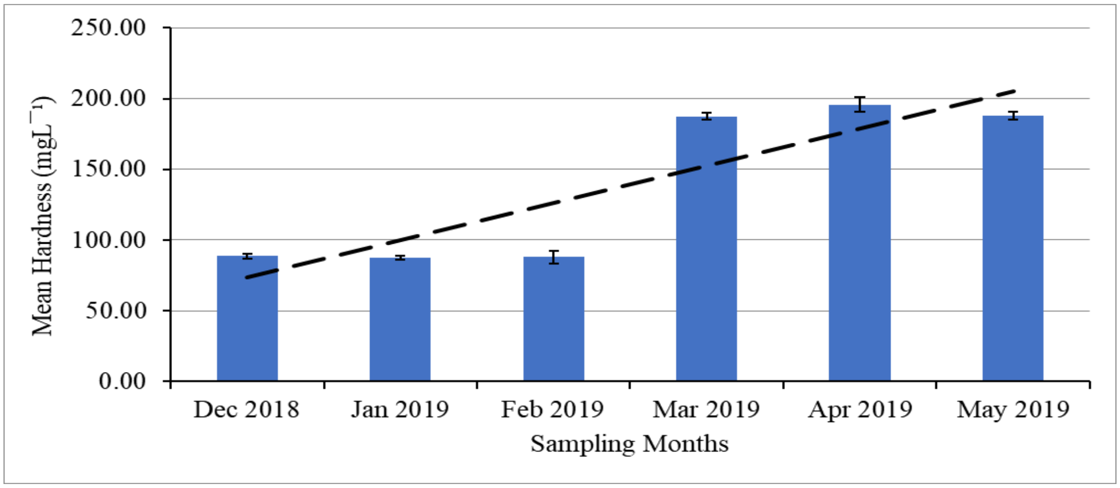

Figure 19: Mean Hardness (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean hardness values between the sampled months ($F(5, 30) = 311.412$, $p = 0.000$). Further analysis by Tukey test revealed that the mean hardness of April 2019 was significantly higher than the mean hardness values of the months of December 2018 (by between 120.86 and 93.47), January 2019 (by between 112.19 and 94.81), and February 2019 (by between 121.53 and 94.14); but not significantly different from the mean hardness values of March 2019 and May 2019. Further, the mean hardness of December 2018 was significantly lower than the mean hardness values of the months of March 2019 (by between 85.14 and 112.53) and May 2019 (by between 85.64 and 113.03); but not significantly different from the mean hardness values of January 2019 and February 2019. Also, the mean hardness of February 2019 was significantly lower than the mean hardness values of the months of March 2019 (by between 85.14 and 112.53) and May 2019 (by between 85.64 and 113.03); but not significantly different from the mean hardness value of January 2019. Finally, the mean hardness of

January 2019 was significantly lower than the mean hardness values of the months of March 2019 (by between 85.14 and 112.53) and May 2019 (by between 85.64 and 113.03).

Seasonally, the mean hardness (190.44 ± 2.12 mgL^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean hardness (88.00 ± 1.53 mgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean hardness values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -39.176$, $p = 0.000$).

4.2.9 Total Dissolved Solids

On the spatial scale, the total dissolved oxygen registered an overall mean of 5372.07 ± 837.27 mgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (5261.67 ± 2191.90 mgL^{-1}) was recorded at ST5 whereas the highest (5450.17 ± 2237.99 mgL^{-1}) was found at ST4. The mean values exhibited a constant trend over time among the stations sampled since all of them recorded almost similar means (Fig. 20).

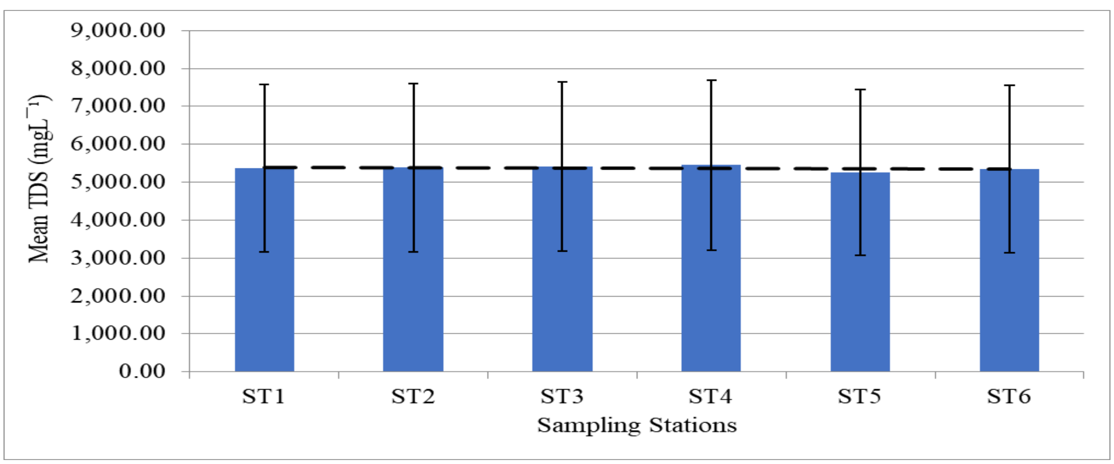

Figure 20: Mean TDS (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean TDS values between the stations sampled ($F(5, 30) = 0.001$, $p = 1.000$).

On the temporal scale, the TDS registered an overall mean of 5372.07 ± 837.27 mgL^{-1} in the lake (Appendix 3). The lowest mean TDS of 55.07 ± 5.36 mgL^{-1} was found in January 2019 while April 2019 ascribed the highest mean value of 11340.67± 19.41 mgL^{-1}. The mean values exhibited an erratic but generally increasing trend over time among the sampled months (Fig. 21).

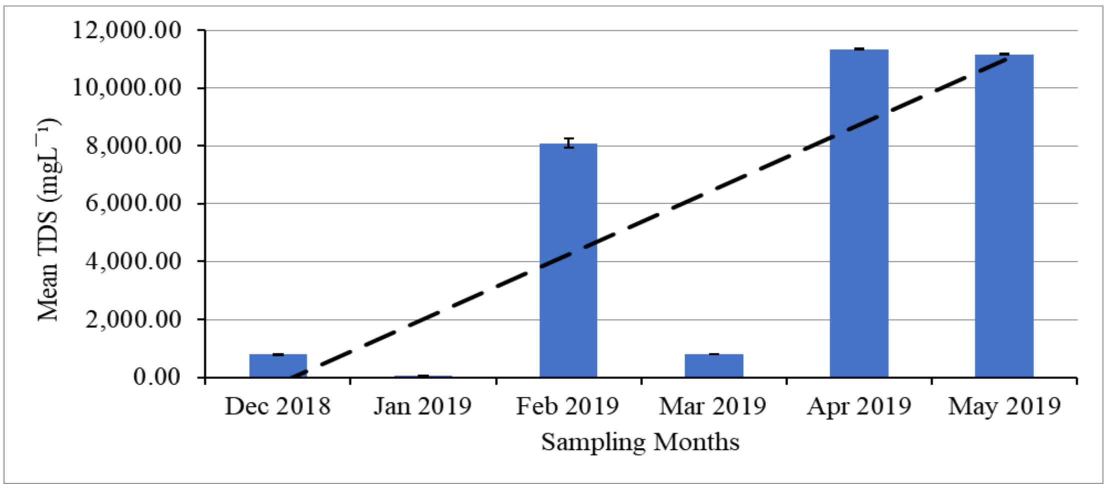

Figure 21: Mean TDS (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean TDS values between the sampled months ($F(5, 30) = 6324.146$, $p = 0.000$). Further analysis by Tukey test revealed that the mean TDS of April 2019 was significantly higher than the mean TDS values of the months of December 2018 (by between 10851and 10265), January 2019 (by between 11579 and 10992), February 2019 (by between 3544 and 2957) and March (by between 10848 and 10261); but not significantly different from the mean TDS values of May 2019. Further, the mean TDS of December 2018 was significantly higher than the mean TDS values of the months of January (by between 1021 and 435); but lower than the mean TDS of February 2019 (by between 7014 and 7600) and May 2019 (by between 10101 and 10688) even though it was not significantly different from the mean TDS values of March 2019. The mean TDS of February 2019 was significantly higher than the mean TDS values of the months of January (by between 8328 and 7742) and March 2019 (by between 7597 and 7010) but significantly lower than the mean TDS of May 2019 (by between 2794 and 3381). The Month of January also had its mean TDS value being significantly lower than the mean TDS values of the month of March 2019 (by between 438 and) and May 2019 (by between 10829 and 11416). Finally, the mean TDS of March 2019 was significantly lower than the mean TDS values of May 2019 (by between 10098 and 10685).

Seasonally, the mean TDS (7768.17 ± 1197.53 mgL^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean TDS (2975.97 ± 881.51 mgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean TDS values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -3.223$, $p = 0.003$).

4.2.10 Turbidity

On the spatial scale, the turbidity registered an overall mean of 81.89 ± 5.07 NTU in Lake Simbi (Appendix 2). The lowest mean value (64.45 ± 4.40 NTU) was recorded at ST4 whereas the highest (131.32 ± 17.68 NTU) was found at ST2. The mean values exhibited a slightly decreasing trend over time among the stations sampled (Fig. 22).

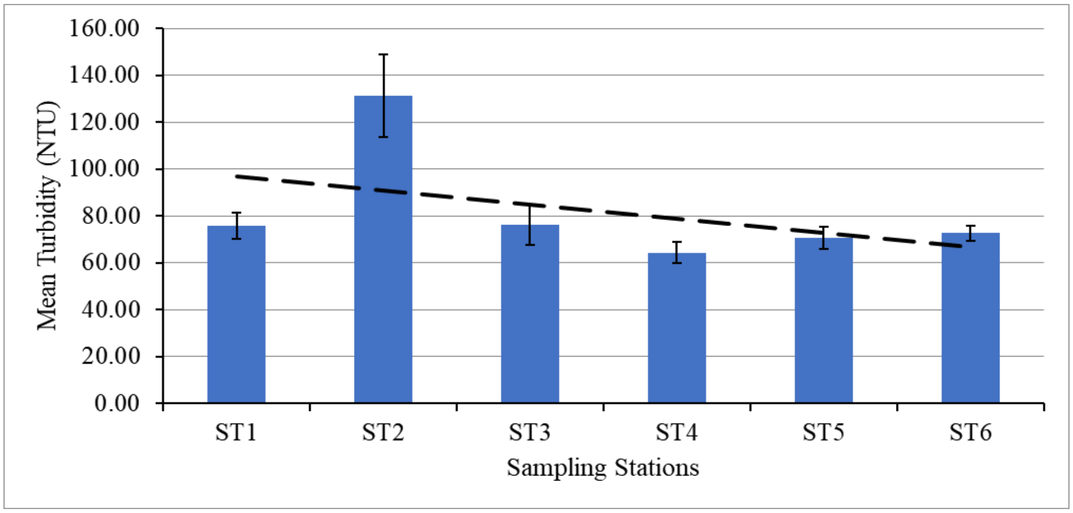

Figure 22: Mean Turbidity (± SE) at different sampling stations

ANOVA revealed that there were significant differences ($p<0.05$) in the mean turbidity values between the stations sampled (F (5, 30) = 7.674, p = 0.000). Further analysis by Tukey test revealed that the mean turbidity of ST1 was significantly lower than the mean turbidity of ST2 (by between 17.16 and 93.52) but was not significantly different from the mean turbidity values of ST3, ST4, ST5 and ST6. On the other hand, the mean turbidity of ST2 was significantly higher than the mean turbidity of ST3 (by between 93.17 and 16.80), ST4 (by between 105.05 and 28.68), ST5 (by between 98.88 and 22.52) and ST6 (by between 96.85 and 20.48).

On the temporal scale, the turbidity registered an overall mean of 81.89 ± 5.07 NTU in the lake (Appendix 3). The lowest mean turbidity of 61.43 ± 2.14 NTU was found in December 2018 while May 2019 ascribed the highest mean value of 96.88 ± 6.75 NTU. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 23).

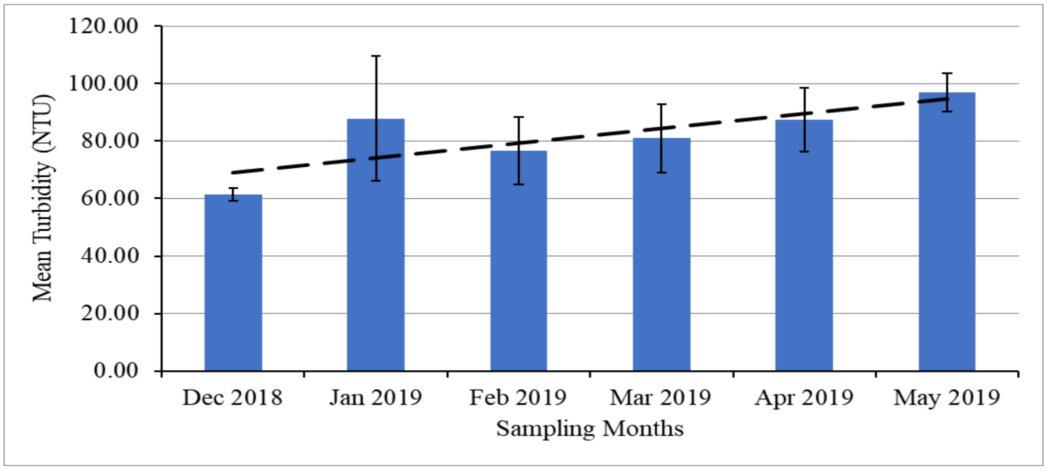

Figure 23: Mean Turbidity (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean turbidity values between the sampled months (F (5, 30) = 0.952, p = 0.462).

Seasonally, the mean turbidity (88.44 ± 5.76 NTU) of wet season (Dec 2018-Feb 2019) was higher than the mean turbidity (75.33 ± 8.22 NTU) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean turbidity values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = -1.307, p = 0.200).

4.2.11 Soluble Reactive Phosphorus

On the spatial scale, the SRP registered an overall mean of 1763.77 ± 35.83 μgL^{-1} in Lake Simbi (Appendix 2). The lowest mean SRP value (1597.28 ± 64.68 μgL^{-1}) was recorded at ST2 whereas the highest (1815.89 ± 108.35 μgL^{-1}) was found at ST1. The mean values exhibited a constant trend among the stations sampled since all stations recorded almost similar means (Fig. 24).

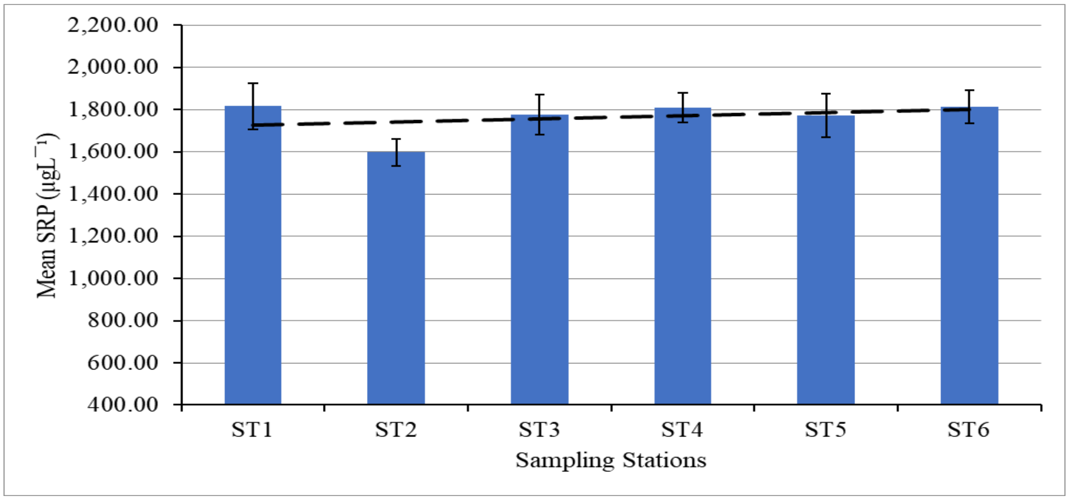

Figure 24: Mean SRP (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean SRP values between the stations sampled (F (5, 30) = 0.894, p = 0.498).

On the temporal scale, the SRP registered an overall mean of 1763.77 ± 35.83 μgL^{-1} in the lake (Appendix 3). The lowest mean SRP of 1599.50 ± 56.50 μgL^{-1} was found in December 2018 while May 2019 ascribed the highest mean value of 2027.28 ± 100.42 μgL^{-1}. The mean values exhibited a slightly increasing trend over time among the sampled months (Fig. 25).

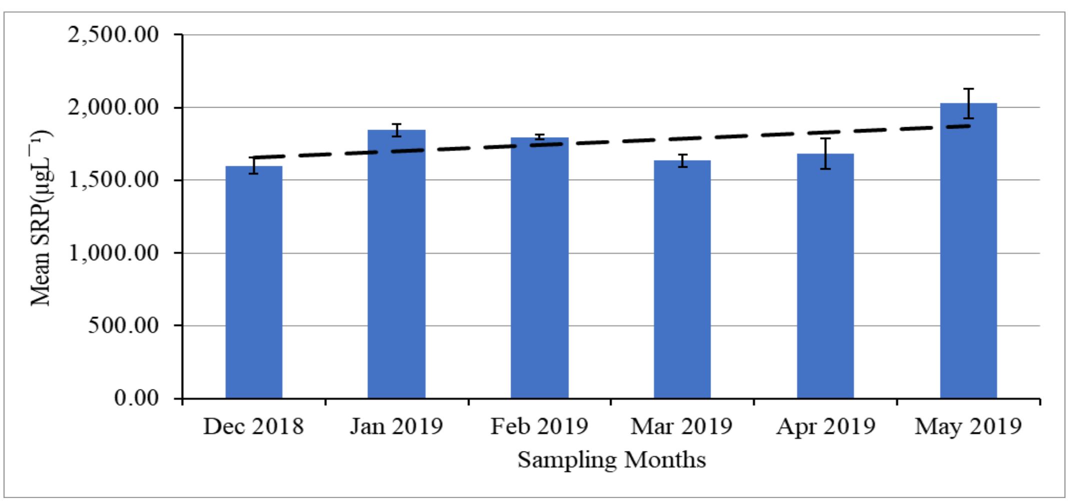

Figure 25: Mean SRP (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean SRP values between the sampled months ($F(5, 30) = 5.388$, $p = 0.001$). Further analysis by Tukey test revealed that the mean SRP of April 2019 was significantly lower than the mean SRP of May 2019 (by between 49.9 and 641.6); but not significantly different from the mean SRP values of December 2018, January 2019, February 2019, March 2019 and May 2019. Further, the mean SRP of December 2018 was significantly lower than the mean SRP of May 2019 (by between 131.9 and 723.6); but not significantly different from the mean SRP values of January 2019, February 2019 and March 2019. Finally, the month of March 2019 had its mean SRP being lower than the mean SRP of May 2019 (by between 97.2 and 688.9).

Seasonally, the mean SRP (1781.01 ± 63.76 μgL^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean SRP (1746.54 ± 34.39 μgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean SRP values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -0.476$, $p = 0.637$).

4.2.13 Total Phosphorus

On the spatial scale, the TP registered an overall mean of 2829.70 ± 77.25 μgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (2729.14 ± 229.80 μgL^{-1}) was recorded at ST2 whereas the highest (3013.60 ± 146.91 μgL^{-1}) was found at ST1. There was no clear trend exhibited over time by the mean values among the stations sampled however, most stations recorded relatively similar means (Fig. 26).

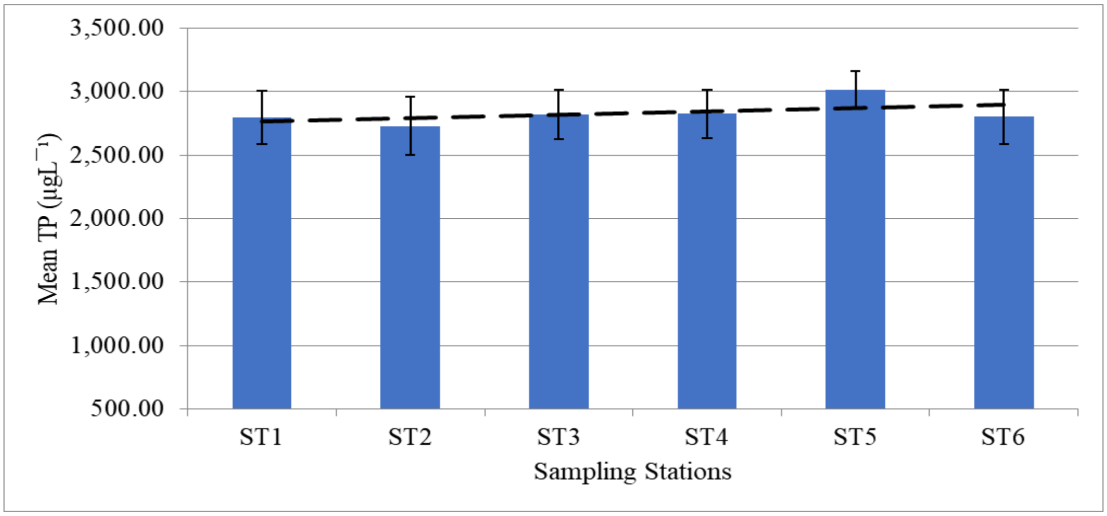

Figure 26: Mean TP (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean TP values between the stations sampled ($F_{(5, 30)} = 0.230$, $p = 0.947$).

On the temporal scale, the TP registered an overall mean of 2829.70 ± 77.25 μgL^{-1} in the lake (Appendix 3). The lowest mean TP of 2212.09 ± 75.51 μgL^{-1} was found in December 2018 while April 2019 ascribed the highest mean value of 3480.43 ± 42.18 μgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 27).

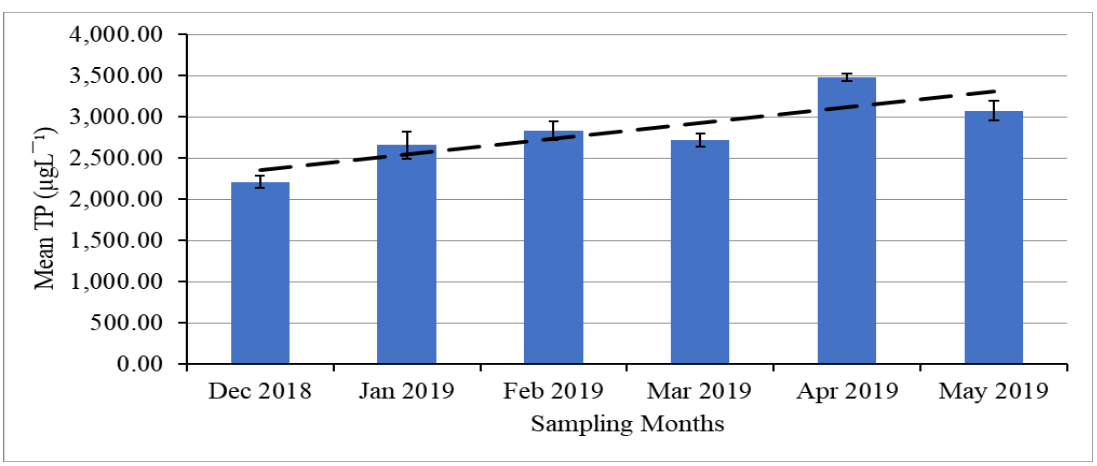

Figure 27: Mean TP (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean TP values between the sampled months ($F(5, 30) = 15.805$, $p = 0.000$). Further analysis by Tukey test revealed that the mean TP of April 2019 was significantly higher than the mean TP of the months of December 2018 (by between 1729.4 and 807.3), January 2019 (by between 1285.3 and 363.3) and February 2019 (by between 1106.3 and 184.2), March 2019 (by between 1223.4 and 301.4); but was not significantly different from the mean TP of May 2019. Finally, the mean TP of December 2018 was significantly lower than the mean TP of the months of February 2019 (by between 162.1 and 1084.1), March 2019 (by between 44.9 and 966.9) and May 2019 (by between 403.2 and 1325.3); but was not significantly different from the mean TP of January 2019.

Seasonally, the mean TP (3091.59 ± 88.92 μgL^{-1}) of wet season (Dec 2018-Feb 2019) was higher than the mean TP (2567.81 ± 92.72 μgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean TP values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -4.077$, $p = 0.000$).

4.2.14 Total Nitrogen

On the spatial scale, the TN registered an overall mean of 456.33 ± 73.45 μgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (331.97 ± 134.91 μgL^{-1}) was recorded at ST5 whereas the highest (618.11 ± 221.11 μgL^{-1}) was found at ST4. The mean values exhibited a general decreasing trend over time among the stations sampled (Fig. 28).

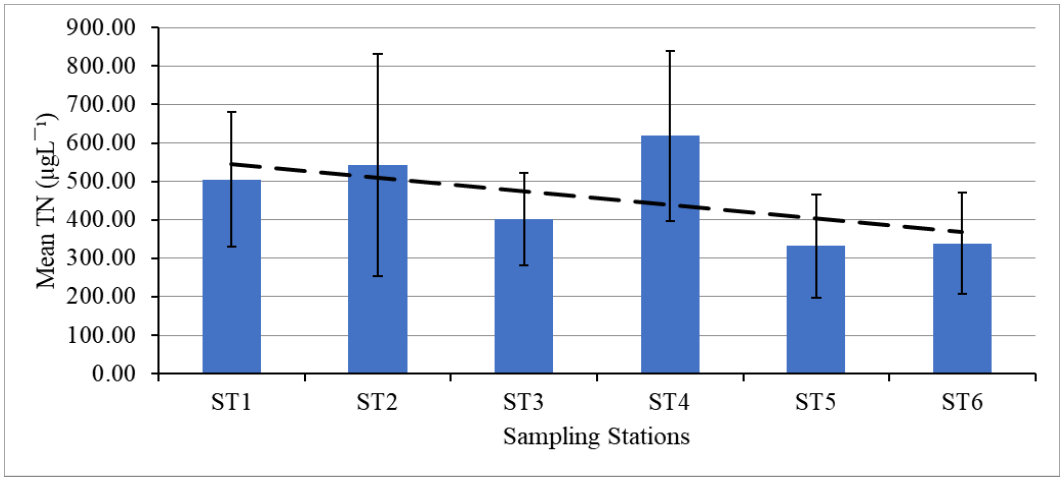

Figure 28: Mean TN (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean TN values between the stations sampled ($F(5, 30) = 0.384$, $p = 0.856$).

On the temporal scale, the TN registered an overall mean of 456.33 ± 73.45 µgL^{-1} in the lake (Appendix 3). The lowest mean TN of 63.37 ± 0.95 µgL^{-1} was found in December 2018 while March 2019 ascribed the highest mean value of 1061.27 ± 214.77 µgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months till April 2019 (Fig. 29).

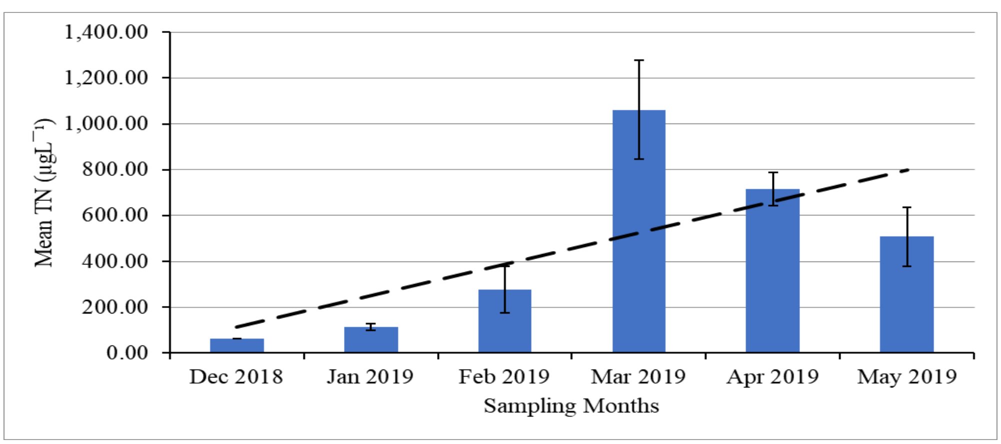

Figure 29: Mean TN (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean TN values between the sampled months ($F(5, 30) = 11.296$, $p = 0.000$). Further analysis by Tukey test revealed that the mean TN of April 2019 was significantly higher than the mean TN of the months of December 2018 (by between 1144.2 and 159.8) and January 2019 (by between 1093.7 and 109.3); but was not significantly different from the mean TN of February 2019 March 2019 and May 2019. Also, the mean TN of December 2019 was significantly lower than the mean TN of March 2019 but was not significantly different from the mean TN of January 2019, February 2019 March 2019 and May 2019. The month of February had a mean TN value being lower than the mean TN value of March 2019 (by between 293.1 and 1277.4) but was not significantly different from the mean TN of January 2019 and May 2019. Further, the mean TN of January 2019 was significantly lower than the mean TN of March 2019 (by between 455.2 and 1439.6) but was not significantly different from the mean TN of May 2019. Finally, the mean TN of March 2019 was significantly higher than the mean TN of May 2019 (by between 1045.3 and 61.0).

Seasonally, the mean TN (761.59 ± 98.72 μgL^{-1}) of the wet season (Dec 2018-Feb 2019) was higher than the mean TN (151.07 ± 38.82 μgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean TN values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -5.755$, $p = 0.000$).

Redfield's Ratios for Lake Simbi

On the spatial scale, the N: P ratios were recorded within the range of 1:5 – 1:9 with the lowest ratio recorded at ST5 whereas the highest was recorded at ST5; while on the temporal scale, the N: P ratios were recorded within the range of 1:3 – 1:35 with the lowest ratio recorded in December 2018 whereas the highest was recorded in March 2019 (Table 11).

Table 11: Spatial and Temporal variation of Nitrogen Phosphorus Ratios for Lake Simbi

Temporal Variation of TN:TP in Lake Simbi					Spatial Variation of TN:TP in Lake Simbi			
Month	**TN**	**TP**	**N:P**		**Station**	**TN**	**TP**	**N:P**
Dec 2018	63.37	2212.09	1:35		ST1	504.95	2795.45	1: 6
Jan 2019	113.85	2656.14	1:23		ST2	542.32	2729.14	1:5
Feb 2019	276.00	2835.19	1:10		ST3	401.48	2817.49	1:7
Mar 2019	1061.27	2718.00	1:3		ST4	618.11	2822.47	1:5
Apr 2019	715.39	3480.43	1: 5		ST5	331.97	3013.60	1:9
May 2019	508.11	3076.33	1:6		ST6	339.16	2800.05	1:8

4.2.15 Nitrates-Nitrogen (NO_3-N)

On the spatial scale, the Nitrates (NO_3-N) registered an overall mean of 8.46 ± 1.11 μgL^{-1} in Lake Simbi (Appendix 2). The lowest mean value (5.33 ± 1.22 μgL^{-1}) was recorded at ST5 whereas the highest (12.42 ± 4.75 μgL^{-1}) was found at ST4. The mean values exhibited a slightly decreasing trend over time among the stations sampled (Fig. 30).

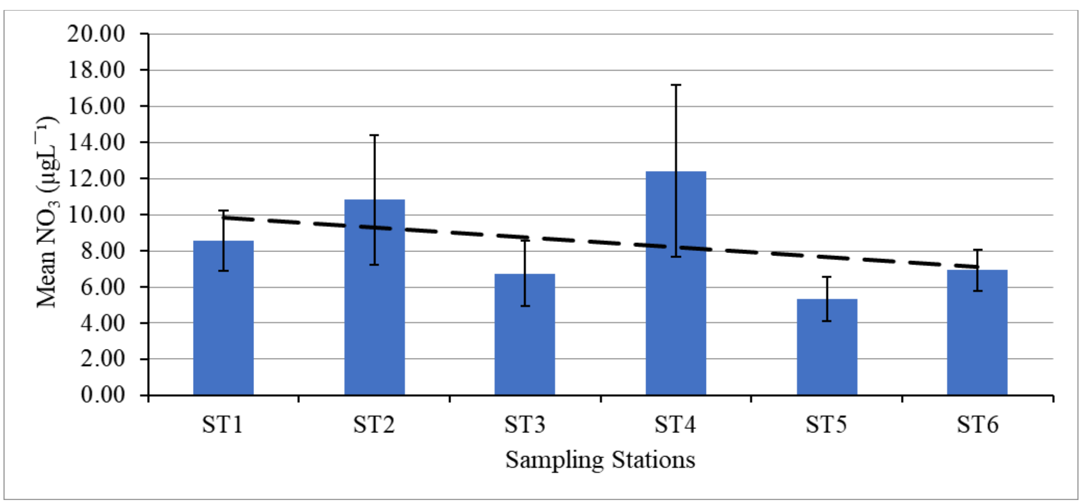

Figure 30: Mean Nitrates (NO_3-N) (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean nitrates (NO_3-N) values between the stations sampled ($F (5, 30) = 0.989$, $p = 0.441$).

On the temporal scale, the Nitrates (NO_3-N) registered an overall mean of 8.46 ± 1.11 μgL^{-1} in the lake (Appendix 3). The lowest mean nitrates (NO_3-N) of 2.94 ± 0.24 μgL^{-1} was found in January 2019 while March 2019 ascribed the highest mean value of 13.75 ± 3.08 μgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 31).

