

CHRISTIAN-ALBRECHTS-UNIVERSITÄT ZU KIEL

Nutrients in Kenya's Coastal Waters: Analyzing Sources, Pathways, and Ecological Implications

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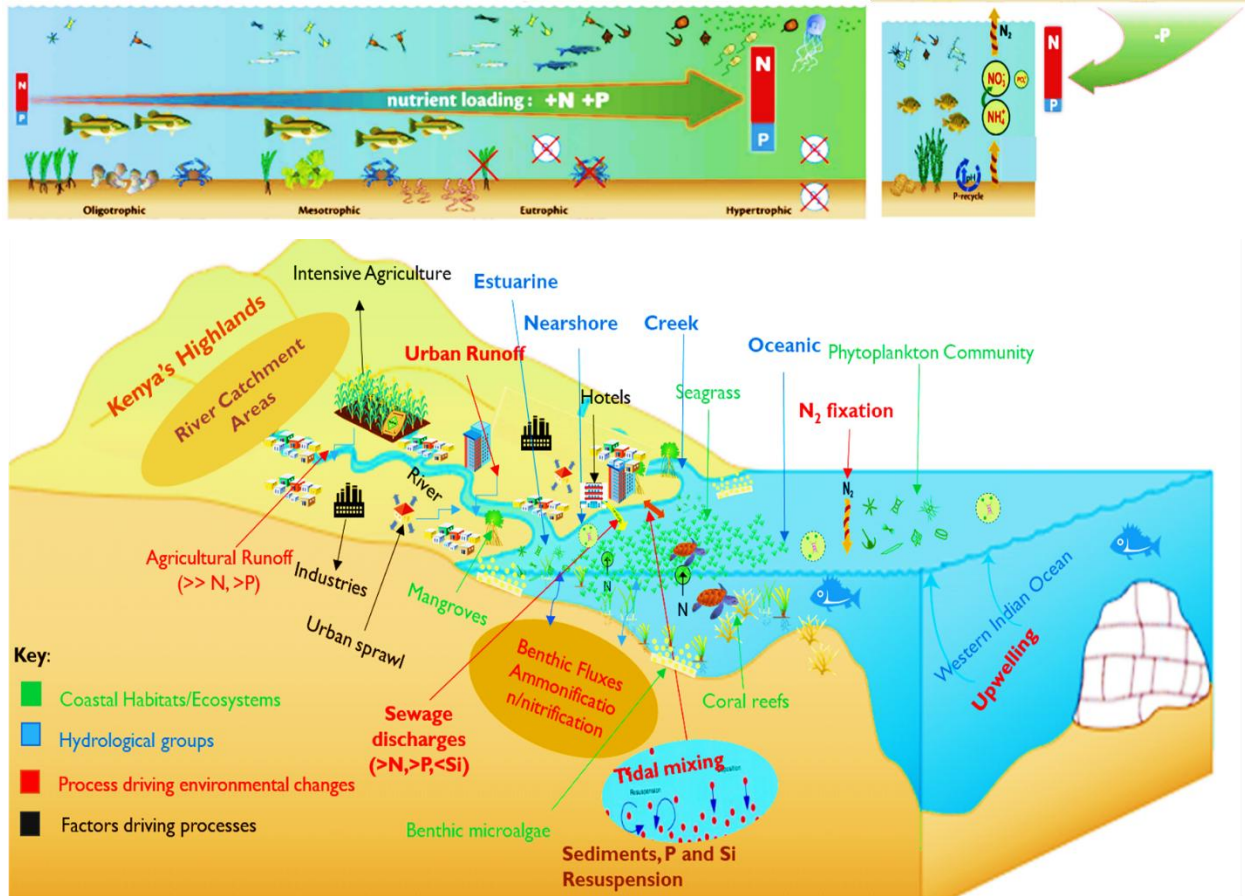
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Dedication

To my late dad who believed in me differently, to my mom who gave her all, to my high school teacher Mr. ABOW who saw my potential against all odds, to my sister Milly who cried on my behalf, and to my kids, Pravyne, Adriel, and Eliana whose love fueled my dreams. This achievement is for you all!

"Faith and effort turn impossible into I'm possible."

Thesis Graphical Overview



Graphical abstract illustrating the ecological habitats, hydrological groupings, nutrient sources, and driving forces along Kenya's coast.

Ecological implications (Changes in food webs; Top Image): Progression from nutrient enrichment \rightarrow algal blooms/HABs \rightarrow hypoxia \rightarrow ecosystem collapse. Modified from Glibert (2015). Bottom Image is the author's creation.

Abstract

Coastal ecosystems around the globe are experiencing heightened pressure from human activities, resulting in nutrient enrichment that poses significant risks of ecological degradation. This phenomenon is evident in Kenya's coastal waters, where rising human-induced nutrient inputs have raised alarm over the declining health of vital coastal and near-shore ecosystems, including coral reefs and seagrass meadows, and the increasing development of potentially harmful microalgae species. This dissertation assessed the dynamics of anthropogenic nutrients in Kenya's coastal waters, emphasizing their effects on phytoplankton communities and the associated risks of harmful algal blooms. It is hypothesized that nutrient enrichment from anthropogenic sources reshapes phytoplankton assemblages while promoting the growth of harmful microalgae. The study investigated spatial and temporal trends in inorganic nutrients and phytoplankton composition, identified nutrient sources and pathways, and analyzed relationships between nutrients and harmful microalgae species. Phytotoxic assessments for Amnesic Shellfish Poisoning (ASP) and Paralytic Shellfish Poisoning (PSP) risks were also conducted. The assessments were conducted through field surveys, laboratory analyses, and literature reviews over a one-year period (August 2021–July 2022).

The study revealed distinct spatio-temporal patterns in nutrient distribution, with highest concentrations found in estuaries and tidal creeks near urban and tourist areas. Tudor Creek consistently showed elevated levels of temperature, salinity, turbidity, and nutrients compared to Kilindini Creek and open waters. Seasonal variations in dissolved inorganic nitrogen (DIN) and silicate correlated negatively with salinity and positively with turbidity, indicating nutrient-rich freshwater influx during wet periods. A significant shift from natural nitrogen-limited conditions to a DIN-enriched state was observed, with phosphorus and silica becoming limiting factors. The observed changes in nutrient composition and availability have led to substantial shifts in phytoplankton dynamics, with a notable increase in the frequency and intensity of harmful algal blooms. These results support the initial hypothesis that anthropogenic nutrient inputs are significantly altering phytoplankton community structure, favoring the proliferation of potentially harmful microalgae. The study identified 59 taxa of potentially harmful microalgae, with 47% capable of producing toxins however; no phytotoxin was detected during the study. Anthropogenic sources, including urban sewage, industrial effluents, and agricultural runoff, were major contributors to nutrient enrichment. Submarine groundwater discharge and atmospheric deposition in high-traffic areas also emerged as crucial nutrient input pathways. These findings highlight the need for conservation and

management of anthropogenic nutrient enrichment in Kenya's coastal waters. The study contributes to understanding nutrient dynamics and their impact on phytoplankton communities in tropical coastal ecosystems, emphasizing the importance of monitoring and mitigating anthropogenic influences to protect vital habitats and marine biodiversity.

