UNIVERSITY OF CAPE COAST

DEVELOPMENT OF AN INTEGRATED WATER QUALITY INDEX FOR MONITORING WATER QUALITY OF SELECTED ESTUARIES IN GHANA

DOROTHY KHASISI LUKHABI

2024

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BY

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Thesis submitted to the Department of Fisheries and Aquatic Sciences, School of Biological Sciences, College of Agriculture and Natural Sciences, University of Cape Coast, in partial fulfilment of the requirements for the award of Doctor of Philosophy degree in Oceanography and Limnology

MAY 2024

DECLARATION

Candidate's Declaration

I hereby declare that this thesis is the result of my original research and that no part of it has been presented for another degree in this university or elsewhere.

Candidate's Signature …………………………… Date …………

Name: Dorothy Khasisi Lukhabi

Supervisors' Declaration

We hereby declare that the preparation and presentation of the thesis were supervised in accordance with the guidelines on supervision of thesis laid down by the University of Cape Coast.

Principal Supervisor's Signature ……………………… Date …………

Name: ……………………………………………………………………

Co-Supervisor's Signature …………………………… Date …………

Name: ………………………………………………………………………

ABSTRACT

There is currently no country-specific water quality index (WQI) for monitoring estuarine water quality in Ghana. This study addresses this gap by developing an integrated WQI for monitoring the quality of estuarine water in Ghana, i.e., $IWQI_{Gh}$. This was achieved by reviewing literature on limitations and potential of adapted WQIs for water quality monitoring in Africa; field assessment of water quality of Ankobra, Volta, Whin and Kakum Estuaries using physicochemical parameters and benthic fauna; and developing the $IWQI_{Gh}$ using multivariate statistical approaches. From results, the Weighted Arithmetic and Canadian Council of Ministers of the Environment WQIs were the most adapted with major limitations occurring in parameter selection and final index classification schemes. Statistical approaches in parameter selection and logical linguistic descriptions in classification schemes were suggested to ensure objectivity in WQI development process. Moreover, Kakum and Whin Estuaries were moderately polluted. Despite the moderate pollution levels, Kakum was the most diverse of the four estuaries. Based on ecological stability, Kakum Estuary was ecologically healthier than Whin, Volta, and Ankobra Estuaries, in that order. The most representative parameters for $IWQI_{Gh}$ that contributed to high index values included nutrients, turbidity, electrical conductivity, chemical oxygen demand as well as pollution-tolerant low scoring taxa. Based on the $IWQI_{Gh}$, the selected estuaries were categorised as "polluted-Class 4". Incorporation of benthic fauna in the WQI development process in further studies was recommended to address region-specific concerns in Africa rather than adapting existing WQIs.

KEYWORDS

Estuaries and estuarine ecosystem health

Benthic macroinvertebrates

Integrated Water Quality Index

Water quality and pollution

Water Quality Parameters

Physicochemical parameters

ACKNOWLEDGEMENTS

I am deeply indebted to my supervisors Dr. Paul Kojo Mensah and Dr. Noble Kwame Asare of the Department of Fisheries and Aquatic Sciences (DFAS-UCC), for their tremendous contribution in supervising this work. I am grateful for the World Bank Group's support in providing me with a scholarship to pursue this program through the Government of Ghana and the Africa Centre of Excellence in Coastal Resilience (ACECoR) Project under the Centre for Coastal Management (CCM). Additionally, I am incredibly grateful to Prof. Denis W. Aheto (Director), Dr. Precious Mattah (Deputy Director), and the entire ACECoR/CCM staff for the wonderful opportunity to be part of ACECoR program. In a special way, I would like to acknowledge the technical assistance provided by the laboratory staff of the Coastal Research Laboratory (DFAS) for their assistance both in fieldwork and in the laboratory. Furthermore, I would like to acknowledge the contributions of Dr. Margaret Fafa Akwetey and Dr. Kwadwo Mireku (DFAS), Dr. Charles Faseyi (CCM), Dr. Ouma O. Kennedy (The Copperbelt University-Zambia), Mr. Thomas Davis, Mr. Prosper Dordunu, Mr. Albert Koomson (Coastal Research Laboratory, DFAS), Mr. Jonathan Ntow (Lab Tech. Dept-UCC), Mr. Samuel Boateng and Ms. Getrude Oboh (Dept. of Water & Sanitation-UCC), Mr. Stephen Kwaw (My fieldwork driver). To all ACECoR project students, especially the PhD 2nd cohort colleagues, Elizabeth Agyekumwa, Janet Frimpomaa, Lavenda Achieng', Namonda Mubiana, Tchaka Kamanga, Jeremiah Kapatu, among many others. Thank you so much for your endless support in the course of this programme. To my parents, siblings and the entire family, thank you for your prayers, endless support and encouragement to ensure the success of this programme. God richly bless you.

DEDICATION

To the entire Lukhabi's family, and most importantly, to my son, Ted

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LIST OF ACRONYMS

AHP	Analytic Hierarchy Process
ANRH	Agence Nationale des Resources Hydrauliques
ANZECC	Australian and New Zealand Environment and Conservation Council
APHA	American Public Health Association
BCC	Banker, Charnes, and Cooper
BMWP	Biological Monitoring Working Party
BOD	Biochemical Oxygen Demand
CA	Cluster Analysis
CCME	Canadian Council Ministers of Environment
COD	Chemical Oxygen Demand
DA	Discriminant Analysis
DEA	Data Envelopment Analysis
DMU	Decision-Making Units
DWARF	Department of Water Affairs and Forestry
EDTA	Ethylenediamine tetra-acetic acid
FA	Factor Analysis
FAO	Food and Agriculture Organisation
FFG	Functional Feeding Groups
HCA	Hierarchical Cluster Analysis
LGMWEP	Lower Great Miami Watershed Enhancement Program
MRA	Multivariate Regression Analysis
MWQI	Microbiological Water Quality Index
NSF	National Sanitation Foundation

NTU	Nephelometric Turbidity Unit
PCA	Principal Component Analysis
RWQI	Ramakrishaniah Water Quality Index
SANS	South African National Standard
SASS	South African Scoring System
SRPB	Solway River Purification Board
TDS	Total Dissolved Solids
TSS	Total Suspended Solids
TWQR	Target Water Quality Ranges
USEPA	United States Environmental Protection Agency
UWQI	Universal Water Quality Index
WA	Weighted Arithmetic
WHO	World Health Organisation
WQI	Water Quality Index
WQPs	Water Quality Parameters

CHAPTER ONE

INTRODUCTION

1.1 Overview

Estuarine ecosystems are characterised by constant mixing of freshwater from rivers with saline water from the marine environment. The continuous mixing of water from these two ecosystems creates new water quality dynamics and modifications. Aquatic organisms in estuaries have to adjust to these changes in relation to their physiological requirements (Ujjania & Dubey, 2015). Estuaries are faced with perturbations ranging from point and non-point sources of pollution to hydro-morphological stressors (Poikane *et al*., 2020). Both point and non-point sources of pollution impair the quality of estuarine water, which in turn poses a threat to aquatic organisms as well as causing detrimental effects to potential downstream users. The most conventional way of determining the degree and extent of pollution in estuaries is through assessing the Water Quality Parameters (WQPs) and this is conveniently performed using a Water Quality Index (WQI) (Hassan *et al*., 2017).

Currently in Ghana, there is no country-specific WQI for monitoring the water quality of estuarine ecosystems found in literature. The WQI presently used is a general-type adapted index initially designed for UK rivers (House, 1989), and later found applicability in South African estuaries (Cooper, 1994). Adapted indices often have inherent abnormalities like eclipsing, ambiguity and rigidity, which give false information about water quality status (Swamee & Tyagi, 2000). For effective water quality monitoring, it is important to develop a customised WQI for monitoring estuarine ecosystems in Ghana since such tools are meant to be source and region specific. Using four selected estuaries

(Ankobra, Whin, Kakum and Volta), the study aimed at reviewing adapted WQIs and their associated challenges and potential for water quality monitoring in Africa, assessing the quality of water using both physicochemical and biological parameters and finally, developing a customised integrated WQI for monitoring quality of water in estuaries along the coast of Ghana.

1.2 Background to the Study

Water Quality Indices (WQIs) have been proposed in water quality evaluation because they are able to summarise a large number of Water Quality Parameters (WQPs) into a single meaningful numerical data that expresses the pollution status of water (Abbasi & Abbasi, 2011; Gorde & Jadhav, 2013; Oni & Fasakin, 2016; Ramesh *et al.*, 2010; Rawat *et al.*, 2019; Zeinalzadeh & Rezaei, 2017). Water Quality Indices reflect the all-round effect caused by various WQPs and provides a comparison of water quality status for different locations in time and space, hence a water quality status prediction tool (Tiwari *et al.*, 2014). Water Quality Indices can be formulated in two ways: (1) Indices based on an increasing scale, whereby index numbers increase with the degree of pollution (water pollution indices) and, (2) indices based on a decreasing scale, in which the index numbers decrease with the degree of pollution (water quality indices) (Rawat *et al.*, 2017). Moreover, classification of WQIs can take the form of descriptive ranks, assuming terms like "poor, marginal, fair, good and excellent" (Banda & Kumarasamy, 2020a).

Water Quality Indices provide an easier and simpler way of understanding for stakeholders in the decision-making process, ranging from policy makers, water authorities, water scientists and the general public (Terrado *et al.*, 2010), about the probability of the overall quality of water to pose a potential threat to various uses, e.g., recreation, irrigation, habitat for aquatic life, etc, (Banda & Kumarasamy, 2020). Water Quality Indices are used to assess the suitability of water sources for various uses, including marine environments (Filatov *et al.*, 2005), river systems (Abdul-Razak *et al.*, 2010; Ewaid *et al.*, 2020; Nwanosike *et al.*, 2010; Shah & Joshi, 2017; Yisa & Oladejo, 2010), drinking water (Chauhan & Singh, 2010; Mukate *et al.*, 2019), groundwater systems (Adimalla & Qian, 2019; Rawat *et al.*, 2019; Saeedi *et al.*, 2010) and reservoirs (Boah & Pelig-ba, 2015). In light of the above, WQIs are considered key in the management of water resources with benefits and uses not limited to (1) comparing water quality from different sources and deciding the appropriate use of the water resource concerned, (2) making more objective and less subjective policy choices, and (3) giving an integral image of the overall quality of the source to make it easier for non-technical stakeholders to understand (Tripathi & Singal, 2019b).

Although WQIs are globally accepted as a baseline for water quality monitoring (Banda & Kumarasamy, 2020b), there is still no definitive and commonly acceptable methodology for developing WQIs (Sutadian *et al.*, 2017). Previous studies have been based on three major aggregation functions namely (i) additive or arithmetic, (ii) multiplicative or geometric, and (iii) logical (Abassi & Abassi, 2012; Uddin *et al.*, 2021). Additive or arithmetic aggregation methods involve combining the transformed parameters through summation. This method has been employed in previous WQIs (Brown *et al.*, 1970; Horton, 1965; Ott, 1978; Prati *et al.*, 1971). The most frequently used additive aggregation function is weighted arithmetic mean due to the simplicity it offers. However, the additive aggregation function lacks sensitivity (Liou *et al.*, 2004; Sutadian *et al.*, 2017).

Multiplicative or geometric aggregation methods combine the transformed parameters through product operation. A few WQIs which assess general water quality have used this function including indices developed by Walski & Parker, (1974), Dinius (1987) and Liou *et al.* (2004). In this multiplicative aggregation, the weighted geometric mean is the most commonly used method. It has been reported to be more viable and unbiased in comparison to the weighted arithmetic mean (Landwehr *et al.*, 1974). In weighted geometric mean, the final index is zero if any one sub-index is zero and this comes as a solution to the eclipsing and ambiguity problems (Liou *et al.*, 2004).

The logical aggregation method involves combining the sub-indices using logical operators. The most common logical operators are the minimum and maximum operators, which have been used in the index by Smith (Smith, 1990). The minimum operator function evades the issues of eclipsing and ambiguity in the final index by using the lowest sub-index values to produce the index value as used by Smith (1990) in New Zealand and Shah & Joshi (2015) in India. On the other hand, the maximum operator aggregation function performs the summation of sub-indices in an increasing scale manner and it has been reported to be suited to applications where an index must report if any of the recommended limits are violated (Abassi & Abassi, 2012).

Although the use of logarithmic functions is the most current practice, most of the researchers use arithmetic (additive) or geometric (multiplicative) aggregations (Ramesh *et al.*, 2010; Sutadian *et al.*, 2017). In the current study, an integrated WQI was developed by combining the summed arithmetic weighted function with other factors as detected by the system status and nature of data collected. Since the inception of the first WQI (Horton, 1965), many other WQIs have been developed. However, the most commonly used and applied WQIs include the Canadian Council of Ministry of Environment (CCME-WQI), National Sanitation Foundation (NSFWQI) and Weighted Arithmetic (WAWQI) (Tyagi *et al.*, 2013).

Through continuous adaption, these three WQIs have found wide applicability in Africa. For example, the CCME-WQI has been adapted and employed in Egypt (Goher *et al.*, 2019) and Ghana (Faseyi *et al.*, 2022; Miyittah et al., 2020). The NSFWQI was used in Nigeria (Kalagbor *et al.*, 2019), while the WAWQI has found applicability in Kenya (Chebet *et al.*, 2020; Njuguna *et al.*, 2020; Robert *et al.*, 2021). Furthermore, in Nigeria, the WAWQI has been widely used (Akoteyon *et al.*, 2011; Nwanosike *et al.*, 2010; Oni & Fasakin, 2016; Yisa & Oladejo, 2010). From these observations, it seems the WQIs applied in most African countries have been adapted from the developed world and are mainly used for assessing surface and ground water quality.

Although it is generally acceptable to adapt indices and modify them in accordance to various legal requirements for water agencies in different countries, WQIs are designed for a particular region and are source-specific (Lukhabi *et al.*, 2023). Before adapting an index for use, it is necessary to understand the bases behind its development and link them to location-specific concerns. This relates to the original conditions supporting the index development as represented by WQPs and the usage (Banda, 2015). If this is not put under consideration, the final index value inherits abnormalities (eclipsing, ambiguity and rigidity). Ambiguous indices suggest worse water quality than expected as a result of differences in sub-index values obtained for all the WQPs, hence parameter impairment (Swamee & Tyagi, 2000). On the other hand, rigid indices are not flexible enough to accommodate WQPs and this occurs if an index is applied in an area with objectives different from those that was developed for (Swamee & Tyagi, 2007).

In Ghana, the use of WQIs is a relatively new concept, which came into existence in 2003 when the Water Resources Commission (WRC) produced a document, "*Ghana Water Quality Guidelines and Criteria-The Adapted Water Quality Index*" and proposed its application for surface water quality. This was pioneered by the works of Ansa-Asare (1998) who adapted and modified the Solway WQI from the Solway River Purification Board (SRPWQI) (Bolton, 1978). The SRPBWQI is a general type of index in which various physical, chemical and microbiological parameters were aggregated to produce an overall index of water quality for rivers in the UK (House, 1989), which later found applicability in South African estuaries (Cooper *et al.*, 1994). The adapted WQI is currently referred to as the Adapted Water Quality Index (AWQI) for surface waters (Ansa-Asare, 1998). Due to the discussed issues surrounding adapted indices, a knowledge gap is evident, providing a scope to develop an index that is customised to Ghana's estuarine ecosystems. Therefore, the overall objective of this study is to develop an integrated WQI as an assessment tool that provides a standardised way of monitoring estuarine water quality for better management of estuarine water resources in Ghana.

1.3 Problem Statement

Understanding the water quality of any system is very crucial as it gives an impression of the monitoring tools required by water managers to easily manage the system. Ghana, just like any other Sub-Saharan African country, does not have a definitive WQI to monitor estuarine water quality and the adapted WQI available is used as a general index for assessing various surface water bodies (Darko *et al.*, 2013). Due to spatial-temporal variations, adapted indices are faced with inherent abnormalities and this translates into inaccuracies and wrong conclusions as far as the quality of water is concerned. Although the adapted WQIs have been used in Ghana to assess and monitor both surface and ground water quality, none has been modified to incorporate benthic macroinvertebrates in the usage, yet these organisms are the most preferred indicators of estuarine ecosystem health (Li *et al.*, 2010).

Moreover, estuarine ecosystems have been immensely studied in Ghana with the aim of assessing environmental quality using various metrics. Among them include studies on water, sediments and benthic macroinvertebrates in Pra Estuary (Faseyi *et al*, 2022a; Faseyi *et al.*, 2022b; Klubi *et al.*, 2018; Nortey *et al.*, 2016); water, sediments and fish in Nyan (Dzakpasu *et al.*, 2015, Nortey *et al.*, 2016) and Whin Estuaries (Faseyi *et al.*, 2022b; Chuku *et al.*, 2023; Agblemanyo, 2021; Sowah, 2019). However, the scope of the studies has been limited as far as water quality indices is concerned.

Due to the dynamic nature of estuaries, potential pollution threats are likely to cause detrimental ecological imbalance to the ecosystem goods provided, with far reaching consequences on the productivity in both the nearshore marine waters and other coastal ecosystems directly in contact with them. Of more than 90 lagoons and about 10 estuaries along the Ghanaian coast (Yankson & Obodai, 1999), water quality studies have largely focused on lagoons due to their abundance (Aggrey-Fynn *et al.*, 2011; Lamptey & Armah,

2008) and associated marshes (Okyere *et al*., 2011). This has further doubled the vulnerability of estuaries to various threats as a result of the little attention received. The present study therefore addresses this gap by assessing the current status of estuaries in Ghana and using the data obtained to develop an integrated WQI that will be used for their water quality monitoring.

1.4 Purpose of the Study

In Ghana, fisheries sustain the livelihoods of over 2.6 million people (Seidu *et al.*, 2022) and 70 % of the total fish stocks are marine in nature (Asiedu *et al*., 2018). Pollution of coastal ecosystems pose a threat to fish stocks and is likely to affect the livelihoods of the majority of Ghanaians depending on them. The results of this study are therefore beneficial to water quality managers by including estuaries in management planning of water resources in Ghana. Through this study, various physicochemical and biological parameters are combined to understand the current status of estuaries, consequently using the data obtained to develop an integrated WQI that will be used for their water quality monitoring.

1.5 Research Aim and Objectives

The overall aim of this study was to develop an integrated WQI as an assessment tool customised for monitoring water quality for better management of estuarine ecosystems in Ghana.

The specific objectives of this study were to;

1. Review the adapted WQIs in the African context and examine their limitations and potential for water quality monitoring using existing literature.

2. Assess the quality of water in selected estuaries (Ankobra, Whin, Kakum and Volta Estuaries) using physicochemical parameters and benthic macroinvertebrate community structure.
3. Develop a customised integrated WQI for monitoring estuarine ecosystems in Ghana using multivariate statistical approaches.

1.6 Research Hypothesis

The developed WQI for estuarine ecosystems in Ghana is more effective in water quality monitoring than adapted indices.

1.7 Significance of the Study

The current study provides valuable information about the status of water quality in selected Ghanaian estuaries (Ankobra, Whin, Kakum and Volta Estuaries) from the West coast, Central coast and East coast of the country. It does this by assessing various physicochemical parameters and benthic macroinvertebrate community structure as well as their interaction in the estuarine ecosystem.

Moreover, the study provides a customised tool for monitoring estuarine water quality along the coast of Ghana. Monitoring of estuarine water quality through an integrated WQI would support existing programmes aimed at sustaining, managing and protecting marine and coastal ecosystems, which is part of taking restoration action in the quest to achieve Sustainable Development Goal (SDG) 14, "Life Below Water, target 2."

It would also contribute efforts towards achieving the African Union Agenda 2063 Goal 6 on sustainable use of marine resources. Furthermore, this study would contribute to the Africa Centre of Excellence in Coastal Resilience (ACECoR) theme on *Ecosystems and Biodiversity*, with one of the theme objectives' being improvement in the management and protection of ecosystems. Furthermore, an integrated Water Quality Index customised for Ghanaian estuaries would be an essential tool meant to reduce flaws such as inaccuracies, cost and time in determining the water quality in future routine monitoring programmes.

1.8 Delimitations

1. The study covered upper, middle and lower reaches of the four selected estuaries: Ankobra, Whin, Kakum and Volta, assessing them within the extent of the mangrove ecosystems with the assumption that the areas covered denoted the extend of the estuaries.
2. Physicochemical parameters collected include surface water temperature, dissolved oxygen, pH, electrical conductivity, salinity, turbidity, total dissolved solids, total suspended solids, nitrate-nitrogen, ammonium-nitrogen, orthophosphates, biochemical oxygen demand, chemical oxygen demand, and benthic macroinvertebrates. Other parameters like heavy metals were not collected due to financial constraints.

1.9 Limitations

1. Sampling was carried out every other month over a hydrological year. However, monthly sampling could have been possible if time and financial resources had permitted.

1.10 Definition of Terms

Adapted WQI - Modified versions of the original WQIs that are used to evaluate the quality of surface and groundwater

Bioindicator - An organism used to assess the quality of environment

Euryhaline species - Species that are able to tolerate a wide range of salinity

Ubiquitous taxa - taxonomic groups of organisms that are commonly found in different environment.

1.11 Organisation of the Study

This thesis is divided into six (6) chapters. The first chapter consists of an introduction, background of the work, statement of the problem, purpose of the study, research aims and objectives, hypotheses, significance of the study, delimitations, limitations, and definition of terms. Chapter two (2) contains the literature reviews on estuaries, their zonation and characteristics, physico-chemical parameters in estuaries, primary productivity in estuaries, the concept of estuaries in Ghana including an overview, benefits and threats, various methods of ecological assessment of estuarine ecosystem health, development of WQIs including evolutionary history, WQI in the context of developing countries and procedure for their development. Chapter three (3) is focussed on drafted article addressing the first objective of the thesis entitled "Adapted Water Quality Indices: Limitations and Potential for Water Quality Monitoring in Africa" which has been published in MDPI- *Water* (*Water* 2023, *15*(9), 1736; *https://doi.org/10.3390/w15091736*). Chapter four (4) contains a drafted article featuring the second objective of the study titled "Benthic macroinvertebrates as indicators of water quality" published in Elsevier's *Heliyon (Heliyon 2024, 10*(7); *https://doi.org/10.1016/j.heliyon. 2024.e28018*). Chapter five (5) contains a drafted article concerning objective three entitled "An Integrated Water Quality Index for Monitoring Estuarine Ecosystem Health in Ghana," that is under review in Taylor and Francis' *African Journal of Aquatic Sciences* (Submitted on 21st October, 2023, submission ID

239125383. Chapter six (6) contains the summary, conclusions, and recommendations. The list of all references cited in the thesis is presented just after chapter six. The appendices come last, containing the supplementary figures and tables from the three articles presented in this thesis.

CHAPTER TWO

LITERATURE REVIEW

2.1 Overview

This chapter reviews relevant literature on the concept of estuaries, characteristics of estuarine water quality in terms of physical, chemical and biological parameters and primary productivity in estuarine ecosystems. The chapter provides an overview of estuaries in Ghana, including their benefits and threats and their various ecological assessment methods. Finally, the chapter details the Water Quality Indices concept including evolutionary history, WQIs in the context of developing countries and procedure for their development.

2.2 Estuarine Ecosystems

Estuaries are semi-enclosed bodies of water freely connected to the open sea, with mixed characteristics of both saline and freshwater (Liou *et al.*, 2004; Ujjania & Dubey, 2015). They are impacted on daily basis by the influence of tides. In most parts of the world, two high tides and two low tides occur every day. The tidal pattern of an estuary is determined by a number of factors, including its geographic position, the structure of the ocean floor and coastline, the water's depth, prevailing winds, and any barriers to water movement. Many estuaries are shielded from the full force of ocean waves, winds, and storms by surrounding reefs, barrier islands, fingers of land, mud, or sand, even if they are heavily impacted by tides and tidal cycles. Every estuary has unique properties that are influenced by the surrounding climate, freshwater input, tidal patterns, and currents (National Oceanic and Atmospheric Administration, 2024).

Estuaries are classified according to mode of formation, nature of sediment, tidal range and salinity. According to Sneli (2012), estuaries are most commonly classified in terms of salinity, hence there are three types: positive, negative and neutral estuaries (Sneli, 2012). Positive estuaries are that whereby lighter freshwater enters the estuary as a surface current and heavier sea water exits as a bottom current. Stratification in the estuary causes salinity to increase from the river head to the sea. Evaporation rates in negative estuaries are higher than freshwater inflows. The resultant hypersaline estuary water sinks and flows into the ocean as a bottom current. Salinity increases from the sea towards river head and the net flow is inward. Lastly, freshwater inflow and evaporation rates are balanced in neutral estuaries. There are no tidal variations or currents, and the salinity profile is constant from top to bottom. Due to the salinity conditions, the physicochemical conditions become unstable and organisms have to develop survival strategies to adjust to the frequent environmental changes (Yankson & Kendall, 2001).

Estuaries have five different salinity zones including the headwaters, upper reaches, middle reaches, lower reaches, and the mouth, according to Montagna *et al.*, (2012). Freshwater enters the estuary at the head, where there is only a little amount of salt penetration and a maximum salinity of 5 ‰. In the upper reaches, the salinity ranges between 5 - 18 ‰ with minimal currents, especially at high tides. The salinity for the middle reaches range between 18 - 25 ‰, while that of lower reaches is between 25-30 ‰. The lower reaches experiences faster currents as it paves way into the mouth. The estuarine mouth has very strong currents with a salinity that is comparable to the nearby seawater. The occurrence of organisms in various estuarine zones is a factor of salinity. Oligohalines can tolerate up to only 5 ‰. Although stenohalines are typical sea dwellers, they can tolerate up to 25 ‰. The majority of living things found in estuaries and the middle of the ocean are considered to be euryhalines. They have a high tolerance range for salinity variations (up to 30 ‰) and stay there to avoid sea competition.

Estuarine ecosystems are among the most productive ecosystems on earth, endowed with both migrant and resident fauna. Some migrant organisms use the estuary seasonally for breeding, feeding, or other life cycle activities. For example, Shellfish species like *pipi spp* and *cockles spp* reside in estuaries at different habitats during various stages of their life cycle. Additionally, salmonids e.g salmon and trout often migrate from the sea to estuaries to spawn. Other fish species are permanent dwellers in estuaries like the tule perch while others like the *snapper spp* and *blue cod spp* only use the estuary as breeding grounds or as nurseries for their juveniles. Estuaries also attract large numbers of seabirds, eg., herons which stopover for feeding in estuaries and all these organisms contribute immensely to the estuarine food-web. Moreover, some economically important estuarine habitats include tidal flats, salt marshes, seagrass beds, oyster reefs, and mangroves (Montagna *et al.*, 2012).

Several authors have proposed various estuarine zonation as shown in Table 1;

Table 1: *Estuarine Zonation According to Salinity*

Salinity classification					Reference
Freshwater <0.2 ‰	Oligohaline 0.2-1.8 ‰	Mesohaline 1.8-18.1 ‰	Polyhaline >18.1 ‰	-	(Redeke, 1922)
Freshwater <0.2 ‰	Oligohaline 0.2-1.8 ‰	Mesohaline 1.8-18.1 ‰	Polyhaline 18.1-30.7 ‰	Marine >30.7 ‰	(Redeke, 1933)
Freshwater <0.2 ‰ (0.5 ‰)	Oligohaline 0.2(0.5)-2(3) ‰	Meio-β-mesohaline 2(3) - (8-10) ‰	Pleio-or (α) mesohaline 8-16.5(10-20) ‰	Polyhaline > 16.5 ‰	(Välikangas, 1933)
Freshwater < 3 ‰	Oligohaline (0.1-0.5)-5 ‰	Mesohaline (5-8) - (15-20) ‰	Polyhaline (15-20) - (25-30) ‰	Marine 30-40‰	(Dahl, 1956)
Limnetic <0.5 ‰	Oligohaline 0.5-5 ‰	Mesohaline 5-18 ‰	Polyhaline 18-30 ‰	Euhaline 30-40 ‰	(Venice System, 1959)

2.2.1 Characteristics of Estuarine Ecosystems

Research by Sims *et al.* (2022) indicate how dynamic estuarine ecosystems are due to frequent exposure to both low and high tides within a 24-hour period, as well as the constant mixing of saline and freshwater. It is important to note that the salinity content of estuaries differ with seasons and during the wet season, there is input of more freshwater, which reduces salinity significantly and vice versa. This ultimately leads to variation in the physicochemical characteristics including salinity, dissolved oxygen (DO), surface water temperature (SWT), electrical conductivity (EC), turbidity, pH among others.

As previously indicated, freshwater imports, tidal variations and location all affect the salinity of an estuary (Devkota & Fang, 2015). The average salinity of seawater is typically 35 ‰, while that of freshwater is 0.5 ‰. Since estuaries are areas where freshwater and seawater mix, their salinity ranges from 0.5 ‰ to 35 ‰. Estuaries receive more freshwater during rainy seasons, which causes dilution and a reduction in salinity. However, during dry seasons, there is less freshwater entering the system, which causes saltwater to move upstream and increase the salinity (Castro, & Huber, 2005). The continuous fluctuation of salinity in estuaries affect the distribution, composition and abundance of estuarine biota, especially in combination with other physicochemical parameters like surface water temperature and dissolved oxygen. The more saline the estuarine water is, the less the soluble oxygen under the same conditions of SWT (Devkota & Fang, 2015).

Aquatic organisms need DO for respiration, and the two main sources of this DO in estuaries are atmospheric oxygen diffusion and photosynthesis by phytoplankton and macrophytes. Large DO quantity is brought in by freshwater inflow and is immediately consumed. The species and abundance of organisms in an estuary are a function of the presence or absence of DO (Dokulil & Qian, 2021).

Estuaries are shallow ecosystems hence their surface water temperature changes rapidly during day and night. Similar to salinity, SWT has an impact on the distribution, abundance and composition of organisms when combined with other physicochemical factors. In particular, the solubility of DO is affected by SWT, which affects the species likely to survive. Since different water inputs occur at various temperatures, the SWT of estuaries fluctuates based on the tidal levels (Bashevkin & Mahardja, 2022).

Turbidity is the suspended particulate matter (SPM) load in water which reduces transparency and restricts the quantity of light that can effectively penetrate the water column for primary production. Estuarine water is more turbid due to sediments carried in by entering water from rivers and from re-suspension of sea tidal currents as a result of the interaction between the sea and the river (Cho, 2007). Other sources of turbidity in estuaries include aquatic weeds and other organic compounds resulting from dead and decayed plant matter which gives water bodies a rust-red colouration (EPA, 1999).

The turbidity maximum, which is a feature of estuaries with considerable tidal motion, indicates a zone where suspended sediment concentrations are notably high, which alters biological conditions and reduces light penetration. The turbidity maximum might have several grams of suspended particles per liter, which significantly impacts the water quality and ecological dynamics of the estuary. Furthermore, prevailing winds significantly affect the turbidity levels of estuaries by altering surface conditions, water velocity, and availability of light. The complex interplay between physical factors and turbidity dynamics in estuaries is highlighted by the fact that wind direction, duration, and generated waves can have a significant impact on the turbidity and light conditions of these environments (Dronkers *et al.*, 2024).

High turbidity in estuaries has negative implications on primary productivity as a result of increased heat absorption capacity of water that causes higher temperatures that subsequently lower oxygen concentration. On the estuarine biota, turbidity impairs normal functioning by decreasing disease resistance as well as clogging fish gills (Faseyi *et al.*, 2022). High turbidity also induces cloudiness in the water and decreases visibility, which hinders breeding, feeding, reproduction, and ultimately the survival of aquatic life. Turbidity levels over 500 NTU are considered harmful to aquatic biota in estuaries (Okyere, 2019).

The majority of aquatic organisms can survive in a pH range of 5.0 to 9.0, and extreme acidity or alkalinity is likely to interfere with their physiological processes. The pH of estuarine water is readily buffered by dissolved carbonate ions in seawater through the reaction with the ions that alter pH. However, elevated levels of pH due to biological activities in estuaries may become problematic to biota (Bednaršek *et al*., 2022).

2.2.2 Primary Productivity of Estuarine Ecosystems

Estuaries depend heavily on nutrients because they regulate primary production. The primary source of nutrients in an estuary comes from the catchment areas through a variety of channels, including a) Nutrient transportation overland into estuaries following a significant downpour; b) transportation and deposition of fine particles from highly urbanised areas in estuaries through aeolian action; and c) nutrients from groundwater from regions with extensive estuarine floodplains (Santos & Eyre, 2011). Factors including plant cover, soil type, slope of landmass, as well as rainfall intensity and volume all impact the quantity and type of nutrients that are transported from catchments to estuaries. Estuarine ecology is greatly influenced by the source and composition of nutrients in a variety of ways, including enormous inputs of total nutrients, such as organics and particulates, that come from forests and extensive grazing catchments. Primary producers first break down these nutrients to make them readily available for uptake (Harris, 2001). Furthermore, catchments with heavy agriculture and urbanisation provide high organic matter content which primary producers like micro and macro-algae easily break down into a more utilisable inorganic form (Smith *et al.*, 2003). The overall nitrogen concentration of catchments that produce coloured dissolved organic matter is very high, and the nitrogen is present as tannins, which have a limited availability to plants (Maie *et al*., 2006). Given the aforementioned elements, it can be deduced that high human population density and high rainfall are directly related to nutrient levels in estuaries (Glibert & Burkholder, 2006). Organic matter in estuaries control the nature of metabolism by microbes, which further controls the cycling of important elements like nitrates, phosphates, sulphur and iron. The time scale by which organic matter is preserved in estuaries is determined by the balance between its production and consumption within the estuarine functional zone. Input of organic matter in estuaries could either be allochthonous or autochthonous. While allochthonous organic matter is brought in from outside its original environment and deposited in the estuarine ecosystem, autochthonous organic matter is produced within the estuarine ecosystem (Pires-Teixeira *et al*., 2023).

Particulate organic matter in estuaries comes from both the total suspended solids that flow in with freshwater intake and direct leaf litter dropping from bordering plants. When terrestrial organic matter decomposes, dissolver organic matter builds up and enters estuaries through runoff and groundwater inputs (Maher *et al*., 2013). The amount and quality of terrestrial particulates and dissolved organic matter imported into an estuary is influenced by vegetation cover, land use, climate, and catchment hydrology (Harris, 2001). Wetlands, surrounding mangroves and salt marshes are additional sources of allochthonous organic matter in estuaries. Mangrove streams provide significant amounts of both organic and inorganic carbon, particularly during out welling of certain tides such the Ebb (Maher *et al*., 2013). According to Douglas *et al*. (2005), flooding during wet season dumps significant volumes of organic matter from nearby wetlands. Light is a limiting factor in estuaries with high turbidity, hence allochthonous organic matter tends to have a higher metabolism requirements than autochthonous organic matter (Middelburg & Herman, 2007).

Pelagic production (phytoplankton) and benthic production (benthic microalgae, macroalgae, and seagrass) are used to describe autochthonous sources of organic matter. The amount of organic matter produced by these two sources is dependent on system size, type, depth, light climate, functional zone and trophic state of the system (Maher *et al*., 2013). More than 90 % of all autochthonous carbon fixation occurs in the upper estuary zone, where larger pelagic zones, deeper channel morphology and greater proximity to watershed nutrient loads are present (Bukaveckas, 2022). Benthic production is limited by a light climate in the middle to upper estuary zones, while in shallow oligotrophic systems, the supply of autochthonous organic matter predominates across the whole estuary.

2.2.3 Estuarine Ecosystem Services and Values

The ecosystem services provided by estuaries makes them very important coastal ecosystems. These services could broadly be classified into four: provisioning, regulatory, cultural and, ecological or supporting (Thrush *et al.*, 2013). The provisioning services in estuaries include food., e.g., shellfish and fish and raw materials including vegetation used as fertiliser, food for fish and livestock, shells for ornamentation and musical instruments, kelp bags for food storage and transportation among others. Shellfish species like pipi (*pipi spp- Paphies australis)* and Common Cockle (*Cerastoderma edule)* reside in estuaries at different habitats during various stages of their life cycle. Some fish species are permanent dwellers in estuaries, like the Tule perch *(Hysterocarpus traskii)*, while others like the Red Snapper *(Lutjanus campechanus)* and Cook Strait Blue Cod *(Parapercis gilliesi)* only use the estuary as breeding grounds or as nurseries for their juveniles. Estuaries also attract large numbers of seabirds like the Great Blue Heron *(Ardea Herodias)* and marine mammals, like the Harbor Seals (*Phoca vitulina)* which contribute immensely to the estuarine food-web (Booi *et al*., 2022).

These regulatory services sustain life-support systems and playing crucial role in respiration, mitigating human impacts and maintaining system integrity. They include waste and climate regulation, storage and nutrient cycling, sediment formation and stability, as well as shoreline protection. For examples, estuarine organisms are responsible for waste modification and contaminant removal through binding, sequestration and burial. Some species of shellfish, e.g., Oysters *(Crassostrea spp.)* and metal reducing bacteria, e.g., some strains of bacteria within the genera Shewanella and Geobacter that live in estuarine sediments break down the toxins in heavy metals. Sewage and other organic wastes are broken down through the effort of estuarine flora, fauna and microbes, which are transported across the food-web. Therefore, in estuaries, all life forms work together to transform energy and matter (Cloern *et al*., 2014). Also, some species of estuarine organisms, e.g., polychaete worms, bivalve molluscs, diatoms, cyanobacteria, etc., are responsible for storage of both organic and inorganic nutrients including their transformation and cycling.

Nutrient cycling takes place in both the water column and sediments (Testa *et al.*, 2018). The rate at which organic matter is broken down and re-mineralised is determined by the animals moving within the sediments (bioturbation) and how this affects pore water flow. Bioturbation initiates several processes in the pore water; impairment of chemical gradient, removal of organic matter, influencing decomposition rate, affecting erodibility and permeability rates of sediments, and releasing of inorganic nutrients from sediments to overlapping waters. Some estuarine organisms involved in bioturbation include Fiddler crabs *(Uca spp.),* Polychaete worms *(Capitella spp., Nereis spp.),* Clams (*Ruditapes spp*.), among others (Testa *et al.*, 2018). The supply of essential nutrients in estuaries like carbon, nitrogen, phosphorus and sulphur is dependent on the above processes.

Furthermore, estuaries regulate climate by facilitating gaseous exchange at the water-sediment-atmosphere interface, including balancing of oxygen and carbon dioxide, as well as regulation of other greenhouse gases (Thrush *et al.*, 2013). Primary producers in estuaries take up carbon dioxide for photosynthesis, but mangroves, seagrasses and other large vegetation provide long-term storage of carbon. Moreover, carbon is sequestrated in biomass stored in sediments in estuaries. Additionally, sediment in estuaries is generated by animals that make shells out of calcium carbonate, like bivalves and snails. Thrush *et al*. (2013) observed that, the shells can stay in sediments for over a century, which affects species diversity, richness and sediment quality. Sediment micro-algae plays a major role in sediment stabilisation through striking a balance between bioturbating animals that disturb the sediment and the resulting growth rate of the micro-algae. Although shell-producing animals have important sediment stabilising effects, some that dig holes in the sediment to feed or move across the sediment-water interface locally destabilise sediments, which makes them vulnerable to transportation by waves and tides (Thrush *et al.*, 2013). In mangroves and seagrasses with sufficient densities of worms and crabs that build structures, sediment erosion is reduced, while deposition rates increased for sediment suspended in water. Finally, fringing vegetation like mangroves and salt marshes are advantageous to downstream users as they hold on to water and control its release to ensure shoreline protection. The advantage particularly comes about during storms by profigating the tidal and wave energy hence reducing their impact as surges and storms (National Geographic, 2023).

As a transition between land, rivers and the sea, estuaries are not only a place for food gathering, but also a linkage to spirituality. Due to the proximity of population centres to estuaries, there is a strong connection between the estuaries and the country's cultural and spiritual heritage and most of the cultural activities involve being in, on or around water to drive customs, practices and values. Products that take cultural significance from estuaries include: cutters and scrappers from mussel shells, tusk bells used in making anklets and necklaces, scallop shells for holding pigments for tattooing (Thrush *et al.*, 2013).

Tourism and recreation activities in estuaries encompasses activities such as water skiing, swimming, diving, sailing, etc that involve direct contact with water. Indirect activities also related to recreation in estuaries are dog walking, bird watching and reclining on the beach (Thrush *et al.*, 2013). Cognitive benefits are the values of estuaries that stimulate cognitive development especially in education and scientific research. Information held in estuarine ecosystems can be harnessed for technological advancement, e.g., development of wear resistant ceramics from studying bivalve shells (Thrush *et al.*, 2013).

Habitat structures are pre-conditions for the provision of goods and services. For instance, estuaries offer refuge and breeding grounds for both resident fish like Tule perch *(Hysterocarpus traskii)* and migrant fish species like the Red Snapper *(Lutjanus campechanus)* and Cook Strait Blue Cod *(Parapercis gilliesi)* (Woke & Wokoma, 2000). In some parts of the world, estuarine flats act as habitats for migratory species of seabirds like Knots- Red Knot (*Calidris canutus*), Herons - Great Blue Heron *(Ardea Herodias)* and Wry bills *(Anarhynchus frontalis)* (Thrush *et al.*, 2013). Also, healthy estuarine ecosystems contain a gene bank of species that can be exploited for use in various industries like pharmaceuticals and aquaculture. Genetic resources maintain genetic diversity, which may become critical in the ability to respond to environmental changes (de Groot *et al.*, 2002).

2.2.4 Threats Facing Estuaries

The major threats faced by estuarine ecosystems are environmental and anthropogenic in nature such as mining, and discharge of domestic, industrial, and agricultural waste that heavily compromise their integrity (Thrush *et al.*, 2013). Sewage from residential and commercial areas does not only contain high content of pathogens, but it is also high oxygen demanding, which causes hypoxic conditions and eutrophication once discharged into estuaries. As a result, there is poor oxygen circulation since the little oxygen received cannot be recirculated (Welsh *et al.*, 1991). Hypoxia leads to death of benthic organisms, fish kills, decrease in the growth rate and reproductive rate, physiologic stress, forced migration, interference with life cycles, reduced spawning grounds, habitat change and increased vulnerability to predation (Sheldon & Alber, 2011). On the other hand, eutrophication contributes to harmful algal blooms increases turbidity, causes shifts in trophic interactions, and leads to loss of aquatic habitat (Rabalais *et al.*, 2002). Furthermore, various pathogens contained in sewage waste including viruses, bacteria and protozoa pose possible human health hazards especially in the face of ingesting contaminated fish and shellfish. The possible infections include typhoid fever from *salmonella* sp, dysentery from *shigella* sp, cholera from *Vibrio cholerae* and other viral infections (EPA, 2023b).

Industrial wastes are highly variable ranging from combustion of fossil fuels and petroleum spillages. Polynuclear Aromatic Hydrocarbons (PAHs) are among commonly encountered industrial waste products. They pose mutagenic, carcinogenic and teratogenic toxic effects to aquatic life and can potentially be transferred to the terrestrial environment. Benthic macroinvertebrates are highly impacted since PAHs are relatively insoluble in water and strongly adsorb to particulate matter (Ankley *et al.*, 2003). Furthermore, a wide range of agricultural activities upstream contaminate estuaries with pesticides, fungicides, insecticide and nutrients, among other oxygen demanding wastes. Majority of these contaminants have fat solubility potential, hence accumulate in lipid tissues of aquatic organisms and become biomagnified in food chains, causing potential health hazards to humans (Mohapatra & Phale, 2021).

Heavy metals (elements with atomic weight between 63-200 and include alkali metals, alkaline earth metals, lanthanides and actinides) are among common pollutants of estuarine ecosystems and become toxic to aquatic life above a certain threshold (Yi *et al.*, 2021). The most common ones are zinc, lead, mercury and copper, which settle in the sediments, biota and the water (Yi *et al.*, 2021). They are deposited in estuaries from freshwater run-off and anthropogenic activities such as ash disposals, smelting, burning of fossil fuels, dredging operations, fishing, farming, automobile emissions, oil refinery effluent, mining of metal ores, metal plating, as well as manufacture of dyes, paints and textiles (Lixia *et al.*, 2021). Mining activities are also a major source of contamination of estuaries with radioactive waste, especially gold and uranium mining and milling (Xiang *et al.*, 2019). Nuclear power plants and scientific research centres dealing with radioactive material are other sources. When ingested by estuarine organisms, they are biomagnified to higher trophic levels, damaging reproductive and somatic cells, leading to chromosomal abnormalities and cancer risks (Lushchak *et al.*, 2018).

2.2.5 Estuaries in Ghana

Ghana is blessed with more than 100 coastal wetlands, comprising more than 90 coastal lagoons and 10 estuaries spread throughout the entire length of the coastline (Yankson & Obodai, 1999). These estuaries include the Volta, Kakum, Pra, Butre, Akwida, Nyan, Owuku, Ankobra, Amanzule and Whin. Salt marshes, mangrove swamps and tidal flats are among coastal wetlands that are close to lagoons and estuaries. Together, these ecosystems form valuable features along the Ghanaian coastline, providing critical habitats for many fish species, as well as wildlife resources that support the country's economy (Aggrey-Fynn *et al.*, 2011). Important Bird Areas (IBAs) are formed by a few of Ghana's estuaries. For instance, the Volta Estuary is a crucial bird site for wintering waterbirds along the coast of Ghana, sustaining over 100,000 birds (BirdLife International, 2023), while the Sakumo Lagoon provides a home for 70 waterbird species with a total population of about 30,000 birds. Additionally, the Sakumo lagoon is a crucial ecological zone for three endangered sea turtle species, including the Olive Radley (*Lepidochelys olivacea*), Green Turtle (*Chelonia mydas*), and Leatherback Turtle (*Dermochelys coriacea*), which breed on the beaches along the Sakumo Lagoon. Blackchin tilapia (*Sarotherodon melanotheron*), which makes up approximately 97 % of the total fish population, is the most prevalent fish species in the lagoon (Zuh *et al*., 2019).

Gold mining operations, including large-scale exploitation for commercial purposes and illicit gold mining (Galamsey), are one of major threats facing Ghanian coastal ecosystems. Ecological health of estuaries is severely hampered by the widespread mining activity throughout Ghana (Essumang & Nortsu, 2008). For example, due to the gold mining operations in the catchment, extremely high turbidity in the Ankobra Estuary has been reported (Faseyi *et al*., 2022). High turbidity as a results of mining activities upstream increases heat absorption capacity of water, leading to higher temperatures that subsequently lower the concentration of oxygen and ultimately affecting primary productivity. High turbidity does not only clog fish gills but is also decreases visibility, which hinders breeding, feeding, reproduction, and ultimately the survival of aquatic life (Faseyi *et al*., 2022; Okyere, 2019).

In addition to the contaminating of estuarine water due to gold mining, heavy metals like mercury, lead and arsenic are also channelled upstream rivers, pass through mining fields, and finally reach coastal ecosystems (Marsden & Iain, 2002). Karikari *et al*. (2009) reported that sewage outfalls caused high rate of pathogen contaminations in estuarine water. Other risks to estuaries in Ghana include; pollution from both land and sea bed sources, accelerated coastal erosion, spread of invasive species, overexploitation of fisheries resources, sea level rise due to climate change, and use as dumping sites (DeGraft-Johnson *et al*., 2010).

2.3 Assessment of Health Status of Estuaries Using Physicochemical and Biological Indicators

Estuaries, just like any other ecological systems, need to be healthy and free from environmental distress, which occurs in case of instability and unsustainability. When faced with external stress, a healthy estuarine ecosystem should maintain its structure and function over time (Chilton *et al*., 2021). Estuarine ecology, human health and socio-economic activities and livelihoods are factors linked to ecosystem services, and they determine estuarine ecosystem health (Tallam & White, 2023). Indicators of estuarine ecosystem health are parameters that reflect the overall health of organisms and include physicochemical and biological indicators.

The assessment of health status of an estuary using physicochemical indicators involves evaluation of physicochemical parameters such as surface water temperature, dissolved oxygen, turbidity, pH, transparency, chlorophyll-*a*, heavy metals, salinity, nutrients, etc. Major characteristics of biological communities, such as productivity, species composition, diversity, and other bio-interactions in estuaries are influenced by physicochemical parameters. Physical and chemical characteristics are employed to measure pollution levels as a result of both anthropogenic and natural sources (Nwanosike *et al*., 2010; Yisa & Oladejo, 2010). Measurement of physicochemical quality of estuarine water could provide conclusion on the pollution status, qualifying this to be a stressor-based approach. In this context, a stressor is any physical or chemical entity that is likely to create an imbalance in an ecosystem (EFSA Scientific Committee, 2016).

Estuarine water quality has been assessed using physico-chemical parameters over the years. Some of the estuaries include Kollidan Estuary (Edward & Ayyakkannu, 1991), Devi Estuary (Pradhan *et al.*, 2009), Narmada Estuary (Isaiah *et al.*, 2012), Mahi Estuary (Isaiah *et al.*, 2013), Tapi Estuary (Nirmal Kumar *et al*., 2009; Ujjania & Dubey, 2015) among others. However, for holistic water quality assessment, there is need for inclusion of biological indicators, a gap that the current study seeks to address.