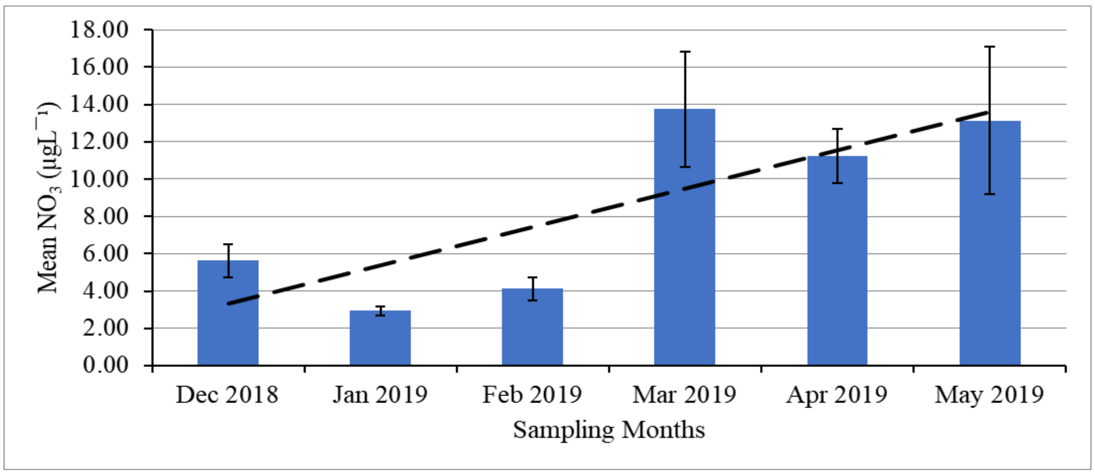

Figure 31: Mean Nitrates (NO_3-N) (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean Nitrates (NO_3-N) values between the sampled months ($F (5, 30) = 4.856$, $p = 0.002$). Further analysis by Tukey test revealed that the mean nitrates (NO_3-N) of February 2019 was significantly lower than the mean nitrates (NO_3-N) of March (by between 0.27 and 18.98) but was not significantly different from the mean nitrates (NO_3-N) values of January 2019 and May 2019. Further, the mean nitrates (NO_3-N) of January 2019 was significantly lower than the mean nitrates (NO_3-N) of both March 2019 (by between 1.45 and 20.16) and May 2019 (by between 0.85 and 19.56).

Seasonally, the mean nitrates (NO_3-N) (12.70 ± 1.65 μgL^{-1}) of the wet season (Dec 2018-Feb 2019) was higher than the mean nitrates (NO_3-N) (4.22 ± 0.43 μgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean nitrates (NO_3-N) values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t (34) = -4.960$, $p = 0.000$).

4.2.16 Nitrites-Nitrogen (NO_2-N)

On the spatial scale, the nitrites (NO_2-N) registered an overall mean of 5.35 ± 0.86 μgL^{-1} in Lake Simbi (Appendix 2). The lowest mean nitrites (NO_2-N) value (3.04 ± 0.78 μgL^{-1}) was recorded at ST5 whereas the highest (7.44 ± 3.45 μgL^{-1}) was found at ST4. The mean values exhibited a general decreasing trend over time among the stations sampled (Fig. 32).

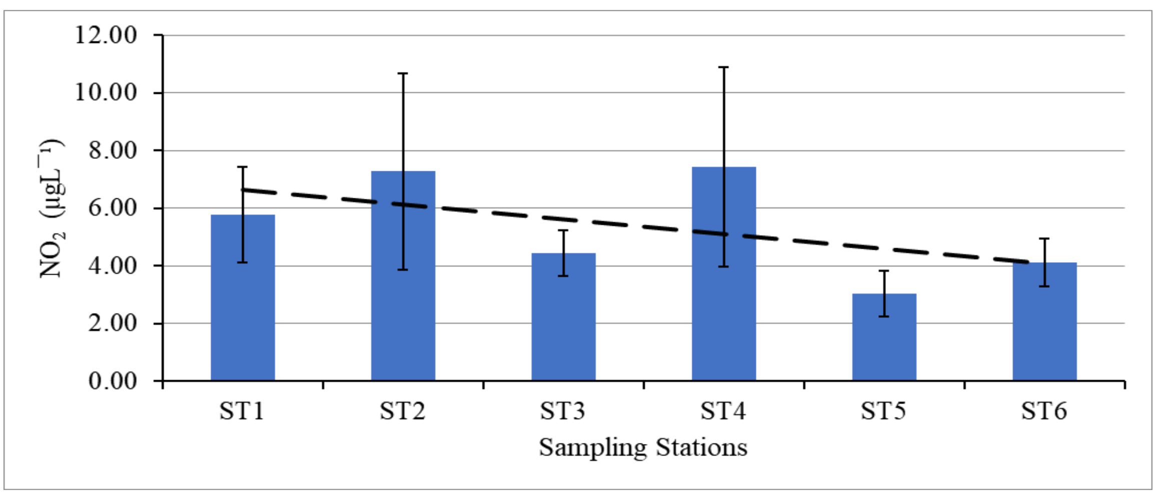

Figure 32: Mean Nitrites (NO_2-N) (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean nitrites (NO_2-N) values between the stations sampled (F (5, 30) = 0.675, p = 0.645).

On the temporal scale, the nitrites (NO_2-N) registered an overall mean of 5.35 ± 0.86 μgL^{-1} in the lake (Appendix 3). The lowest mean nitrites (NO_2-N) of 2.70 ± 0.49 μgL^{-1} was found in January 2019 while March 2019 ascribed the highest mean value of 9.20 ± 3.11 μgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 33).

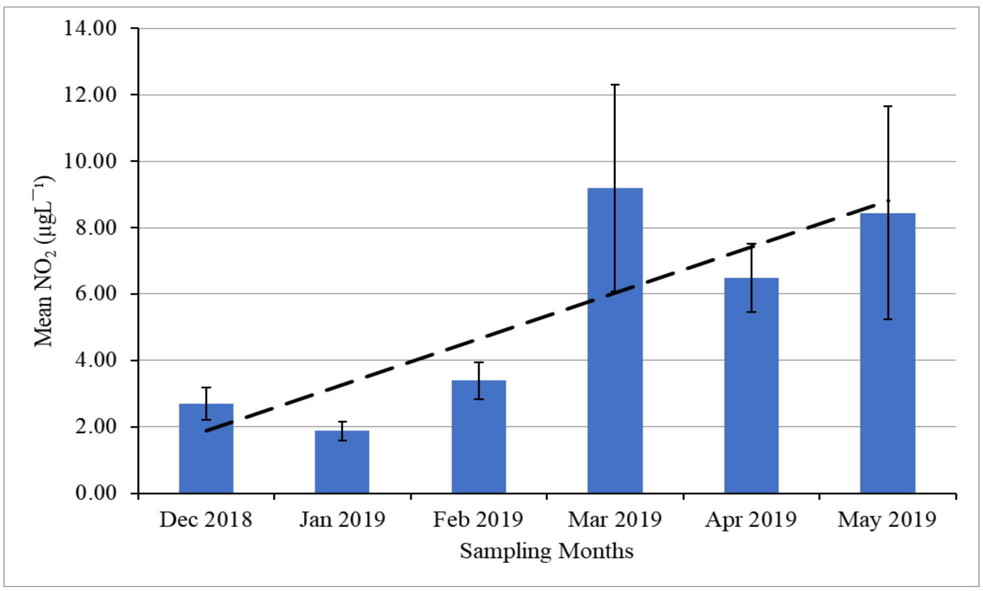

Figure 33: Mean Nitrites (NO_2-N) (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean nitrites (NO_2-N) values between the sampled months (F (5, 30) = 2.694, p = 0.400).

Seasonally, the mean nitrites (NO_2-N) (8.04 ± 1.46 μgL^{-1}) of the wet season (Dec 2018-Feb 2019) was higher than the mean nitrites (NO_2-N) (2.66 ± 0.29 μgL^{-1}) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean nitrites

(NO_2-N) values with the independent t-test showed that there was a significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -3.614$, $p = 0.001$).

4.2.17 Ammonia-Nitrogen (NH_3-N)

On the spatial scale, the ammonia (NH_3-N) registered an overall mean of 79.84 ± 40.25 µgL^{-1} in Lake Simbi (Appendix 2). The lowest mean ammonia (NH_3-N) value (29.34 ± 5.62 µgL^{-1}) was recorded at ST2 whereas the highest (271.96 ± 238.23 µgL^{-1}) was found at ST1. The mean values exhibited a general decreasing trend over time among the stations sampled (Fig. 34).

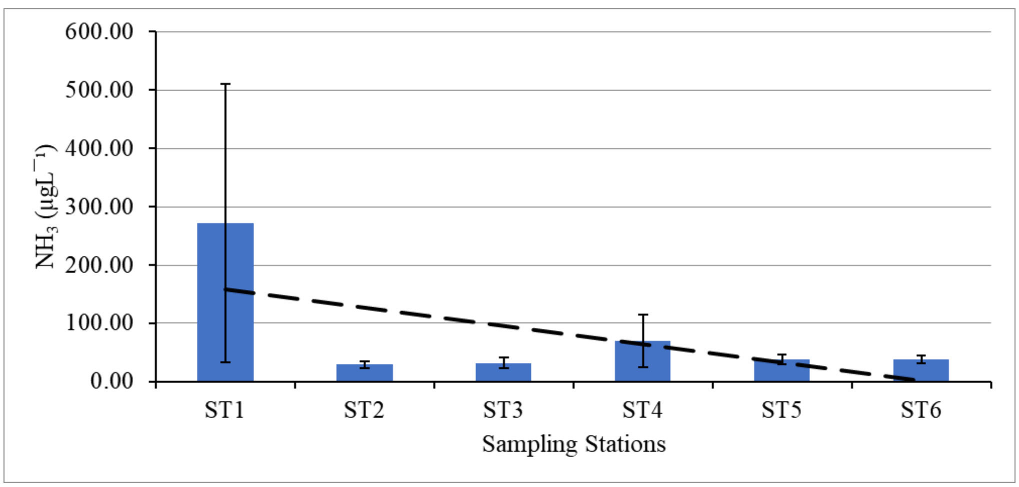

Figure 34: Mean Ammonia (NH_3-N) (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean ammonia (NH_3-N) values between the stations sampled ($F(5, 30) = 0.923$, $p = 0.480$).

On the temporal scale, the ammonia (NH_3-N) registered an overall mean of 79.84 ± 40.25 µgL^{-1} in the lake (Appendix 3). The lowest mean ammonia (NH_3-N) of 11.15 ± 2.03 µgL^{-1} was found in February 2019 while March 2019 ascribed the highest mean value of 314.27 ±

233.38 µgL¯¹. There was no clear trend exhibited by the mean values among the sampled months since most of them recorded relatively small but almost similar means except for March 2019 which registered much higher mean value (Fig. 35).

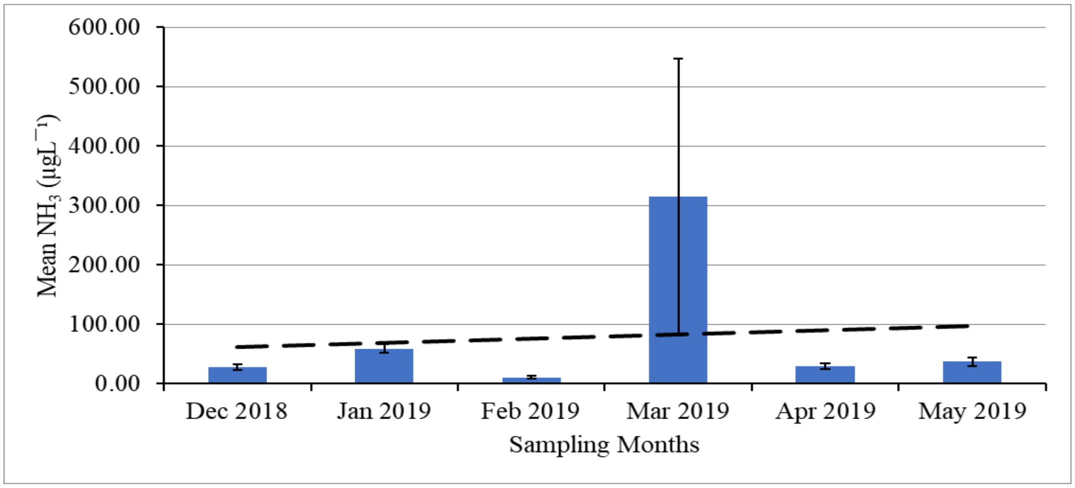

Figure 35: Mean Ammonia (NH_3-N) (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences (p<0.05) in the mean ammonia (NH_3-N) values between the sampled months (F (5, 30) = 1.476, *p* = 0.227).

Seasonally, the mean ammonia (NH_3-N) (126.80 ± 79.88 µgL¯¹) of wet season (Dec 2018-Feb 2019) was higher than the mean ammonia (NH_3-N) (32.89 ± (5.54 µgL¯¹) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean ammonia (NH3-N) values with the independent t-test showed that there was no significant difference (p<0.05) between the dry season and the wet season of the sampling period (t (34) = -1.173, p = 0.249).

4.2.18 Silicates (SiO_2)

On the spatial scale, the silicates (SiO_2) registered an overall mean of 65.93 ± 1.17 µgL^{-1} in Lake Simbi (Appendix 2). The lowest mean $_{value}$ (55.13 ± 1.52 µgL^{-1}) was recorded at ST4 whereas the highest (72.57 ± 1.84 µgL^{-1}) was found at ST6. There was no clear trend exhibited by the mean values among the stations sampled however, most stations recorded relatively similar means (Fig. 36).

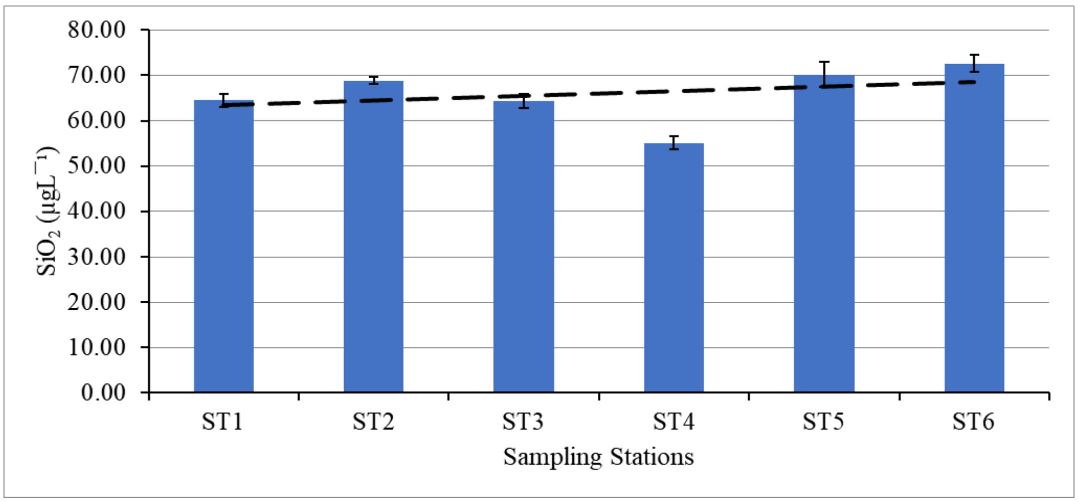

Figure 36: Mean SiO_2 (± SE) at different sampling stations

ANOVA revealed that there were significant differences ($p<0.05$) in the mean silicates (SiO_2) values between the stations sampled ($F (5, 30) = 12.274$, $p = 0.000$). Further analysis by Tukey test revealed that the mean silicates (SiO_2) of ST1 was significantly higher than the mean silicates (SiO_2) of ST4 (by between 16.98 and 1.77) and was significantly lower than the mean silicates (SiO_2) of ST6 (by between 0.46 and 15.67); but was not significantly different from the mean silicates (SiO_2) values of ST2, ST3 and ST5. On the other hand, the mean silicates (SiO_2) of ST2 was significantly higher than the mean silicates (SiO_2) of ST4 (by between 21.32 and 6.12); but was not significantly different from the mean silicates

(SiO_2) values of ST3, ST5 and ST6. Further, the mean silicates (SiO_2) of ST3 was significantly higher than the mean silicates (SiO_2) of ST4 (by between 16.81 and 1.60) and significantly lower than the mean silicates (SiO_2) of ST6 (by between 0.63 and 15.84) though it was not significantly different from the mean silicates (SiO_2) values of ST5. Finally, the mean silicates (SiO_2) of ST4 was significantly lower than the mean SiO^2 values of both ST5 (by between 7.43 and 22.64) and ST6 (by between 9.41 and 25.05).

On the temporal scale, the silicates (SiO_2) registered an overall mean of 65.93 ± 1.17 μgL^{-1} in the lake (Appendix 3). The lowest mean SiO_2 of 64.32 ± 3.07 μgL^{-1} was found in January 2019 while May 2019 ascribed the highest mean value of 67.80 ± 3.13 μgL^{-1}. The mean values exhibited a general increasing trend over time among the sampled months (Fig. 37).

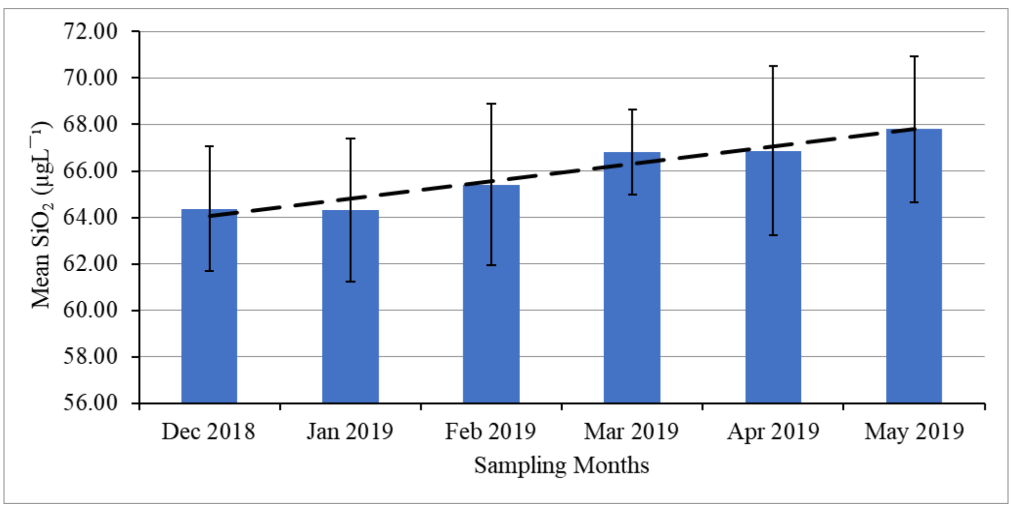

Figure 37: Mean SiO2 (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean silicates (SiO_2) values between the sampled months ($F(5, 30) = 0.228$, $p = 0.948$).

Seasonally, the mean silicates (SiO_2) (67.16 ± 1.61 µgL¯¹) of wet season (Dec 2018-Feb 2019) was higher than the mean silicates (SiO_2) (64.70 ± 1.68 µgL¯¹) of the dry season (Mar 2019-May 2019) of the sampling period (Table 9). Further comparison of the mean silicates (SiO_2) values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = -1.055$, $p = 0.299$).

4.2.19 Vertical Profiles of Some Selected Physico-chemical Parameters in Lake Simbi

The results for the vertical profiles of temperature, DO, salinity, conductivity and TDS conducted at the deepest part (depth of 27.7m) of Lake Simbi on 10th April of 2019 (which represents a transitional period between the dry season and wet seasons) are presented in the Table 12 and Fig. 38 below.

Table 12: Vertical profile for some selected physico-chemical parameters

Depth (m)	**Temperature (°C)**	**DO (mg l^{-1})**	**pH**	**Salinity (ppt)**	**Conductivity (µScm^{-1})**	**TDS (mgL¯¹)**
0	30.8	8.01	8.17	10.24	19487	11388
0.5	29.6	8	8	10.23	18990	11342
1	28.4	6.6	7.99	10.18	18452	11297
1.5	28.2	6.4	7.16	10.16	18394	11290
2	27.9	5.6	6.9	10.17	18306	11271
3	27.6	4.5	6.4	10.16	18197	11264
4	27.6	4	5.68	10.16	18127	11258
5	26.9	3.8	5.3	10.5	18418	11557
6	26.7	3.9	5.4	10.96	19062	12051
7	26.5	3.5	5	11.8	19268	12170
8	26.5	3.2	4.8	11.29	19597	12381
9	26.5	2.6	4	11.36	19714	12460
10	26.5	2	4	11.44	19849	12546

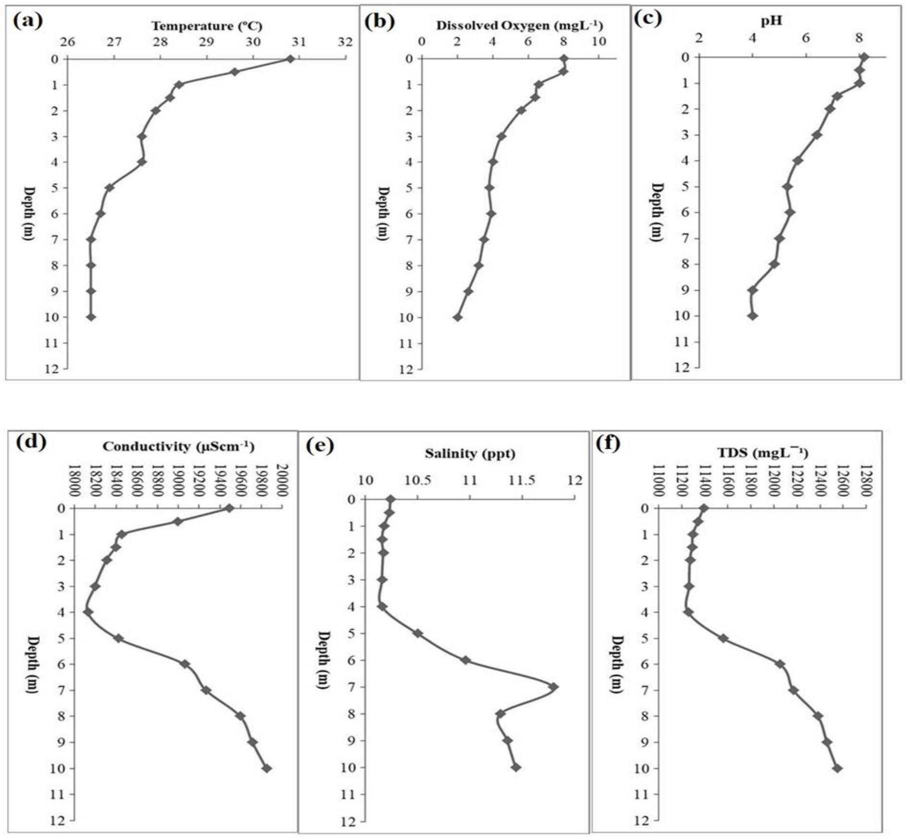

Figure 38: Vertical profile for some selected physico-chemical parameters

As a usual occurrence in most lakes found in the tropics, Lake Simbi is thermally stratified. The vertical thermal stratification in Lake Simbi illustrated in the Fig. 38(a) above indicates that a thin layer of epilimnion exists up to 1m of depth; a thick layer of metalimnion exists between 1m and 5m and a thicker hypolimnion exists between 5m and 10m. The thermocline was established to exist between 0.5m and 1m of depth.

The vertical profile of dissolved oxygen from 0m to 10m of depth carried out on 10th April 2019 in Lake Simbi is presented in Fig. 38(b) above. The range of concentration of dissolved oxygen within the measured depth was between 2 to 8.01 mg L^{-1}. The DO-Depth graph in Fig. 38(b) indicates that hypoxic conditions exist at depths below 4m while anoxic conditions exist at depths below10m. The oxycline was established at the same depth range as the thermocline. Both temperature and DO profiles showed a pattern of decreasing with increasing depth in the water column. It can be seen from Figure 38 that there exists a direct relationship between temperatures and DO since as temperature decreases with depth so do the concentration of dissolved oxygen in water.

A similar trend observed in the vertical profiles for both temperature and DO was also observed in the values of pH as illustrated in Fig. 38(c). The range of pH within the measured depth was between 4 to 8.17. The water column of Lake Simbi is stratified by pH into three distinct layers; a thin upper alkaline layer of pH range 7.16-8.17 sitting up to the depth of 1.5m of depth, followed by a thinner middle neutral layer of pH range 6.9-7.0 existing between the depth of 1.5m and 2m, and lastly a thick acidic layer existing at depths below 3m. There was a somewhat uniform variation of pH with depth of water with a pronounced chemocline observed at layers above the 4m depth.

The range of salinity within the measured depth was between 10.16 ppt to 11.80 ppt. The vertical salinity depth profile shown in Fig. 38(e) indicates the stratification of Lake Simbi based on salinity (density) where the waters above 4m of depth are less dense than the waters below it. There appears to be little variation of salinity with depth in the upper water column above 4m since the water is fairly mixed followed by an establishment of a halocline between the depth of 6m and 7m, and then little variation of salinity with depth occurs again downwards. The increasing salinity with depth generally results in a corresponding decreasing oxygen concentration with depth in Lake Simbi. This inverse relationship is clearly illustrated in the Fig. 38 above.

The vertical conductivity and TDS profiles for Lake Simbi both follow the vertical salinity profile. The Fig. 38 (d), (e) and (f) illustrates that an increase in salinity with depth generally produces a corresponding increase in both conductivity and TSD. The ranges for conductivity and TDS within the measured depth were between 18127 - 19849 μScm^{-1} and 11258 – 12546 mgL^{-1} respectively. Just like salinity, both conductivity and TDS decrease with depth on the upper water column above 4m before an establishment of the halocline after which they increase with increasing depth. However, the vertical conductivity profile showed establishment of two haloclines, the minor one corresponding to the thermocline and oxycline at the depth of between 0.5m and 1m and the major one between the depth of 6m and 7m corresponding to the one observed in salinity profile.

4.2.19 Comparison of some basic water quality parameters of Lake Simbi with NEMA and WHO standards

Parameter	Dry Season		Wet Season		Overall Sampling Period		Drinking Water Standards (Permissible Limits)	
	Range	Mean (S.E)	Range	Mean (S.E)	Range	Mean (S.E)	NEMA (2006)	WHO (2011)
Secchi Depth (m)	0.51 - 0.68	0.62 ± (0.01)^{a}	0.51 - 0.61	0.56 ± (0.01)^{b}	0.51 - 0.68	0.59 ± (0.01)	-	-
Temperature (°C)	26.60 - 30.74	28.65 ± (0.36)^{a}	28.90 - 31.50	30.13 ± (0.17)^{b}	26.60 - 31.50	29.39 ± (0.23)	-	-
Dissolved Oxygen (mgL^{-1})	3.86 - 6.06	4.49 ± (0.14)^{a}	4.35 - 8.76	6.01 ± (0.33)^{b}	3.86 - 8.76	5.25 ± (0.21)	8	7
pH	9.40 - 10.50	10.27 ± (0.06)^{a}	8.20 - 12.00	10.19 ± (0.21)^{a}	8.20 - 12.00	10.23 ± (0.11)	6.5 -8.5	6.5 -8.5
Conductivity (μScm^{-1})	15460.00 - 18713.00	16956.67 ± (244.69)^{a}	18106.00 - 19487.00	18792.83 ± (84.42)^{b}	15460.00 - 19487.00	17874.75 ± (200.88)	2500	2500
Salinity (ppt)	9.04 - 9.96	9.52 ± (0.08)^{a}	9.93 - 10.24	10.10 ± (0.025)^{b}	9.04 - 10.24	9.81 ± (0.06)	-	-
Alkalinity (mgL^{-1})	6900.00 - 10138.00	8441.28 ± (205.48)^{a}	7296.00 - 8622.00	7788.94 ± (90.73)^{b}	6900.00 - 10138.00	8115.11 ± (123.66)	500	500
Hardness (mgL^{-1})	79.00 - 104.00	88.00 ± (1.53)^{a}	180.00 - 211.00	190.44 ± (2.12)^{b}	79.00 - 211.00	139.22 ± (8.75)	500	500
TDS (mgL^{-1})	45.00 - 8441.00	2975.97 ± (881.51)^{a}	774.80 - 11388.00	7768.17 ± (1197.53)^{b}	45.00 - 11388.00	5372.07 ± (837.27)	1200	1500
Turbidity (NTU)	51.60 - 194.00	75.33 ± (8.22)^{a}	58.50 - 140.00	88.44 ± (5.76)^{a}	51.600.00 - 194.000	81.89 ± (5.07)	5	5
SRP (μgL^{-1})	1385.33 - 1950.33	1746.54 ± (34.39)^{a}	1443.67 - 2217.00	1781.01 ± (63.76)^{a}	1385.33 - 2217.00	1763.77 ± (35.83)	-	-
TP (μgL^{-1})	2034.71 - 3183.29	2567.81 ± (92.72)^{a}	2467.29 - 3590.00	3091.59 ± (88.92)^{b}	2034.71 - 3590.00	2829.70 ± (77.25)	2000	2000
TN (μgL^{-1})	60.47 - 729.95	151.07 ± (38.82)^{a}	215.21 - 1912.58	761.59 ± (98.72)^{b}	60.47 - 1912.58	456.33 ± (73.45)	2000	2000

NO_3 (μgL^{-1})	2.03 - 9.61	4.22 ± (0.43)[a]	5.06 - 28.39	12.70 ± (1.65)[b]	2.03 - 28.39	8.46 ± (1.11)	10000	10000
NO_2 (μgL^{-1})	0.42 - 5.06	2.66 ± (0.29)[a]	1.12 - 24.15	8.04 ± (1.46)[b]	0.42 - 24.15	5.35 ± (0.86)	3000	100
NH_3 (μgL^{-1})	4.06 - 75.94	32.89 ± (5.54)[a]	15.98 - 1462.19	126.80 ± (79.88)[a]	4.06 - 1462.19	79.84 ± (40.25)	500	500
SiO_2 (μgL^{-1})	52.74 - 79.30	64.70 ± (1.68)[a]	50.20 - 76.51	67.16 ± (1.61)[a]	50.20 - 79.30	65.93 ± (1.17)	-	-
Chl-a (μgL^{-1})	12.60 - 1367.68	246.79 ± (98.14)[a]	13.09 - 661.85	135.05 ± (39.65)[a]	12.60 - 1367.68	190.92 ± (53.01)	-	-

Note: Mean values in the same row that do not share a superscript letter are significantly different ($p<0.05$).

4.3 Qualitative and quantitative analysis of the of the phytoplankton community characteristics of Lake Simbi

4.3.1 Phytoplankton community structure

The phytoplankton community structure of Lake Simbi during the entire study period comprised of 84 species belonging to six major phytoplankton families (Table 13). Among these phytoplankton families, cyanophyceae was the most dominant comprising of 36 species (44%), followed by chlorophyceae (25 species, 30%), bacillariophyceae (11 species, 13%), zygnematophyceae (4 species, 5%), dinophyceae (3 species, 4%) and euglenophyceae (3 species, 4%) (Fig. 39). The most dominant species in the cyanophyceae family was *Microcystis flos-aqua* (33%); in the chlorophyceae family was *Scenedesmus obliquus* (80%); in the bacillariophyceae family was *Fragilaria pinnata* (31%); in the zygnematophyceae family was *Hyalotheca mucosa* (50%); in the dinophyceae family was *Ceratinium branchyceros* (81%) and finally in the euglenophyceae family *Euglena acus* (63%) (Appendix 4).