Zusammenfassung

Küstenökosysteme weltweit stehen unter erhöhtem Druck durch menschliche Aktivitäten, was zu einer Nährstoffanreicherung führt, die erhebliche Risiken für die ökologische Degradation mit sich bringt. Dieses Phänomen zeigt sich auch in den Küstengewässern Kenias, wo steigende anthropogene Nährstoffeinträge in Verdacht stehen eine abnehmende Gesundheit lebenswichtiger Küsten- und küstennaher Ökosysteme, einschließlich Korallenriffe und Seegraswiesen, sowie die zunehmende Entwicklung potenziell schädlicher Mikroalgenarten hervorgerufen zu haben. Diese Dissertation untersuchte die Dynamik von Nährstoffen in Kenias Küstengewässern mit Schwerpunkt auf deren Auswirkungen auf Phytoplanktongemeinschaften und die damit verbundenen Risiken schädlicher Algenblüten. Die zentrale Hypothese der Arbeit ist, dass die Nährstoffanreicherung aus anthropogenen Quellen die Zusammensetzung des Phytoplanktons verändert und gleichzeitig das Wachstum schädlicher Mikroalgen fördert. Die Studie untersuchte räumliche und zeitliche Trends bei anorganischen Nährstoffen und der Zusammensetzung des Phytoplanktons, identifizierte Nährstoffquellen und -wege und analysierte die Beziehungen zwischen Nährstoffen und schädlichen Mikroalgenarten. Zudem wurden phytotoxische Bewertungen für die Risiken der Amnesischen Schalentiervergiftung (ASP) und der Paralytischen Schalentiervergiftung (PSP) durchgeführt. Die Geländearbeit wurde im Zeitraum August 2021–Juli 2022 durchgeführt.

Die Studie zeigte deutliche räumlich-zeitliche Muster in der Nährstoffverteilung, wobei die höchsten Konzentrationen in Ästuaren und Gezeitenkanälen in der Nähe von städtischen Gebieten und Hotels gefunden wurden. Der Tudor Creek wies im Vergleich zum Kilindini Creek und Messtellen im offenen Küstengewässer durchweg erhöhte Werte für Temperatur, Salzgehalt, Trübung und Nährstoffe auf. Saisonale Schwankungen des gelösten anorganischen Stickstoffs (DIN) und Silikats korrelierten negativ mit dem Salzgehalt und positiv mit der Trübung, was auf einen nährstoffreichen Süßwasserzufluss während feuchter Perioden hindeutet. Es wurde eine signifikante Verschiebung von natürlichen stickstofflimitierten Bedingungen zu einem DIN-angereicherten Zustand beobachtet, wobei Phosphor und Silizium zu limitierenden Nährstoffen wurden. Die beobachteten Veränderungen in der Nährstoffzusammensetzung und -verfügbarkeit haben zu erheblichen Verschiebungen in der Phytoplanktodynamik geführt, mit einer bemerkenswerten Zunahme der Häufigkeit und Intensität schädlicher Algenblüten. Diese Ergebnisse stützen die anfängliche Hypothese, dass anthropogene Nährstoffeinträge die Struktur der Phytoplanktongemeinschaft signifikant verändern und die Vermehrung potenziell schädlicher Mikroalgen begünstigen. Die Studie identifizierte 59 Taxa potenziell schädlicher Mikroalgen, von denen 47% in der Lage sind, Toxine zu produzieren. Allerdings

wurden keine Phytotoxine nachgewiesen. Anthropogene Quellen, einschließlich städtischer Abwässer, industrieller Abflüsse und landwirtschaftlicher Abflüsse, waren Hauptverursacher der Nährstoffanreicherung. Auch der submarine Grundwasserabfluss und atmosphärische Depositionen in Gebieten mit hohem Verkehrsaufkommen erwiesen sich als wichtige Nährstoffeintragswege. Diese Erkenntnisse unterstreichen die Notwendigkeit des Schutzes und Managements der anthropogenen Nährstoffanreicherung in Kenias Küstengewässern. Die Studie trägt zum Verständnis der Nährstoffdynamik und ihrer Auswirkungen auf Phytoplanktongemeinschaften in tropischen Küstenökosystemen bei und betont die Bedeutung der Überwachung und Minderung anthropogener Einflüsse zum Schutz lebenswichtiger Lebensräume und der marinen Biodiversität.

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Preface

This cumulative thesis focused on assessment of anthropogenic nutrient pollution and harmful algal blooms (HABs) in Kenya's coastal and marine waters, where there is increased interest in blue economy investments including fisheries, and aquaculture to increase the country's economic prospects. The PhD work was conducted at Leibniz Centre for Tropical Marine Research (ZMT) and Technical University of Mombasa (TUM) from October 2020 to February 2025. Three specific objectives are addressed in this thesis, organized into independent publications and chapters. Two objectives have been published in refereed journals, while two manuscripts are currently under review, with preprints available.

Chapter 2: Publication 1

Oduor, N. A., Munga, C. N., Ong'anda, H. O., Botwe, P. K., & Moosdorf, N. (2023). Nutrients and harmful algal blooms in Kenya's coastal and marine waters: A review. *Ocean & Coastal Management*, 233(December 2022), 106454. <https://doi.org/10.1016/j.ocecoaman.2022.106454>

Chapter 3: Publication 2

Oduor, N. A., Munga, C. N., Imbayi, L. K., Botwe, P. K., Nyanjong, E. O., Muthama, C. M., Mise, N. A., & Moosdorf, N. (2024). Anthropogenic nutrients and phytoplankton diversity in Kenya's coastal waters: An ecological quality assessment of sea turtle foraging sites. *Marine Pollution Bulletin*, 199(October 2023), 115897. <https://doi.org/10.1016/j.marpolbul.2023.115897>

Chapter 4: Publication 3

Oduor, N. A., Botwe, P., Munga, C. N., Kremp, A., Harrison, O., & Moosdorf, N. (2024). Potentially Harmful Algal Bloom (HAB) Causing Species in Kenya's Coastal Waters: Diversity, Distribution and Effect Implications. Preprints of the previous version are available at <https://doi.org/10.2139/ssrn.4869883>. The chapter's revised article is under Preparation for submission to the *Journal of Ocean & Coastal Management*.

Chapter 4 associated Data: **Oduor, N. A.,** Imbayi, K. L., Mulusa, V. O., Odhiambo, M., & Nyanjong, E. (2024). Domoic acid (DA), Paralytic Shellfish Toxins (PST), the abundance of potential DA and PST microalgae causative agents, and environmental data in Kenya's coastal waters in July 2022, *PANGAEA-Data Publisher for Earth & Environmental Science*. Available at: <https://doi.org/https://doi.pangaea.de/10.1594/PANGAEA.964978>

Chapter 5: Publication 4

Oduor, N. A., Kay, L. D., Cátia M. E. Von Ahn, & Moosdorf, N. Distribution of nutrients and organic matter in Mombasa Coastal Waters. Manuscript in Preparation for *Marine Pollution Bulletin*.