The assessment of health status of estuaries using biological indicators entails estimating the biomass generated by diverse communities of organisms such as phytoplankton, macrophytes, macroinvertebrates and fish. Benthic macroinvertebrates are the most popular of these communities since they are generally simpler, cheaper, and easier to collect and identify using current diversity monitoring indices (Edegbene *et al.*, 2021). Additionally, different species of benthic macroinvertebrates react differently to pollution and habitat change, making it easy to quantify their diversity and density in a given locality, which also gives an idea of the prevailing environmental conditions (Edegbene *et al.*, 2021). For instance, pollution-sensitive species, like stoneflies (Plecoptera), mayflies (Ephemeroptera), caddisflies (Trichoptera), flatworms, and leeches, are highly responsive to environmental pollution and are used as bioindicators to assess ecosystem health (Nerbonne & Vondracek, 2001; Nunkumar, 2002). Their absence or scarcity indicates pollution or environmental degradation (Pinto et al., 2009). On the other hand, pollution-tolerant species, including midge larvae, oligochaeta, scuds, copepods, and snails, thrive in polluted or disturbed aquatic ecosystems (Barrilli *et al.*, 2021;Zhang *et al.*, 2019).

Furthermore, various species vary in sensitivity to hydrological stress, bearing in mind that most are non-mobile and ubiquitous, enabling detections of perturbation and recolonisation patterns for both long- and short-term life cycles (Abbasi & Abbasi, 2011). The use of bioindicators is a response-based approach since they are more expressive and because of this they have gained more applicability as key elements in water resource management policy formulations (Holt & Miller, 2010). Except in few countries like South Africa and Serbia, most developing countries have not fully embraced bioindicators in water quality monitoring (Abbasi & Abbasi, 2011).

Benthic macroinvertebrates, which form the basis of the trophic level are a source of detritus when it comes to decomposition, play the role of detritus feeders and predator's food at secondary and tertiary levels in food chains, respectively (Sharma & Chowdhary, 2011). Benthic macroinvertebrates also play a vital role in sediment formation, structure and mineralisation process (Tampo *et al*., 2021). They facilitate physicochemical processes occurring in the interface zones between sediment and water, and in the interstitial water, including processes such as breakdown and distribution of toxins like trace metals and organic matter (Adámek & Maršálek, 2013). Moreover, benthic communities are sources of ecosystem services that impact the quality of water and that of sediments in estuaries (Tampo *et al.*, 2021). Suspension feeders enhance water clarity hence allowing deposition of organic matter from superimposed water, and this increases the thriving ability of submerged aquatic plants. Soil aeration is as well enhanced by the burrowing benthos. Research by Yankson & Kendall (2001) reports that, environments with plenty deposits of organic matter support high abundance of capitellid polychaetes due to their ability to survive under low oxygen levels. They therefore indicate potential pollution when occurring in association with other benthos.

2.4 Assessment of Estuarine Ecosystem Health Using WQIs

2.4.1 Evolution of the Development of WQIs

The concept of categorising water quality using a numerical value based on biological, physical, and chemical parameters dates back to the mid-20th century when the first Water Quality Index was developed in the United States (Horton, 1965). By applying expert opinion (Delphi technique), Horton selected 10 Water Quality Parameters including dissolved oxygen, pH, specific conductance, alkalinity, temperature, carbon chloroform extract, faecal coliforms (FC), chloride, percentage of population upstream that is connected to a sanitation facility and obvious pollution to develop a general use WQI. For each parameter, a rating scale of 0 to 100 was used and weightage of the parameters ranged between 1 and 4. The final index aggregation involved weighting the sum of the sub-indices, divided by the sum of weights and multiplied by two coefficients that depended on temperature and level of pollution of water. Although this index was and has continuously been used to compare the efforts put in place by water quality improvement programmes, it does not absolutely asses the quality of water. Besides, it excludes the concept of toxicity in water (Kachroud, 2019). As an improvement methodology to Horton's work, Brown and his colleagues in 1970 established a new WQI using the Rand Corporation's Delphi Technique (Brown *et al.*, 1970). The methodology was purely based on expert opinion in both parameter selection and assignment of sub-indices. Brown selected nine parameters including: total phosphates (TP), biochemical oxygen demand, dissolved oxygen, pH, temperature, FC, total nitrates (TN), total solids (TS) and turbidity. Brown singled out 142 water quality experts, whose opinion was used to reduce the parameters from 44 to 9, assigned the weightings to each variable and establish 5 water quality classes, ranging from red (very poor), orange (poor), yellow (average), green (good) to blue (excellent). The overall aggregation of this index was arithmetic in nature, but when a single variable exceeds the norm, the index becomes less sensitive (Lumb *et al.*, 2011). These works were supported by the United States National Foundation (NSF), hence the index was named NSFWQI, (1970). To improve on this scenario, geometric aggregation was adopted by Brown *et al*. (1972).

In 1971, a similar WQI to the NSFWQI was proposed by Deininger and Landwehr (Dinius, 1972a) consisting of 12 parameters for surface waters and 14 parameters for groundwater, including temperature, dissolved solids, DO, FC, pH, BOD, nitrates, phosphates, phenol, turbidity, hardness and colour for surface waters, and the same parameters, in addition to iron and fluoride concentrations, for groundwater. Within the same year, another index was proposed by Prati *et al*., (1971), based on water quality standards, utilising 13 parameters, including BOD, DO, COD, pH, iron, alkyl benzene sulphonates, nitrate, ammonium, carbon chloroform extract, concentrations of permanganate, chloride, manganese, and suspended solids (SS). The scheme behind this index was to convert pollutant concentrations into pollution levels.

From 1971 to 1990, several other indices were proposed, mostly modifying and improving on the already existing indices. These included; the general WQI modified index based on Horton's index (Dinius, 1972b), modified index based on Dalkey's work (Dinius, 1987), pollution load WQI (Bhargava, 1983), modified index from Horton and Brown and his colleagues. Later, the Canadian Council of Ministers of the Environment (CCME), through the "Water Quality Guidelines Task Group" in the mid-1990s modified the original British Columbia Water Quality Index into the CCME Water Quality Index (WQI), and, subsequently endorsed by the CCME (2001). This was a non-liner index that established its roots on frequency of sampling and measurement, frequency of values outside the required objectives and the deviation from recommended value of each variable (Lumb *et al.*, 2006). It comprises of three factors: Factor 1 (F_1) deals with scope that assesses the extent of water quality guideline non-compliance over the time period of interest; Factor 2 (F_2) deals with frequency, i.e., how many occasions the observed value was off the acceptable limits; and Factor 3 (F_3) deals with the amplitude of deviation or the amount by which the objectives are not met. This index is flexible in the selection of input parameters and involves simple calculation processes (CCME, 2001).

Since the 1990s, a lot more methods have been employed, computing various water quality parameters for the purpose of deducing the environmental quality of waters (Ludwig, 1996). The NSFWQI was improved in 1996 by the Lower Great Miami Watershed Enhancement Program (WEP) in Dayton, Ohio, through developing an index (WEPWQI) that comprised physicochemical and biological parameters of a river, including flow measurements, water quality parameters and water clarity. This was an improved index from the NSFWQI, since it considered contamination by pesticides and Polycyclic Aromatic Hydrocarbons (Said *et al*., 2004). The most recent method of calculating WQI is based on fuzzy logic, as instituted by Icaga (2007). This model is a multi-valued logic that expresses the partial truth between being false or true, and takes into account both subjective and non-quantitative data. The model is based on a numerical scale that represents water quality and it increases the sensitivity of the method, giving a rigorous framework to evaluation (Kachroud, 2019).

Development of WQI has encountered several shifts, including the focus on specific use of water resources rather than general purpose (Richardson, 1997). Nevertheless, a stepwise shift has been recognised, developing from general purpose indices, to specific use indices and finally to planning and statistical based approach between 1960s to 1980s (Couillard & Lefebvre, 1985). Afterwards, majority of the WQIs being developed focus more on general use of water. Secondly, there has been a shift from pure water to brackish water systems. The already developed WQIs focus on the quality of both lotic and lentic systems, waterways and pure water (Dinius, 1987), however, very few indices focus on estuarine (Richardson, 1997) and marine environments (Vollenweider *et al*., 1998). The third trend is how the final transfigured results are expressed (Couillard & Lefebvre, 1985). As a norm, WQI scores are expressed as a single numerical value. However, recent developers are incorporating numerical or alphanumerical values to ascertain the robustness of the value obtained (Rawat *et al*., 2015).

2.5 Summary

This chapter critically reviewed key concepts related to estuarine ecosystems, WQIs and their relevance and applicability globally, regionally and in Ghana. Estuarine ecosystems were discussed in general with respect to types, structure and functioning. Characteristics of estuaries were well illustrated, narrowing down to some of the most important physicochemical parameters relevant to the current study, i.e., salinity, temperature, DO, pH and turbidity. The chapter went further and broadly examined the estuarine values and uses, scaling deeper into the four major ecosystem services including provisioning, regulatory, cultural and habitat and ecological community services. Furthermore, assessment of estuarine ecosystem health using physicochemical and biological indicators was reviewed. Finally, the chapter wrapped up by reviewing development of WQI, including evolution, application and processes.

ADAPTED WATER QUALITY INDICES: LIMITATIONS AND POTENTIAL FOR WATER QUALITY MONITORING IN AFRICA

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Published: MDPI -Water

(*Water* 2023, *15*(9),1736; https://doi.org/10.3390/w15091736).

Statement of Joint Authorshop Contributions

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P.K.M: Review, Editing and Supervision (**Principal Supervisor**)

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Abstract

A Water Quality Index (WQI) is a tool that describes the overall water quality by combining complex and technical water quality information into a single meaningful unitless numerical value. WQIs express water quality since they reflect the impact of multiple Water Quality Parameters (WQPs) and allow for spatial-temporal comparison of water quality status. Most African countries employ adapted WQIs by modifying the original index (or indices) and propose their concepts for evaluating the quality of surface and groundwater, which is normally accompanied by irregularities. The current review examined the process(es) involved in WQI modifications for monitoring water quality in Africa, explored associated limitations, and suggested areas for improvement. A review of 42 research articles from five databases in the last 10 years (2012–2022) was conducted. The findings indicated Weighted Arithmetic (WAWQI) and the Canadian Council of Ministers of Environment (CCME-WQI) as the most adapted WQIs. Several limitations were encountered in WQI developmental steps, largely in parameter selection and classification schemes used for the final index value. Incorporation of biological parameters, use of less subjective statistical methods in parameter selection, and logical linguistic descriptions in classification schemes were some recommendations for remedying the limitations to register the full potential of adapted WQIs for water quality monitoring in Africa.

Keywords: Water Quality Index; Water Quality Parameters; physicochemical parameters; multivariate statistics; benthic macroinvertebrates; microbiological parameters.

3.1 Introduction

The destruction of natural resources, especially the contamination of aquatic habitats, has been hastened by continued worldwide population increase and socio-economic development. The quality of surface water has been and continues to be impaired by the increased discharge of physical, chemical, and biological contaminants in water sources, which also stresses aquatic life (Aljanabi *et al*., 2021). These contaminants either come from non-point sources (such as surface runoff, airborne contaminants, and sewage outflows), point sources (such as industries and direct effluent disposal), and/or hydro-morphological sources (such as those related to natural processes and human activities including water abstraction). It is essential to develop a management strategy to reduce any potential threats to aquatic life and public health in light of the sources of contaminants in aquatic ecosystems. The amount and nature of contaminants present in water should be considered when deciding whether it is suitable for any usage (drinking, irrigation, recreation, habitat for aquatic life, industrial operations, etc.). Normally, these contaminants are expressed using contamination parameters (Soumaila *et al*., 2019) defined in water quality indices WQIs.

A WQI is a tool that describes the overall water quality by combining or summarising complex and technical water quality information into a single, meaningful, unitless numerical value (Lumb *et al*., 2011; Zeinalzadeh & Rezaei, 2017). Therefore, WQIs are used to express water quality status since they reflect the overall impact of multiple WQPs and allow for comparing water quality status across time and space (Tiwari & Mishra, 1985). Specifically, WQIs are used to: (i) communicate information about water quality to the general public, policymakers, and non-water experts in a straightforward manner (Poonam *et al.*, 2013) and (ii) improve the understanding of general water quality issues by stakeholders in the decision-making process.

3.1.1 The Development of a WQI

Development of most WQIs involves the following four fundamental steps: (i) Parameter selection; (ii) Estimation of sub-index values for parameter comparison on a common scale; (iii) Weighting of parameters based on their relative significance to the overall water quality; and (iv) Formulation and computation of the overall index (Tyagi *et al*., 2013). Selecting the right water quality parameters under specific environmental conditions is the most challenging of the four steps (Boyacioglu, 2007). A comprehensive analysis of WQI development steps is provided in the later sections of this review. To reduce biases and select the right number and types of parameters, two approaches have been proposed that eventually yield different classes of indices. These are (1) approaches that rely on expert opinion (Rand Corporation's Delphi Technique) and (2) statistical-based approaches (Banda & Kumarasamy, 2020c). Using the premise that "Two heads are better than one," the Delphi approach elicits and refines a group's judgment. It was first developed by "The Rand Corporation" in the USA and included three basic features; anonymity of responses, repetitive and controlled feedback, and a group response (an appropriate aggregate of individual opinions on final round).

Although the features mentioned above are designed to reduce the biasedness of dominant individuals, irrelevant communication, and groups' pressure towards conformity, the final WQI value is mostly subjective as it is solely based on the advice of consulted experts (Kachroud, 2019). Some indices have been developed through such expert opinion, for example, the NSFWQI (Brown *et al.*, 1970), a public index used to monitor general water quality, Oregon and British Columbia WQI (Banda, 2015), and the WQI by Hallock and Ehinger (Hallock & Ehinger, 2003), a planning index used as a decision-making tool for designing water quality management projects. On the other hand, statistical-based approaches aim to lessen subjectivity and increase the accuracy of the final index. Principal Component Analysis (PCA), Discriminant Analysis (DA), Cluster Analysis (CA), and Factor Analysis (FA) are some of the multivariate techniques employed (Liu *et al.*, 2011). Table 2 provides a summarised history of WQI development along with the progress of each step in a chronological manner.

Table 2: *Chronological Evolution of WQI Development*

WQI Model	Parameters	Standardisation	Weighting	Aggregation Function	Reference
Horton index	DO, pH, SC, Alk, temp, CCE, FC, Cl^-, PP, OP	Parameters value used as sub-index value	Fixed and unequal system	Weighted Arithmetic Average $QI = \frac{\sum_{i=1}^{n} Wili}{\sum_{i=1} Wi} M_1M_2$	(Horton, 1965)
NSFWQI (1)	DO, BOD, pH, temp, FC, Turb, nitrates, phosphates, TDS	Rating curves; Experts' opinion	Expert questionnaire	Weighted Arithmetic Average $WQI = \sum_{i=1}^{n} WiQi$	(Brown *et al.*, 1970)
Prati's implicit index	BOD, pH, COD, DO, conc. of permanganate, ammonium, nitrates, Cl^-, Fe, Mn, CCE, SS Alkyl Benzene sulphonates	Linear and parabolic function	No	Additive $I = {}^{1}/_{13} \sum_{1=i}^{13} li$	(Prati *et al.*, 1971)
Dinius index (1)	Temp, DO, pH, EC, colour, BOD, Alk, FC, Cl^-, hardness, *E. coli*	Linear and nonlinear function	Delphi technique	Weighted $DWQI = \frac{1}{21} \prod_{i-1}^{11} li^{wi}$	(Dinius, 1972a)
NSFWQI (2)	DO, BOD, PH, temp, FC, Turb, nitrates, phosphates, TDS	Rating curves; Experts' opinion	Expert questionnaire	Weighted Geometric Average $WQI\ (M) = \prod_{i=1}^{n} Qi^{wi}$	(Brown *et al.*, 1972)
Stoner's index	Irrigation: EC, SAR, SC, Mn, B, As, Cd, Be, Al, Co, Cr, V, Ni, Cu, Zn, F Public water: Cl, MBAs, phenols, nitrates, ammonia, colour, pH, Cu, FC, F, Fe, Zi, sulphates	Limits classes: nonlinear functions	Researcher's experience	Additive $I = \sum_{i=1}^{m} li + \sum_{j=1}^{n} Wjlj$	(Stoner, 1978)
Dinius Index (2)	Temp, DO, pH, EC, colour, BOD, Alk, FC, Cl^-, hardness, *E. coli*, nitrates	The linear and non-linear function	Delphi technique	Geometrical average $IWQ = \prod_{i=1}^{n} li^{wi}$	(Dinius, 1987)
Bhargava index	According to the use	Formulas	Weighted Product	Additive $WQI = [\prod_{i=1}^{n} fi(pi)] \times 100^{1/n}$	(Bhargava, 1985)
Smith index	BOD, temp, Turb, SS, DO, ammonia, FC	Rating curves; Experts' opinion	Delphi technique	Minimum operator $I\ min = \sum min\ (Isub1, Isub2, \ldots \ldots Isubn)$	(Smith, 1990)
CCME-WQI	Minimum of 4, not specified	Standard values	No	Arithmetic average $100 - \frac{\sqrt{F1^2+F2^2+F3^2}}{1.732}$	(CCME, 2001)

Table 2, *continued*

New WQI	DO, TP, FC, Turb, SC	No need	Ranking	Logarithmic aggregation SAID WQI = log $\left(\frac{(DO)^{1.5}}{(3.8)^{TP}(Turb)^{0.15}(15^{Fcol}/_{10000}+0.4(SC)^{0.5}}\right)$	(Said *et al.*, 2004a)
Ewaid index	COD, TDS, DO, total hardness, Cl^-, TC	Rating curves; Experts' opinion	Expert opinion	Formula [(−0.019 TDS + 84.587) × 0.2] + [(−0.006 TC + 86.231) × 0.2] + [10 DO × 0.2] + [((−0.119 TH) + 113.68) × 0.15] + [(−5.886 COD +99.846) × 0.1] + [(−0.12 Cl+ 106.58) × 0.15]	(Ewaid *et al.*, 2020)

The Table provides an overview of various WQIs used overtime to monitor water quality. Each WQI has been explained in terms of parameters used, the methods of sub-index development and weighting and finally computation of the final index. All the WQIs in the table are original indices developed outside the shores of Africa, from the period between 1965 and 2020. Sourced from; (Aljanabi et al., 2021; Kachroud, 2019; Lumb *et al*., 2011; Poonam *et al.*, 2013; Smith, 1990; Walsh & Wheeler, 2012; Swamee & Tyagi, 2000). Notes; SC-Specific Conductivity, DO-Dissolved Oxygen, Alk-Alkalinity, CCE- Carbon Chloroform Extract, FC-Faecal Coliforms, Cl- Chloride, PP- Percentage of Population, OP-Obvious Pollution, Turb-Turbidity, BOD-Biochemical Oxygen Demand, TDS-Total Dissolved Solids, Mn-Manganese, Fe-Iron, SS-Suspended Solids, EC-Electrical Conductivity, SAR- Sodium Adsorption Ratio, B-Boron, Be-Beryllium, V-Vanadium, F-Fluorine, TP-Total Phosphorus, COD-Chemical Oxygen Demand, TC-Total Coliforms, As-Arsenic, Pb-Lead, Al-Aluminium, Co-Cobalt, Cr-Chromium, Ni-Nickel, Cu-Copper, Zn-Zinc, MBAs-Methylene Blue Active Substances.

3.1.2 Application of WQIs: The African Perspective

The concept of indexing water using a numerical value based on biological, physical, and chemical parameters dates back to the mid-20th century when the first WQI was developed in the United States (Horton, 1965) and applied in the UK and Europe in the 1970s and later in Africa and Asia (Ramesh *et al.*, 2010). Since then, there has been significant improvement and modification to existing indices as well as the development of new models. Some of the available WQIs include the Scatter score index (Kim & Cardone, 2005), Index of River Water Quality (Liou *et al.*, 2004), Overall Index of

Pollution (Sargaonkar & Deshpande, 2003), Chemical WQI (Tsegaye *et al.*, 2006), Universal WQI (Boyacioglu, 2007), CCME-WQI (CCME, 2001), NSFWQI (Kumar & Alappat, 2009), Oregon WQI (Dinius, 1987), and Weighted Arithmetic WQI (Manju, *et al.*, 2014). These and many other WQIs have been developed with global and regional applicability. However, the most commonly used and applied indices are the NSFWQI, CCME-WQI, and WAWQI, according to reviews of WQIs by Aljanabi *et al.* (2021), Poonam *et al.* (2013) and Tyagi *et al.* (2013).

The methodology for developing the NSFWQI was purely based on expert opinion in parameter selection and sub-indices assignment. Out of 35 possible parameters from which 142 experts were expected to select, only nine were selected for index construction (Brown *et al.*, 1970). This indexing is easily communicable to non-water experts, while the single index value obtained is considered objective and reproducible (Wills & Irvine, 1996). However, the index only represents general water quality and has a high data loss rate. Since it is implemented with only nine input parameters, any additional parameters require extra effort and careful consideration (Noori *et al.*, 2019). Due to this, the index has found limited applicability, especially in Africa, with only Nigeria adapting and applying it to assess drinking water quality as applied by Kalagbor *et al.* (2019).

The CCME-WQI was incepted in 2001 for use within the Canadian jurisdiction. It comprises three factors: Factor 1 (F_1) deals with a scope that assesses the extent of water quality guidelines for non-compliance over the stipulated period. Factor 2 (F_2) deals with frequency, i.e., how often the observed value was off the acceptable limits. Factor 3 (F_3) deals with the amplitude of deviation or the amount by which the objectives are not met. This index is flexible in the selection of input parameters and involves simple calculation processes. The calculation formulae involved in final index computation for CCME-WQI are comprehensively discussed in later sections of this review. Due to its flexibility in adapting to various WQPs and legal requirements by water agencies in different countries with little modifications (Abbasi & Abbasi, 2011), the index has found both global and regional applicability. In North Africa, the index has been applied in Egypt to monitor surface water for irrigation purposes (Goher *et al.*, 2019) and for evaluating water for the protection of aquatic systems (Abukila, 2015; Goher *et al.*, 2019). In Ghana, West Africa, the CCME index was used to assess the surface water quality of Aby Lagoon for the protection of aquatic life (Miyittah *et al.*, 2020), while river water for domestic use was also assessed (Egbi *et al.*, 2019). Furthermore, in East Africa, the CCME index has been applied to test the suitability of groundwater for drinking around Lake Victoria goldfields of northwestern Tanzania (Ligate *et al.*, 2022). Despite the versatile applicability, this index applies similar importance to all parameters, is highly subjective, and does not provide guidelines about the objectives specific to each location and particular water use (Terrado *et al.*, 2010). In addition, the index calculation does not involve sub-index generation for the parameters, establishment of weights, and classical index aggregation (CCME, 2001).

The WAWQI is overly calculated by linearly aggregating the sub-index values with the unit weight. With its methodology being modified over time, the WAWQI is among the top three indices universally used since adjustments can be made depending on the parameters in place and system status.

Furthermore, the WAWQI is the only index among all specific use indices requiring the least parameters (Tyagi *et al.,* 2013). It is also suitable for assessing ground and surface water meant for human consumption (Yogendra & Puttaiah, 2008). As a result, the index has been extensively applied in Africa.

In North Africa, several authors have employed WAWQI in the evaluation of groundwater for drinking purposes, for example in Lybia (Salem *et al.*, 2022) and Egypt (Hagage *et al.*, 2022; Rabeiy, 2018) as well as in Tunisia for irrigation and protection of aquatic life (Khmila *et al.*, 2021). Also, this index has been embraced in East Africa to examine surface and groundwater for various uses. In Kenya, both lake and river water were assessed for their ability to cause human health risks (Githaiga *et al*., 2021; Njuguna *et al.*, 2020), while the potability of river water was investigated for use (Chebet *et al*., 2020; Robert *et al*., 2021). Likewise, WAWQI has been applied by Terrado *et al.* (2010) and Yogendra & Puttaiah, (2008) to examine the appropriateness of surface water for human use in Ethiopia.

Many studies have been performed in West Africa using WAWQI, especially in Nigeria. Both surface and groundwater have been appraised for drinking and other domestic purposes (Nwanosike *et al.*, 2010; Yisa & Oladejo, 2010 Akoteyon *et al.*, 2011; Oni & Fasakin, 2016; Ochelebe & Kudamnya, 2022). Additionally, in Ghana, it has been put under usage to assess the portability of river and dam water for drinking (Boah *et al*., 2015; Akoto *et al.*, 2021) as well as groundwater suitability for both drinking and other domestic uses (Boateng *et al.*, 2016). Aside Nigeria and Ghana, a study involving WAWQI was documented in Chad where the suitability of groundwater as drinking water was established (Bon *et al.*, 2021). Despite the wide application, this index cannot meet many uses of the water quality data, and, at the same time, some of the water quality parameters may not be included in the overall index (Akoteyon *et al.*, 2011).

In South Africa, new methodologies have been proposed to establish the applicability of surface water for various purposes besides adapting the existing WQI models. Among the new models are the Equitable raw water pricing model (Banda, 2015), Universal WQI for South African river catchments (Banda & Kumarasamy, 2020b), and Surrogate WQI for South African watersheds (Banda & Kumarasamy, 2020a). Similar to other African regional blocs, the fitness of both surface and groundwater has been assessed for drinking and domestic purposes using both WAWQI (Molekoa *et al*., 2019; Belle *et al.*, 2021; Molekoa *et al.*, 2021; Mandindi *et al.*, 2022; Molekoa *et al.*, 2022) and CCME-WQI (Namugize & Jewitt, 2018), respectively.

From the above discourse, it is apparent that there exists a tendency to use WQIs adapted from developing countries in Europe and the Americas. The WQI adaptation process is by modifying the original WQIs and proposing their use in evaluating the surface and groundwater quality. Notably, WQIs are developed for a given location and are source-specific. Despite that fact, it is generally acceptable to adapt and modify WQIs in compliance with varied regulatory criteria for water agencies in different nations (Sutadian *et al.*, 2017).

However, before modifying an index for use, it is important to comprehend its development and relationship to local contexts. This evaluation pertains to the initial factors that supported its construction, as represented by WQPs (Cude, 2001), and their applications (Smith, 1990). If this is not considered, the index picks up irregularities, including ambiguity, rigidity, and eclipsing. Because sub-index values for all WQPs are obtained differently, ambiguous indices suggest worse water quality than expected. Rigid indices, on the other hand, are not adaptable enough to include extra or substitute WQPs. Rigidity happens when impairment develops in a parameter excluded from the WQI or when an index is used in a setting with different usage objectives for which it was designed (Swamee & Tyagi, 2007). Lastly, eclipsing issues frequently arise when a low sub-index value is concealed by a high overall WQI value (Swamee & Tyagi, 2000). From the African perspective, WQIs have been adapted to address various societal purposes guided by the different water uses. The current review, therefore, examined how WQIs have been modified and adapted for monitoring water quality in Africa. Additionally, it explored limitations in the modified WQIs and suggested areas for improvement for their application and full potential to be realised.

3.2 Materials and Methods

3.2.1 Data sources, inclusion and exclusion criteria, analyses

Articles were collected and analysed between March 2021 and June 2022 from five electronic databases: ScienceDirect, Springer, Google Scholar, ResearchGate, and semantic scholar. The search keywords included: “water quality index’, “Africa”, “surface water”, and “groundwater” for the past 10 years. Only studies that evaluated water quality using the original WQI of another author or authors incorporating physical, chemical, and microbiological parameters were included. The different modification approaches included replacing the type and/or quantity of WQPs, modification of either of the developmental steps, and changing the application or usage of the WQI. Articles that developed a new WQI approach were excluded because the objective of the review was to explore adapted WQI and not original models. MS Excel 2019 was used for graphic presentations, Sigma Plot v.14 for descriptive statistics, and Xlstat for multivariate analysis to identify the most popular WQP combinations and average linkage between the WQPs as used by authors.

3.3 Results and Discussion

Figure 1 displays the flowchart of how articles were located, evaluated, and selected. The five databases produced a total of 165 articles. Forty-two articles were included in this study to eliminate duplications.

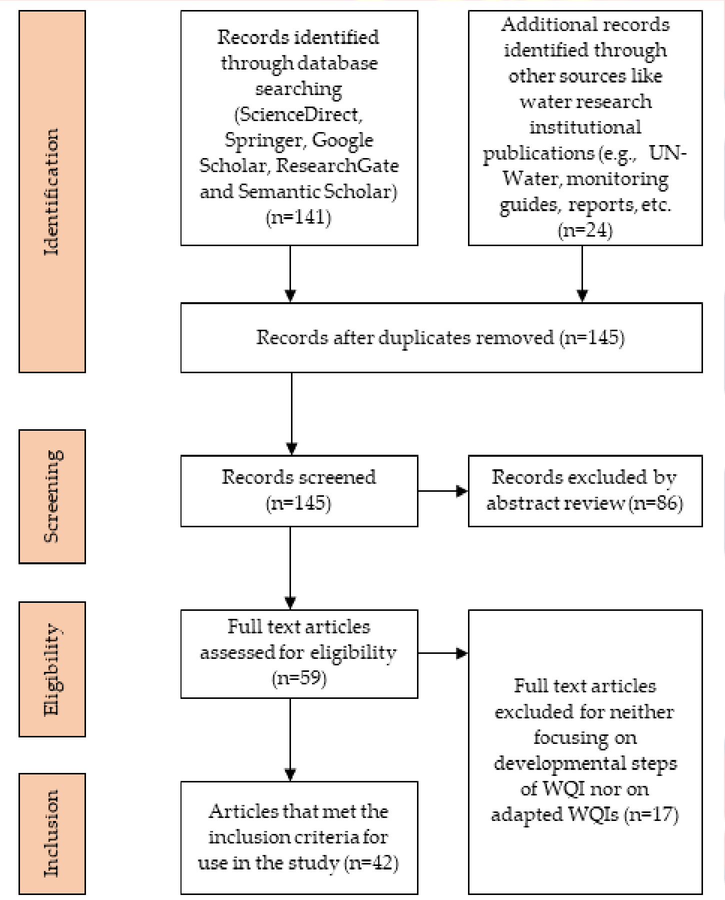

Figure 1: Flow diagram of the research protocol and selection of articles
Adapted from Moher *et al.* (2009)

The reviewed articles were a collection from 12 African countries (Figure 2), with Nigeria recording the highest number of modified WQIs. Tanzania, Mozambique, Libya, Chad, Tunisia, and Namibia with the least.

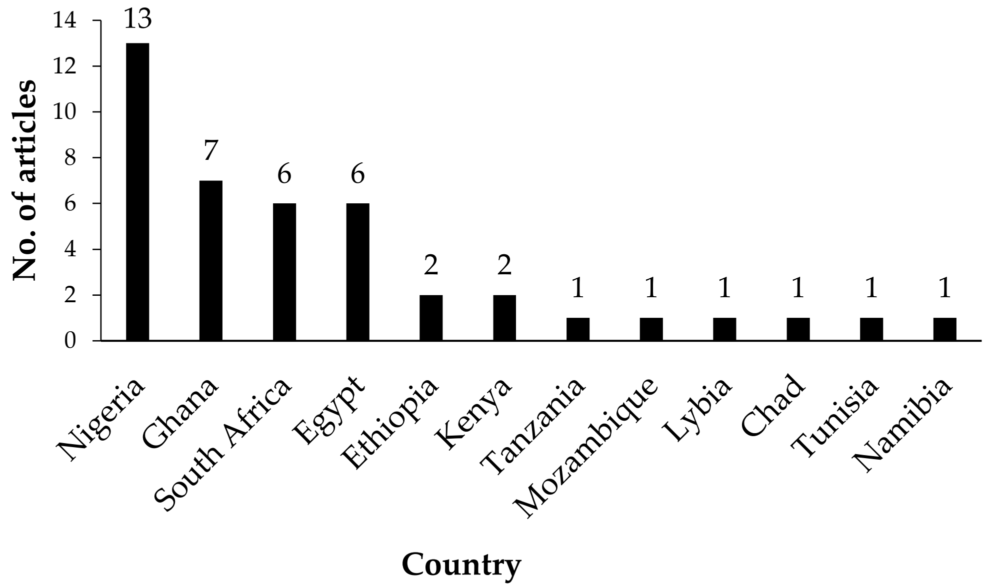

Figure 2: Number of articles with adapted WQI over the last 10 years

The CCME-WQI was adapted by eight articles, including (Goher *et al.*, 2019; Abukila, 2015; Miyittah *et al.*, 2020; Egbi *et al.*, 2019; Ligate *et al.*, 2022; Namugize & Jewitt, 2018; Sirunda *et al.*, 2022), whereas the WAWQI was adapted by 34 articles, including (Khmila *et al.*, 2021; Njuguna *et al.*, 2020; Akoto *et al.*, 2021; Belle *et al.*, 2021; Anim-Gyampo *et al.*, 2019; Bankole *et al.*, 2022; Marove *et al.*, 2022; Solihu & Bilewu, 2022; Wali *et al.*, 2022). The three developmental steps used by the articles that adapted CCME-WQI were the Scope (F_1), Frequency (F_2), and Amplitude (F_3) calculations. On the other hand, those that adapted the WAWQI followed the four main aforementioned developmental steps. From parameter selection, the generation of parameter sub-indices (step 2) and assignment of parameter weights (step 3) was done interchangeably depending on the author's preference before the final computation of the WQI using an aggregation function (Figure 3).

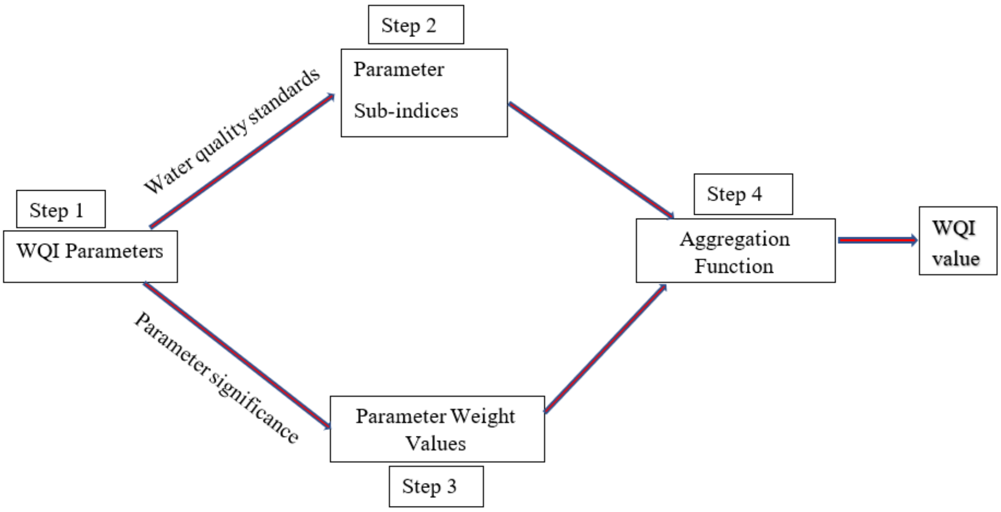

Figure 3: Development steps of WAWQI in adapted indices

Based on the overall index value, the water quality was rated using a categorisation scheme (Uddin *et al.*, 2021). About 66.7 %, 28.6 %, and 4.7 % of the articles focused on surface water, (Goher *et al.*, 2019; Abukila, 2015; Miyittah *et al.*, 2020; Egbi *et al.*, 2019; Oni & Fasakin, 2016; Molekoa *et al.*, 2019; Molekoa *et al.*, 2021; Molekoa *et al.*, 2022), groundwater (Hagage *et al.*, 2022; Khmila *et al.*, 2021; Boateng *et al.*, 2016; Uddin *et al.*, 2021; Ekere *et* al., 2019; Idehen, 2016), and a combination of both surface and groundwater (Ochelebe & Kudamnya, 2022; Marove *et al.*, 2022; Berhe, 2020; Mgbenu & Egbueri, 2019) respectively. The types and extent of the various societal needs addressed by the articles are shown in Figure 4. More than half of the reviewed articles addressed water for drinking purposes.

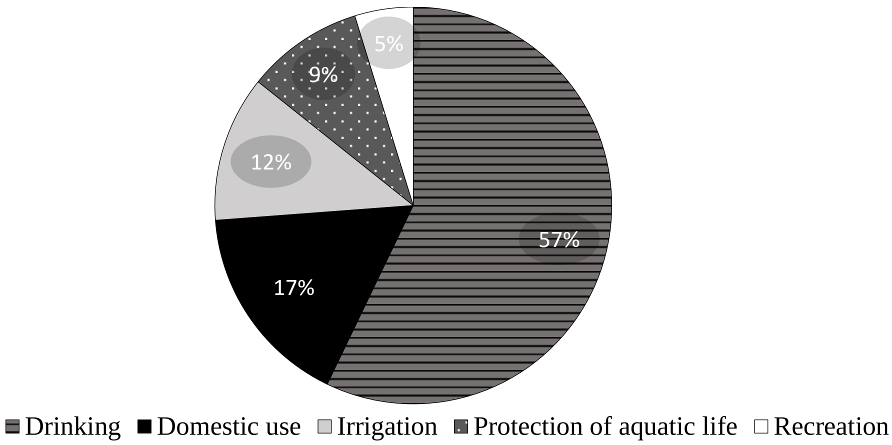

Figure 4: Percentage distribution of societal needs in the adapted indices

3.3.1 Parameter selection criteria in the adapted WQIs

From the articles reviewed, 65 WQPs were identified. On average, the articles focused on 14 WQPs with a minimum of 4 and maximum of 24, that examined the quality of swimming pool water for bathing (Ibanga *et al*., 2020) and the applicability of lake water for irrigation (Goher *et al.*, 2019) respectively. It has been affirmed that various physical, chemical, and biological factors influence the level of contamination in a specific aquatic system (Poonam *et al*., 2013). These categories were employed by authors in the current investigation and are detailed in Table 3.

Table 3: *The WQP Categories Utilised by Authors in the 42 Articles Reviewed*

Chemical Parameters (52)	**Physical Parameters (9)**	**Microbiological Parameters (4)**
Dissolved Oxygen (DO), pH, Total Phosphorus (TP), chloride (Cl^-), Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD), manganese (Mn), Soluble Reactive Phosphorus (SRP), cadmium (Cd), lead (Pb), chromium (Cr), copper (Cu), zinc (Zn), sulphates (SO_4^{2-}), arsenic (As), fluoride (F^-), mercury (Hg), selenium (Se), cobalt (Co), vanadium (V), magnesium (Mg), sodium adsorption ratio (SAR),Residual Sodium Carbonate (RSC), Magnesium Adsorption Ratio (MAR), nickel (Ni), calcium (Ca^{2+}), nitrites (NO_2^-), Total Nitrogen (TN), sodium (Na^+), total acidity, total alkalinity, calcium hardness (Ca H), magnesium hardness (Mg H), silica (Si), salinity, boron (Bo), electrode potential (Eh), bicarbonate (HCO_3^-), chlorophyll-a (Chl-a), iron oxide (Fe), Osmotic Potential (OP), barium (Ba), aluminium (Al), potassium (K^+), molybdenum (Mo), strontium (Sr), Uranium (U), titanium (Ti), carbonates (CO_3^{2-}), ammonium nitrogen (NH_4-N), nitrate nitrogen (NO_3-N), Isotopes 2-H and 18-O.	Hardness, temperature, colour, total solids (TS), turbidity, suspended solids (SS), total dissolved solids (TDS), and electrical conductivity (EC).	Fecal coliform (FC), *Escherichia coli* (*E. coli*), Total Fungi (TF), Total Coliforms (TC).

The table illustrates the various WQI categories employed in the reviewed articles. Chemical WQPs (81.5 %) were the most often employed characteristics, followed by physical WQPs (12.3 %) and microbiological WQPs (6.2 %).

With the input of a panel of 142 experts, the original WAWQI used the Delphi technique to determine the parameters used for index development (Brown *et al.*, 1970). In the reviewed articles, parameter selection was at the authors' discretion based on their relevance to water quality. With 57 % of articles focusing on water for drinking purposes, the parameters selected included *in situ* physicochemical parameters, nutrients, heavy metals, faecal indicator bacteria, and organic matter. For other domestic uses, stable isotopes 2-H and 18-O parameters selected for drinking water were also added while exempting colour, alkalinity, total acid, and faecal indicator bacteria. For irrigation purposes, similar parameters to those of drinking water were used with the addition of SAR, RSC, MAR, and TF excluding the stable isotopes. Physical parameters, nutrients, and heavy metals were considered in water to protect aquatic life and recreation. However, for recreation purposes, heavy metals were excluded as *E. coli* was included (Table 4).

Table 4: *Parameters Selected for Various Societal Needs in Adapted Indices*

	Parameters
Drinking	Temp, DO, BOD, TDS, TSS, EC, pH, hardness, colour, Alk, total acid, turb, F, Cl^- Mn, HCO_3^-, CO_3^{2-}, SO_4, NO_2-N, NO_3-N, PO_4-P, NH_4-N, Cu^{2+}, Fe^{2+}, Fe^{3+}, Cr, K^+, Na^+, Zn^{2+}, Ca^{2+}, Mg^{2+}, Pb, As, Ar, Si, Al, Ba, B, U, Se, Mo, Bo, Cd, Hg, Ni, Co, V, *E. coli*, FC, TC
Domestic use	Temp, pH, turb, EC, TDS, TS, hardness, DO, BOD, COD, NO_3-N, NH_4-N, PO_4-P, Cl^-, SO_4, Mg^{2+}, B, Fe, F, As, Cd, Si, Cr, Sr, Ti, Pb, Ni, Hg, Se, Al, Mn, CO_3^{2-}, K^+, Na^+ Zn^{2+}, Cu^{2+} Ca^{2+}, Mg^{2+} HCO_3^-, SO_4, FC, Eh, salt, SiO_2, Isotopes δ^2H and $\delta^{18}O$
Agriculture and irrigation	Temp, EC, TDS, TS, pH, BOD, DO, Alk, turb, colour, K^+, Na^+, Zn^{2+}, Ca^{2+}, Mg^{2+}, SO_4, Cl^-, SO_4, HCO_3^-, CO_3^{2-}, NO_3-N, PO_4-P, NH_4-N, SAR, RSC, HCO_3^-, MAR, hardness, B, Al, As, Cd, Co, Cr, Cu, Fe, Mn, Mo, Ni, Pb, Se, V, TC, FC, *E. coli*, Fungi
Protection of aquatic life	Temp, EC, pH, DO, turb, BOD, COD, TSS, TDS, Cl^-, Chl-a, F, NH_4-N, PO_4-N, NO_3-N, NO_2-N, SO_4, OP, Mn, Cu^{2+}, Zn^{2+}, Pb, Cd, Cr, Ni, Fe, TC
Recreation	Temp, pH, turb, EC, Cl^-, TSS, SRP, TP, NH_4-N, NO_3-N, *E-coli*

It is noteworthy that given the resource constraints with infrastructure, human capital, and financial resources in most regions of Africa, chemical parameters, especially heavy metals, were prioritised, followed by physical parameters and nutrients for societal needs. The original general use of WAWQI established nine fixed WQPs: DO, FC, pH, BOD, temperature, TP, TN, turbidity, and TS (Brown *et al.*, 1970). However, out of the 65 WQPs utilised in the current review, pH (88.1%), Cl^- (81 %), EC (73.8 %), NO_3-N (69 %), TDS (66.7 %), Ca^{2-} (66.7 %), Mg^{2+} %, SO_4^{2-} (57.1 %), Na^+ (47.6 %), and HCO^{3-} (42.9 %) were the 10 most often used WQPs in water quality analysis. Only pH from the original nine recommended parameters made an appearance in the 10 most popular WQPs in the adapted indices. This may be related to location-specific dimensions (Banda & Kumarasamy, 2020c), allocation, and usage (Kachroud, 2019). Additionally, the difference in parameters selected between the original WAWQI and adapted ones can only be interpreted considering that the original index was designed for the USA.

Furthermore, the original WAWQI was developed with a fixed set of WQPs. There is, therefore, a high likelihood that the final index scores in the reviewed articles faced the effects associated with parameter modification caused by index rigidity (Swamee & Tyagi, 2007). However, the initial CCME-WQI was designed with a minimum of four WQPs and no upper limit (Aljanabi *et al.*, 2021). The current study discovered that the eight articles that adapted CCME-WQI to construct their WQIs employed between 7 (Miyittah *et al.*, 2020) and 24 (Goher *et al.*, 2019) parameters. Since CCME-WQI offers the ability to incorporate more parameters based on existing environmental quality guidelines and local circumstances, the choice of the quantity of WQPs selected in the adapted WQIs was justifiable. Yet, to calculate index values, four of the selected WQP must have been sampled at least four times throughout the necessary sampling period (Misaghi *et al.*, 2017). Six of the eight articles in the current analysis showed that sampling was done at least four times. However, Egbi *et al.* (2019) and Ligate *et al.* (2022) did not specify the sampling frequency.

Also, when only one application needs to be evaluated using CCME-WQI, it is advisable to employ a core set of parameters, such as nutrients, heavy metals, physical parameters, etc. This inclusion is vital because too few parameters or too much covariance between them could enhance or decrease the significance of any one parameter, giving factor F_1 (scope) too much weight for determining the final index score (Terrado *et al.*, 2010). The chosen core set of parameters must also address the major environmental stress faced by the system to retain the relevance and correctness of the final index (CCME, 2001). According to the results of the current investigation, only Miyittah *et al.* (2020) maintained a single core set of parameters (nutrients). Figures 5 and 6 show the WQP utilisation in percentages employed by the author(s) to develop customised WQIs.

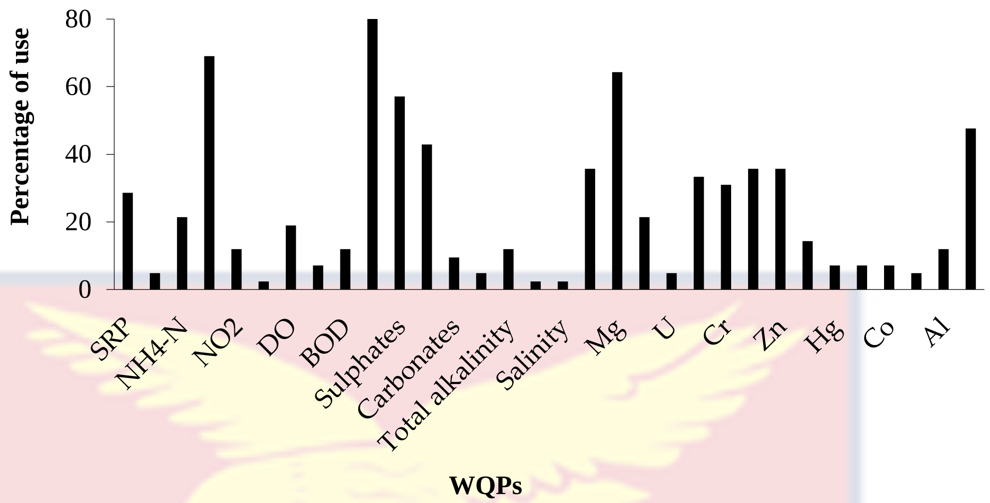

Figure 5: Percentage of WQPs used among the 42 articles

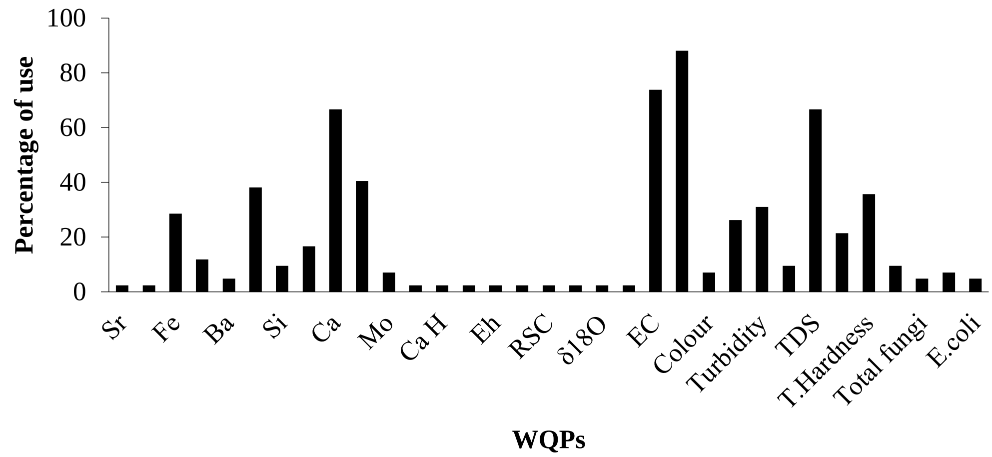

Figure 6: More WQPs as a continuation of Figure 5

Multiple WQPs were clustered to determine the frequently employed pairing by the articles examined (Figure 7). The cluster analysis connected WQPs based on the distance between parameters, and the more dissimilar the parameters were, the larger the distance was between them. All WQPs were grouped into four sub-clusters and two major clusters (Clusters 1 and 2) and (Sub-clusters 1.1, 1.2, 2.1 & 2.2). Cluster 2 was more heterogeneous with more homogeneous characteristics, while Cluster 1 was more homogeneous with a flatter dendrogram. Both clusters had 2 sub-clusters, with each sub-cluster bearing various combinations.