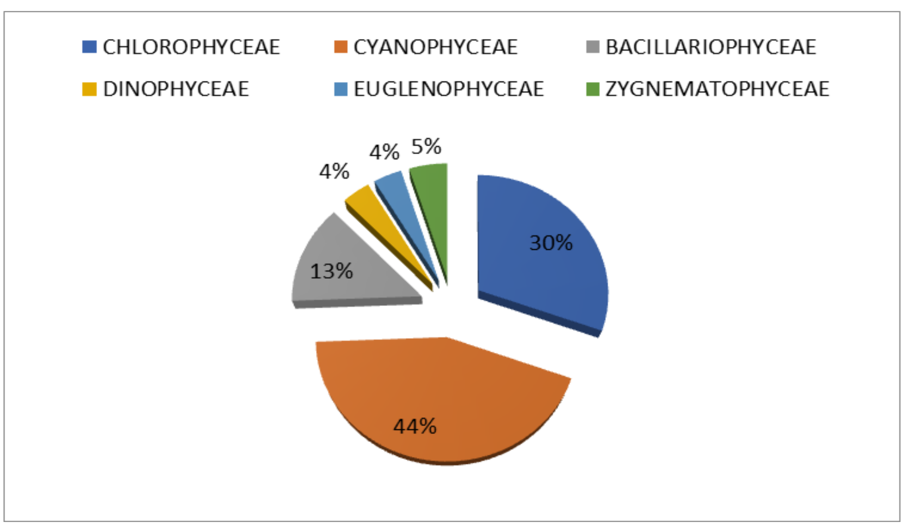

Figure 39: Phytoplankton community structure of Lake Simbi

Table 13: Species composition and distribution along the sampling stations

Family	Species	Sampling Stations					
		ST1	ST2	ST3	ST4	ST5	ST6
Chlorophyceae	*Oocystis parva*	+	-	-	-	-	-
	Pediastum boryanum	++	+	++	+++	+++	-
	Botryococcus braunii	+++	+++	++	+++	-	+++
	Monoraphidium braunii	+	++	-	-	-	-
	Chlorellla vulgaris	++	++	+++	+++	+++	+++
	Romeria elegans	-	++	+++	+	+++	-
	Pediastum duplex	-	-	++	-	-	-
	Tetraedron trigonum	-	-	++	-	-	++
	Kirchneriella obesa	-	-	++	-	-	-
	Monoraphidium caribeum	-	-	-	++	-	-
	Ankistrodesmus falcatus	-	-	-	+++	-	-
	Tetredron arthromisforme	-	-	-	-	++	-
	Coelastrum microporum	-	+	-	-	-	-
	Monoraphidium sp	+	-	-	-	-	++
	Pediastum muticum	-	-	++	-	++	-
	Scenedesmus acuminatus	+	-	-	++	-	-
	Scenedesmus obliquus	-	-	+	+++	++	+++
	Kirchnelliera lunaris	+++	+++	+++	-	++	-
	Ankistrodesmus gracilis	-	-	++	-	-	+++
	Oocystis nageli	+	-	-	-	-	-

	Crucigenia heteracantha	+	++	-	-	-	-
	Characium sp	-	++	-	-	-	-
	Kirchneriella Schmidle	-	-	++	-	-	-
	Crucigenia sp	-	+	-	-	-	-
Cyanophyceae	*Microcystis flos-aqua*	+++	-	-	-	-	+++
	Fragilaria aethiopica	+	-	-	-	-	-
	Planktolyngbya tallingii	++	-	-	-	-	-
	Planktolyngbya circumcreta	+	++	-	-	++	-
	Spirulina princeps	+++	+++	+++	++	++	+
	Chroococcus turgidus	+++	++	+++	+++	++	+++
	Oscillatoria sp	++	++	++	+++	++	++
	Oscillatoria tenuis	+++	+++	+++	++	+++	+
	Planktolyngbya limnetica	-	++	+	+++	+++	++
	Anabaena flos-aquae	-	++	-	+++	-	-
	Cylindrospermopsis africana	++	++	+	++	-	+++
	Spirulina major	+++	+++	+++	+++	+++	+++
	Spirulina gigantea	++	+++	++	+	-	+++
	Anabaena hylina	+++	+++	+++	+++	+++	+++
	Pseudo anabaena sp	+	++	-	++	-	-
	Pseudo-anabaena circularis	++	-	-	-	-	-
	Oscillatoria geminata	++	-	-	-	-	-
	Aphanocapsa rivularis	-	++	-	-	-	+++
	Spirulina laxissima	-	-	+++	-	-	+++
	Arthrospira fusiformis	+++	+++	++	+++	+++	+++
	Planktolyngbya contrata	++	-	-	-	-	-

	Oscillatoria splendida	+	-	-	-	-	-
	Anabaenopsis circularis	-	++	+++	++	++	+++
	Oscillatoria tanganyikae	-	++	-	+++	++	++
	Microcystis aeruginosa	-	++	-	+++	+++	+++
	Chroococcus dispersus	-	++	+++	+++	-	+++
	Cylindrospermopsis sp	-	-	+++	-	+	-
	Microcystis robusta	+++	+++	-	-	++	-
	Pseudo-anabaena tanganyikae	+	+++	++	++	++	+++
	Romeria ankensis	-	++	-	++	+++	+++
	Merismopedia tenuissima	-	-	-	-	+++	-
	Microcystis wasenbergii	-	-	++	++	++	-
	Anabaenopsis tanganyikae	-	++	-	+++	-	+++
	Chroococcus limneticus	+	-	-	+++	-	-
	Coelomoron sp	-	-	+++	-	+++	-
	Coelomoron vestitus	-	-	-	+++	-	-
Bacillariophyceae	*Fragilaria pinnata*	+	+++	+	-	++	+++
	Cyclotella ocellata	-	-	+	++	-	-
	Aulacoseira nyansenssis	+	-	-	-	-	-
	Nitzschia palea	-	-	-	+	-	-
	Nitzschia acicuraris	-	-	-	+	++	-
	Fragillaria virescens	-	-	++	+++	-	-
	Navicula sp	-	-	++	-	-	-
	Nitzschia sp	-	-	++	-	-	-
	Surirella sp.	++	-	-	-	-	-

	Fragilaria construens	-	-	-	-	-	-
Dinophyceae	*Ceratinium branchyceros*	-	-	-	+++	++	+++
	Glenordinium bernardii	-	-	-	++	-	-
	Ceratinium sp	-	++	-	-	-	-
Euglenophyceae	*Euglena acus*	+++	++	-	-	++	+++
	Euglena viridis	-	-	-	++	-	-
	Phacus sp	-	-	-	+++	-	-
Zygnematophyceae	*Cosmarium succisum*	-	-	-	-	+	-
	Cosmarium muticum	+	++	-	-	-	-
	Closterium abruptum	-	-	+++	-	-	-
	Hyalotheca mucosa	++	++	-	++	+	-

Key:

+++ : The phytoplankton species were dominant at the station (> 500,000 $Ind.L^{-1}$)

++ : The phytoplankton species were common at the station (< 500,000 $Ind.L^{-1}$)

\+ : The phytoplankton species were rare at the station (< 10,000 $Ind.L^{-1}$)

\- : The phytoplankton species were absent at the station (0 $Ind.L^{-1}$)

4.3.2 Spatial and temporal variation of algal biomass

Table 14: Descriptive Statistics for the spatial, temporal and seasonal variation of algal biomass of Lake Simbi

Descriptive Statistics for Chlorophyll-a concentration in Lake Simbi								
Spatial Variation			Temporal Variation			Seasonal Variation		
Station		Chl-a (μgL^{-1})	Month		Chl-a (μgL^{-1})	Season		Chl-a (μgL^{-1})
ST1	Mean	81.14 [a]	Dec 2018	Mean	49.5 [a]	Dry season	Mean	246.79 [a]
	S.E	24.94		S.E	11.56		S.E	98.14
ST2	Mean	537.35 [a]	Jan 2019	Mean	345.64 [a]	Wet season	Mean	135.05 [a]
	S.E	266.41		S.E	208.67		S.E	39.65
ST3	Mean	181.33 [a]	Feb 2019	Mean	345.24 [a]	Total	Mean	190.92
	S.E	69.11		S.E	207.09		S.E	53.01
ST4	Mean	126.15 [a]	Mar 2019	Mean	167.62 [a]			
	S.E	23.32		S.E	103.1			
ST5	Mean	164.65 [a]	Apr 2019	Mean	52.61 [a]			
	S.E	99.57		S.E	3.07			
ST6	Mean	54.92 [a]	May 2019	Mean	184.94 [a]			
	S.E	8.86		S.E	57.67			
Total	Mean	190.92	Total	Mean	190.92			
	S.E	53.01		S.E	53.01			

Note: Mean values in the same column that do not share a superscript letter are significantly different ($p<0.05$).

On the spatial scale, the Chl-a registered an overall mean of 190.92 ± 53.01 μgL^{-1} in Lake Simbi (Table 14). The lowest mean value (54.92 ± 8.86 μgL^{-1}) was recorded at ST6 whereas the highest (537.35 ± 266.41 μgL^{-1}) was found at ST2. There was no clear trend exhibited over time by the mean values among the stations sampled (Fig. 40).

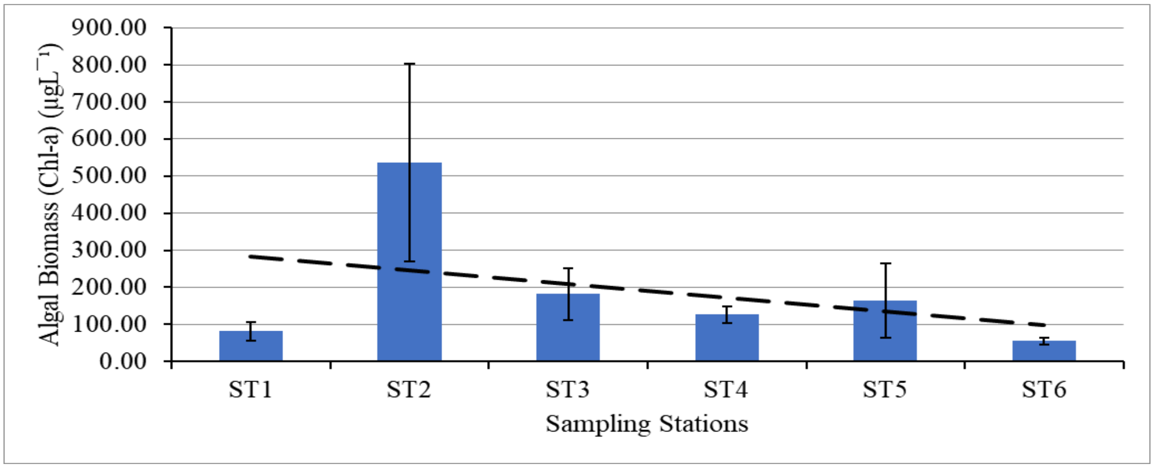

Figure 40: Mean Chl-a (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Chl-a values between the stations sampled ($F_{(5, 30)} = 2.148$, $p = 0.087$).

On the temporal scale, the Chl-a registered an overall mean of 190.92 ± 53.01 μgL^{-1} in the lake (Table 14). The lowest mean Chl-a of 49.50 ± 11.56 μgL^{-1} was found in December 2018 while January 2019 ascribed the highest mean value of 345.64 ± 208.67 μgL^{-1}. There was no clear trend exhibited over time by the mean values among the sampled months (Fig. 41).

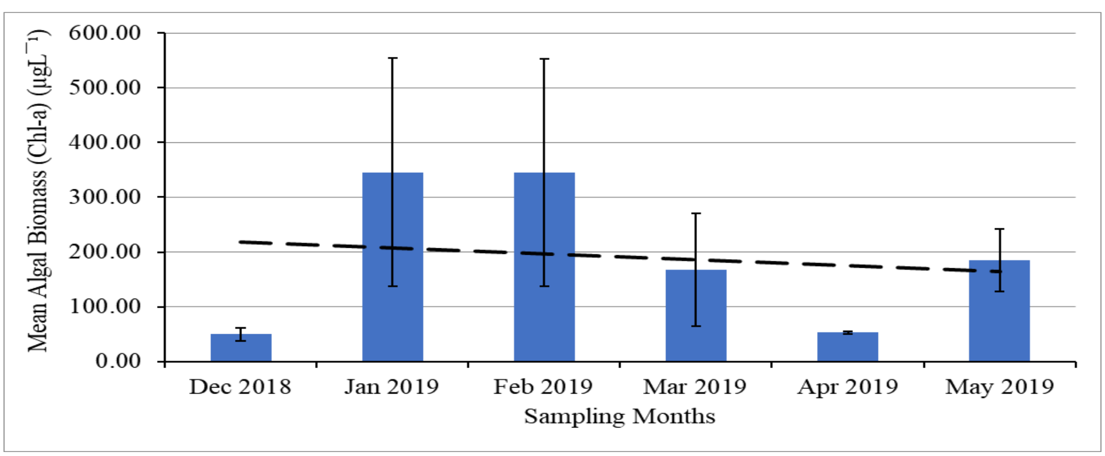

Figure 41: Mean Chl-a (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Chl-a values between the sampled months ($F (5, 30) = 1.044$, $p = 0.410$).

Seasonally, the mean Chl-a ($246.79 \pm 98.14\ \mu gL^{-1}$) of dry season (Dec 2018-Feb 2019) was higher than the mean Chl-a ($135.05 \pm 39.65\ \mu gL^{-1}$) of the wet season (Mar 2019-May 2019) of the sampling period (Table 14). Further comparison of the mean Chl-a values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t (34) = 1.056$, $p = 0.299$).

4.3.3 Spatial and temporal variation of phytoplankton abundance

Table 15: Descriptive Statistics for the spatial, temporal and seasonal variation of phytoplankton abundance of Lake Simbi

Descriptive Statistics for phytoplankton abundance in Lake Simbi										
Spatial Variation				Temporal Variation				Season Variation		
Month		Abundance ($Ind.L^{-1} \times 10^{3}$)		Month		Abundance ($Ind.L^{-1} \times 10^{3}$)		Season		Abundance ($Ind.L^{-1} \times 10^{3}$)
ST1	Mean	61993.82 [a]		Dec-18	Mean	80749.19 [a]		Dry	Mean	73056.68 [a]
	S.E	25001.52			S.E	31569.52			S.E	30703.71
ST2	Mean	52416.14 [a]		Jan-19	Mean	124798.75 [a]		Wet	Mean	27818.24 [a]
	S.E	30362.64			S.E	85724.07			S.E	9239.67
ST3	Mean	15505.40 [a]		Feb-19	Mean	13622.09 [a]		Total	Mean	50437.46
	S.E	7824.18			S.E	4085.64			S.E	16257.21
ST4	Mean	23327.54 [a]		Mar-19	Mean	31599.68 [a]				
	S.E	13442.59			S.E	17371.38				
ST5	Mean	48549.51 [a]		Apr-19	Mean	28444.16 [a]				
	S.E	27414.54			S.E	16492.51				
ST6	Mean	100832.33 [a]		May-19	Mean	23410.88 [a]				
	S.E	87366.17			S.E	17035.21				
Total	Mean	50437.46		Total	Mean	50437.46				
	S.E	16257.21			S.E	16257.21				

Note: Mean values in the same column that do not share a superscript letter are significantly different ($p<0.05$).

On the spatial scale, the phytoplankton abundance registered an overall mean of 50437.46 ± 16257.21 (Ind.L^{-1} x 10^3) in Lake Simbi (Table 15). The lowest mean phytoplankton abundance value (15505.40 ± 7824.18 Ind.L^{-1} x 10^3) was recorded at ST3 whereas the highest (100832.33 ± 87366.17 Ind.L^{-1} x 10^3) was found at ST6. The mean generally had a slightly increasing trend over time among the stations sampled (Fig. 42).

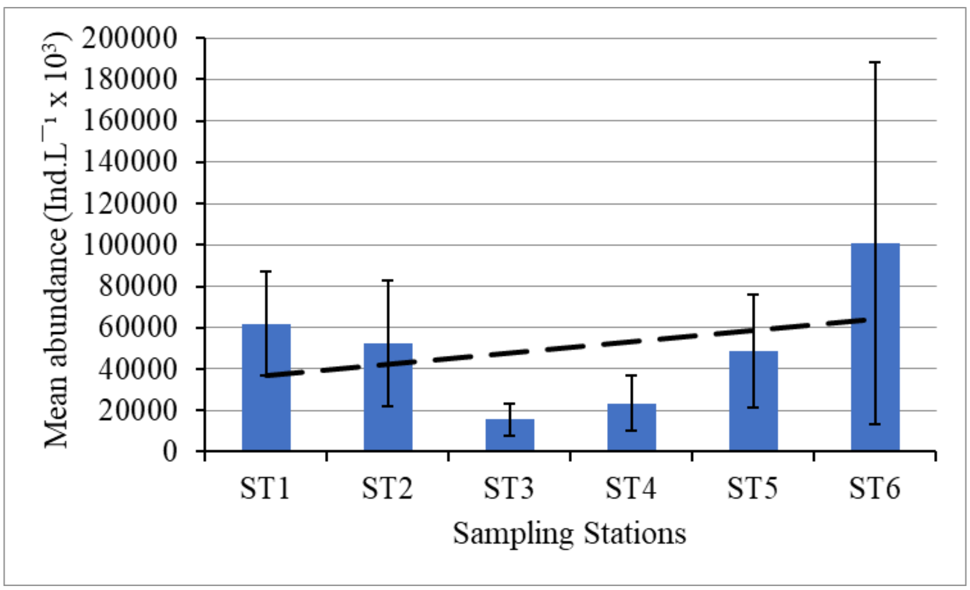

Figure 42: Mean phytoplankton abundance (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean phytoplankton abundance values between the stations sampled ($F(5, 30) = 0.547$, $p = 0.739$).

On the temporal scale, the phytoplankton abundance registered an overall mean of 50437.46 ± 16257.21 (Ind.L^{-1} x 10^3) in the lake (Table 15). The lowest mean phytoplankton abundance of 13622.09 ± 4085.64 (Ind.L^{-1} x 10^3) was found in February 2019 while January

2019 ascribed the highest mean value of 124798.75 ± 85724.07 (Ind.L^{-1} x 10^3). The mean generally had a decreasing trend over time during the sampled months (Fig. 43). ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean phytoplankton abundance values between the sampled months (F (5, 30) = 1.219, p = 0.324).

Seasonally, the mean phytoplankton abundance (73056.68 ± 30703.71 Ind.L^{-1} x 10^3) of dry season (Dec 2018-Feb 2019) was higher than the mean phytoplankton abundance (27818.24 ± 9239.67 Ind.L^{-1} x 10^3) of the wet season (Mar 2019-May 2019) of the sampling period (Table 15). Further comparison of the mean phytoplankton abundance values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = 1.411, p = 0.167).

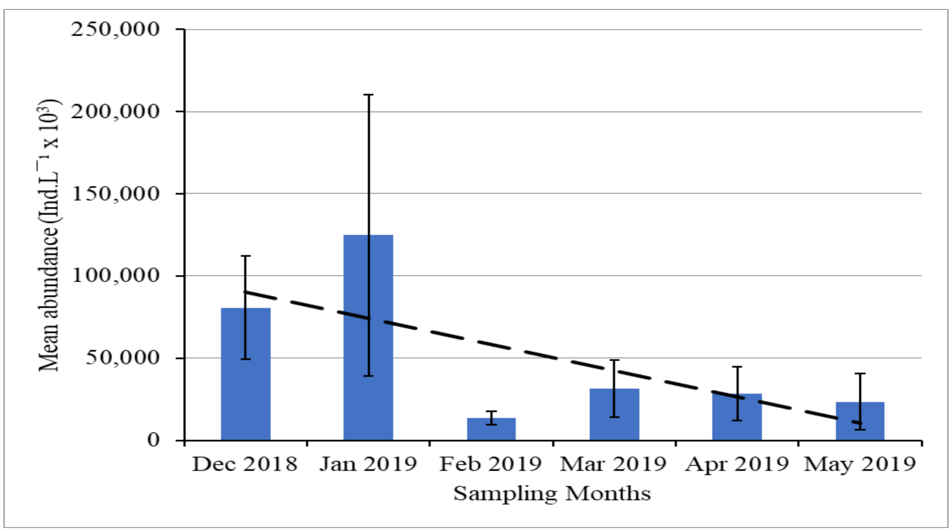

Figure 43: Mean phytoplankton abundance (± SE) in different sampling months

4.3.4 Spatial and temporal variation of the relative abundance (percentage composition) of the phytoplankton families in Lake Simbi

On a spatial scale, the relative abundance of cyanophyceae was the highest in all sampling stations contributing between 74 - 97%, followed by chlorophyceae contributing between 1.50 - 24.77% and the rest (bacillariophyceae, dinophyceae, euglenophyceae and zygnematophyceae) each contributing below 1.30% to the total relative abundance for each of the sampled stations (Fig. 44).

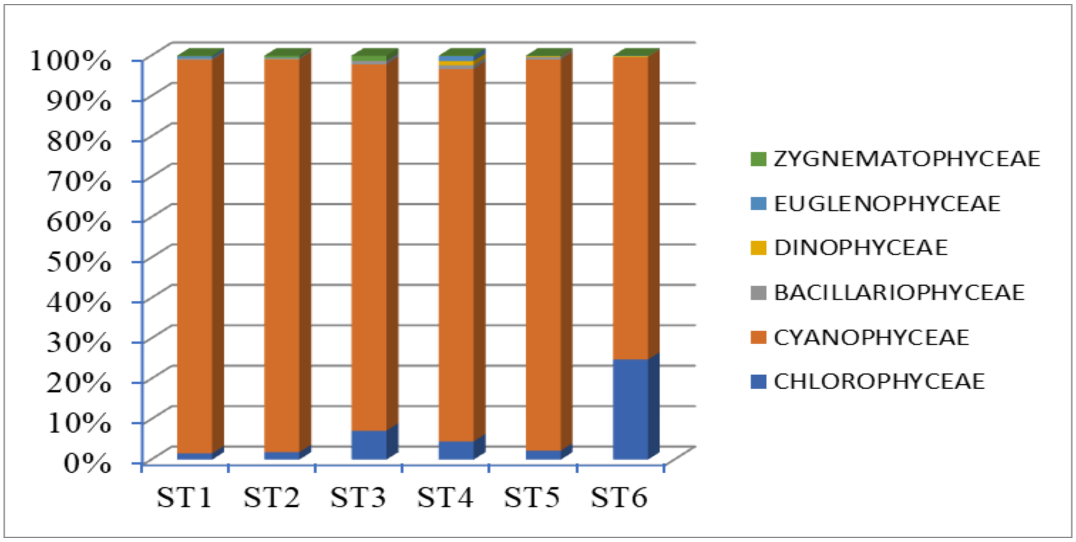

Figure 44: Phytoplankton relative abundance (percentage composition) at different sampling stations

On the temporal scale, the relative abundance of cyanophyceae was the highest in all the months sampling contributing between 77.98 - 97.43%, followed by chlorophyceae contributing between 1.37 - 21.33% and the rest (bacillariophyceae, dinophyceae, euglenophyceae and zygnematophyceae) each contributing below 1.57% to the total relative abundance for each of the months sampled (Fig. 45).

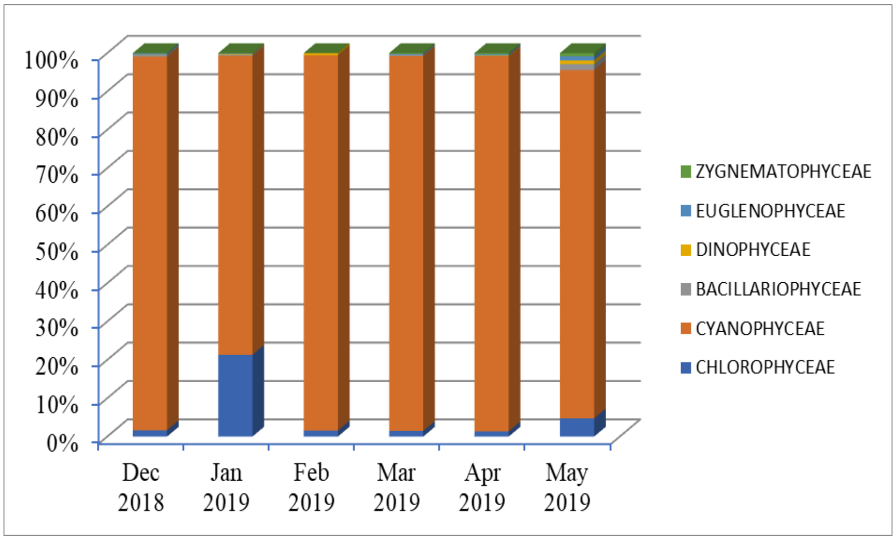

Figure 45: Phytoplankton relative abundance (percentage composition) in different sampling months

Seasonally, the relative abundance for chlorophyceae, cyanophyceae, bacillariophyceae and dinophyceae families were higher during the dry season than the wet season. On the other hand, the euglenophyceae and zygnematophyceae families both had their relative abundances for the wet season being higher than the relative abundances for the dry season (Fig. 46).

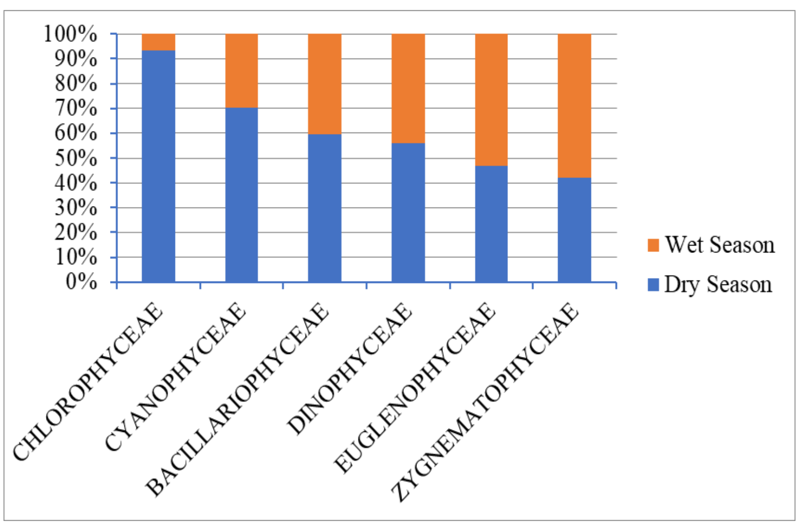

Figure 46: Phytoplankton relative abundance (percentage composition) in different sampling seasons

4.3.4 The mean phytoplankton species diversity indices variations

Table 16: Descriptive Statistics for the spatial and temporal variation of phytoplankton diversity indices of Lake Simbi

Descriptive Statistics for phytoplankton diversity indices in Lake Simbi

Spatial Variation

Month		Richness Index	Shannon Index	Evenness Index	Simpson Index
ST1	Mean	0.58 [a]	1 [a]	0.42 [a]	0.59 [a]
	S.E	0.05	0.31	0.13	0.13
ST2	Mean	0.55 [a]	1.3 [a]	0.58 [a]	0.42 [a]
	S.E	0.07	0.32	0.15	0.12
ST3	Mean	0.55 [a]	1.61 [a]	0.71 [a]	0.29 [a]
	S.E	0.05	0.15	0.06	0.05
ST4	Mean	0.57 [a]	1.42 [a]	0.61 [a]	0.39 [a]
	S.E	0.09	0.27	0.11	0.11
ST5	Mean	0.45 [a]	1.01 [a]	0.49 [a]	0.52 [a]
	S.E	0.05	0.24	0.12	0.1
ST6	Mean	0.4 [a]	1.39 [a]	0.69 [a]	0.37 [a]
	S.E	0.05	0.18	0.08	0.08
Total	Mean	0.52	1.29	0.59	0.43
	S.E	0.03	0.1	0.05	0.04

Temporal Variation

Month		Richness Index	Shannon Index	Evenness Index	Simpson Index
Dec 2018	Mean	0.48 [a]	0.91 [a]	0.42 [a]	0.55 [a]
	S.E	0.06	0.25	0.12	0.1
Jan 2019	Mean	0.68 [a]	1.5 [a]	0.59 [a]	0.37 [a]
	S.E	0.07	0.32	0.12	0.12
Feb 2019	Mean	0.43 [a]	1.28 [a]	0.62 [a]	0.43 [a]
	S.E	0.07	0.19	0.07	0.08
Mar 2019	Mean	0.5 [a]	1.18 [a]	0.55 [a]	0.48 [a]
	S.E	0.05	0.27	0.13	0.13
Apr-19	Mean	0.5 [a]	1.36 [a]	0.63 [a]	0.4 [a]
	S.E	0.06	0.25	0.11	0.1
May 2019	Mean	0.49 [a]	1.48 [a]	0.7 [a]	0.34 [a]
	S.E	0.05	0.24	0.11	0.11
Total	Mean	0.52	1.29	0.59	0.43
	S.E	0.03	0.1	0.05	0.04

Note: Mean values in the same column that do not share a superscript letter are significantly different ($p<0.05$).

Table 17: Descriptive Statistics for the seasonal variation of phytoplankton diversity indices of Lake Simbi

Seasonal Variation					
Seasonal		Richness Index	Shannon Index	Evenness Index	Simpson Index
Dry	Mean	0.53 [a]	1.23 [a]	0.54 [a]	0.45 [a]
	S.E	0.04	0.15	0.06	0.06
Wet	Mean	0.5 [a]	1.34 [a]	0.63 [a]	0.41 [a]
	S.E	0.03	0.14	0.07	0.06
Total	Mean	0.52	1.29	0.59	0.43
	S.E	0.03	0.1	0.05	0.04

4.3.4.1 Margalef's Species Richness Index

On the spatial scale, the species richness index registered an overall mean of 0.52 ± 0.03 in Lake Simbi (Table 16). The lowest mean species richness index value (0.4 ± 0.05) was recorded at ST6 whereas the highest (0.58 ± 0.05) was found at ST1. The mean had a general decreasing trend over time among the stations sampled (Fig. 47).

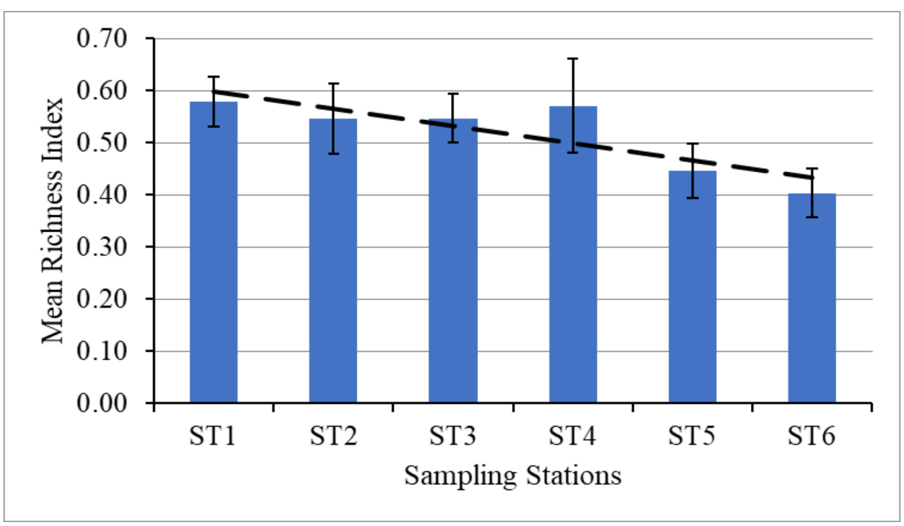

Figure 47: Mean species richness index (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean species richness index values between the stations sampled ($F (5, 30) = 1.443$, $p = 0.038$).

On the temporal scale, the species richness index registered an overall mean of 0.52 ± 0.03 in the lake (Table 16). The lowest mean species richness index of 0.43 ± 0.07 was found in February 2019 while January 2019 ascribed the highest mean value of 0.68 ± 0.07. There was no clear trend exhibited over time by the mean values among the sampled months since most months had relatively similar means (Fig. 48).

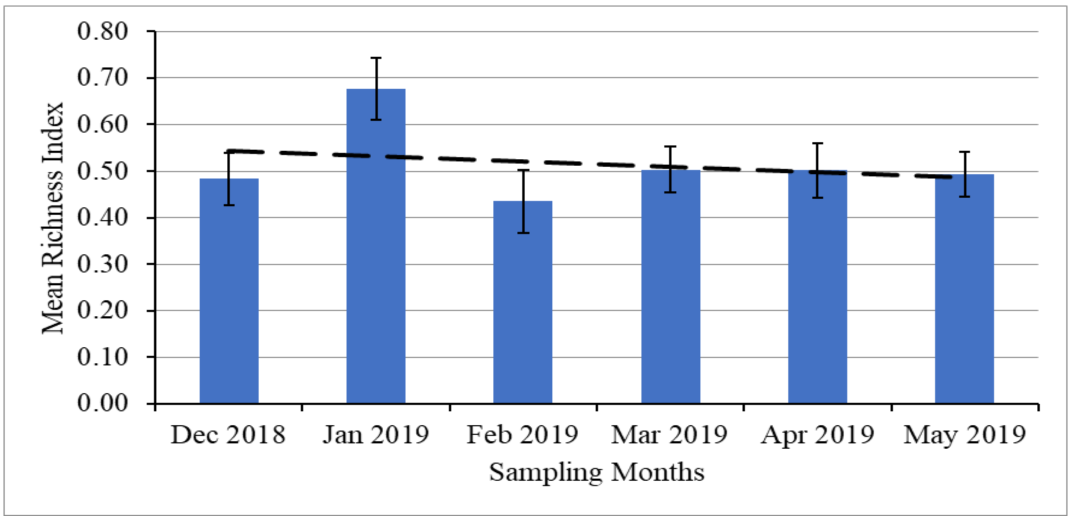

Figure 48: Mean species richness index (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean species richness index values between the sampled months ($F (5, 30) = 2.040$, $p = 0.101$).

Seasonally, the mean species richness index (0.53 ± 0.04) of dry season (Dec 2018-Feb 2019) was higher than the mean species richness index (0.5 ± 0.03) of the wet season (Mar 2019-May 2019) of the sampling period (Table 17). Further comparison of the mean species richness index values with the independent t-test showed that there was no significant difference (p<0.05) between the dry season and the wet season of the sampling period (t (34) = 0.632, p = 0.532).

4.3.4.2 Shannon-Wiener's Species Diversity Index

On the spatial scale, the Shannon-Wiener's Species Diversity Index registered an overall mean of 1.29 ± 0.01 in Lake Simbi (Table 16). The lowest mean Shannon-Wiener's Species diversity Index value (1 ± 0.31) was recorded at ST1 whereas the highest (1.61± 0.15) was found at ST3. There was no clear trend exhibited over time by the mean values among the stations sampled (Fig. 49).