List of Abbreviations & Acronyms

<i>Abbreviation</i>	<i>Definition</i>
<i>ASCLME</i>	— UNDP-GEF Agulhas and Somali Current Project
<i>ASP</i>	— Amnesic Shellfish Poisoning
<i>BMU</i>	— Beach Management Unit
<i>Chl a</i>	— Chlorophyll a
<i>CBO</i>	— Community-Based Organization
<i>CFP</i>	— Ciguatera Fish poisoning
<i>DIC</i>	— Dissolved inorganic carbon
<i>DIN</i>	— Dissolved inorganic nitrogen
<i>DL</i>	— Detection limit
<i>DIP</i>	— Dissolved inorganic phosphate
<i>DO</i>	— Dissolved Oxygen
<i>DSi</i>	— Dissolved silicate
<i>DSP</i>	— Diarrheic Shellfish Poisoning
<i>EEZ</i>	— Exclusive Economic Zone
<i>EQR</i>	— Ecological quality ratios
<i>EQS</i>	— Ecological Quality Status
<i>FP</i>	— Fibro-papillomatosis
<i>FG</i>	— Functional Group
<i>GCDN</i>	— Gill clogging dinoflagellates
<i>GCDT</i>	— Gill clogging diatoms
<i>GCSL</i>	— Gill clogging silicoflagellates
<i>HABs</i>	— Harmful algal blooms
<i>HBDN</i>	— High biomass-producing dinoflagellate
<i>HBDT</i>	— High biomass-producing diatoms
<i>HSP</i>	— Hepatotoxic shellfish poisoning
<i>KeFS</i>	— Kenya Fisheries Service
<i>KLD</i>	— Kilindini Creek
<i>KMFRI</i>	— Kenya Marine and Fisheries Research Institute
<i>KWS</i>	— Kenya Wildlife Service

<i>LOQ</i>	— Limit of quantification
<i>MMNPR</i>	— Mombasa Marine National Park and Reserve
<i>MPAs</i>	— Marine Protected Areas
<i>NEM</i>	— Northeast monsoon winds
<i>NO_x⁻</i>	— Nitrates + Nitrites
<i>OPN</i>	— Open waters
<i>POC</i>	— Particulate organic carbon
<i>PON</i>	— Particulate organic nitrogen
<i>PSP</i>	— Paralytic Shellfish Poisoning
<i>Ref.con</i>	— Reference condition
<i>AcDev</i>	— Actual Deviation
<i>SDGs</i>	— Sustainable Development Goals
<i>SEM</i>	— Southeast Monsoon winds
<i>SGD</i>	— Submarine Groundwater Discharge
<i>TCG</i>	— Turtle Conservation Group
<i>TDC</i>	— Tudor Creek
<i>TKM</i>	— Tudor-Kilindini Mouth
<i>TPCY</i>	— Toxin-producing cyanobacteria
<i>TPDN</i>	— Toxin-producing dinoflagellates
<i>TPDT</i>	— Toxin-producing diatoms
<i>WIO</i>	— Western Indian Ocean
<i>WIOHAB</i>	— Western Indian Ocean Harmful Algal Blooms
<i>WIOMSA</i>	— Western Indian Ocean Marine Science Association
<i>WMNPR</i>	— Watamu Marine National Park and Reserve
<i>ZMT</i>	— Leibniz Centre for Tropical Marine Research

Definition of terms

<i>Term</i>	<i>Definition</i>
Algal bloom	— The rapid increase of the algal population in the aquatic environment
Anthropogenic nutrients	— Nitrogen and phosphorus originating from human activities
Ecosystem health/ quality	— Ecological structure and functioning of aquatic ecosystems
Essential nutrients	— Nitrogen, Phosphorus, and silicates needed for primary production
Estuaries	— River mouths or areas where river waters are drained/mix with ocean waters
Eutrophication	— Accelerated growth and accumulation of algal biomass in an ecosystem due to nutrient enrichment (nitrogen/phosphorus)
Nearshore	— Areas located < 500m from shore
Nutrient	— Essential elements, specifically N, P, and Si, required by microalgae for energy and cell development
Nutrient enrichment	— Supply of nutrients in the coastal waters
Oceanic	— Island areas located \geq 500m from shore with less human effects.
Runoff	— Overflow of rainwater from a farm or urban areas.
Tidal Creeks	— Tidal channels with saline water and low river influence

Chapter One: General Introduction

State of Research

1.1.Chapter Summary

Coastal ecosystems play vital ecological and socio-economic roles in supporting human well-being, but face increasing threats from nutrient pollution due to anthropogenic activities. Excessive supply of nitrogen (N) and phosphorus (P) is altering global nutrient dynamics and concentrations in coastal waters worldwide. This has accelerated eutrophication and promoted the proliferation of harmful algal blooms (HABs), with adverse ecological, economic, and public health consequences. Global studies have documented a geographical expansion and increased toxicity of HABs. However, tropical coastal regions in developing African countries, characterized by year-round high temperatures and rapid coastal development, remain understudied despite their potential vulnerability to these phenomena. This study investigated anthropogenic nutrient fluxes in Kenya's coastal waters, assessing their impact on phytoplankton community structure and examining the ecological implications for microalgae diversity, distribution, and risks of HABs formation. Kenya is a tropical developing country, with a coastline spanning 640 km in the Western Indian Ocean's East African Current Large Marine Ecosystem. The region hosts critical habitats supporting endangered species and sustaining vulnerable coastal communities' livelihoods. This study thus contributes to the understanding of nutrient dynamics, phytoplankton assemblages, and potentially harmful microalgae species in tropical developing regions. This chapter provides a comprehensive overview of the thesis research project, encompassing the background information, preceded by the research problem, aims, specific objectives and questions, significance, and finally its scope and limitations.

1.2.Background

The 20th Century witnessed unprecedented human development and urbanization, resulting in a massive increase in nitrogen and phosphorus emissions, leading to widespread nutrient enrichment in coastal waters worldwide. This enrichment compromises coastal ecosystem resilience, increasing vulnerability to emerging threats like ocean acidification, warming of sea surface waters, rising sea levels, and other associated climate change impacts (Adams *et al.*, 2020; Heisler *et al.*, 2008; IPCC, 2014). Additionally, it is directly linked to eutrophication, which causes significant degradation of coastal ecosystem health and functionality (Bastos *et al.*, 2022; Gu *et al.*, 2022; Gilbert, 2020; Li *et al.*, 2019). Eutrophication has severe consequences, including increased occurrences of harmful algal

blooms (HABs) (e.g., [Glibert, 2017](#)), development of dead zones ([Diaz et al., 2019](#)), degradation and losses of benthic habitats ([Hoegh-Guldberg et al., 2017](#); [Waycott et al., 2009](#)).