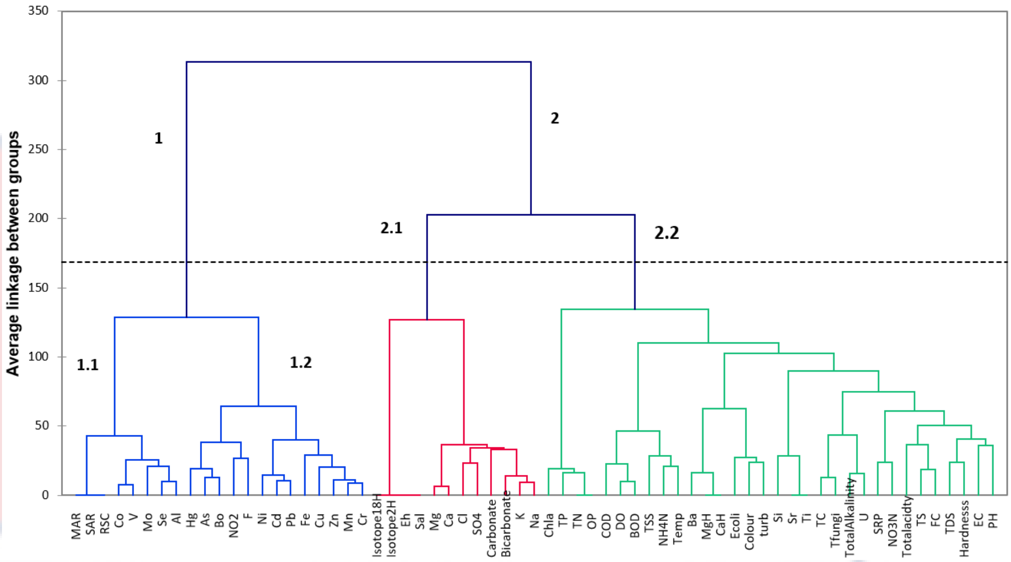

Figure 7: A parameter combination dendrogram from cluster analysis

The following were some of the authors' more inventive combo uses:

•Combination one; Zn^{2+}, Mn, Cu^{2+}, Fe, Pb, Cd, Ni, Bo, As, Hg, Al, Se, Mo, and Co with Cr, V, MAR, SAR, and RSC as outliers.

•Combination two; Mg, Ca, Cl^-, SO_4, HCO_3, K^+, and Na^+ with Eh, Sal, and the stable isotopes as outliers.

•Combination three; COD, DO, BOD, TSS, NH_4-N, Temp, *E. coli*, colour, turbidity, TC, Total fungi, Total Alkalinity, SRP, NO_3-N, Total acidity, TS, FC, TDS, Hardness, EC and pH with outliers like OP, MgH, CaH, Sr, Ti, and U.

Combination one consisted of a cluster of heavy metals associated with groundwater contamination, among other factors. Most of the authors who employed this combination investigated groundwater suitability for drinking, domestic use, or its suitability for irrigation. For example, (Rabeiy, 2018) and (Khmila *et al.*, 2021). On the other hand, combination three comprised physicochemical parameters in conjunction with nutrients and faecal indicator bacteria. These are parameters used to assess the ecosystem health of any water body, and almost all the authors included physicochemical parameters and nutrients for all the societal needs addressed. The faecal indicator bacteria parameter was included by authors who investigated water for drinking and recreation since faecal indicator bacteria indicates recent contamination of the system with faecal matter, hence the presence of faecal bacteria.

Generation of parameter sub-indices

In this developmental step, parameter concentrations and levels are compared on a similar scale and transformed into unit-less sub-index values from different units like ppm, saturation, %, mg/l, and counts (Terrado *et al.*, 2010). One can use expert judgment, water quality standards, and statistical techniques to create sub-indices.

Use of expert judgment

Judgement can come from a single expert or a team of specialists who create the critical points of rating curves and draw graphs to illustrate each parameter's effects on water quality at various concentration levels. The graphs are also transformed into linear or non-linear sub-index functions (Brown *et al*., 1970).

Use of water quality standards

The developed rating curves are transformed into sub-index functions using the recommended water quality standards. Because the critical points on the graphs are obtained using the recommended values for each particular parameter, this technique is less arbitrary than expert judgment. Sub-index values can be obtained through categorical scaling, linear interpolation rescaling, and comparison with recommended limits. Each recommended limit is allocated to the appropriate water quality class and a matching sub-index in linear interpolation, where the sub-indices vary from 0-100 or 0-1, much like the water quality classes listed in a sequence (Sutadian *et al.*, 2017). For illustration;

a) Recommended standards-20, 30, 40, 80, 120

b) Sub-index ranges-100, 75, 50, 25, 1

c) Pairing; class 1(20-100), class 2 (30-75), class 3 (40-50), class 4 (25-80), and class 5 (1-120) are used. The paired data are the bases for sub-index development since they are the key points of the rating curves. If the actual measured value falls between two classes, the sub-index value is obtained using mathematical equations. Equation 1 is used when a parameter decreases the level of water quality with an increase in the parameter value.

$$S_i = S1-\left[(S1 - S2)\left(\frac{xi-x1}{x2-x1}\right)\right] \qquad (1)$$

Where; *si* is the i^{th} sub-index value; S1 and S2 are the sub-index values for the upper and lower classes, respectively; *xi* is the i^{th} parameter value; and *x1* and *x2* are values of permissible limits for the upper and lower class.

In a case where a parameter increases the level of water quality with an increase in parameter value, equation 2 is employed.

$$Si = S1-\left[(S1 - S2)\left(\frac{x1-xi}{x1-x2}\right)\right] \qquad (2)$$

In categorical scaling, the actual parameter values are converted into sub-index values by using constant values of either 0 or 1. The values 0 or 1 are assigned to parameter levels when concentrations exceed and fall below the recommended standard, respectively. These mathematical functions are important in this technique;

(a) si = 0 if *xi* > recommended standard

(b) si = 1 if *xi* < recommended standard

The measured parameter values and the recommended standards are compared from the sub-indices produced based on the established water quality criteria. The values range between 0 and 1, as shown in Equation 3;

$$Si = \frac{xi}{xmax} \tag{3}$$

where *si* is the i^{th} sub-index value; *xi* is the i^{th} actual parameter value (mg/l); and *xmax* is the maximum value of the recommended standard (mg/l)(Banda & Kumarasamy, 2020c; Stoner, 1978).

Statistical methods

Statistical analysis and historical parameter data are used to identify critical points for the generation of sub-index values. The metrics in question and their consistently measured average values and multiple quantiles are used. The individual sub-index values are multiplied by the parameter weightage values to generate the final index value. This technique has been used by Dunnett and Bhargava indices (Banda & Kumarasamy, 2020c).

3.3.2 Sub-index Development in the Adapted Indices

This step was bypassed by the eight WQIs that utilised the CCME-WQI. This procedure follows CCME, (2001), which developed a multivariate statistical procedure to combine the initial parameter values without sub-indices. In the original WAWQI, respondents were asked to create a rating curve for each of the nine parameters and sub-indices determined by expert judgment (Brown *et al.*, 1970). In the current review, all the articles that adapted WAWQI used water quality standards to develop sub-index values by comparing the measured parameters with existing recommended standards, both internationally and locally. The internationally adopted standards were WHO guidelines for drinking water (Boateng *et al.*, 2016), FAO guidelines for irrigation (Goher *et al.*, 2019), ANZECC guidelines for the conservation of aquatic areas (Sirunda *et al.*, 2022), WHO standards for swimming pools and similar environments (Ibanga *et al.*, 2020) among others. On the other hand, the local standards included but not limited to; SANS241-1:2015 (Mandindi *et al.*, 2022), LNCSM (Salem *et al.*, 2022), Ghana's WRC guidelines for domestic use and protection of aquatic life (Miyittah *et al.*, 2020) among others.

Going by equation 4 (Liou *et al.*, 2004), 61.8 % including (Githaiga *et al.*, 2021; Akoto *et al.*, 2021; Boateng *et al.*, 2016; Bon *et al.*, 2021; Belle *et al.*, 2021; Akakuru *et al.*, 2021), among others, assigned a quality rating scale (q_1) by dividing the concentration in each water sample with its corresponding standard following the suggested recommendations and multiplying the result by 100.

$$q_i = \left(\frac{Ci}{Si}\right) \text{x } 100 \quad (4)$$

where: *qi* is the quality rating, *Ci* is the concentration of each parameter in each water sample in mg/l, *Si* is the maximum permissible guideline limit for each parameter in mg/l.

On the other hand, 32.4 % of articles, including (Hagage *et al.*, 2022; Boah *et al.*, 2015; Bankole *et al.*, 2022; Idehen, 2016; Ibanga *et al.*, 2020; Ayogu *et al.*, 2020), considered using a function that multiplied the result by 100 and included the ideal WQP values in addition to the maximum allowable guidelines, as illustrated in equation 5.

$$\text{Qn} = 100 \left[\frac{vn-vo}{sn-vo}\right] \quad (5)$$

Where; *Vn* is the observed value of the nth parameter, *Vo* is the ideal value of the nth parameter in pure water. *Vo* = 0, except for pH = 7.0 and DO = 14.6 mg/I, *Sn* is the recommended standard value of the nth parameter.

Assignment of parameter weights

In essence, parameters are given weighted values based on their relative importance to the overall quality of the water (Sutadian *et al*., 2017). While some WQI models give each parameter the same weight and see them as equally important to water quality, most WQIs give each parameter an unequal weight while ensuring the sum of all weights equals 1. This weighting approach is appropriate because the overall impact of WQP shouldn't be greater than 100 % (Banda, 2015). The integrity of the final index score is negatively impacted and is regarded as dysfunctional if improperly conducted, giving a parameter more or less relevance than it deserves. Therefore, care should be taken when assigning unequal parameter weights (Uddin *et al*., 2021). This consideration ensures that the final index value reflects the water quality status. There are two approaches to establishing parameter weights: (i) the Delphi Method and (ii) the Analytic Hierarchy Process (AHP).

The Delphi Technique

This approach seeks professional judgment from significant players in the water quality field to weight parameters. They typically base their weightings on environmental relevance, the recommended guideline values, and the application to the particular water body (Uddin *et al*., 2021). In certain circumstances, some authors establish parameter weights based on existing values in the literature using a scale of 1-5 (Jagaba *et al*., 2020) or 1-4 to compare the environmental significance of different factors. To achieve relative weightings between 0 and 1 for the least influential and most influential parameters, respectively, all of the ratings are pooled, and their arithmetic mean values are determined by mathematical functions or compared to existing standards (Banda & Kumarasamy, 2020c; Jagaba *et al.*, 2020). This technique was employed in Horton and Brown indices.

Analytic Hierarchy Process

With this multidisciplinary technique, the decision-making process considers both quantitative and qualitative factors (Banda & Kumarasamy, 2020c). AHP uses pair-wise comparison principles in WQI development, where experts present their preferred option by contrasting many choices from a complex collection of factors (Banda & Kumarasamy, 2020c). Sutadian *et al.* (2017) have effectively used this technique, which enables the reliability check of the evaluations being made and reduces subjectivity in the decision-making process (Uddin *et al.*, 2021).

3.3.3 Assignment of parameter weights in the adapted indices

The parameters in the eight articles that used the CCME-WQI were not assigned weights. This pre-condition is consistent with the original CCME-WQI model, which assumes that all parameters have equal weights and does not call for weight values when predicting the final index score (Soumaila *et al.*, 2019). Brown and his associates applied the Delphi method to the original WAWQI to give parameter weight values. When the respondents' replies were compiled, the unit weight was summed up to 1, and they were asked to compare the weights of several parameters using a scale of 1 (highest) to 5 (lowest) (Brown *et al.*, 1970).

Articles that adapted the WAWQI used two approaches to assign weights to parameters; (1) assignment of parameter weighs by Delphi technique and fitting the values into equation 6 (Anim-Gyampo *et al.*, 2019; Njuguna *et al.*, 2020; Khmila *et al.*, 2021; Bankole *et al.*, 2022; Hagan *et al.*, 2022; Solihu

& Bilewu, 2022) and (2), assignment of parameter weights by Delphi technique and fitting values into equation 7 through the application of a value inverse of recommended guideline (Idehen, 2016; Ayogu *et al.*, 2020; Teshome, 2020; Akakuru *et al.*, 2021; Salem *et al.*, 2022;). Although parameter weights were assigned in accordance to (Brown *et al.*, 1970) with a scale of 1 (highest) to 5 (lowest), (Rabeiy, 2018) and (Wali *et al.*, 2022) employed a scale of 1 (highest) to 4 (lowest).

$$\text{Wi} = {}^{wi}\!/_{\sum_{i=1}^{n} wi} \quad (6)$$

Where: *Wi* is the unit weight, *wi* is the weight of each parameter and *n* is the number of parameters.

$$Wi = {}^{k}\!/_{Sn} \quad (7)$$

Where; *k* is a proportionality constant determined as;

$$k = \frac{1}{\sum_{i=i}^{n} \frac{1}{si}} \quad (8)$$

Where; *Si* is the standard permissible value for the i^{th} parameter.

Final computation of the WQI

This last step combines sub-indices and weighted factors from all metrics using various aggregation methods to get a unitless value representing the overall water quality status (Banda & Kumarasamy, 2020). The most often utilised are the multiplicative (geometric) and additive (arithmetic) functions (Terrado *et al.*, 2010). Continuous efforts have been made since the initial WQI to address the shortcomings of earlier aggregating functions. For example, moving from the weighted arithmetic average (Brown *et al.*, 1970; Horton, 1965) to the weighted geometric average (Brown et al., 1972) weighted product

(Bhargava, 1983), harmonic mean (Cude, 2001), minimum operator (Smith, 1990), and finally to, logarithmic-based functions (Icaga, 2007).

The overall WQI is directly impacted by the parameter values in any given aggregation approach. Regardless of how the parameters are weighted, indices produced with arithmetic average functions are most frequently affected by extreme values of the parameters (eclipsing) (Mophin-Kani & Murugesan, 2011). To offset this shortcoming, weighted geometric mean is suggested for aggregation, equation 9; (Bhargava 1985; Dinius 1987; Liou *et al.,* 2004). It has been reported to be more viable and unbiased in comparison to the weighted arithmetic mean (Landwehr *et al.*, 1974).

$$\text{WQI} = \prod_{i=1}^{n} Si^{wi} \qquad (9)$$

Where; WQI is the final WQI value, n is the number of parameters, Si is the sub-index value of the i^{th} parameter, wi is weight of the i^{th} parameter. Furthermore, in geometric average functions, the WQI tends to be zero if the value of one of the parameters is near zero. To offset this short coming, minimum operator function is suggested. The minimum operator function uses the lowest sub-index value as the final WQI value and does not consider weights of each parameter (Smith 1990; Swamee & Tyagi, 2000; Shah & Joshi, 2015).

$$\text{WQI} = \min(s_1, s_2, s_3, \ldots\ldots s_n) \qquad (10)$$

Weighted arithmetic and weighted geometric averages are thought to be outperformed by the unweighted harmonic square average. This function has been found to reduce the eclipsing effect while accounting for other indicators' impact by being more sensitive to the most degraded indicators (Walsh & Wheeler, 2012). Although the most recent method uses logarithmic functions, most researchers still use arithmetic or geometric aggregations.

Aggregation function for the adapted CCME-WQI

The original aggregation formula (CCME, 2001) was employed by the eight articles that adopted the CCME-WQI to get the overall index value, which is based on three parameters;

Factor 1 (F_1): Scope - assesses the extent of water quality guideline non-compliance during the period of interest.

$$F_1 = \left(\frac{Number\ of\ failed\ variables}{Total\ number\ of\ variables}\right) X\ 100 \qquad (11)$$

Where variables indicate those WQP with objectives that were tested during the period for the index calculation.

Factor 2 (F_2): Frequen**cy** represents the mean frequency and number of times the tested or observed value was out of acceptable limits or standards.

$$F_2 = \left(\frac{Number\ of\ failed\ tests}{Total\ number\ of\ variables}\right) X\ 100 \qquad (12)$$

Factor 3 (F_3): Amplitude - It represents the amount by which the failed test values do not meet their objectives and is calculated in three steps:

i) Calculation of Excursion

Excursion is the number of times an individual concentration is greater than (or less than, when the objective is a minimum) the objective.

-When the test value must not exceed the objective;

$$Excursion_i = \left(\frac{FailedTest\ value\ i}{Objective\ j}\right) - 1 \qquad (13)$$

-When the test value must not fall below the objective;

$$Excursion_i = \left(\frac{Objective\ j}{FailedTest\ value\ i}\right) - 1 \qquad (14)$$

ii) Calculation of Normalised Sum of Excursions

The normalised sum of excursions, *nse*, is the collective amount by which individual tests are out of compliance. This value is calculated by summing the excursions of individual tests from their objectives and dividing them by the total number of tests (both those meeting objectives and those not meeting objectives).

$$nse = \frac{\sum_{i=0}^{n} excursion\ i}{Number\ of\ tests} \quad (15)$$

iii) Calculation of F_3

F_3 is calculated by an asymptotic function that scales the normalized sum of the excursions from objectives to yield a range from 0 to 100.

$$F_3 = \left(\frac{nse}{0.01nse + 0.01}\right) \quad (16)$$

The WQI is then calculated as;

$$\text{WQI} = 100 - \left(\frac{\sqrt{F1^2 + F2^2 + F3^2}}{1.732}\right) \quad (17)$$

The factor of 1.732 arises because each of the three index factors can range as high as 100. Therefore, the vector length can reach;

$\sqrt{100^2 + 100^2 + 100^2} = \sqrt{30000} = 173.2$ as a maximum. A division using 1.732 reduces the vector length to 100 as a maximum.

Aggregation Function for the adapted WAWQI

All 34 articles in this model employed the WA aggregation function to compute the final index value. After weighting the parameters and sub-indices, two approaches were employed to calculate WQI.

i) 67.6% of the 34 articles, including (Boateng *et al.*, 2016; Abuzaid, 2018; Anim-Gyampo *et al.*, 2019; Berhe, 2020; Akoto *et al.*, 2021; Belle *et al.*, 2021; Githaiga *et al.*, 2021; Bankole *et al.*, 2022) employed an additive function to combine relative weight from expert opinion and water quality ratings from recommended guidelines in these two equations;

$$Sli = Wi \ x\ qi \quad (18)$$

$$\text{WQI} = \sum_{i=1}^{n} Sli \tag{19}$$

Where; *Sli* is the sub-index value of the i^{th} parameter, *Wi* is the relative weight of the i^{th} parameter, *n* is the number of parameters.

ii) The remaining 32.4 % (Ekere *et al.*, 2019; Ayogu *et al.*, 2020; Ibanga *et al.*, 2020; Umoh *et al.*, 2020; Hagage *et al.*, 2022) who incorporated the quality rating and relative weights from suggested standards employed an additive function;

$$\text{WQI} = \sum \frac{QnWn}{Wn} \tag{20}$$

Where Qn is the quality rating, *Wn* is the relative weight, summing up to unity.

Classification of WQIs and index scores

The assignment of final WQI values to classes or categories is known as classification or categorisation, and it can be done using one of two sets of categorisation scales : (i) an increasing scale, where the index value rises with the level of contamination, and, (ii) a decreasing scale, where the index value falls with the level of contamination. In both instances, the final objective is to express the water quality status by determining the degree of contamination (Banda & Kumarasamy, 2020c). Classifying WQI values should be based on the public's expectations for water quality, professional judgment, and the most up-to-date information (CCME, 2001). Typically, the index values range between 0 and 100. Further, the values are grouped into classes 1 through 5, depending on whether the author employs an increasing or decreasing scale. Tables 5 and 6 showcase the various WQI developed using increasing and decreasing scales.

Table 5: *WQI Classification using Increasing Scale*

	a. CCME-WQI		b. OWQI	
Class	**Rank**	**Index score**	**Rank**	**Index score**
Class 1	Excellent	95 - 100	Excellent	90 – 100
Class 2	Good	80 - 94	Good	85 – 89
Class 3	Fair	60 - 79	Fair	80 – 84
Class 4	Marginal	45 - 59	Poor	60 -79
Class 5	Poor	0 - 44	Very poor	0 – 59

Source: a (CCME, 2001; Lumb *et al.*, 2011)**;** b (Dinius, 1987; Dunnette, 1979)
Note: CCME-WQI (Canada); OWQI: Oregon WQI (Oregon)

Table 6: *WQI Classification using Decreasing Scale*

	a. TMWQI		b. RWQI	
Class	**Rank**	**Index score**	**Rank**	**Index score**
Class 1	Excellent	< 26	Excellent	<50
Class 2	Good	26 - 50	Good	50 – 100
Class 3	Medium	51 - 75	Poor	100 – 200
Class 4	Poor	76 - 100	Very poor	200 – 300
Class 5	Unsuitable	>100	Unsuitable	>100

Source: a (Tiwari & Mishra, 1985); b (Ramakrishnaiah *et al*., 2009).
Note: TMWQI; WQI proposed by Tiwari and Mishra (India), RWQI; Ramakrishaniah WQI (India).

3.3.4 Classification of water quality in the adapted WQIs

As shown in Table 5 part a, the original CCME-WQI used the CCME categorisation scheme described in (Lumb *et al.*, 2011) and Canadian Council of Ministers of the Environment, 2001). All eight articles that used the CCME-WQI followed the same trend. On the other hand, Brown and his colleagues' original WAWQI plan called for using colour schemes to categorise water quality across the state (Table 7).

Table 7: *WQI Classification using Colour Schemes*

Colour	Rank	Index value
Dark red	Very poor	0-10
Orange	Poor	*
Yellow	Medium/Average	50
Green	Good	**
Dark blue	Excellent	90-100

Source: (Kachroud, 2019; Lumb et al., 2011)
Notes: * and ** are not given, and this review assumes >10WQI<50 and >50WQI<90 for poor and good water quality, respectively.

All the 34 papers that applied the WAWQI used a decreasing scale to classify their subjects, ranging from 0 to 100 (Boah *et al.*, 2015; Abuzaid, 2018; Ayogu *et al.*, 2020; Ibanga *et al.*, 2020; Umoh *et al.*, 2020; Bankole *et al.*, 2022) in some cases, and 0 to 300 in others, (Anim-Gyampo *et al.*, 2019; Akoto *et al.*, 2021; Solihu & Bilewu, 2022; Wali *et al.*, 2022) while (Bankole *et al.*, 2022) and (Ayogu *et al.*, 2020) used 6 and 4 classes, respectively. Majority of reviewed articles (approx. 94 %) ranked the WQI values in 5 classes. Of the 34, 19 had their final index values classified (Table 8 part a) following (Sahu & Sikdar, 2008), and 6 (Table 8 part b) based on (Tyagi *et al.*, 2013).

Table 8: *Classification Scale for WQI Scores in the Adapted Indices*

	a.		b.	
Class	**Rank**	**Index score**	**Rank**	**Index score**
Class 1	<50	Excellent	0 – 25	Excellent
Class 2	50 -100	Good	26 – 50	Good
Class 3	100 -200	Poor	51 – 75	Poor
Class 4	200 - 300	Very poor	76 – 100	Very poor
Class 5	>300	Unsuitable for human consumption	>100	Unsuitable for human consumption

Source: a (Sahu & Sikdar, 2008) and b (Tyagi *et al.*, 2013).

3.3.5 Limitations identified in the classification scales of adapted indices in reviewed articles

In the Sahu & Sikdar (2008) classification (Table 8 part a), the final index score may fall within 2 classes. For instance, a score of 100 could either fall within class 2 ("good") or class 3 ("poor"), while 200 could fall within class 3 ("poor") or class 4 ("very poor"). Such a limitation has been reported elsewhere (Shah & Joshi, 2017). To solve this shortcoming, (Aljanabi *et al.*, 2021) modified the CCME-WQI classification scale by minimising the difference between classes to a decimal fraction. In the adapted indices, two articles (Rabeiy, 2018) and (Boateng *et al.*, 2016) introduced decimal fractions in their classification scheme; <50 ("excellent"), 50-100.1 ("good"), 100-200.1 ("poor"), 200-300.1 ("very poor"), and >300 ("unsuitable for human consumption"). Nevertheless, such a classification scheme still faces the challenge of some scores falling within 2 classes, like 100 in class 2 ("good") and class 3 ("poor,") as well as a score of 200 falling between class 3 ("good") and class 4 ("very poor"). Also, an overlap exists in classes whereby class 2 ("50-100.1") suggests good water quality compared to class 3 ("100-200. 1") of poor quality yet 100.1 is less than 200.1.

The classification by Tyagi and his colleagues (Table 8 part b) makes it impossible for some final index scores to be accommodated. With regard to this, there is no provision for index scores between 25-26, 50-51 and 75-76, unless the index score is rounded off to the nearest whole number. Other works with similar limitations in class assigning index values have been reported (Banda & Kumarasamy, 2020c; Kannel *et al.*, 2007). Finally, there is a possibility of lack of representation for index score values falling below 50 in the scheme used by

Akoto *et al.* (2021) which runs from 50 to > 300, i.e., 50 ("excellent") 50-100 ("good") 101-200 ("poor") 201-300 ("very poor") >300 ("unsuitable for human consumption"). It can be noted that unless performed carefully, classification schemes will certainly fail to accommodate all the achievable index scores. Despite the dynamic nature of aquatic ecosystems due to the continuous influence by allochthonous and autochthonous factors, a final index value must be attained and categorised. It is, therefore, imperative to introduce logical linguistic descriptions like less than, equal to and greater than to ensure inclusivity of all index scores, as demonstrated in Table 9.

Table 9: *Classification of WQI Scores using Logical Linguistic Descriptions*

	a.		b.	
Class	**Rank**	**Index score**	**Rank**	**Index score**
Class 1	91≤ Index ≤100	Good	< 2.8	Excellent
Class 2	61≤ Index <91	Acceptable	2.3 - 2.8	Good
Class 3	31≤ Index <61	Regular	< 2.3	Poor
Class 4	16≤ Index <31	Bad		
Class 5	0 ≤ Index <16	Very bad		

Source: a (Abrahão *et al*., 2007) adapted from the Bascaron method (1979) ; b (Rubio-Arias *et al*., 2012).

3.3.6 Potential solutions to uncertainties associated with adapted indices

The concerns center on how differences in a WQI's developmental stages might impact the ultimate index score. These may arise at any of the WQI developmental steps, including parameter selection, sub-indexing and weighting (Sutadian *et al.,* 2017), and the final aggregation function (Smith, 1990).

Parameter selection uncertainties

There are no set criteria or procedures for choosing WQPs to be included in the various WQI models. The researcher could be directed toward the best WQPs to choose by aspects like data accessibility, environmental relevance (Debels *et al.*, 2005), and use or purpose of the water body (Kannel *et al.*, 2007). It is important to emphasise that lack of data availability is a major cause for concern when selecting parameters, especially in developing nations (Uddin *et al.*, 2021). This is mostly because water quality monitoring programs require much labour and have significant analytical costs, which likely explains why most authors in the examined articles only used the fundamental criteria to assess water quality. Additionally, the inability of researchers to access comprehensive data on water quality is hampered by the lack of contemporary analytical laboratory facilities due to insufficient capital, such as financial assistance and competent human resources (Debels *et al.*, 2005).

Approximately 80 % of the articles in the current review focused on drinking water or water for human use, while the minority concentrated on recreation, irrigation, protecting aquatic life, and health risk issues. As can be deduced, practically all of the authors that focused on drinking water largely prioritised heavy metals in conjunction with other ancillary parameters, though the list was not exhaustive. Heavy metals, especially in gold mining areas, are a source of environmental contamination and threaten human health. For instance, Pb, Hg, Cd, and As are carcinogenic if they exceed the maximum tolerable upper intake levels (WHO, 2011). Additionally, Cr, Cu, Zn, and Ni threaten all aquatic inhabitants by disrupting food chains via bioaccumulation. Some heavy metals, like Ni nanoparticles, damage liver cells (Birniwa *et al.*, 2021). Furthermore, chronic exposure to Pb may result in mental retardation, congenital disabilities, psychosis, autism, allergies, dyslexia, weight loss, hyperactivity, paralysis, muscular weakness, brain damage, and kidney damage and may even cause death (Fang *et al.*, 2016).

On the other hand, Fe in water may chemically bond with free hydrogen radicals, hence attacking DNA cells, leading to mutations and malignant transformations which cause a myriad of diseases (Jaishankar *et al.*, 2014). In addition, long-term ingestion of water with a high concentration of Zn has been found to cause the death of human brain cells, trauma, and prostate cancer (Nriagu, 2011). Sometimes, water samples for drinking may comply with the state or international set standards, especially for physical parameters like colour, odour, and turbidity. However, harmful hazardous, and hardly detectable compounds may also be present due to the universal solvent nature of water. This presents a challenge in analysing each of the chemicals present in water (Abbasi & Abbasi, 2011). Even though it is key to prioritise WQPs, it is important to include data on physical, chemical, biological, and hazardous factors to thoroughly analyze and portray the ideal water quality conditions (Ongley & Booty, 2009). Notably, none of the articles under consideration used radioactive or hazardous components to assess the water quality.

Additionally, four WQPs were used in the modified indices to evaluate the swimming pool water quality in hotels in Nigeria (Ibanga *et al.*, 2020), while at the other extreme, 24 WQPs were chosen to evaluate the appropriateness of surface water for agriculture in Egypt (Abuzaid, 2018). It should be emphasised that since water quality varies on a wide range of natural and anthropogenic factors, too few parameters may not provide a good picture of the final WQI.

Likewise, it is doubtful that all the data from the 24 collected parameters would be readily available in addition to being data of high quality. Although the original CCME-WQI and West Java WQI, which used four and 26 WQPs, respectively, may have been provided as inspiration for the author(s). Considerable care must be taken to ensure that the parameters chosen are just enough, neither too few nor too many (Banda & Kumarasamy, 2020c), and based on the study at hand and end-use objectives (Bhargava, 1983).

Given resource constraints, it was important to prioritise parameters that directly impacted water quality. These included parameters related to eutrophication, dissolved chemicals, dissolved oxygen levels, physical qualities, and health issues (Dunnette, 1979). The relevance of prioritised parameters for whatsoever societal needs cannot be overemphasised. For example, DO concentration is normally used to indicate water quality to the extent that high DO concentrations indicate good water quality. Low DO concentration could mean reduced organismal growth, disruption of life cycles, migration to avoid poor conditions, and even death of benthic organisms and fish (reviewed in Vaquer-Sunyer and Duarte 2008; cited in (Sheldon & Alber, 2011).

On the other hand, inorganic nutrients (mainly nitrogen and phosphorus) may stimulate the growth of algal blooms causing eutrophication. In addition, nitrites, reduced forms of nitrates, have been proven to cause blue baby syndrome (Methemoglobinemia) in infants after long-term ingestion. Physical attributes like suspended solids and turbidity, odour and colour affect the suitability of water for some domestic uses like washing as well as drinking.

High turbidity may also clog gills of some benthic macroinvertebrates, causing death.

For microbiological water quality, faecal coliforms (FC) have been conventionally used as indicator organisms for faecal contamination, and some WQI models have been incorporating them as part of the WQPs, e.g., (Odonkor & Ampofo, 2013). However, WHO commonly accepts *E. coli* as a better indicator of faecal and microbiological water contamination and recommends it for use instead of FC. This is the case where water for drinking purposes is being assessed (Ashbolt, 2001). In the articles reviewed, where more than three-quarters of articles focused on drinking water, microbiological constituents (TC, FC, TF, and *E. coli*) accounted for less than 10 %, while *E. coli* alone accounted for only 4.8 % of the WQPs used.

Nevertheless, bio-assessments using aquatic organisms such as plankton, macroinvertebrates, and fish have proved to be a fair reflection of the current water quality status and the overall ecosystem health. This aspect was overlooked in all the reviewed articles. Biological components of water bodies, especially benthic macroinvertebrates, phytoplankton, and microbiota, are therefore highly recommended for integration with physicochemical parameters for adapted WQIs since they can indicate the future direction of the overall aquatic ecosystem health.

Eclipsing, Ambiguity and Rigidity

In additive/arithmetic models, eclipsing—where the total WQI conceals the underlying nature of water quality—is frequent (Swamee & Tyagi, 2000). Eclipsing might result from faulty sub-indexing, weighting, or an inadequate aggregation procedure (Smith, 1990). Ambiguity is a situation where the overall index shows that the water quality is worse than expected based on the sub-index values for all WQPs. Due to how weights are applied and because these indices base their categorisation on the parameter with the greatest impairment, ambiguity is mostly observed in weighted indices (Swamee & Tyagi, 2000). On the other hand, rigidity is the condition in which an index is not adaptable enough to consider new or alternative criteria (Swamee & Tyagi, 2007). It frequently happens when an impairment arises in a parameter or parameters that the index does not consider or when an index is applied to a circumstance where the concerns differ from those for which it was designed (Swamee & Tyagi, 2007).

There was a high likelihood of articles that adapted the WAWQI to develop the abnormalities mentioned above. This was owed to the fact that WAWQI is an additive, weighted model, and was also employed in a field with distinct concerns from the ones from which it was derived. Additionally, the WA-adapted model was adjusted throughout the process of adapting from an original index with a set number of parameters, changing both the type and the number of parameters. On the other hand, articles that adapted CCME-WQI were less likely to experience the mentioned abnormalities as the original index is not explicit about the parameters to be chosen and gives the flexibility to accommodate more or alternative factors. Moreover, CCME-WQI does not use either parameter weighting or standard arithmetic computation. Therefore, CCME-WQI uses a variety of intricate aggregating procedures, which may cause the final index value uncertain (Sutadian *et al.*, 2017).

Either weighted or unweighted multiplicative models can be used as a potential remedy for eclipsing, or new methods of calculating sub-index values can be used (Swamee & Tyagi, 2007). If a crucial parameter was concealed in the overall WQI, the lowest scoring parameter buried within the overall high WQI value might also be reported along with the WQI. To lessen eclipsing abnormalities, Smith (1990) also suggested using the minimum operator aggregation function. Finally, in parameter selection, applying multivariate statistical tools like Principal Component Analysis, Cluster Analysis, and Discriminant Analysis could significantly minimise ambiguity and eclipsing and establish new sub-index weights. Location-specific issues regarding rigidity can be reflected through the adjustment of sub-index parameters. To increase uniformity among the WQI models, great care must also be taken to employ local water quality parameters that are in sync with international guidelines (Swamee & Tyagi, 2000). The Analytical Hierarchical Process technique is advised for weight assignment because it reduces abnormalities brought about by improper parameter weighting by determining parameter significance (Uddin *et al.*, 2021).

3.3.7 Applied analysis and comparison of adapted WQIs and new models

Since the review's objective was to analyse the development stages for WQIs that were modified versions of already existing WQIs, techniques that created a new model were among those that were disqualified under the exclusion and inclusion criteria. This section offers a summary of four novel approach WQIs from an African perspective and compares their similarities and differences to the adapted indices.

WQI based on Data Envelopment Analysis (DEA): Application to Algerian dams (Soltani et al., 2021)

The methodology was applied to a sample of 47 dams situated in hydrographic basin areas in Algeria's Tellian region, specified by ten physicochemical parameters. The input variables, dubbed "optimistic closeness values," were skillfully constructed from the hydrochemical parameter values before applying a DEA model.

Because of its objective data-driven nature, DEA is one strategy that avoids using a priori elicited weights (Al-Mezeini *et al.*, 2020). The power of DEA has been demonstrated for performance evaluation of Decision-Making Units (DMUs), which employ many inputs to produce numerous outputs, permitting explicit segmentation of these DMUs into efficient and inefficient units (Oukil & Al-Zidi, 2018). WQIs, benchmark frequencies, and slack values from the DEA model are used to determine the bounds of the quality ranges, rank the dams, and design a priority scale for treating the hydrological parameters. When analysing the performance of a DMU, the DEA model operates on the fundamental concept of using fewer inputs to generate more outputs.

As a result, when viewed implicitly as an element of a production process, every input has to adhere to the maxim that "less is better"(Cook *et al*., 2014). The WHO (WHO, 2011) and the ANRH (Soltani *et al.*, 2021) drinking water standards were used in this article to classify each hydrochemical parameter into one of four quality levels: Excellent, Acceptable, Poor, and Unsuitable. The Acceptable, Poor, and Unsuitable ranges, on the other hand, required basic, refined, and highly advanced treatments, respectively, to achieve the necessary water quality, while the Excellent interval expressed the satisfactory quality of the water that could be utilized without any special requirements.

Sturges's rule (Sturges, 1926), was used to determine the best number of intervals to classify data samples of size K, and in this case,

$$I = 1 + 3.322\log_{10}(K) \tag{21}$$

Where K= number of dams (47), and the number of adequate classes for the samples of WQIs would be I= 7.

The DEA-WQI model is not only a risk-ranking tool, but it also has great potential to support vital decisions on the water treatment of vulnerable dams.

UWQI: South African catchments (Banda & Kumarasamy, 2020b)

The information was gathered from six sampling stations spread across four distinct catchments that fall under the purview of the Pongola-Mtamvuna Water Management Area, located in the South African province of KwaZulu-Natal. The four watershed regions are the Umgeni, Umdloti, Nungwane, and Umzinto/uMuziwezinto River catchments. The four catchments employed were sufficient to determine the model's functionality, and the procedure was a step toward the ultimate objective of testing the model against the majority, if not all, of the catchment areas in South Africa. This model employed the four classical steps involved in the development of conventional WQIs.

Parameter selection- A fixed set of 10 parameters were established using expert opinions.

Weight coefficients- Ratings of parameter significance were provided based on data gathered from the existing literature and via Delphi questionnaires. The preliminary ratings from the two methodologies were then combined to create parameter significance ratings (bi). Equation 20 was used to calculate the relative weight coefficients (wi), which were directly proportional to the significant ratings and obtained by dividing the parameter significance rating value (bi) by the summation of all ratings (bi);

$$\text{wi} = \frac{bi}{\sum_{i=1}^{n}(bi)} \qquad (22)$$

Where *bi* is the assigned significance rating of the ith water parameter; wi is the final weight coefficient for the ith water parameter; *n* is the total number of the rated water quality parameters.

Formation of sub-indices- The allowable concentration limits were used to graphically establish the fixed key points of the rating curves. The plotted points were converged using straight lines to create a sequence of linear graphs, which were then transformed into linear sub-index functions. The procedure included consultation with the Target Water Quality Ranges (TWQRs), laid out by DWAF (1996).

Aggregation formula- The final UWQI, an improved version of the weighted sum approach, was developed by modifying and aligning the model with local conditions using scenario-based analysis. The model equation obtains the overall water quality status as a unitless number ranging from 0 to 100, as stated in the equations below by integrating sub-index values of selected parameters regarding the defined weights.

$$\text{WQI} = \frac{1}{100}\left[\sum_{i=0}^{n} siwi\right]^{2} \quad (23) \longrightarrow \frac{2}{3}\left[\sum_{i=0}^{n} siwi\right]^{1.0880563} \quad (24)$$

(Modified weighted sum model) (Final universal water quality model)

Classification of WQI scores in UWQI

An increasing scale index serves as the basis for the classification mechanism. The UWQI model produces WQI values ranging from 0 to 100. The WQI scores are thus divided into classes ranging from one to five, with "Class 1" designating water of the highest degree of purity with a maximum possible score of one hundred and, conversely, "Class 5" designating water quality of the lowest degree with index scores close to or equal to zero. The classification involves the use of appropriate mathematical operations with logical linguistic descriptors, such as, but not limited to, "greater than," "less than," and "equal to," to evaluate WQI scores that were assigned to each category to fill in gaps in some of the existing classification scales. This classification has also been used in literature by Banda & Kumarasamy (2020c). Due to the fixed number of parameters, this model can be used in various catchments without affecting its structure or operation. Stakeholders may be able to compare the water quality of various sites and establish a more impartial management prioritisation. Furthermore, expert opinion has the benefit of promoting the acceptability of the model because most of the experts involved are also the model's intended end users. As a result, their participation in the development of the UWQI may eventually bring about acceptance through a sense of ownership.

The Surrogate WQI based on multivariate statistical analysis: South African watersheds (Banda & Kumarasamy, 2020a)

The parameters for water quality in this model were determined via a two-phase testing process that encompassed (i) the Delphi method used for the UWQI, where twenty-one parameters were reduced to thirteen variables, and (ii) further reducing the parameters to four proxy variables using statistical analysis, including electrical conductivity, chlorophyll-a, pH, and turbidity. Pattern recognition and elucidating the underlying dataset's structure were both accomplished during this process using PCA (Bouza-Deaño *et al.*, 2008).

The most significant parameters that can be employed as proxy variables were identified, and it also offered important statistical data on the intercorrelated parameters. Furthermore, Hierarchical Cluster Analysis (HCA) was used to demonstrate the intuitive correlations between various water quality data. In the process, it produced a dendrogram illustrating how the clusters were arranged and how close the various parameters were to one (Zhao *et al.*, 2012). The resulting regression equation and coefficients represent the surrogate WQI model. The surrogate model outline displays the structure of the surrogate WQI with the four-proxy water quality input variables x_1, x_2, x_3, and x_4; their corresponding coefficients b_1 to b_4, intercept term b_0, error term for the regression model symbolized as ε, and the regression model function

$$f(x) = b_0 + b_1x_1 + b_2x_2 + \ldots + b_4x_4 + \varepsilon \qquad (25)$$

The beneficial aspect of this model is that, regardless of the absence of the entire data set, optimally chosen parameters could still reflect water quality (Zhao *et al.*, 2012). Likewise, it conforms with the requirements of the study and offers an essential quick guide equal to the outcome of a high-fidelity model.

The Hounsinou scale

Its development and use to determine the overall quality of groundwater used for drinking and bathing in the municipality of Abomey-Calavi in Benin (Hounsinou, 2021). Twenty-three physicochemical parameters and three microbiological parameters were used to evaluate the overall water quality of 68 wells in the municipality of Abomey-Calavi in Benin using the Hounsinou scale. The Hounsinou scale, which is novel and superior to the conventional WQIs, independently indicates the chemical and microbiological properties of water. The Hounsinou scale combines the CWQI scale and the MWQI scale to provide a final chemical and microbiological contamination assessment.

Weighted arithmetic sums of the values of the physicochemical and microbiological parameters and of the WHO standards are used to calculate the CWQI and the MWQI. CWQI computation involves several steps: parameter selection, determination of the ideal values of the parameters, development of sub-indices, assignment of weights, and aggregation of sub-indices to produce an overall index expressed as;

$$\text{CWQI}\sum QnWn \,/\, \sum Wn \qquad (26)$$

Akoteyon (2013) provides inspiration for the classification scheme used in this model whereby CWQI ≤ 50 means water is very excellent to >500, water is very unsuitable for drinking. As for the MWQI, the model is based on the contents per mL of water of TC, FC, and intestinal enterococci (IE). The final index is expressed as;

$$MWQI = 1/3\ (10/9\ TC + 20\ FC + 1000\ EI) \quad (27)$$

The classification scale in MWQI ranges from 'Absence of any germ for excellent water' to 'Presence in water of pathogen germs with or without TC, FC or EI,' meaning the consumption of this water exposes users to water-born illness within a short time. This new scale is superior to the water quality index that has been used to date because it is more accurate, provides the public with more comprehensive information on water quality, and is constantly applicable across the globe. International institutions can use this new scale, which combines the CWQI and MWQI scales to provide a total rating of chemical and microbial contamination, to rate and monitor the quality of water resources in all countries, and to evaluate the efforts made by those countries to safeguard and clean up their water resources. A summary of the comparisons between the new models and adapted WQIs is shown in Table 10.

Table 10: *Comparison Table on Adapted WQIs and New Approach Models*

Metrics of comparison	Adapted WQIs	The DEA Model	UWQI	Surrogate WQI	Hounsinou scale
1. Parameter selection	Researchers' discretion	Value observation	Expert opinion	Delphi technique multivariate statistics	Researchers' discretion
2. No. of parameters	average: 14	10	13	4	27
3. Type of parameters	Physical, chemical, microbiological	Physical, chemical	Physical, chemical	Physical, chemical	Physical, chemical, microbiological
4. Index development steps	Parameter selection, establishment of parameter weights, formation of sub-indices and final aggregation function	Creation of input variables, classification to obtain optimistic closeness values, Banker Charnes Cooper, (BCC) model with a single output, WQI	Parameter selection, establishment of weight coefficients, formation of sub-indices and aggregation formula	Delphi method, PCA, Hierarchical Cluster Analysis (HCA), Multivariate Regression Analysis (MRA), Regression model	Parameter selection, establishment of parameter weights, formation of sub-indices and final aggregation function
5. Index aggregation function	Weighted Arithmetic function $WQI = \sum \frac{QnWn}{Wn}$	$WQI = E^{*}_{dd}$	Modified weighted sum to to Final UWQI function WQI= $\frac{1}{100}[\sum_{i=0}^{n} siwi]^{2}$ to $\frac{2}{3}[\sum_{i=0}^{n} siwi]^{1.0880563}$	Surrogate WQI= $f(x) = b_0 + b_1x_1 + b_2x_2 + \ldots + b_4x_4 + \varepsilon$	$CWQI = \sum \frac{QnWn}{Wn}$ MWQI = 1/ 3 (10/ 9 TC + 20 FC + 1000 EI)

Table 10, *continued*

6. Water quality standards	Sub-index development, WHO (2011), FAO (1994), SANS (2015)	Classification of each hydrochemical parameter into quality categories, WHO (2017), Union (2014, ANRH (2019)	Sub-index development, DWAF (1996)	-	Sub-index development, WHO,2006
7. Index categorisation scheme	Adapted CCMEWQI-increasing categorisation scale with 5 classes Adapted WAWQI-decreasing scale with 5 classes	Sturges's rule $I = 1 + 3.322\log_{10}(K)$	Increasing categorisation scale with 5 classes including logical linguistic descriptors	Increasing categorisation scale with 5 classes	CWQI-decreasing categorisation scale with 7 classes MWQI-decreasing categorisation scale with 8 classes

The table provides a comparison of similarities and differences between the adapted WQIs and the new models in the African perspective

3.4 Future Perspectives

Given the historical adaptation and contextualisation of WQIs for water resource management in Africa, the continent is well-aligned to develop its region-specific (North, Western, Central, Eastern, and Southern) integrated water quality monitoring indices. We believe there is a plethora of scientific data on water science research and emerging technologies in the continent from which the requisite WQPs can be obtained, harmonised, and incorporated within the index development frameworks for specific regions. An additional advantage in WQI development is facilitating access to scientific data repositories and standardisation of water quality monitoring and data processing protocols across these regions. Such a transition towards customisation of WQIs will improve the accuracy and predictive power of Africa-based models while increasing confidence in the interpretation of water quality assessment and implementation of water resource management interventions.

3.5 Conclusions

Effective water quality monitoring and management in Africa is hampered by a lack of "indigenous" or region-specific WQIs due to the long-term trend in adapting and adopting WQIs from outside the continent. Ten-year trends in adapting WQIs for water quality monitoring in Africa to examine performance and potential for water quality monitoring in the continent were reviewed. The most commonly adapted indices for water quality monitoring in Africa were WAWQI and CCME-WQI, which exhibit a general bias towards physical and chemical parameters over biological metrics. In addition, these indices tend to suffer from abnormalities such as ambiguity, eclipsing, and rigidity, which limits their application potential. There is a need to integrate physicochemical, biological, and hazard indicators in adapting or developing WQI for the African context to address the wide spectrum of WQP requirements. Non-subjective statistical approaches could further provide uniformity in WQI model development. Nevertheless, the potential for developing Africa-derived WQIs that provide region-specific water quality status of the region’s aquatic ecosystems is unlimited.

CHAPTER FOUR

BENTHIC MACROINVERTEBRATES AS INDICATORS OF WATER QUALITY: A CASE STUDY OF ESTUARINE ECOSYSTEMS ALONG THE COAST OF GHANA

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Published: Elsevier-*Heliyon* (Heliyon 2024, 10(7); https://doi.org/10.1016/j.heliyon. 2024.e28018).