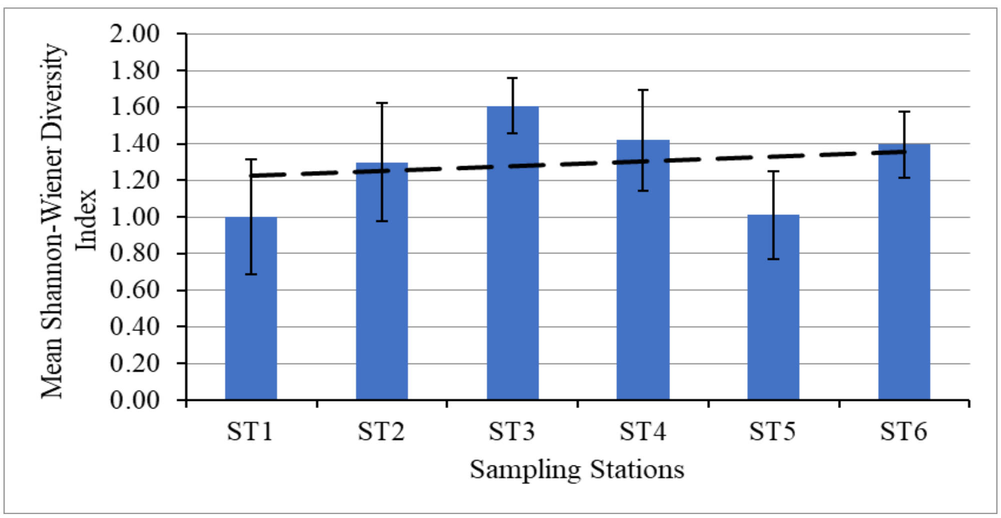

Figure 49: Mean Shannon-Wiener's Species diversity Index (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences (p<0.05) in the mean Shannon-Wiener's Species diversity index values between the stations sampled (F (5, 30) = 0.889, *p* = 0.501).

On the temporal scale, the Shannon-Wiener's species diversity index registered an overall mean of 1.29 ± 0.01 in the lake (Table 16). The lowest mean Shannon-Wiener's species diversity index of 0.91 ± 0.25 was found in December 2018 while January 2019 ascribed the highest mean value of 1.5 ± 0.32. The mean had a general increasing trend over time among the sampled months (Fig. 50).

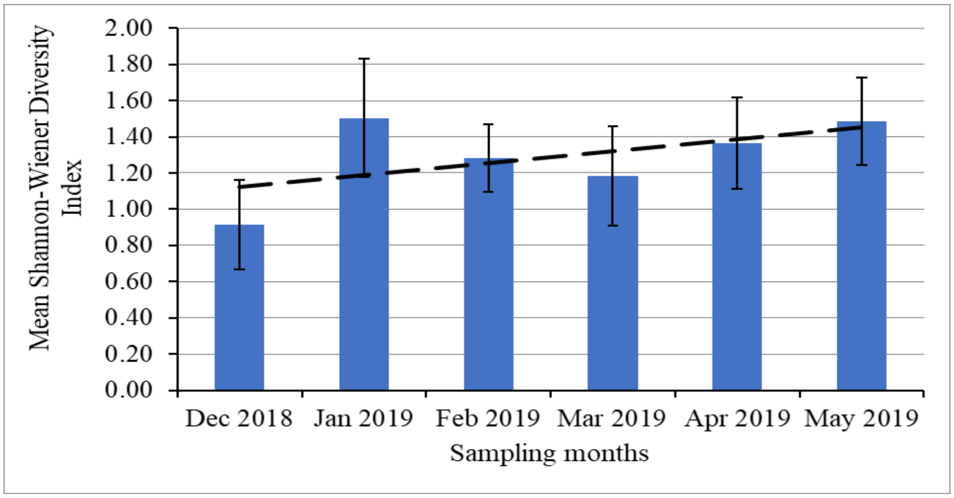

Figure 50: Mean Shannon-Wiener's Species diversity Index (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Shannon-Wiener's species diversity index values between the sampled months (F (5, 30) = 0.729, p = 0.607).

Seasonally, the mean Shannon-Wiener's species diversity index (1.34 ± 0.14) of the wet season (Dec 2018-Feb 2019) was higher than the mean Shannon-Wiener's species diversity index (1.23 ± 0.15) of the dry season (Mar 2019-May 2019) of the sampling period (Table 17). Further comparison of the mean Shannon-Wiener's species diversity index values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = -0.526, p = 0.602).

4.3.4.3 Pielou's Species Evenness Index

On the spatial scale, the species evenness index registered an overall mean of 0.59 ± 0.05 in Lake Simbi (Table 16). The lowest mean species evenness index value (0.42 ± 0.13) was recorded at ST1 whereas the highest (0.71 ± 0.06) was found at ST3. The mean had a general increasing trend over time among the stations sampled (Fig. 51).

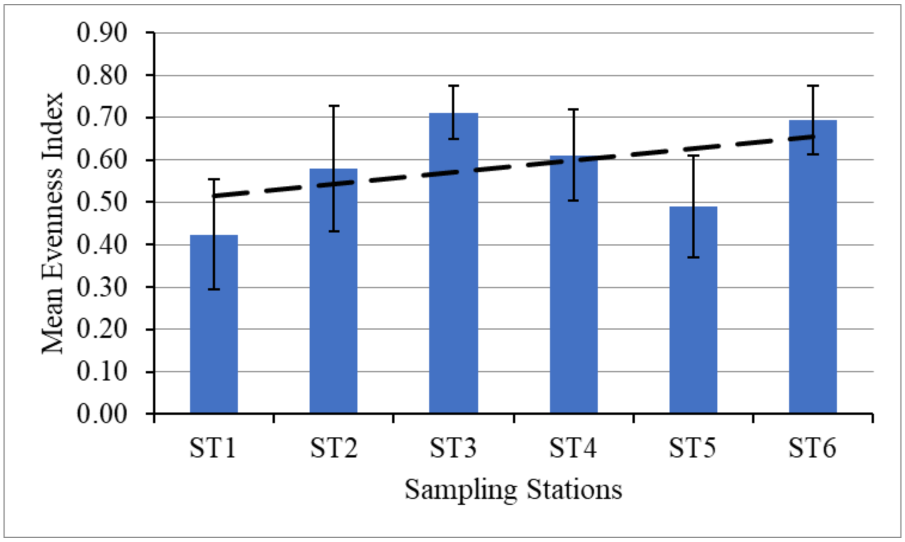

Figure 51: Mean species evenness index (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean species evenness index values between the stations sampled ($F(5, 30) = 1.015$, $p = 0.426$).

On the temporal scale, the species evenness index registered an overall mean of 0.59 ± 0.05 in the lake (Table 16). The lowest mean species evenness index of 0.42 ± 0.12 was found in December 2018 while May 2019 ascribed the highest mean value of 0.7 ± 0.11. The mean had a general increasing trend over time among the sampled months (Fig. 52).

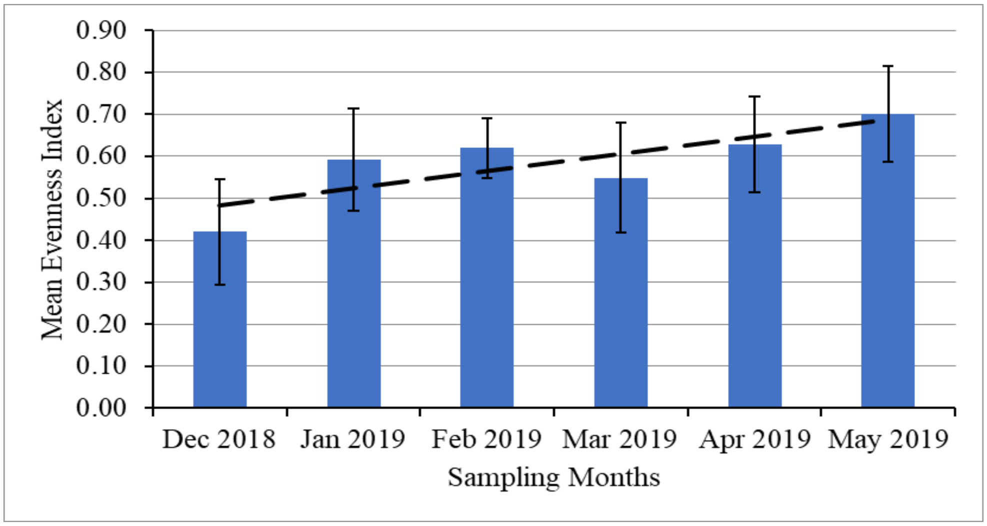

Figure 52: Mean species evenness index (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean species evenness index values between the sampled months ($F\ (5, 30) = 0.686$, $p = 0.638$).

Seasonally, the mean species evenness index (0.63 ± 0.07) of the wet season (Dec 2018-Feb 2019) was higher than the mean species evenness index (0.54 ± 0.06) of the dry season (Mar 2019-May 2019) of the sampling period (Table 17). Further comparison of the mean species evenness index values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t\ (34) = -0.894$, $p = 0.377$).

4.3.4.4 Simpson's Species Diversity index

On the spatial scale, the Simpson's species diversity index registered an overall mean of 0.43 ± 0.04 in Lake Simbi (Table 16). The lowest mean Simpson's species diversity index value (0.29 ± 0.05) was recorded at ST3 whereas the highest (0.59 ± 0.13) was found at ST1. The mean had a slightly decreasing trend over time among the stations sampled (Fig. 53).

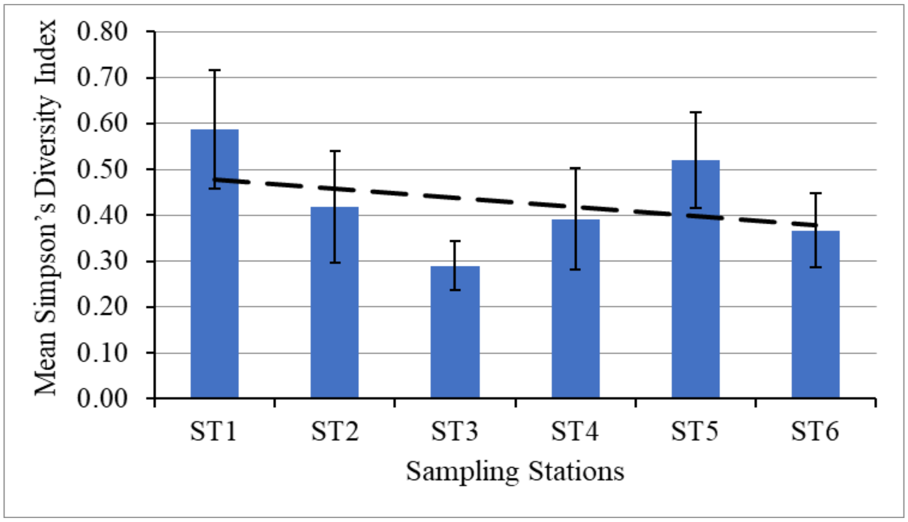

Figure 53: Mean Simpson's species diversity index (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Simpson's species diversity index values between the stations sampled ($F_{(5, 30)} = 1.085$, $p = 0.389$).

On the temporal scale, the Simpson's species diversity index registered an overall mean of 0.43 ± 0.04 in the lake (Table 16). The lowest mean Simpson's species diversity index of 0.34 ± 0.11 was found in May 2019 while December 2018 ascribed the highest mean value of 0.55 ± 0.1. The mean had a general decreasing trend over time among the sampled months (Fig. 54).

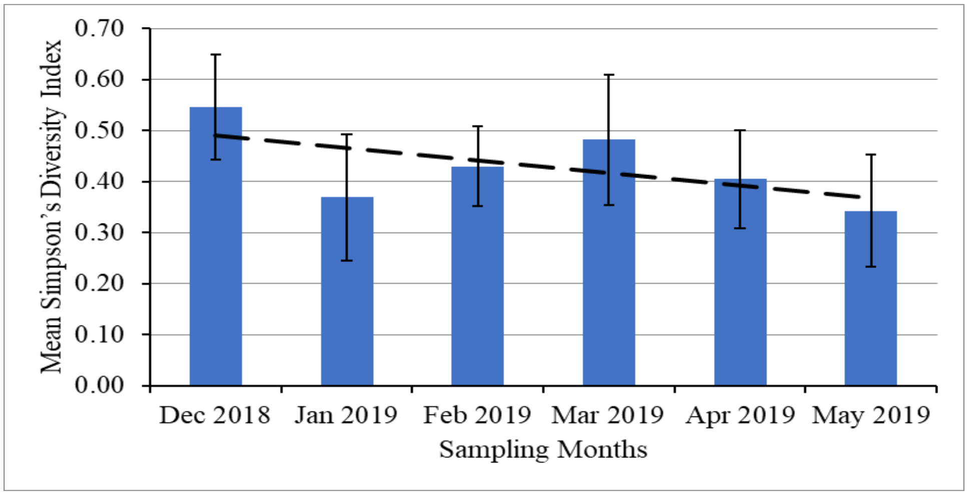

Figure 54: Mean Simpson's species diversity index (± SE) in different sampling months

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean Simpson's species diversity index values between the sampled months ($F(5, 30) = 0.488$, $p = 0.782$).

Seasonally, the mean Simpson's species diversity index (0.45 ± 0.06) of the dry season (Dec 2018-Feb 2019) was higher than the mean Simpson's species diversity index (0.41 ± 0.06) of the dry season (Mar 2019-May 2019) of the sampling period (Table 17). Further comparison of the mean Simpson's species diversity index values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period ($t(34) = 0.450$, $p = 0.656$).

4.3.5 The Nygaard's Trophic state Indices for the phytoplankton quotients of Lake Simbi

Table 18: No. of species found from each phytoplankton family per station

FAMILY	SPECIES IDENTIFIED PER STATIONS					
	ST 1	ST 2	ST 3	ST 4	ST 5	ST 6
Bacillariophyceae	3	1	5	4	2	1
Chlorophyceae	10	10	12	9	7	7
Cyanophyceae (Myxophycea)	21	22	14	22	19	18
Dinophyceae	0	1	0	2	1	1
Euglenophyceae	1	1	0	2	1	1
Zygnematophyceae (Desmidaceae)	2	2	1	1	2	0

Nygaard's phytoplankton quotients were calculated as biological indicators of pollution (Table 19). This was based on the species distribution among the sampling stations for the various phytoplankton families observed in Lake Simbi (Table 18). The Myxophycean index recorded in Lake Simbi ranged from 0 – 22 which classifies the lake as eutrophic. The Chlorophycean index ranged between 0 – 12 also classifying Lake Simbi as eutrophic. The euglenophycean index on the other hand recorded between 0 – 0.06 indicating that the water is oligotrophic differing from the other two indices. But the overall index known as compound co-efficient generally recorded a higher range of 0 – 37 effectively and definitively classifying the lake as eutrophic. In fact, it could be hyper-eutrophic because it is way past the range of 0.01 – 2.5 proposed by Nygaard (1949). The hyper-eutrophic status though is not part of Nygaard's trophic status classification so based on this consideration, the lake is just eutrophic.

Table 19: Nygaard's Trophic state Indices for the phytoplankton quotients in Lake Simbi

Nyagaard's Index	Trophic Status Indices	ST1	ST2	ST3	ST4	ST5	ST6	Range	Conclusion
Myxophycean	Oligotrophic (0.0 – 0.4) Eutrophic (0.1 – 3.0)	10.5	11	14	22	9.5	0	0 - 22	Eutrophic
Chlorophycean	Oligotrophic (0.0 – 0.7) Eutrophic (0.2 – 9.0)	5	5	12	9	3.5	0	0 - 12	Eutrophic
Euglenophycean	Oligotrophic (0.0 – 0.7) Eutrophic (0.0 -1.0)	0.03	0.03	0	0.06	0.04	0.04	0 - 0.06	Oligotrophic
Compound Co-efficient	Oligotrophic (< 0.01) Eutrophic (0.01- 2.5)	17.5	17	31	37	14.5	0	0 - 37	Eutrophic

4.4 Spatial and temporal variation of the Trophic State Index (TSI) characteristics of Lake Simbi

Table 20: Mean Trophic State Indicators (± SE) at different sampling stations

Spatial Variation of the Trophic State Indicators of Lake Simbi					
Month		TSI (SD)	TSI (Chl-a)	TSI (TP)	CTSI
ST1	Mean ± SE	66.68 ± 0.33 ^{b}	72.00 ± 2.44 ^{a}	118.37 ± 1.12 ^{a}	85.68 ± 1.01 ^{a}
ST2	Mean ± SE	66.77 ± 0.46 ^{b}	81.75 ± 7.31 ^{a}	117.97 ± 1.24 ^{a}	88.83 ± 2.29 ^{a}
ST3	Mean ± SE	67.50 ± 0.33 ^{ab}	74.08 ± 6.44 ^{a}	118.52 ± 1.02 ^{a}	86.70 ± 2.35 ^{a}
ST4	Mean ± SE	68.76 ± 0.31 ^{a}	77.08 ± 2.03 ^{a}	118.55 ± 1.01 ^{a}	88.13 ± 0.67 ^{a}
ST5	Mean ± SE	68.45 ± 0.26 ^{a}	75.20 ± 3.92 ^{a}	119.58 ± 0.70 ^{a}	87.74 ± 1.25 ^{a}
ST6	Mean ± SE	67.42 ± 0.35 ^{ab}	69.04 ± 1.96 ^{a}	118.39 ± 1.11 ^{a}	84.95 ± 0.93 ^{a}
Total	Mean ± SE	67.60 ± 0.19	74.86 ± 1.85	118.56 ± 0.40	87.01 ± 0.63

Table 21: State Indicators (± SE) in different sampling months

Temporal Variation of the Trophic State Indicators of Lake Simbi					
Month		TSI (SD)	TSI (Chl-a)	TSI (TP)	CTSI
Dec 2018	Mean ± SE	66.62 ± 0.35 ^{c}	67.80 ± 2.50 ^{a}	115.36 ± 0.45 ^{c}	83.26 ± 0.99 ^{b}
Jan 2019	Mean ± SE	67.00 ± 0.37 ^{bc}	82.81 ± 5.50 ^{a}	117.94 ± 1.07 ^{b}	89.25 ± 1.62 ^{ab}
Feb 2019	Mean ± SE	67.12 ± 0.48 ^{abc}	81.12 ± 4.79 ^{a}	118.72 ± 0.63 ^{b}	88.99 ± 1.55 ^{a}
Mar 2019	Mean ± SE	68.50 ± 0.30 ^{a}	71.34 ± 6.18 ^{a}	118.15 ± 0.41 ^{b}	86.00 ± 2.11 ^{ab}
Apr 2019	Mean ± SE	68.30 ± 0.39 ^{ab}	69.39 ± 0.59 ^{a}	121.74 ± 0.18 ^{a}	86.47 ± 0.20 ^{ab}
May 2019	Mean ± SE	68.11 ± 0.27 ^{abc}	79.20 ± 3.23 ^{a}	119.91 ± 0.58 ^{ab}	89.07 ± 0.94 ^{a}
Total	Mean ± SE	67.60 ± 0.19	74.86 ± 1.85	118.56 ± 0.40	87.01 ± 0.63

Note: Mean values in the same column that do not share a superscript letter are significantly different ($p<0.05$).

Table 22: Mean Trophic State Indicators (± SE) at different sampling seasons

Seasonal Variation of the Trophic State Indicators of Lake Simbi					
Season		TSI (SD)	TSI (Chl-a)	TSI (TP)	CTSI
Dry	Mean ± SE	66.89 ± 0.28 [a]	76.41 ± 4.08 [a]	117.19 ± 1.79 [a]	86.83 ± 2.38 [a]
Wet	Mean ± SE	68.31 ± 0.03 [a]	73.30 ± 2.89 [a]	119.93 ± 0.15 [a]	87.18 ± 0.11 [a]

4.4.1 Trophic State Indicators

On the spatial scale, the TSI (SD) registered an overall mean value of 67.60 ± 0.19, with the lowest (66.62 ± 0.35) and highest (68.50 ± 0.30) means recorded in December 2018 and March 2019 respectively; the TSI (Chl-a) registered an overall mean value of 74.86 ± 1.85, with the lowest (72.00 ± 2.44) and highest (81.75 ± 7.31) means recorded in ST1 and ST2 respectively; the TSI (TP) registered an overall mean value of 118.56 ± 0.40, with the lowest (117.97 ± 1.24) and highest (119.58 ± 0.70) means recorded in ST2 and ST5 respectively (Table 20, Figure 55).

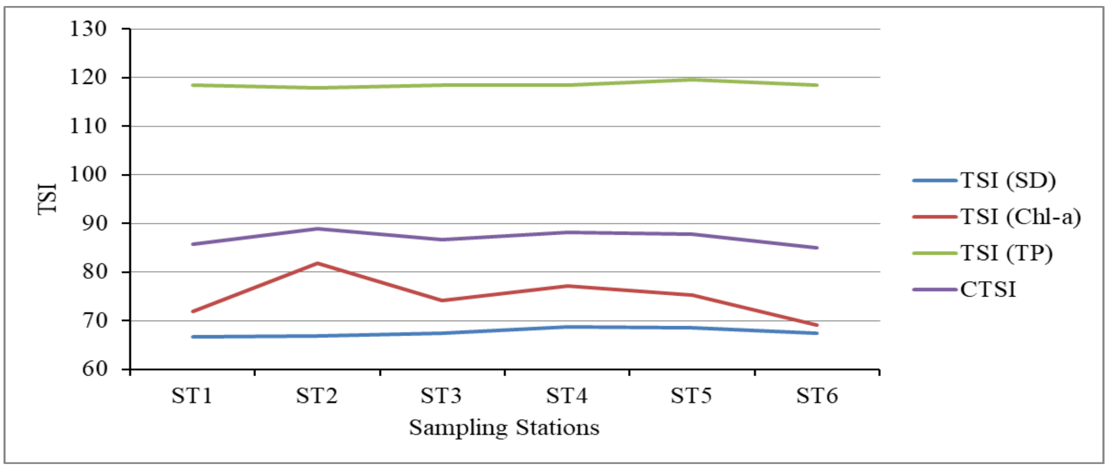

Figure 55: Mean TSI indices (± SE) at different sampling stations

On the temporal scale, the TSI (SD) registered an overall mean value of 67.60 ± 0.19, with the lowest (66.68 ± 0.33) and highest (68.76 ± 0.31) means recorded in ST1 and ST4 respectively; the TSI (Chl-a) registered an overall mean value of 74.86 ± 1.85, with the lowest (67.80 ± 2.50) and highest (82.81 ± 5.50) means recorded in December 2018 and January respectively; the TSI (TP) registered an overall mean value of 118.56 ± 0.40, with the lowest (115.36 ± 0.45) and highest (121.74 ± 0.18) means recorded in December 2018 and April 2019 respectively (Table 21, Figure 56).

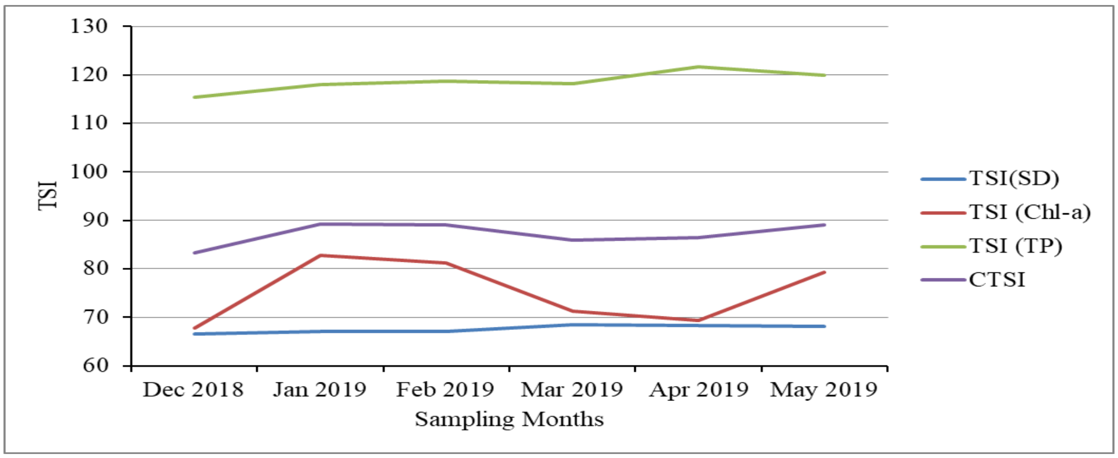

Figure 56: Mean TSI indices (± SE) at different sampling months

All the four indicators exhibited relatively slight differences between the dry season and the wet season with most of the indicators registering almost similar values for both seasons (Fig. 57). The wet season recorded higher values than the dry season for the indicators of TSI (SD), TSI (TP) and CTSI leaving only the TSI (Chl-a) recording higher values during the dry season than the wet season (Table 22, Fig. 57).

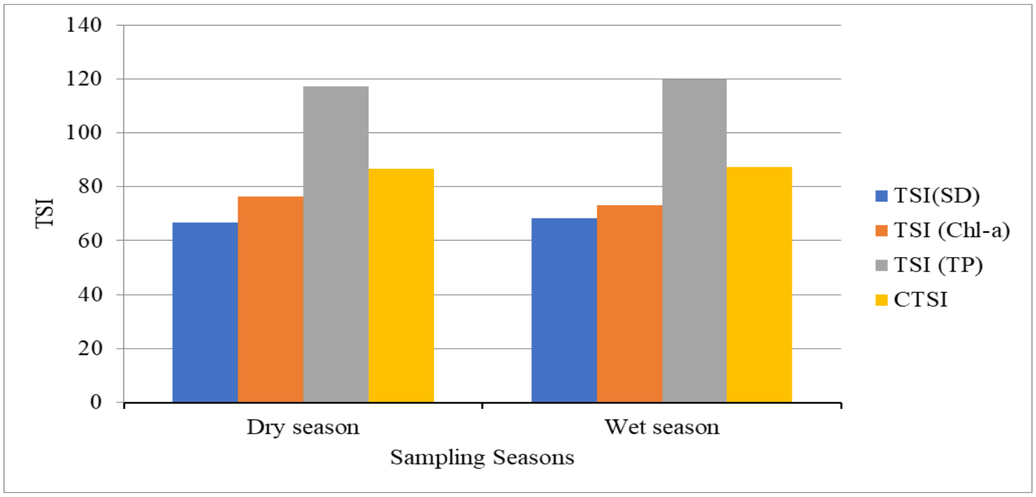

Figure 57: Mean TSI indices (± SE) at different seasons of the sampling period

Among all the three TSI indicators (TSI-SD, TSI-Chl-a and TSI-TP) together with the average overall indicator (CTSI) measured during the sampling period in Lake Simbi, the TSI-SD generally recorded the lowest values and the TSI-TP generally recorded the highest values on the TSI scale. The range of TSI (SD) value classifies the lake in the eutrophic category, the range of TSI (Chl-a) value classifies the in the hyper-eutrophic category, and the range of TSI (TP) classifies the lake in the hypereutrophic category on the TSI scale (Table 20). The general trophic status therefore ranged from eutrophic to hyper-eutrophic (Fig. 55 and Fig. 56).

Table 23: Carlson's Trophic State Index Classification

Trophic Index	Chlorophyll a	Total phosphorus	Secchi depth	Trophic class
<30 – 40	0 - 2.6	0 – 12	>8 – 4	Oligotrophic
40 – 50	2.6 – 7.3	12 – 24	4 – 2	Mesotrophic
50 – 70	7.3 – 56	24 – 96	2 – 0.5	Eutrophic
70 – 100	56 – 155 +	96 – 384 +	0.5 - < 0.25	Hypereutrophic

4.4.2 Carlson's Trophic State Index (CTSI) of Lake Simbi

On the spatial scale, the CTSI registered an overall mean of 87.01 ± 0.63 in Lake Simbi which effectively classifies the lake as hyper-eutrophic based on the CTSI scale (Table 16). The lowest mean CTSI value (84.95 ± 0.93) was recorded at ST6 whereas the highest (88.83 ± 2.29) was found at ST2. There was no clear trend exhibited over time by the mean values among the stations sampled since most stations had relatively similar means indicating hyper-eutrophic status (Fig. 58).

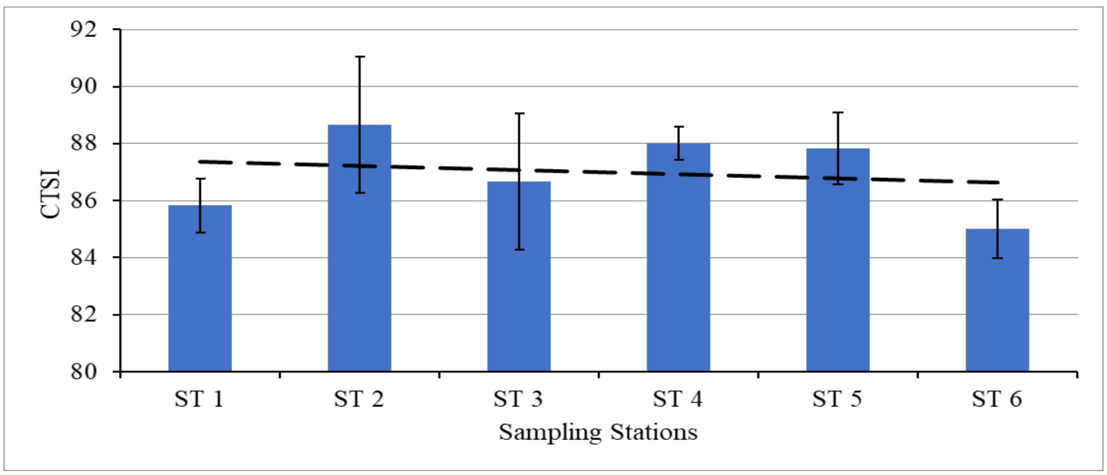

Figure 58: Mean CTSI (± SE) at different sampling stations

ANOVA however, revealed that there were no significant differences ($p<0.05$) in the mean CTSI values between the stations sampled (F (5, 30) = 0.916, p = 0.484).

On the temporal scale, the CTSI registered an overall mean of 87.01 ± 0.63 in the lake which effectively classifies the lake as hyper-eutrophic based on the CTSI scale (Table 16). The lowest mean CTSI of 82.98 ± 1.12 was found in December 2018 while May 2019 ascribed the highest mean value of 89.07 ± 0.94. The mean values exhibited a general increasing hyper-eutrophic trend over time among the sampled months (Fig. 59).

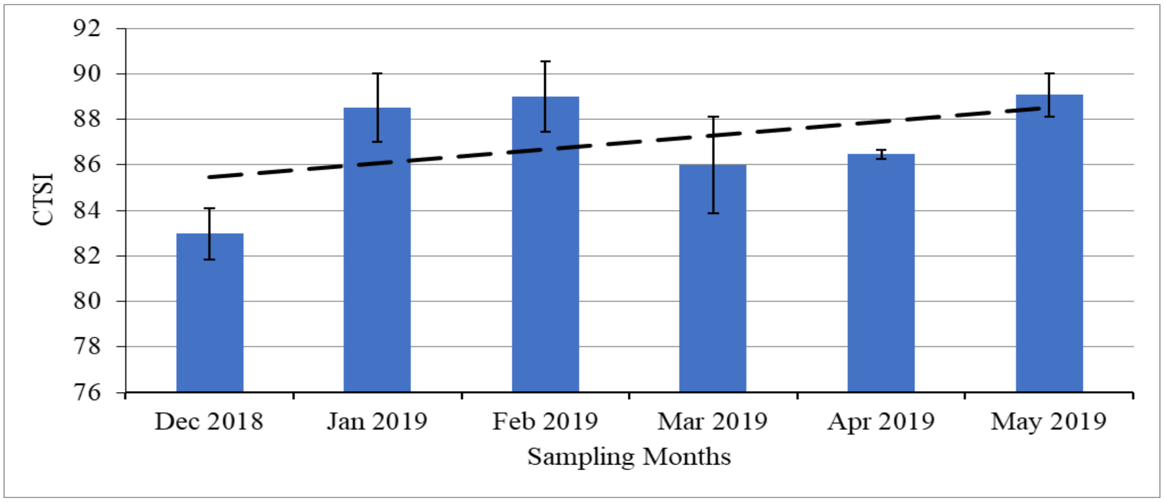

Figure 59: Mean CTSI (± SE) in different sampling months

ANOVA revealed that there were significant differences ($p<0.05$) in the mean CTSI values between the sampled months (F (5, 30) = 2.97, *p* = 0.027). Further analysis by Tukey test revealed that the mean CTSI of December 2018 was significantly lower than the mean CTSI of both February 2019 (by between 0.10 and 11.92) and May 2019 (by between 0.19 and 12.00); but was not significantly different from the mean CTSI of January 2019 and March 2019. Seasonally, the mean CTSI (87.18 ± 0.11) of the wet season (Dec 2018-Feb 2019) was higher than the mean CTSI (86.83 ± 0.38) of the dry season (Mar 2019-May 2019) of the sampling period in Lake Simbi however, generally the trophic status of the lake during both seasons was indicated as hyper-eutrophic by the CTSI scale (Table 17). Further comparison of the mean CTSI values with the independent t-test showed that there was no significant difference ($p<0.05$) between the dry season and the wet season of the sampling period (t (34) = -0.270, p = 0.789).

4.5 Lake Habitat Quality Survey (LHS) characteristics of Lake Simbi

The summarized data indicating the various anthropogenic pressures and non-natural land-uses documented in each Hab-plot of Lake Simbi is given in the Table 24. Each Hab-plot consists of three zones i.e. the littoral zone (an area which covers about 10m into the lake from the waterline), exposed shore zone (an area of variable width bound between the edge of the bank and the waterline, may be present or absent) and the riparian zone (covers an area of about 15m outwards from the banks of the shore). These various zones are shown in the pictures of Hab-plots D and F (Figure 60).