Nitrogen (N) and phosphorus (P) are essential nutrients in aquatic ecosystems, driving phytoplankton production and cellular functions ([Razaq et al., 2017](#)). These elements are crucial for growth and development in all organisms and limit primary production in aquatic environments ([Glibert et al., 2018](#)). Specifically, N is vital for protein synthesis, while P is essential for DNA formation, cell membranes, and energy production (ATP) ([Davidson et al., 2012](#)). Their availability and forms in aquatic environments significantly influence community structure and ecological processes, indicating their importance in coastal and marine ecosystems. Besides N and P, silicon (Si) significantly influences phytoplankton productivity and community structure, since it is crucial for diatom growth due to its role in frustule formation ([Gilpin et al., 2004](#)). In this study, the term "critical" or "essential" nutrients refers to nitrogen (N), phosphorus (P), and silicon (Si), which are fundamental for primary production in aquatic ecosystems.

1.2.1. Nutrient Balance and Limitation

The essential nutrients occur in aquatic systems in proportions similar to phytoplankton cells ([Turner et al., 1998](#)). While nitrogen (N) limitation is common in coastal waters and phosphorus (P) in freshwaters, some studies have revealed excess N relative to P or co-limitation in coastal areas with high nitrogen inputs ([Turner & Rabalais, 2013](#)). Additionally, recent evidence challenges traditional nutrient limitation patterns, suggesting that N limitation also occurs in estuarine, temperate, and polar coastal ecosystems ([Glibert, 2017](#); [Schindler et al., 2016](#)), expanding our understanding of nutrient dynamics in diverse aquatic environments.

Nutrient enrichment from human activities have significantly altered nutrient dynamics in coastal ecosystems, particularly affecting N and P balance ([Michalak et al., 2013](#)). Major sources of excessive nutrient enrichment include sewage, industrial discharges, agricultural runoff, and groundwater leachates ([Malone & Newton, 2020](#)). Over the past century, anthropogenic activities have doubled global dissolved inorganic nitrogen supply ([Jickells et al., 2017](#)) and increased overall N loading tenfold ([Adams et al., 2020](#)), dramatically impacting coastal nutrient loads. The addition of limiting nutrients, especially N and P, is considered the primary cause of eutrophication and harmful algal blooms in coastal waters ([Dai et al., 2023](#); [Glibert et al., 2018](#)). While these nutrients are typically

beneficial, anthropogenic inputs often result in imbalanced N: P: Si ratios compared to natural requirements for primary production (Turner *et al.*, 1998). Human activities also introduce chemically reduced forms of nitrogen, such as urea and ammonium (Jickells *et al.*, 2017), which are known to favor the growth of specific non-diatom species, particularly the harmful ones (Glibert *et al.*, 2012, 2018). Nutrient source and pathway thus significantly influence the stoichiometric composition and form of available essential nutrients in coastal waters, potentially disrupting ecosystem balance and function (Glibert, 2017; IOC-UNESCO, 2008).

1.2.2. Nutrient Sources and Pathways

Different nutrient sources are characterized by specific nutrient profiles: agricultural runoff is high in urea (Glibert *et al.*, 2006, 2005), atmospheric deposition contributes soluble reactive N (Jickells *et al.*, 2017), and domestic sewage is rich in ammonium-NH₄⁺ and phosphate-PO₄³⁻ concentrations (Oduor *et al.*, 2023; Tett *et al.*, 2003). Organic matter from land-based activities can promote NH₄⁺ generation in coastal waters through bacterial decomposition under anoxic conditions (Hensen *et al.*, 2006; Kaiser *et al.*, 2015). Surface runoff and groundwater supplied through river and submarine groundwater discharges contribute elevated nitrates and silicates relative to phosphates. However, these sources can differ significantly in phosphate contributions due to its varying mobility in different environments (Mwaura *et al.*, 2017; Rodellas *et al.*, 2014). Additionally, nutrients supplied to the surface seawater from pore water and bottom sediments through advection and upwelling processes are enriched with nitrates and silicates, respectively (Jacobs *et al.*, 2020; Sospedra *et al.*, 2018). Besides the sources, nutrient conditions in coastal waters are significantly influenced by anthropogenic activity intensity, transport pathways, and ecosystem geomorphology (IOC-UNESCO, 2008). While coastal systems near populated urban areas and large catchment estuaries typically show higher nutrient pollution levels (Li *et al.*, 2023; Mwaura *et al.*, 2017; Paerl *et al.*, 2014), smaller populations can also exhibit comparable groundwater and coastal seawater nutrient levels (Adyasari *et al.*, 2018). This complexity highlights the challenge in understanding diverse nutrient sources contributing to coastal water inputs, emphasizing the need for comprehensive assessment of local and regional factors.

1.2.3. Effects of Nutrient Enrichment in Coastal Waters

Human activities have drastically altered coastal nutrient dynamics, leading to pollution and HABs globally. Changes in dissolved inorganic N, P, and Si availability, form and composition are reshaping

coastal biodiversity and phytoplankton communities, with both nutrient excess and limitations impacting phytoplankton structure (Glibert, 2017; Glibert *et al.*, 2018). Nutrients from untreated sewage, wastewater, and agricultural runoff, with imbalanced proportions and reduced nitrogen forms, can promote harmful microalgae proliferation, potentially increasing the abundance and toxicity of certain species (Glibert, 2017; Glibert & Burkholder, 2011). Phosphorus limitation can shift communities from diatoms to dinoflagellates (Lu *et al.*, 2022; Nishimura *et al.*, 2020), while nitrogen limitation can lead to nitrogen-fixing cyanobacteria dominance (Bergman *et al.*, 2013). Additionally, nutrient enrichment, particularly with chemically reduced nitrogen forms, can promote blooms of cyanobacteria, dinoflagellates, and chlorophytes. Elevated NH_4^+ relative to NO_3^- favors non-diatomic populations due to their higher affinity for NH_4^+ (Glibert *et al.*, 2018; Lundgren *et al.*, 2016). This shift occurs because NH_4^+ uptake requires less energy than NO_3^- utilization, and NO_3^- uptake is repressed when NH_4^+ exceeds 1 $\mu\text{mol/L}$ (L'Helguen *et al.*, 2008). Consequently, anthropogenic nutrient enrichment often leads to persistent eutrophication, characterized by large blooms of low species diversity, predominantly harmful species (Glibert, 2020). Secondary effects include hypoxia, reduced primary production, food web alterations, ecosystem modifications, and decreased fisheries production in coastal ecosystems (Berdalet *et al.*, 2016; Turner & Rabalais, 2013).

1.3. Research Problem

Since the late 20th Century, nutrient pollution, eutrophication, and harmful algal blooms (HABs) have garnered increasing attention due to their escalating ecological, socio-economic, and public health impacts. Overall, there is extensive literature documenting global rise in nutrient enrichment eutrophication and HABs occurrences in coastal waters (Adams *et al.*, 2020; Glibert, 2020). However, research has predominantly focused on developed nations, creating a critical data gap in developing countries, particularly in tropical regions (Painter *et al.*, 2022). This scarcity of research is especially concerning for Africa, which is notably absent from most recent global eutrophication and HABs status reports (de Raús Maúre *et al.*, 2021; Hallegraeff *et al.*, 2021). The lack of information is alarming given these regions' vulnerability to cultural eutrophication and HABs events, especially in the context of global climate change impacts (Rosenzweig *et al.*, 2008).