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P.K.M- Supervision, Review and Editing (**Principal Supervisor**)

N.K.A- Supervision, Review and Editing (**Co-Supervisor**)

C.A.F- Data Analysis, Review and Editing (**Co-Author**)

M.F.A.A- Identification of benthic macroinvertebrates, Review and Editing (**Co-Author**)

Abstract

Increasing human activities in coastal areas of Ghana have led to the degradation of many surface waterbodies, with significant consequences for the ecosystems in the affected areas. Thus, this degradation extremely affects the health of ecosystems and disrupts the essential services they provide. The present study explored the use of benthic macroinvertebrates as an indicator of estuarine degradation along the coast of Ghana. Water and sediment samples were collected bimonthly from Ankobra, Kakum, Whin and Volta Estuaries for physicochemical parameters, nutrients and benthic macroinvertebrates. The findings revealed the dominance of pollution-tolerant taxa such as *Capitella* sp., *Nereis* sp., *Heteromastus* sp., *Tubifex* sp., *Cossura* sp. and *Chironomous* sp. in Kakum and Whin Estuaries while pollution-sensitive taxa such as *Scoloplos* sp., *Eurydice* sp., *Lumbriconereis* sp. and *Pachymelania* sp. in the Volta Estuary. Moreover, a salinity indicator taxon (*Penaeus* sp.) dominated the Ankobra Estuary. Although Kakum and Whin Estuaries were dominated by a wide range of pollution tolerant taxa, Pearson correlation analysis revealed weak and moderate correlations (both positive and negative) in both estuaries, suggesting moderate levels of organic pollution. Dissolved oxygen, temperature, salinity, orthophosphate, nitrates, ammonium, electrical conductivity, turbidity, and chemical oxygen demand were the most significant parameters that complemented the use of benthic macroinvertebrates as indicators of environmental quality in the studied estuaries. Despite the moderate pollution levels, Kakum was the most diverse of the four estuaries. The study ranked these estuaries based on ecological stability using biological indices, suggesting Kakum Estuary as ecologically healthier than Whin, Volta, and Ankobra

Estuaries, in that order. The results highlighted the need to protect estuarine ecosystem health from further degradation with anthropogenic sources of contaminants. The findings further emphasised the need to integrate data obtained from benthic macroinvertebrates and physicochemical parameters that indicate the status of water quality into a water quality monitoring model for holistic assessment of estuarine ecosystem health in Ghana.

Keywords: Estuaries, physicochemical parameters, benthic macroinvertebrates estuarine ecosystem health, pollution tolerant species, pollution sensitive species.

4.1 Introduction

Estuaries are semi-enclosed areas of water that are openly connected to the ocean and have a combination of freshwater and saltwater features (Potter *et al*., 2010; Ujjania & Dubey, 2015). These ecosystems, being among the world's most productive, play a crucial role in supporting diverse life forms, including fish, shellfish, migratory birds, benthic organisms, and aquatic plants (Barbier *et al*., 2011; Day *et al*., 2012; Meire *et al*., 2005; Thrush *et al*., 2013). The distribution, and abundance of these life forms are influenced by biophysical forces such as tides, waves, and temperature (Ducrotoy *et al*., 2019). Estuaries provide essential ecosystem services with provisioning, supporting, regulatory, and cultural functions for the environment and humans, resulting in a high dependence on them (Van Niekerk *et al*., 2013; Wiethüchter, 2008).

The health of estuarine ecosystems is evaluated through various indicators and parameters reflecting their overall condition (Borja *et al*., 2013; Harwell *et al*., 2019; Lu *et al*., 2015; Vugteveen *et al*., 2006). Biological monitoring, specifically biomonitoring, employs bioindicators, such as benthic macroinvertebrates, to assess ecosystem conditions based on organisms' reactions to environmental changes (Lavoie & Campeau, 2010; Oertel & Salánki, 2003; Parmar *et al*., 2016; Tampo *et al.*, 2021). High numerical abundance, high sensitivity to environmental stressors, wide distribution, low mobility, and a high capacity for measurement and standardisation are all desirable qualities for benthic macroinvertebrates as bioindicators (Bressler *et al.*, 2006; Gresens *et al.*, 2010; Reynoldson & Metcalfe-Smith, 1992). Benthic macroinvertebrates, sensitive to environmental changes, serve as valuable tools for monitoring aquatic ecosystem health, being either pollution-sensitive or pollution-tolerant (Karmakar *et al*., 2022; Nerbonne & Vondracek, 2001).

Pollution-sensitive species, like stoneflies, mayflies, caddisflies, flatworms, and leeches, are highly responsive to environmental pollution and are used as bioindicators to assess ecosystem health (Ferraro *et al.*, 1991; Karmakar *et al*., 2022; Nerbonne & Vondracek, 2001; Nunkumar, 2002; Pinto *et al.*, 2009; Van Dolah *et al.*, 1999). Their absence or scarcity indicates pollution or environmental degradation (Ferraro *et al*., 1991; Pinto *et al.*, 2009; Van Dolah *et al.*, 1999).

On the other hand, pollution-tolerant species, including midge larvae, oligochaeta, scuds, copepods, and snails, thrive in polluted or disturbed aquatic ecosystems (Barrilli *et al.*, 2021; Gordon, 2000; Lamptey & Armah, 2008; Zhang *et al.*, 2019). While previous studies in Ghana have explored marine benthic macroinvertebrates, more data is needed to monitor coastal water and regulate anthropogenic activities degrading surface waterbodies (Aggrey-Fynn *et al.*, 2011; Armah *et al.*, 2012; Dzakpasu, 2019; Dzakpasu *et al.*, 2015;

Gordon, 2000; Lamptey & Armah, 2008; Okyere & Nortey, 2018). Table 11 lists other coastal waterbodies in the country that have previously been assessed.

Table 11: *Some Coastal Waterbodies Previously Assessed for Environmental Quality in Ghana*

Waterbodies	**Matrices assessed**	**Reference**
Pra Estuary	Water, sediment, benthic macroinvertebrates	(Dzakpasu, 2019; Faseyi, *et al.*, 2022a; Faseyi *et al.*, 2022b; Klubi *et al.*, 2018; Okyere & Nortey, 2018)
Nyan Estuary	Water, sediment, benthic macroinvertebrates, fish	(Dzakpasu *et al.*, 2015; Nortey *et al.*, 2016)
Muni Lagoon	Water, fish	(Ansa-Asare *et al.*, 2009; Okyere *et al.*, 2023)
Whin Estuary	Water, sediment, fish	(Agblemanyo, 2021b; Chuku et al., 2023; Nortey *et al.*, 2016; Sowah, 2019)
Chemu Lagoon	Water	(Okyere *et al.*, 2023)
Benya Lagoon	Water, sediment, benthic macroinvertebrates, fish	(Armah *et al.*, 2012; Dzakpasu, 2019; Vowotor *et al.*, 2014)
Narkwa Lagoon	Water, bivalves	(Ansa-Asare *et al.*, 2009; Chuku *et al.*, 2023; Dodoo *et al.*, 2013; Essumang, 2010; Sowah, 2019)
Amisa Lagoon	Sediment	(Mahu *et al.*, 2016)
Fosu Lagoon	Water, sediment, benthic macroinvertebrates, fish	(Armah, Ason, *et al.*, 2012; Assiam, 2020; Dankwa *et al.*, 2016)

The degradation of surface water bodies is caused by human activities such as mining, illegal fishing, improper waste disposal, open defecation, and the use of harmful chemicals in farming (Faseyi *et al.*, 2023).

This study therefore provides further information on the abundance, composition, and diversity of benthic macroinvertebrates, using the data to assess the condition of four notable Ghanaian estuarine ecosystems (i.e., Ankobra, Kakum, Volta and Whin), while examining their relationship with environmental factors prevailing in the estuaries. Additionally, the study also hypothesised that anthropogenic impacts on environmental factors significantly influenced benthic macroinvertebrates' composition and diversity.

4.2 Materials and Methods

4.2.1 Study area

The study was carried out along the coast of Ghana, which is mainly a high energy coast about 540 km stretching from Aflao (Togo border) to the La Cote d'Ivoire border. Ghana's coastline is divided into three based on geomorphologic characteristics: West, Central and East coasts (Figure 4.1) (Boateng, 2012). The West coast is generally a low energy coast, covering a distance of abou 95 km of shoreline from the estuary of the Ankobra River to the border with La Cote D'Ivoire. It is characterised by flat and wide beaches backed by coastal lagoons. The Central Coast is the most developed part of Ghana's coastlines, extending for approximately 296 km from the west of Prampram to Cape Three Points (the south most point of Ghana). It is a medium energy coast endowed with bays comprising of sand bars, rocky headlands and spits that encircle coastal lagoons. The East coast is a high-energy coast with wave heights often exceeding 1 m in the surf zone (Ly 1980), and it extends approximately 149 km from Aflao (Togo Border) in the East to the west of Prampram. The East coast is made up of medium to coarse sand, and its elevation is about 2 meters above sea level. The shoreline is mainly sandy, equipped with spits and barrier lagoons. For the purpose of this study, three estuaries; Ankobra, Kakum and Volta, from the West, Central and East coasts were selected to represent the various sections of the coastline. Additionally, Whin Estuary was also selected to serve as a reference condition following earlier research designating the estuary as relatively pristine by Atindana *et al*. (2019) and CRC/FoN (2010).

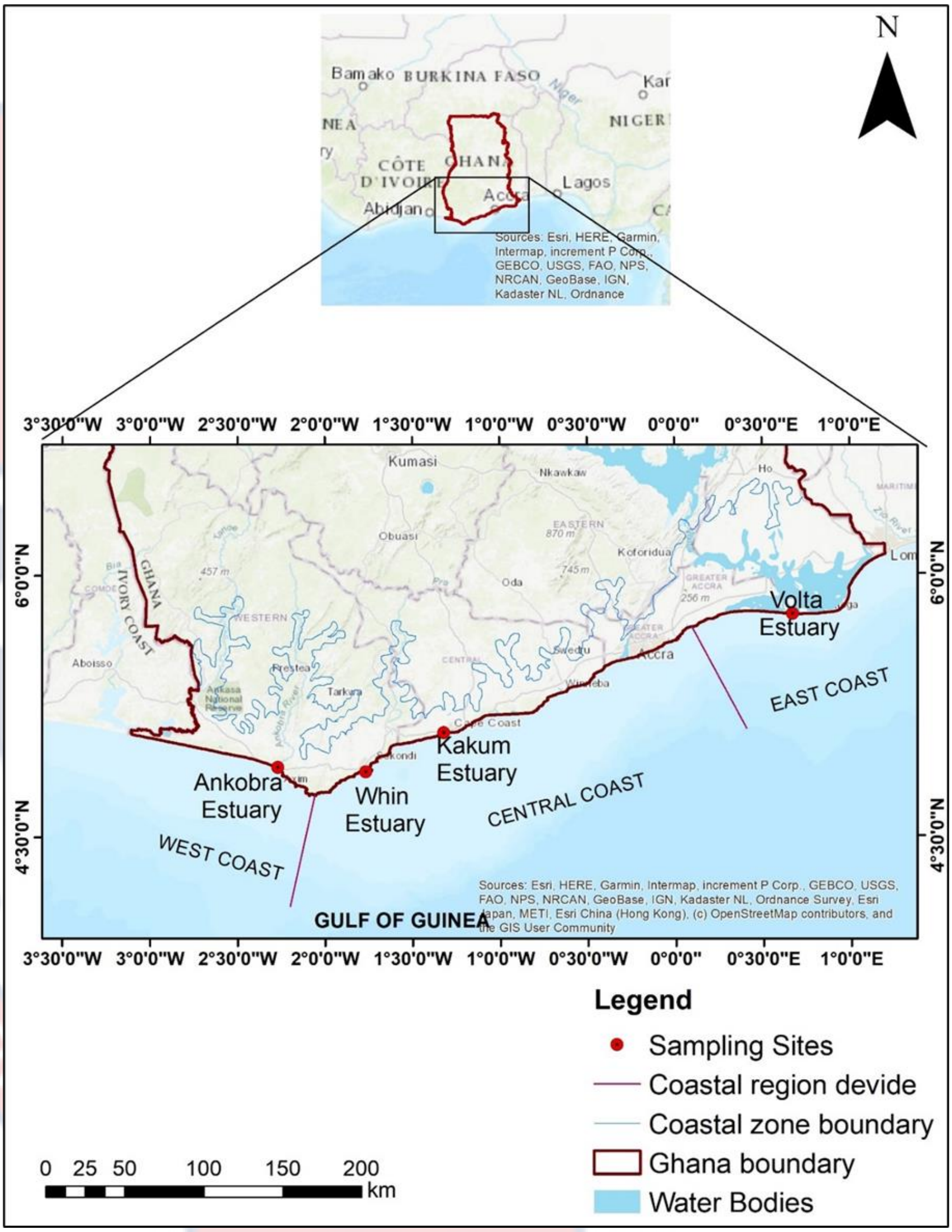


Figure 8: Map of Ghana's coastline showing study locations

The Ankobra Estuary

The Ankobra Estuary lies between latitudes 4°55’N and 4°54’N, and longitudes 2°17’W and 2°15’W. It is approximately 10 km within the mangrove ecosystems and discharges into the Gulf of Guinea at Asanta in Ellembelle district, just a few kilometres westward of Axim, the Western Region of Ghana. It is bound to the west by Boblama, to the east by Nzema East district and to the south by the Gulf of Guinea (Osman *et al*., 2016). The Estuary, which forms part of the Ankobra basin and lower section of Ankobra river, takes its source from the North hills of Basindare, near Bibiani. The Ankobra river is joined in the mid-section by rivers Mansi, Ankasa and Bonsa. The entrance of the estuary could be described as a sand ramp with abrupt increase in depth towards the freshwater end. The water column is extremely turbid. The riparian vegetation, mainly mangrove and strand with patches of rubber plantations provides niches for avifauna (Klubi *et al*., 2018). Topographically, it is a low-lying area with over 80 % of the landmass below 14 m above sea level (United States Geological Survey, 2011). Ankobra estuary is located within the South-Western Equatorial Climatic Zone of Ghana (Minerals Commission Ghana, 2011). The rainfall forms a bimodal regime having February–July as its major season and August–November as the minor season (Aduah *et al*., 2015). The soil type is ferralsols, which is low in fertility, has low infiltration and is highly prone to erosion (Minerals Commission Ghana, 2011). The Ankobra basin has a total surface area of approximately 8,400 km^2 and runs through Dompem, Prestea, Bogoso, Asankragua, Awaso, Tarkwa, Egyembra, Esiama and Axim townships. The two communities inhabiting the area around Ankobra estuary are Asanta and Sanwoma, distributed in a total of 1328 houses and structures inhabited by over 4069 people (Ghana Statistical Service, 2010). The main economic activities along the drainage system include both legal and illegal gold mining, cash crop farming and fishing. Illegal mining activities from upstream which form the major challenge in this estuary and these result in massive siltation that threatens the estuarine ecosystem health and consequently, its biodiversity. Additionally, surface run-offs from upland agricultural fields, as well as residential and municipal effluents contribute to the pollution of this estuary (Okyere & Nortey, 2018).

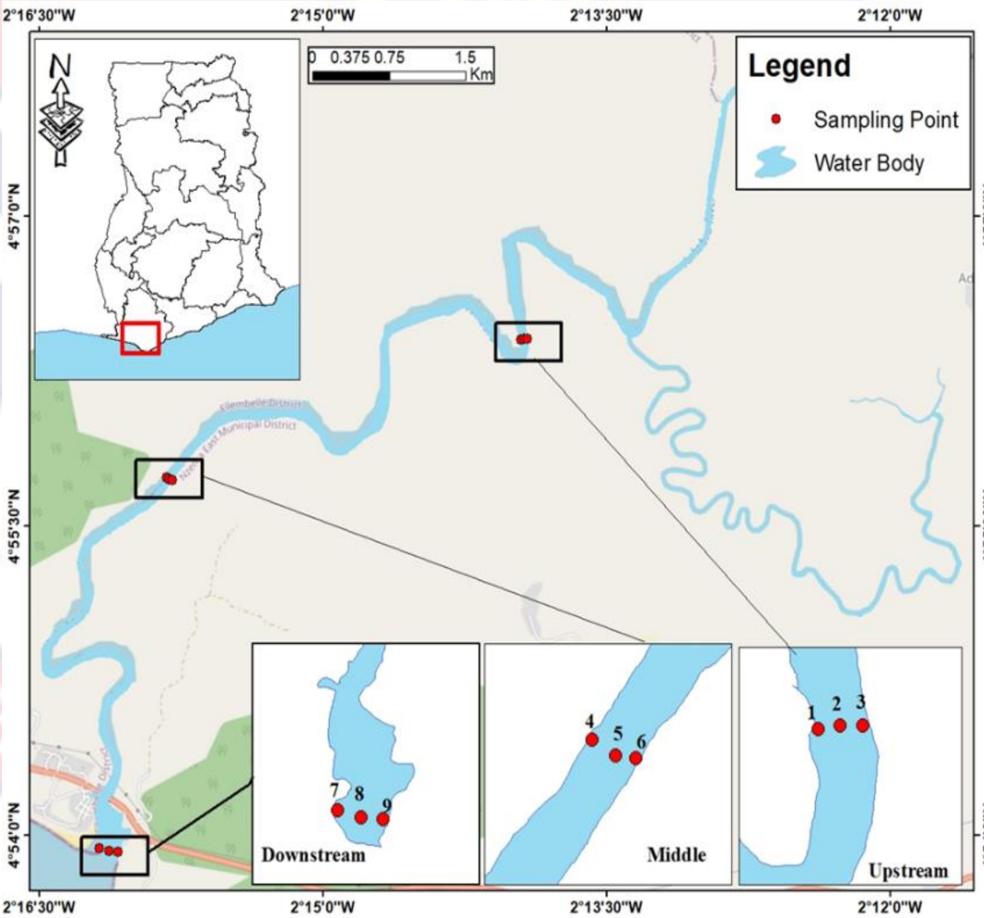


Figure 9: Map of Ankobra Estuary showing the sampling sub-stations upstream, midstream and downstream

The Kakum Estuary

The Kakum Estuary lies between latitudes 5º30'N and 5º47'N, and longitudes 0º12'W and 0º35'W. It is located along the Cape Coast-Takoradi highway near Iture community in the Cape Coast metropolis, Central Region and Central section of the coastline. The estuary is about 2 km west of the University of Cape Coast and about 3 km east of Elmina. It is formed by the Kakum river and Sweet (Sorowie) river, which drain from a rapidly urbanised area of the Central Region. The Kakum river is relatively bigger than the Sweet river, explaining the naming of the estuary after Kakum (Dzakpasu et al., 2015). The estuary is fringed by mangroves, forming the Kakum mangrove forest reported to have the highest diversity of mangroves in Ghana (DeGraft-Johnson *et al*., 2010); (Sackey *et al.*, 2011). Five different species of mangroves (red mangroves: *Rhizophora mangle*, *R. racemosa*, *R. harrisonii* and white mangroves: *Laguncularia racemosa* and *Avicennia germinans*) are found along this estuary (Sackey *et al.*, 1993). Occurring within the mangrove system are patches of marshes and sparsely distributed but interconnected pools through tidal exchanges. The pools serve as microhabitats within the mangrove swamp and provide refuge for a variety of fauna (Aheto *et al*., 2014). The estuary discharges into the Atlantic Ocean at Iture.

Kakum Estuary is located in the dry Equatorial Climatic Zone of Ghana. The average annual rainfall in the area is about 1,000 mm and the vegetation type is coastal savannah with grassland and few trees (Government of Ghana Official Portal-Central Region, 2024). The wettest months in this area are May/June and September/October while the drier periods occur in December – February with a brief period in August. Mean monthly temperature ranges from

24 °C in August to about 30 °C in March-April. The geological formation of the area is referred to as Dahomeyan. It is underlain by the Cape Coast basin type granitoids which have wide range of chemical and mineralogical composition (Junner, 1940). The rock types include ortho and para gneisses, schist and magmatites many of which are rich in garnet, hornblende and biotite. The soil consists mainly of coastal savanna and coastal sand. Two small communities, namely Iture and Abakam, are found along the estuary. The Iture community constitutes part of the Cape Coast-Elmina coastal plain. Its morphology and surficial geology bear characteristic signatures of Cape Coast granites, which have wide range of chemical and mineralogical composition (Junner, 1940). The inhabitants of Iture and Abakan communities depend on the estuary as their major source of fisheries and water for domestic activities. Sand winning is common at the Sweet river as a source of livelihood for the youth (Dzakpasu & Yankson, 2015; Fianko *et al.*, 2007).

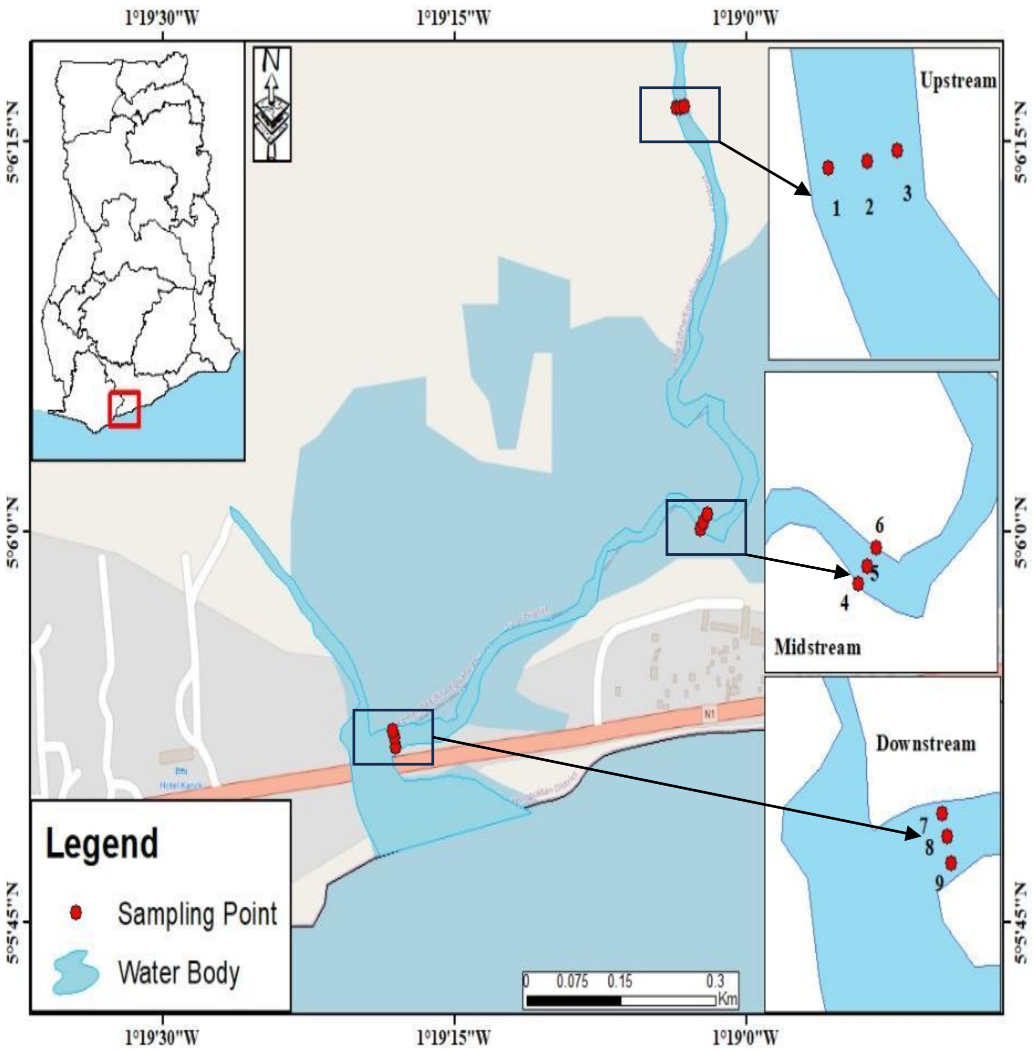


Figure 10: Map of Kakum Estuary showing the sampling sub-stations upstream, midstream and downstream

The Volta Estuary

The Volta Estuary lies between latitudes 5°46'N and 5°48'N and longitudes 0°37'E and 0°41'E. It is located in the lower basin of the Volta River at Ada- Foah in the Danghe East District of Greater Accra about 90 km from Accra. The Volta Estuary is located in a region within the coastal savannah zone with an annual rainfall of 750–1,250 mm (Dickson & Benneh, 1977). At the discharging point into the sea. The estuary is associated with a relatively large spit, about 1.2 km wide. The large spit is as result of a direct outgrowth of a natural change in the location of the mouth of the river (Anthony *et al.*, 2016)

The Volta River, a transboundary system and the largest river basin in Ghana, has a drainage area covering approximately 379,000 km^2 (Finlayson *et al.*, 2000). Its' sources are the White Volta, Black Volta, Red Volta and the Oti River all of which originate from the Burkina Faso (UNEP-GEP Volta Project, 2012). The Volta River basin cuts across six riparian countries as its catchment and comprises the major sediment supply to the Gulf of Guinea. The countries are; Burkina Faso (43 %), Ghana (42 %), Togo (15 %), Benin, Cote d'Ivoire and Mali (15 %) (Barry *et al.*, 2005). This river drains about 70 % of the Ghana's hydrological basin and its quality and quantity are dependent on the Akosombo and Kpong dams built on the river in 1965 and 1982, respectively, for hydro-electric power production (Nyarko *et al.*, 2017).

The geology is of the region is generally Quaternary which is made up of alluvial sand, silt and clay (Jayson-Quashigah *et al.*, 2013). The vegetation cover of the area is dominantly coconut and mangroves at its peripheries while the bottom floor provides suitable substrate for the *Galatea paradoxa*, as well as other benthic fauna (Klubi *et al.*, 2018). The area is the second-most important bird site for wintering waterbirds on the Ghanaian coast, supporting estimated maximum numbers of over 1,00,000 birds (BirdLife International, 2023). The beaches adjacent to this estuary serve as nesting grounds for three species of threatened marine turtle; *Lepidochelys olivacea*, *Dermochelys coriacea* and *Chelonia mydas*. Human activities in and around the estuary comprise mainly of farming, fishing and salt production (Mahu, 2016). The estuary also provides transport facilities for the communities and it is a recognised touristic site (Klubi *et al*., 2018).

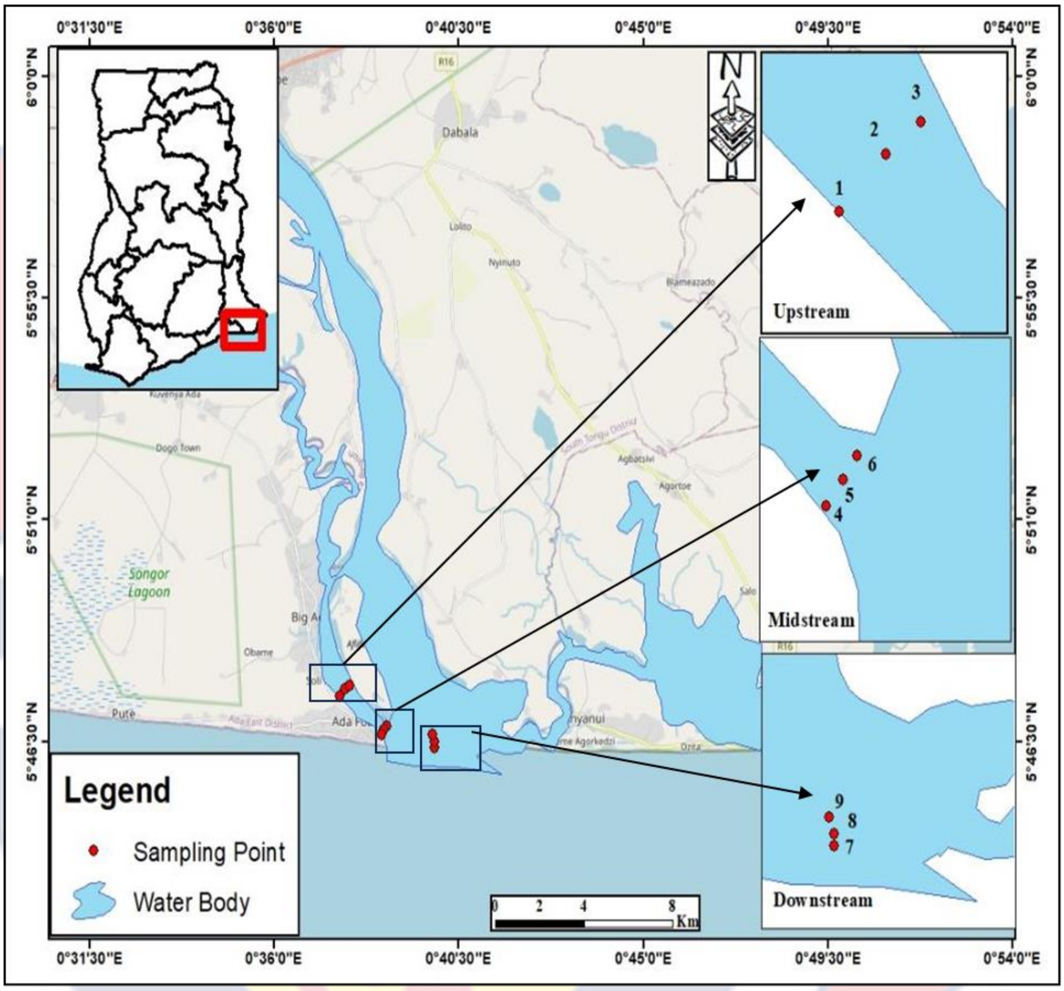


Figure 11: Map of Volta Estuary showing the sampling sub-stations upstream, midstream and downstream

The Whin Estuary

The Whin Estuary is located in the westernmost section of the Sekondi-Takoradi Metropolis in Ahanta West District on longitude 1˚ 48' W and 1˚ 48' W; and latitude 4˚ 52' N and 4˚ 56' N of the Western Region of Ghana. It is classified as an urban wetland due to its proximity to urban development (CRC/FoN, 2010), being a recipient of major effluents from near-by industrial area (Chuku *et al.*, 2023). It lies perpendicular to the sea relative to the orientation of the prevailing shoreline at its narrow mouth, which is about 90 m wide compared to further upstream where it spans over 350 m wide. It is a partially enclosed estuary with a free connection with the open sea (Sneli, 2012), relatively shallow and constantly fed with freshwater from the Whin River (Chuku *et al.*, 2023). The sources of the estuarine water are land drainage, direct rain and the sea (Yankson & Kendall, 2001). The Whin River is stretched into two branches forming a funnel-shaped structure joining and pouring into the estuary. The larger arm lies on the Western side of Adakope, a suburb of Takoradi while the smaller arm is sandwiched between Adakope and Kokompe on the Eastern side of Adakope. The mouth of the estuary lies on the Western side of the African Beach Hotel located on the Habour Road parallel to the shoreline (CRC/FoN, 2010). Its' size is estimated to be 652,202 km^2, with its' banks heavily vegetated with thickets of mangrove stands, an indication of a closed vegetation pattern surrounding the estuary. The mangrove vegetation provides suitable habitat for the *C. tulipa* including the relatively deep rocky sections (>1m) at the western north area which serves as their natural sanctuary (Chuku *et al*., 2023). Coconut trees and other coastal vegetation compliment the mangroves (Nortey *et al.*, 2016). The geology of the Whin estuary area is predominantly sandstone and grits (Ghana Minerals Commission, 2011) cited in Nortey *et al.*,2016). The bank of the estuary is a mix of beach sand to the Western side and rocky beaches to the Eastern side (CRC/FoN, 2010). Within the estuary are three neighbouring communities, New Amanfrul, Adakope and Apremdo, whose major source of livelihood is obtained through artisanal fishing and subsistence farming. The estuary forms a fishing hub for more than fifty fishermen (CRC/FoN, 2010).

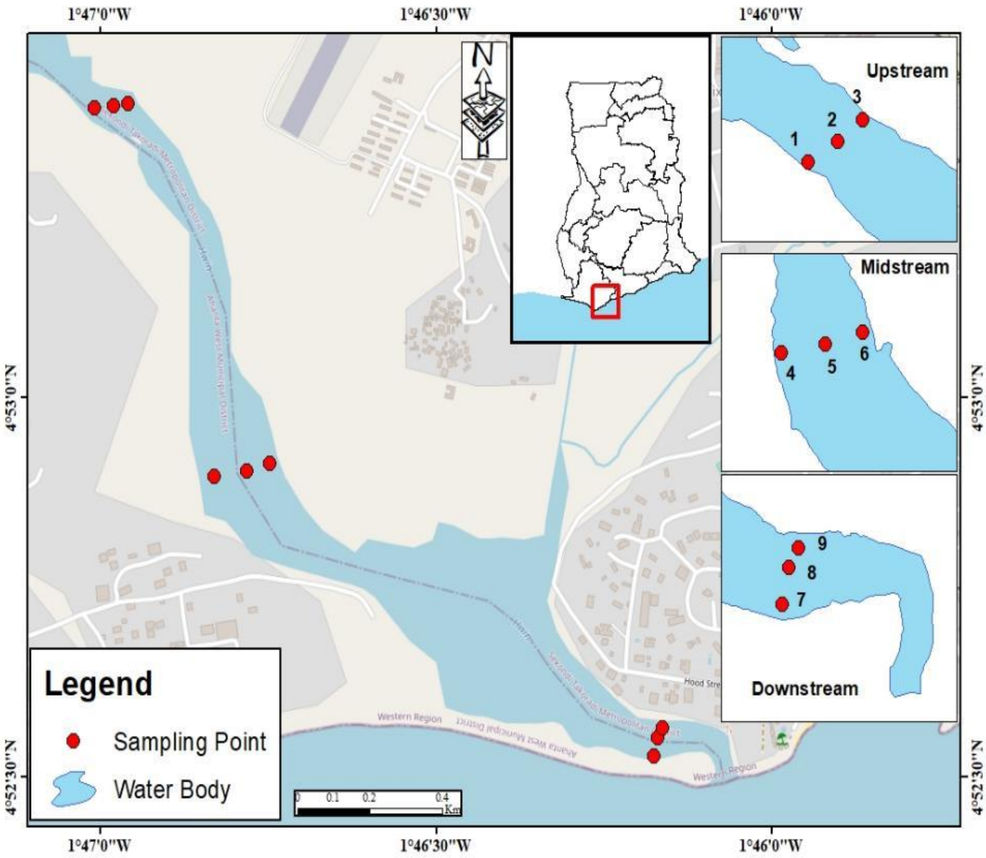


Figure 12: Map of Whin Estuary showing the sampling sub-stations upstream, midstream and downstream

4.2.2 Sample collection

Each estuary was demarcated into three zones; upstream, mid-stream and downstream to ensure collection of representative samples. Zonation of the estuaries was based on proximity to the riverine system using observation of mangroves and distance with the aid of a handheld Global Positioning System (GPS) gadget (Garmi etrex 60). Samples were collected bi-monthly between April 2022 and February 2023 to cover one hydrological year at low tide following a tide table (https://tides4fishing.com/af/ghana).

Water

In situ measurements of surface water temperature (SWT), dissolved oxygen (DO), electrical conductivity (EC), salinity and pH were performed using a HORIBA water quality monitor, model U-5000 (JAPAN) with multi-parametric probes. Turbidity and total suspended solids (TSS) were measured using a pre-calibrated multi-parametric photometer (DR 900). Water samples were collected in pre-cleaned 350-mL plastic polyethylene bottles rinsed with deionised water. Before transportation, water samples were kept under ice for laboratory analysis of chemical oxygen demand (COD), nitrate-nitrogen (NO_3-N), ammonium-nitrogen (NH_4-N) and orthophosphate (PO_4^{3-}).

Benthic macroinvertebrates

Three replicate samples of benthic macroinvertebrates were collected at each sampling station using Ekman grab (15 cm x 15 cm) at low tide alongside the water samples. Samples were screened in the field using a set of sieves with mesh sizes of 4.0 mm, 2.0 mm and 0.5 mm. During the sieving process, the larger sieves were stacked above the smaller ones and fauna retained on the sieves were preserved in 10 % formalin for further laboratory examination. The preserved sieved samples were stained with eosin dye before sorting to enhance visibility. The benthic macroinvertebrates found were observed under a dissecting microscope and identified to the lowest possible taxonomic level with the aid of relevant manuals and keys (Chapman, 2007; Lawson, 2011; Smith, 2004; Yankson & Kendall, 2001). The counts of different taxa groups and individual species were recorded for further analysis.

4.2.4 Laboratory analyses

Analysis of physicochemical parameters in water

a. Nitrate-Nitrogen (NO_3-N) determination

NO_3-N was determined using UV spectrophotometric method (APHA/AWWA/WEF, 2018). In this method, 1 ml of hydrochloric acid (HCl) was added to 50 ml of filtered sample and then mixed. The absorbance was read against re-distilled water set at zero absorbance. A wavelength of 220 nm was used to obtain NO_3-N reading, while interference due to dissolved organic matter was obtained by reading the wavelength at 275 nm. To obtain absorbance due to NO_3-N, the absorbance reading at 275 nm was subtracted from the reading at 220 nm and concentration was calculated using equation 1 (R^2=0.99) obtained from a nitrate-nitrogen standard calibration curve.

$$y = 0.1719x \quad (28)$$

b. Ammonium-Nitrogen (NH_4-N) determination

Ammonium-Nitrogen was determined using Nesslerisation method (APHA/AWWA/WEF, 2018). In this method, 1 ml of $ZnSO_4$ solution and 0.4 ml NaOH were added to 100 ml of the sample to obtain a pH of 10.5. This was allowed to settle and a suitable aliquot of the sample (25 ml) was taken after filtration. One (1) drop of EDTA reagent was added, together with 3 ml of Nessler reagent and the contents were thoroughly mixed. A blank was prepared in the same way by using distilled water instead of the sample. Absorbance was read after 10 minutes at 410 nm from a UV scanning spectrophotometer, then used to calculate NH_4-N concentration using equation 3 (R^2=0.98) obtained from the ammonium-nitrogen standard calibration curve.

$$y = 0.0023x \quad (29)$$

c. Orthophosphate (PO_4^{3-}) determination

Orthophosphate was determined using Ascorbic acid method (APHA/AWWA/WEF, 2018). The prepared reagents of sulphuric acid (A), potassium antimonyl tartrate solution (B), ammonium molybdate solution (C) and ascorbic acid solution (D) were mixed in the ratio of 10:1:3:6, respectively. In 50 ml of the sample, a drop of phenolphthalein indicator was added, together with 8 ml of the combined reagent and mixed. After 10 minutes of reaction time, the absorbance of each sample was read at 880 nm. A blank reagent was used as the reference. For turbid samples, a sample blank was prepared by adding all reagents except ascorbic acid and potassium antimonyl tartrate to the sample, and the blank absorbance was subtracted from the sample absorbance reading. The concentration of orthophosphate was calculated using equation 2 (R^2=0.99) obtained from the orthophosphate standard calibration curve.

$$y = 0.5543x \quad (30)$$

d. Chemical Oxygen Demand (COD) determination

Chemical Oxygen Demand was determined on unfiltered samples by dichromate oxidation using the closed reflux, titrimetric method according to Standard Methods for Examination of Water and Wastewater (APHA/AWWA/WEF, 2018). For this analysis, standard 10 ml borosilicate ampules were used as digestion vessels while standard potassium dichromate was use as the digestion solution. The ampules were washed with 20 % H_2SO_4 before use to prevent contamination. Volumetric measurements were made in the order of 2.5 ml of the sample, 1.5 ml digestion solution and 3.5 ml sulphuric acid reagent. The sample was placed in the ampule and the digestion solution added. The sulphuric acid reagent was carefully run down the inside of the ampules so an acid layer was formed under the sample-digestion solution layer. The ampules were tightly capped and inverted several times each for homogenisation. The ampules were placed in block digester pre-heated to 150 °C and refluxed for 2 hours behind a protective shield then allowed to cool to room temperature. Using standardised 0.01M ferrous ammonium sulphate (FAS) solution and 2-3 drops of ferroin indicator, the mixture was titrated until the end point was achieved. In the same manner, a blank containing the reagents and a volume of distilled water equal to that of the sample was refluxed and titrated. COD was calculated using the formula:

COD (mg/l) = $\frac{(A-B) X\ M\ X\ 8000}{ml\ sample}$ where:

A = ml FAS used for blank, B = ml FAS used for sample, M = molarity of FAS used, 8000 = milliequivalent weight of oxygen x 1000 ml/l.

Analysis of nutrients in sediments

a. Nitrate-Nitrogen and orthophosphate

Nitrate-Nitrogen and orthophosphate in the estuarine sediments were first extracted using calcium sulphate and Mehlich 2 extraction methods for nitrogen and orthophosphate respectively using the Hach method (Hach, 2001). After extraction, the filtrate was subjected to UV spectrophotometric and Ascorbic acid methods (APHA/AWWA/WEF, 2018), respectively.

4.2.4 Statistical analysis of data

Descriptive statistics were carried out to summarise physicochemical data and results were presented in a table and graphs. The differences in median concentrations of physicochemical parameters in the estuaries were tested using One-way non-parametric ANOVA (Kruskal-Wallis test). To check whether the residuals followed a normal distribution, the Shapiro-Wilk test was applied. The Shapiro -Wilk test was calculated as; W = $\frac{(\sum_{i=1}^{n} a_i\, x_{(i)})^2}{(\sum_{i=1}^{n}(x_i - \bar{x})}$ where a_i are the constants generated by the expression, $x_{(i)}$ are the ordered sample values, i.e., the i^{th} smallest number in the sample and $\bar{x}$ is the sample mean, i.e., $(x_1 + \cdots + x_n)/n$. In case of significant differences among groups, Tukey's Honest Significant Difference test (HSD) was performed. The Turkey's HSD test was determined as; HSD = q.$\sqrt{\frac{MS_w}{n_k}}$ where q is a constant from the Studentised range q table, MS_w is the mean square within, n_k is the number in each category (n for one condition). Descriptive statistics, ANOVA and normality tests were carried out in SigmaPlot software (v.14.0) with a significance threshold set at $\alpha = 0.05$. Macroinvertebrate structure was described through the species richness (d), the Shannon-Wiener diversity index (H'), and evenness (J'). Richness was determined from Margalef's index (d) calculated as d $= \frac{s-1}{\ln N}$ where s is the number of species in a sample and N is the total number of individuals in the sample. Species diversity was calculated using the equation given as H'$= -\sum_{i=1}^{s} Pi\,(\ln Pi)$ where Pi is the proportion of the ith species and s is the number of species in a sample. Evenness was determined using Pielou's index calculated as J'$= \frac{H'}{(\ln s)}$ where s is the number of species and H' the Shannon-Wiener diversity index. Spearman's rank order correlation analysis was used to establish the relationship between physicochemical parameters and macroinvertebrate abundance which was performed in Palaeontological Statistics (PAST) software (v. 4.03).

4.3 Results

4.3.1 Spatial variation in physicochemical parameters in the study sites

There was variation in median SWT in the estuaries over the study period ($P<0.05$; $H=28.3$; $df=2$ Kruskal-Wallis test). The SWT in Whin (29.8 °C) and Volta Estuaries (29.7 °C) differed significantly with that recorded in both Kakum (27.9 °C) and Ankobra Estuaries, (27.6 °C), ($P<0.05$; Tukey's post hoc test), (Table 4.1). However, there was no significant difference in SWT in

Ankobra and Kakum Estuaries (P =0.985; Tukey's post hoc test), as well as Whin and Volta Estuaries (P =1.0; Tukey's post hoc test); Table 12.

The median DO concentration did not vary significantly among all the estuaries ($P \geq 0.05$; Kruskal-Wallis test); Table 12). Moreover, there was no significant difference in pH values among the four estuaries ($P \geq 0.05$; Kruskal-Wallis test), Table 12.

The highest and lowest median EC concentrations were recorded in Whin (16500.0 µS/cm) and Ankobra Estuaries (93.0 µS/cm), respectively (Table 12). The EC concentration in the estuaries varied significantly ($P<0.05$; $H=49.7$; $df=2$; Kruskal-Wallis test). The median concentration of EC in Whin and Kakum Estuaries was statistically different from both Ankobra and Volta Estuaries ($P<0.05$; Tukey's post hoc test). Nevertheless, the median EC concentration in the Volta and Kakum Estuaries did not show any significant difference ($P>0.05$; Tukey's post hoc test), (Table 12).

The highest and lowest median turbidity concentrations were recorded in Ankobra (751.5 NTU) and Volta Estuaries (5.0 NTU), respectively (Table 4.1). Turbidity in the estuaries varied considerably, ranging from as low as 0.0 NTU in the Volta Estuary to above 3000 NTU in Ankobra Estuary ($P<0.05$; $H=135.9$; $df=2$; Kruskal-Wallis test), Table 12. Turbidity in the Volta Estuary differed significantly from Kaum, Volta and Whin estuaries ($P<0.05$; Tukey's post hoc test), Table 12.

Generally, NO_3-N concentration in the water column was low across the estuaries, with values ranging between 0.01 mg/L to a maximum of 21.8 mg/L, (Table 4.1). Variation in NO_3-N concentration in estuaries was noted ($P<0.05$; $H=47.9$; $df=2$; Kruskal-Wallis test). The median NO_3-N concentration in Volta

Estuary (0.9 mg/L) differed significantly with both Kakum (2.3 mg/L) and Ankobra (1.5 mg/L) Estuaries ($P<0.05$; Tukey's post hoc test), Table 12. However, no significant difference was observed in median concentration of NO_3-N in the water column between Kakum and Ankobra Estuaries ($P>0.05$), Table 12. In the sediment matrices, no notable significant difference was observed in NO_3-N concentration among the four estuaries ($P\geq0.05$; Kruskal Wallis test).

Median orthophosphate concentration in the water column varied significantly across the estuaries ($P<0.05$; $H=59.2$, $df=2$; Kruskal Wallis). The concentration of orthophosphate in the water column in Whin (3.4 mg/L) and Ankobra Estuaries (11.0 mg/L) were significantly different from both Volta (1.9 mg/L) and Kakum (2.7 mg/L) Estuaries ($P<0.05$; Tukey's post hoc test), Table 12. Nevertheless, orthophosphate concentration in the water column in Volta and Kakum Estuaries did not differ statistically, just like in both Whin and Ankobra Estuaries ($P>0.05$; Kruskal Wallis test), Table 12. In the sediment matrices, significant differences existed among the estuaries ($P<0.05$; Kruskal Wallis test). The highest orthophosphate concentration in sediments was recorded in both Kakum (18.1 mg/L) and Whin (17.6 mg/L) Estuaries, with no significant difference between them ($P>0.05$; Kruskal Wallis test). Although the concentration of orthophosphate in the sediments differed between Whin and Volta Estuaries ($P<0.05$; Tukey's post hoc); no statistical difference occurred between Ankobra and Volta Estuaries ($P>0.05$; Kruskal Wallis test), Table 12.

Among the nutrients sampled, NH_4-N was the highest in concentration across the four estuaries in comparison to NO_3-N and orthophosphate. Among the estuaries, only the Whin differed significantly in NH_4-N concentration from the rest (P<0.05; Tukey’s post hoc test), which did not show any variation in NH_4-N concentration (P>0.05; Kruskal Wallis test). In essence, a relatively wide range of values was recorded from as low as 0.01 mg/L to above 1200 mg/L in Ankobra Estuary (Table 12).

A significant variation was observed in mean COD concentration among the estuaries (P<0.05; H=44.9; df=2; Kruskal-Wallis test). The median COD concentration in Ankobra Estuary (2640 µS/cm) differed significantly with that recorded in Kakum (474 mg/L) and Volta (252 mg/L) Estuaries (P<0.001; Tukey’s post hoc test), Table 12. Nonetheless, the difference in median COD concentration in Kakum and Volta Estuaries was not statistically significant (P=0.466; Tukey’s post hoc test), Table 12.