Hab-plot B was recorded as containing the highest number of various anthropogenic pressures (10) occurring within its entire length (0-50m), followed by Hab-plot C (9), then Hab-plots A, D, I and J (7), then Hab-plots E, F and G (5) and finally Hab-plot H which recorded the least number of pressures (4). In all the Hab-plots, the anthropogenic pressures observed stretched landwards from 15m to 50m. All Hab-plots used in Lake Simbi LHS study had more than 3 pressures occurring within both the 0-15m band and >15 – 50m band. This is an indication that when pressures occur, they often most probably stretch landwards into the riparian zones. From these observations, the study concluded that the shoreline of Lake Simbi is experiencing an intense pressure from various competing anthropogenic activities as well as non-natural land uses as illustrated by GIS remote sensed imagery in Fig. 2. Some of the most widespread pressures include farming, development of residential homes, deforestation and erosion in riparian areas (Fig. 2 and 61).

The Table 25 presents a summary of the Lake Simbi shoreline land cover types recorded for both perimeter bands of each Hab-plots (i.e. 0m-15m & >15m – 50m). The greatest diversity in terms of the land cover types was recorded in Hab-plots A, C and E (each had 4 various types), followed by D (had 3 types), then B and F (each had 2 various types) and the rest (Hab-plots G, H, I and J) recorded only 1 type of land cover. The natural land cover along the shoreline habitat for Lake Simbi was established to be predominantly consisting of scrub and shrubs (with coverage ranging between 10% and 40% of almost every Hab-plot) interspersed by few tall trees with the rest of the shoreline under bare exposed ground as illustrated in Fig. 2 and 62.

Table 26 presents a summary of the substrate characteristics recorded for each Hab-plot (A-J). The shoreline substrate material was predominantly silt and clay which was observed in all the Hab-plots with coverage of above 40%. Sand was the second most dominant substrate material in the shoreline of Lake Simbi. It was recorded in almost all Hab-plots, with coverage of less than 10 % observed in Hab-plots A, B, E and G, and coverage of between 10 - 40% in Hab-plots F, H and J. Only 4 Hab-plots (F, G, H and I) recorded bedrock as part of their substrate characteristics with areal coverage of less than 10%.

The physical habitat of Lake Simbi comprised of mainly scrubs and shrub vegetation as seen in the remote sensed image in Fig. 1. The predominant species included *Acacia tortilis, Balanites aegyptiaca, Combretum molle,* Senna siamea and *Striga hermonthica* (Striga weed). The Lake's habitat is also having small coverage of sisal, Aloe vera and cactus plants in some hill slopes along the shores.

Figure 60: Images of Hab-plots D (picture a) and F (picture b) depicting the zones of various Hab-plots; Littoral zone, Shore zone and the Riparian zone. Hab-plot D has all the three zones while Hab-plot F has only two zones.

Figure 61: An image of the whole lake (picture a), residential homes sitting on the riparian zone between Hab-plot C and D (picture b) and boating activities on the waters of Lake Simbi (picture c)

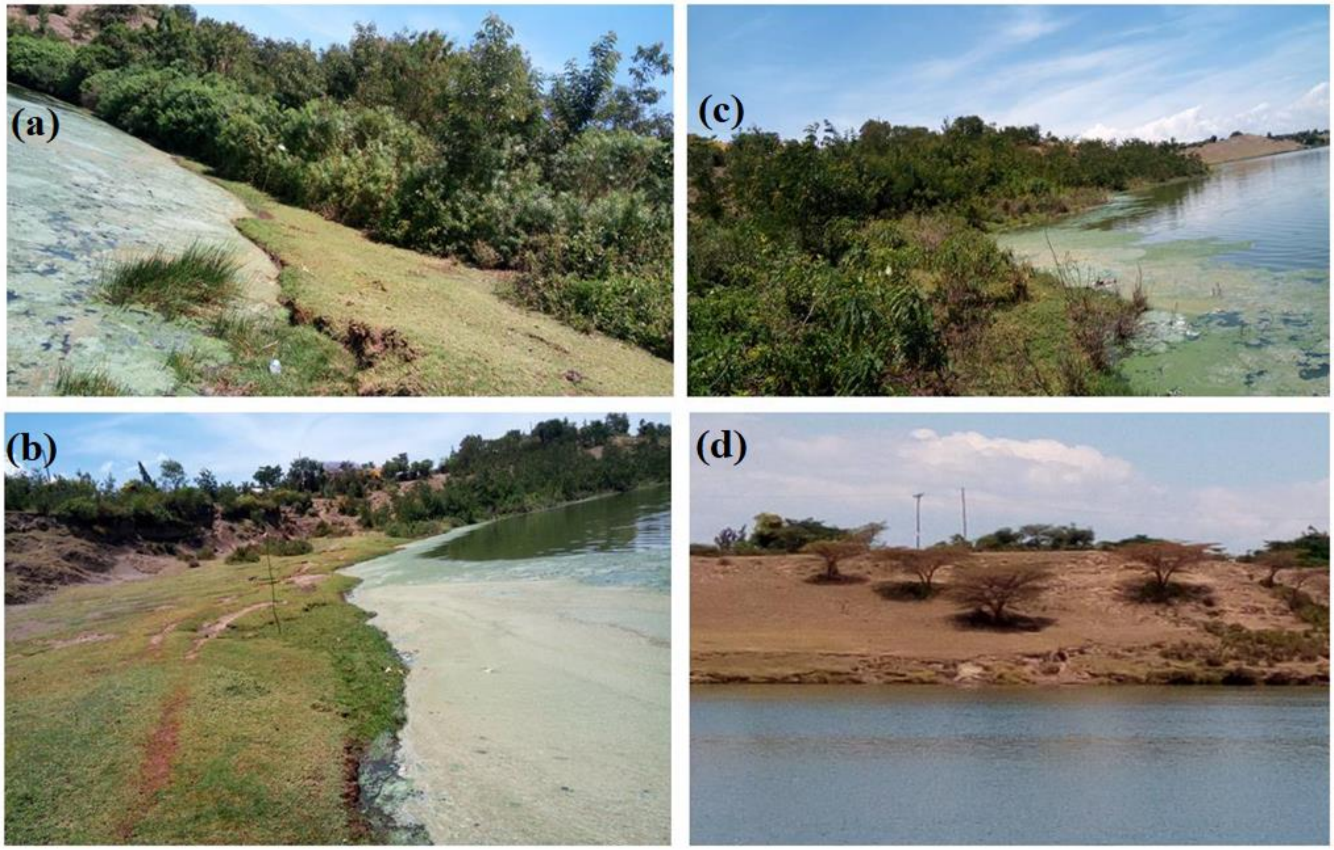

Figure 62: Bushy shrubs touching the waterline on the riparian zone and algal scum floating on the littoral zone of Hab-plot A (picture a). Hab-plot B having highly eroded banks and grassy shores (picture b). Dense vegetation of majorly Kassod trees on the riparian zone and algal scum in littoral zone of Hab-plot C (picture c). Riparian zone characterized by few sparsely distributed Acacia trees on a bare ground and boulders on the bank face of Hab-plot I (picture d).

Table 24: Summarized data for shoreline anthropogenic pressures recorded within the 15m and between 15m to 50m for each Hab-Plot in Lake Simbi LHS study (April 2019) expressed as extent of the entire perimeter of the lake.

Anthropogenic pressures and non-natural land-use	**Hab-Plots**																			
	A		**B**		**C**		**D**		**E**		**F**		**G**		**H**		**I**		**J**	
	15	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**
Commercial activities	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Residential areas	0	2	0	2	0	2	0	2	0	2	0	2	0	2	0	2	0	2	0	2
Roads or railways	1	1	0	1	1	1	0	1	0	1	0	1	0	1	0	1	0	1	0	1
Parks and gardens	0	2	0	1	0	2	0	2	0	1	0	2	0	1	0	0	0	1	0	2
Recreational beaches	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Educational activities	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Litter, dump, landfill	1	1	1	1	1	1	1	0	1	0	1	0	1	0	1	0	1	1	1	1
Quarrying or mining	1	0	1	1	1	1	1	1	0	0	0	0	0	0	0	0	1	0	1	0
Coniferous plantation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Evidence recent logging	2	0	1	0	0	0	1	1	1	1	0	0	0	0	0	0	0	0	0	0
Pasture	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Observed grazing	1	0	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Tilled land	0	2	0	2	0	2	0	2	0	0	0	2	0	2	0	2	0	2	0	2
Orchard	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Erosion	2	1	2	1	2	1	2	0	1	0	1	0	2	0	1	0	1	1	0	1
Number of pressures	**6**	**7**	**5**	**10**	**5**	**9**	**5**	**7**	**4**	**5**	**3**	**5**	**3**	**5**	**3**	**4**	**4**	**7**	**3**	**7**

Note:

(i) 15 represent an area of between 0m to 15 m while 50 represent an area of between 15m to 50 m of the Hab-Plot.

(ii) 0, 1, 2, 3 and 4 represents areal coverage of between (0 – 1 %), (>1 – 10 %), (> 10 – 40 %), (> 40 – 75 %) and (> 75 %) respectively.

Table 25: Summarized data for shoreline habitat land cover types recorded within the 15m and between 15m to 50m for each Hab-Plot in Lake Simbi LHS study (April 2019) expressed as extent of the entire perimeter of the lake.

Lake Habitat Land Cover Types	**Hab-Plots**																			
	A		**B**		**C**		**D**		**E**		**F**		**G**		**H**		**I**		**J**	
	15	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**	**15**	**50**
Broadleaf/Mixed woodland	1	2	0	1	2	2	2	1	0	1	0	0	0	0	0	0	0	0	0	0
Broadleaf/Mixed plantation	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Coniferous woodland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Scrub and shrubs	1	3	0	0	1	3	0	2	0	3	2	1	0	0	0	0	3	0	0	2
Moorland/heath	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Open water	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Rough grassland	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tall herb/rank vegetation	1	1	0	1	1	1	1	1	0	1	0	1	0	1	0	0	0	0	0	0
Rock, scree or dunes	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Fringing reed banks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Wet woodlands	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Alders	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bogs	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Quaking banks	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Other (e.g. fen, marsh)	2	1	0	0	2	1	0	0	2	1	0	0	0	0	2	1	0	0	0	0
Extent of the predominant cover	**4**	**4**	**0**	**2**	**4**	**4**	**2**	**3**	**1**	**4**	**1**	**2**	**0**	**1**	**1**	**1**	**1**	**0**	**0**	**1**
Diversity of land cover types	**4**		**2**		**4**		**3**		**4**		**2**		**1**		**1**		**1**		**1**	

Table 26 : Summarized data for predominant shoreline substrate characteristics found in the different Hab-Plots (A-J) for the littoral zones in Lake Simbi during the LHS study (April 2019).

Predominant shore-forming material in Lake Simbi										
Note: 0, 1, 2, 3 and 4 represents areal coverages of between (0 – 1 %), (>1 – 10 %), (> 10 – 40 %), (> 40 – 75 %) and (> 75 %) respectively.										
Littoral substrate texture	**Hab-Plot**									
	A	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**
Bedrock	0	0	0	0	0	1	1	1	1	0
Boulders (>256 mm)	0	0	0	0	0	1	1	1	0	0
Cobbles (>64–256 mm)	0	0	0	0	0	2	0	0	0	0
Pebbles (>2–64 mm)	0	0	0	0	1	0	0	1	0	1
Sand (>0.063–2 mm)	1	1	0	0	1	2	1	2	0	2
Silt/clay (<0.063 mm)	4	4	4	4	4	3	3	3	4	3

Table 27: Features and Scores for Lake Habitat Quality Assessment (LHQA) for Lake Simbi

Zone	Measurable LHS feature	Counts of features across lake, or number of Hab-Plots with a feature	Score allocated
Riparian	Complex or simple veg.	7	3
	> 10% large trees	4	2
	Natural/semi natural veg.	8	3
	No. natural types	3	3
	No. bank top features	2	2
Shore	Earth/sand bank	4	1
	Trash line	4	2
	Natural bank material	5	2
	No. natural types	4	4
	Natural beach material	7	3
	No. natural types	2	2
Littoral	Coefficient variation	0	0
	Natural littoral substrate	9	4
	No. natural types	3	3
	Total macrophyte cover	1	1
	Extend lake wards?	5	2
	No. macrophyte types	3	3
	Total fish cover	0	0
	No. littoral features	4	4
Whole lake	No. wetland habitats	4	20
	No. islands	0	0
	No. deltaic deposits	2	4
Vegetation structure	Introduced species	1	2
LHQA total	**score (out of 112)**		**70**

Table 28: Features and scores for Lake Habitat Modification Score (LHMS) for Lake Simbi

Pressure	Score
Shore zone modification	2
Shore zone intensive use	4
In-lake use	4
Hydrology	2
Sediment regime	6
Nuisance Species	0
LHMS total score (out of 42)	**18**

Habitat Quality Indicators

Based on the information collected in the entire survey, the aggregate scores for habitat quality indices of LHQA and LHMS were generated (Tables 27 and 28 respectively), and used to describe the condition of the lake habitat based on the guidelines contained in the Ratings for Habitat Quality of EPA Victoria (2010) (Table 8).

Lake Simbi recorded a much higher LHQA score of 70/112 (which is indicative that the habitat quality is “good”) and a relatively lower LHMS score of 18/42 (which is indicative that the habitat modification is “moderate”). These indices collectively suggested that the Lake Simbi habitat is moderately pristine since its hydromorphology is moderately modified by the various pressures which are still operating at marginal scales.

4.6 Correlation analyses between some selected physico-chemical parameters, trophic state index and the phytoplankton community characteristics of Lake Simbi

Pearson correlation analyses were carried out for various variables (Table 29). Temperature had a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with conductivity, salinity, hardness and TP, and a medium positive correlation ($p < 0.05$, $0.3 < R^2 < 0.5$) with DO concentration, TDS, turbidity and CTSI. The DO concentration on the other hand also showed a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with conductivity, salinity, hardness, TP and TDS. Conductivity shown a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with salinity, hardness, TDS, TP and TN but negatively correlated with total alkalinity. Salinity had a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with hardness, TDS, TP and TN and a medium negative correlation with total alkalinity. Alkalinity negatively correlated with total hardness. Hardness showed a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with TP and TN, and a medium positive correlation with TDS. TDS posted a a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with TP. Turbidity had a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with Chl-a and a medium positive correlation with CTSI. Chl-a had a strong positive correlation ($p < 0.05$, $R^2 > 0.5$) with CTSI. Shannon-Wiener diversity index had a strong negative correlation with Simpson’s diversity index. Evenness index also had a strong negative correlation with Simpson’s diversity index.

Table 29: Pearson correlation analysis for some physico-chemical parameters, CTSI, phytoplankton abundance and diversity indices

Correlations																				
		WT	DO	pH	EC	Sal	TA	TH	TDS	Turb	TP	TN	SiO_2	Chl-a	CTSI	Abundance	Richness	Shannon	Evenness	Simpson
WT	Pearson Correlation	1	.479**	-0.159	.674**	.640**	-0.290	.546**	.440**	.475**	.566**	.349*	0.074	0.280	.460**	-0.229	0.081	0.288	0.277	-0.233
	Sig. (2-tailed)		0.003	0.355	0.000	0.000	0.087	0.001	0.007	0.003	0.000	0.037	0.667	0.098	0.005	0.178	0.637	0.089	0.101	0.172
DO	Pearson Correlation	.479**	1	-.370*	.583**	.554**	-0.270	.643**	.563**	0.113	.611**	0.254	0.023	-0.149	0.107	-0.083	0.014	0.056	0.065	-0.041
	Sig. (2-tailed)	0.003		0.026	0.000	0.000	0.111	0.000	0.000	0.513	0.000	0.135	0.892	0.384	0.536	0.630	0.935	0.747	0.707	0.812
pH	Pearson Correlation	-0.159	-.370*	1	-0.170	-0.130	0.003	-0.115	-0.298	0.135	-0.229	-0.060	0.088	0.080	0.001	-0.003	0.166	0.085	0.072	-0.044
	Sig. (2-tailed)	0.355	0.026		0.321	0.450	0.986	0.504	0.077	0.432	0.180	0.729	0.610	0.643	0.993	0.985	0.334	0.622	0.675	0.799
EC	Pearson Correlation	.674**	.583**	-0.170	1	.849**	-.443**	.776**	.705**	0.188	.701**	.558**	0.215	-0.103	0.238	-.355*	-0.295	0.077	0.199	-0.080
	Sig. (2-tailed)	0.000	0.000	0.321		0.000	0.007	0.000	0.000	0.272	0.000	0.000	0.208	0.552	0.162	0.034	0.080	0.657	0.244	0.641
Sal	Pearson Correlation	.640**	.554**	-0.130	.849**	1	-.465**	.774**	.585**	0.111	.626**	.607**	0.073	-0.174	0.179	-0.320	-0.214	0.054	0.140	-0.068
	Sig. (2-tailed)	0.000	0.000	0.450	0.000		0.004	0.000	0.000	0.519	0.000	0.000	0.674	0.310	0.297	0.057	0.210	0.755	0.416	0.695

	N	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36
TA	Pearson Correlation	-0.290	-0.270	0.003	-.443**	-.465**	1	-.435**	-.419*	-0.047	-.353*	-.336*	-0.313	0.149	-0.005	0.055	.420*	0.091	-0.019	-0.053
	Sig. (2-tailed)	0.087	0.111	0.986	0.007	0.004		0.008	0.011	0.785	0.035	0.045	0.063	0.386	0.976	0.750	0.011	0.597	0.914	0.758
TH	Pearson Correlation	.546**	.643**	-0.115	.776**	.774**	-.435**	1	.499**	0.194	.590**	.672**	0.188	-0.199	0.021	-0.229	-0.142	0.043	0.111	-0.035
	Sig. (2-tailed)	0.001	0.000	0.504	0.000	0.000	0.008		0.002	0.257	0.000	0.000	0.273	0.246	0.904	0.178	0.409	0.803	0.518	0.840
TDS	Pearson Correlation	.440**	.563**	-0.298	.705**	.585**	-.419*	.499**	1	0.193	.705**	0.174	0.130	-0.049	0.267	-0.293	-0.241	0.148	0.239	-0.153
	Sig. (2-tailed)	0.007	0.000	0.077	0.000	0.000	0.011	0.002		0.259	0.000	0.311	0.450	0.778	0.115	0.083	0.158	0.389	0.160	0.375
Turb	Pearson Correlation	.475**	0.113	0.135	0.188	0.111	-0.047	0.194	0.193	1	0.167	0.164	0.203	.636**	.441**	-0.183	0.186	0.258	0.230	-0.198
	Sig. (2-tailed)	0.003	0.513	0.432	0.272	0.519	0.785	0.257	0.259		0.329	0.339	0.234	0.000	0.007	0.284	0.277	0.129	0.178	0.247
TP	Pearson Correlation	.566**	.611**	-0.229	.701**	.626**	-.353*	.590**	.705**	0.167	1	.337*	0.121	-0.108	0.266	-0.302	-0.031	0.088	0.125	-0.054
	Sig. (2-tailed)	0.000	0.000	0.180	0.000	0.000	0.035	0.000	0.000	0.329		0.045	0.482	0.531	0.116	0.074	0.859	0.611	0.467	0.753
TN	Pearson Correlation	.349*	0.254	-0.060	.558**	.607**	-.336*	.672**	0.174	0.164	.337*	1	0.000	-0.160	-0.055	-0.088	-0.071	-0.198	-0.173	0.249
	Sig. (2-tailed)	0.037	0.135	0.729	0.000	0.000	0.045	0.000	0.311	0.339	0.045		0.999	0.350	0.748	0.609	0.679	0.247	0.313	0.144

	N	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36
SiO_2	Pearson Correlation	0.074	0.023	0.088	0.215	0.073	-0.313	0.188	0.130	0.203	0.121	0.000	1	0.068	0.013	0.256	-.414*	-0.042	0.067	0.003
	Sig. (2-tailed)	0.667	0.892	0.610	0.208	0.674	0.063	0.273	0.450	0.234	0.482	0.999		0.692	0.942	0.132	0.012	0.808	0.699	0.988
Chl-a	Pearson Correlation	0.280	-0.149	0.080	-0.103	-0.174	0.149	-0.199	-0.049	**.636****	-0.108	-0.160	0.068	1	**.756****	-0.166	0.300	.353*	0.277	-0.311
	Sig. (2-tailed)	0.098	0.384	0.643	0.552	0.310	0.386	0.246	0.778	0.000	0.531	0.350	0.692		0.000	0.333	0.075	0.035	0.102	0.065
CTSI	Pearson Correlation	.460**	0.107	0.001	0.238	0.179	-0.005	0.021	0.267	.441**	0.266	-0.055	0.013	**.756****	1	-0.250	0.131	.370*	.348*	-.337*
	Sig. (2-tailed)	0.005	0.536	0.993	0.162	0.297	0.976	0.904	0.115	0.007	0.116	0.748	0.942	0.000		0.142	0.446	0.026	0.038	0.044
Abundance	Pearson Correlation	-0.229	-0.083	-0.003	-.355*	-0.320	0.055	-0.229	-0.293	-0.183	-0.302	-0.088	0.256	-0.166	-0.250	1	-0.053	-0.211	-0.291	0.251
	Sig. (2-tailed)	0.178	0.630	0.985	0.034	0.057	0.750	0.178	0.083	0.284	0.074	0.609	0.132	0.333	0.142		0.758	0.217	0.085	0.140
	N	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36	36
Richness	Pearson Correlation	0.081	0.014	0.166	-0.295	-0.214	.420*	-0.142	-0.241	0.186	-0.031	-0.071	-.414*	0.300	0.131	-0.053	1	0.311	0.041	-0.175
	Sig. (2-tailed)	0.637	0.935	0.334	0.080	0.210	0.011	0.409	0.158	0.277	0.859	0.679	0.012	0.075	0.446	0.758		0.065	0.811	0.307

Shannon	Pearson Correlation	0.288	0.056	0.085	0.077	0.054	0.091	0.043	0.148	0.258	0.088	-0.198	-0.042	.353*	.370*	-0.211	0.311	1	.953**	**-.969****
	Sig. (2-tailed)	0.089	0.747	0.622	0.657	0.755	0.597	0.803	0.389	0.129	0.611	0.247	0.808	0.035	0.026	0.217	0.065		0.000	0.000
Evenness	Pearson Correlation	0.277	0.065	0.072	0.199	0.140	-0.019	0.111	0.239	0.230	0.125	-0.173	0.067	0.277	.348*	-0.291	0.041	.953**	1	**-.971****
	Sig. (2-tailed)	0.101	0.707	0.675	0.244	0.416	0.914	0.518	0.160	0.178	0.467	0.313	0.699	0.102	0.038	0.085	0.811	0.000		0.000
Simpson	Pearson Correlation	-0.233	-0.041	-0.044	-0.080	-0.068	-0.053	-0.035	-0.153	-0.198	-0.054	0.249	0.003	-0.311	-.337*	0.251	-0.175	**-.969****	**-.971****	1
	Sig. (2-tailed)	0.172	0.812	0.799	0.641	0.695	0.758	0.840	0.375	0.247	0.753	0.144	0.988	0.065	0.044	0.140	0.307	0.000	0.000	

**. Correlation is significant at the 0.01 level (2-tailed).

*. Correlation is significant at the 0.05 level (2-tailed).

CHAPTER FIVE

DISCUSSION

5.1 Introduction

This chapter presents the discussions on the results of water quality (physico-chemical) parameters, phytoplankton community structure, trophic state index (TSI) and habitat quality survey of Lake Simbi. The results are discussed in-depth while being compared with other relevant studies including various international standards like WHO (2011).

5.2 Physico-chemical parameters

5.2.1 Secchi Depth (Transparency)

The Secchi depth influences the amount of light intensity reaching the water column (Tiseer, Tanimu & Chia, 2008). The spatial variations observed in Secchi depths among the stations in Lake Simbi might be due to variations in depth among stations. The onshore stations (ST4 and ST5) possess shallow depth and therefore have their settled sediments at the bottom vulnerable to wind perturbations such that a small wind blowing over the water surface is enough to stir up the sediments and result in low Secchi depths. The offshore stations (ST2, ST3 and ST6) possess greater depths and therefore have their settled sediments at the bottom away from the reach of wind action and so are not brought up in suspension in the water column which explains why they recorded high Secchi depths. Also, the high Secchi depths recorded in ST1, ST2 and ST3 could be due to the high vegetation cover on the shoreline that tightly holds the soil hence preventing erosive floods from sweeping away sediments and depositing them in the lake.

The temporal variations observed in Secchi depths in Lake Simbi can be mainly attributed to the seasonal variations in the amounts of precipitation received within that region. The low depths recorded during the months in the wet season could be due to high run off from the surface and increased flooding brought by the high discharge. This corresponds with the observations by Mwaura (2000) that attributed low Secchi depths in small water bodies found in the rift valley to large inflows from flooding. These inflows are characterized by enormous quantities of suspended particles which when deposited in the water decreases the transparency (Bronmark & Hansson, 2005)

5.2.2 Temperature

Water temperature is an important factor that influences physiological activity of organisms. The temperature variations observed in Lake Simbi can be attributed to variations in nature of solar radiation exposure at sampling stations and variations in the prevailing weather conditions during the sampling period. The higher temperatures generally recorded in Lake Simbi could be because of the high air temperatures of the arid climatic conditions of the region. The temperatures at the water surface have been known to closely follow the temperatures of the surrounding air (Breen, Chutter, Merona & Saad, 1981). Since, the whole lake is located in an open area that is directly exposed to the sun, only small variations of temperature were realized among sampling stations since they all receive equal direct heating from the insolation. The general higher temperatures could also be attributed to the fact that Lake Simbi is a closed lake which gets little to know inflow except for the surface run-off and no outflow therefore there is constant warming of the water.

The variations in the exact time of sampling could also be the cause of the slight variations observed during the sampling period in Lake Simbi. Those sampling stations sampled during the early hours of the morning (especially ST1) registered higher temperatures than the ones which might have been sampled during the late hours of the morning (such as ST4 and ST5). The temporal variations in temperature could probably be attributed to the cloud cover during some months of sampling. Some sampling months had total cloud cover which made them to register low temperatures as compared to those that had clear sky during sampling.

The seasonal variation of temperatures exhibited an unusual occurrence whereby the temperatures of the wet season were found to be higher than those of the dry season which is inconsistent with the common occurrence of dry season temperatures being always higher than the wet season temperatures. This phenomenon could be attributed to the cooling effect due to evaporation and the occasional rainfall that occurred during some weeks in the dry season. Further, the global effects of climate change could also be attributed to this weird occurrence.

5.2.3 Dissolved Oxygen Concentration

The concentration of DO in the water supports the metabolism of all aquatic life possessing aerobic respiration as part of their biochemistry (Goldman & Horne, 1983), and therefore it is a crucial indicator of water quality. The concentration of DO recorded in all seasons in Lake Simbi was generally oscillating around 5.25 (± 0.21) mgL^{-1} which is recommended for fish survival by USEPA (2005) but it is insufficient to support the demands of most other aquatic organisms. This was lower than the 8 mgL^{-1} and 7 mgL^{-1} recommended by NEMA (2006) and the WHO (2011) respectively. This shows that Lake Simbi has poor water quality. The general increasing trend of DO concentration among the months sampled could be due to the increased photosynthetic activity which utilizes CO_2 and releases a substantial quantity of oxygen to the water. This corresponds to the general increasing trend observed in temperatures among the sampled months since high temperature influences high photosynthetic activity and high diffusion of oxygen in the water column.

The spatial variations in DO concentration observed in Lake Simbi among the sampling stations can be attributed to variations in the prevailing weather conditions (e.g. cloud cover) and the differences in the exact time of sampling. Those sampling stations initially sampled during the early hours of the morning (especially ST1 and ST2) registered lower DO concentration than the ones which were sampled during the late hours of the morning (such as ST3, ST4, ST5 and ST6). This is because during the early morning hours the temperatures are still low in the water which reduces the photosynthetic rates while during the late morning hours there are high temperatures in the water which increases photosynthetic rates. The high temperatures in the late morning hours also causes high diffusion rates of oxygen in the water as compared to the early morning hours. The extremely low DO observed in ST1 could be because of a combination of low temperature and shallower depth which stimulates the breakdown of organic matter by microbial organisms. The offshore stations (especially ST3 and ST6) recorded high DO concentration levels as opposed to the ones on shores because the ones closer to the shores experience high nutrient loading from the nitrogen and phosphate based fertilizers coming from the subsistence agricultural activities carried out around the lake's habitat. This nutrients causes eutrophication of these areas hence depletion of oxygen levels by the ensuing algal growth.

The temporal variations in temperature could probably be attributed to the cloud cover during some months of sampling. Some sampling months had total cloud cover which made them to register low temperatures as compared to those that had clear sky during sampling. The months which occur during the transitional period between the dry season and wet season (i.e. February 2019 and March 2019) registered low concentration of DO compared to the other months since they had almost 60% cloud cover which therefore slowed the photosynthetic activity.

Seasonally, the higher concentration observed during the wet season as compared to the dry season follows the unusual temperature trend observed in Lake Simbi in which the higher temperatures was recorded during the wet season as compared to the dry season.

5.2.4 pH

Naturally, the of pH in water is dependent upon the concentration of dissolved carbon (iv) oxide, alkalinity, carbonates and bicarbonates while its level is determined by the photosynthesis and respiration in the aquatic ecosystem (Skelton, 2001). The mean pH levels observed in Lake Simbi were generally highly alkaline oscillating around 10.23 (± 0.11). However, they were within a range that supports aquatic life (6.5 – 9.0) as prescribed by USEPA (2005) but fall above the recommendations for drinking water (6.5 – 8.5) as set by both NEMA (2006) and WHO (2011). This therefore means the water has poor quality. The general alkaline inclination of the pH observed in Lake Simbi can be attributed to the high photosynthetic rates occasioned by a combination of high temperatures, dense algal growth and eutrophication of water. The high photosynthetic rates of the abundant phytoplankton biomass catalyzed by high temperatures cause the removal of carbon (IV) oxide in the water therefore raising the pH in it. The high pH could also be attributed to the high salinity levels of the lake.

The slight variations in pH observed in Lake Simbi among the sampling stations can be attributed to variations in the prevailing weather conditions (e.g. cloud cover) and the differences in the exact time of sampling. Stations sampling during the cloudy days or when the temperatures were low experienced decreased photosynthetic rates and hence low pH but the ones sampled during clear skies and high temperatures experienced high photosynthetic rates (Harris, Piccinin & Ryn, 1983), which decreased the CO_2 concentration in the water and hence raised pH. The microbial decomposition of organic matter dumped into the lake by the surrounding residents in terms of wastes could also raise the pH since it manufactures humic acids which reduce the level of pH. During most of the sampling sessions the cloud cover was relatively stable not surpassing 10% which explains the relative stability of pH levels observed during the entire sampling period in Lake Simbi.

The slightly higher pH observed during the dry season as compared to the wet season could probably be as a result of decomposition of the organic matter in the decreased water level during the dry season while the slightly lower pH observed during the wet season could be due to the stirring effect brought by the flood inflows which causes the mixing of the acidic water at the bottom and the highly alkaline water at the surface, lowering the pH level. It is noteworthy that the Lake is pH stratified into acidic layer at the bottom, slightly alkaline in the middle layer and a highly alkaline surface.

5.2.5 Electrical Conductivity, TDS and Salinity

Measuring conductivity of "natural water" can provide a clear picture of the concentration of nutrients dissolved in water such as nitrates and phosphates (Fatoki *et al.*, 2003). In his assessment he states that the conductivity of "natural water" ranges between 20- 1500 µs/cm and since the conductivity measured in Lake Simbi was much higher than this range, its waters can't be described as "natural" but rather "polluted" by excess nutrients from its catchment area. The levels of conductivity in water masses are governed by the levels of TDS in the water and therefore they both exhibited similar variability across stations and months. The level of TDS is in turn governed by the residence time of the water, geology, soil type and the anthropogenic activities occurring within the catchment (Ansa-Asare & Asante, 1996). The TDS levels recorded in Lake Simbi oscillated around 5372.07 (± 837.27) mgL^{-1} which fall above the permissible limits for drinking water (1200 mgL^{-1}) as set by NEMA (2006) and (1500 mgL^{-1}) set by the WHO (2011) which therefore suggests that the lake has poor water quality.

The variations in both electrical conductivity and TDS in Lake Simbi could be attributed to the variations in the precipitation and temperatures received in the region. This assumption is in line with the study by Chapman and Kramer (1991) which indicated that the conductivity of water bodies in the tropics is greatly influenced by changes in the precipitation and temperatures within their locality. A study by Maitland (1994) which revealed high amounts of TDS in cultivated areas is in agreement with this study since high levels of TDS were recorded in Lake Simbi arising from the heavily cultivated farms around the lake. The high TDS was responsible for the high conductivity.

The general increasing trend of both conductivity and TDS among months of the sampling period could be because of the increasing rainfall and temperature among the sampled months from December 2018 to May 2019. The dry season had low values for both conductivity and TDS as compared to the wet season which had high conductivities and TDS, because during the wet season the high precipitation caused elevated levels of surface runoff that deposited a lot of dissolved solid substances into the lake. This phenomenon of high conductivity and high TDS in the wet season than the dry season is unusual since normally the dry season is expected to have high conductivity and TDS than the wet season because the wet season experiences dilution brought about by the rains. However, pollution and contamination arising from the high rates of erosion from the geology, agricultural farms in the catchment, livestock grazing around the lake, washing of clothes and the organic wastes dumping in the lake could possibly lead to such high conductivity and TDS levels of Lake Simbi. Several studies (WHO, 1996; NEMA, 1999; USEPA, 2007) have established that TDS level beyond 1200 mgL^{-1} is related to the presence of toxic contaminants in the water body. Also, since the lake lacks any outlet or inlet, the residence time of its water is much longer and this coupled with high temperatures could justify its ever high conductivity and TDS values.