Tropical coastal developing countries face heightened vulnerability to nutrient pollution and HABs due to several factors. These regions often rely heavily on coastal resources, with up to 80% of local livelihoods depending on marine ecosystems (Ocean & Coasts, 2024). Warm temperatures and sun light intensities also create ideal conditions for algal growth, particularly harmful species (Ho &

Michalak, 2020), while rapid population growth and urbanization increase nutrient inputs from poorly managed waste and agricultural runoff (Adyasari *et al.*, 2018; Glibert, 2020). Additionally, limited research capacity hinders comprehensive water quality assessments and HABs monitoring in many developing countries. These factors underscore the need for research to identify and mitigate risks associated with nutrient enrichment, eutrophication, and HABs to safeguard coastal communities' livelihoods, food security, and public health (Berdalet *et al.*, 2016; Glibert, 2020).

In East Africa, including Kenya, anthropogenic nutrient pollution and HABs have earlier emerged as pressing environmental issues due to rapid urban and industrial expansion (UNEP-GEF, 2006). Studies have revealed elevated nutrient levels and occurrences of potentially harmful microalgae in coastal waters near major towns, tourism centers, and estuaries (Kiteresi *et al.*, 2013; Mwaura *et al.*, 2017). However, existing research is often localized and sporadic, lacking comprehensive data to guide effective management decisions. Limitations include infrequent sampling, narrow focus on either nutrients or phytoplankton, restricted analysis of water column nutrients, and unsubstantiated assumptions about nutrient sources (e.g. Ogongo *et al.*, 2015; Ongore *et al.*, 2013).

Whereas some few studies in Kenya have examined nutrient sources and their connections to biological communities (Mwaura *et al.*, 2017; Ohowa *et al.*, 1997), most have not investigated the pathways and contributions of various inputs essential for effective management. Existing studies have linked the development of potentially harmful microalgae to anthropogenic nutrient enrichment (Kiteresi *et al.*, 2013; Ochieng *et al.*, 2015; Ogongo *et al.*, 2015), but lack a comprehensive understanding of nutrient conditions, including quantity, forms, and stoichiometric composition. Furthermore, the drivers of eutrophication and HABs are often location-specific, influenced by hydrological, geomorphological, and internal factors, as well as external pressures like climate change and overfishing (GEOHAB, 2016; Wells & Karlson, 2018). This data gap hampers the development of effective strategies to manage nutrient enrichment, eutrophication, and harmful algal blooms, underscoring the need for systematic assessments of coastal water quality and ecosystem health, which this study in Kenya aims to address.

1.4. Research Aims, Objectives, and Questions

This thesis aimed to evaluate anthropogenic nutrient dynamics and their ecological impacts in Kenya's coastal waters by examining nutrient fluxes, their influence on phytoplankton communities, and factors driving potentially harmful microalgae development, providing a foundation for management

strategies to mitigate nutrient pollution, eutrophication, and HAB occurrences. The specific objectives of this study were:

Objective 1:

- i. To assess the spatio-temporal distribution of anthropogenic nutrients and phytoplankton assemblages in Kenya's coastal waters;

Objective2:

- ii. To investigate the distribution, sources and pathways of anthropogenic nutrients in Kenya's coastal waters, focusing on Mombasa.

Objective3:

- iii. To determine the composition and distribution of potentially harmful microalgae species and bloom formation risks in Kenya's coastal waters.

Objective4:

- iv. To determine the relationship between the composition of potentially harmful microalgae species with environmental factors.

This study addressed four major research questions including:

1. How do anthropogenic nutrients and phytoplankton assemblages vary spatially and temporally in Kenya's coastal waters?
2. What are the distribution patterns, sources, and pathways of anthropogenic nutrients in Kenya's coastal waters, with a specific focus on the Mombasa region?
3. What is the composition and distribution of potentially harmful microalgae species in Kenya's coastal waters, and what are the associated risks of bloom formation?
4. How do environmental factors influence the composition of potentially harmful microalgae, especially the toxin-producing species in Kenya's coastal waters?

1.5.Research Significance/ Justification

Kenya's coastal and marine waters are integral to the Somali Current Upwelling system, featuring diverse habitats such as coral reefs, mangroves, seagrass beds, and estuaries. These ecosystems support a variety of marine species, including globally endangered sea turtles and corals (Jacobs *et al.*,

2020; Kirkman *et al.*, 2020; Olendo *et al.*, 2019). Studying nutrient pollution, eutrophication, and HABs in this region is essential due to its recognition as one of six global marine biodiversity hotspots (ASCLME, 2012; Ryabinin, 2020). The coastal area serves as a critical habitat for migratory species, providing nesting sites for sea turtles and other marine biodiversity in the Western Indian Ocean (WIO) region.

Mangroves and seagrasses also function as vital carbon sinks, contributing to carbon sequestration and climate change mitigation (Githaiga *et al.*, 2017). Coastal ecosystems are crucial for local economies, significantly supporting livelihoods through fisheries and tourism. Approximately 4.3 million Kenyans depend on these resources, with 80% of household income derived from coastal fisheries and tourism (County Government of Mombasa, 2023). As Kenya aims to harness its blue economy potential—encompassing fisheries, aquaculture, maritime activities, and tourism—under its Vision 2030 development agenda (Government of Kenya, 2023; Muigua, 2018; Nyonje, 2018), the sustainability of these ecological functions relies on healthy coastal ecosystems.

However, coastal urbanization and industrialization increasingly compromise ecosystem health through escalating human-induced environmental pressures (Kirkman *et al.*, 2020; UNEP, 2017). Approximately 34% of global coastlines face high degradation risks due to development pressures, with around 245,000 square kilometers of marine ecosystems affected by nutrient pollution (GEF-8, 2023; Ocean & Coasts, 2024). Rapid population growth in East Africa has led to extensive deforestation and uncontrolled agricultural expansion without adequate sewage and wastewater management systems (Hamisi & Mamboya, 2014; Kithiia & Majambo, 2020; Scheren *et al.*, 2016). These activities contribute to eutrophication and the proliferation of HABs in coastal waters (Malone & Newton, 2020).

The synergistic effects of human-induced eutrophication and rising temperatures can exacerbate the development and toxicity of HABs (Hallegraeff *et al.*, 2021; Glibert and Burkholder, 2011). Current research has linked land-based nutrient inputs to declining water quality (Okuku *et al.*, 2011), increased potential for HAB development (Kiteresi *et al.*, 2012), changes in community dynamics, and degradation of critical marine ecosystems in Kenya's coastal waters (Mohamed *et al.*, 2009; Mwaura *et al.*, 2017; Okello *et al.*, 2022). Without proper nutrient management plans, these impacts are likely to worsen with ongoing population growth and planned expansions in agriculture, industry, aquaculture, and maritime transport (USAID, 2023). The global rise in harmful algal blooms suggests potential threats to Kenya's coastal waters, particularly given the presence of species known to cause these

blooms. Ciguatera fish poisoning risks contaminating seafood products, potentially leading to international market bans (Hansen et al., 2001; Kibler et al., 2015; Kiteresi et al., 2012; Ochieng et al., 2015; Ogongo et al., 2015). Climate change is expected to further exacerbate nutrient dynamics and HAB issues through altered precipitation patterns and rising temperatures (Kibler et al., 2015; Kithia & Majambo, 2020; Malone & Newton, 2020). Additionally, the ongoing coastal infrastructure development may enhance HAB development by altering coastal geomorphology and hydrological conditions (GEOHAB, 2016).