Table 12: *Summary of Physicochemical Characteristics*

	Estuaries			
Parameter	**Whin**	**Ankobra**	**Kakum**	**Volta**
Temp (°C)	(29.8±2.4) ^{a} **24.2-32.9**	(27.6±1.4) ^{b} **25.9-30.1**	(27.9±1.4) ^{b} **24.4-29.6**	(29.7±1.3) ^{a} **27.1-31.9**
DO (mg/l)	(4.7±3.3) ^{a} **0.7-13.6**	(5.4±2.7) ^{a} **0.0-10.9**	(5.6±2.4) ^{a} **0.23-9.4**	(6.1±0.9) ^{a} **4.4-9.2**
pH	(6.7±2.8) ^{a} **0.1-9.7**	(6.2±2.4) ^{a} **0.9-8.7**	(5.0±2.3) ^{a} **0.7-8.9**	(6.4±2.2) ^{a} **1.2-8.8**
EC (µs/cm)	(16500.0±18410.6) ^{a} **153.0-52700.0**	(93.0±7004.4) ^{c} **48.0-29800.0**	(7580.0±12516.11) ^{b} **160-40500**	(1420.0±4648.4) ^{c} **80-29900**
Salinity (ppt)	(3.6±12.5) ^{a} **0.0-35.7**	(0.1±4.7) ^{b} **0.0-20.6**	(0.3±6.7) ^{b} **0.0-20.7**	(0.4±2.9) ^{b} **0-19.5**
Turbidity (NTU)	(42.5±98.9) ^{b} **10.0-304.0**	(751.5±940.4) ^{a} **47.3-3292.0**	(44.5±32.2) ^{b} **11-151.3)**	(5.0±7.1) ^{b} **0-41.0**
w. Nitr (mg/l)	(1.5±2.5) ^{ab} **0.1-13.2**	(1.5±3.9) ^{a} **0.0-21.8**	(2.3±1.1) ^{a} **0.0-4.9**	(0.9±0.4) ^{b} **0.0-2.1**
w. Orth (mg/l)	(3.4±2.5) ^{a} **0.5-54.9**	(11.0±8.9) ^{a} **0.2-54.3**	(2.7±5.6) ^{b} **0.4-26.1**	(1.5±5.3) ^{b} **0.0-22.8**
Amm (mg/l)	(76.3± 373.4) ^{a} **0.0-1281.7**	(52.9±135.0) ^{b} **15.2-658.3**	(38.8±45.1) ^{b} **3.9-161.9**	(53.3±57.2) ^{b} **-0.87-277.8**
COD (mg/l)	(1504.0±4815.6) ^{ab} **0.0-30960.0**	(2160±8725.6) ^{a} **0.0-56160**	(474.0±1592.3) ^{b} **0.0-8480**	(230.4±1954.8) ^{b} **0.0-6960**
S. Nitr (mg/l)	(6.4±42.8) ^{a}	(1.8±59.6) ^{a}	(4.1±14.7) ^{a}	(6.6±3.4) ^{a}

	0.0-299.5	**-0.3-299.9**	**0.0-67.8**	**0-10.7**

Table 12, *continued*

S. Orth (mg/l)	(17.6±23.7) [a]	(3.2±21.2) [b]	(18.1±21.4) [ab]	(6.6±24.1) [b]
	1.4-79.1	**0.0-81.2**	**2.2-81.8**	**0.0-100**

Note: *The numbers in parenthesis are values for median and standard deviations, while those in bold represent the range. On each row, the medians with the same superscript letter are* **not** *significantly different at p = 0.05 level; n = 54 while those with different superscript letters indicate significant differences.* ***Abbreviations:*** *Temp-Temperature, DO-dissolved oxygen, EC-electrical conductivity, w. Nitr-nitrate-nitrogen concentration in water, w. Orth-orthophosphate concentration in water, Amm-ammonium-nitrogen, COD-chemical oxygen demand, S. Nitr-nitrate-nitrogen concentration in sediments, and S. Orth-orthophosphate concentration in sediments*

4.3.2 Occurrence of macroinvertebrates

Table 13 shows 28 taxa belonging to 26 families and from six classes in all the estuaries recorded during the sampling period. The highest occurrence of organisms was recorded in Whin Estuary, totalling to 1497 specimens. These were distributed in four classes with 13 polychaetes, one oligochaete, three crustaceans and one insecta. *Nephtys* sp. was the most abundant (324 specimens), followed closely by *Nereis* sp. (296 specimens) and *Capitella* sp. (289 specimens). Others occurring in large abundances included *Notomastus sp., Heteromastus sp*. and *Chironomous* sp. The highest abundance occurred in the upstream (1165 specimens), with a few in the midstream (254 specimens) and lowest abundance occurring in the downstream (78 specimens). Some taxa, e.g. *Glycera* sp and *Rhodine* sp, were only found in this estuary, with one and three specimens occurring in the upstream, respectively.

In the Kakum Estuary, the six classes included 11 polychaeta, four crustacea, and one taxon belonging to oligochaeta, clitellata and insecta in each case (Table 4.3). Polychaete worms were the most dominant group and they included *Sigambra* sp. (103 specimens), *Capitella* sp. (89 specimens), *Nephtys* sp. (88 specimens), *Nereis* sp. (57 specimens), *Heteromastus* sp. (42 specimens) and Notomastus sp. (39 specimens). Other polychaetes in low abundances were *Scoloplos* sp., *Polyphysia* sp., *Cossura* sp., *Lopadorrhynchus* sp. and *Phyllodoce* sp. Among the crustaceans in Kakum Estuary, *Bemlos* sp. recorded the highest abundance (146 specimens). Other crustaceans occurring in low abundances in this estuary were *Gammarus* sp., *Uca* sp. and *Elasmopus* sp. *Chironomous* sp, the only taxa in class insecta, had the highest abundance (169 specimens) in this estuary. Also, most of the organisms occurred in the upstream

(311 specimens) with relatively fewer in the downstream (261 specimens) and lowest number in the mid-stream (241 specimens), (Table 13).

In the Volta Estuary, five classes occurred and were represented by eight polychaeta, five crustaceans, two gastropoda and one specimen belonging to oligochaeta and insecta in each case, respectively (Table 13). Polychaetes dominated the estuary with *Capitella* sp. (157 specimens), *Scoloplos* sp. (63 specimens) and *Nereis* sp. (41 specimens). Other polychaetes occurring in low abundances were *Notomastus* sp., *Sigambra* sp., *Nephtys* sp., *Lumbriconereis* sp and *Syllis* sp. Moreover, the crustaceans in the Volta Estuary were *Eurydice* sp. (34 specimens) with others in low abundance like *Bemlos* sp., *Coenobita* sp., *Penaeus* sp. and *Mysis* sp. While low abundances of *Tubifex* sp. and *Chironomous* sp. occurred in the Volta Estuary, a high abundance of *Pachymelania* sp (150 specimens) and *Tympanotonus* sp. (43 specimens) belonging to class gastropoda were recorded. These particular taxa were only encountered in the Volta Estuary. The highest and lowest abundance of organisms in general was observed in the upstream (403 specimens) and downstream (38 specimens), respectively (Table 13).

Finally, the Ankobra estuary recorded the least number of taxa (7) represented by four polychaeta, two crustaceans and one oligochaeta. *Penaeus* sp was the most dominant organism with 154 specimens who dominated the upstream area of the estuary.

Table 13: *Occurrence of Benthic Macroinvertebrates at Various Stations in the Four Estuaries*

				Estuaries											
				Volta			**Ankobra**			**Whin**			**Kakum**		
Phylum	Class	Family	Organism name	St.1	St.2	St.3	St.1	St.2	St.3	St.1	St.2	St.3	St.1	St.2	St.3
Annelida	Polychaeta	Capitellidae	*Capitella* sp	+	+	+	+	-	-	+	+	+	+	+	+
		Capitellidae	*Notomastus* sp.	+	-	-	-	-	-	+	+	+	+	+	+
		Capitellidae	*Heteromastus* sp	-	-	-	-	-	-	+	+	+	+	+	+
		Nereididae	*Nereis* sp.	+	+	+	+	-	-	+	+	+	+	+	+
		Orbiniidae	*Scoloplos* sp.	+	+	+	-	-	-	+	-	-	+	+	+
		Pilargidae	*Sigambra* sp	+	+	-	-	-	-	+	+	+	+	+	+
		Nephtydae	*Nephtys* sp.	+	-	-	+	-	+	+	+	+	+	+	+
		Scalibregmatidae	*Polyphysia* sp	-	-	-	+	-	-	+	+	+	+	-	-
		Lumbrineridae	*Lumbriconereis* sp	+	+	-	-	-	-	+	-	-	-	-	-
		Syllidae	*Syllis* sp.	-	-	+	-	-	-	+	-	-	-	-	-
		Glyceridae	*Glycera* sp.	-	-	-	-	-	-	+	-	-	-	-	-
		Maldanidae	*Rhodine* sp.	-	-	-	-	-	-	+	-	-			
		Cossuridae	*Cossura* sp.	-	-	-	-	-	-	+	-	-	+	-	-
		Lopadorrhynchidae	*Lopadorrhynchus* sp	-	-	-	-	-	-	-	-	-	+	+	+
		Phyllodocidae	*Phylodoce* sp.	-	-	-	-	-	-	-	-	-	-	+	+
	Oligochaeta	Naididae	*Tubifex* sp.	+	-	+	+	-	-	+	+	+	+	+	+
	Clitellata	Glossiphoniidae	*Glossiphonia* sp	-	-	-	-	-	-	-	-	-	-	-	+
Arthropoda	Crustacea	Mysidae	*Mysis* sp.	+	-	-	-	-	-	-	-	-			
		Penaeidae	*Penaeus* sp.	+	-	+	+	+	+	+	-	-			
		Aoridae	*Bemlos* sp.	+	-	+	-	-	-	+	+	+	+	+	+
		Gammaridae	*Gammarus* sp.	-	-	-	-	-	-	+	-	-	+	+	+
		Ocypodidae	*Uca* sp.	-	-	-	+	-	-	-	-	-	+	-	-

		Cirolanidae	*Eurydice* sp.	+	+	+	-	-	-	+	-	-			
		Maeridae	*Elasmopus* sp.	-	-	-	-	-	-	-	-	-	+	-	+
		Coenobitidae	*Coenobita* sp.	+	+	-	-	-	-	-	-	-			
	Insecta	Chironomodae	*Chironomous* sp.	+	-	-	-	-	-	+	+	+	+	+	+
Mollusca	Gastropoda	Hemisinidae	*Pachymelania* sp.	+	+	+	-	-	-	-	-	-	-	-	-
		Potaminidae	*Tympanotonus* sp.	+	-	-	-	-	-	-	-	-	-	-	-

Note: St.1, St.2 and St.3 indicates upstream, midstream and downstream, respectively. (+) indicates present, (-) indicates absent

4.3.3 Composition of benthic macroinvertebrates

Polychaetes were the most dominant macroinvertebrates in the estuaries, accounting for 55 %, 82 %, 46 and 5 % of the total macroinvertebrate fauna recorded in Kakum (Figure 13a), Whin (Figure 13b) and Volta Estuaries (Figure 14a), respectively. Whereas, in Ankobra Estuary (Figure 14b), crustaceans were the most dominant group represented by *Penaeus* sp., which accounted for 94 % of all the benthic fauna encountered. Moreover, polychaetes and crustaceans were ubiquitous in all the four estuaries. In this study, gastropods (*Pachymelania* sp. and *Tympanotonus* sp.) were only present in the Volta Estuary, accounting for 31 % of all benthic fauna encountered (Figure 14a). Additionally, insecta (*Chironomous* sp.) were only recorded in Kakum (Figure 13a) and Whin (Figure 13b) Estuaries, accounting for 21 % and 8 % of all benthic fauna in Kakum and Whin Estuaries, respectively. There was no record of gastropods, insects and clitellates in the Ankobra Estuary (Figure 14b).

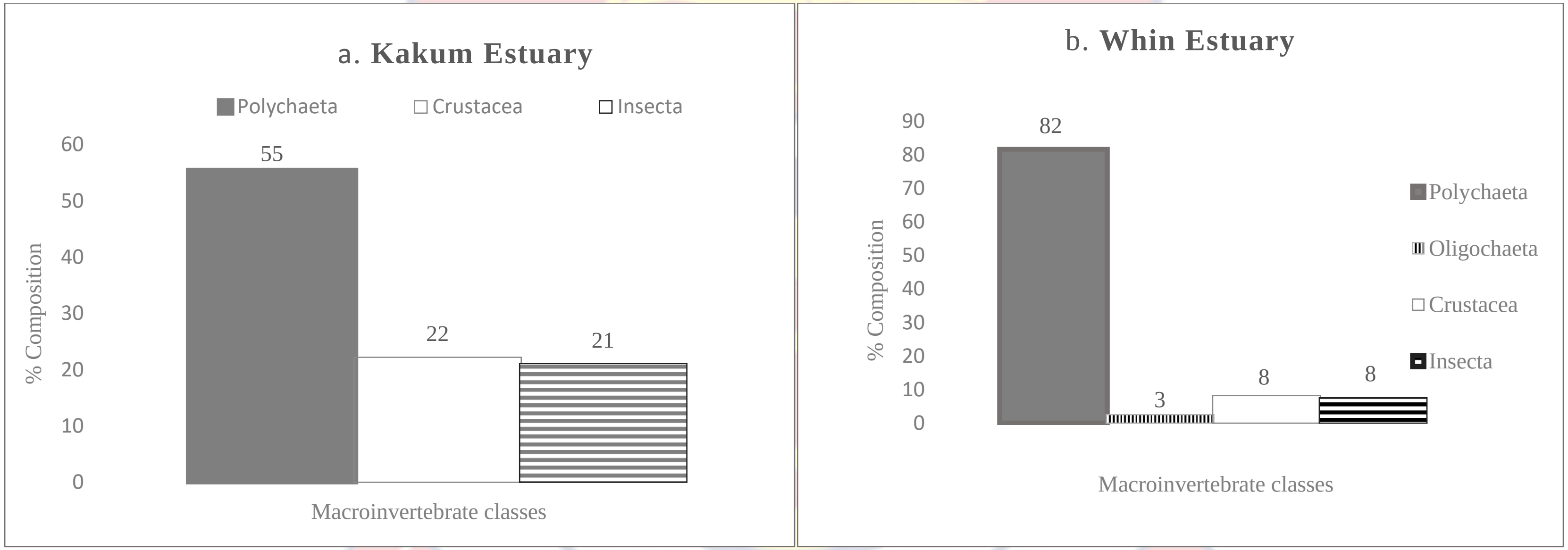


Figure 13: Percentage composition of macroinvertebrate classes in (a) Kakum and (b) Whin Estuaries

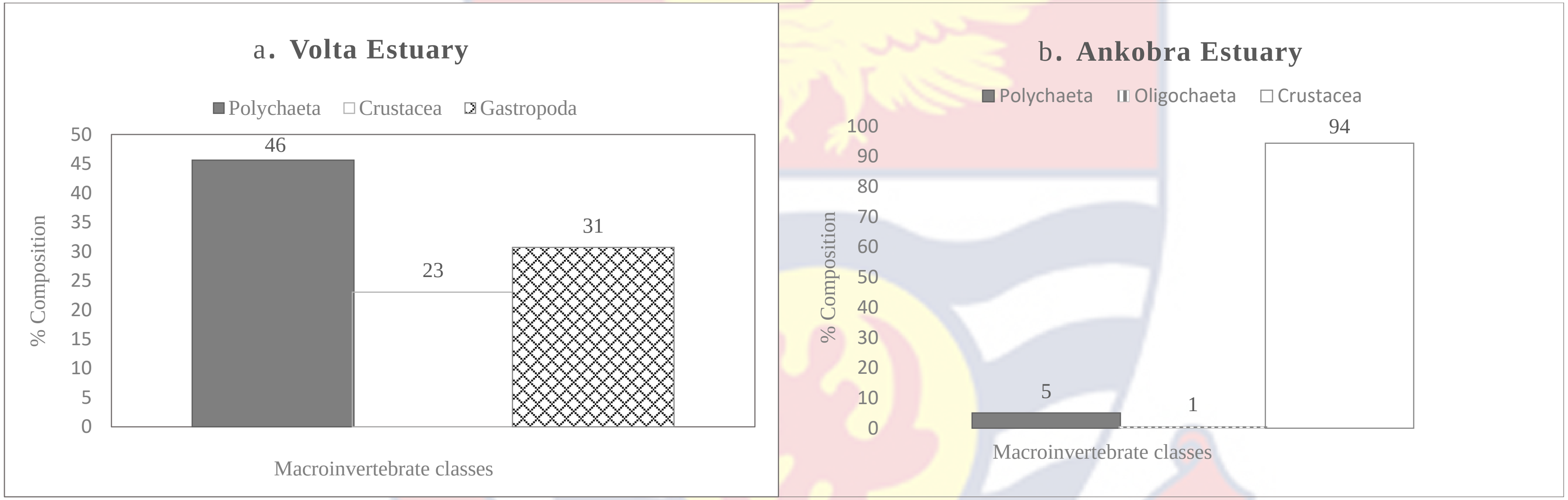


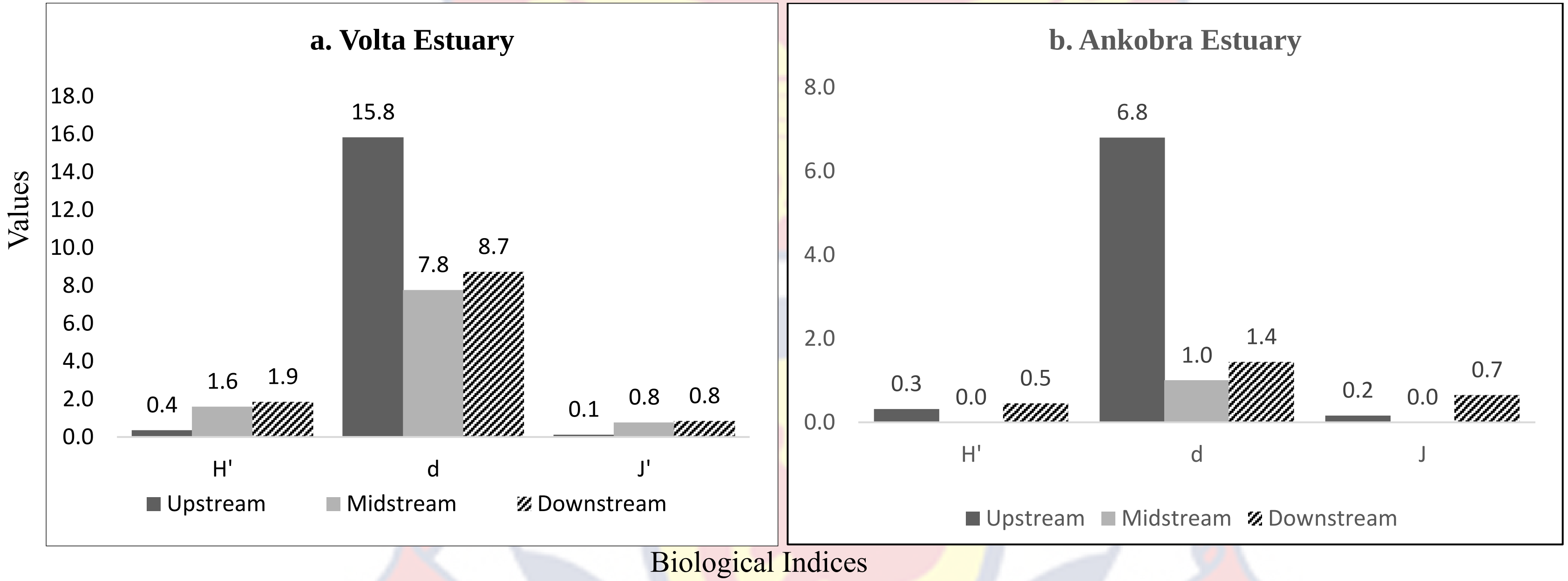
Figure 14: Percentage composition of macroinvertebrate classes in (a) Volta and (b) Ankobra Estuaries

4.3.4 Spatial variation in diversity of macroinvertebrates among the estuaries

The biological indices for benthic macroinvertebrates in specific stations (upstream, midstream and downstream) of each estuary are shown in Figure 15. Species diversity varied with each estuary. Species diversity was highest in the downstream of Volta (1.9) and Ankobra (0.5) Estuaries, Figures 15(a) and (b). In Whin Estuary, species diversity in the upstream (2.1) was similar to that recorded downstream, Figure 15(c), while in Kakum Estuary species diversity decreased towards downstream, Figure 15(d).

Species richness was highest in the upstream in all the estuaries, with Volta, Ankobra, Whin and Kakum Estuaries recording richness of 15.8, 6.8, 17.9 and 15.8, respectively, Figure 15. The general trend indicated low species richness in the midstream of all the estuaries apart from Whin Estuary, where similar species richness was recorded in both midstream and downstream, Figure 15.

Species evenness increased towards the downstream in Volta, Ankobra and Whin Estuaries, while it was highest in the upstream of Kakum Estuary, Figure 15. Generally, relatively higher values ($\geq$0.7) of species evenness were recorded in the four estuaries.


a. Volta Estuary
Values
18.0
16.0
14.0
12.0
10.0
8.0
6.0
4.0
2.0
0.0
0.4
1.6
1.9
15.8
7.8
8.7
0.1
0.8
0.8
H'
d
J'
Upstream
Midstream
Downstream
b. Ankobra Estuary
8.0
6.0
4.0
2.0
0.0
0.3
0.0
0.5
6.8
1.0
1.4
0.2
0.0
0.7
H'
d
J
Upstream
Midstream
Downstream
Biological Indices

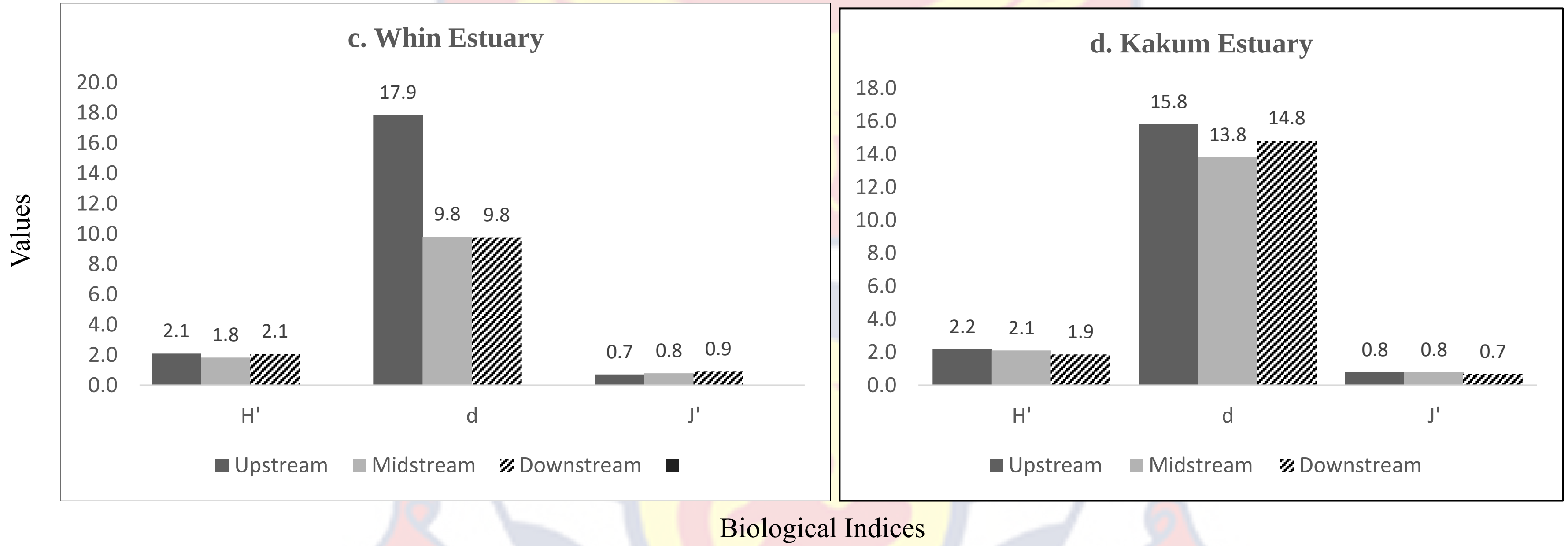


Figure 15: Biological indices in (a)Volta, (b) Ankobra (c) Whin and (d) Kakum Estuaries (H': diversity index, J': evenness index and d: richness index)

The number of individual specimens, taxa and biological indices for Whin, Kakum, Volta and Ankobra Estuaries in general is illustrated in Table 14. The Whin estuary recorded the highest number of individuals (1497) and taxa (18), while the lowest number was recorded in Ankobra estuary (164 individuals and 7 taxa). The highest and lowest species diversity was encountered in Kakum (H'=2.28) and Ankobra Estuary (H'= 0.33), respectively (Table 14). Additionally, relatively high species evenness (J'=0.79) was found in Kakum and lowest in Ankobra (J'=0.17) estuaries. On the other hand, the Whin Estuary was found to have the highest species richness (d=17.86) as compared to Kakum (d=17.85), Volta (d=16.84) and Ankobra (d=6.8) estuaries, Table 14.

Table 14: *Biological Indices of Macroinvertebrate Community at the Various Estuaries*

Biological Indices	**Estuaries**			
	Whin	**Kakum**	**Volta**	**Ankobra**
No. of individuals	1497	813	519	164
No. of Taxa	18	18	17	7
Shannon wiener (H')	2.09	2.28	1.86	0.33
Pielou's (J)	0.72	0.79	0.66	0.17
Margalef's (d)	17.86	17.85	16.84	6.80

4.3.5 Species-environment interactions

Associations were observed among individual physicochemical parameters and different benthic fauna using the Spearman's Rank Order Correlation. In the Whin estuary, most of the associations occurred among individual physicochemical parameters and individual benthic macroinvertebrates (Figure 16). However, *Chironomous* species displayed weak negative correlations with temperature (r=-0.272) and DO (-0.277) but was positively correlated with nitrates in the water column (w. Nitr), (r=0.289).

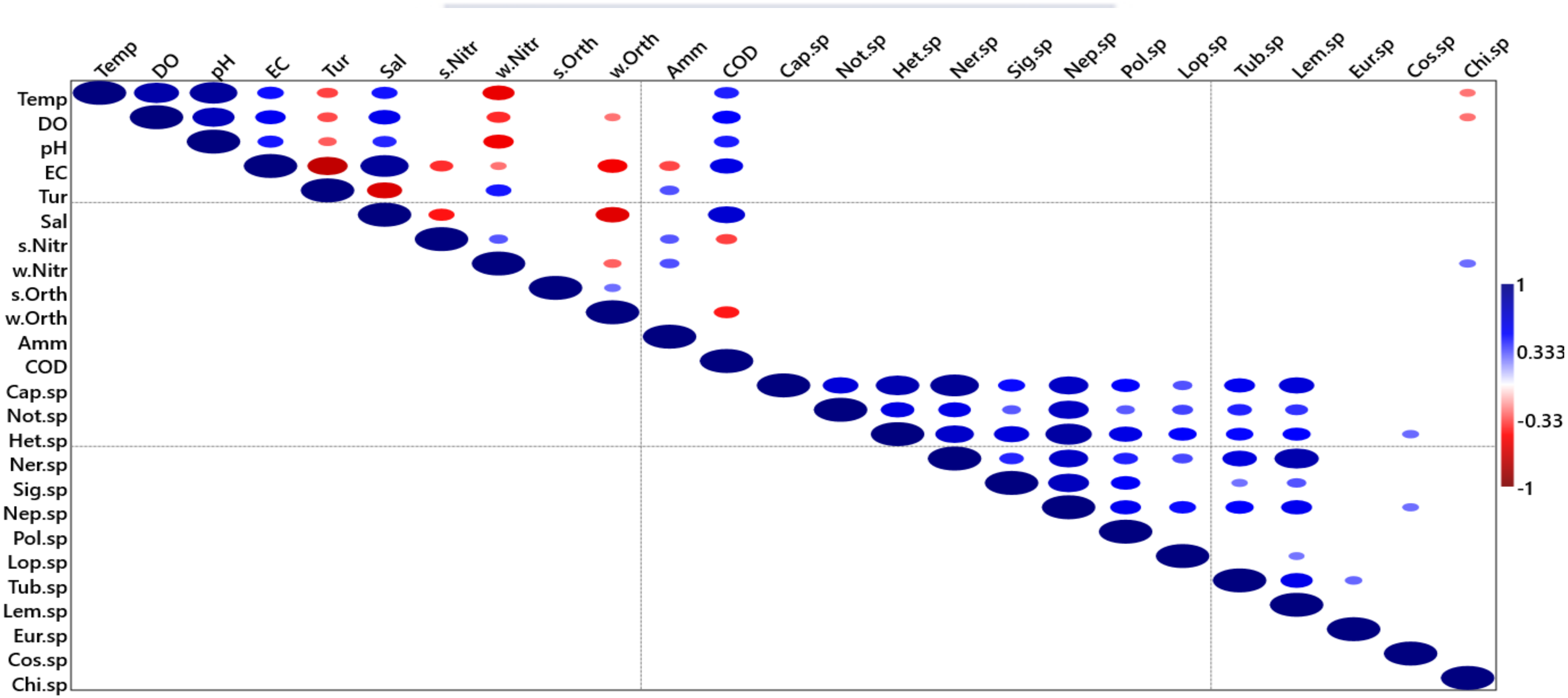


Figure 16: Spearman's rank order correlation for Whin Estuary

Note*: In the correlation matrices, deep blue and deep red colour indicate positive and negative correlations, respectively. The deeper the colour the stronger the correlation. The circles indicate p<0.05 and the larger the size of the circle the stronger the correlation.*

Abbreviations*: Temp-temperature, DO-dissolved oxygen, EC- electrical conductivity, COD-chemical oxygen demand, Tur-turbidity, Sal-salinity, w.Nitr-nitrogen-nitrogen in water, S.Nitr-nitrogen-nitrogen in sediment, w.Orth-orthophosphate in water, S.Orth-orthophosphate in sediment, Amm-ammonium-nitrogen, Cap-Capitella, Not-Notomastus, Het-Heteromastus, Ner-Nereis, Sig-Sigambra, Neph-Nephtys, Pol-Polyphysia, Tubi-tubifex, Chir -Chironomous, Lem-Lembos, Eur-Euridyce, Cos-Cossura, Lop- Lopadorrhynchus,*

In the Kakum Estuary, a number of associations were observed among individual physicochemical parameters and among the different benthic fauna (Figure 17).

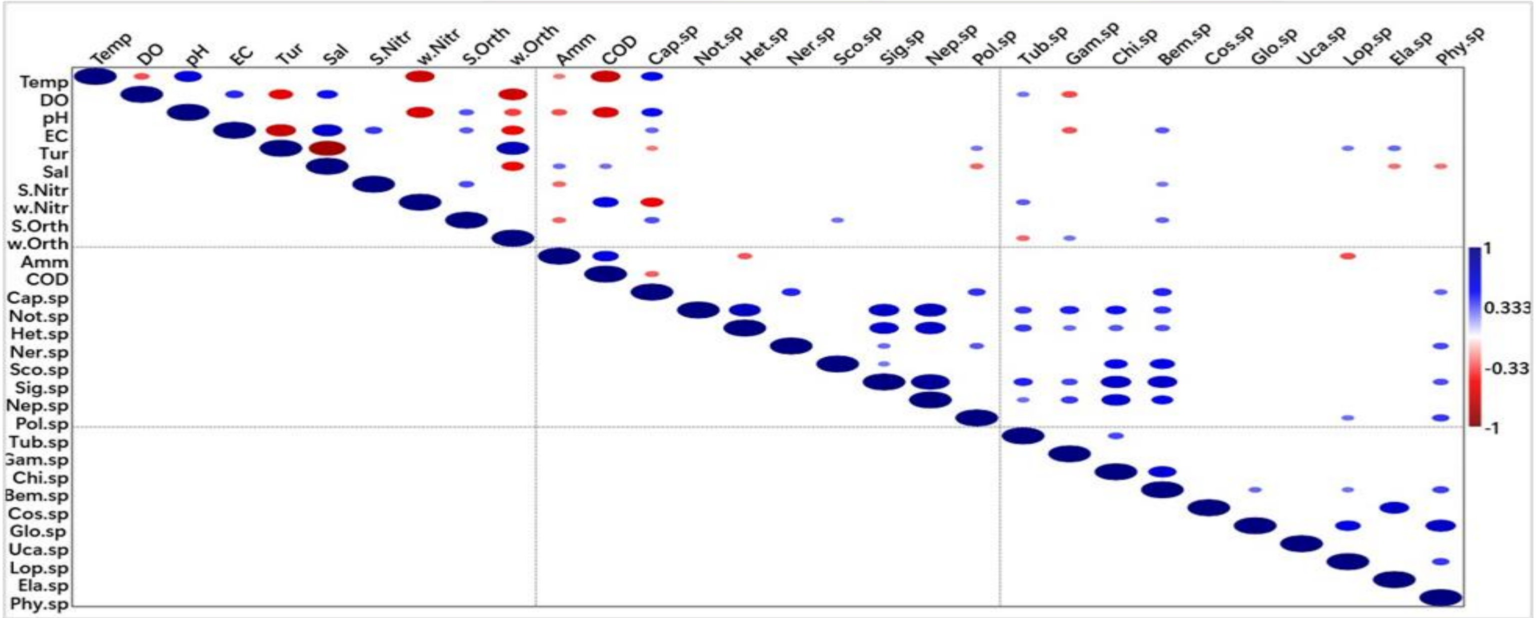


Figure 17: Spearman's rank order correlation for Kakum Estuary

***Note**: In the correlation matrices, deep blue and deep red colour indicate positive and negative correlations, respectively. The deeper the colour the stronger the correlation. The circles indicate p<0.05 and the larger the size of the circle the stronger the correlation.* ***Abbreviations**: Temp-temperature, DO-dissolved oxygen, EC- electrical conductivity, COD-chemical oxygen demand, Turb-turbidity, Sal-salinity, w-Nitr-nitrogen-nitrogen in water, S-Nitr-nitrogen-nitrogen in sediment, w-Orth-orthophosphate in water, S-Orth-orthophosphate in sediment, Amm-ammonium-nitrogen, Cap-Capitella, Not-Notomastus, Het-Heteromastus, Ner-Nereis, Sco-Scoloplos, Sig-Sigambra, Neph-Nephtys, Poly-Polyphysia, Tubi-tubifex, Gam-Gammarus, Chir -Chironomous, Bem -Bemlos, Cos-Cossura, Glossi -Glossiphonia, Lop- Lopadorrhynchus, Elas-Elasmopus, Phyll-Phyllodoce.*

Furthermore, in the Kakum Estuary, more associations in the interest of the present study were documented between physicochemical parameters and benthic macroinvertebrates. For instance, a weak positive but significant correlation (r=0.284) was observed between DO and *Tubifex* sp. Also, significant moderate correlations existed between EC and *Capitella* sp. (r=0.307) as well as EC and *Bemlos* sp. (r=0.332). Turbidity and COD were weakly (r = -0.270) and moderately (r = - 0.323) correlated with *Capitella* sp. The two correlations were both negative but significant at p = 0.05, Figure 4.10.

In the same Kakum Estuary, turbidity correlated weakly but positively with both Polyphysia sp. (r=0.274) and Lopadorrhynchus sp. (r=0.276). However, it showed a moderate but positive correlation with Elasmopus sp. (r=0.302). All the correlations between benthic fauna and turbidity were significant at p=0.05. Just like turbidity, salinity displayed significant associations with benthic fauna at p=0.05. While salinity and Polyphysia sp. correlated moderately and negatively (r = -0.315), weak negative associations were observed between salinity and both Elasmopus sp. and Phyllodoce sp. (r =0.29). Moreover, nutrients correlated significantly with benthic fauna at p=0.05, with nitrates and orthophosphates demonstrating positive correlations while negative correlations were observed in ammonium-nitrogen. Bemlos sp. was weakly and moderately correlated with nitrate-nitrogen (r=0.275) and orthophosphates (r=0.315), respectively. In addition to that, weak positive correlations were recorded between orthophosphates and Scoloplos. sp (r= 0.285) as moderate negative associations existed between ammonium-nitrogen and Heteromastus sp. (r = -0.338), Figure 17.

In the Volta Estuary, the only correlations between physicochemical parameters and benthic macroinvertebrates existed between *Tubifex* sp. and turbidity (r=-0.278) as well as *Tubifex* sp. and orthophosphates in the water column (r=-0.312). Although the correlations were negative and significant at p=0.05. In Figure 18, the other associations were among individual physicochemical parameters as well as among individual macroinvertebrate fauna.

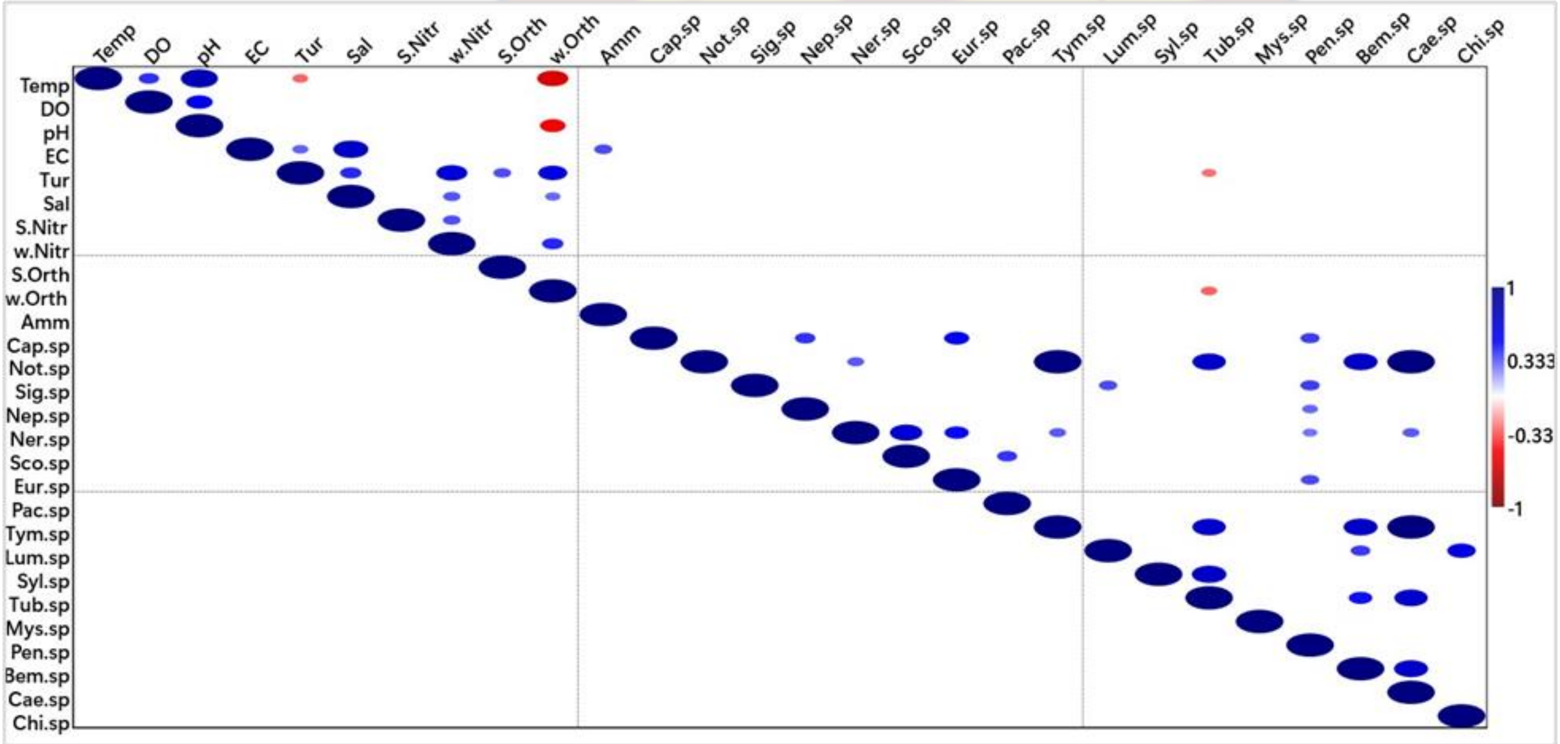


Figure 18: Spearman' rank order correlation for Volta Estuary

Note*: In the correlation matrices, deep blue and deep red colour indicate positive and negative correlations, respectively. The deeper the colour the stronger the correlation. The circles indicate p<0.05 and the larger the size of the circle the stronger the correlation.* ***Abbreviations****: Temp-temperature, DO-dissolved oxygen, EC- electrical conductivity, COD-chemical oxygen demand, Turb-turbidity, Sal-salinity, w-Nitr-nitrogen-nitrogen in water, S-Nitr-nitrogen-nitrogen in sediment, w-Orth-orthophosphate in water, S-Orth-orthophosphate in sediment, Amm-ammonium-nitrogen, Cap-Capitella, Not-Notomastus, Sig-Sigambra, Neph-Nephtys, Ner-Nereis, Sco-Scoloplos, Eur-Eurydice, Pach-Pachymelania, Lumbr-Lumbriconereis, Syll-Syllis, Tubi-Tubifex, Mys-Mysis, Pen-Penaeus, Bem -Bemlos, Caen-Caenobita, Chiron -Chironomid.*

In the Ankobra Estuary, salinity depicted a negative correlation with *Penaeus* sp. as seen in Figure 19. However, the correlation was weak (r=-0.293) whereas no other significant associations existed between physicochemical parameters and faunal specimen in Ankobra Estuary.

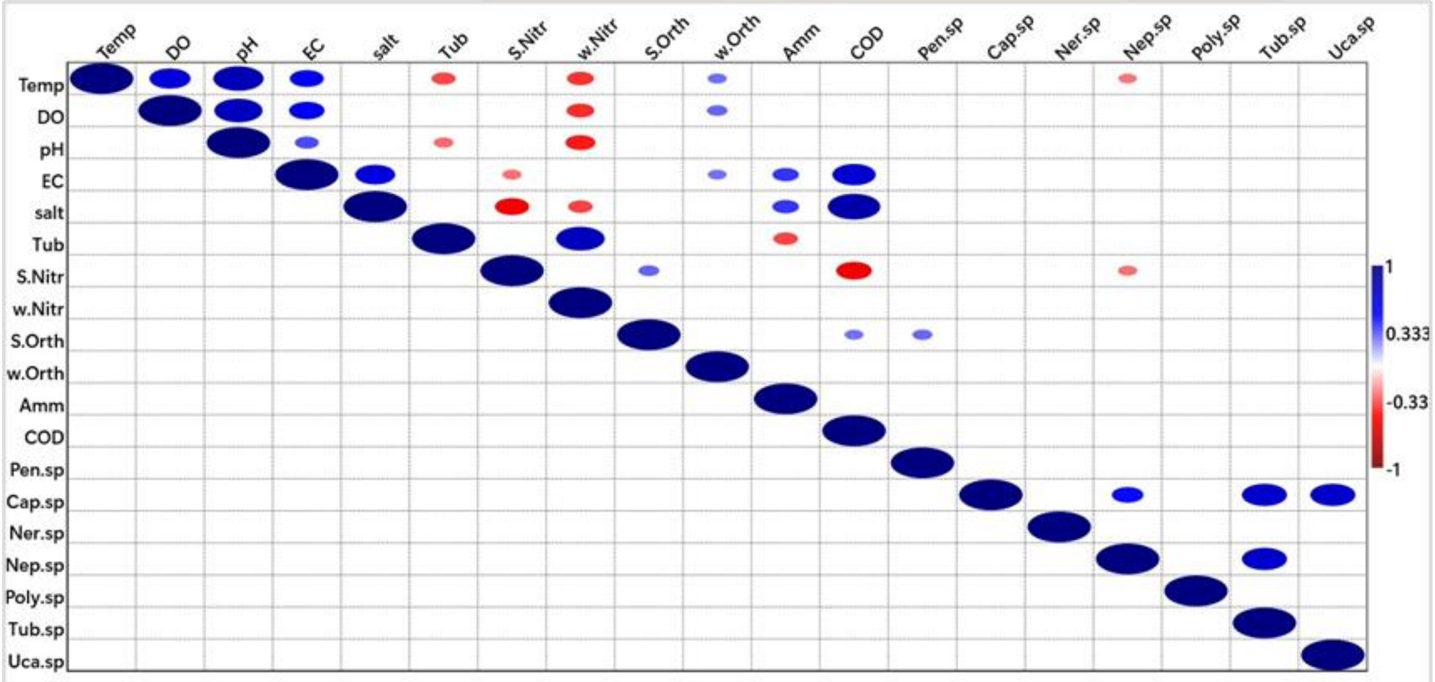


Figure 19: Spearman' rank order correlations for Ankobra Estuary

Note*: In the correlation matrices, deep blue and deep red colour indicate positive and negative correlations, respectively. The deeper the colour the stronger the correlation. The circles indicate p<0.05 and the larger the size of the circle the stronger the correlation.*
Abbreviations*: Temp-temperature, DO-dissolved oxygen, EC- electrical conductivity, Tub-turbidity, Salt-salinity, w.Nitr-nitrogen-nitrogen in water, S.Nitr-nitrogen-nitrogen in sediment, w.Orth-orthophosphate in water, S.Orth-orthophosphate in sediment, Amm-ammonium-nitrogen, COD-chemical oxygen demand, Pen-Penaeus, Cap-Capitella, Ner-Nereis, Neph-Nephtys, Poly-Polyphysia, Tub-Tubifex.*

4.4 Discussion

4.4.1 Physicochemical parameters

The most restricting environmental elements in aquatic habitats include physicochemical parameters such as temperature, DO, EC, pH, and turbidity (Lawson, 2011). The distribution of organisms, as well as other processes regulating metabolism and other changes in water bodies are all significantly influenced by SWT (Smith, 2004). The relatively low SWT in Kakum (27.9˚C) and Ankobra (27.6˚C) Estuaries could be due to shading from mangrove trees observed at either side of the estuarine banks. Kakum Estuary has been listed as the most diverse mangrove forest in Ghana (Dali, 2023). The SWT in the current study is within the range of values reported in other studies within Ghana. For instance, similar findings of SWT by authors like Adjei-Boateng *et al.* (2010) and Madkour *et al.* (2011) (Volta Estuary), Dzakpasu *et al.* (2015) and Fianko *et al.* (2007) (Kakum Estuary) and Faseyi *et al.* (2022) and Soetan *et al.* (2021) (Ankobra Estuary) corroborate the findings in the current study.

Relatively low DO concentrations were recorded in Whin (4.7 mg/L) and Ankobra Estuaries (5.4 mg/L). This could be attributed to high siltation from gold mining activities in the catchment area, which affects sunlight penetration hence reducing photosynthetic activities that consequently affects DO supply. On the other hand, the maximum DO levels in the Volta Estuary (6.1 mg/L) were attributed to the limited amount of silt and suspended matter that reaches the estuary, which is a result of the river being dammed at Akosombo. This results in limited biodegradation and organic decomposition activity. Additionally, there are fewer impacts, especially from gold mining, which is much less common in the Volta Estuary compared to the Ankobra and

Whin Estuaries. Similar findings have been reported in the literature, for instance, Dzakpasu *et al.* 2015 in Kakum Estuary and those of Adjei-Boateng *et al.* (2010) and Madkour *et al.* (2011) in the Volta Estuary fall within the ranges (Table 12) of the findings in the current study. Contrary to this, wider DO ranges were recorded in Pra Estuary (Okyere, 2019) and some coastal waters in Nigeria (Abdus-Salam *et al.*, 2010). Concentrations of DO under 5 mg/L may have a major impact on the survival and normal functioning of biological communities while DO levels below 2 mg/L may result in hypoxic conditions (Rogers *et al.*, 2016). According to the Ghana Raw Water Criteria and Guidelines, the Target Water Quality Range (TWQR) of DO for sustaining aquatic life is 5.0-8.0mg/L (WRC, 2003). The fact that there was no significant difference in the median DO concentration in both Whin (4.7 mg/L) and Ankobra (5.4 mg/L) Estuaries at the time of sampling, aquatic life in the two waterbodies are adapted to survive in low DO environment. However, the DO values in Kakum and Volta Estuaries are suitable for aquatic life in this circumstance.

Electrical conductivity refers to the capacity of water to conduct electric current based on the amount of ionized compounds present in it (Radojevic & Bashkin, 2006). According to the TWQR of the Ghana Raw Water Quality Criteria and Guidelines, any water with an EC of 0 µS/cm to 70 µS/cm (Darko *et al.*, 2013) is best for preserving the well-being of aquatic ecosystems. Results from the current study indicate that the median EC values recorded in the four estuaries were significantly above the maximum limit of TWQR range. Also, reports from other studies in similar environments that corroborate the current study include Aheto *et al.* (2011) in Whin Estuary, Okyere *et al.* (2011) in

Kakum Estuary, Faseyi *et al*. (2022) in Ankobra and Pra Estuaries. Nevertheless, contrary to the current study, lower EC values were reported by Adjei-Boateng *et al*. (2010) and Madkour *et al*. (2011) in the Volta Estuary. The findings of this study suggest that the four estuaries have significant concentrations of dissolved ions, including inorganic salt and organic litter which may be due to domestic wastewater discharges and surface runoffs from cultivated fields in the catchment.

When combined with other environmental factors, the pH of water can have a significant impact on aquatic life due to the acidity or alkalinity of waterbodies. According to National Estuarine Research Research (NERR), 1997), aquatic life is said to be most adaptive in a pH range of 5 to 9. The acidic or alkaline state of estuaries could be influenced by the amount of alkaline ions in seawater from the bedrock of the waterbody (Dzakpasu *et al*., 2015). The present study recorded wider pH ranges, suggesting a potential buffering impact of seawater. Similar observations have been made in Tapi Estuary, India (Nirmal *et al*., 2009). In a different view, other studies have reported narrow pH ranges (Faseyi *et al*., 2022) and (Tufuor *et al*., 2007) in the Pra Estuary, (Adjei-Boateng *et al*., 2010) and (Madkour *et al*., 2011) in the Volta Estuary and (Dzakpasu *et al*., 2015) in Kakum Estuary. The results reported in the current study indicate that the estuaries can support aquatic life.

High turbidity in water could result from siltation, watershed runoffs, aquatic weeds and other organic compounds produced by dead and decayed plant matter which gives waterbodies a rust-red colouration (EPA, 1999). In this study, the maximum turbidity levels were above 3000 NTU in the Ankobra Estuary and as low as 0.0 NTU in the Volta Estuary. The turbidity levels above

3000 NTU, which is of concern in the Ankobra Estuary may be due to significant amounts of silt inflow from upstream regions where illegal gold mining activities occur. Consequently, high turbidity increases the heat absorption capacity of water, leading to higher temperatures that subsequently lower the concentration of oxygen and ultimately affect primary productivity. Also, it impairs biological activities by decreasing disease resistance and clogging fish gills (Faseyi *et al.*, 2022). High turbidity also induces cloudiness in the water and decreases visibility, which hinders breeding, feeding, reproduction, and ultimately the survival of aquatic life (Bilotta & Brazier, 2008; Okyere *et al.*, 2019). The turbidity in the current study was found to be comparably higher than what was recorded in earlier works in the same estuary (Faseyi *et al.*, 2022; Soetan *et al.*, 2021). This pattern indicates increasing impacts of siltation with time as a result of illegal mining activities. Likewise, the results of the current study support those from the Pra Estuary, where turbidity levels above 1000 NTU have been reported (Okyere, 2019). According to Bilotta & Brazier, (2008), the maximum turbidity for aquatic life is 100 NTU, whereas turbidity levels over 500 NTU are considered harmful to aquatic life (I. Okyere, 2019). With the exception of Ankobra Estuary, the turbidity for Whin, Kakum and Volta Estuaries in the current study are suitable for supporting aquatic life.