Salinity measures the level of salts concentration in the water and therefore it has a direct relationship with both conductivity and TDS, which explains why the parameters had the same trends in both spatial and temporal scales. High salinity comes from the dissolved salts washed off by the surface runoff and deposited into the lake. High salinity favors the production of cyanobacteria which explains the occurrence of blue-green algal bloom in Lake

Simbi. Krienitz *et al.* (2013) study in Lake Oloidien realized an enormous increase in cyanobacteria as the waters become more saline proving the connection between the two.

5.2.7 Alkalinity

Alkalinity is seen basically as a concentration of three major components namely CO_2, carbonates and bicarbonates. It is important in aquatic ecosystems since it protects the aquatic life from abrupt pH variations in water. The alkalinity levels recorded in lake Simbi oscillated around 8115.11 (± 123.66) mgL^{-1} which fall above the permissible limits for drinking water (500 mgL^{-1}) as set by both NEMA (2006) and WHO (2011) which indicates that the lake's water quality is poor.

The total alkalinity variation in Lake Simbi could be due to the variations in the four processes of respiration, photosynthesis, evaporation and weathering from its geology. The low levels of total alkalinity observed in the months of wet season (March, April and May) could be as a result of the floods of high velocity generated by the surface runoff limiting the amount of contact time between the water and the parent rock which consequently reduces weathering and so low levels of alkalinity. On the other hand, the high alkalinity recorded during the dry season is caused by the combination of high evaporation rates and greater weathering of the limestone rock which forms the framework of the geology of the area. The high alkalinity recorded in the dry season as compared to the wet season can also be due the accumulation of large amounts of carbonates and bicarbonates by high evaporation during the dry season and to dilution of water during the wet season by the large amounts of rainfall. Generally, the lack of any outlet which ensures greater contact time between the water and the limestone parent rock coupled with high evaporation rates results in greater weathering which explains high alkalinity in Lake Simbi. Moreover, the high rates of photosynthesis and evaporation characterizing Lake Simbi could be also contributing to the high alkalinity in Lake Simbi. This assumption is in agreement to the study by (Idowu and Ugwumba, 2005) in Nigerian waters. Furthermore, the high total alkalinity could be attributed to various sources of pollutants draining into the lake from the catchment area.

5.2.7 Hardness

Total hardness comes from the occurrence of a combination of ions but the most predominant ones are magnesium and calcium (Olawale, 2016). The total hardness levels recorded in Lake Simbi oscillated around 139.22 (± 8.75) mgL^{-1} which fall below the permissible limits for drinking water (500 mgL^{-1}) as set by both NEMA (2006) and WHO (2011) which therefore suggest that even though lake Simbi has poor water quality, the hardness level is acceptable.

Hardness followed the trend of alkalinity where the wet season recorded higher values than the dry season. This could be attributed to the increased accumulation of the Mg^{2+} and Ca^{2+} ions in the lake from the weathering of these ions from its geology and probably from the fertilizers eroded from the nearby agricultural farms which are undergoing intense cultivation regimes. Since the lake is small, there is even distribution of hardness and that explains the relatively similar values among stations. The low hardness observed during the dry season could be attributed to the dilution brought about by the short rainy season prior to the dry season encompassing the months of December through February.

5.2.7 Turbidity

Turbidity is influenced by the quantities of dissolved solids in the water column which limits the light penetration in water (Usman, 2016). The turbidity levels recorded in Lake Simbi oscillated around 81.89 ± (5.07) NTU which fall above the permissible limits for drinking water (5 NTU) as set by both NEMA (2006) and WHO (2011) which therefore suggest that water quality of Lake Simbi is poor. Turbidity and TDS are related because they both involve solids or sediments which explains why the observed high turbidity can be linked to the high TDS. The high turbidity recorded during the wet season as compared to the dry season arises from the deposition of dissolved solids by the high surface runoff. The run-off could have also caused re-suspension of sediments settled at the bottom (Balance & Batram, 1996). The high turbidity values corresponded to the low Secchi depths while the low turbidity values corresponded to high Secchi depths recorded in Lake Simbi. Generally, the high turbidity of Lake Simbi can be attributed to the sedimentation from erosion, dense phytoplankton in the water, indiscriminate wastes dumping by the local residents and institutions like hospitals, decomposition of organic material, surface runoff and re-suspension of settled sediments at the bottom. This is in agreement with Dallas and Day (2004) study which documented the factors influencing turbidity in water.

5.2.7 Nutrients

Aquatic life requires nutrients for growth and development. All the nutrients sampled in Lake Simbi fell below the permissible limits set by both NEMA (2006) and WHO (2011) except for TP which was slightly above the set limits. This shows that the water quality in terms of nutrients concentration in Lake Simbi is generally poor due to the high phosphate inputs. The levels of TN and TP were generally high in all the sampled stations. This could be attributed to the high agricultural activities in the catchment. Agricultural activities have been reported by Omernik (1976) to cause increased nutrient loading in the aquatic ecosystem. In aquatic ecosystems, TN and TP have been reported to be the major nutrients limiting productivity. The ratios of concentration of these two nutrients have often been used to establish which one is responsible for regulating phytoplankton growth. Redfield (1958) stated that for a healthy ecosystem, the ratio of N: P should be 1:16 but the one recorded in Lake Simbi was oscillating around 1:10 which according to the Redfield's ratio reference depicts that N is the limiting nutrient in Lake Simbi and the lake's ecosystem is therefore unhealthy. The nutrient enrichment of the lake could be attributed to the intensive fertilizer use in the surrounding area which during the rains are swept and deposited into the lake by flood inflows. It could also be attributed to the natural environmental processes. These assumptions concurs with Guldin (1989) who opined that apart from the sources emanating from human activities such as mining, industrial operations and poor agricultural practices, nutrients ending up in water masses can occur from the natural processes of the atmosphere and fixation by micro-organisms and lightning.

All the nitrogen-based and phosphate-based nutrients showed general increasing temporal trend among the sampling months with the wet season recording higher values than the dry season for all the nitrogen-based and all the phosphate-based nutrients. This could be attributed to the increased nutrient deposition that occurs during the wet season whereby the high rainfall discharge causes the surface runoff and flooding that sweeps the various nutrients from the heavily cultivated farms with loosed soil and no vegetation cover. It was established that most farmers were applying CAN and DAP fertilizers in their farms to increase productivity since the area is in a semi-arid region. Other potential sources of nutrients in this lake could be the washing of clothes using detergents on its waters and also grazing of the livestock which drops nutrient-rich faeces around the lake during which then leaches to the water during the rainy season. The variations in TP concentration results from the human activities, the geochemistry of the catchment, waste disposal, detergent effluents emanating from washing clothes, agricultural runoff and the climatic conditions.

The spatial variation in nutrient concentration can be attributed to the closeness of the sampling stations to the shores, the intensity of agricultural practices nearby and the vegetation cover. The stations located offshores (especially ST3 and ST5) recorded much lower levels for all nutrients since being in deep waters they are far from the points of entry of the various nutrients eroded from the nearby farms. Low levels of ammonia were recorded in all the station except ST4 which is had high values. This is because it is located in one of the most accessible areas of the lake and therefore it is prone to pollution from dumping of the residential and agricultural wastes which then decomposes microbially to produce ammonia. The low levels of nitrites and nitrates recorded in most stations in Lake Simbi could be due to the uptake by the phytoplankton and utilization during some biological processes. The lack of vegetation cover around the lake caused the intensity of nutrient enrichment since there are no buffers which can hold and utilize the nutrients thereby preventing them from reaching the lake. Rijsdijk, Bruijnzee & Prins (2004) opined that the water bodies surrounded by vegetation cover usually records low nutrient concentration.

The silicates recorded from lake Simbi could be arising from the weathering of silica from the lithology of the catchment of lake Simbi under the influence of CO_2 (Hutchinson, 1994). The temporal variations in silica concentration could be coming from the variations in the precipitation patterns which weather the rocks of the catchment. The low silicates observed during the dry season correspond to the high bacillariophyceae observed during the dry season. This is because the bacillariophyceae have been known to utilize silica for their biology. Sommer and Stabel (1983) reported that silica makes part of the structure of bacillariophyceae and it's crucial for the manufacture of chlorophyll.

5.2.8 Vertical stratification and classification of Lake Simbi

Based on mixing regimes, Lake Simbi can be described as meromictic since it experiences little mixing. Being a closed lake which lacks both inlet and outlet, mixing in the lake Simbi only occurs through the action of the wind. The partial mixing occurring in it is brought about the effect of wind forces blowing over it from the shores of Lake Victoria. But since Lake Simbi sits on a depression (sunken ground), it is mostly shielded from the disruptive action of strong wind forces by the highly elevated banks, and its deep depth would also limits better mixing. These winds barely mix the epilimnion layer of the water leaving the deep water layers intact. The chemocline initially observed by Ochumba and Kibaara (1988), still exists at the same depth of 4m. This implies that the meteorological and hydrological conditions in the region around the lake, and the morphometry of the lake have maintained a relatively stable stratification in the lake over the years. The thermal stratification generally influences the water quality as well as the planktonic community structure in the lake. The concentration of dissolved oxygen and the nutrient cycling are both impacted by the thermal stratification. The partial mixing potentially enables some of the nitrogen and phosphorus nutrients to be released from the sediments into the hypolimnion layer and consequently stimulating the formation of the cyanobacteria blooms in form of algal scum.

The floating algal scum observed in Lake Simbi is a bloom which comes from the blue-green algae. It is always seen as a symptom for a sophisticated level of eutrophication (Paerl & Ustach, 1982). This cyanobacterial scum is responsible for the foul smell in Lake Simbi. According to Paerl and Ustach (1982), they algal scum lowers the concentration of dissolved oxygen which in turn leads to fish mortalities, elevated toxicity and general decline in aesthetic value of the water body. These impacts have economic implications on tourism, public health and businesses. The eutrophication occurring in Lake Simbi originates from the extensive agricultural farms that surround the lake to the extent of covering even most of the riparian zones. The nutrients are carried from the upper regions of the catchment by storm floods and drained into the lake which sits in a sunken ground surrounded by steep slopes. These nutrients mainly from nitrogen and phosphate based fertilizers enrich the waters leading to the formation of algal bloom which is normally seen floating on the water surface. Since this lake lacks any outlet, all the nutrients deposited build up over time. The floating occurs because the surface algal scum becomes increasingly buoyant when the cells of the dominant phytoplankton species such as of Anabaena and Microcystis form gas vacuoles (Booker & Walsby, 1981).

With hypoxic conditions (DO $\leq$ 4 mg l^{-1}) beginning at the depth of 4m and anoxic conditions (DO = 0 mg l^{-1}) beginning at depths below 10m, the waters of Lake Simbi seem to be devoid of sufficient oxygen that can support fisheries. The oxycline exists at the same level as the thermocline since the rapid drop in temperature produced a corresponding rapid drop in dissolved oxygen concentration in the water. The decreasing oxygen concentration with depth results from the decreasing temperature with depth. A high temperature stimulates high dissolution of oxygen from the atmosphere into the water and vice versa, so when the temperature decreases less oxygen diffuses from the atmosphere into the water. Apart from the decreasing temperatures with depth, the oxygen depletion could have probably been caused by increased concentration of sulfide and organic carbon in the water or by increased microbial respiration which utilizes all the oxygen to depletion. Since salinity also directly affects the concentration of dissolved oxygen, the low dissolved oxygen concentration observed in Lake Simbi must have also been caused by the high salinity levels measured. This combination of factors is responsible for the hypoxic and almost anoxic conditions experienced in Lake Simbi.

The elevated pH values observed on the upper surface of the water is indicative of an elevated rate of photosynthetic activity, which signifies that the lake is undergoing a sophisticated level of eutrophication. This elevated rate of photosynthetic activity is fueled by the protracted dry period (occurring annually between Novembers to March) which causes the pH level of the water to rise (Wetzel, 2001). The evidence of eutrophication was seen on top of the water surface which appeared dark green in color and had a thick algal scum floating in several areas. This cyanobacteria scum was established to be coming from the phytoplankton species of genera *Microcystis* which dominated most parts of Lake Simbi. This indicates high nutrient enrichment causing eutrophication. The low values of pH observed in deeper water column of the lake usually stems from high concentration of CO_2 released from the respiration by microbes in the water (Widdel, 1988).

According to Wetzel (2001), salinity measures the level of salt concentration dissolved in a water body but these salts undergo dissociation in the water column to give cations and anions, which forms the foundation for conductivity which in turn measures the capability of water to allow an electrical current to pass through it. Total Dissolved Solids is a measure of the aggregate summation of all the ions in the water whose size is less than 2µm present, (Environmental Protection Agency, 2012). This demonstrates that both conductivity and TDS are intricately interrelated to salinity, which explains why both salinity and TDS can be computed directly from conductivity values (Fondriest Environmental Inc., 2014). As salinity increases, both conductivity and TDS also increases and vice versa. Thompson (2006) observes that a "clean water" should be one having equal amounts of both TDS and salinity. The high TDS, salinity and conductivity registered in Lake Simbi comes from the geology of the lake (including the clay soils), pollution from illegal dumping and the anthropogenic perturbations in terms of deforestation and agricultural activities. EPA (2012) reveals that the clay sediments washed off from the surrounding soils and the minerals from phosphates and nitrates based fertilizers washed from the nearby agricultural farms into the lake become ionized and hence contribute to TDS, salinity and conductivity.

The similarity observed in the annual occurrence pattern of the thermocline, oxycline and the chemocline in Lake Simbi suggests that the most biogeochemical processes of mineralization and nutrients transformation in the waters of Lake Simbi occurs majorly at the depths above 4m.

5.3 Phytoplankton community characteristics

5.3.1 Phytoplankton biomass, density and diversity

Chlorophyll-a is a green pigment used by the phytoplankton to absorb sunlight required for the photosynthetic process. It serves as important indicator for both the quality and quantity of the phytoplankton in water bodies and, hence a crucial variable for determining the water quality (USEPA, 2001b). The high chlorophyll-a biomass recorded in Lake Simbi corresponds with the high phytoplankton abundance observed. During the investigation in the lake, there was high Chl-a biomass of about 190.92 ± 53.01 μgL^{-1} in the waters and the waters also had a strong foul smell. A study by KDHEBW (2011) associated water bodies with more than 10 μgL^{-1} of Chl-a with poor odor and taste. Taking these standards into consideration, it can be deduced that the waters of Lake Simbi are of very low quality. The high algal productivity indicated by the Chl-a biomass caused an explosive algal bloom in the lake waters which explains the occurrence of the algal scum that covered most parts of the lake. The algal scum impaired the quality of water and is responsible for the foul smell that comes out of the lake. For the poor taste of the water, the toxins produced from the decomposing algal biomass. The algal bloom occurrence has always been linked to the increased nutrients loading especially TN and TP (Harding & Perry, 1997). The high chlorophyll-a observed in Lake Simbi could be as a result of high nutrient loads carried by the surface runoff and flood inflows from the fertilizer used in the heavily cultivated catchment. The bloom indicates that the phytoplankton species found in the Lake Simbi are tolerant to pollution conditions since the water is suffering from advanced eutrophication.

The spatial variations in Chl-a observed in lake Simbi can be attributed to the variations in nutrient concentrations among sampling stations. Sampling stations (ST1, ST2 and ST3) which occurred in areas closer to points of entry of nutrient loads from runoff recorded higher Chl-a than the ones located away from these points (ST4, ST5 and ST6). On the other hand, the temporal variations in Chl-a observed in lake Simbi can be attributed to the variations in the precipitation patterns, temperature, light intensity conditions during sampling. The stations sampled during the early morning hours recorded low Chl-a because of the reduced light intensity but the ones sampled during the late hours of the morning recorded higher values of Chl-a due to the increased light intensity, which explains the variations observed among stations. The low concentration of Chl-a occurring in wet season months could be due to the reduced phytoplankton productivity arising from increased turbidity caused by the heavy rains. The high turbidity of the wet season hindered light penetration which in turn impaired photosynthesis and so low algal production. Furthermore, the high Chl-a biomass recorded in dry season was higher as compared to the wet season because during the dry season there's high photosynthetic rate from the direct sunlight but during the wet season the photosynthetic rate is reduced by the cloud cover.

On the phytoplankton density (abundance) exhibited seasonal variation in terms of dominance. The phytoplankton density recorded in Lake Simbi followed the chl-a biomass trend. In terms of density, the dry season recorded high counts of individuals as compared to the wet season because of the left-over nutrients which were made available by the rainy season preceding the short dry spell of December through February. Since the lake is closed, there's no flushing out of nutrients. The increased solar intensity and reduced turbidity could also be the reason for high phytoplankton abundance during the dry season (Ewa *et al.*, 2013). The reducing trend of the phytoplankton abundance from the months in the dry season to the months of the wet season could have been probably caused by a combination of decreased transparency in the water, diminished light period from the cloud cover and the dilution factor brought by the pounding rains.

The phytoplankton diversity indices exhibited spatial, temporal and seasonal variation which could be attributed to variations in nutrient dynamics and environmental conditions

(Reynolds, 1984; Lewis, 1996). While some diversity indices recorded higher values for the dry season, others recorded high values during the wet season. The two phytoplankton indices of Simpson's diversity and richness recorded higher values during the dry season as compared to the wet season which can be probably attributed to the short-term environmental variations which in turn influenced the nutrient dynamics in the lake making an assortment of species at various growth stages to co-exist - with certain species increasing while others are decreasing. The environmental variations raised the depth of the mixed layer which Lewis (1996) found to have the effect of improving nutrient dynamics that favor a situation whereby a mixture of species would exist with each other, with others decreasing and others increasing. This observation in Lake Simbi concurs with the findings of Kotut *et al.* (1998b) which stated that the reduced amount of flood inflows during the dry season lowered the availability of nutrients which have a corresponding effect of reducing species stability hence contributing to a rather high diversity. This is similar also to an assessment by Fogg (1975) that found species richness is seasonally influenced by temperature and nutrient levels.

The other two phytoplankton diversity indices (Shannon-Wiener and evenness) on the other hand, recorded high values in wet season than the dry season since the increased availability of sufficient nutrients prevents competition among various species hence enabling various families to co-exist. The quality of water in Lake Simbi recorded a mean value below 2 on the Shannon-Wiener Index which effectively categorizes it as heavily polluted. This characterization is based on a proposed classification by Shekhar *et al.* (2008) which categorizes the water as clean (> 4 Shannon-Wiener index value), mildly polluted (3 - 4 Shannon-Wiener index value) and lastly heavily polluted (<2 Shannon-Wiener index value). This investigation constantly registered low values of phytoplankton species which is indicative of inequitable abundance of different phytoplankton groups during the period of this study.

5.3.1 Phytoplankton community composition and seasonal succession

Six families of phytoplankton were recorded in Lake Simbi. Of all the six families, Cyanophyceae dominated in both spatial and temporal scales because it was highly represented by both the number of species and density while the zygnematophyceae, dinophyceae and euglenophyceae families were the least represented.

The Cyanophyceae family is known by other names; myxophyceae, cyanobacteria and blue-green algae. Newman and Barret (1993) stated that the dominance of some phytoplankton species can be utilized in categorizing the trophic status of the water. The dominance of the cyanophyceae shows that the lake is highly polluted. A condition which is further confirmed by the little occurrence of the bacillariophyceae at about 13% in terms of relative abundance indicating that the waters of Lake Simbi have poor water quality since, the occurrence of bacillariophyceae in a water body is indicative of good water quality (Tan, Ma & Xa, 2013). The dominance of cyanophyceae in alkaline-saline lakes could be due to the elevated levels of salinity, conductivity and alkalinity which are brought by the continuous weathering of the lithology and the closed nature of the lake. The dominance of the cyanophyceae could also be stimulated by the excessive nutrient loading from the catchment of Lake Simbi where the intensive anthropogenic activities contribute to the interference with the natural balance of phosphate and nitrates in the water. This situation is worsened by the fact that Lake Simbi is a totally closed lake which ensures that all the nutrients deposited are never washed off by outflows but remain in high concentration for consumption by the pollutant tolerant species, especially of the family cyanophyceae. The low bacillariophyceae could be attributed to the polluted nature of the lake which therefore can't support the species in this family and low silicate levels especially during the dry season.

Cyanophyceae family of Lake Simbi was dominated by the species *Microcystis flos-aqua* (33%), *Microcystis robusta* (25%), *Anabaena hylina* (24%) and *Microcystis aeruginosa* (7%) which is contrary to the previous findings of Ballot et al. (2005) which documented *Arthrospira fusiformis* and *Anabaenopsis abijatae* as the most abundant cyanophycean species. This shift in the phytoplankton composition can be attributed to the changes in the water quality regime of Lake Simbi. This shift in the dominance and composition of cyanobacterial species seems to be the plausible reason for the declining numbers of flamingos that inhabit Lake Simbi since their major food resources (*Arthrospira fusiformis* and *Anabaenopsis abijatae*) have been dwindling. This agrees with Tuite (1979) observation that whenever there is massive biomass of *A. fusiformis* in the saline lakes, there is high flamingo population too since it's part of their food. The declining numbers of the flamingo in Lake Simbi can also be attributed to the toxins (anatoxin-a and microcystins) which were identified in Lake Simbi by Ballot *et al.* (2005) from these dominant *Microcystis* species (microcystins) and *Anabaena* species (anatoxin-a). The algal bloom observed in Lake Simbi not only impaired the water quality but also contaminated the little food resources available for the lesser flamingos which could either be making them die off or migrate to other lakes such as Lake Nakuru. The algal blooms from blue-green algae are widely documented to produce harmful cyanotoxins (Codd *et al*., 1999).

According to Mason (1991), the occurrence of certain phytoplankton species (especially from the cholorophyceae family) explains the acidic nature of the water especially the deeper layers. The species belonging to *Romeria* and *Cyclotella* genera were found in Lake Simbi which implies that the waters are somewhat acidic since the species found in these genera have always been known to tolerate acidic conditions. The acidic nature of the lake comes from the low pH levels in the lake but it could also be attributed to the organic wastes dumped into the lake by the various institutions (schools, church and hospitals) and homes surrounding the lake. These acidic tolerant Chlorophyceans could be thriving in the deeper waters since the top waters of Lake Simbi are permanently highly alkaline. Chlorophyceae family of Lake Simbi was dominated by the species Scenedesmus obliquus (80%), Chlorella vulgaris (3%), Pediastum boryanum (3%) and Botryococcus braunii (2%). Chlorophyceae overtook bacillariophyceae earlier recorded by Ballot et al. (2005), as the second most dominant phytoplankton family. This could be due to the changing water quality conditions and climate change effects. Especially the increasing alkalinity in the lake could be a possible reason for the sudden rise of Chlorophyceae over Bacillariophyceae. Abubakar (2017) states that some aquatic organisms such as the *Chlophyta spp* requires the protection from erratic pH variations and so they would thrive well in alkaline waters. Generally, nutrient enrichment in aquatic ecosystems has been known to regulate the phytoplankton primary production and succession (Githaiga, 2003).

5.4 Trophic status classification

The trophic state is a response to the elevated nutrient concentration levels in a water body (Naumann, 1929). Although no particular trend in the spatial variations of the trophic status was realized in Lake Simbi, an increasing a slight hyper-eutrophic trend was observed in the temporal scale which could be attributed to the monthly variations in the climatic conditions such as temperature and rainfall amounts received within the catchment. Nevertheless, all the TSI indicators definitively classified the lake as hyper-eutrophic which reflects the magnitude of nutrients concentration in the lake. This high nutrients concentration comes from several years of nutrients inflows and pollution of the lake from the catchment areas experiencing intense agricultural activities among other unsustainable land uses. The reason for the higher hyper-eutrophic condition observed in the wet season is because of high amounts of rains which brought in loads of nutrients into the lake and coupled with high temperatures recorded, stimulated high Chl-a production, turbidity and nutrients accumulation. This observation is contrary to what has been normally observed in other small lakes where the dry season is more eutrophic than the wet season because the wet season is characterized by heavy rains which dilute pollutants, flushing out of algae and limited growth of algae due to unavailability of sunlight. All these conditions don’t apply to Lake Simbi because it is a closed lake in a semi-arid region with generally high temperatures throughout the year. The TSI for Chl-a was greater than the TSI for SD which implies that the turbidity and consequently light attenuation was because of the substantial amounts of suspended algal biomass and not because of the mineral or solid substances. Carlson (1977) had opined that whenever TSI (Chl-a) is greater than the TSI (SD), it is an indication that algae in the water dominates the light attenuation.

The hyper-eutrophic nature of Lake Simbi was also confirmed by Nygaard’s trophic state indices for algal quotients calculated for the lake. The phytoplankton quotients calculated in Lake Simbi were way higher on the eutrophic scale surpassing even the range established by Nygaard (1976). This reflects the cultural eutrophication of Lake Simbi emanating from the pressure caused by anthropogenic disruptions, erosion and siltation, domestic wastes dumping, agricultural fertilizer use in the catchment area which has altered the water quality. The hyper-eutrophic status is evidenced by the cyanobacterial bloom (in form of algal) observed in Lake Simbi throughout the study period. Moschini-Carlos *et al.* (2009) observed that the occurrence of cyanobacterial bloom in aquatic ecosystem is the greatest symptom for rapid eutrophication.

The Redfield's TN/TP ratios calculated for Lake Simbi further confirms its eutrophic status since N was established to be the limiting nutrient. Ryding and Rast (1989) established that in the tropical region, N is generally limited in eutrophic waters while P is generally limited in oligotrophic waters. The authors further stated that the metabolism of the organisms in the water can change the TN/TP dynamics. Moreover, they observed that heavily polluted waters (such as Lake Simbi) experience N limitation because of the following reasons; the loss of N from the bottom of the waters due to denitrification, the release of P trapped by the bottom sediments and probably sewage disposal into the lake. These reasons could all be responsible for the N limitation of Lake Simbi because the lake is heavily polluted by wastes dumping, heavy sedimentation and dense algal growth.

5.5 Lake Habitat Quality

LHQA is an LHS index that is designed to quantify the degree of diversity and naturalness of the physical lake habitat. Lake Simbi recorded a much higher LHQA score of 70/112, indicative that the physical habitat quality is "good". This implies that the lake's physical habitat quality can be regarded as a pristine but moderately modified which is starting to experience the impacts of anthropogenic interferences. Drawn comparison from similar studies in the tropics across the continent shows that Lake Simbi scored lower than both Cleveland (78) and Malilangwe (76) but slightly higher than higher than Lake Chivero (62), all of which are water bodies in Zimbabwe surveyed by Dalu *et al.* (2013; 2016). However, it scored higher than all the Victorian lakes in Australia surveyed by EPA Victoria (2009) and most lakes in the UK studied by Rowan *et al.* (2005). These findings suggest that the lakes in the tropics might be having better physical habitat quality as compared to their counterparts in the temperate regions that are rapidly losing their naturalness and level of diversity due to the influence of anthropogenic activities. The “good” physical habitat quality recorded in Lake Simbi is quite impressive considering the fact that it totally lacks any protective regime even though it is an important national bird sanctuary in Kenya recognized by law and placed under management of the KWS by law. The “good” habitat quality recorded in Lake Simbi shows that the recent conservation efforts by various concerned stakeholders in Lake Simbi are paying off and so the government agencies need to support these conservation efforts. Lake Simbi fared better on the LHQA index despite an established fact by McGoff and Irvine (2009) that small alkaline lakes (such as Lake Simbi) always score lower on LHQA index notwithstanding their naturalness, compared to large highly alkaline lakes because they normally possess relatively small number of emergent macrophytes and little habitat diversity. Alkaline-saline lakes such as Lake Simbi are saline environments possessing “salt stress” which limits the growth and development aquatic plants (Sim, Davis, Chambers & Strehlow, 2006). According to Haller (1974), only few macrophytes which have developed strong adaptation mechanisms in their structure, physiology and biochemistry can withstand the extreme conditions of saline lakes.

LHMS is an LHS index that is designed to classify the habitat quality in terms of the degree of hydro-morphological alterations in the lake habitat resulting from the various anthropogenic activities. It essentially measures the pressures occurring in the lake habitat that might have impacts on its "ecological status". Lake Simbi scored a relatively lower LHMS score of 18/42, indicative that the habitat modification is "moderate". This suggests that the lake's hydro-morphology is moderately impacted by impacted by the pressures from anthropogenic and non-natural land uses. By comparison, this score ranks lower than Lake Chivero (32) but higher than Malilangwe reservoir (16) and Cleveland Reservoir (10), all of which are tropical water bodies in Zimbabwe surveyed by Dalu *et al.* (2013; 2016). The LHMS score for Lake Simbi ranked higher than most Victorian lakes in Australia surveyed by EPA Victoria (2009), (such as Hattah and Locke which both scored 0 – no modification) and lower than some Victorian lakes (Reedy Lake and Longmore Lagoon which both scored 36- greater modification). Lake Simbi is impacted by numerous pressures including widespread erosion, extensive agricultural activities, grazing, logging, indiscriminate dumping of wastes, and quarry/mining of salt locally known as *bala* and encroachment by residential developments. Despite harboring these numerous pressures, the habitat of Lake Simbi was found to be moderately modified because these pressures are still operating at marginal scales. The lake suffers from constant pollution from the dumping of wastes both medical and domestic from the nearby health dispensary, schools and surrounding homes, sedimentation resulting from widespread erosion along its banks occasioned by salt (*bala*) mining and grazing, encroachment of the riparian zones through construction of residential homes by the local community, and deforestation from clearing of bushes to create space for agricultural activities. This encroachment and neglect by responsible government agencies results from the fact that the lake is viewed by the locals as a “dead lake” since it doesn’t support fisheries.

The LHS survey of Lake Simbi generally established that the physical habitat quality of the lake is moderately pristine since its hydromorphology is moderately modified by the numerous pressures which are operating at marginal scales. The continuous activity of these anthropogenic invasions coupled with climate change impacts will in the future assessments, make Lake Simbi score low on LHQA scale and high on LHMS scale, generally indicating an impaired and degraded ecological integrity unless appropriate and effective measures are put in place to eliminate the identified pressures and combat their associated impacts on the lake’s ecological condition.

The scrubs and shrub vegetation is typical of semi-arid conditions around the lake characterized by high temperatures, low rainfall and clay soil type. No aquatic plants were identified in the Lake’s habitat since their growth is restricted by both elevated salinity and limited concentration of dissolved oxygen in the water.

The USEPA (2016) regards sediments as a crucial “physical habitat indicator of biological stress”. Sediments make up an important part of the lake habitat since they provide substrates for macrophytes growth which in turn provides shelters for fish avoiding predation as well as food for some macro-invertebrates. Kaufmann and Whittier (1997) opined that the size and properties of a substrate are significant contributing factors to the habitat behavior for fish and other aquatic organisms such as macro-invertebrates because they influence hydrologic alterations. The shoreline substrate material observed in Lake Simbi was predominantly fine sediments of silt and clay observed in all the Hab-plots with coverage of above 40%. These fine sediments indicate a response to the latest alterations in the flow and sediment supply resulting from increased land use practices such as intense agricultural activities, grazing, deforestation, quarry/mining of salt locally known as *bala* and construction of residential developments occurring within the catchment including in the riparian areas all around Lake Simbi. These unsustainable anthropogenic activities have increased erosivity which then results in high fine sediment load deposited by floods into Lake Simbi. This sedimentation coupled with high evaporation rates threatens the existence of this lake since being an endorheic, it lacks the mechanisms for getting rid of some of this sediments which end up building up overtime on the it's floor. The USEPA (2016) warns that such excess buildup of fine sediments can block the habitat spaces between the much larger substrates in the water such as rocks thereby impairing those habitats by suffocating the aquatic organisms that live in them together with their breeding grounds.

The lands cover and land uses in Lake Simbi catchment area were accurately categorized through remote sensing images. The extent of some of the pressures such as crop lands and built areas on the lake's habitat could also be clearly identified from the GIS remote sensed images. Even though the remote sensed information couldn't provide finer details on the structure of the vegetation cover, it proved to be an important tool that should be used to complement the field assessment in carrying out the Lake Habitat Survey (LHS). It's important to note that the remote sensing by GIS can't be used alone as the only method for carrying out LHS assessments. This is the third study in the tropics that has tested the applicability of LHS in evaluating the ecological status of a water body. The two earlier studies had tested it in the ecological monitoring of small reservoirs in Zimbabwe (Dalu *et al.,* 2013, 2016). This study therefore recommends that since Kenya (and generally Africa) lacks any environmental monitoring tool for the lakes' habitats (Rowan *et al.*, 2005), the LHS should be adopted as a national standard tool for assessing ecological health of water bodies of conservation value in Kenya (such as Ramsar sites) because it possesses crucial indices useful for easier and rapid tracking of the scale of anthropogenic modifications which can inform rapid decision making by the management.

The LHS technique is effective for monitoring ecological condition of tropical water bodies for management and conservation purposes since it can help the relevant government agencies identify the water bodies at *risk* of habitat quality degradation so that they prioritize them for conservation programs. However, the LHS falls short in some aspects as an environmental monitoring tool for aquatic ecosystems. It fails to integrate the measurements for the basic physico-chemical variables into the scoring system of any of the two indices (LHQA and LHMS) despite having it as a requisite part of LHS. These measurements ought to be compared with the relevant international standards as a baseline and integrated into the scoring criteria for rating the most appropriate index. In agreement with Dalu *et al*. (2013), since LHS was initially developed for temperate regions, it requires further improvements so as to strengthen its effectiveness in environmental monitoring and management for aquatic ecosystems in the tropics.