This thesis provides a more comprehensive analysis, significantly advancing the understanding of nutrient dynamics, phytoplankton, and potentially harmful microalgae in Kenya. The findings can be used to model future trends in areas with similar environmental exposures but limited data. By identifying anthropogenic nutrient sources and pathways, this research informs improvements in land management practices and guides sustainable development strategies in coastal zones. The scientific evidence produced regarding HAB toxins is crucial for influencing policy debates and raising public awareness about environmental issues related to food security and public health in the area. Furthermore, this research aligns with global Sustainable Development Goals (SDGs) and the Kunming-Montreal Global Biodiversity Framework by addressing SDG 6 (Clean Water and Sanitation), SDG 11 (Sustainable Cities and Communities), and SDG 14 (Life Below Water) through insights into anthropogenic nutrient sources and their impact on coastal water quality (Landrigan et al., 2020; United Nations, 2016). By contributing to the understanding of ecological integrity in Kenya's coastal waters, this study supports the preservation of biodiversity while enhancing community livelihoods through sustainable resource management.

1.6.Scope and Limitations

This study on nutrient pollution and ecological implications in Kenya's coastal waters encountered several limitations that shaped its focus. Resource constraints allowed for a concentrated examination of nitrogen and phosphorus pollution and their effects on key phytoplankton assemblages, particularly harmful microalgae. While limited sampling materials and equipment restricted the collection of macro algae and phytotoxins, this allowed for a concentrated analysis of microalgae species $\geq 20 \mu\text{m}$, which are critical to understanding phytoplankton assemblages and potentially harmful microalgae dynamics. The in situ measurements in this study spanned a 264 km stretch from Vanga to Marereni,

covering 44% of Kenya's coastline, representing the most comprehensive spatial and temporal analysis of the research topic in the region, providing an unprecedented continuous assessment. However, certain areas prone to HABs, such as the North Kenya Banks, were not assessed due to resource constraints. Monthly surveys focused primarily on Mombasa, offering valuable insights into coastal ecosystem dynamics. The study's targeted emphasis on harmful bloom-associated species enabled an in-depth analysis of these ecologically significant microalgae.

Methodologically, the research assumed bottom-up control of phytoplankton assemblages without considering top-down influences such as predation and competition. The one-year duration of the study may not have been sufficient to capture long-term trends and seasonal variations in nutrient fluxes and ecological processes affecting potentially harmful microalgae species. The one-year study may not suffice to understand fully the nutrient fluxes and ecological processes. However, it offered a valuable snapshot of nutrient dynamics and phytoplankton, including potentially harmful species, in coastal environments. This foundation sets the stage for future long-term research, offering crucial insights into seasonal patterns and ecological processes within these complex ecosystems. The study encountered some challenges related to obtaining permits under the NAGOYA protocols, which limited the scope of phytotoxic analysis to water samples. However, this focus allowed for a thorough examination of critical water quality indicators relevant to HABs effects. Additionally, the COVID-19 pandemic posed challenges that necessitated innovative adaptations, such as leveraging online resources for literature reviews and adjusting fieldwork plans to maximize data collection within accessible areas.

1.7. Thesis Structural Outline

This thesis is structured into six chapters, four of which comprise either of individual papers that have been published or in preparation for submission.

Chapter One introduces the research context, outlining the aims, objectives, questions, significance, scope, and limitations of the study.

Chapter Two reviews the status of anthropogenic nutrients and phytoplankton assemblages, including harmful algal blooms (HABs), in Kenya's coastal and marine waters.

- **Status:** Published in *Oduor et al. (2023)*(<https://doi.org/10.1016/j.ocecoaman.2022.106454>).

Chapter Three analyzes the spatial distribution of nutrients and phytoplankton assemblages in Kenya's coastal waters. It examines their associations and implications for coastal ecosystem health.

- **Status:** Published in Oduor *et al.* (2024) <https://doi.org/10.1016/j.marpolbul.2023.115897>).

Chapter Four focuses on the composition and distribution of potentially harmful microalgae. It analyzes their distribution, potential effects, and examines the driving factors behind the development of toxin-producing species in the study area. This chapter also explores the association between anthropogenic nutrient fluxes and occurrences of HAB-causing species.

- **Status:** The chapter's revised article is under Preparation for submission in the *Journal of Ocean & Coastal Management*

Chapter Five assesses nutrient fluxes and associated drivers in Kenya's coastal waters, with a specific focus on Mombasa coastal city.

- **Status:** The Chapter Manuscript in Preparation for *Marine Pollution Bulletin*

The final chapter, **Chapter 6** is a general discussion that synthesizes the findings from previous chapters, addressing how research questions were answered. It discusses long-term trends and seasonal patterns in phytoplankton community structure and harmful algal blooms (HABs) in Kenya's coastal waters. The chapter compares HABs dynamics to other coastal ecosystems globally, assessing environmental and climate change effects. It concludes by emphasizing the importance of high-resolution monitoring, offering policy recommendations, and identifying knowledge gaps for future research.

Chapter Two: Literature Review



Modified from,

Oduor, N. A., Munga, C. N., Ong'anda, H. O., Botwe, P. K., & Moosdorf, N. (2023). Nutrients and harmful algal blooms in Kenya's coastal and marine waters: A review. *Ocean & Coastal Management*, 233(December 2022), 106454. <https://doi.org/10.1016/j.ocecoaman.2022.106454>

Nutrients and Harmful Algal Blooms in Kenya's Coastal and Marine Waters: A Review

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2.1.Chapter Summary

Nutrients and Phytoplankton composition are important indicators of coastal water quality used for fisheries, tourism, and recreational purposes, which are the cornerstone of the blue economy. This is a comprehensive review and synthesis of the outcome of previous empirical studies conducted on anthropogenic nutrients (nitrogen and phosphorus) and phytoplankton productivity and community composition with a particular focus on harmful algal blooms (HABs) in Kenya's coastal and marine waters. Our results indicate that localized nutrient pollution occurs in estuaries and creeks adjacent to urban areas. Nitrogen speciation was characterized by high ammonium relative to nitrate concentrations with evidence of nitrogen limitation ($N:P < 16$). Based on all studies' average chlorophyll a concentration, seawater in coral and seagrass ecosystems was classified as mesotrophic, mangroves and nearshore as higher mesotrophic, and estuarine and creeks as eutrophic. Additionally, 44 species of potential toxin-producing HAB-forming species belonging to 22 genera were established in low abundances (3 ± 2 - 210 ± 11 cells/l) below the standards of HAB's warning thresholds. Their drivers, spatial and temporal trends are, however, unclear. This review revealed the existence of localized nutrient pollution and a low abundance of toxin-producing HABs species in Kenya's coastal waters.