Although high nutrient concentrations in estuaries increase primary productivity, high turbidity levels in estuaries make it difficult for light to penetrate the water column. During periods of intense rainfall, significant volumes of deposited nitrates in soils from industrial and agricultural operations are carried into estuaries, and this results in increased turbidity (Okyere (2019;

Iida & Shock, 2007). In the current study, the concentration of NO_3-N in the water column was lower than in sediment matrices in the four estuaries. This is due to the fact that, sediments offer a larger surface area and porous environment that allows for the retention of nitrates. Particularly, nitrates can bind to sediment particles, making it less prone to rapid removal through processes like denitrification or assimilation by organisms in the water column (Sanders & Laanbroek, 2018). According to NOAA/EPA (1988), NO_3-N concentration of 1.0 mg/L in estuaries is recommended to prevent algal blooms. However, all the estuaries recorded NO_3-N concentration above this limit both in the sediment matrices and in the water column. A similar finding of high NO_3-N concentrations has been reported in Pra Estuary (Tufuor et al., 2007). On the other hand, lower concentrations below 2.0 mg/L were reported from similar waterbodies by other studies, including Faseyi *et al.* (2022) in the Pra and Ankobra Estuaries, and CRC/FoN (2010) in the Ankobra Estuary.

Furthermore, the concentration of orthophosphates was generally higher in sediment matrices than in the water column in Whin, Kakum and Volta Estuaries. Sediments can host specific microbial populations that contribute to phosphorus transformations, such as phosphorus-solubilising bacteria and phosphorus-accumulating bacteria. These microbes can alter the balance of phosphorus species in sediments, leading to higher concentrations relative to the water column (Howell, 2010). The average orthophosphate concentration in the four estuaries was above the recommended 0.1 mg/L for estuaries and other coastal ecosystems (NOAA/EPA, 1988). These results contradict what Tufuor *et al.* (2007) and Faseyi *et al.*(2022) reported in the Pra Estuary as well as reports from the collaborative report from CRC/FoN (2010) in the Ankobra Estuary where lower values were recorded.

In comparison to NO_3-N and orthophosphates, NH_4-N levels were higher (≥39 mg/L) in all estuaries. In areas of low DO concentration, NO_3-N is easily transformed to NH_4-N and this could be true for the current study since relatively low DO values (≤6.1 mg/L) were recorded in the estuaries. Moreover, elevated levels of NH_4-N in waterbodies point to potential presence of contaminants from human or animal waste, hence a danger to human and animal health (WHO, 2003). Also, NH_4-N in elevated concentrations is toxic to aquatic life. It poses a challenge to be efficiently excreted, hence the build-up in tissues of organisms could lead to death (EPA, 2023a).

4.4.2 Occurrence and composition of benthic macroinvertebrates

Annelids, crustaceans, and molluscs are the three most common macroinvertebrates found in estuaries (Balogun *et al.*, 2011). It has been demonstrated that these groupings accurately represent a variety of aquatic environments in West Africa, including estuaries and lagoons (Adam *et al.*, 2019). In the current investigation, 28 taxa belonging to 26 families and six classes were encountered, summing up to 2993 specimens collected during the sampling period. Individually, the four estuaries recorded 18, 18, 17 and 7 taxa, respectively, corresponding to Kakum, Whin, Volta and Ankobra Estuaries. Individually, these numbers of taxa encountered in Whin, Kakum and Volta Estuaries are comparable to the 17 taxa reported in the Gambia River Estuary (Adam *et al.*, 2019). On the contrary, the summation of all the taxa in the four estuaries is comparatively lower than the 87 taxa reported in Brazilian tropical Estuaries (Adam *et al.*, 2019; Dzakpasu, 2019; Okyere & Nortey, 2018).

In all the four estuaries, polychaete worms were the most dominant taxa especially *Capitella* sp. and *Nereis* sp. Both fauna occupied only the upstream of Ankobra Estuary, they occurred in all the stations (upstream, midstream and downstream) in Whin, Kakum and Volta Estuaries. Some taxa such as *Capitella* sp., *Nereis* sp. and *Nephtys* sp. were found to be ubiquitous in all the four estuaries. A similar observation where polychaetes dominate estuarine water have been documented (Adam *et al*., 2019). In the Ankobra Estuary, the most abundant taxa were crustaceans of the *Penaeus* sp, which occupied all the stations sampled from upstream to downstream, but their occurrence dominated the upstream section of the estuary.

Pollution tolerant species like *Capitella* sp. have been reported to tolerate low oxygen conditions hence considered an indicator of organic pollution (Dean, 2008; Méndez *et al*., 2013). Additionally, *Nereis* sp. and *Nephtys* sp. have been observed to tolerate wide ranges of salinity (Woke & Wokoma, 2007) and according to Balogun *et al.* (2011), *Nereis* sp. can indicate pollution in an aquatic environment. *Cossura* sp., with one and five specimens recorded in Kakum and Whin Estuaries, respectively, has been designated as pollution tolerant and its presence or absence can be used to determine ecosystem health (Dean, 2008; Woke & Wokoma, 2007).

Chironomous sp. was found in high abundance in all the stations of Whin and Kakum Estuary, with only one specimen in the upstream of the Volta Estuary and zero occurrence in Ankobra Estuary. *Chironomous* sp. has been observed to survive high levels of organic pollution and low DO environments (Aggrey-Fynn *et al*., 2011; Lencioni *et al*., 2012; Rafia & Ashok, 2014; Sharma & Chowdhary, 2011).

Additionally, *Tubifex* sp. occurred in all stations of Whin (35 specimens), Kakum (9 specimens) Estuaries, and only three and one specimen in the entire Volta and Ankobra Estuaries, respectively. Studies indicate that *Tubifex* sp. can withstand and thrive in severely enriched environments with organic pollution (Barrilli *et al*., 2021; Bouchard, 2004; Lencioni *et al.*, 2012). The dominance, abundance and presence of pollution tolerant taxa like *Capitella* sp., *Nereis* sp., *Heteromastus* sp., *Tubifex* sp., *Cossura* sp. and *Chironomous* sp. in Kakum and Whin Estuaries is an indication of organic pollution. This study is in agreement with other studies that have demonstrated the presence of pollution indicator species in brackish water in West Africa (Adam *et al*., 2019; Aggrey-Fynn *et al*., 2011; Dzakpasu *et al.*, 2015; Okyere & Nortey, 2018).

Scoloplos sp. was encountered in relatively high abundance in the Volta Estuary (63 specimens), low abundances in Whin (14 specimens) and Kakum (four specimens) Estuaries, and zero abundance in Ankobra Estuary. *Scoloplos* sp. has been pointed out to be sensitive to pollution (Belan, 2003). This could be indicative of a conducive ecosystem health in the Volta Estuary in comparison to the other estuaries where the conditions are not conducive for survival of the organism. Additionally, *Gammarus* sp., which was particularly recorded in Kakum Estuary in low abundance (26 specimens, 18 of which occurred in the downstream), and zero occurrence in the three other estuaries, has been observed to dominate less polluted waters hence relatively sensitive to pollution (Bloor & Banks, 2006). The low abundance of *Gammarus* sp. in Kakum Estuary and zero occurrence in the other estuaries and could point to some level of pollution in the estuaries.

Tympanotonus sp. and *Pachymelania* sp. (gastropods) were documented in the Volta Estuary and seemed endemic to this estuary. Although *Tympanotonus* sp. has been reported to be pollution tolerant (Nkwoji *et al.*, 2020; Onyena *et al.*, 2021), *Pachymelania* sp. are sensitive to environmental changes including the presence of contaminants in water (Balogun *et al.*, 2011; Nkwoji *et al*., 2020). The occurrence of *Pachymelania* sp in Volta Estuary could be as a result of existence of food resources such as detritus for their survival and less predation pressure. Consequently, the absence of *Pachymelania* sp. in Kakum, Whin and Ankobra Estuaries is indicative of contamination in these three estuaries.

Some polychaetes in the Lumbrineridae family are negative indictors of poor benthic conditions and their absence in a community. They are sensitive to changes in environmental conditions, particularly pollution levels, sediment quality and oxygen levels. Their absence, presence and abundance can provide valuable insights into the health of a water body. High abundance of *Lumbriconereis* sp. may indicate good water quality while low or zero abundance is an indicator of poor environmental conditions (Dean, 2008). With eight specimens of *Lumbriconereis* sp. documented in the Volta Estuary, two specimens in Whin and zero occurrence in Kakum and Ankobra Estuaries, it is an indication that all the estuaries have poor environmental conditions, except Volta with relatively good conditions.

The presence of pollution sensitive taxa like *Scoloplos* sp., *Eurydice* sp., *Lumbriconereis* sp. and *Pachymelania* sp. in the Volta Estuary could be an indication of a relatively less polluted environment. The high abundance of *Penaeus* sp. in Ankobra Estuary could be explained by their ability to tolerate wider salinity ranges of whereas other organisms could not (Li *et al.*, 2021), hence could be used as an indicator of salinity. The low abundance of benthic macroinvertebrates in Ankobra Estuary in general is attributed to elevated turbidity levels that increases siltation. On the contrary, in the Volta Estuary, the water is free from turbidity and siltation, attracting largely pollution sensitive organisms. In relation to various stations, most of the organisms occurred in the upstream along the banks with majority hiding in the sediments obtained from rock bottoms, in the mangroves and other objects like logs, decreasing towards the sea side. This could be explained by the unstable salinity gradient towards the sea side, as well as the swift flow of water. These observations corroborate the findings made in Northwest Florida Estuary (Nestlerode *et al.*, 2020).

4.4.3 Diversity of benthic macroinvertebrates

Environmental factors and stressors present in a particular area have a significant impact on the number and diversity of benthic macrofauna (Arslan *et al.*, 2007). The structure, distribution, diversity, and composition of the benthic macroinvertebrate communities are also influenced by habitat physiographic features and microhabitat variety (Aggrey-Fynn *et al*., 2011). In the current study, the Ankobra Estuary recorded the lowest species diversity, richness, and evenness. On the other hand, high species diversity and evenness occurred in Kakum Estuary, with similar species richness in Kakum and Whin estuaries.

Generally, high species richness was found in Whin (d=17.86) and Kakum (d=17.87) Estuaries with lowest richness occurring in Ankobra Estuary (d=6.8). It has earlier been observed that Whin and Kakum Estuaries have records of high abundance of pollution tolerant taxa, which explains the high species richness. In Ankobra Estuary, the fine sediments have high organic matter content hence increased bacterial decomposition that limits oxygen levels. This coupled with limited food availability and unstable sediments makes it less habitable, contributing to low species richness (Fuller *et al.*, 2021). The species richness values in this study are comparatively higher than the maximum values obtained for species richness in Pra River Estuary (Okyere & Nortey, 2018). Moreover, species richness was relatively highest in the upstream of all the estuaries with the general trend indicating low species richness in the midstream of all estuaries apart from Whin Estuary, where similar species richness was recorded in both midstream and downstream. As seen earlier, the highest species abundance occurred in the upstream, which explains high species richness in the same station. The upstream is prone to less wave action, hence the less disturbance allow thriving of species and similar observations were made in Nyan and Kakum Estuaries (Dzakpasu *et al.*, 2015).

The highest species evenness was encountered in Kakum Estuary (J'=0.79) and lowest in Ankobra Estuary (J'=0.17). In Whin, Volta and Ankobra Estuaries, species evenness was highest upstream while in Kakum Estuary, it was highest in the downstream. However, in all the stations, individuals were fairly distributed among the species (J'≥0.7) in all the estuaries. Similar observations with J'≥0.6 were made in Pra Estuary (Okyere & Nortey, 2018). The distribution of species on a spatial scale in individual estuaries showed a wide range in Volta (0.1-0.8) and Ankobra (0.0-07) Estuaries and a narrower range in Whin (J'=0.7-0.9) and Kakum (J'=0.7-0.8) Estuaries.

In previous studies, narrower ranges have been reported in Kakum (J'=0.68-0.73) and Nyan (J'=0.78-0.80) Estuaries (Dzakpasu *et al.*, 2015).

Species diversity largely depends on species richness, abundance and distribution in an ecosystem. In the current study, Kakum Estuary, which had the highest species evenness and richness, recorded the highest species diversity (H'=2.29) while the Ankobra, with lowest species evenness and richness, had the lowest species diversity (H'=0.33). Due to the dynamic nature of estuaries, there was spatial differences in species diversity within specific estuaries. In Volta and Ankobra Estuaries, species diversity increased from downstream to upstream, while in Kakum Estuary, diversity decreased towards the downstream with Whin Estuary depicting similar diversity both downstream and upstream.

The findings of the current study in Kakum Estuary where diversity, H'=2.29 are similar to the highest value obtained in Pra Estuary, H'=2.3. However, the lowest value recorded in Ankobra Estuary in the current study is lower than what was found in Pra Estuary, H'=1.0 (Okyere & Nortey, 2018). Moreover, the results from Kakum Estuary indicating the least diversity downstream are similar to what Dzakpasu *et al.* (2018) found in Kakum and Nyan Estuaries. Generally, the highest species diversity in the current study in specific stations (Kakum downstream, H'=2.2) is higher than the findings in the Gambia River Estuary; H'<2, (Adam *et al.*, 2019) and lower than those in the Negombo Estuary; H'=3.72 (Dahanayaka and Aratne, *cited in* Dzakpasu *et al.*, 2015).

In general, high diversity index values above 3 (H'>3) point to a stable, balanced habitat, while values below one (H'>1) are indicative of a polluted and degraded habitat (Turkmen & Kazanci, 2010). Pielou's evenness index, on the other hand, ranges from 0 to 1, and the closer it is to 1 the more evenly distributed a habitat is (Pielou, 1966). Since the Margalef index has no upper bound, it is used for spatial comparison (Kocataş, 1992). Species diversity, $H' \leq 3$, species evenness, $J' \leq 0.8$, and species richness, $d \leq 18$ for the four estuaries is not necessarily indicative of contamination, instability and imbalance. Estuaries are very dynamic ecosystems, and very few taxa are able to adapt to the frequently changing environmental conditions and cope with the fluctuating environmental stress. Species diversity, since it encompasses species richness and species richness, is a prime aspect of biological monitoring and is considered a valuable parameter in determining ecosystem health (Marques, 2001). Therefore, ranking of the four estuaries in terms of stability using the biological indices implies that Kakum Estuary is ecologically healthier than Whin Estuary, which is healthier than Volta Estuary that is in turn healthier than Ankobra Estuary, ie., Estuarine ecological health ranking; Kakum >Whin> Volta > Ankobra Estuary.

4.4.4 Species-environment interactions

Physicochemical factors can have a favourable or detrimental impact on the entire benthic population in any aquatic habitat, depending on their sources. The richness, abundance and composition of benthic communities can change over time due to changes in physicochemical factors. A diverse population of benthic fauna shows that the water quality is appropriate for their existence in the entire ecosystem. The abundance and diversity of macroinvertebrates in aquatic environments have been linked to some parameters such as temperature, salinity, DO, nutrient concentrations, pH, turbidity, and organic matter content (Mistri *et al.*, 2000).

Salinity is one of the key determinants of macroinvertebrate diversity and abundance in Ghanaian coastal waters and benthic macroinvertebrate communities react differently to variations in water salinity (Lamptey & Armah, 2008). In the Kakum Estuary, some polychaete taxa like *Polyphysia* sp. and *Phyllodoce* sp. demonstrated moderate and weak negative correlations, respectively, with salinity. Furthermore, some crustaceans like *Elasmopus* sp. and *Penaeus* sp. showed significant but weak negative correlations with salinity in Kakum and Ankobra Estuaries, respectively. The findings are contradictory to the fact that polychaetes as deposit feeders increase in abundance with salinity, but agrees with observations that suspension feeders like crustaceans decrease in abundance with salinity (Kim & Montagna, 2012). Tachet *et al.*, (2010) points out that lower salinity levels favour the development of sensitive species like *Penaeus* sp. In Ankobra Estuary, high abundance of *Penaeus* sp. occurred upstream (146 specimens, relative to three specimens recorded midstream and five specimens downstream), indicating high abundance with decreasing salinity. Given the euryhaline nature of *Penaeus* sp., (Li *et al.*, 2021) and its high percentage composition (94 %) in Ankobra Estuary, the taxon can be considered an indicator taxon of salinity.

The DO concentration threshold that supports aquatic life 5.0 mg/L to 8.0 mg/L (WRC, 2003). In aquatic life settings where DO concentration ranges from 3.5 mg/L to 6.5 mg/L, a greater macroinvertebrate fauna is favoured (Okyere & Nortey, 2018). However, some benthic faunae are able to thrive well in heavily organically contaminated environments such as organisms in the families of Tubificidae, Capitellidae and Cirratulidae, indicating stressed communities (Dean 2008). In the Whin Estuary, DO negatively correlated with

Chironomous sp. (r=-0.277). Given that *Chironomus* sp. is able to survive in organically contaminated low DO environments (Lencioni *et al.*, 2012), *Chironomous* sp.occured in all stations (upstream, midstream and downstream).

These findings agree with the fact that the lowest DO concentration was encountered in this particular estuary. The negative corelation is an indication of an organically contaminated low DO environment that supports a high abundance of pollution tolerant taxa like *Chironomous* sp. in the Whin Estuary. Also, in the Kakum Estuary, *Tubifex* sp. demonstrated a weak but significant positive correlation with DO (r=0.284). Studies have indicated that *Tubifex* sp. can reach very high densities in organically polluted systems as they feed on organic matter from oxygen-poor sediments (Barrilli *et al.*, 2021; Okyere & Nortey, 2018). A positive correlation between *Tubifex* sp. and DO with only nine specimens of *Tubifex* sp. occurring in the entire Estuary with even distribution from upstream to downstream could be indicative of moderate organic pollution. Other studies have recorded contradicting results from the findings in the current study, indicating a negative correlation between *Tubifex sp.* and DO (Barrilli *et al.*, 2021; Ertaş & Yorulmaz., 2021)

Electrical conductivity is the ability of water to conduct electric current and is an index of both dissolved ions and salts in water (Radojevic & Bashkin, 2006). Increased EC in water could have been as a result of high surface run-off of organic debris from anthropogenic sources like domestic sewage since the Kakum river drains a densely populated and highly urbanised Central Region of Ghana. Electrical conductivity positively and significantly showed moderate correlation with *Capitella* sp. in the Kakum Estuary (r=0.307). With a relatively high EC (7580 µS/cm), Kakum Estuary can be said to be contaminated with dissolved substances that extend further upstream given the small distance covered by the estuary (1.2 km). *Capitella* sp. occurred in all the stations but due to its ability to survive contaminated waters, the highest abundance occurred upstream. Similar findings where EC positively and moderately correlated with *Capitella* sp. in coastal waters have been reported (Andem *et al*., 2013; Fernández-Rodríguez *et al*., 2016).

Effects of turbidity on benthic macroinvertebrates have been demonstrated (Faseyi *et al*., 2022; Okyere, 2019). The outcome of the effects of turbidity on benthic macroinvertebrates is probably the reason for the negative correlation between turbidity and *Capitella* sp. despite their ability to colonise organically enriched environments. High abundance of *Capitella sp*. occurred in the upstream of Kakum Estuary due to reduced turbidity in this particular station, which also explains the negative correlations.

Additionally, the positive correlations observed between turbidity and some polychaete taxa like *Polyphysi*a sp. in Kakum Estuary is because, *Polyphysia* sp. is associated with high turbidity areas, which are associated with abundant food resources like detritus and plankton. Some other *Polyphysia* sp. like *Scalibregma crassa*, are known to thrive in areas with moderate to high turbidity, indicating tolerance to high conditions (Okyere & Nortey, 2018. In the Volta Estuary, very low turbidity values recorded (sometimes 0.0 NTU) throughout all the stations. Since *Tubifex* sp. thrives in very turbid and organically contaminated conditions, the Volta Estuary was not favourable for this particular taxon, hence the negative correlation. Also, only three specimens of *Tubifex* sp. were recorded in the Volta Estuary (one specimen upstream and

2 specimens downstream) which explains the negative correlation with turbidity.

High concentration of nutrients in water are a characteristic of diffuse sources of organic and inorganic matter from anthropogenic activities in the catchment (Jun *et al*., 2011). Generally, high nutrient concentration in water can lead to increased primary productivity which may support higher population of benthic macroinvertebrates due to food availability. However, the influence of nutrients on benthic fauna in Kakum Estuary was less impactful as seen from the weak positive correlations between some pollution sensitive taxa like *Bemlos* sp. and *Scoloplos* sp. with NO_3-N and orthophosphates. Also, the weak associations could be related to the very low levels of NO_3-N and orthophosphate recorded in all stations within the Estuary. According to Ertaş & Yorulmaz (2021), the distribution of pollution tolerant taxa positively correlated with NO_3-N, orthophosphate and NH_4-N, and this contradicts the present study as far as NO_3-N and orthophosphate are concerned. These findings in the current study suggest that Kakum Estuary is low on nutrient enrichment.

Tubifex sp. and *Chironomous* sp. have been established as pollution tolerant taxa. Occurrence of only three specimens of *Tubifex* sp. in the Volta Estuary explains the negative correlation with orthophosphates (r=-0.312). On the other hand, high abundance of *Chironomous* sp. evenly distributed in all the sampling stations coupled with high NH_4-N concentration (76 mg/L) explains the positive correlation (r=0.289) between the two variables in Whin Estuary.

The study gives an overview of the spatial distribution of benthic macroinvertebrates within and among four estuaries along the coastal part of Ghana, the implication of existing benthic communities on water quality and their association with physicochemical parameters. Pollution tolerant taxa (*Capitella* sp., *Nereis* sp., *Heteromastus* sp., *Tubifex* sp., *Cossura* sp. and *Chironomous* sp.) occurred in in Kakum and Whin Estuaries. Pollution sensitive taxa (*Scoloplos* sp., *Eurydice* sp., *Lumbriconereis* sp. and *Pachymelania* sp.) occurred in the Volta Estuary while and salinity indicator taxon (*Penaeus* sp.) occurred in Ankobra Estuary. Although Kakum and Whin Estuaries are dominated by a wide range of pollution tolerant taxa, Pearson correlation analysis shows weak and moderate correlations (both positive and negative) in both estuaries, suggesting that they are moderate on organic pollution. Moreover, Kakum Estuary is the most diverse estuary despite the moderate pollution levels. However, in all the four estuaries, the quality of water is a factor of anthropogenic activities in the catchment areas, which has negative implications as expressed through physicochemical parameters like DO, turbidity, nutrients, EC and COD. As seen from literature, estuarine ecosystem health has been indicated using physicochemical and benthic macroinvertebrate data in isolation. The study recommends integration of the various water quality metrics into a model that will provide a holistic view of estuarine ecosystem health, including frequent monitoring of water quality.

4.5 Conclusion

The current study has revealed significant acumens into the presence and extent of pollution in estuarine water along the coast of Ghana. The dominance of pollution-sensitive, pollution-tolerant and salinity indicator species in the various estuaries, in conjunction with their associations with crucial environmental factors indicate variation in levels of organic pollution. In terms of ecological health status, Ankobra Estuary emerged as the least healthy, followed by Volta Estuary, then the Whin Estuary and finally the Kakum Estuary being the healthiest. These results highlight the need to protect estuarine ecosystem health from further degradation with anthropogenic sources of contaminants, especially the Ankobra, Whin and Kakum Estuaries. The findings further emphasise the need to integrate data obtained from benthic macroinvertebrates and physicochemical parameters that indicate the status of water quality into a water quality monitoring model for easy assessment of estuarine ecosystem health in Ghana.

CHAPTER FIVE

AN INTEGRATED WATER QUALITY INDEX FOR MONITORING ESTUARINE ECOSYSTEM HEALTH IN GHANA

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Under review: Taylor & Francis' *African Journal of Aquatic Sciences* Journal (Submitted on 21st October, 2023, submission ID *239125383).*

Statement of Joint Authorship Contributions

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Abstract

Estuarine ecosystems in Ghana display stress symptoms as a result of both point and non-point source pollutants. The degree and extent of pollution in selected Ghanaian estuaries was assessed using a modified Integrated Water Quality Index ($IWQI_{Gh}$) by combining physicochemical and benthic macroinvertebrate indices. Multivariate statistics (Principal Component Analysis and Correlation Analysis) and descriptive statistics were used to reduce 15 physicochemical parameters common among four estuaries to eight in Ankobra, seven each in Volta and Whin, and six in Kakum Estuary. Generally, the selection criteria yielded nine parameters representative of water quality along the coast of Ghana; dissolved oxygen (DO), pH, electrical conductivity (EC), chemical oxygen demand (COD), turbidity, total suspended solids (TSS), NH_4-N, NO_3-N and orthophosphate. Sub-index values and relative weights were generated from mathematical functions incorporating the maximum permissible guideline limits of brackish water ecosystems computed from other studies in similar environments in Ghana while missing information obtained in studies from other tropical countries. The final index was computed by aggregating the sub-indices and relative weights to construct a modified Weighted Arithmetic (WA) WQI using an additive function. Results indicate that the estuaries under study are polluted with $IWQI_{Gh}$ placing them under ecological category 4 "polluted," and nutrients (nitrates, orthophosphate and ammonium), turbidity, EC and COD as the main contributors to high index values. Furthermore, the selected estuaries are dominated by low scoring taxa that were highly tolerant to organic pollution. The results could be attributed to the impacts of human activities like agriculture, sewage disposal and gold mining in the catchment areas of these

water bodies, and acting as the greatest contributors to the deteriorated water quality.

Keywords: Water Quality Index, Estuary, Physicochemical parameters, Benthic Macroinvertebrates, Principal Component Analysis

5.1 Introduction

One of the most complex and highly dynamic ecosystems on earth are estuaries (Vorwerk *et al.*, 2003), occurring at the intersection of freshwater and marine water. Their strategic location (Araújo *et al.*, 2016) and complexity make them relevant in many aspects of usage in provisioning, regulatory, cultural, habitat, and ecological community services (Thrush *et al.*, 2013). Ghana is endowed with more than 10 estuaries along her coastline, providing critical habitats for many fish species as well as wildlife resources that support the country's economy (Aheto *et al.*, 2011). According to Sasu (2022), fishing in Ghana generated 1.1 % of the nation's GDP in 2020, representing close to 1.6 billion Ghanaian cedis (GHc), which is about 263.2 million US dollars.

Among the major threats facing these ecosystems are environmental and anthropogenic factors such as mining, discharge of domestic, industrial and agricultural waste that heavily compromise their health (Thrush *et al.*, 2013). Large- scale exploitation of gold for commercial purposes as well as illegal small-scale gold mining activities play a significant role in compromising the ecological well-being of estuaries (Essumang & Nortsu, 2008). Other threats include over-exploitation of fisheries resources, pollution from both land and sea bed sources, accelerated coastal erosion, habitat loss, climate change and conversion of estuaries into waste dumpsites (DeGraft-Johnson *et al.*, 2010).

One way of doing this is frequent monitoring of estuaries using a Water Quality Index (WQI). This is a tool that describes the overall water quality by combining complex and technical water quality information into a single unitless numerical value (Kachroud, 2019; Lumb *et al.*, 2011; Zeinalzadeh & Rezaei, 2017). It describes water quality status by reflecting the overall impact of multiple Water Quality Parameters (WQPs) and allowing for spatial-temporal comparison of physical, chemical and biological attributes of water. Ghana, just like any other Sub-Saharan African country, does not have a customised WQI to specifically monitor her estuarine water quality.

Estuarine ecosystem health can be monitored by either using physicochemical parameters, bioindicators or incorporating both. For a better understanding of the biological communities and pollution levels, physicochemical factors such as DO, turbidity, Surface Water Temperature (SWT), nutrients, pH, transparency, salinity, chlorophyll-a and heavy metals are evaluated. They are stressor-based methods that assess pollution levels causing ecological imbalances as a result of both anthropogenic and natural sources (Nwanosike *et al.*, 2010; Yisa & Oladejo, 2010). On the other hand, bioindicators such as fish, macrophytes, benthic macroinvertebrates, and phytoplankton are used to predict estuarine ecological conditions by estimating their biomass (Lavoie *et al.*, 2008; Maggioni *et al*., 2009). Use of bioindicators is a response based approach since thy are more expressive than physicochemical parameters (Norris & Morris, 1995). Among them, benthic macroinvertebrates are the most preferred because they are generally simpler, cheaper, and easier to collect and identify using current diversity monitoring indices (Nazarova *et al*., 2004). Furthermore, various benthic macroinvertebrate species respond differently to the adverse impacts of pollution and habitat loss (Nazarova *et al.*, 2004). For instance, pollution tolerant taxa are able to survive and even thrive in elevated levels of pollution and environmental stressors. They include midge larvae (chironomidae), oligochaeta, scuds (amphipoda), copepods, etc., (Barrilli *et al.*, 2021; Zhang *et al.*, 2019). On the other hand, pollution sensitive taxa are highly subtle to environmental pollution and are adversely affected even at low pollutant concentrations. They are often used as bioindicators to assess the overall health and quality of an ecosystem. They include stoneflies (Plecoptera), mayflies (Ephemeroptera), caddisflies (Trichoptera), flatworms (turbellaria), etc., (Karmakar *et al*., 2022).

The concept of WQIs in Ghana is relatively new, dating back to exactly two decades ago when the Water Resources Commission (WRC) produced a document, "Adapted Water Quality Index" and proposed its application for surface water quality. This was pioneered by the works of Ansa-Asare (1998) who adapted and modified the Solway WQI from the Solway River Purification Board (SRPB). The SRPBWQI is a general type of index incorporating physical, chemical and microbiological parameters to produce an overall index of water quality for rivers in the United Kingdom (House, 1989) and has been applied in South African Estuaries (Cooper *et al.*, 1994). The adapted WQI is currently referred to as "Adapted Weighted Raw Water Quality Index (AWQI) for Ghanaian River Systems" (WRC, 2003). The ideas behind its development were the need to share and communicate the technical results from monitoring water with the general public and to provide a general means of comparing and ranking various bodies of water throughout Ghana (WRC, 2003).

Generally, WQIs are designed to be location and source-specific (Lukhabi *et al.*, 2023; Banda & Kumarasamy, 2020b), and it is acceptable to adapt and modify WQIs as long as they conform to the varying regulatory criteria for water agencies in different states (Sutadian *et al.*, 2017). Nevertheless, it is essential to relate the index being modified to local context. This pertains to the original objective for which the index was developed as directed by the parameters in question and specific index usage (Banda, 2015; Banda & Kumarasamy, 2020b).

Failure to observe these, abnormalities such as rigidity, eclipsing and ambiguity come into play in the adaption process, rendering the index less useful and may be deemed dysfunctional. A rigid index is not adaptable enough to include extra or substitute WQPs. This occurs when impairment develops in a parameter excluded from the WQI or when an index is used in a setting with different use objectives for which it was designed (Swamee & Tyagi, 2007). On the other hand, ambiguous indices suggest worse water quality than expected because they obtain sub-index values for all WQPs differently. Finally, eclipsing issues frequently arise when a low sub-index value is masked by a high overall WQI value (Swamee & Tyagi, 2000). Therefore, the above overview exposes an evident knowledge gap that the current study seeks to address. It endeavours to develop a customised WQI that could be used as an assessment tool to provide a standardised way of monitoring Ghana's estuarine ecosystems to support the country's water resource management agenda.

5.2 Materials and Methods

5.2.1 Study location

The study was carried out in four estuaries along the coast of Ghana; Ankobra (West coast), Volta (East coast), Kakum (Central coast) and Whin (West coast) to represent the various sections of the coastline. The Ghanaian coastline is mainly a high energy coast about 540 km stretching from Aflao (Togo border) to La Cote D'Ivoire border, (Figure 20) (Boateng, 2012).

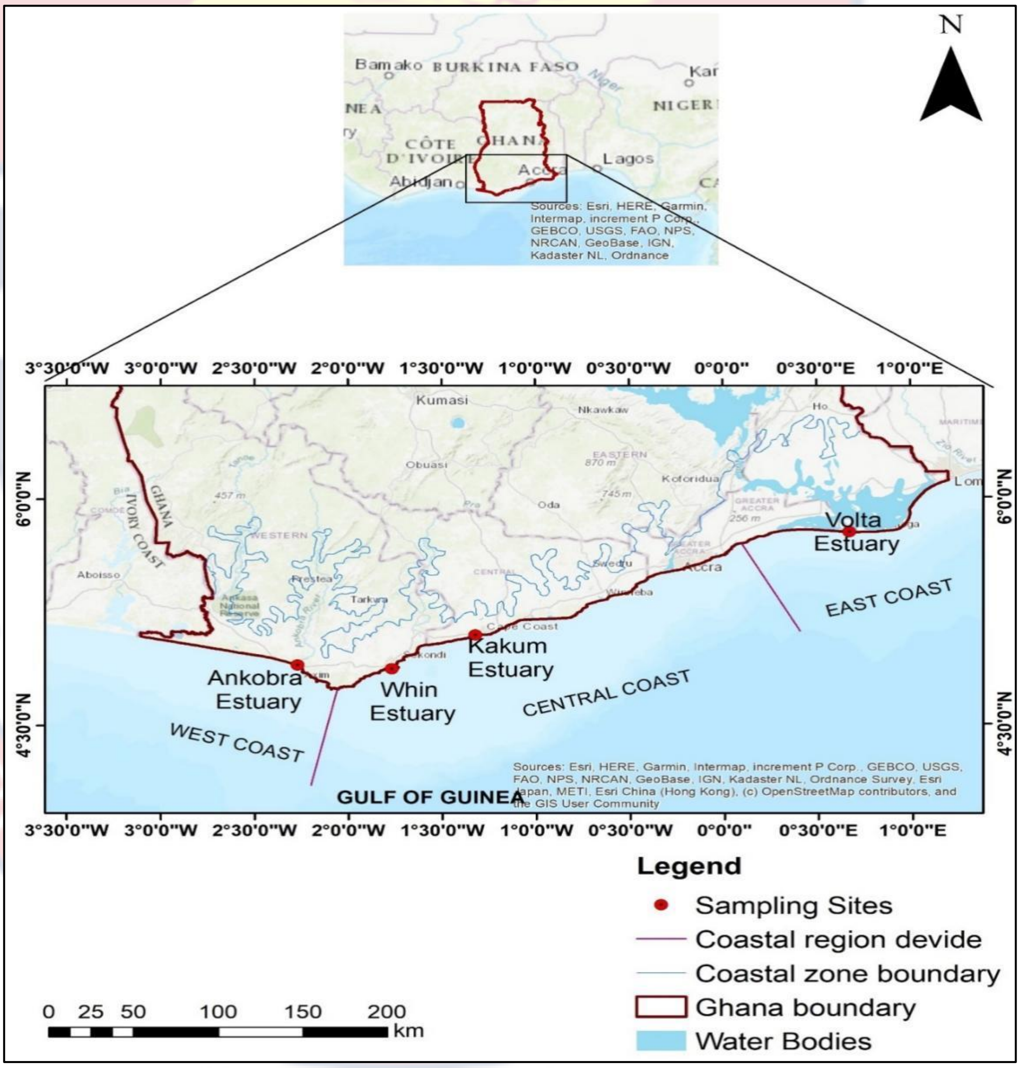

Figure 20: Map of Ghana's coastline showing the study locations
Note: Every sampling site has three sampling stations; upper, middle and lower reaches

The Ankobra Estuary

The Ankobra Estuary lies between latitudes 4°55’N and 4°54’N, and longitudes 2°17’W and 2°15’W. It is approximately 10 km within the mangrove ecosystems and discharges into the Gulf of Guinea at Asanta. It is bound to the east by Nzema East district and to the South by the Gulf of Guinea (Osman *et al.*, 2016). The Estuary forms part of the Ankobra basin, which has a total surface area of approximately 8,400 km^2 and runs through Dompem, Prestea, Bogoso, Asankragua, Awaso, Tarkwa, Egyembra, Esiama and Axim townships. The main economic activities along the drainage system include both legal and illegal gold mining, cash crop farming and fishing. Illegal mining activities from upstream form the major challenge in this estuary, resulting in massive siltation that threatens the estuarine ecosystem health and its biodiversity. Additionally, surface run-offs from upland agricultural fields, as well as residential and municipal effluents contribute to the pollution in this estuary (Okyere & Nortey, 2018).

The Kakum Estuary

This estuary lies between latitudes 5º30’N and 5º47’N, and longitudes 0º 12’ W and 0º 35’ W. It is located along the Cape Coast-Takoradi highway in the Cape Coast metropolis, Central region and within the central section of the coastline. It is formed by the Kakum River and Sweet River, which drain from a rapidly urbanised area of the Central Region. The main community bordering Kakum Estuary is the Iture community, which constitutes part of the Cape Coast-Elmina coastal plain. The major economic activities practiced by communities along the estuary include sand winning and fishing (Fianko *et al.*, 2007).

The Volta Estuary

The Volta Estuary lies between latitudes 5°46'N and 5°48'N and longitudes 0°37'E and 0°41'E, located in the lower basin of River Volta at Ada in Greater Accra region within the coastal savannah zone and at the discharging point into the sea, the estuary is 1.2 km wide. The Volta Riis the largest river basin in Ghana, with a drainage area covering approximately 379,000 km^2. Its quality and quantity are dependent on the Akosombo and Kpong dams built on the river. The Black, White, Red Volta and Oti Rivers constitute the sources of the Volta River, originating from Burkina Faso. The Volta River basin cuts across six riparian countries as its catchment and comprises the major sediment supply to the Gulf of Guinea. The countries are; Burkina Faso (43 %), Ghana (42 %), Togo (15 %), Benin, Cote d'Ivoire and Mali (15 %) (Barry *et al.*, 2005).

The Whin Estuary

It is located in the Ahanta West District between latitudes 4°52'52''N and 4°52'30''N and longitudes 1°46'47''W and 1°46'04''W in the Western region of Ghana in Sekonde-Takoradi (Chuku *et al.*, 2023). Its' size is estimated to be 652,202 km^2, with its banks characterised by thick vegetation with thickets of mangrove stands. It forms a y-shaped structure that pours into the estuary with the longer arm lying on western side of Adakope village while the shorter arm is sandwiched between Adakope and Kokompe on the eastern side of Adakope (Sneli, 2012). The major water sources for the estuary are land drainage, the sea and direct rain water. The major anthropogenic activity in the estuary is fishing (CRC/FoN, 2010).

5.2.2 Sampling

Water and sediment samples were collected from the upper, middle and lower reaches of each estuary. Zonation of the estuaries was based on proximity to riverine ecosystem using observation of mangroves and distance with the aid of handheld Global Positioning System (GPS) gadget. Sampling was conducted during low tides every other month between April 2022 and February 2023 to cover one hydrological year using a tide table (Tides4fishing, 2023).

Water samples were collected at each sampling point in pre-cleaned 350 ml polyethylene bottles for analyses of nutrients (nitrate-nitrogen-NO_3-N, ammonium-nitrogen-NH_4-N, and orthophosphate), Biochemical Oxygen Demand (BOD) and Chemical Oxygen Demand (COD). Similarly, sediment samples were collected in pre-cleaned and labelled plastic containers at each sampling point, one set for nutrient analysis (in sediment matrices) and another set for the analysis of benthic macroinvertebrates (BM). SWT, DO, EC, salinity, total dissolved solids (TDS) and pH were measured in situ using a HORIBA water quality monitor, model U-5000 (JAPAN) with multi-parametric probes. Turbidity and total suspended solids (TSS) were measured by reading their concentration from a pre-calibrated multi-parametric photometer (DR 900). Sediment samples were collected using Ekman grab (15 cm x 15 cm). Three replicate grab samples were taken at each sampling point. The organisms were screened in the field using a set of sieves with mesh sizes of 4.0 mm, 2.0 mm and 0.5 mm. During the sieving process, the larger mesh size sieves were stacked above the smaller ones and organisms retained on the sieves were preserved in 10 % formalin for further laboratory examinations.

5.2.3 Analytical methods

Deionised water was used throughout the sample analysis wherever applicable. Analytical methods used for water samples varied depending on the parameters of interest. All field and laboratory determinations were carried out according to the standard methods (APHA 2018) in Table 15.

Table 15: *Analytical Methods Employed in the Laboratory*

Parameter	Analytical method	Reference
Nutrient extraction	Nitrates: Calcium sulphate extraction Phosphorus: Mehlich 2 extraction	Hach Company, 2001
NO_3-N	UV Spectrophotometric	APHA, 2018
NH_4-N	Nesslerisation	APHA, 2018
orthophosphate	Ascorbic acid	APHA, 2018
BOD	Winker, 5-day incubation at 20°C	University of Idaho, 2023
COD	Closed reflux, titrimetric	APHA, 2018,5220 C

5.2.4 Identification of macroinvertebrates

Preserved sediment samples were stained with eosin dye prior to sorting to enhance visibility. The associated organisms were observed under a dissecting microscope and identified to the lowest possible taxonomic level with the aid of relevant identification manuals and keys (Chapman, 2007; Day, 1967; Edmunds, 1978; Yankson & Kendall, 2001). Counts of different taxa groups were recorded for further analysis.

5.2.5 Water Quality Index Method

The current study modified the Weighted Arithmetic Water Quality Index (WAWQI) method by Brown *et al.* (1970) following the four classical steps involved in index development. These included;

(1) parameter selection

(2) estimation of sub-index values

(3) weighting of parameters

(4) formulating and computing of the overall index (Tyagi *et al.*, 2013; Abbasi & Abbasi, 2012).

Parameter selection

Multivariate statistical methods; Principal Component Analysis (PCA and Correlation Coefficient "r" (CA) and descriptive statistics (% of variance from standard deviation: mean ratio) were used in parameter selection.

Principal Component Analysis is a powerful tool that transforms complex multivariate datasets to a minimal and manageable number of factors without loss of information (Ewaid *et al.*, 2020; Tripathi & Singal, 2019; Quevedo-Castro *et al.*, 2018). Principal Component Analysis does this by preserving and transforming the structure and pattern of the original dataset containing physicochemical and biological parameters to the maximum extent possible (Tripathi & Singal, 2019b).

The multivariate statistical methods followed a stepwise criterion as discussed below;

Identification of initial physicochemical parameters for PCA

A total of 15 physicochemical parameters (SWT, DO, pH, EC, salinity, TSS, TDS, turbidity, NO_3-N (water), orthophosphate (water), NO_3-N (sediments), orthophosphate (sediments), NH_4-N, BOD and COD common to the four estuaries were considered for PCA.

Transformation of parameters

All the parameters were tested for normality and further transformed by calculating their z- scores (Normalisation). The dataset of z-score values of all the parameters from the four estuaries was used in all parameter reduction stages up to the final parameter selection.

Testing parameter suitability for PCA

To examine the suitability of the dataset for PCA, Kaiser–Meyer–Olkin (KMO) and Bartlett's tests of Sphericity were performed. The KMO is a measure of sampling adequacy that indicates the proportion of variance caused by underlying Principal Components (PCs). A higher value (closer to 1) generally indicates that the data set may be excellent to be used for PCA. On the other hand, Bartlett's test of Sphericity examines whether the correlation matrix is an identity matrix. If the correlation matrix is an identity matrix, then all parameters become unrelated making PCA model inappropriate and unsuitable statistical tool for advanced data analysis. The Null Hypothesis of Bartlett's test assumes that Correlation Matrix is an identity matrix (i.e., there is no scope for dimensionality reduction) (Tripathi & Singal, 2019b).

Selection of Factor loadings

PCA was performed on normalised data using Varimax rotation with Kaiser Normalisation separately for each of the four estuaries. Factor loadings were classified as "strong", "moderate", and "weak", corresponding to absolute loading values of > 0.75, 0.75–0.50, and 0.50–0.30, respectively (Liu *et al.*, 2003). These were further subjected to Pearson correlation analysis to obtain the least correlated parameters to further reduce them to a manageable number (Tripathi & Singal., 2019).

Pearson's Correlation Coefficient "r"

The Pearson's correlation analysis was used to generate the correlation matrix and identify the number of possible parameters that would provide the same importance between parameters to be discriminated against. This is because some of the shortlisted parameters were highly correlated hence prone to redundancy, while some could be simply calculated from others. Nevertheless, in scenarios where the highly correlated parameters were extremely important for representing water quality and neither of them could be overlooked, their inclusion was based on author's judgement. Where parameter selection did not satisfy the multiparametric criteria, descriptive statistical data (% of variance from standard deviation: mean ratio) was used for parameter selection, as illustrated in Quevedo-Castro *et al.* (2018). The final selected parameters as a result of statistical tools implementation (PCA, Pearson correlation Coefficient "r" and descriptive statistics) were classified into either of the four categories; physicochemical, particulate matter, nutrients and organic matter.

5.2.6 Development of sub-indices

The ranges of levels to which different parameters can occur vary greatly from parameter to parameter (Abbasi & Abbasi, 2011). Therefore, this step aims to transform the WQPs into a common scale since the actual parameter values have different units. The sub-indices were developed by establishing sub-index functions based on the permissible water quality guidelines calculated from previous studies in brackish water ecosystems in Ghana (Appendix D). However, for the TSS, NH_4-N, BOD and COD scenarios that lacked guidelines from brackish water ecosystems in Ghana, they were developed from brackish water ecosystems in other tropical countries (Appendix E). The standards were adjusted to minimum and maximum values of the permissible guideline limits through best judgement basing on observed data, the available knowledge about estuarine ecosystems in Ghana and general principles of estuarine ecology as shown in Table 16. In cases where the observed value was higher than the maximum permissible guideline limit of a specific parameter, the observed value assumed the maximum value of the permissible guideline limit. Measurements for each parameter were converted to values on an interval scale ranging from 0 (best) – 100 (worst) in accordance with the degree of water quality, implying that higher sub-index values corresponded to more polluted systems (Sahoo *et al*., 2015). The sub-indices were developed from the quality rating function in equation 29 (Liou *et al*., 2004).

$$Qi = \left(\frac{Ci}{Si}\right) \text{ x } 100 \qquad (29)$$

Where QI = sub-index value of the i^{th} parameter, *Ci* = observed of the i^{th} parameter value (mg/L), *Si* = maximum permissible guideline limit of the i^{th} parameter value (mg/L).

Table 16: *Standards for Brackish Water Ecosystems in Ghana*

Parameter	Studies on brackish water ecosystems within Ghana	Studies on brackish water ecosystems from other tropical countries	Range
	Mean ± SE	**Mean ± SE**	**Min-Max**
Temp (°C)	28.8 ±0.4	28.1±0.5	27-29.5
DO (mg/L)	4.6± 0.3	5.7±0.5	5-8
pH	7.4±0.1	7.2±0.1	6-8
EC (µS/cm)	9930 ±3471	9057.5±2755.6	2000-5000
Turbidity	160±5	51.5±13.0	50-160
Salinity	17±3.1	37±17.0	5-20
TDS (mg/L)	2931.7±797.6	6101.1±2286.9	1000-3000
TSS (mg/L)	-	142.3±87.6	20-100
NO_3-N (mg/L)	15±3.4	7.7±2.3	1-10
Orthophosphate (mg/L)	4.1±1.8	8.6±5.4	1-3
NH_4-N (mg/L)	-	0.6±0.2	1-25

Note: Temp-temperature; DO-dissolved oxygen; EC-electrical conductivity; TDS- total dissolved solids; TSS-total suspended solids; NO_3-N-nitrate-nitrogen; NH_4-N-ammonium-nitrate; BOD-biochemical oxygen demand; COD-chemical oxygen demand.

5.2.7 Weighting of parameters

Relative weights of each parameter were computed by a value inversely proportional to the recommended maximum permissible guideline limit of the corresponding i^{th} parameter inequation 30 (Ekere *et al.*, 2019; Marove *et al.*, 2022; Salem et al., 2022).

$$Wi = \frac{K}{Si} \quad (30)$$

Where; Wi = Relative weight of the i^{th} parameter, *k* is the constant of proportionality calculated using the expression in equation 31;

$$K = {}^{1}\!\Big/\!\sum\left(\frac{1}{Si}\right) \quad (31)$$

5.2.8 Aggregation of the overall index

The overall WQI was calculated linearly by aggregating the sub-indices (Qi) with relative weights computed for each parameter (Wi) and dividing by a factor 10 as illustrated in equation 32.

$$WQI = \frac{1}{10}\sum_{i=1}^{i=n} WixQi \quad (32)$$

Where; WQI = index representing estuaries along the coast of Ghana, *Wi* = Relative weight of i^{th} parameter (0-1), Qi = Sub-index of the i^{th} parameter (0-100), *n* = number of parameters.