CHAPTER SIX

CONCLUSION AND RECOMMENDATIONS

6.1 Introduction

This chapter presents conclusions based on the results already discussed in the previous chapter, before making recommendations on the way forward on the management of Lake Simbi and finally suggesting areas that requires further research.

6.2 Conclusions

The study established that the water quality of Lake Simbi is heavily polluted and unsafe for any domestic usage since almost all the basic water quality variables (DO, pH, TDS, alkalinity, hardness and turbidity) measured exceeded the maximum permissible limits set by both NEMA (2006) and WHO (2011). It was established though, that it could support the demands of other aquatic life including fish. It can also be used for large scale commercial irrigation purposes. All the selected water quality (physico-chemical) parameters (with exception of pH) exhibited significant temporal variations but not significant spatial variations. These findings therefore didn't support the stated null hypothesis on the spatial aspect but supported the stated null hypothesis on the temporal aspect. The variations observed in the water quality variables were found to be highly responsible for the phytoplankton composition and succession in Lake Simbi with special mention of the occurrence of toxic algal blooms.

Six phytoplankton families were recorded in Lake Simbi of which the Cyanophyceae family was the most dominant. Even though the Cyanophyceae still dominated the waters in Lake Simbi, this study observed a shift in the dominance of the cyanobacterial composition whereby the cyanobacterial species which form the primary food resources for the lesser flamingos (*Arthrospira fusiformis* and *Anabaenopsis abijatae*) were overtaken by the toxin-producing *Microcystis* species (microcystins) and *Anabaena species (*anatoxin-a) in terms of abundance. The decreasing food resources coupled with food contamination by the toxins might have caused the migration and deaths of some flamingos respectively, hence the declining population observed in Lake Simbi over the recent years. All the phytoplankton characteristics of biomass, density and diversity indices measured in Lake Simbi exhibited no significant spatial and temporal variations. These findings therefore generally supported the stated null hypothesis, so the study accepted null hypothesis.

Lake Simbi was diagnosed to be hypereutrophic by both CTSI and Nygaard's trophic state indices which reflects the magnitude of nutrients concentration in the lake and the poor state of its ecosystem. The Redfield's TN/TP ratios established that N was the limiting nutrient, which is characteristic of highly eutrophic lakes. The symptom of this hyper-eutrophic condition was observed on the proliferation of cyanobacterial bloom. This further confirmed that the waters are of poor quality. With the exception of TSI (SD), all the trophic state indicators exhibited no significant spatial variations but shown highly significant temporal variations with TSI (SD) included. These findings therefore didn't support the stated null hypothesis on the spatial aspect but supported the stated null hypothesis on the temporal aspect.

On the LHS carried out determine the quality of physical habitat of Lake Simbi, the lake scored much higher on the LHQA index with a score of 70/112 (which is indicative that the habitat quality is "good") and relatively lower on the LHMS index with a score of 18/42 (which is indicative that the habitat modification is "moderate"). These indices collectively suggest that the Lake Simbi physical habitat can be regarded as a natural ecosystem despite the numerous pressures it experiences since these human-induced pressures are still operating on relatively smaller scales. The two habitat quality survey indices didn't show any statistical temporal or spatial variations since they are designed to evaluate the physical habitat of the lake as a whole and therefore the findings didn't support the stated null hypothesis which is therefore rejected.

Several relationships were established between various water quality parameters, phytoplankton characteristics and the trophic state index, which therefore makes the stated null hypothesis rejected. Generally, the results indicated that Lake Simbi suffers from poor water quality, algal bloom, hyper-eutrophication and an increasingly deteriorating physical habitat quality hence a retarded the ecosystem which can be attributed to the unsustainable anthropogenic activities and the changing environmental conditions occurring in its catchment.

Finally, this study concluded that multivariate ecological integrity indices provides an effective, easier and rapid tracking of the ecological status of aquatic environments which can inform rapid decision making for conservation and management purposes and should therefore be adopted for lake environmental assessments and monitoring.

6.3 Recommendations.

6.3.1 Lake Simbi Management

For a proper and effective conservation and management of Lake Simbi ecosystem, the following recommendations can be made to the relevant government agencies:

- The Kenya Wildlife Service (KWS) and the Homabay County Government both have the responsibility of conserving and managing the Lake Simbi national bird sanctuary and so they should cooperate in the protection of Lake Simbi ecosystem from anthropogenic encroachment activities such as agricultural practices, deforestation, illegal waste dumping and residential developments which threaten the biodiversity in the lake. This can be achieved by barbed wire-fencing the lake to bar unauthorized access.
- This study recommends that since Kenya lacks any environmental monitoring tool for the lakes' habitats, the LHS should be adopted as a national standard tool for assessing ecological health of water bodies of conservation value in Kenya (such as Ramsar sites) because it possesses crucial indices useful for easier and rapid tracking of the scale of anthropogenic intrusions which can inform rapid decision making by the management.
- The high erosion rates causing sedimentation problem in the Lake can be cushioned through sustainable agricultural practices, soil conservation measures and protection of the sources of water such as streams in the catchment area of Lake Simbi ecosystem.
- To remedy the hyper-eutrophic status of the lake, an efficient multi-sectoral plan should be adopted to monitor and control nutrients and other pollutants input into the

lake. This can be done through the establishment of vegetation around the lake to act as buffers to the nutrient loads flowing from the nearby agricultural farms.

- The local communities and the conservation groups such as the Lake Simbi Conservancy group should be enlightened on the impacts of their anthropogenic undertakings on the Lake's ecosystem and they affect water quality and hence aquatic life. By doing this, they then can be involved in the advocacy, conservation and management of the ecosystem.
- Generally, activities that help to improve and maintain ecosystem integrity need to be adopted by all stakeholders to promote sustainability of all water resources in Kenya.
- Lake Simbi national sanctuary is a natural resource with great economic potential and should therefore be effectively and sustainable harnessed to generate revenue and employment opportunities which can improve the quality of life of the local community.

6.3.2 Suggested areas for further research

To better understand the ecological status of Lake Simbi national bird sanctuary, the following aspects requires further research:

- The zooplankton assemblages and the macroinvertebrates should be investigated since they were not part of the present study.
- Since the present study has revealed that Lake Simbi is dominated by cyanobacteria, there's need to conduct toxicology studies to help precisely identify the cyanotoxins that might be harmful to the already "nearly threatened" lesser flamingos and other aquatic life in its ecosystem.
- Constant and periodic ecological integrity surveillances should be carried out on both the biotic and abiotic elements of Lake Simbi ecosystem to always screen for negative changes that deserves prompt remedial measures.
- Currently, Lake Simbi doesn't support fisheries but there could be a potential for fisheries in it and so further studies should be carried out by fisheries specialists to ascertain the potential for fish introduction.
- Lastly, a similar study may be replicated using many more sampling stations over a longer length of sampling period like two years or so for comparability of results.

REFERENCES

Abubakar, S. (2017). *Aspects of Physico-Chemical Parameters and Phytoplankton Assemblage of River Saye, Zaria, Kaduna State, Nigeria* (Master's Thesis). Ahmadu Bello University, Zaria, Nigeria.

Adakole, J. A., Abulode, D. S., & Balarabe, M. L. (2009). Assessment of water quality of a man-made lake in Zaria, Nigeria. *Proceedings of Taal*, the 12th World Lake Conference, 1373-1382.

Adhoni, S. A., Shivasharan, C. T., & Kaliwal, B. B. (2015). Physicochemical studies of freshwater lakes in Dharwad and screening of algal species for biofuel production. *European Journal of Experimental Biology*, 5(4):60-67

Adoni, A. D., Joshi, G., Ghosh, K., Chourasia, S. K., Vaishya, A. K., & Yadav, M. (1985). Work book on Limnology, Department of Botany, Dr. *Harisingh Gour Vishwavidyalaya Sagar, India*. pp. 1 -166

American Public Health Association (APHA). (2012). *Standard Methods for the Examination of Water and Wastewater, 22nd edition*. American Public Health Association, Washington, DC.

Ansa-Asare, O. D., & Asante, K. A. (1998). A comparative study of the nutrient status of two reservoirs in southeast Ghana. *Lakes & Reservoirs: Research & Management*, 3(3-4), 205-217.

ANZECC & ARMCANZ (1994b). Water quality management: an outline of the policies. In: *National Water Quality Management Strategy.* Australian and New Zealand Environment and Conservation Council and Agricultural and Resource Management Council of Australia and New Zealand. Retrieved from https://www.environment.gov.au>files. [On 6/8/2018]

ANZECC & ARMCANZ (2000). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australian and New Zealand Environment and Conservation Council and Agricultural and Resource Management Council of Australia and New Zealand. Retrieved from https://www.environment.gov.au>files. [On 6/8/2018]

Averett, R. C., & Schroder, L. J. (1994). *A guide to the design of surface-water-quality studies* (No. 93-105). US Geological Survey: USGS Earth Science Information Center, Open-File Reports Section, 39 p.

Bade, D.L. (2004). Ecosystem Carbon cycles: Whole lake fluxes estimated, with multiple isotopes. *Journal of Geophysical Research*, 62: 321-329.

Ballot, A., Krienitz, L., Kotut, K., Wiegand, C., & Pflugmacher, S. (2005). Cyanobacteria and cyanobacterial toxins in the alkaline crater lakes Sonachi and Simbi, Kenya. *Harmful algae*, *4*(1), 139-150.

Bartram, J., & Ballance, R. (Eds.). (1996). *Water quality monitoring: a practical guide to the design and implementation of freshwater quality studies and monitoring programmes*. CRC Press.

Biddle, C. (2008). Oxygen: the two-faced elixir of life. *AANA journal*, *76*(1).

Bildstein, K. L., Golden, C. B., McCraith, B. J., & Bohmke, B. W. (1993). Feeding behavior, aggression, and the conservation biology of flamingos: integrating studies of captive and free-ranging birds. *American Zoologist*, *33*(2), 117-125.

Birdlife International (2012) Phoeniconaias minor. The IUCN Red List of Threatened Species. Version 2014.2 Retrieved from www.iucnredlist.org. [On 20/6/2018]

Boesch, D. F., Brinsfield, R. B., & Magnien, R. E. (2001). Chesapeake bay eutrophication. *Journal of Environmental Quality*, *30*(2), 303-320.

Booker, M. J., & Walsby, A. E. (1981). Bloom formation and stratification by a planktonic blue-green alga in an experimental water column. *British Phycological Journal*, *16*(4), 411-421.

Breen, C.M., Chutter, F.H., Merona, B. & Saad, M.A.H. (1981) Rivers. In: Symoens, J.1. Burgis, M. & Gaduet, J.1. (eds.), The Ecology and Utilization of African Inland Waters. UNEP Reports and Proceedings, 1: pp 83-92.

Brönmark, C., & Hansson, L. A. (2005). The Biology of Lakes and Ponds, –Oxford University Press.

Brown, L. (1973). *The mystery of the flamingos* (p. 116). London: Country Life.

Carlson, R. E. 1977. A trophic state index for lakes. Limnology and Oceanography, 22 (2):361- 369.

Carlson, R. E., & Simpson, J. (1996). A coordinator's guide to volunteer lake monitoring methods. *North American Lake Management Society*, *96*, 305.

CEN (Comité Européen de Normalisation – European Standards Agency). 2011. *Water quality: guidance standard on assessing the hydromorphological features of lakes*. Ref. No. EN 16039:2011. Brussels: European Standards Agency.

Chapman, L. J., & Kramer, D. L. (1991). Limnological observations of an intermittent tropical dry forest stream. *Hydrobiologia*, *226*(3), 153-166.

Childress, B., Nagy, S., Hughes, B., & Abebe, Y. B. (2008). *International Single Species Action Plan for the Conservation of the Lesser Flamingo (Phoeniconaias minor).* CMS Technical Series No. 18, AEWA Technical Series No. 34. Bonn, Germany.

Clark, E. H., Haverkamp, J. A., & Chapman, W. (1985). *Eroding soils: the off-farm impacts.* The Conservation Foundation, Washington, D.C. pp 252.

Cocquyt , C., Vyverman, W., & Compére, P., (1993). A Check-List of the Algal Flora of the East African Great Lakes; Lake Malawi, Lake Tanganyika and Lake Victoria. Scripta Bot. Belg. 8:56 pp.

Codd, G. A., Bell, S. G., Kaya, K., Ward, C. J., Beattie, K. A., & Metcalf, J. S. (1999). Cyanobacterial toxins, exposure routes and human health. *European Journal of Phycology*, *34*(4), 405-415.

Cohen, A. S. (2003). *Paleolimnology: the history and evolution of lake systems*. Oxford University Press.

Dallas, H. F., & Day, J. A. (2004). *The effect of water quality variables on aquatic ecosystems: a review*. Pretoria: Water Research Commission, 221pp.

Dalu, T., Clegg, B., & Nhiwatiwa, T. (2013). Application of the Lake Habitat Survey method on the tropical Malilangwe reservoir, south-eastern lowveld, Zimbabwe. *African journal of aquatic science*, *38*(1), 21-34.

Dalu, T., Tambara, E., Chari, L. D., Moyo, S., & Nhiwatiwa, T. (2016). A test of the Lake Habitat Survey method in Cleveland Reservoir and Lake Chivero (Manyame River Basin, Zimbabwe). *Water SA*, *42*(1), 102-111.

David C. L., & William M. S. (1999). *Potential climate change effects on Great Lakes hydrodynamics and water quality*. Ed. ASCE Publications, 232 p

De Deckker, P. (1988). Biological and sedimentary facies of Australian salt lakes. *Palaeogeography, Palaeoclimatology, Palaeoecology*, *62*(1-4), 237-270.

Defew, E. C., Perkins, R. G., & Paterson, D. M. (2004). The influence of light and temperature interactions on a natural estuarine microphytobenthic assemblage. *Biofilms*, *1*(1), 21-30.

Deloitte. (2014). *The Lake Region Economic Blueprint: A better life.* Retrieved from https://www2.deloitte.com/content/dam/Deloitte/ke/Documents/public-sector/PS_Lake%20Basin_DeloitteKenya.pdf [On 10/5/2018]

Demenocal, P. B. (1995). Plio-pleistocene African climate. *Science*, *270*(5233), 53-59.

Deocampo DM, Jones BF (2014) Geochemistry of saline lakes. In: Holland HD, Turekian KK (eds) Treatise on geochemistry, vol 7, 2nd edn. Elsevier, Oxford, pp 437–469

Deocampo, D. M. (2010). The geochemistry of continental carbonates. *Developments in Sedimentology*, *62*, 1-59.

Deocampo, D. M., & Renaut, R. W. (2016). Geochemistry of African soda lakes. In *Soda Lakes of East Africa* (pp. 77-93). Springer, Cham.

Department of Environmental Quality-Oregon, (2015). Nitrate in drinking water. Fact Sheet. State of Oregon. Retrieved from www.deq.state.or.us. [On 26/7/2018]

Dhanam, S., Sathya, A., & Elayaraj, B. (2016). Study of physico-chemical parameters and phytoplankton diversity of Ousteri lake in Puducherry. *World Scientific News*, *54*, 153-164.

Dhanya S. (2017). *Trophic state and phytoplankton blooms of shallow tropical ponds- a case study from Pallippuram, Kerala* (Doctorate Thesis). Cochin University of Science and Technology, Kerala, India.

Dinnes, D. L., Karlen, D. L., Jaynes, D. B., Kaspar, T. C., Hatfield, J. L., Colvin, T. S., & Cambardella, C. A. (2002). Nitrogen Management Strategies to Reduce Nitrate Leaching in Tile-Drained Midwestern Soils. *Agronomy Journal 94*:153-171.

Dodds, W. K., & Cole, J. J. (2007). Expanding the concept of trophic state in aquatic ecosystems: it's not just the autotrophs. *Aquatic Sciences*, *69*(4), 427-439.

Domingues, R. B., Barbosa, A. B., Sommer, U., & Galvão, H. M. (2012). Phytoplankton composition, growth and production in the Guadiana estuary (SW Iberia):

Unraveling changes induced after dam construction. *Science of the Total Environment*, *416*, 300-313.

Dupont, D. P. (1992). Economic assessment of the performance of alternative environmental policy instruments as they pertain to agriculture and water quality. *Agriculture and water quality*, 23-24.

Earman, S., Phillips, F. M., & McPherson, B. J. (2005). The role of "excess" CO2 in the formation of trona deposits. *Applied Geochemistry*, *20*(12), 2217-2232.

EPA Victoria (2009). A Snapshot of the environmental condition of Victorian lakes. EPA Victoria (EPA publication 1303). Melbourne. Retrieved from https://www.epa.vic.gov.au/~/media/Publications/1303.pdf [On 1/9/2018]

EPA Victoria (2010). Environmental quality guidelines for Victorian lakes (EPA publication 1303), Melbourne. Retrieved from http://www.epa.vic.gov.au/~/media/Publications/1303.pdf [On 1/9/2018]

EPA. (2012). Total Dissolved Solids. In Water: Monitoring and Assessment. Retrieved from http://water.epa.gov/type/rsl/monitoring/vms58.cfm. [On 6/12/2018]

Eugster, H. P., & Hardie, L. A. (1978). Saline lakes. In *Lakes*(pp. 237-293). Springer, New York, NY.

Eugster, H. P., & Jones, B. F. (1979). Behavior of major solutes during closed-basin brine evolution. *American journal of science*, *279*(6), 609-631.

European Commission. (2000). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. *Official Journal of the European Communities* . 2002, *L327*, 66–106.

Ewa, E.E., Iwara, A.I. and Ekelemu, J.K. (2013). Spatio-temporal Distribution of Phytoplankton in the Industrial area of Calabar River. *Advances in Environmental Biology*, 7(3), 466-470.

Fatoki, O. S., and Giovane, P. (2003). Pollution assessment in the Keiskamma River and in the impoundment downstream. *Water SA*, *29*(2), 183-188.

Finlay, B. J., Curds, C. R., Bamforth, S. S., & Bafort, J. M. (1987). Ciliated protozoa and other microorganisms from two African soda lakes (Lake Nakuru and Lake Simbi, Kenya). *Archiv für Protistenkunde*, *133*(1-2), 81-91.

Flöder, S., Urabe, J., & Kawabata, Z. I. (2002). The influence of fluctuating light intensities on species composition and diversity of natural phytoplankton communities. *Oecologia*, *133*(3), 395-401.

Florida LAKEWATCH (2018). Trophic State: A Waterbody's Ability to Support Plants Fish and Wildlife at http://lakewatch.ifas.ufl.edu/LWcirc.html. [On 28/6/2018]

Fogg, G. E. (1975). Algal Cultures and Phytoplankton Ecology, 2nd edn. 175 pp.

Fondriest Environmental, Inc. "Conductivity, Salinity and Total Dissolved Solids." Fundamentals of Environmental Measurements. Retrieved from https://www.fondriest.com/environmental-measurements/parameters/water-quality/conductivity-salinity-tds/. [On 6/4/2019]

Forster, J., Hirst, A. G., & Atkinson, D. (2012). Warming-induced reductions in body size are greater in aquatic than terrestrial species. *Proceedings of the National Academy of Sciences*, *109*(47), 19310-19314.

Garg, R. K., Saksena, D. N., & Rao, R. J. (2006). Assessment of physico-chemical water quality of Harsi Reservoir, district Gwalior, Madhya Pradesh. *Journal of Ecophysiology and Occupational Health*, *6*(1), 33-40.

George Tchobanoglous & Edward D Schroeder. (1987). *Water quality: characterization, modeling, modification* (Master's Thesis). University of California, California, USA

Gilbert, L. (1991). *The Ecology of Urban Habitats.* Chapman and Hall, London. 369p.

Githaiga, J.M. (2003). *Ecological factors determining utilization patterns and inter-lake movement of lesser flamingos (Phoeniconaias minor) in Kenyan alkaline lakes (Doctorate Thesis)*. University of Nairobi, Nairobi, Kenya.

Goldman CR, Horne AJ (1983) Limnology. McGraw-Hill, New York, p 464

Grace. A., & Michael, S. (2010). Phytoplankton productivity in newly dug fish ponds within Lake Victoria wetlands (Uganda). *African Journal of Environmental Science and Technology*, *4*(6).

Grant, W. D. (2004). Introductory chapter: half a lifetime in soda lakes. In *Halophilic microorganisms* (pp. 17-31). Springer, Berlin, Heidelberg.

Grant, W. D., Mwatha, W. E., & Jones, B. E. (1990). Alkaliphiles: ecology, diversity and applications. *FEMS Microbiology Reviews*, *6*(2-3), 255-269.

Guinotte, J. M., & Fabry, V. J. (2008). Ocean acidification and its potential effects on marine ecosystems. *Annals of the New York Academy of Sciences*, *1134*(1), 320-342.

Guldin, R. W. (1989). An analysis of the water situation in the United States: 1989-2040. *US Government Documents (Utah Regional Depository)*, 507.

Gupta, S. K. & Gupta, R. C. (2006). *General and Applied Ichthyology (Fish and Fisheries)* S. Chand and Campany Ltd. Ram Niger, New Delhi, p 1130.

Hall, L. S., Krausman, P. R., & Morrison, M. L. (1997). The habitat concept and a plea for standard terminology. *Wildlife Society Bulletin*, 173-182.

Haller, W. T., Sutton, D. L., & Barlowe, W. C. (1974). Effects of salinity on growth of several aquatic macrophytes. *Ecology*, *55*(4), 891-894.

Hammer, U. T., 1981.Primary production in saline lakes. *Hydrobiologia*. 81: 47-57.

Harding, L. W., & Jr. Perry, E. S. (1997). Long-term increase of phytoplankton biomass in Chesapeake Bay, 1950-1994. *Marine Ecology Progress Series*. *157*:39-52.

Hargrave, J. E., Hicks, M. K., & Scholz, C. A. (2014). Lacustrine carbonates from Lake Turkana, Kenya: a depositional model of carbonates in an extensional basin. *Journal of Sedimentary Research*, *84*(3), 224-237.

Harper, D.M., Childress, R.B., Harper, M.M., Boar, R.R., Hickley, p., Mills, S.C., Otieno, N., Drane, T., Vareschi, E., Nasirwa, O., Mwatha, W.E., Darlington, J.P. and Escute-Gasulla, X. (2003). Aquatic biodiversity and saline lakes: Lake Bogoria National Reserve, Kenya. *Hydrobiologia.* 500: 259-276.

Harris, G.B., Piccinin, B.B & Ryn, J. (1983) Physiological and environmental constraints in the ecology of the planktonic dinoflgellate *Ceratium hirundinella. Archiv. Hydrobiol.*, 98: 215-39.

Hayombe, P., Agong', M., Mossberg, L., Malbert, B., & Odede, F. A. (2014). Simbi Nyaima: An Interplay of Nature and Mythology in the Lake Victoria Region; Planning and Management for Ecotourism Transformation in Kenya. *Journal of Arts and Humanities*, *3*(1), 173-187.

Hecky, R. E., & Kilham, P. (1988). Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment 1. *Limnology and Oceanography*, *33*(4part2), 796-822.

HELCOM (2017). Manual for Marine Monitoring in the COMBINE Programme of HELCOM, Annex C-6: Guidelines concerning phytoplankton species composition, abundance and biomass. 1-17. Retrieved from http://www.helcom.fi/Lists/Publications/Guidelines%20for%20monitoring%20phytoplankton%20species%20composition,%20abundance%20and%20biomass.pdf [On 24/6/2018]

Huber-Pestalozzi, G., (1968). *Das Phytoplankton des Süsswassers. Systematik and Biologie*. 3. Teil: Cryptophyceae, Chloromonadophyceae, Dinophyceae. E. Schweizerbart'sche Verlagsbuchhandlung (Nägele u Obermiller), Stuttgart, Germany. pp 322.

Hunt, R. J. and Christiansen, I. H. (2000). Dissolved oxygen information kit. A CRC sugar *Technical Publication.* CRC for sustainable sugar production, Townsville, pp 27.

Hutchinson, G. E. (1944). Limnological studies in Connecticut. VII. A critical examination of the supposed relationship between phytoplakton periodicity and chemical changes in lake waters. *Ecology*, *25*(1), 3-26.

Huynh, M., & Serediak, N. (2006). Algae Identification Field Guide Agriculture and Agric Food Canada. In *Report number: Cat. No. A125-8/2-2011E-PDF*. 40 pages.

Ibelings, B. W., Gsell, A. S., Mooij, W. M., Van Donk, E., Van Den WYNGAERT, S. I. L. K. E., & De Senerpont Domis, L. N. (2011). Chytrid infections and diatom spring blooms: paradoxical effects of climate warming on fungal epidemics in lakes. *Freshwater Biology*, *56*(4), 754-766.

Idowu, E. O., & Ugwumba, A. A. A. (2005). Physical, Chemical and Benthic Faunal Characteristics of a Southern Nigerian Reservoir. *The Zoologist, 3,* 15 – 25.

Idu, A. J., (2015). Threats to water resources development in Nigeria. *Journal of Geology and Geophysics*. 4(3), 1-10.

Indabawa .I.I. (2009). Studies on Limnological parameters and Phytoplankton dynamic of Nguru lake Yobe state Nigeria. *Bioscience Resources Communication,* 21(4): 57 - 80.

International Union for Conservation of Nature (IUCN). 2010. IUCN Red List of Threatened Species. 2010. Retrieved from http://www.iucnredlist.org/apps/redlist/details/144723/0 [On 16/3/2018]

Ivanković, A., Ćosić, N., Knezović, Z., & Vasilj, V. (2018). Trophic Status of Blidinje Lake (Bosnia and Herzegovina) Based on the Determination of the Trophic State Index (TSI). *Ekológia (Bratislava), 37*(1), 1-10.

Jaji, M. O., Bamgbose, O., Odukoya, O. O., & Arowolo, T. A. (2007). Water quality assessment of Ogun River, south west Nigeria. *Environmental monitoring and assessment*, *133*(1-3), 473-482.

James V. Q., Vallarino, L. M. (1969). *Chemistry*. Ed. Prentice-Hall, University of California. 844p

Jekatierynczuk-Rudczyk, E., Zieliński, P., Grabowska, M., Ejsmont-Karabin, J., Karpowicz, M., & Więcko, A. (2014). The trophic status of Suwałki Landscape Park lakes based on selected parameters (NE Poland). *Environmental monitoring and assessment*, *186*(8), 5101-5121.

Jenkin, P. M. (1957). The filter-feeding and food of flamingoes (Phoenicopteri). *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, 401-493.

Jeppesen, E., Peder Jensen, J., SØndergaard, M., Lauridsen, T., & Landkildehus, F. (2000). Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. *Freshwater biology*, *45*(2), 201-218.

Jones, B. E., & Grant, W. D. (1999). Microbial diversity and ecology of the Soda Lakes of East Africa. In *Microbial Biosystems: New Frontiers: Proceedings of the 8th International Symposium for Microbial Ecology (2000). Atlantic Canada Society for Microbial Ecology, Halifax, Canada* (pp. 681-687).

Jørgensen, S.E. (1980). *Lake management water development, supply and management* (Master's Thesis). University of Copenhagen, Denmark.

Kaggwa, M. N., Burian, A., Oduor, S. O., & Schagerl, M. (2012). Ecomorphological variability of A rthrospira fusiformis (C yanoprokaryota) in A frican soda lakes. *MicrobiologyOpen*, *2*(5), 881-891.

Kansas Department of Health and Environment Bureau of Water. (KDHEBW) (2011). Water Quality Standards White Paper. Chlorophyll-a Criteria for Public Water Supply Lakes or Reservoirs. pp 1 – 8.

Kaufmann, P. R., Hughes, R. M., Van Sickle, J., Whittier, T. R., Seeliger, C. W., & Paulsen, S. G. (2014c). Lakeshore and littoral physical habitat structure: A field survey method and its precision. *Lake and Reservoir Management*, *30*(2), 157-176.

Kaufmann, P. R., Peck, D. V., Paulsen, S. G., Seeliger, C. W., Hughes, R. M., Whittier, T. R., & Kamman, N. C. (2014a). Lakeshore and littoral physical habitat structure in a national lakes assessment. *Lake and Reservoir Management*, *30*(2), 192-215.

Kaufmann, P.R., & Whittier, T.R. (1997). Habitat characterization. Section 5. In: Baker, J.R., Peck, D.V., Sutton, D.W., (eds). Environmental Monitoring and Assessment Program Surface Waters: Field Operations Manual for Lakes. U.S. Environmental Protection Agency, Washington, D.C. EPA/620/R-97/001

Kenya Wildlife Servce, KWS. (2018). "For the Birds". Retrieved from http://www.kws.go.ke/content/lake-simbi-national-sanctuary [On 6/5/2018]

Kigamba, B.M. (2005). Limnological Studies in a fish pond at Birdman, West Bengal, India, Japan. *Journal of Limnology*, 15(2), 10-18.

Kingsolver, J. G. (2009). The Well-Temperatured Biologist: (American Society of Naturalists Presidential Address). *The American Naturalist, 174*(6), 755-768.

Kotut, K., Krienitz, L., & Muthuri, F. M. (1998). Temporal changes in phytoplankton structure and composition at the Turkwel Gorge Reservoir, Kenya. *Hydrobiologia*, *368*(1-3), 41-59.

Krienitz, L., Dadheech, P. K., & Kotut, K. (2013). Mass developments of the cyanobacteria Anabaenopsis and Cyanospira (Nostocales) in the soda lakes of Kenya: ecological and systematic implications. *Hydrobiologia*, *703*(1), 79-93.

Kyalo, M. (2012). *Assessment of Alternative Food Resources of The Lesser Flamingo (Phoeniconaias minor) in some Rift Valley Saline Lakes in Kenya and Tanzania* (Master's Thesis). University of Nairobi, Nairobi, Kenya.

Latinopoulos, D., Ntislidou, C., & Kagalou, I. (2016). A multi-approach Lake Habitat Survey method for impact assessment in two heavily modified lakes: A case of two Northern Greek lakes. *Environmental monitoring and assessment*, *190*(11), 658.

Lewis, W.M. (1996) Tropical lakes: How low latitude lakes a difference. In: Sciemer, F. & Boland, K.T. (eds.), Perspectives in tropical limnology, 43-64. SPD Academic Publishing, B.V. Amsterdam.

Liu, Y., Fan, Y. W., & Wang, Q. X. (2015). Study on the achnanthoid diatoms from the Great Xing'an moutains. *Acta Hydrobiologica Sinica*, *39*(3), 554-563.

Lugonzo M. (2014). *Alkaline active α-amylases from alkaliphilic bacillus sp.: screening and enzyme production* (Master's Thesis). University of Nairobi, Nairobi, Kenya.

Maitland, P.S. (1994) Biology of fresh waters (2nd edition). Chapman and Hall, London.

Manahan, S. E. (1993). Fundamentals of environmental chemistry.

Manuel, S. E., Colmenarejo, M. F., Vicente, J., Rubio, A., García, M. G., Travieso, L., & Borja, R. (2007). Use of the water quality index and dissolved oxygen deficit as simple indicators of watersheds pollution. *Ecological indicators, 7*(2), 315-328.

Margalef R. (1978). Life forms of phytoplankton as survival alternatives in an unstable environment. *Oceanologia Acta.* 1: 493-509

Margalef, R. (1958). Temporal succession and spatial heterogeneity in phytoplankton. *Perspectives in marine biology*, 323-349.

Marshall, H. G., Lacouture, R. V., Buchanan, C., & Johnson, J. M. (2006). Phytoplankton assemblages associated with water quality and salinity regions in Chesapeake Bay, USA. *Estuarine, Coastal and Shelf Science*, *69*(1-2), 10-18.

Mason, C. F. (1991). Biology of Fresh Water Pollution. 2nd Ed., Wesley and Sons Inc., New York, pp: 351.

Masters, G. M., & Ela, W. P. (1991). *Introduction to environmental engineering and science* (Vol. 3). Englewood Cliffs, NJ: Prentice Hall.

McCulloch, G. P., Irvine, K., Eckardt, F. D., & Bryant, R. (2008). Hydrochemical fluctuations and crustacean community composition in an ephemeral saline lake (Sua Pan, Makgadikgadi Botswana). *Hydrobiologia*, *596*(1), 31-46.

McGoff, E., & Irvine, K. (2009). A test of the association between Lake Habitat Quality Assessment and macroinvertebrate community structure. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *19*(5), 520-533.

Medudhula, T.C.H. and Samath, C. (2012). Analysis of water quality using physiochemical parameters in lower manair reservoir of Karignagar district Andhra Pradesh. *International Journal of Environmental Science*, 3: 1-5.

Melack, J. M. (1979). Photosynthesis and growth of Spirulina platensis (Cyanophyta) in an equatorial lake (Lake Simbi, Kenya) 1. *Limnology and Oceanography*, *24*(4), 753-760.

Melack, J. M., & Kilham, P. (1974). Photosynthetic rates of phytoplankton in East African alkaline, saline lakes 1. *Limnology and Oceanography*, *19*(5), 743-755.

Melack, J.M. (1976): *Limnology and dynamics of phytoplankton in equatorial African lakes* (Doctorate Thesis). Duke University, Durham, North Carolina, U.S.A.