The existing data is, however, very fragmented and inadequate to guide coastal managers in addressing the problem of anthropogenic nutrient pollution and HABs in the area. We, therefore, recommend the inclusion of anthropogenic nutrients and toxic HABs in routine water quality monitoring in the area, especially in potential fisheries, aquaculture, tourism, and recreational sites.

2.2.Introduction

Coastal fisheries and tourism support the livelihoods of about 4.3 million people in Kenya ([County Government of Mombasa, 2019](#)) with the former supplying 95% of the country's total marine catch, generating approximately US\$ 3.2 million per year ([ASCLME, 2012](#)). Fisheries and tourism depend on healthy coastal ecosystems to thrive. However, the health of coastal and marine ecosystems are globally threatened by the enrichment of nutrients (nitrogen-N and phosphorus-P) originating from anthropogenic sources like agricultural run-offs, untreated sewage, and industrial discharges through the surface and groundwater ([Long *et al.*, 2014](#); [Malone & Newton, 2020](#)). Additionally, benthic fluxes from pore water are also a major source of nitrate, ammonium, phosphate, and silicate nutrients in bottom waters and shallow coastal environments ([Mathis *et al.*, 2014](#)).

Nitrogen (N) and phosphorus (P) are essential for primary production with P limitation occurring in most freshwaters, N limitation usually in marine and estuarine waters, and a co-limitation of both N and P reported in tropical coastal and marine waters ([Anderson *et al.*, 2002](#); [Glibert *et al.*, 2011](#)). Diatoms require silicon (Si) as a major nutrient in addition to N and P to form their silicified cell walls. The availability, forms, and stoichiometry of these nutrients, therefore, govern the growth and composition of phytoplankton species that determine fisheries productivity in coastal waters ([Heil *et al.*, 2007](#)).

Increased use of inorganic fertilizers, fossil fuels, and generation of sewage effluents has caused the elevation of P and N relative to Si to occur in coastal waters ([Howarth *et al.*, 2002](#); [Malone and Newton, 2020](#)). The resulting imbalance in P, N, and Si can transform the phytoplankton community composition favoring the development of harmful algal blooms (HABs), which include dinoflagellates, cyanobacteria, toxin-producing pelagic diatoms, and flagellates ([Davidson *et al.*, 2012, 2014](#); [Dhar *et al.*, 2015](#); [Mead *et al.*, 2013](#)). The resulting transformation in phytoplankton composition can affect the entire food web with serious consequences to marine life, human health, and local livelihoods ([Berdalet *et al.*, 2016](#)).

There is currently a global increase in frequency, amplitude, and toxicity of HABs in coastal waters, which has been associated with the increasing temperatures and anthropogenic supply of N and P (Berdalet *et al.*, 2016; Glibert *et al.*, 2005; Glibert, 2020). Tropical developing countries are particularly vulnerable to the occurrences and impacts of HABs due to high reliance on artisanal fisheries, increasing use of artificial fertilizers in river catchment areas (Malone & Newton, 2020), and poor management of sewage discharges in coastal areas (e.g., Kithiia and Majambo, 2020). Moreover, in these areas, anthropogenic nutrient pollution and the dynamics of HABs in coastal waters are understudied and are thus poorly understood.

Like many other developing coastal countries, Kenya is currently diversifying its economy by exploiting its blue economy potential to achieve Sustainable Development Goals (SDGs). It also has set economic development goals, namely ‘The BIG 4 Agenda and Vision 2030’ (cn.invest.go.ke, 2017). The country plans to expand agriculture, aquaculture, and maritime transport to deliver food security and employment opportunities, with the latter two being the cornerstone of the blue economy. Thus, understanding the coastal and marine ecosystem’s dynamics, including nutrient fluxes and consequent occurrence of HABs, is important to realize the full potential of the blue economy. Furthermore, the planned expansions may increase the supply of nutrients N and P from agricultural fertilizers, sewage, and ballast water if no early management plans are put in place.

Efficient management of anthropogenic nutrient pollution and its associated effects, like HABs, requires sufficient scientific information for policy formulation. This review, therefore, examines the status and trends of anthropogenic nutrient enrichment and HABs as presented in the previous empirical studies. It was conducted to identify *the major drivers, sources, pathways, and forms of anthropogenic nutrient (N and P) pollution, highlight links between nutrient conditions and phytoplankton community composition and primary production, and lastly establish the status of harmful algal bloom species in Kenya’s coastal and marine waters.* It thus provides insights into the available information and existing gaps that need special consideration in future research to understand the drivers of nutrient pollution and HABs development, as well as the coastal ecosystem’s responses for management purposes to ensure full utilization of the blue economy potential.

2.3. Methodology

2.3.1. Study area

Kenya is located on the east coast of Africa with a coastline length of about 640 km extending from 1°30'S to 4°30'S. Kenya's coast is characterized by many creeks, estuaries, fringing coral reefs, seagrass beds, and mangrove swamps (Figure 1). It experiences semi-diurnal tides with amplitudes between 1 and 3 m (Kitheka *et al.*, 1996), and a mean sea surface temperature (SST) of 28.2°C (ASCLME, 2012). The annual rainfall ranges between 1000 and 1200 mm and experiences a modified equatorial climate with two rainy seasons (October to December and May to July) (ASCLME, 2012; Obura, 2001). The region has two major river systems, Tana and Athi-Sabaki, which drain into the north coast, providing greater sediment and nutrient inputs than the smaller, semi-permanent rivers (Ramisi, Umba, Mwache, Mkurumuji, Rare, and Kombeni) on the south coast.

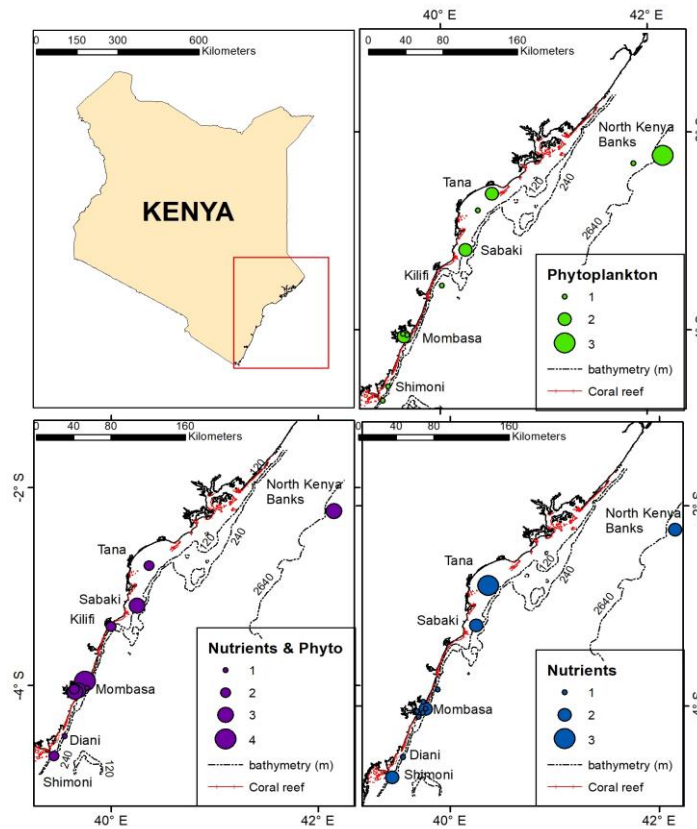


Figure 1. A map of the study area showing the location of studies on phytoplankton (top right), nutrients (bottom left), and a combination of the two (bottom left) in Kenya's coastal and marine environment