5.2.9 Benthic Macroinvertebrate Index Method

For this index, the South African Scoring System version 5 (SASS5) method was employed. The SASS5 is based on the Biological Monitoring Working Party (BMWP) method to determine the ecological water quality of an aquatic ecosystem (Armitage *et al*., 1983). The BMWP calculation was performed based on the abundance of macroinvertebrate assemblages, where each taxon was associated with a specified tolerance score. BMWP score for each family ranges from 1 (most tolerant taxa) to 10 (most sensitive taxa). The total score per site was calculated by summing the taxon scores and the value divided by the number of taxa to determine the average score per taxon (ASPT) for each estuary. The BMWP index was formulated using the mathematical function in equation 33;

$$\text{BMWPI} = \frac{\sum B(n)}{N} \tag{33}$$

Where, BMWPI= Biological Monitoring Working Party Index, B = BMWP scores of each family, n = the number of individuals in each family, N = the total number of individuals of all organisms.

5.2.10 Integrated WQI

A harmonised index was computed through integration of physicochemical and benthic macroinvertebrate indices using equation 34;

$$\text{IWQI}_E = \frac{(a+b)}{c} \tag{34}$$

Where; IWQI_E = Integrated WQI for a specific estuary; *a* = physicochemical index; b = benthic macroinvertebrate index; *c* = number of indices, in this case, *c* =2.

An integrated WQI to represent the quality selected brackish water ecosystems along the coast of Ghana was obtained using equation 35;

$$\text{IWQI}_{Gh} = \sum \frac{(a+b)}{n} \tag{35}$$

Where IWQI_{Gh} = Integrated WQI for brackish water ecosystems in Ghana; n = number of brackish water ecosystems, and in this case, n=4.

5.2.11 Classification of Final Index Scores

To ease integration of BMWPI and physicochemical index on a harmonised scale for comparison of various ecological categories, the scale proposed by Armitage *et al.* (1983) and Tiwari & Mishra (1985) for BMWPI and physicochemical index were modified. The former is an increasing scale where the index values rise with the level of pollution while the latter is decreasing scale with index values falling with the level of pollution. The index values range between 0 and 100 and are grouped into classes 1 through 5 in both scales. Modification involved standardisation of the scales by a factor 10 in order to accommodate all the index values obtained and ensuring they ranged between 0 (less disturbed/unpolluted) to 10 (heavily polluted), being distributed among five classes. A final categorisation schema based on a decreasing scale was obtained as illustrated in Table 17.

Table 17: *IWQI Modified Categorisation Scale for Ranking Final Values*

Class	Ecological category	Index score
Class 1	Unpolluted/Less disturbed	0-2.0
Class 2	Slightly polluted	2.1-4.0
Class 3	Moderately polluted	4.1-6.0
Class 4	Polluted	6.1-8.0
Class 5	Heavily polluted	8.1-10.0

5.2.12 Statistical analysis of data

Data was organized in SPSS IBM Statistics v. 25 where normality checks and standardisation of data were performed, followed by PCA for parameter selection. Pearson correlation was used to further reduce the parameters by determining the most correlated parameters. Other computations were performed in MS Excel 2019.

5.3 Results

5.3.1 Multivariate Statistics in Parameter Selection

KMO and Bartlett's test

The resulting KMO values for Whin, Ankobra, Kakum and Volta Estuaries respectively, were; 0.735, 0.619, 0.534 and 0.505, (with respective p-values of 0.000) (Table 18), an indication that relationships among the parameters were significant and the dataset was appropriate for PCA hence rejecting the null hypothesis.

Table 18: *Kaiser–Meyer–Olkin and Bartlett's Test*

Test		**Estuary**			
		Whin	**Ankobra**	**Kakum**	**Volta**
KMO measure of sampling adequacy		0.735	0.619	0.534	0.505
Bartlett's Test of Sphericity	Approx. Chi-Square	1000.297	1248.707	456.659	406.469
	df	105	105	105	105
	Sig.	.000	.000	.000	.000

5.3.2 Selection of parameters representing physicochemical category

Physicochemical parameters are considered essential for understanding the dynamics of other physicochemical contaminants and primary productivity and their effect on water quality. In the current study, this category was composed of SWT, DO and pH. The selection and/or elimination of the three parameters in the four estuaries was based on facts that;

(1) Between SWT and DO, DO was selected since it is the most significant parameter and a key factor for aquatic life development as well as a determinant of other chemical characteristics of water.

Irrespective of the factor loadings from PCA, the significance of DO in water quality cannot be overlooked.

(2) Between SWT and pH, correlation analysis was first considered and in case where the two were strongly correlated, the parameter with high percentage of variation (% standard deviation/mean) from descriptive statistics was selected since it was indicative of high significance in the data.

Consequently, DO was selected and SWT eliminated in all the four estuaries. To add on that, pH was selected as SWT was eliminated since pH indicated high percentage of variation in all the four estuaries. The PCs, parameter loadings, eigenvalues and variances accounted for by physicochemical parameters are presented in Appendices F, H, J and L while correlation matrices are presented in Appendices G, I, K and M corresponding to Volta, Ankobra, Kakum and Whin Estuaries.

5.3.3 Selection of parameters representing particulate matter category

This category was further sub-divided into two sub-categories. Sub-category one encompassed EC, salinity and TDS while sub-category two comprised of TSS and turbidity. The selection and/or elimination criteria were guided by the facts that;

(1) In sub-category one, EC was selected while salinity and TDS were eliminated because, the three are related in such a way that EC is a function of total concentration of dissolved salts in water. Moreover, salinity is a derivative of TDS which is in turn a derivative of EC and therefore EC encapsulates the measures of both TDS and salinity.

(2) In sub-category two, turbidity and TSS are closely related in water. Turbidity qualitatively measures the amount of suspended particles in water while TSS is a quantitative measure of the same. In case where only one of them was represented on the PCs, it was selected. Further to that, if the two demonstrated a weak correlation from the correlation matrix, they were both selected. In case of a strong correlation between them, it was necessary to avoid redundancy by selecting only one with high variance percentage using descriptive statistics.

From sub-category one, EC was selected while both salinity and TDS were eliminated in the four estuaries. On the other hand, turbidity was selected in the Volta Estuary since it was the only one represented on the PCs. Both turbidity and TSS were selected in Ankobra Estuary because of their weak association while TSS was selected in both Kakum and Whin Estuaries due to its significant variation as indicated by high variance percentage from descriptive statistics. The PCs, parameter loadings, eigenvalues and variances accounted for by particulate matter are presented in Appendices F, H, J and L while correlation matrices are presented in Appendices G, I, K and M corresponding to Volta, Ankobra, Kakum and Whin Estuaries.

5.3.4 Selection of parameters representing nutrients category

This category was composed of NO_3-N (sediments), NO_3-N (water), orthophosphate (sediments), orthophosphate (water) and NH_4-N. These were further distinguished into nitrogen compounds (NO_3-N, NH_4-N) and phosphorous compounds (orthophosphate). The facts that provided a baseline for elimination and selection of specific nutrient species include;

(1) If the nutrients represented belonged to different compounds, they were both selected regardless of whether they were found in water column or sediment matrices.

(2) In case of strong correlation existing in nutrients of a particular compound, the one with higher factor loadings was selected.

(3) Where NO_3-N and NH_4-N existed, NO_3-N was selected because it is the most oxidised chemical species in the nitrogen cycle while NH_4-N is the most reduced form of nitrogen and can be converted into NO_3-N through nitrification process. The role of NO_3-N in water quality monitoring with particular interest on eutrophication and its potential implications to aquatic life cannot be overemphasised.

(4) Where a nutrient species existed in both water and sediments, the one in sediments was selected because, in sediments nutrients serve as an essential reservoir and contribute to nutrient cycling in the aquatic ecosystem while in water column, concentration is lower as they are more readily available for uptake by aquatic organisms.

Therefore, in the Volta Estuary, NO_3-N (sediments) and orthophosphate (sediments) were both selected since they belonged to different nutrient compounds. In Ankobra Estuary, NH_4-N was the only nitrogen compound hence selected. Additionally, due to higher factor loading, orthophosphate (sediments) was selected as orthophosphate (water) was eliminated. Moreover, in Kakum Estuary, NO_3-N (sediments) was selected as NH_4-N was eliminated. Finally, in Whin Estuary, orthophosphate (water) (orthophosphate concentration in the water column) was the only nutrient representing phosphorus compounds and was therefore selected. Additionally, NO_3-N (sediment) was selected as NO_3-N (water) was eliminated in Whin Estuary. The PCs, parameter loadings, eigenvalues and variances accounted for by nutrients are presented in Appendices F, H, J and L while correlation matrices are presented in Appendices G, I, K and M corresponding to Volta, Ankobra, Kakum and Whin Estuaries.

5.3.5 Selection of parameters representing organic matter category

This category comprised of BOD and COD. COD includes both biodegradable and non-biodegradable organic and inorganic material, unlike BOD which considers only biodegradable matter hence COD>BOD in water. COD has a greater data representativeness than BOD and it is the reason that informed the selection of COD and elimination of BOD in the four estuaries. The PCs, parameter loadings, eigenvalues values and variances accounted for organic matter are presented in Appendices F, H, J and L while correlation matrices are presented in Appendices G, I, K and M corresponding to Volta, Ankobra, Kakum and Whin Estuaries.

Eventually, multivariate analysis and descriptive statistics were able to reduce the initial 15 parameters to eight in Ankobra, seven respectively in the Volta and Whin Estuaries, and six in Kakum Estuary as shown in Table 19.

Table 19: *Selected Parameters to Represent Water Quality*

	Estuary			
Categories of parameters	**Volta**	**Ankobra**	**Kakum**	**Whin**
Physicochemical	DO, pH	DO, pH	DO, pH,	DO, pH
Particulate matter	EC, Turbidity	EC, turbidity, TSS	EC, TSS	EC, TSS
Nutrients	NO_3-N (sediments), orthophosphate (sediments)	Orthophosphate (sediments) NH_4-N	NO_3-N (sediments)	NO_3-N (sediments), orthophosphate (water)
Organic matter	COD	COD	COD	COD

5.3.6 Physicochemical WQI

Based on the analyses using the modified WAWQI for physicochemical parameters, final index values were computed as 8.0, 8.6, 7.2 and 9.1 for the Volta, Ankobra, Kakum and Whin Estuaries, respectively. The Volta and Kakum Estuaries therefore fall under Class 4 (polluted) while the Ankobra and Whin Estuaries fall under Class 5 (heavily polluted). The WQPs, observed values, WQI values, permissible guidelines and various WQI class descriptions for the four estuaries are presented in Table 20.

Table 20: *WQI from Physicochemical Parameters*

Volta Estuary			**Ankobra Estuary**			**Kakum Estuary**			**Whin Estuary**			**Permissible guideline limits**
WQP	***Ci***	$\frac{1}{10}(WiQi)$	**WQP**	***Ci***	$\frac{1}{10}(WiQi)$	**WQP**	***Ci***	$\frac{1}{10}(WiQi)$	**WQP**	***Ci***	$\frac{1}{10}(WiQi)$	**(*Si*)**
DO	6.2	1.4	DO	5.1	1.3	DO	5.1	2.2	DO	5.9	1.2	DO: 8
pH	5.7	1.3	pH	5.4	1.3	pH	5.2	2.3	pH	6.2	1.3	pH: 8
EC	2697.4	0.0	EC	2198.7	0.0	EC	5000.0	0.0	EC	5000.0	0.0	EC: 5000
Turb	6.9	0.0	Turb	160.0	0.1	TSS	38.2	0.1	TSS	87.0	0.1	TSS: 100
NO_3-N (S)	2.9	0.4	Ortho (S)	3.0	5.3	NO_3-N (S)	9.6	2.7	NO_3-N (S)	10.0	1.3	Turb: 160
Ortho (S)	3.0	4.8	NH_4-N	25.0	0.6	COD	1082.0	0.0	Ortho (W)	3.0	4.5	NO_3-N: 10 NH_4-N: 25
COD	1074.9	0.0	COD	2000.0	0.0				COD	2000.0	0.0	Ortho: 3
												COD: 2000
$\frac{1}{10}\sum_{i=1}^{i=n} WiQi$		**= 8.0**			**= 8.6**			**= 7.2**			**= 9.1**	
Ecological category		Polluted (Class 4)			Heavily polluted (Class 5)			Polluted (Class 4)			Heavily polluted (Class 5)	

Note: WQP- Water Quality Parameter; *Ci*- Observed value; WiQi-Water quality index value (*Wi*-Weightage, *Qi*-Sub-index value); *Si*- Maximum permissible limit; DO-dissolved oxygen (mg/L); EC-electrical conductivity (µS/cm); Turb-turbidity (NTU); NO_3-N (S)-Nitrate nitrogen (mg/L);

Ortho- orthophosphate (mg/L); (S)-sediment, (W)-water; NH_4-N-Ammonium nitrogen (mg/L); TSS-total suspended solids (mg/L), BOD-biochemical oxygen demand (mg/L), COD-chemical oxygen demand (mg/L).

5.3.7 Benthic macroinvertebrate Index

The taxa, abundance, tolerance scores, the BMWPI values and various BMWPI class descriptions for the four estuaries are summarised in Table 21.

Table 21*: Benthic Macroinvertebrate Index*

		Whin		Kakum		Volta		Ankobra	
Family	**Taxon**	**(A)**	**S**	**(A)**	**S**	**(A)**	**(S)**	**(A)**	**S**
Capitellidae	*Capitella* sp.	289	1	33	1	157	1	2	1
Capitellidae	*Notomastus sp.*	174	3	43	3	1	3	-	-
Capitellidae	*Heteromastus* sp.	109	3	42	3	-	-	-	-
Nereididae	*Nereis* sp.	296	4	55	4	34	4	3	4
Orbiniidae	*Scoloplos* sp.	4	5	24	5	63	5	-	-
Pilargidae	*Sigambra* sp	16	4	113	4	3	4	-	-
Nephtydae	*Nephtys* sp.	324	4	131	4	9	4	2	4
Scalibregmatidae	*Polyphysia* sp	22	4	12	4	-	-	1	3
Lumbrineridae	*Lumbriconereis* sp	2	5	-	-	8	5	-	-
Syllidae	*Syllis* sp.	2	5	-	-	3	5	-	-
Glyceridae	*Glycera* sp.	1	5	-	-	-	-	-	-
Maldanidae	*Rhodine* sp.	3	5	-	-	-	-	-	-

Table 21, *continued*

Cossuridae	*Cossura* sp.	5	4	1	4	-	-	-	-
Phyllodocidae	*Phyllodoce* sp.	-	-	2	4	-	-	-	-
Naididae	*Tubifex* sp.	35	1	9	1	3	1	1	1
Glossiphoniidae	*Glossiphonia* sp	-	-	1	5	-	-	-	-
Mysidae	*Mysis* sp.	-	-	-	-	1	8	-	-
Penaeidae	*Penaeus* sp.	1	8	-	-	4	8	154	8
Aoridae	*Bemlos* sp.	91	9	146	9	4	9	-	-
Gammaridae	*Gammarus* sp.	1	6	26	6	-	-	-	-
Ocypodidae	*Uca* sp.	-	-	1	3	-	-	1	3
Cirolanidae	*Eurydice* sp.	19	7	-	-	34	7	-	-
Maeridae	*Elasmopus* sp.	-	-	5	9	-	-	-	-
Coenobitidae	*Coenobita sp.*	-	-	-	-	1	3	-	-
Chironomidae	*Chironomous* sp.	104	1	169	1	1	1	-	-
Hemisinidae	*Pachymelania* sp.	-	-	-	-	150	3	-	-
Potaminidae	*Tympanotonus* sp.	-	-	-	-	43	4	-	-
	Total Score	1498	**84**	813	**70**	519	**75**	164	**24**
	ASPT		**4.4**		**4.1**		**4.4**		**3.4**
	BMWP Index		**3.3**		**4.1**		**3.2**		**7.7**
	Ecological category	Slightly polluted		Moderately polluted		Slightly polluted		Polluted	

Note: (A) – abundance; S- tolerance score; ASPT-Average Score Per Taxa; BMWPI-Biological Monitoring Working Party Index. Adapted from South African Scoring System, SASS5 (Dickens & Graham, 2002).Where several organisms occurred in one family, scores were based on individual taxon)

From the results in Table 21, Ankobra recorded the lowest ASPT while Whin and Volta Estuaries recorded the highest ASPT values. The relatively high ASPT value in the Whin and Volta Estuaries indicated presence of a few sensitive high scoring taxa hence relatively good water quality while low ASPT value in Ankobra implied presence of more pollution tolerant species pointing to poor water quality. The BMWPI values were computed as 3.3, 4.1, 3.2 and 7.7 corresponding to Whin, Kakum, Volta and Ankobra Estuaries (Table 21). Basing on the classification schema in Table 17, Whin and Volta Estuaries are slightly polluted (Class 2), Kakum Estuary is moderately polluted (Class 3) while Ankobra Estuary is polluted (Class 4).

5.3.8 Integrated WQI

Integrated WQI (IWQI) computed from both physicochemical and benthic macroinvertebrate indices is presented in Table 22. Considering individual estuaries, the classification schema (Table 20) shows that Whin Estuary (6.2) is polluted (Class 4), Volta (5.6) and Kakum (5.7) Estuaries are moderately polluted (Class 3) while Ankobra Estuary (8.2) is heavily polluted (Class 5). Eventually, the $IWQI_E$ pulls together the four estuaries and the final index value representing the status of water quality is 6.4 corresponding to polluted water (Class 4).

Table 22: *Integrated WQI for Selected Estuaries Along the Ghanaian Coast*

Estuary	Physicochemical Index	BMWPI	IWQIE
Whin	9.1	3.3	6.2 (polluted)
Volta	8.0	3.2	5.6 (moderately polluted)
Ankobra	8.6	7.7	8.2 (heavily polluted)
Kakum	7.2	4.1	5.7 (moderately polluted)
$IWQI_{Gh}$ Ecological category	$\sum \frac{(a+b)}{n}$ =		**6.4 (polluted) (Class 4)**

Note: a = BMWP index; b = physicochemical index; n = number of estuaries, in this case n=4; $IWQI_E$ = Integrated WQI for each estuary; $IWQI_{Gh}$ = Integrated.

5.4 Discussion

The proposed $IWQI_{Gh}$ for monitoring ecological health of brackish water ecosystems in Ghana has been developed by incorporating physicochemical and benthic macroinvertebrate indices. The most appropriate parameters to be utilised in the physicochemical index formulation have been selected through multivariate statistical methods (Principal Component Analysis, Pearson Correlation Coefficient ‘r’) and descriptive statistics (% of variance from standard deviation: mean ratio).

5.4.1 PCA in selection of parameters for $IWQI_{Gh}$ development

Principal Component Analysis (PCA) is a powerful tool that transforms complex multivariate datasets to a minimal and manageable number of factors without loss of information (Ewaid *et al.*, 2020; Tripathi & Singal, 2019; Quevedo-Castro *et al.*, 2018). Principal Component Analysis does this by preserving and transforming the structure and pattern of the original dataset containing physicochemical and biological parameters to the maximum extent possible (Tripathi & Singal, 2019b). In PCA, a minimum of 150-300 cases is recommended to obtain satisfactory results (Sutadian *et al.*, 2017; Tripathi &

Singal, 2019). The current study satisfied this criterion with a total of 216 test scores collected from nine stations in each of the four estuaries for six sampling episodes for the period sampled. The whole idea of parameter reduction is to considerably reduce excessive use of resources in terms of assessment costs, time, effort, human resources etc., hence promoting routine monitoring (Tripathi & Singal, 2019; Banda & Kumarasamy, 2020a).

In the current study, through multivariate statistics, the original 15 physicochemical parameters common to the four estuaries have been reduced to eight (Ankobra Estuary), seven (Volta and Whin Estuaries) and six (Kakum Estuary). Parameter selection using PCA and Pearson Correlation Coefficient "r" has also been applied elsewhere with parameter reduction from 25 to 6 (Ewaid *et al.*, 2020) and 20 to 9 (Tripathi & Singal, 2019b).

Since environmental parameters have different units, z-score transformation (normalisation) has been carried out to bring them all to a common platform with a mean of zero and standard deviation of one. Normalisation of data in PCA is a common practice which eases value aggregation and has been practiced in other researches (Abuzaid, 2018; Liu *et al.*, 2003; Njuguna *et al.*, 2020; Tripathi & Singal, 2019). Moreover, KMO and Bartlett's test of Sphericity performed to authenticate the suitability of the data set for PCA revealed ≥0.5 and ≤0.05 as values for KMO and significance, respectively hence the data was satisfactory to handle PCA. The results agree with those of Banda & Kumarasamy (2020a) whose KMO value = 0.510 but are slightly lower than those of Tripathi & Singal (2019) whose KMO value was 0.722. In order to purposefully conclude PCA results, the number of PCs to be retained is characterised as; related eigenvalues greater than one (> 1.0), initial eigenvalues percentage of variance of greater than ten percent (> 10 %), and cumulative percentage of variance of greater than sixty percent (> 60 %) (Tripathi & Singal, 2019a). The current study satisfied the second criterion where the extracted PCs had initial eigenvalues percentage of variance of greater than ten percent (>10%) although the cumulative variance varied with each estuary such that the Whin, Kakum, Ankobra and Volta Estuaries corresponded to 68, 56, 67 and 48 % with three 3PCs extracted in each case. The first two PCs accounted for the highest percentage of variance in the dataset and therefore 3PCs were sufficient. Some studies used PCs with higher cumulative variances in comparison to the current study, e.g. 81.88 % with three PCs (Ewaid *et al*., 2020), 90.36 % with 5PCs (Tripathi & Singal, 2019b) and 89.34 % with 8 PCs (Quevedo-Castro *et al*., 2018).

Factor loadings are classified as “strong”, “moderate”, and “weak” corresponding to absolute loading values of >0.75, 0.75–0.50, and 0.50–0.30, respectively (Liu et al., 2003). The current study selected parameters with moderate and strong factor loadings contributing (≥ 0.50; positive or negative) to the first 3PCs constituting the shortlisted parameters. In other studies, factors with lower loading values have been selected to represent water quality like the case of >±0.35 (Tripathi & Singal, 2019b). After selecting the moderately and strongly loaded parameters, they were exposed to Pearson’s correlation Coefficient “r” which generated the correlation matrix for identifying the number of possible parameters with same importance to be discriminated against. Notably, significant information is obtained from parameters containing the lowest relationship between them (An *et al*., 2015). Moreover, parameters not significantly correlated are the most representative (Varol &

Davraz, 2015). From the correlation matrix, correlation coefficients (r ≥-0.3 to r ≥+0.3) were considered as strong correlations as per Wang (2018). PCA/Pearson correlation coefficient "r" in parameter selection is therefore a widely employed technique in various aquatic ecosystems in WQI development ecosystems ranging from river systems (Sahoo *et al.*, 2015; Zeinalzadeh & Rezaei, 2017), tropical estuaries (Looi *et al.*, 2013), coastal bays (Al-Mutairi *et al.*, 2014), etc.

5.4.2 Physicochemical indicators of the $IWQI_{Gh}$

In the physicochemical index, the nine parameters selected to represent the quality of brackish ecosystems in Ghana include DO, pH, EC, COD, turbidity, TSS, NH_4-N, NO_3-N and orthophosphate. The selected parameters fulfil the Dunnette (1979) criteria of the five commonly recognized impairment categories viz; (1) oxygen status (DO,COD), (2) eutrophication (NH_4-N, NO_3-N and orthophosphate), (3) health aspects, (4) physical characteristics (pH, turbidity, TSS), and (5) dissolved substances (EC). The parameters have been carefully and procedurally selected to ensure only those with greatest influence on the quality of brackish water are retained. Using a fixed system, four fixed parameters (DO, pH, EC and COD) common to the four estuaries have been selected. Other indices that considered a fixed system in parameter selection include the National Sanitation Foundation Index; DO, Faecal Coliforms (FC), pH, BOD, temperature, total phosphorus (TP), NO_3-N, turbidity and total solids (Brown *et al.*, 1970), a generalized WQI for Taiwan; DO, BOD, NH-$_4$N, suspended solids, turbidity, FC, temperature, toxic parameters, pH (Liou *et al.*, 2004) and an innovative index for evaluating water quality in streams; DO, TP, FC, turbidity, specific conductance (Said *et al.*, 2004). Nevertheless, to avoid "rigidity" as an abnormality in the final index, five additional parameters have been selected since they are significant for water quality evaluation in specific estuaries. These are TSS, turbidity, NH_4-N, NO_3-N and orthophosphate. The open system of including additional parameters is advantageous as it gives the flexibility to users to incorporate as many parameters from the list of potential parameters and has also been practiced by the CCME-WQI (CCME, 2001). Essentially, the current study has employed a mixed system (incorporating both fixed and open systems) which has so far been observed to be the best, however, there is no method that can achieve 100 % accuracy in parameter selection (Abbasi & Abbasi, 2012).

The final physicochemical index values indicate that the Volta and Kakum Estuaries are polluted (Class 4) while Ankobra and Whin Estuaries are heavily polluted (Class 5), Table 5.6. The parameters that contribute to poor status of water in the estuaries include orthophosphate (Volta Estuary), EC (Kakum Estuary), EC, NO_3-N, orthophosphate, COD (Whin Estuary) and turbidity, NH_4-N, orthophosphate, COD (Ankobra Estuary). These are parameters that exceed the maximum limit of permissible guidelines in the respective estuaries hence contributing highly to the final index values. Anthropogenic activities in the catchment including illegal gold mining, industrial discharges, run-offs from phosphorus-rich sewage, agricultural run-offs, and high levels of organic matter decomposition from the surrounding mangrove ecosystems are possible sources of contaminants in the estuaries in the current study. The combined effort of excessive nitrogen and phosphorus-based compounds cause eutrophication that drastically accelerate aquatic plant matter that in-turn reduces concentration of DO and affects other WQPs due to creation of hypoxic and anoxic conditions (USEPA, 2012). Such conditions have been found to distabilise ecological health of estuarine ecosystems including; altering the ecological structure of communities especially the less mobile ones, disturbing suitable habitats, interfering with trophic predator-prey interactions, etc., which render the estuaries unstable and less functional (Ecological Society of America, 2000). Similar findings where high nutrient concentration has contributed to high final WQI value and consequently poor water quality in estuaries have been recorded (Ezekwe & Edoghotu, 2015;Ujjania & Dubey, 2015). Nonetheless, other WQIs have been constructed with low records of nutrients, attributing the results to uptake of nutrients by aquatic plants and lack of anthropogenic activities in the upstream (Al-Musawi *et al.*, 2018; Shah & Joshi, 2017).

Elevated turbidity levels have the potential of altering physiological processes, e.g., photosynthesis due to reduced light penetration. Soil erosion, large bottom feeders that disturb sediments, waste discharge and urban run-off, contribute to increased turbidity levels in water (USEPA, 2012). Further to that, EC in water has been reported to increase with mining activities and dredging upstream, and it is indicative of dissolved ionised organic compounds. COD tests on the other hand estimate the need for oxygen during the oxidation of inorganic compounds and the breakdown of organic materials. Supposedly, higher COD concentration suggest contaminated water. Armah *et al.* (2012) reported turbidity as one of parameters contributing to high WQI in the Tarkwa mining area in Ghana. Ujjania & Dubey (2015) observed that turbidity, COD and EC exceeded limits of Indian drinking water quality standards, which contributed to a high WQI hence rendering Tapi Estuary polluted. EC was among parameters discovered to contribute to the high WQI value thus poor water quality of Al Hammar marsh (Al-Musawi *et al.*, 2018). The final physicochemical WQI values for each estuary in the present study suggest that the water quality is polluted or heavily polluted. This is contrary to physicochemical analysis of water in Vamsadhara Estuary which showed that the estuary is pollution free and ecologically balanced, and the final WQI was characterised as good (Pradesh *et al.*, 2020).

5.4.3 Macroinvertebrates indicators of the $IWQI_{Gh}$

The ability of an aquatic ecosystem to support macroinvertebrates depends on its habitat quality, physicochemical conditions, and local taxonomic diversity. Therefore, a wide range of brackish habitats and water chemistry offer the possibility for a great diversity of brackish water macroinvertebrates (Likens, 2010). In the present study, Whin Estuary recorded the highest macroinvertebrate abundance while the lowest was recorded in Ankobra which corroborated results of total scores. The Whin, Kakum and Volta Estuaries are relatively diverse with both large numbers of tolerant and intolerant taxa in comparison to the Ankobra Estuary and their ASPT values ≥4. Following an argument by Armitage *et al.* (1983), ASPT scores greater than four is an indication of clean water as a result of contribution from a large number of high scoring taxa. Nonetheless, the final BMWPI scores placed the estuaries in various ecological categories; Whin and Volta (slightly polluted), Kakum (moderately polluted) and Ankobra (heavily polluted), Table 21.

Generally, Ankobra Estuary performed poorly in terms of taxa abundance, total score, ASPT and final BMWQI. The availability of only one sensitive family (Penaeidae) pointed to worse water quality ($IWQI_A$=8.2), contributing to the estuary being heavily polluted. Ankobra Estuary has been reported to be a receptacle of gold mine-wash from the catchment. Alluvial mining deposits like heavy metals are discharged into the Ankobra River hence finding their way down the estuary. This poses serious ecological implications as well as threats to the health of communities that depend on the water body for drinking, domestic use and fishing (Faseyi *et al.*, 2022). This could be the reason the estuary is less dominated by pollution sensitive taxa and by extension less dominance of other taxa.

5.4.4 Integrated WQI ($IWQI_{Gh}$)

The $IWQI_E$ places Volta and Kakum Estuaries in moderately polluted category. The Whin Estuary is in the polluted category and the Ankobra Estuary is in the heavily polluted category.

The $IWQI_{Gh}$ concludes that the selected estuaries representing the quality of brackish water habitats in Ghana are polluted ($IWQI_{Gh}$=6.4, Class 4). The present study notes the high abundance of low scoring taxa to be indicative of pollution in the estuaries which suggests that the estuaries are undergoing environmental stress. Moreover, the premise that Chironomids are good colonists and occur under a variety of conditions (Likens, 2010) further buttresses the current findings since *Chirononomous* sp. were unibiquitous, they were among the least sensitive taxa to organic pollution and therefore they pointed to polluted systems. To further support the observations in the present study, none of the pollution sensitive taxa within the orders of Ephemeroptera, Plecoptera and Trichoptera (EPT) were recorded in the estuaries, yet they have been dubbed as indicators of good water quality (Olomukoro & Dirisu, 2014).

It is impossible to overstate the value of benthic macroinvertebrate indices in the evaluation of water quality. However, there are numerous challenges accompanying the efforts to utilise them for water quality monitoring especially in Africa and more specifically in Ghana. There is paucity of taxonomical and ecological information on macroinvertebrates communities that explicitly distinguishes between reference conditions and moderately impacted sites as well as those that distinguishes between reference conditions and highly impacted sites (Elias *et al.*, 2014). Most of aquatic systems have been heavily impacted by anthropogenic activities, consequently reducing reference sites (Kaaya *et al.*, 2015). Whin Estuary was designated as "pristine" and was applied as "reference site" in the present study. From the $IWQI_E$ results, Whin Estuary is polluted ($IWQI_W$=6.2, Class 4) and its quality is even worse than Volta ($IWQI_V$=5.6) and Kakum ($IWQI_K$=5.7) which are moderately polluted and fall under Class 3. The present study therefore uses this basis to refute the pristine nature of Whin as earlier reported by Atindana *et al*. (2020) and CRC/FoN (2010).

Finally, the guide for scoring tolerance levels for benthic macroinvertebrates used in the present study was adapted from South African Scoring System v.5 (SASS5). This is problematic given the estuarine ecology with regards to taxa uniqueness. To ensure effective bioassessment, there is need to develop a benthic macroinvertebrate scoring system for brackish water ecosystems that are specific to Ghana for effective interpretation of tolerance and sensitivity of occurring taxa based on local conditions. As far as the current study is concerned, there is no existing index (neither physicochemical nor biological) for assessing the quality of Ghanaian brackish water ecosystems.

The proposed $IWQI_{Gh}$ is one of its kind and can be improved to ensure its customisation for the sole purpose of assessment, frequent monitoring and comparison of brackish water ecosystems in Ghana.

5.5 Conclusion

The $IWQI_{Gh}$ can be applied in monitoring brackish water ecosystems along the coast of Ghana by researchers and policy makers. Accordingly, informative decisions on formulating regulatory measures against pollution related activities in the catchment can be made for sustainable utilisation of brackish water ecosystems. The future perspective is to devote efforts towards integrating habitat assessment in the integrated index in order to have a comprehensive outlook of the water quality status.

CHAPTER SIX

SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

6.1 Summary

Before this study, there was no country-specific water quality index (WQI) for monitoring the water quality of estuaries. Therefore, the study's primary aim was to address this gap by developing an integrated WQI as an assessment tool customised for monitoring water quality for better management of estuarine ecosystems in Ghana. In order to achieve this aim, three specific objectives were set to achieve, including (1) To review the adapted WQIs in the African context and examine their limitations and potential for water quality monitoring using existing literature; (2) To assess the quality of water in selected estuaries using physicochemical parameters and benthic macroinvertebrate community structure; and (3) To develop a customised integrated WQI for monitoring estuarine ecosystems in Ghana using multivariate statistical approaches. It is important to report that each of these objectives were successfully achieved and the outcomes published in Chapters 3, 4 and 5, respectively, as part of this thesis. Summaries of the published articles presented as follows:

Adapted Water Quality Indices: Limitations and potential for water quality monitoring in Africa

This article presents the findings of the process(es) involved in WQI modifications for monitoring water quality in Africa, associated limitations and suggests ways of improving on the limitations. From a review of 42 research articles from five databases in the last 10 years (2012–2022), the WAWQI and the CCME-WQI are the most adapted WQIs. The major limitations were encountered in WQI developmental steps, largely in parameter selection and classification schemes used for the final index value.

Benthic macroinvertebrates as indicators of water quality: A case study of brackish habitats along the coast of Ghana

This article presents findings of the assessment of water quality using physicochemical parameters and benthic macroinvertebrate community structure in the Volta, Kakum and Ankobra Estuaries along the coast of Ghana. Most of pollution tolerant taxa like *Capitella* sp., *Nereis* sp., *Heteromastus* sp., *Tubifex* sp., *Cossura* sp. and *Chironomous* sp. dominated Kakum and Whin Estuaries while pollution sensitive taxa like *Scoloplos* sp., *Eurydice* sp., *Lumbriconereis* sp. and *Pachymelania* sp. dominated Volta Estuary. The species-environment interactions listed DO, SWT, orthophosphate, nitrates, ammonium, EC, turbidity, and COD as the most significant parameters affecting the spatial distribution of macroinvertebrates in the studied estuaries. Although Kakum and Whin Estuaries are dominated by a wide range of pollution tolerant taxa, Pearson correlation analysis shows weak and moderate correlations (both positive and negative) in both estuaries, suggesting that they are moderate on organic pollution. Moreover, Kakum Estuary is the most diverse estuary despite the moderate pollution levels while lowest species diversity occurs in Ankobra Estuary. Consequently, the study ranks the four estuaries in terms of stability using the biological indices and the implication is that Kakum Estuary is ecologically healthier than Whin Estuary, which is healthier than Volta Estuary that is in turn healthier than Ankobra Estuary, i.e., Estuarine ecological health ranking; Kakum >Whin> Volta > Ankobra Estuary.

Development of an integrated water quality index for monitoring estuarine water quality in Ghana

This article presents findings of a customised WQI for monitoring estuarine ecosystems in Ghana using multivariate statistical approaches. The selection criteria yielded nine parameters representative of water quality along the coast of Ghana; DO, pH, EC, COD, turbidity, TSS, NH_4-N, NO_3-N and orthophosphate. A modified WAWQI has been computed by aggregating sub-index values and relative weights from functions incorporating the maximum permissible guideline limits of brackish water ecosystems computed from other studies in similar environments in Ghana and other tropical countries. The index developed has incorporated both physicochemical parameters (physicochemical index) and benthic macroinvertebrates (BMWP index) in the four estuaries to represent water quality along the coast of Ghana ($IWQI_{Gh}$). The estuaries under study are polluted with $IWQI_{Gh}$ placing them under ecological category 4 "polluted," and nutrients (nitrates, orthophosphate and ammonium), turbidity, EC and COD are the major contributors to high index values. Furthermore, the selected estuaries are dominated by low scoring taxa that is highly tolerant to organic pollution. The pollution status of the studied waterbodies is attributed to the impacts of human activities like agriculture, sewage disposal and illegal gold mining in the catchment areas of these waterbodies, and acting as the greatest contributors to the deteriorated water quality.

6.2 Conclusions

Based on the outcome of this study, the following conclusions are made:

1. The most commonly adapted indices for water quality monitoring in Africa are Weighted Arithmetic Water Quality Index and Canadian

Council of Ministers of Environment Water Quality Index which exhibit a general bias towards physicochemical parameters over biological metrics. The indices tend to suffer from abnormalities such as ambiguity, eclipsing, and rigidity, which limit their application potential.

2. Effective water quality monitoring in most developing countries is hampered by a lack of indigenous or region-specific WQIs due to over reliance on using adapted WQIs.
3. Kakum and Whin Estuaries are dominated by pollution-tolerant taxa. However, the estuaries are facing moderate organic pollution and Kakum has a relatively high species diversity compared to the Whin, Volta and Ankobra Estuaries.
4. The Volta Estuary is dominated by pollution-sensitive taxa and is low on organic pollution.
5. The most significant environmental parameters affecting the spatial distribution of macroinvertebrates in the studied estuaries are dissolved oxygen, salinity, orthophosphate, nitrates, ammonium, electrical conductivity, turbidity and chemical oxygen demand.
6. The Ankobra Estuary is facing serious environmental stress with extremely high turbidity, which has negatively impacted the benthic fauna and therefore the current study finds it to be the least healthy ecologically among the four estuaries. The observed level of turbidity loads could be attributed to anthropogenic activities such as agriculture, sewage disposal and gold mining in the catchment area.

7. The most significant water quality indicator parameters in estuaries of Ghana include DO, pH, EC, COD, turbidity, TSS, NH_4-N, NO_3-N and orthophosphate.
8. Using the $IWQI_E$ for specific estuaries, Ankobra Estuary ($IWQI_A$=8.2) is heavily polluted. Moreover, the level of pollution in Whin Estuary ($IWQI_W$=6.2) is higher than that of Volta ($IWQI_V$=5.6) and Kakum ($IWQI_K$=5.7) Estuaries. Therefore, Whin Estuary may have lost its hold on suggestions as being the most pristine estuary in Ghana.
9. The $IWQI_{Gh}$ classifies selected estuaries along the coast of Ghana as "polluted", which could be attributed to low abundance of high scoring benthic macroinvertebrates and significant presence of nutrients, turbidity, EC and COD.
10. Incorporation of both physicochemical parameters and benthic macroinvertebrates in developing an integrated WQI gives a holistic outlook of estuarine ecosystems health status.

6.3 Recommendations

The following recommendations are made based on the study:

1. The Government of Ghana needs to take actions to halt further the degradation water bodies such as the Ankobra Estuary in order to enhance biodiversity conservation and livelihoods of nearby communities.
2. The development of WQIs in Ghana must integrate physicochemical and biological indicators into assessment protocols of pollution in order to have a holistic picture of water quality status.

3. African water quality researchers must collaborate in developing WQIs to address the continent's water quality problems.
4. New WQI models should embrace non-subjective statistical approaches and logical linguistic descriptions in classification schemes to ensure objectivity in WQI development.
5. Benthic macroinvertebrate ecologists in Ghana should develop country-specific identification keys and scoring systems for aquatic ecosystems to ensure effective classification of taxa based on local conditions.

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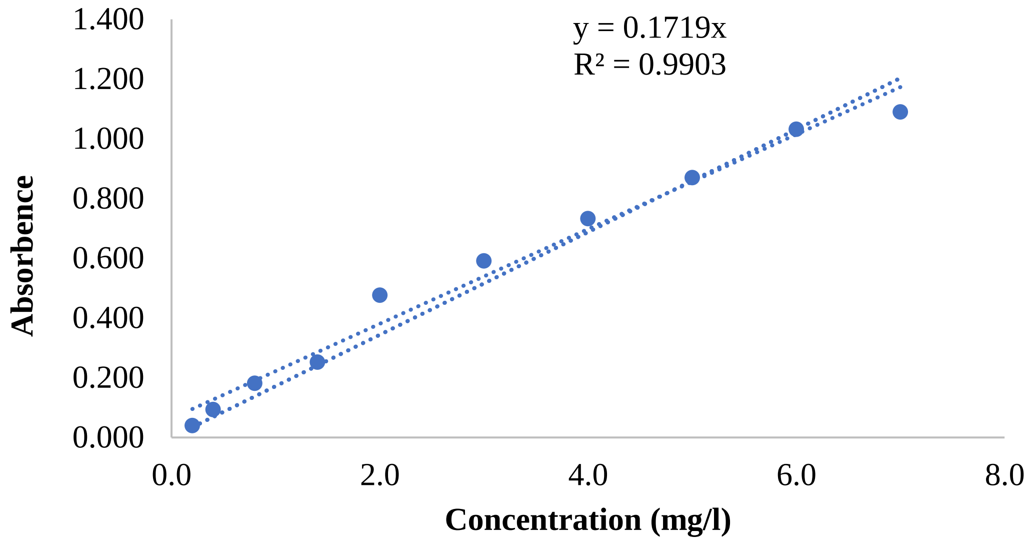
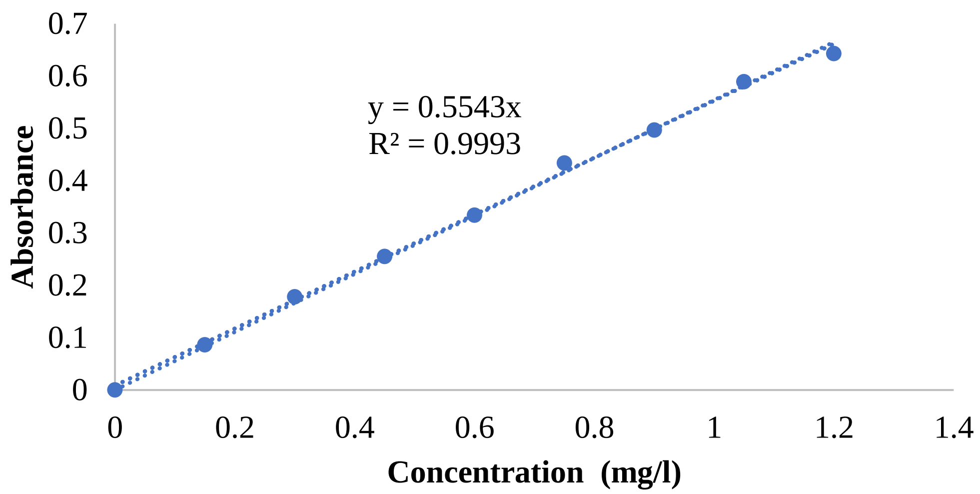
APPENDICES

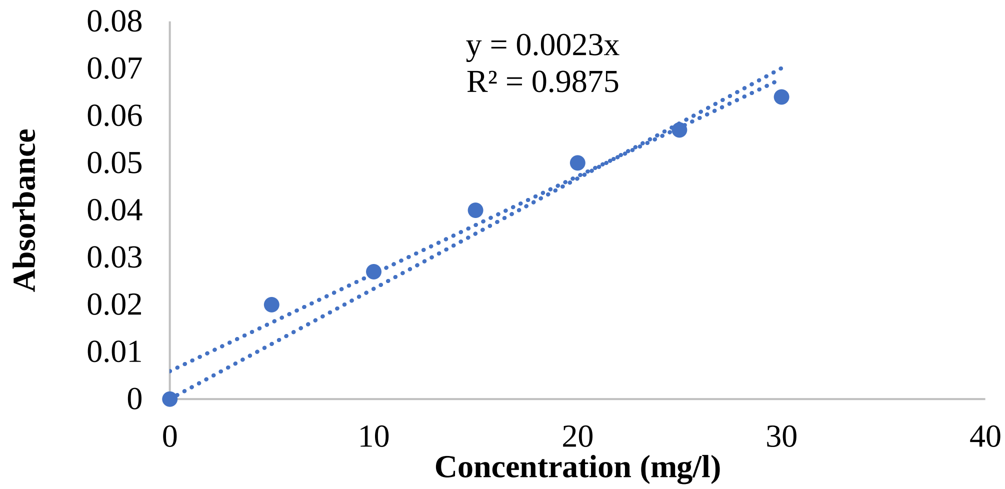
Appendix A. Standard calibration curve for Nitrate-nitrogen (NO_3-N)



Appendix B. Standard calibration curve for Orthophosphates



Appendix C. Standard calibration curve for Ammonium-Nitrogen (NH_4-N)



Appendix D. Parameters from brackish water in other studies within Ghana

No.	Parameter	Range (midpoint) OR Average	Water body	Reference
1.	Temp (°C)	27.28-29.22 (28.25)	Volta Estuary	Adjei-Boateng *et al.*, 2010
		28.50-29.30 (28.9)	Domini Lagoon	Aggrey-Fynn *et al.*, 2011
		23.20-31.10 (27.15)	Amansuri Lagoon	Aggrey-Fynn *et al.*, 2011
		(32.9)	Butuah Lagoon	Aheto *et al.*, 2011
		(22.4)	Whin Estuary	Aheto *et al.*, 2011
		25.67-28.91 (27.58)	Nyan Estuary	Dzakpasu &Yankson, 2015
		25.06-29.21 (27.14)	Kakum Estuary	Dzakpasu &Yankson, 2015
		26.50-29.00 (27.75)	Kakum Estuary	(Fianko *et al.*, 2007)
		28.60-35.30 (31.95)	Korle lagoon	Karikari *et al.*, 2006
		27.00-32.00 (29.50)	Ankobra Estuary	Soetan *et al.*, 2022
		25.08-31.38 (28.23)	Tendo Lagoon	Miyittah *et al.*, 2020

Appendix D, continued

		(32.95)	Butuah Lagoon	CRC/FoN, 2010
		(25.98)	Whin Estuary	CRC/FoN, 2010
		(25.57)	Essei Lagoon	CRC/FoN, 2010
		27.72- 30.94 (29.33	Keta Lagoon	Lamptey & Amah, 2008
		32.60-37.80 (35.20)	Kakum Estuary	Okyere *et al.*, 2011
		24.85-29.11 (26.98)	Kakum Estuary	Dzakpasu, 2019
		26.70-29.04 (27.87)	Pra Estuary	Dzakpasu, 2019
		23.62-30.93 (27.28	Benya Lagoon	Dzakpasu, 2019
		26.48-30.91 (28.69)	Sakumo II Lagoon	Dzakpasu, 2019
		28.58-31.71 (30.15)	Fosu Lagoon	Dzakpasu, 2019
		26.33-33.36 (29.85)	Muni Lagoon	Dzakpasu, 2019
		26.50-29.80 (28.15)	Pra Estuary	Okyere &Nortey., 2018
		27.28 - 29.59 (28.44)	Volta Estuary	Madkour *et al.*, 2011
		24.40-29.70 (27.05)	Pra Estuary	Faseyi *et al.*, 2022
		24.20-30.00 (27.10)	Ankobra Estuary	Faseyi *et al.*, 2022
		(28.99)	Keta Lagoon	Dankwa *et al.*, 2004
		(29.29)	Songor Lagoon	Dankwa *et al.*, 2004
		(29.49)	Benya Lagoon	Armah *et al.*, 2012
		(31.37)	Fosu Lagoon	Armah *et al.*, 2012
		30.10- 32.07 (31.01)	Brenu Lagoon	Akwetey *et al.*, 2021
	Mean	**28.79**		
	Std Error	**0.44**		
2.	DO (mg/L)	2.48-8.76 (5.62)	Volta Estuary	Adjei-Boateng *et al.*, 2010
		5.90-6.00 (5.95)	Domini Lagoon	Aggrey-Fynn *et al.*, 2011
		4.70-6.50 (5.60)	Amansuri Lagoon	Aggrey-Fynn *et al.*, 2011

Appendix D, continued

	(3.70)	Butuah Lagoon	Aheto *et al*., 2011
	(3.10)	Whin Estuary	Aheto *et al*., 2011
	2.73-4.42 (3.59)	Nyan Estuary	Dzakpasu & Yankson, 2015
	2.43-4.38 (3.41)	Kakum Estuary	Dzakpasu & Yankson, 2015
	0.20-6.47 (3.34)	Korle lagoon	Karikari *et al*., 2006
	5.38-7.37 (6.38)	Ankobra Estuary	Soetan *et al*., 2022
	0.43-5.52 (2.98)	Tendo Lagoon	Miyittah *et al*., 2020
	4.00-7.00 (5.50)	Pra Estuary	Okyere, 2019
	(9.51)	Butuah Lagoon	CRC/FoN, 2010
	(3.11)	Whin Estuary	CRC.Fon, 2010
	(0.10)	Essei lagoon	CRC.Fon, 2010
	3.40-5.20 (4.30)	Kakum Estuary	Okyere *et al*., 2011
	2.79-6.07 (4.43)	Kakum Estuary	Dzakpasu, 2019
	2.07-6.92 (4.49)	Pra Estuary	Dzakpasu, 2019
	1.90-5.92 (3.91)	Benya Lagoon	Dzakpasu, 2019
	2.16-5.91 (4.04)	Sakumo II Lagoon	Dzakpasu, 2019
	2.13-9.23 (5.68)	Fosu Lagoon	Dzakpasu, 2019
	2.13-9.23 (5.68)	Muni Lagoon	Dzakpasu, 2019
	4.00-6.00 (5.00)	Pra Estuary	Okyere & Nortey., 2018
	1.52 - 8.76 (5.14)	Volta Estuary	Madkour *et al.*, 2011
	0.70-7.73 (4.22)	Pra Estuary	Faseyi *et al*., 2022
	0.50-8.28 (4.39)	Ankobra Estuary	Faseyi *et al*., 2022
	(4.20)	Keta Lagoon	Dankwa *et al*., 2004
	(4.71)	Songor Lagoon	Dankwa *et al*., 2004
	(1.24)	Benya Lagoon	Armah *et al*., 2012