Melcher, A. A. (2013). A trophic state analysis of lakes in Yellowstone National Park. *All Theses and Dissertations*. Retrieved from https://scholarsarchive.byu.edu/etd/3522 [On 15/4/2018]

Meybeck, M. (1995). Les lacs et leur bassin. *Limnologie générale*. *Masson*, *Paris*, 7-59.

Mills AA, Sims MR (1995). pH of the Martian surface. *Planetary and Space Science,* 5 (43): 695 - 696

Molles, M. (2005). Ecology: Concepts and Applications. Boston, McGraw Hill. 608 p

Moschini-Carlos, V., Bortoli, S., Pinto, E., Nishimura, P. Y., Gomes de Freitas, L., Pompêo, M. L., & Dörr, F. (2009). Cyanobacteria and cyanotoxin in the Billings reservoir (São Paulo, SP, Brazil). *Limnetica*, *28*(2), 0273-282.

Moss, B. (1994). Brackish and freshwater shallow lakes—different systems or variations on the same theme? In *Nutrient Dynamics and Biological Structure in Shallow Freshwater and Brackish Lakes* (pp. 1-14). Springer, Dordrecht.

Murrell, M. C., & Lores, E. M. (2004). Phytoplankton and zooplankton seasonal dynamics in a subtropical estuary: importance of cyanobacteria. *Journal of Plankton Research*, *26*(3), 371-382.

Mwaura, F., Mavuti, K. M., & Wamicha, W. N. (2002). Biodiversity characteristics of small high-altitude tropical man-made reservoirs in the Eastern Rift Valley, Kenya. *Lakes & Reservoirs: Research & Management*, 7(1), 1-12.

National Environmental Management Authority, NEMA. (2004). "The National Environment (Standards of discharge of effluent into water or on land regulations, S.I.No.153-4)". In *Environmental Legislation of Kenya*, pp 2.

National Geographic (2018). Habitat Needs. Retrieved from https://www.nationalgeographic.org/activity/habitat-needs/ [On 6/7/2018]

National Geographic, (2016). Oceans Information and Facts. Retrieved from https:// https://www.nationalgeographic.com/environment/habitats/ocean/ [On 6/8/2018]

Naumann, E. (1929). The scope and chief problems of regional limnology. *Internationale Revue der gesamten Hydrobiologie und Hydrographie*, *22*(1), 423-444.

Nellemann, C., Kullerud, L., Vistnes, I., & Forbes, B. (2001). *GLOBIO Global methodology for mapping human impacts on the biosphere: the Arctic 2050 scenario and global application*. UNEP. DEWA Technical Report 3.

Newman, J. R., & Barrett, P. R. F. (1993). Control of Microcystis aeruginosa by decomposing barley straw. *Journal of aquatic plant Management*, *31*, 203-203.

Nielsen, J. M., & Dahi, E. (1995). The occurrence of fluoride contaminated magadi (Trona) in Kenya and Tanzania. In *Dahi E, Bregnhoj H. Proceedings of the 1 st International workshop on fluorosis prevention and defluoridation of water.*

Ngurdoto, Tanzania. International Society for Fluorosis Research (ISFR), New Zealand, in collaboration with Centre for Developing Countries (DANIDA).pp 17-22

NMK, KWS (2010). Brochure of the Directorate of Research and Collections. National Museums of Kenya (NMK), Nairobi.

Nowrouzi, S., & Valavi, H. (2011). Effects of environmental factors on phytoplankton abundance and diversity in Kaftar Lake. *Journal of Fisheries and Aquatic Science*, *6*(2), 130-140.

Nygaard, G. (1949). Hydrobiological studies on some Danish ponds and lakes. *Kongl. Dansk Vid. Selskab., Biol. Skr. Kobenhavn, 7*(1), 1-293.

Ochumba, P. B., & Kibaara, D. I. (1988). An instance of thermal instability in Lake Simbi, Kenya. *Hydrobiologia*, *158*(1), 247-252.

Oduor, B. (2018, July 20). Pollution in Lake Simbi Nyaima sends away flamingos. *Daily Nation*. Retrieved from https://www.nation.co.ke/news/Pollution-in-Lake-Nyaima-Simbi-sends-away-flamingos/1056-4672182-10n994k/index.html [On 6/8/2018]

Oduor, S. O., & Schagerl, M. (2007). Phytoplankton primary productivity characteristics in response to photosynthetically active radiation in three Kenyan Rift Valley saline–alkaline lakes. *Journal of Plankton Research*, *29*(12), 1041-1050.

Ogendi, M. (2017). *Effects of Anthropogenic Activities on Water Quality of Nyakomisaro-Riana and Nyanchwa-Riana Rivers Draining Kisii Municipality, South-West Kenya* (Doctorate Thesis) Kisii University, Kisii, Kenya.

Olawale, S. A. (2016). Physico Chemical Analysis of Water from Asa River, Ilorin, Nigeria. *Imperial Journal of Interdisciplinary Research (IJIR)*, *2*, 122-129.

Omernik, J. M. (1976). *The influence of land use on stream nutrient levels* (Vol. 76, No. 14). US Environmental Protection Agency, Office of Research and Development, Corvallis Environmental Research Laboratory, Eutrophication Survey Branch.

Otterbine. (2003). Integrated Pond and Lake Management Manual and Guide. Otterbine Barebo. Emmaus,PA. Retrieved from http://www.otterbine.com/assets/base/resources/PondAndLakeManual.pdf

Paerl, H. W., & Ustach, J. F. (1982). Blue‐green algal scums: An explanation for their occurrence during freshwater blooms1. *Limnology and oceanography*, *27*(2), 212-217.

Pecoraino, G., D'Alessandro, W., & Inguaggiato, S. (2015). The other side of the coin: geochemistry of alkaline lakes in volcanic areas. In *Volcanic lakes* (pp. 219-237). Springer, Berlin, Heidelberg.

Pielou, E. C. (1966). The measurement of diversity in different types of biological collections. *Journal of theoretical biology*, *13*, 131-144.

Pionke, H. B., Gburek, W. J., Sharpley, A. N., & Schnabel, R. R. (1995). Flow and nutrient export patterns for an agricultural hill-land watershed. *Water Resources Research*, *32*(6), 1795-1804.

Prasad, A. D. (2012). Siddaraju. *Carlson's Trophic State Index for the assessment of trophic status of two Lakes in Mandya district, Advances in Applied Science Research*, *3*(5), 2992-2996.

Radulović, S., Laketić, D., Popović, Ž., & Teodorović, I. (2010). Towards candidature of the Crno jezero (Black Lake)(Durmitor, Montenegro) as a high ecological status (HES) site of the dinaric western Balkan Ecoregion. *Archives of Biological Sciences, Belgrade*, *62*(4), 1101-1117.

Rajdeep, R., Anil P., Gauns M. & Naqvi, S.W. (2006). *Estuarine, Coastal and Shelf Science*, 69: 189-195.

Ramachandra, T. V., Rishiram, R., & Karthick, B. (2006). Zooplankton as bioindicators: Hydro-Biological investigations in selected Bangalore lakes. *The Ministry of Science and Technology Government of India Technical Report*, *115*.

Rani, R., Gupta, B. & Srivastava, K. (2004). Studies on water quality assessment in Satna city (M. P.): Seasonal parametric variations. *Nature, Environment and Pollution Technology*, *3*(4), 563-565.

Rao, S. (1989). *Acid stress and aquatic microbial interactions*. Ed. CRC Press.176 pages

Rao, T. R., & Kumar, R. (2003). Patterns of prey selectivity in the cyclopoid copepod Mesocyclops thermocyclopoides. *Aquatic Ecology*, *36*(3), 411-424.

Rasolofomanana, L. V. (2009). *Characterization of Ranomafana lake water quality-antsirabe Madagascar* (Master's thesis). University of Stavanger, Norway.

Redfield, A. C. (1958). The biological control of chemical factors in the environment. *American scientist*, *46*(3), 230A-221.

Renaut, R. W., & Tiercelin, J. J. (1994). Lake Bogoria, Kenya Rift Valley—a sedimentological overview. In: Renaut RW, Last WM (eds) Sedimentology and geochemistry of modern and ancient saline lakes. SEPM Special Publication 50, pp 101–124

Reynolds, C. S. (1984). Phytoplankton periodicity: the interactions of form, function and environmental variability. *Freshwater biology*, *14*(2), 111-142.

Reynolds, C. S. (2005). *The ecology of phytoplankton*. Cambridge University Press.

Richardson, J. L., & Richardson, A. E. (1972). History of an African rift lake and its climatic implications. *Ecological monographs*, *42*(4), 499-534.

Ridley, M. W., Moss, B. L., & Percy, R. C. (1955). The food of flamingoes in Kenya colony. *Journal of East Africa Natural History Society*.22 (5), 147 -158.

Rijsdijk, A. Bruijnzee, L.A & Prins, T.M. (2004) Sediment yield from gullies, riparian mass wasting and bank erosion in the upper Konto catchment, East Java, Indonesia. *Agriculture, Ecosystem and Environment*, 128: 4225-238.

Robarts, R. D., & Zohary, T. (1987). Temperature effects on photosynthetic capacity, respiration, and growth rates of bloom-forming cyanobacteria. *New Zealand Journal of Marine and Freshwater Research*, *21*(3), 391-399.

Roger, A.Y. (2010). The Effects of Turbulence on phytoplankton, Aerospace Technology Enterprise. NTRS. pp 220-230.

Rosen, M. R. (1994). The importance of groundwater in playas: A review of playa classifications and. *Paleoclimate and basin evolution of playa systems*, *289*, 1.

Rowan, J. (2005). Development of a technique for Lake Habitat Survey. Retrieved from http://epa.gov/nheerl/arm/indicators/lhs_report.htm [On 3/7/2018]

Rowan, J. S., Carwardine, J., Duck, R. W., Bragg, O. M., Black, A. R., Cutler, M. E. J., ... & Boon, P. J. (2006). Development of a technique for lake habitat survey (LHS) with applications for the European Union Water Framework Directive. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *16*(6), 637-657.

Rowan, J., Soutar, I., Bragg, O., Carwadine, J., & Cutler, M. (2008) Lake Habitat Survey in the United Kingdom field survey guidance manual version 3.1. Final Report Project WFD 42. Scotland and Northern Ireland Forum for Environmental Research (SNIFFER), Edinburgh, pp. 4-36.

Ryding, S. O., & Rast, W. (1989).. The Control of Eutrophication of Lakes and Reservoirs, UNESCO, Paris, 314 pp.

Salmaso, N., Buzzi, F., Garibaldi, L., Morabito, G., & Simona, M. (2012). Effects of nutrient availability and temperature on phytoplankton development: a case study from large lakes south of the Alps. *Aquatic Sciences*, *74*(3), 555-570.

Schagerl, M., & Oduor, S. O. (2008). Phytoplankton community relationship to environmental variables in three Kenyan Rift Valley saline-alkaline lakes. *Marine and Freshwater Research*, *59*(2), 125-136.

Shannon, C. E., & Wiener, W. (1949). The mathematical theory of communication– University of Illinois Press. *Urbana*, pp *117*.

Sharma, M. P., Kumar, A., & Rajvanshi, S. (2010). Assessment of trophic state of lakes: a case of Mansi Ganga Lake in India. *Hydro Nepal: Journal of Water, Energy and Environment*, *6*, 65-72.

Shekhar, T. S., Kiran, B. R., Puttaiah, E. T., Shivaraj, Y., & Mahadevan, K. M. (2008). Phytoplankton as index of water quality with reference to industrial pollution. *Journal of Environmental Biology*, *29*(2), 233.

Sim, L. L., Davis, J. A., Chambers, J. M., & Strehlow, K. (2006). What evidence exists for alternative ecological regimes in salinising wetlands?. *Freshwater Biology*, *51*(7), 1229-1248.

Skelton P. H. (2001). A complete guide to the freshwater fishes of Southern Africa 2nd edition. Struik Publishers, Cape Town: pp.395.

Smol, J.P. & Stoermer, E.F. (2010). *The Diatoms. Applications for the Environmental and Earth Sciences*. 2nd Edition, Cambridge University Press, Cambridge

Sommer, U., & Stabel, H. H. (1983). Silicon consumption and population density changes of dominant planktonic diatoms in Lake Constance. *Journal of Ecology*, *71,* 119-130.

Stella, J. C., Rodríguez-González, P. M., Dufour, S., & Bendix, J. (2013). Riparian vegetation research in Mediterranean-climate regions: common patterns, ecological processes, and considerations for management. *Hydrobiologia*, *719*(1), 291-315.

Stevenson, R. J., & Pan, Y. (1999). Assessing environmental conditions in rivers and streams with diatoms. *The diatoms: applications for the environmental and earth sciences, 1*(4).

Stevenson, T., & Fanshawe, J. (2002). Field guide to the birds of East Africa. Kenya, Tanzania, Uganda, Rwanda, Burundi. T and A.D. Poyser, London. Pp 34-35.

Talling, J. F., & Driver, D. (1961). Some problems in the estimation of chlorophyll a in phytoplankton, *In* M. S. Doty (ed.), proc. Conf. Primary Production Measurement Marine freshwater, *University of Hawaii, U.S. Atomic Energy Commission Publications*. TID 7633, pp 142-146.

Talling, J. F., Wood, R. B., Prosser, M. V., & Baxter, R. M. (1973). The upper limit of photosynthetic productivity by phytoplankton: evidence from Ethiopian soda lakes. *Freshwater Biology*, *3*(1), 53-76.

Tan, X., Ma, P., Xia, X., & Zhang, Q. (2014). Spatial pattern of benthic diatoms and water quality assessment using diatom indices in a subtropical river, China. *CLEAN–Soil, Air, Water*, *42*(1), 20-28.

Thakur, R. K., Jindal, R., Singh, U. B., & Ahluwalia, A. S. (2013). Plankton diversity and water quality assessment of three freshwater lakes of Mandi (Himachal Pradesh, India) with special reference to planktonic indicators. *Environmental monitoring and assessment*, *185*(10), 8355-8373.

Thirupathacah, M. (2012). Diversity of Zooplankton in lower manair reservoir, Karimuagar AP India. *Journal of Biological Science*,1 (7): 27-32.

Thomas, R., Meybeck M. & Beim, A. (1996). Water Quality Assessments - A Guide to Use of Biota, Sediments and Water in Environmental Monitoring - Second Edition Edited by Deborah Chapman © 1992, 1996 UNESCO/WHO/UNEP

Thompson, K. (2006). *Characterizing and managing salinity loadings in reclaimed water systems*. American Water Works Association.

Tiseer, F. A., Tanimu, Y., & Chia, A. M. (2008). Seasonal occurrence of algae and physicochemical parameters of Samaru stream, Zaria, Nigeria. *Asian Journal of Earth Sciences*, *2*(3), 101-107.

Trauth, M. H., Maslin, M. A., Deino, A., & Strecker, M. R. (2005). Late cenozoic moisture history of East Africa. *Science*, *309*(5743), 2051-2053.

Trishala. K., Rawtani, D., & Agrawal, Y. K. (2016). Bioindicators: the natural indicator of environmental pollution. *Frontiers in life science*, *9*(2), 110-118.

Tuite, C. H. (1979). Population size, distribution and biomass density of the lesser flamingo in the Eastern Rift valley, 1974-76. *Journal of Applied ecology*. 16 (3): 765-775.

Tuite, C. H. (1981). Standing crop densities and distribution of Spirulina and benthic diatoms in East African alkaline saline lakes. *Freshwater Biology*, *11*(4), 345-360.

U.S. Geological Survey (USGS), 2018, Lakes and reservoirs—Guidelines for study design and sampling: U.S. Geological Survey Techniques and Methods, book 9, chap. A10, 48 p.

United State Environmental Protection Agency (USEPA) (2005). Report to Congress Nonpoint Source Pollution in the US Gov't Print Office Washington D.C 35Pp.

United States Environmental Protection Agency (USEPA) (1995). National Water Quality Inventory: 1994 report to Congress. U.S. Environmental Protection Agency Office of Water. *E.P.A. Publication 841*- R-95-005. Washington, D.C. pp 566.

United States Environmental Protection Agency (USEPA). (1992). *Secondary drinking water Regulations*: Guidance for Nuisance Chemicals. Pp 1-30. Retrieved from http://www.epa.gov/safewater /consumer/2ndstandards.html. [On 13/5/2019]

United States Environmental Protection Agency (USEPA). (2001b). Nutrient Criteria Technical Guidance Manual - Estuarine and Coastal Marine Waters. EPA- 822/B-01/003. Office of Water, Washington, DC. pp 1-362.

United States Environmental Protection Agency (USEPA). (2007). *The Water Sourcebooks*: Grade Level 9 - 12 Fact Sheets. Retrieved from http://www.epa.gov/safewater/kids/wsb/index.html#about.

United States Environmental Protection Agency (USEPA). (2013). *Aquatic Life ambient water quality criteria for Ammonia-Freshwater*: pp 8-22.

USEPA. (2016). Aquatic resource surveys. Retrieved from http://www.epa.gov/national-aquatic-resource-surveys/indicators-streambed-sediments [On 25/2/2019]

Usman, L. U. (2016). Some Limnological and Biological Aspects of Ajiwa Reservoir, Katsina State, Nigeria. (M.Sc. Dissertation). Department of Biological Sciences, Ahmadu Bello University, Zaria. Pp 112-118.

Vareschi, E. (1978). The ecology of Lake Nakuru (Kenya). *Oecologia*, *32*(1), 11-35.

Vareschi, E. (1982). The ecology of Lake Nakuru (Kenya): III. Abiotic factors and primary production. *Oecologia*, *55*(1), 81-101.

Vareschi, E., & Jacobs, J. (1985). The ecology of Lake Nakuru. *Oecologia*, *65*(3), 412-424.

Vareschi, E., Melack, M.J. & Kilham, P. (1981): Saline waters. – In: Symoens, J.J.; Burgis, M. & Gaudet, J. (eds): The Ecology and Utilization of African Inland Waters. – pp. 93–102, UNEP, Nairobi.

Verschuren, D., Cumming, B. F., & Laird, K. R. (2004). Quantitative reconstruction of past salinity variations in African lakes: assessment of chironomid-based inference models (Insecta: Diptera) in space and time. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(6), 986-998.

Walkuska, G., & Wilczek, A. (2010). Influence of Discharged Heated Water on Aquatic Ecosystem Fauna. *Polish Journal of Environmental Studies*, *19*(3).

Wang L, Wei W, Zhou P. (2013). Phytoplankton community structure and its relationship with environmental factors in the rivers of Tongling city, Anhui Province of East China in Winter. *Chinese Journal of Applied Ecology*, 24 (1): 243-250

Ward, S. (2015). *Are changes in the lesser flamingo population a natural consequence of soda lake dynamics?* (Doctorate Thesis). University of Southampton, UK.

Wetzel, R.G. and Liken, G.E. (2001). Limnological Analyses, Third Edition springe, New York p 429.

Widdel, F. (1988). Microbiology and ecology of sulfate-and sulfur-reducing bacteria. *Biology of anaerobic microorganisms.*, 469-585.

Williams, W. D. (1981). Inland salt lakes: an introduction. In *Salt Lakes* (pp. 1-14). Springer, Dordrecht.

WordWeb dictionaries online. (2018). Retrieved from http://www.wordwebonline.com [On 17/3/2018]

World Health Organization, WHO. (1996). *Total dissolved solids in drinking water*. Guidelines for drinking-water quality, 2nd edition., pp 2.

World Health Organization, WHO. (2011). *Guidelines for drinking-water quality [electronic resource]: incorporating 1st and 2nd addenda, Vol. 1, Recommendations.* , 3rd edition. Retrieved from www.who.int/water_sanitation_health/publications/2011/dwq_chapters/en/ [On 8/5/2019]

APPENDICES

Appendix 1: Research Permit

NATIONAL COMMISSION FOR SCIENCE, TECHNOLOGY AND INNOVATION

Telephone:+254-20-2213471,
2241349,3310571,2219420
Fax:+254-20-318245,318249
Email: dg@nacosti.go.ke
Website : www.nacosti.go.ke
When replying please quote

NACOSTI, Upper Kabete
Off Waiyaki Way
P.O. Box 30623-00100
NAIROBI-KENYA

Ref: No. **NACOSTI/P/19/84544/29997**

Date: **27th May, 2019**

Stephen Balaka Opiyo
Kisii University
P.O. Box 408-40200
KISII.

RE: RESEARCH AUTHORIZATION

Following your application for authority to carry out research on ***"Integrated ecological integrity assessment based on phytoplankton indices, water quality dynamics, habitat quality survey and trophic state indices in Lake Simbi, deep alkaline-saline lake in Kenya"*** I am pleased to inform you that you have been authorized to undertake research in **Homa Bay County** for the period ending **27th May, 2020.**

You are advised to report to **the County Commissioner and the County Director of Education, Homa Bay County** before embarking on the research project.

Kindly note that, as an applicant who has been licensed under the Science, Technology and Innovation Act, 2013 to conduct research in Kenya, you shall deposit **a copy** of the final research report to the Commission within **one year** of completion. The soft copy of the same should be submitted through the Online Research Information System.

BONIFACE WANYAMA
FOR: DIRECTOR-GENERAL/CEO

Copy to:

The County Commissioner
Homa Bay County.

The County Director of Education
Homa Bay County.

Appendix 2: Descriptive statistics for spatial scale

Station		SD	Temp	DO	pH	EC	Sal.	TA	TH	TDS	Turb	SRP	TP	TN	NO_3-N	NO_2-N	NH_3-N	SiO_2
ST1	Mean	0.63^{a}	28.91^{a}	4.73^{a}	10.26^{a}	17954.67^{a}	9.78^{a}	8873.67^{a}	140.17^{a}	5365.28^{a}	75.98^{b}	1815.89^{a}	2795.45^{a}	504.95^{a}	8.55^{a}	5.77^{a}	271.96^{a}	64.51^{b}
	S.E	0.01	0.51	0.30	0.21	373.05	0.17	412.90	19.39	2204.87	5.67	108.35	212.70	175.55	1.65	1.65	238.23	1.47
ST2	Mean	0.63^{a}	29.94^{a}	4.93^{a}	10.57^{a}	17755.33^{a}	9.80^{a}	7713.67^{a}	134.67^{a}	5385.53^{a}	131.32^{a}	1597.28^{a}	2729.14^{a}	542.32^{a}	10.82^{a}	7.28^{a}	29.34^{a}	68.86^{ab}
	S.E	0.02	0.59	0.36	0.30	574.44	0.14	439.81	21.98	2218.45	17.68	64.68	229.80	288.05	3.57	3.41	5.62	0.77
ST3	Mean	0.60^{ab}	29.34^{a}	5.53^{a}	10.02^{a}	17937.50^{a}	9.85^{a}	8202.00^{a}	133.83^{a}	5420.60^{a}	76.33^{b}	1776.17^{a}	2817.49^{a}	401.48^{a}	6.74^{a}	4.46^{a}	32.25^{a}	64.34^{b}
	S.E	0.01	0.60	0.55	0.32	518.19	0.16	129.60	22.64	2229.69	8.74	95.81	195.10	121.00	1.80	0.79	9.31	1.52
ST4	Mean	0.55^{b}	29.54^{a}	5.47^{a}	10.14^{a}	17658.83^{a}	9.91^{a}	8111.83^{a}	138.50^{a}	5450.17^{a}	64.45^{b}	1807.83^{a}	2822.47^{a}	618.11^{a}	12.42^{a}	7.44^{a}	69.90^{a}	55.13^{c}
	S.E	0.01	0.74	0.67	0.20	571.66	0.16	95.42	24.44	2237.99	4.40	70.62	192.89	221.11	4.75	3.45	44.60	1.52
ST5	Mean	0.56^{b}	29.36^{a}	5.32^{a}	10.17^{a}	18017.33^{a}	9.79^{a}	7828.67^{a}	141.33^{a}	5261.67^{a}	70.62^{b}	1773.67^{a}	3013.60^{a}	331.97^{a}	5.33^{a}	3.04^{a}	37.71^{a}	70.17^{ab}
	S.E	0.01	0.59	0.63	0.41	591.58	0.19	256.81	24.58	2191.90	4.73	103.01	146.91	134.91	1.22	0.78	8.06	2.82
ST6	Mean	0.60^{ab}	29.25^{a}	5.54^{a}	10.23^{a}	17924.83^{a}	9.76^{a}	7960.83^{a}	146.83^{a}	5349.17^{a}	72.65^{b}	1811.81^{a}	2800.05^{a}	339.16^{a}	6.93^{a}	4.13 a	37.90^{a}	72.57^{a}
	S.E	0.01	0.52	0.70	0.09	511.48	0.17	128.55	24.95	2207.12	3.23	78.84	215.63	132.11	1.13	0.84	6.50	1.84
Total	Mean	0.59	29.39	5.25	10.23	17874.75	9.81	8115.11	139.22	5372.07	81.89	1763.77	2829.70	456.33	8.46	5.35	79.84	65.93
	S.E	0.01	0.23	0.22	0.11	200.88	0.06	123.66	8.75	837.27	5.07	35.83	77.25	73.45	1.11	0.86	40.25	1.17

Note: Mean values in the same column that do not share a superscript letter are significantly different ($p < 0.05$).

Appendix 3: Descriptive statistics for temporal scale

Month		SD	Temp	DO	pH	EC	Sal.	T-Alk	T-Hard	TDS	Turb	SRP	TP	TN	NO_3-N	NO_2-N	NH_3-N	SiO_2
Dec 2018	Mean	0.63 ^{a}	26.91 ^{b}	4.20 ^{c}	10.32 ^{a}	16060.00 ^{c}	9.26 ^{d}	8279.00 ^{ab}	88.67 ^{b}	782.85 ^{c}	61.43 ^{a}	1599.50 ^{b}	2212.09 ^{c}	63.37 ^{c}	5.62 ^{abc}	2.70 ^{a}	28.25 ^{a}	64.37 ^{a}
	S.E	0.02	0.10	0.06	0.04	151.59	0.13	431.19	1.74	4.24	2.14	56.50	75.51	0.95	0.88	0.49	4.48	2.68
Jan 2019	Mean	0.62 ^{ab}	29.54 ^{a}	5.01 ^{bc}	10.33 ^{a}	16698.83 ^{c}	9.52 ^{cd}	9087.67 ^{a}	87.33 ^{b}	55.07 ^{d}	87.87 ^{a}	1844.78 ^{ab}	2656.14 ^{bc}	113.85 ^{c}	2.94 ^{c}	1.88 ^{a}	59.27 ^{a}	64.32 ^{a}
	S.E	0.01	0.45	0.31	0.01	347.08	0.06	307.74	1.43	5.36	21.78	43.65	166.40	14.07	0.24	0.28	7.10	3.07
Feb 2019	Mean	0.61 ^{ab}	29.50 ^{a}	4.26 ^{c}	10.17 ^{a}	18111.17 ^{b}	9.79 ^{bc}	7957.17 ^{b}	88.00 ^{b}	8090.00 ^{b}	76.70 ^{a}	1795.33 ^{ab}	2835.19 ^{b}	276.00 ^{bc}	4.12 ^{bc}	3.40 ^{a}	11.15 ^{a}	65.41 ^{a}
	S.E	0.02	0.43	0.14	0.19	163.07	0.11	119.70	4.30	164.49	11.80	15.27	114.60	101.19	0.60	0.56	2.03	3.46
Mar 2019	Mean	0.56 ^{b}	29.95 ^{a}	4.81 ^{bc}	10.59 ^{a}	18491.67 ^{ab}	10.07 ^{ab}	7833.00 ^{b}	187.50 ^{a}	786.25 ^{c}	80.94 ^{a}	1634.22 ^{b}	2718.00 ^{b}	1061.27 ^{a}	13.75 ^{a}	9.20 ^{a}	314.27 ^{a}	66.80 ^{a}
	S.E	0.01	0.23	0.13	0.12	92.03	0.04	162.32	2.59	4.19	11.95	40.98	77.45	214.77	3.08	3.11	233.38	1.82
Apr 2019	Mean	0.56 ^{b}	29.83 ^{a}	7.24 ^{a}	9.77 ^{a}	19026.67 ^{a}	10.21 ^{a}	7795.33 ^{b}	195.83 ^{a}	11340.67 ^{a}	87.52 ^{a}	1681.54 ^{b}	3480.43 ^{a}	715.39 ^{ab}	11.22 ^{abc}	6.49 ^{a}	29.14 ^{a}	66.87 ^{a}
	S.E	0.01	0.34	0.69	0.56	153.87	0.01	204.33	4.89	19.41	11.12	106.33	42.18	72.38	1.46	1.04	4.59	3.65
May 2019	Mean	0.57 ^{ab}	30.60 ^{a}	5.98 ^{ab}	10.22 ^{a}	18860.17 ^{ab}	10.04 ^{ab}	7738.50 ^{b}	188.00 ^{a}	11177.58 ^{a}	96.88 ^{a}	2027.28 ^{a}	3076.33 ^{ab}	508.11 ^{bc}	13.14 ^{ab}	8.45 ^{a}	36.98 ^{a}	67.80 ^{a}
	S.E	0.01	0.24	0.07	0.21	102.77	0.03	122.34	2.58	20.27	6.75	100.42	121.11	129.63	3.95	3.20	6.61	3.13
Total	Mean	0.59	29.39	5.25	10.23	17874.75	9.81	8115.11	139.22	5372.07	81.89	1763.77	2829.70	456.33	8.46	5.35	79.84	65.93
	S.E	0.01	0.23	0.22	0.11	200.88	0.06	123.66	8.75	837.27	5.07	35.83	77.25	73.45	1.11	0.86	40.25	1.17

Appendix 4: Checklist for Species in Lake Simbi

CYANOPHYCEAE		CHLOROPHYCEAE		BACILLARIOPHYCEAE	
Species	%	Species	%	Species	%
Microcystis flos-aqua	*33.33*	*Kirchneriella lunaris*	*0.53*	*Fragilaria pinnata*	*31.40*
Fragilaria aethiopica	*0.05*	*Oocystis parva*	*0.06*	*Cyclotella ocellata*	*10.63*
Planktolyngbya tallingii	*0.04*	*Pediastum boryanum*	*3.08*	*Aulacoseira nyansenssis*	*16.43*
Planktolyngbya circumcreta	*0.25*	*Botryococcus braunii*	*2.45*	*Nitzschia palea*	*1.45*
Spirulina princeps	*1.37*	*Monoraphidium braunii*	*0.28*	*Nitzschia acicuraris*	*18.36*
Chroococcus turgidus	*0.66*	*Chlorella vulgaris*	*3.41*	*Fragillaria virescens*	*9.66*
Oscillatoria sp	*0.24*	*Romeria elegans*	*1.11*	*Navicula sp*	*2.90*
Oscillatoria tenuis	*0.45*	*Pediastum duplex*	*0.11*	*Nitzschia sp*	*3.86*
Planktolyngbya limnetica	*0.31*	*Tetraedron trigonum*	*0.41*	*Surirella sp.*	*5.31*
Anabaena flos-aquae	*0.29*	*Kirchneriella obesa*	*0.13*	*Fragilaria construens*	*0.00*
Cylindrospermopsis africana	*0.45*	*Monoraphidium caribeum*	*0.26*	***TOTAL***	***100***
Spirulina major	*1.67*	*Ankistrodesmus falcatus*	*0.92*	DINOPHYCEAE	
Spirulina gigantea	*0.39*	*Tetredron arthromisforme*	*0.33*	*Ceratinium branchyceros*	*81.90*
Anabaena hylina	*23.83*	*Coelastrum microporum*	*0.07*	*Glenordinium bernardii*	*6.03*
Pseudo anabaena sp	*0.18*	*Monoraphidium sp*	*0.37*	*Ceratinium sp*	*12.07*
Pseudo-anabaena circularis	*0.03*	*Pediastum muticum*	*0.55*	***TOTAL***	***100***
Oscillatoria geminata	*0.04*	*Scenedesmus acuminatus*	*0.26*	EUGLENOPHYCEAE	
Aphanocapsa rivularis	*0.18*	*Scenedesmus obliquus*	*80.77*	*Euglena acus*	*63.83*
Spirulina laxissima	*0.18*	*Kirchnelliera lunaris*	*1.94*	*Euglena viridis*	*4.26*
Arthrospira fusiformis	*1.07*	*Ankistrodesmus*	*1.44*	*Phacus sp*	*31.91*

		gracilis			
Planktolyngbya contrata	*0.02*	*Oocystis nageli*	*0.07*	***TOTAL***	***100***
Oscillatoria splendida	*0.08*	*Crucigenia heteracantha*	*0.94*	ZYGNEMATOPHYCEAE	
Anabaenopsis circularis	*0.49*	*Characium sp*	*0.35*	*Cosmarium succisum*	*4.59*
Oscillatoria tanganyikae	*0.07*	*Kirchneriella Schmidle*	*0.09*	*Cosmarium muticum*	*12.84*
Microcystis aeruginosa	*7.22*	*Crucigenia sp*	*0.07*	*Closterium abruptum*	*33.03*
Chroococcus dispersus	*0.15*	***TOTAL***	***100***	*Hyalotheca mucosa*	*49.54*
Cylindrospermopsis sp	*0.04*			***TOTAL***	***100***
Microcystis robusta	*24.88*				
Pseudo-anabaena tanganyikae	0.23				
Romeria ankensis	0.16				
Merismopedia tenuissima	0.08				
Microcystis wasenbergii	0.13				
Anabaenopsis tanganyikae	0.10				
Chroococcus limneticus	0.19				
Coelomoron sp	0.92				
Coelomoron vestitus	0.24				
TOTAL	**100**				