Appendix D, continued

		(8.74)	Fosu Lagoon	Armah et al., 2012
		5.93- 6.70 (6.32)	Brenu Lagoon	Akwetey *et al.*, 2021
	Mean	**4.61**		
	Std Error	**0.34**		
3.	EC (µSem/cm)	60.00-70.00 (65.00)	Volta Estuary	(Adjei-Boateng et al, 2010)
		(0.00)	Omini Lagoon	Aggrey-fynn *et al.*, 2011
		42.40-46.50 (44.45)	Amansuri Lagoon	Aggrey-fynn *et al.*, 2011
		(19000.00)	Butuah Lagoon	Aheto *et al.*, 2011
		(55600.00)	Whin Estuary	Aheto *et al.*, 2011
		1800.00-57000.00 (29400.00)	Korle Lagoon	Karikari *et al.*, 2006
		73.28-268.26 (170.77)	Tendo Lagoon	Miyittah *et al.*, 2020
		(2.97)	Butuah Lagoon	CRC/FoN, 2010
		(55.61)	Whin Estuary	CRC/FoN, 2010
		35860.00-67880.00 (51870.00)	Keta Lagoon	Lamptey & Amah, 2008
		4636.00-7355.00 (5995.50)	Kakum Estuary	Okyere *et al.*, 2011
		16.89-6932.00 (3474.45)	Kakum Estuary	Dzakpasu, 2019
		10.35-7166.00 (3588.18)	Pra Estuary	Dzakpasu, 2019
		42.52-54.50 (48.51)	Benya Lagoon	Dzakpasu, 2019
		760.00-2594.00 (1677.00)	Sakumo II Lagoon	Dzakpasu, 2019
		11.00-9941.00 (4976.00)	Fosu Lagoon	Dzakpasu, 2019
		1.33-82.70 (42.02)	Muni Lagoon	Dzakpasu, 2019
		52.00 – 70.00 (61.00)	Volta Estuary	Madkour *et al.*, 2011
		35.00-9001.00 (4518.00)	Pra Estuary	Faseyi *et al.*, 2022

Appendix D, continued

		44.00-9808.00 (9852.00)	Ankobra Estuary	Faseyi *et al.*, 2022
		(33701.50)	Benya Lagoon	Armah et al., 2012
		(4446.80)	Fosu Lagoon	Armah *et al.*, 2012
		27.14- 33.29 (30.22)	Brenu Lagoon	Akwetey *et al.*, 2021
	Mean	**9939.99**		
	Std Error	**3471.02**		
4.	pH	6.48-6.99 (6.74)	Volta Estuary	Adjei-Boateng *et al.*, 2010
		7.4-7.7 (7.55)	Domini Lagoon	Aggrey-fynn et *al.*, 2011
		5.5-6.0 (5.75)	Amansuri Lagoon	Aggrey-fynn *et al.*, 2011
		(7.6)	Butuah Lagoon	Aheto *et al.*, 2011
		(8.1)	Whin Estuary	Aheto *et al.*, 2011
		5.30-6.8 (6.05)	Nyan Estuary	Dzakpasu &Yankson, 2015
		6.05-6.8 (6.43)	Kakum Estuary	Dzakpasu &Yankson, 2015
		6.8-7.7 (7.25)	Kakum Estuary	Franko *et al.*, 2006
		6.1-7.6 (6.85)	Korle Lagoon	Karikari *et al.*, 2006
		6.64-8.26 (7.45)	Ankobra Estuary	Soetan *et al.*, 2022
		5.08-6.83 (5.96)	Tendo Lagoon	Miyittah *et al.*, 2020
		6.9-8.0 (7.45)	Pra Estuary	Okyere, 2019
		(7.62)	Butuah Lagoon	CRC/FoN, 2010
		(8.08)	Whin Estuary	CRC/FoN, 2010
		(7.76)	Essei Lagoon	CRC/FoN, 2010
		7.4-7.9 (7.65)	Kakum Estuary	Okyere et al., 2019
		6.40-7.73 (7.07)	Kakum Estuary	Dzakpasu, 2019
		6.33-7.73 (7.03)	Pra Estuary	Dzakpasu, 2019
		7.34-7.73 (7.54)	Benya Lagoon	Dzakpasu, 2019
		6.72-8.09 (7.41)	Sakumo II Lagoon	Dzakpasu, 2019
		7.26-8.96 (8.11)	Fosu Lagoon	Dzakpasu, 2019

Appendix D, continued

		7.15-8.70 (7.93)	Muni Lagoon	Dzakpasu, 2019
		6.18-8.50 (7.34)	Volta Estuary	Madkour et al., 2011
		6.22-8.24 (7.23)	Pra Estuary	Faseyi *et al*., 2022
		5.58-8.09 (6.84)	Ankobra Estuary	Faseyi *et al*., 2022
		7.77-8.53 (8.15)	Pra Estuary	Tufuor *et al*., 2007
		(8.33)	Keta Lagoon	Dankwa *et al*., 2004
		(8.41)	Songor Lagoon	Dankwa *et al*., 2004
		(7.46)	Benya Lagoon	Armah *et al*., 2012
		(8.22)	Fosu Lagoon	Armah *et al*., 2012
		8.67- 9.37 (9.02)	Brenu Lagoon	(Akwetey *et al*., 2021)
	Mean	**7.43**		
	Std Error	**0.13**		
5.	Turbidity	0.00-98.30 (49.15)	Domini Lagoon	Aggrey-fynn *et al*., 2011
		15.10-20.00 (17.55)	Amansuri Lagoon	Aggrey-fynn *et al*., 2011
		(540.30)	Butuah Lagoon	Aheto *et al*., 2011
		42.3ppm	Whin Estuary	Aheto *et al*., 2011
		6.17-57.67 (31.92)	Nyan Estuary	Dzakpasu &Yankson, 2015
		9.33-12.36 (10.85)	Kakum Estuary	Dzakpasu &Yankson, 2015
		7.00-30.20 (18.60)	Tendo Lagoon	Miyittah *et al*., 2020
		60.00-1000.00 (530.00)	Pra Estuary	Okyere, 2019
		(180.07)	Butuah Lagoon	CRC/FoN, 2010
		(42.29)	Whin Estuary	CRC/FoN, 2010
		(55.79)	Essei Lagoon	CRC/FoN, 2010
		(69.20)	Keta Lagoon	Lamptey & Amah, 2008
		3.06-177.00 (90.03)	Kakum Estuary	Dzakpasu, 2019
		14.35-902.00 (458.18)	Pra Estuary	Dzakpasu, 2019
		5.73-186.00 (95.87)	Benya Lagoon	Dzakpasu, 2019
		8.61-825.00 (416.81)	Sakumo II Lagoon	Dzakpasu, 2019

Appendix D, continued

		7.60-258.00 (132.80)	Fosu Lagoon	Dzakpasu, 2019
		3.14-115.00 (59.07)	Muni Lagoon	Dzakpasu, 2019
		365.00-949.00 (657.00)	Pra Estuary	Okyere & Nortey, 2018
		(5.50)	Benya Lagoon	Armah *et al.*, 2012
		(46.30)	Fosu Lagoon	Armah *et al.*, 2012
	Mean	**160.03**		
	Std error	**45.69**		
6.	Salinity	(0.03)	Volta Estuary	Adjei-Boateng *et al.*, 2010
		(0.00)	Domini Lagoon	Aggrey-fynn *et al.*, 2011
		26.60-29.40 (28.00)	Amansuri Lagoon	Aggrey-fynn *et al.*, 2011
		1.90	Butuah Lagoon	Aheto *et al.*, 2011
		3.70	Whin Estuary	Aheto *et al.*, 2011
		2.88-34.50 (18.69)	Nyan Estuary	Dzakpasu &Yankson, 2015
		0.00-34.48 (17.24)	Kakum Estuary	Dzakpasu &Yankson, 2015
		(19.01)	Butuah Lagoon	CRC/FoN, 2010
		(37.01)	Whin Estuary	CRC/FoN, 2010
		(18.78)	Essei Lagoon	CRC/FoN, 2010
		27.90- 61.80 (44.85)	Keta Lagoon	Lamptey & Amah, 2008
		1.90-3.20 (2.55)	Kakum Estuary	Okyere et al., 2011
		0.13-25.25 (12.69)	Kakum Estuary	Dzakpasu, 2019
		0.04-7.96 (4.00)	Pra Estuary	Dzakpasu, 2019
		4.23-30.46 (17.35)	Benya Lagoon	Dzakpasu, 2019
		0.43-5.50 (2.97)	Sakumo II Lagoon	Dzakpasu, 2019
		2.11-7.50 (4.81)	Fosu Lagoon	Dzakpasu, 2019
		6.55-38.25 (22.40)	Muni Lagoon	Dzakpasu, 2019
		0.02 - 0.03 (0.03)	Volta Estuary	Madkour *et al.*, 2011

Appendix D, continued

		0.02-33.4 (16.71)	Pra Estuary	Faseyi *et al.*, 20122
		0.03-17.14 (8.59)	Ankobra Estuary	Faseyi *et al.*, 20122
		0.01-28.18 (28.19)	Pra Estuary	Tufuor *et al.*, 2007
		(14.55)	Keta Lagoon	Dankwa *et al.*, 2004
		(60.45)	Songor Lagoon	Dankwa *et al.*, 2004
		(33.86)	Benya Lagoon	Armah *et al.*, 2012
		(2.14)	Fosu Lagoon	Armah *et al.*, 2012
		39.68- 42.14 (40.91)	Brenu Lagoon	Akwetey *et al.*, 2021
	Mean	**17.09**		
	Std error	**3.05**		
7.	TDS	310.0-35.00 (33.00)	Volta Estuary	Adjei-Boateng *et al.*, 2010
		10.13-8091.00 (4050.57)	Kakum Estuary	Dzakpasu, 2019
		(10.12)	Pra Estuary	Dzakpasu, 2019
		(6738.00)	Benya Lagoon	Dzakpasu, 2019
		387.00-1223.00 (805.00)	Sakumo II Lagoon	Dzakpasu, 2019
		1863.00-9326.00 (5594.50)	Fosu Lagoon	Dzakpasu, 2019
		13.10-7625.00 (3819.05)	Muni Lagoon	Dzakpasu, 2019
		27.00- 35.00 (31.00)	Volta Estuary	Madkour *et al.*, 2011
		17.00-6842.00 (3429.50)	Pra Estuary	Faseyi *et al.*, 2022
		22.00-9593.00 (4807.50	Ankobra Estuary	Faseyi *et al.*, 2022
	Mean	**2931.72**		
	Std error	**797.58**		
8.	NO_3-N	0.30-29.94 (15.12)	Tendo Lagoon	Miyittah *et al.*, 2020
		2.00-78.20 (40.10)	Pra Estuary	Okyere, 2019

Appendix D, continued

		0.33- 1.08 (0.71)	Keta Lagoon	Lamptey & Amah, 2008
		0.00-18.60 (9.30)	Kakum Estuary	Dzakpasu, 2019
		0.00-17.60 (8.80)	Pra Estuary	Dzakpasu, 2019
		0.00-17.16 (8.80)	Benya Lagoon	Dzakpasu, 2019
		0.00-68.00 (34.00)	Sakumo II Lagoon	Dzakpasu, 2019
		0.00-39.60 (19.80)	Fosu Lagoon	Dzakpasu, 2019
		0.00-13.97 (6.99)	Muni Lagoon	Dzakpasu, 2019
		0.30-17.20 (8.75)	Pra Estuary	Faseyi *et al*., 2022
		0.30-20.90 (10.60)	Ankobra Estuary	Faseyi *et al*., 2022
		8.27-39.86 (24.07)	Pra Estuary	Tufuor *et al*., 2007
	Mean	**15.59**		
	Std err	**3.40**		
9.	Orthophosphate	0.06-1.26 (0.66)	Tendo Lagoon	Miyittah *et al*., 2020
		0.01-0.41 (0.21)	Pra Estuary	Okyere, 2019
		0.20-0.27 (0.24)	Keta Lagoon	Lamptey & Amah, 2008
		0.03-5.56 (2.79)	Kakum Estuary	Dzakpasu, 2019
		0.00-4.46 (2.23)	Pra Estuary	Dzakpasu, 2019
		0.00-9.08 (4.54)	Benya Lagoon	Dzakpasu, 2019
		0.00-49.10 (24.55)	Sakumo II Lagoon	Dzakpasu, 2019
		0.00-9.46 (4.73)	Fosu Lagoon	Dzakpasu, 2019
		0.00-5.22 (2.61)	Muni Lagoon	Dzakpasu, 2019
		0.02-2.10 (2.12)	Pra Estuary	Faseyi *et al*., 2022
		0.02-4.68 (2.35)	Ankobra Estuary	Faseyi *et al*., 2022
		0.02-3.95 (1.99)	Pra Estuary	Tufuor *et al*., 2007
	Mean	**4.09**		
	Std error	**1.75**		

10.	NH_4-N	0.10-0.46 (0.28)	Pra Estuary	Tufuor *et al*., 2007
11.	TSS	80.00-1260.00 (670.00)	Korle Lagoon	Karikari *et al*., 2006

Appendix E. Parameters from brackish water studies in other tropical countries

NO.	Parameter	Range (midpoint) OR Average	Water body, Location	Reference
1.	Temp (°C)	21.00-30.00 (25.17)	Tapi Estuary, India	Ujjania & Dubey, 2015
		20.00-30.50 (25.25)	Mangrove swamps, Nigeria	Lawson, 2011
		29.81-31.69 (30.75)	Gambia R. Estuary, Gambia	Adam *et al.*, 2018
		25.14-26.01 (14.08)	Chilika Lagoon, India	Mahapatro *et al.*, 2009
		29.80-31.50 (30.65)	Omoku Creek, Nigeria	Ewa *et al.*, 2011
		23.50-33.50 (28.50)	Tapi Estuary, India	Nirmal Kumar *et al.*, 2009
		(26.47)	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		27.00-30.20 (28.58)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al.*, 2017
		27.60-30.00 (28.60)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017
		25.30-28.90 (27.10)	Qua Iboe Estuary, Nigeria	Akan & Nsikak, 2010
		27.14-29.21 (28.18)	Cross River Estuary, Nigeria	Akan &Nsikak, 2010
		26.20-28.70 (27.45)	Andoni River Estuary, Nigeria	Ezekwe & Edoghotu, 2015
		28.80-35.00 (31.90)	Arasalar Estuary, India	Raju *et al.*, 2017
		26.30-31.60 (28.95)	Lagoon Aghien, Ivory Coast	Ahoutou *et al.*, 2021
		23.57-28.70 (26.00)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
		27.01-27.97 (27.49)	Lagune Ebrie, Cote D’ivoire	Mireille *et al.*, 2020
		27.30-30.50 (28.90)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2010
		20.90-32.60 (26.75)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		28.10-32.80 (30.45)	Poxim River Estuary, Brazil	Nilin *et al.*, 2019
		27.46-30.20 (28.83)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		24.90-28.05 (30.90)	Aghien Lagoon, Cote D’ivoire	Effebi et al., 2017

Appendix E, continued

		26.00-33.00 (29.42)	Iyagbe Lagoon, Nigeria	Onyena *et al.*, 2021
		26.30-29.50 (27.90)	Aby Lagoon, Ivory Coast	Kambiré *et al.*, 2014
		27.00-29.20 (28.45)	Agniyar Estuary, India	Sugumaran, 2016
		28.50-30.15 (29.15)	Lagos Lagoon, Nigeria	Oyeleke *et al.*, 2019
		26.85-31.31 (29.08)	Muthupet Estuary, India	Suganthi *et al.*, 2020
		27.00-33.40 (30.20)	Rajakkamangalam Estuary, India	Banu et al., 2018
		21.70-33.70 (27.70)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021
		26.40-33.35 (29.88)	Aby Lagoon, Code D'lvore	Assemian-niango et al., 2020
		26.42-29.00 (27.71)	Buenaventura Bay Estuary, Colombia	Duque *et al.*, 2020
		30.50-31.75 (31.13)	Vettar Estuary, India	Nanjappa *et al.*, 2023
		25.19-33.50 (29.35)	Potou Lagoon, Cote D'lvoire	Marthe *et al.*, 2015
		24.00-27.00 (25.50)	Lagos Lagoon, Nigeria	Ajibare & Loto, 2022
	Mean	**28.07**		
	Std Error	**0.53**		
2.	DO (mg/L)	0.80-6.00 (2.73)	Tapi Estuary, India	Ujjania & Dubey, 2015
		0.58-10.00 (5.29)	Mangrove swamps, Nigeria	Lawson, 2011
		4.70-7.30 (6.00)	Gambia R. Estuary, Gambia	Adam *et al.*, 2018
		38.20-41.50 (22.35)	Omoku Creek, Nigeria	Ewa et al., 2011
		7.20-8.50 (7.85)	Tapi Estuary, India	Nirmal Kumar *et al.*, 2009
		7.15	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		4.13-5.74 (4.94)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al.*, 2015
		2.85-7.50 (5.18)	Upper Bonny Estuary, Nigeria	Ngah et al., 2017

Appendix E, continued

	2.82-7.21 (5.02)	Qua Iboe Estuary	Akan &Nsikak, 2010
	4.42-11.21 (7.82)	Cross River Estuary	Akan &Nsikak, 2010

	4.02-6.60 (5.31)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
	3.50-7.20 (5.10)	Arasalar Estuary, India	Raju *et al.*, 2015
	2.58-5.88 (4.23)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
	0.63-4.98 (2.80)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
	1.38-9.06 (5.22)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
	4.05-5.60 (4.83)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
	1.41-9.78 (5.60)	Poxim River Estuary, Brazil	Nilin *et al.*, 2019
	1.87-4.64 (3.23)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
	1.46-8.72 (5.09)	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017
	4.40-6.70 (5.55)	Tana River Estuary, Kenya	Ongore *et al.*, 2013
	5.70-10.40 (8.05)	Sabaki River Estuary	Ongore *et al.*, 2013
	4.00-5.60 (4.67)	Iyagbe Lagoon, Nigeria	Onyema, 2013
	5.74-6.43 (6.09)	Aby Lagoon, Ivory Coast	Kambire *et* al., 2014
	(3.77)	Cross River Estuary, Nigeria	Ebong & John, 2021
	(3.82)	Imo River Estuary, Nigeria	Ebong & John, 2021
	(3.64)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
	2.57-3.41 (2.99)	Agniyar Estuary, India	Sugumaran, 2016
	1.60-6.40 (4.00)	Lagos Lagoon	Oyeleke *et al.*, 2019
	6.81- 10.07 (8.44)	Muthupet Estuary, India	Suganthi *et al.*, 2020
	0.80-8.00 (4.4)	Rajakkamangalam Estuary, India	Banu *et al.*, 2018
	5.90-8.40 (7.15)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021

Appendix E, continued

1.39-11.13 (6.26)	Aby Lagoon, Code D'lvore	Assemian-niango et al, 2020
4.93-7.18 (6.06)	Buenaventura Bay Estuary, Colombia	Duque *et al.*, 2020
(4.40)	Lagos Lagoon, Nigeria	Nkwoji *et al*., 2020
3.48-5.84 (4.66)	Vettar Estuary, India	Nanjappa *et al*., 2023

Appendix E, continued

		2.34-8.91 (5.63)	Potou Lagoon, Cote D'lvoire	Marthe *et al*., 2015
		3.90-4.60 (4.25)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean	**5.66**		
	Std Error	**0.52**		
3.	EC (µS/cm)	950.00-12470.00 (6710.00)	Tapi Estuary, India	Ujjania & Dubey, 2015
		391.00-462.00 (426.50)	Omoku Creek, Nigeria	Ewa *et al*., 2011
		(28.45)	Eastern Obolo Estuary, Nigeria	Udoh *et al*., 2013
		27169.33-39851.33 (33541.50)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al*., 2015
		12775.00-28250.00 (20512.50)	Upper Bonny Estuary, Nigeria	Ngah *et al*., 2017
		16460.00-43500.00 (29980.00)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		12242.90-21337.00 (16789.95)	Nyong Estuary, Cameroon	Anselme *et al*., 2018
		6074.00-8583.00 (7328.50)	Lagune Ebrie, Cote D'ivoire	Mireille *et al*., 2020
		47.73-89.33 (68.53)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
		10-13910 (6960.00)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		29.80-42.86 (36.33)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		48.30-161.10 (104.70)	Aghien Lagoon, Cote D'ivoire	Effebi *et al*., 2017
		110.00-40850.00 (13208.59)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		705.20-2440.00 (1572.60)	Aby Lagoon, Ivory Coast	Kambire *et al*., 2014
		(443.35)	Cross River Estuary, Nigeria	Ebong & John, 2021

Appendix E, continued

		(561.06)	Imo River Estuary, Nigeria	Ebong & John, 2021
		(574.13)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
		0.18-15.20 (7.69)	Lagos Lagoon	Oyeleke *et al.*, 2019
		50.47-8181.07 (4115.77)	Aby Lagoon, Code D'lvore	Assemian-niango *et al*, 2020
		25110.00-51250.00 (38180.00)	Vettar Estuary, India	Nanjappa *et al.*, 2023
	Mean	**9057.51**		
	Std Error	**2755.63**		
4.	pH	7.40-8.10 (7.78)	Tapi Estuary, India	Ujjania & Dubey, 2015
		1.89-8.50 (5.20)	Mangrove swamps, Nigeria	Lawson, 2011
		7.30-7.61 (7.46)	Gambia R. Estuary, Gambia	Adam *et al.*, 2018
		7.67-8.06 (7.87)	Chilika Lagoon, India	Mahapatro *et al.*, 2009
		5.40-6.80 (6.10)	Omoku Creek, Nigeria	Ewa *et al.*, 2011
		(7.64)	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		7.42-8.51 (7.76)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al.*, 2015
		6.40-7.60 (7.00)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017
		6.81-7.42 (7.2)	Qua Iboe Estuary	Akan & Nsikak, 2010
		6.34-7.01 (6.68)	Cross River Estuary	Akan & Nsikak, 2010
		7.77-8.22 (7.80)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		7.10-8.20 (7.65)	Arasalar Estuary, India	Raju *et al.*, 2015
		6.63-7.40 (7.02)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
		6.38-6.88 (6.33)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
		6.90-7.57 (7.24)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001

Appendix E, continued

		6.50-8.40 (7.45)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		6.90-8.00 (7.45)	Poxim River Estuary, Brazil	Nilin *et al.*, 2019
		6.06-6.90 (6.48)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		5.67-7.99 (6.83)	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017
		6.70-8.42 (7.40)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		6.96-7.80 (7.38)	Aby Lagoon	Kambire *et al.*, 2014
		(6.87)	Cross River Estuary, Nigeria	Ebong & John, 2021
		(6.84)	Imo River Estuary, Nigeria	Ebong & John, 2021
		(7.14)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
		7.86-8.51 (8.19)	Lagos Lagoon, Nigeria	Oyeleke *et al.*, 2019
		7.20-8.07 (7.64)	Muthupet Estuary, India	Suganthi *et al.*, 2020
		5.82-8.12 (6.97)	Rajakkamangalam Estuary, India	Banu *et al.*, 2018
		7.10-7.90 (7.50)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021
		5.44-8.90 (7.17)	Aby Lagoon, Code D'lvore	Assemian-niango *et al.*, 2020
		7.80-8.10 (7.95)	Vettar Estuary, India	Nanjappa *et al.*, 2023
		5.20-7.80 (6.50)	Potou Lagoon, Cote D'lvoire	Marthe *et al.*, 2015
		7.26-8.38 (7.82)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean	**7.20**		
	Std Error	**0.11**		
5.	Turbidity	64.00-89.00 (73.79)	Tapi Estuary, India	Ujjania & Dubey, 2015
		26.40-31.80 29.10)	Omoku Creek, Nigeria	Ewa *et al.*, 2011
		0.40-9.67 (5.28)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al.*, 2015
		0.24-1.33 (0.79)	Qua Iboe Estuary	Akan & Nsikak, 2010
		0.24-1.21 (0.23)	Cross River Estuary	Akan & Nsikak, 2010

Appendix E, continued

		0.90-10 (5.45)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		7.24-198.00 (102.62)	Lagoon Aghien, Ivory Coast	Ahoutou *et al.*, 2021
		11.67-28.00 (19.84)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
		5.00-8.60 (6.80)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		17.13-273.00 (145.07)	Aghien Lagoon, Cote D'ivoire	Effebi *et al*., 2017
		(74.80)	Cross River Estuary, Nigeria	Ebong & John, 2021
		(110.36)	Imo River Estuary, Nigeria	Ebong & John, 2021
		(97.31)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
		(50.10)	Lagos Lagoon, Nigeria	Nkwoji *et al.*, 2020
	Mean	**51.54**		
	Std error	**13.04**		
6.	Salinity (ppt)	0.20-16.75 (8.48)	Mangrove swamps, Nigeria	Lawson, 2011
		29.57-35.04 (32.31)	Gambia R. Estuary, Gambia	Adam *et al*., 2018
		0.80-14.00 (7.40)	Chilika Lagoon, India	Mahapatro *et al*., 2009
		18.70-20.80 (19.75)	Omoku Creek, Nigeria	Ewa *et al*., 2011
		0.11-32.00 (16.06)	Tapi Estuary, India	Nirmal Kumar et al., 2009
		(1.79)	Eastern Obolo Estuary, Nigeria	Udoh *et al*., 2013
		10.33-18.00 (15.39)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al*., 2015
		7.00-17.20 (12.10)	Upper Bonny Estuary, Nigeria	Ngah *et al*., 2017
		0.94-2.62 (1.78)	Qua Iboe Estuary	Akan & Nsikak, 2010
		0.87-1.97 (1.42)	Cross River Estuary	Akan & Nsikak, 2010
		10. 53-27.10 (18.82)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		5.50-34.00 (19.75)	Arasalar Estuary, India	Raju *et al*., 2015

Appendix E, continued

		3.10-4.58 (3.84)	Lagune Ebrie, Cote D'ivoire	Mireille et *al.*, 2020
		1.00-8.00 (4.50)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		0.00-35.00 (17.50)	Poxim River Estuary, Brazil	Nilin *et al.*, 2019
		(0.00)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		0.06-35.10 (14.43)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		0.28-1.28 (0.78)	Aby Lagoon, Ivory Coast	Kambire *et al.*, 2014
		180.00 - 290.00 (235.00)	Agniyar Estuary, India	Sugumaran, 2016
		0.00-160.00 (80.00)	Lagos Lagoon, Nigeria	Oyeleke *et al.*, 2019
		95.50-320.70 (416.20)	Muthupet Estuary, India	Suganthi *et al.*, 2020
		0.00-12.00 (6.00)	Rajakkamangalam Estuary, India	Banu *et al.*, 2018
		8.50-16.20 (12.35)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021
		0.00-4.75 (2.38)	Aby Lagoon, Code D'lvore	Assemian-niango *et al*, 2020
		14.53-25.56 (20.05)	Buenaventura Bay Estuary, Colombia	Duque *et al.*, 2020
		15.16-33.68 (24.42)	Vettar Estuary, India	Nanjappa *et al.*, 2023
		2.50-10.50 (6.50)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean	**37.00**		
	Std error	**16.99**		
7.	TDS	88.00-2560.00 (1324.00)	Mangrove swamps, Nigeria	Lawson, 2011
		196.00-231.00 (213.50)	Omoku Creek, Nigeria	Ewa *et al.*, 2011
		20,706.62	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		10185.00-26250.00 (18217.50)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017

Appendix E, continued

		14900.00-27840.00 (21370.00)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		3.74-5.36 (4.55)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
		33.00-62.00 (47.50)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
		0.33-6.94 (3.64)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		90.00-25000 (8467.65)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		(558.53)	Cross River Estuary, Nigeria	Ebong & John, 2021
		(654.10)	Imo River Estuary, Nigeria	Ebong & John, 2021
		(622.48)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
		80.00-355.00 (217.50)	Rajakkamangalam Estuary, India	Banu *et al.*, 2018
		9.77-16.91 (13.34)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021
		12560.00-25630.00 (19095.00)	Vettar Estuary, India	Nanjappa *et al.*, 2023
	Mean	**6101.06**		
	Std Error	**2286.89**		
8.	NO_3-N	0.21-54.46 (17.23)	Tapi Estuary, India	Ujjania & Dubey, 2015
		0.67-2.09 (1.38)	Tapi Estuary, India	Nirmal Kumar *et al.*, 2009
		0.15-2.51 (1.33)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017
		0.60-1.50 (1.05)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
		(0.00)	Arasalar Estuary, India	Raju *et al.*, 2015
		0.29-1.06 (0.68)	Lagoon Aghien, Ivory Coast	Ahoutou *et al.*, 2021
		0.27-1.42 (0.85)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
		(0.1)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
		0.02-0.28 (0.15)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
		0.02-17.30 (8.66)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012

Appendix E, continued

		0.07-0.10 (0.09)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		0.23-6.72 (3.48)	Aghien Lagoon, Cote D'ivoire	Effebi *et al*., 2017
		3.30-59.80 (10.54)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		(28.44)	Cross River Estuary	Ebong & John, 2021
		(41.27)	Imo River Estuary	Ebong & John, 2021
		(40.61)	Qua Iboe River Estuary	Ebong & John, 2021
		0.06-1.20 (0.63)	Agniyar Estuary, India	
		1.53-4.43 (2.98)	Muthupet Estuary, India	Suganthi *et al*., 2020
		0.02-0.31 (0.17)	Rajakkamangalam Estuary, India	Banu *et al*., 2018
		0.01-0.08 (0.05)	Pasur R. Estuary, Bangladesh	Shefat *et al*., 2021
		0.00-9.20 (4.60)	Aby Lagoon, Code D'lvore	Assemian-niango *et al*, 2020
		0.96-2.56 (1.76)	Buenaventura Bay Estuary, Colombia	Duque *et al*., 2020
		(8.50)	Lagos Lagoon, Nigeria	Nkwoji *et al*., 2020
		5.80-13.68 (9.74)	Vettar Estuary, India	Nanjappa *et al*., 2023
		0.17-7.14 (3.66)	Potou Lagoon, Cote D'lvoire	Marthe *et al*., 2015
		(11.11)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean	**7.68**		
	Std Error	**2.31**		
9.	Orthophosphate	0.17-0.88 (0.36)	Tapi Estuary, India	Ujjania & Dubey, 2015
		0.03-0.90 (0.47)	Tapi Estuary, India	Nirmal Kumar *et al*., 2009

Appendix E, continued

(0.05)	Andoni River Estuary, Nigeria	Ezekwe & Odoghotu
(0.00)	Arasalar Estuary, India	Raju *et al.*, 2015
0.04-0.12 (0.08)	Lagoon Aghien, Ivory Coast	Ahoutou *et al.*, 2021
0.36-1.90 (1.13)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
0.54-0.65 (0.59)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
0.09-0.47 (0.28)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
0.01-5.70 (2.86)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
(0.50)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
0.02-0.33 (0.18)	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017
0.01-0.01 (0.01	Tana River Estuary, Kenya	Ongore *et al.*, 2013
0.01-0.22 (0.12)	Sabaki River Estuary	Ongore *et al.*, 2013
0.01-1.68 (0.26)	Iyagbe Lagoon, Nigeria	Onyema, 2013
563.00-1193.00	Aby Lagoon, Ivory Coast	Kambire *et al.*, 2014
(1.37)	Cross River Estuary	Ebong & John, 2021
(3.11)	Imo River Estuary	Ebong & John, 2021
(2.87)	Qua Iboe River Estuary	Ebong & John, 2021
0.11-0.16 (0.14)	Agniyar Estuary, India	Sugumaran, 2016
0.53-1.49 (1.01)	Muthupet Estuary, India	Suganthi *et al.*, 2020
0.07-5.82 (2.95)	Pasur R. Estuary, Bangladesh	Shefat *et al.*, 2021
0.01-0.28 (0.15)	Aby Lagoon, Code D'lvore	Assemian-niango *et al*, 2020
0.06-0.18 (0.18)	Buenaventura Bay Estuary, Colombia	Duque *et al.* 2020

Appendix E, Continued

		5.30-195.00 (102.80)	Potou Lagoon, Cote D'lvoire	Marthe *et al.*, 2015
		(0.85)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean Std Error	**8.6** **5.39**		
10.	NH_4-N	0.18-1.59 (0.60)	Tapi Estuary, India	Ujjania & Dubey, 2015
		0.05-0.15 (0.10)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017
		0.03-0.46 (0.25)	Lagoon Aghien	Ahoutou *et al.*, 2021
		0.80-3.00 (1.9)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
		0.01-0.21 (0.11)	Epie Creek, Nigeria	Izonfuo & Bariweni, 2001
		0.25-3.00 (1.63)	Poxim River Estuary, Brazil	Nilin *et al.*, 2019
		0.04-2.69 (1.37)	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017
		0.00-0.01 (0.005)	Tana River Estuary, Kenya	Ongore *et al.*, 2013
		0.00-0.02 (0.01)	Sabaki River Estuary	Ongore *et al.*, 2013
		0.01-0.09 (0.05)	Aby Lagoon, Ivory Coast	Kambire *et al.*, 2014
		0.11-2.11 (1.11)	Pasur R. Estuary, Bangladesh	Shefat et al., 2021
		0.02-0.46 (0.24)	Aby Lagoon, Code D'lvore	Assemian-niango *et al*, 2020
		0.00-0.16 (0.08)	Potou Lagoon, Cote D'lvoire	Marthe *et al.*, 2015
	Mean Std Error	**0.57** **0.19**		
11.	TSS	220.00-22094.00 (11157.00)	Mangrove swamps, Nigeria	Lawson, 2011
		19.80-24.90 (22.35)	Omoku Creek, Nigeria	Ewa *et al.*, 2011
		(288.65)	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		3.68-6.31 (5.00)	Qua Iboe Estuary	Akan & Nsikak, 2010

		3.69-5.57 (4.63)	Cross River Estuary	Akan & Nsikak, 2010
		6.18-17.04 (11.61)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
		(20.00)	Lagune Ebrie, Cote D'ivoire	Mireille *et al.*, 2020
		2.50-50.00 (26.25)	Aghien Lagoon, Cote D'lvoire	Effebi *et al.*, 2017

		18.00-2310.00 (172.48)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		(17.72)	Cross River Estuary	Ebong & John, 2021
		(21.05)	Imo River Estuary	Ebong & John, 2021
		(23.25)	Qua Iboe River Estuary	Ebong & John, 2021
		(79.50)	Lagos Lagoon, Nigeria	Nkwoji *et al.*, 2020
	Mean	**142.27**		
	Std error	**87.63**		
12.	BOD	0.40-2.90 (1.60)	Tapi Estuary, India	Ujjania & Dubey, 2015
		38.00-59.00 (48.50)	Omoku Creek, Nigeria	Ewa et *al.*, 2011
		(0.33)	Eastern Obolo Estuary, Nigeria	Udoh *et al.*, 2013
		0.42-2.80 (1.70)	Bonny/New Calabar Estuary, Nigeria	Onojake *et al.*, 2015
		0.45-7.50 (3.98)	Upper Bonny Estuary, Nigeria	Ngah *et al.*, 2017
		0.87-2.21 (1.54)	Qua Iboe Estuary	Akan & Nsikak, 2010
		0.27-0.62 (0.45)	Cross River Estuary	Akan & Nsikak, 2010
		12.60-45.88 (29.24)	Nyong Estuary, Cameroon	Anselme *et al.*, 2018
		0.31-6.77 (3.54)	Epie Creek, Nigeria	
		8.00-65.00 (36.50)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		1.02-2.47 (1.75)	Taylor Creek, Nigeria	Alagoa & Leleye-Wokoma, 2012
		3.00-6.00 (4.50(	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017

		4.70-6.60 (5.65)	Tana River Estuary, Kenya	Ongore *et al.*, 2013
		2.60-5.80 (4.20)	Tana River Estuary, Kenya	Ongore *et al.*, 2013
		2.00-22.00 (7.15)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		11.70-28.16 (19.93)	Aby Lagoon, Ivory Coast	Kambire *et al.*, 2014
		(6.58)	Cross River Estuary, Nigeria	Ebong & John, 2021
		(6.76)	Imo River Estuary, Nigeria	Ebong & John, 2021
		(6.42)	Qua Iboe River Estuary, Nigeria	Ebong & John, 2021
		17.20-35.20 (26.20)	Lagos Lagoon, Nigeria	Nkwoji *et al.*, 2020
		3.50-10.00 (6.75)	Lagos Lagoon, Nigeria	Ajibare & Loto 2021
	Mean	**10.63**		
	Std error	**2.92**		
13.	COD	140.00-852.00 (666.00)	Tapi Estuary	Ujjania & Dubey, 2015
		12.00-120.00 (66.00)	Majidun Creek, Nigeria	Adesalu & Kunrunmi, 2012
		17.0-58.70 (37.85)	Aghien Lagoon, Cote D'ivoire	Effebi *et al.*, 2017
		8.00-89.00 (30.21)	Iyagbe Lagoon, Nigeria	Onyema, 2013
		27.37-64.29 (45.83)	Aby Lagoon, Ivory Coast	Kambire *et al.*, 2014
	Mean	**169.18**		
	Std Error	**124.35**		

Appendix F. Factor loadings and eigenvalues for Volta Estuary

Parameter	**PC1**	**PC2**	**PC3**
Temp (°C)	**0.90**	-0.05	0.12
DO (mg/L)	**0.68**	-0.22	-0.12
pH	**0.91**	0.08	0.10
EC (µS/cm)	-0.07	**0.93**	0.07
Turbidity (NTU)	-0.24	0.10	**0.66**
Salinity (ppt)	-0.10	**0.90**	0.09
TDS (mg/L)	0.35	-0.29	0.09
TSS (mg/L)	-0.24	-0.32	0.35
NO3-N(water) (mg/L)	-0.43	-0.15	0.22
NO3-N (sed)	-0.19	**-0.51**	0.07
Ortho (water) (mg/L)	-0.24	-0.09	-0.36
Ortho (sed) (mg/L)	0.05	-0.04	**0.61**
NH_4-N (mg/L)	-0.17	0.32	0.11
BOD (mg/L)	0.00	0.37	**-0.54**
COD (mg/L)	0.01	0.21	**0.75**
Eigenvalue	2.64	2.50	2.02
Variance (%)	17.60	16.65	13.48
Cum Var (%)	17.60	34.25	47.72

Extraction method: PCA. Rotation Method: Varimax with Kaiser Normalisation.

Bold values indicate the parameter with the highest correlation on each principal component

Appendix G. Pearson correlation matrix for the Volta Estuary

	Temp	DO	pH	EC	Turb	Sal	NO_3-N (sed)	Ortho (sed)	BOD	COD
Temp	1.00									
DO	**0.46**	1.00								
pH	**0.92**	**0.47**	1.00							
EC	-0.14	-0.18	0.03	1.00						
Turb	-0.13	-0.22	-0.10	0.13	1.00					
Sal	-0.16	-0.19	-0.01	**0.98**	0.14	1.00				
NO_3-N (sed)	-0.11	-0.13	-0.17	**-0.32**	-0.06	-0.27	1.00			
Ortho (sed)	-0.01	0.07	0.03	0.01	0.09	0.01	0.07	1.00		
BOD	-0.08	0.06	-0.01	0.27	-0.21	0.21	-0.17	**-0.30**	1.00	
COD	0.16	-0.17	0.05	0.15	**0.40**	0.13	-0.09	**0.35**	-0.18	1.00

The bold correlation coefficients are considered as strong correlations (r ≥-0.3 to r ≥+0.3)

Appendix H. Rotated factor loadings and eigenvalues for Ankobra Estuary

Parameter	**PC1**	**PC2**	**PC3**
Temp (°C)	0.20	-0.02	**0.88**
DO (mg/L)	0.12	0.15	**0.73**
pH	0.09	0.06	**0.87**
EC (µS/cm)	**0.93**	-0.12	0.28
Turbidity (NTU)	**0.84**	0.45	0.22
Salinity (ppt)	**0.92**	-0.11	0.29
TDS (mg/L)	-0.25	0.01	-0.36
TSS (mg/L)	-0.03	**0.97**	-0.07
NO_3-N(water) (mg/L)	-0.35	-0.12	0.09
NO_3-N (sed) (mg/L)	0.05	-0.06	-0.33
Ortho (water) (mg/L)	-0.05	0.50	**0.52**
Ortho (sed) (mg/L)	0.04	**0.96**	0.15
NH_4-N (mg/L)	**0.94**	-0.15	0.13
BOD (mg/L)	0.06	**0.98**	0.07
COD (mg/L)	0.26	-0.08	**0.50**
Eigenvalue	3.63	3.38	3.10
Variance (%)	24.17	22.51	20.64
Cum Var (%)	24.17	46.68	67.32

Extraction method: PCA. Rotation method: Varimax with Kaiser Normalisation.

Bold values indicate the parameter with the highest correlation on each principal component.

Appendix I. Pearson correlation matrix for the Ankobra Estuary

	Temp	DO	pH	EC	Sal	TSS	Turb	Ortho (water)	Ortho (sed)	NH_4-N	BOD	COD
Temp	1.00											
DO	**0.51**	1.00										
pH	**0.79**	**0.65**	1.00									
EC	**0.42**	**0.34**	**0.34**	1.00								
Sal	**0.38**	**0.32**	**0.32**	**0.79**	1.00							
TSS	**-0.41**	-0.14	**-0.41**	-0.23	-0.24	1.00						
Turb	-0.07	0.11	0.02	-0.13	**0.42**	0.23	1.00					
Ortho (water)	**0.41**	0.40	0.24	0.01	0.26	-0.16	**0.39**	1.00				
Ortho (sed)	0.13	0.23	0.21	-0.04	**0.49**	-0.13	**0.90**	**0.50**	1.00			
NH_4-N	**0.32**	0.16	0.15	**0.91**	**0.77**	-0.23	-0.18	0.01	-0.11	1.00		
BOD	0.09	0.17	0.17	-0.05	**0.52**	-0.14	**0.93**	**0.45**	**0.96**	-0.10	1.00	
COD	**0.47**	0.25	0.21	**0.34**	0.25	-0.11	-0.12	**0.41**	0.04	**0.33**	-0.08	1.00

The bold correlation coefficients are considered as strong correlations ($r \geq -0.3$ to $r \geq +0.3$)

Appendix J. Rotated factor loadings and eigenvalues for Kakum Estuary

Parameter	PC1	PC2	PC3
Temp (°C)	-0.32	**-0.83**	-0.07
DO (mg/L)	**0.76**	-0.07	0.28
pH	0.21	**-0.87**	0.01
EC (µS/cm)	**0.83**	0.17	-0.02
Turbidity (NTU)	**-0.75**	-0.09	0.35
Salinity (ppt)	**0.79**	0.27	-0.17
TDS (mg/L)	0.32	0.03	**-0.66**
TSS (mg/L)	-0.32	0.00	**0.81**
NO_3-N(water) (mg/L)	0.22	0.20	0.44
NO_3-N (sed) (mg/L)	0.09	**0.54**	0.03
Ortho (water) (mg/L)	-0.04	-0.05	0.39
Ortho (sed) (mg/L)	-0.13	0.08	-0.23
NH_4-N (mg/L)	-0.07	**0.60**	-0.51
BOD (mg/L)	0.01	0.03	**0.73**
COD (mg/L)	0.30	**0.85**	-0.07
Eigenvalue	2.97	2.97	2.51
Variance (%)	19.80	19.78	16.74
Cum Var (%)	19.80	39.58	56.32

Extraction method: PCA. Rotation method: Varimax with Kaiser Normalisation.

Bold numbers indicate the parameter with the highest correlation on each principal component.

Appendix K. Pearson correlation matrix for the Kakum Estuary

	Temp	DO	pH	EC	Turb	Sal	TDS	TSS	NO_3-N (sed)	NH-N	BOD	COD
Temp	1											
DO	**-0.34**	1.00										
pH	**0.68**	0.20	1.00									
EC	-0.28	**0.41**	0.02	1.00								
Turb	**0.31**	**-0.41**	-0.01	**-0.57**	1.00							
Sal	**-0.34**	**0.40**	-0.10	**0.85**	**-0.60**	1.00						
TDS	-0.10	0.15	0.15	0.20	-0.25	0.30	1.00					
TSS	0.14	-0.03	0.02	-0.16	**0.68**	-0.23	**-0.54**	1.00				
NO_3-N (sed)	**-0.42**	0.14	**-0.36**	0.13	-0.09	0.15	0.13	-0.01	1.00			
NH_4-N	**-0.37**	-0.29	**-0.46**	0.13	-0.19	0.28	0.29	**-0.30**	0.03	1.00		
BOD	-0.07	0.13	0.04	0.00	0.16	-0.10	**-0.38**	**0.57**	0.13	-0.33	1.00	
COD	**-0.73**	0.08	**-0.55**	**0.40**	-0.28	**0.49**	0.24	-0.11	**0.43**	**0.55**	-0.04	1.00

The bold correlation coefficients are considered as strong correlations ($r \geq -0.3$ to $r \geq +0.3$)

Appendix L. Rotated factor loadings and eigenvalues for Whin Estuary

Parameter	PC1	PC2	PC3
Temp (°C)	0.20	**0.88**	-0.26
DO (mg/L)	0.43	**0.79**	0.12
pH	0.26	**0.81**	-0.19
EC (µS/cm)	**0.95**	0.20	-0.02
Salinity (ppt)	**0.85**	0.33	0.13
TDS (mg/L)	**0.95**	0.20	-0.02
TSS (mg/L)	**-0.76**	-0.19	0.44
Turbidity (NTU)	**-0.74**	-0.20	0.44
NO_3-N (water) (mg/L)	0.16	**-0.68**	0.07
NO_3-N (sed) (mg/L)	-0.25	-0.18	**0.51**
Ortho (water) (mg/L)	-0.32	0.10	**-0.81**
Ortho (sed) (mg/L)	-0.08	-0.18	-0.45
NH_4-N (mg/L)	-0.35	-0.15	**0.61**
BOD (mg/L)	**0.71**	-0.12	0.45
COD (mg/L)	0.31	**0.51**	0.18
Eigenvalue	4.83	3.17	2.24
Variance (%)	32.19	21.10	14.94
Cum Var (%)	32.19	53.29	68.23

Extraction method: PCA. Rotation method: Varimax with Kaiser Normalisation. Bold values indicate the parameter with the highest correlation on each principal component.

Appendix M. Pearson Correlation matrix for the Whin Estuary

	Temp	DO	pH	EC	Sal	TDS	TSS	Turb	NO_3-N (water)	NO_3-N (sed)	Ortho (water)	NH_4-N	BOD	COD
Temp	1													
DO	**0.77**	1.00												
pH	**0.84**	**0.71**	1.00											
EC	**0.36**	**0.55**	**0.44**	1.00										
Sal	**0.42**	**0.59**	**0.43**	**0.90**	1.00									
TDS	**0.36**	**0.56**	**0.44**	**1.00**	**0.90**	1.00								
TSS	**-0.42**	**-0.45**	**-0.36**	**-0.71**	**-0.60**	**-0.71**	1.00							
Turb	**-0.41**	**-0.45**	**-0.37**	**-0.68**	**-0.57**	**-0.68**	**1.00**	1.00						
NO_3-N (water)	**-0.49**	**-0.36**	**-0.37**	0.04	-0.22	0.03	0.11	0.12	1.00					
NO_3-N (sed)	**-0.33**	-0.19	-0.25	-0.26	-0.20	-0.27	0.27	0.24	0.04	1.00				
Ortho (water)	0.18	-0.16	0.15	-0.25	**-0.36**	-0.25	-0.23	-0.25	-0.25	-0.15	1.00			
NH_4-N	**-0.38**	-0.13	**-0.32**	**-0.37**	-0.26	**-0.37**	**0.39**	**0.36**	0.03	**0.59**	-0.19	1.00		
BOD	-0.05	0.22	0.05	**0.57**	**0.59**	**0.57**	**-0.32**	**-0.31**	0.09	0.05	**-0.55**	0.08	1.00	
COD	**0.39**	**0.44**	**0.43**	**0.38**	**0.35**	**0.37**	-0.23	-0.22	-0.11	-0.14	-0.25	-0.18	0.13	1.00

The bold correlation coefficients are considered strong correlations ($r \geq -0.3$ to ≥ 0.3