2.3.2. Literature search and publications screening

Systematic literature searches were conducted between June 2021 and March 2022 with search terms targeting studies on anthropogenic nutrients (nitrogen and phosphorus) sources, pathways, conditions (concentrations and ratios), and phytoplankton composition, including the occurrence of HABs in Kenyan waters (Figure 2). We searched Web of Science, Science Direct, and Google Scholar for topics with keywords including nutrients, phytoplankton, primary production, HABs, coasts, or marine and Kenya. Inclusion criteria were journal articles, book chapters, and technical reports with empirical data, published in 2021 or earlier, and a study focus on Kenya's coastal and marine waters (Search string in Appendix 1). Exclusion criteria were editorial and meeting proceeding articles, conference abstracts, and lack of information on nutrient pollution, phytoplankton, and HABs (Figure 2). The articles were then manually screened for inclusion based on the language of publication (only those in English were considered) and the availability of information on nutrients, phytoplankton, and HABs. The initial list of literature was established, and then yielded further sources through snowball and citation searches. We decided to include seven articles from grey literature sources, such as technical reports from government bodies or non-governmental organizations (NGOs), and theses, because of limited available data in the peer-reviewed literature (Appendix 2). Data were extracted from original empirical research articles only to avoid duplication. MendeleyDesktop Version 1.19.4.0 was used for article organization and the removal of duplicates.

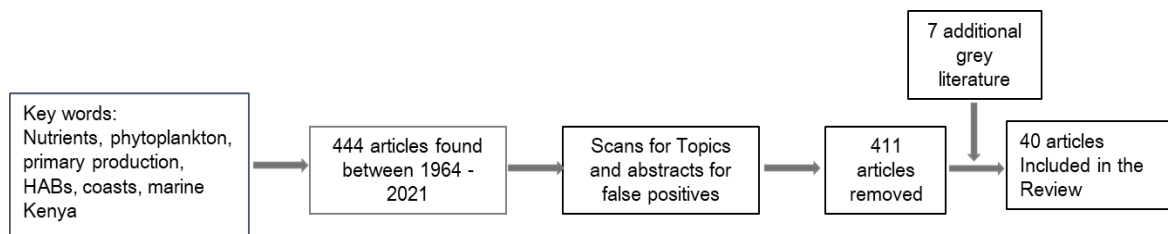


Figure 2: Systematic search and review process undertaken

2.3.3. Data collection

To establish anthropogenic nutrient sources, pathways, and drivers of pollution and HABs development, data were extracted for the type of water (e.g., seawater, pore-water, and groundwater) and the type of ecosystem from the selected articles. The ecosystems were classified based on hydrological features or ecological conditions. These categories were estuarine (areas lying within river mouths), creek (tidal channels with mostly saline water and low river influence), mangroves (mangrove-covered areas), seagrass (areas covered by seagrass), and coral reefs (open areas covered by coral reefs),

nearshore (areas close to land with no mangrove, seagrass or coral coverage), offshore (open waters). Data were extracted for nutrient concentrations and ratios (dissolved inorganic nitrogen –nitrates and ammonium, phosphates and silicates), primary production (phytoplankton biomass- chlorophyll a (*Chl-a*) and daily primary production), phytoplankton composition, abundance, and diversity where they existed. The sampling year, month, and season were also recorded. Studies from similar environments in other areas were used to aid the interpretation of the established results.

2.3.4. Data analysis

To examine anthropogenic nutrient conditions (nitrate - NO_3^- and ammonium- NH_4^+ and phosphate- PO_4^- , and N: P ratio) and phytoplankton biomass (*Chl-a*) and composition (HABs) in Kenyan coastal waters, descriptive statistics and an F-test Two Sample Variance was conducted, followed by a t-test at $\alpha = 0.05$ significance level. All nutrient concentration data were converted into $\mu\text{mol/L}$ and *Chl-a* to $\mu\text{g/L}$. Threshold levels for nutrients and *Chl-a* were defined based on standard water quality variables criteria (Simboura *et al.*, 2005). All analyses and data presentations were conducted using MS Excel 2016, Sigma Plot 14.0. and PRIMER-E (v7.).

2.4.Results

The results from a total of 40 articles with empirical studies indicated that the mean nitrate and phosphate concentrations across all studies ranged between 2.07, 1,254 $\mu\text{mol/L}$, 1.22, and 19.26 $\mu\text{mol/L}$ in Kenya's coastal and marine waters. The mean *Chl-a* and daily carbon primary production were $2.17 \pm 0.31 \mu\text{g/l}$ and $185.66 \pm 28.10 \text{ mgC/L}^{-\text{d}}$, respectively, and 56 species of potentially harmful algal blooms were recorded during the study.

2.4.1. Articles composition and distribution

A total of 40 articles published between 1983 and 2021 were found relevant, containing empirical data used in this review. About 78% were on nutrients, 18% on HABs, and 28% on phytoplankton composition while only 45% covered both nutrients and phytoplankton community composition including HABs. Most studies were conducted on the creeks ($n = 24$) and estuaries ($n = 16$) with the creeks around Mombasa (Tudor, Kilindini, and Mtwapa) and Sabaki as the most studied creeks ($n = 5$) and estuary ($n = 7$), respectively. Most of the studies were published after 1990, with an average of twelve studies per decade.

2.4.2. Sources and pathways of nutrients in Kenya's coastal and marine waters

Average nitrate and silicate concentrations were substantially higher in groundwater compared to pore and seawater. The highest mean phosphate concentration in the study was registered in the pore water while the highest N: P ratio was in groundwater (Table 1). Elevated nitrate levels were observed in both ground and pore water in urbanized nearshore waters (Nyali in Mombasa). The mean and ranges for dissolved inorganic nitrogen (NO_3^- and NH_4^+), PO_4^- and dissolved silicate- SiO_3^{2-} , as well as N: P and N: Si in ground, sea, and pore water established from this review varied across groundwater, pore water and seawater (Table 1).

Table 1: The Mean (\pm S.D.), and ranges of nutrient concentrations (NO_3^- , NH_4^+ , PO_4^- , SiO_3^{2-}) and ratios (N: P and N: Si) in the ground, pore, and seawater in Kenya's coastal and marine environment

Nutrients Condi- tions	Groundwater (n= 6) ^A		Pore water (n = 3)		Sea water (n = 38)	
	Mean \pm S.D.	Range	Mean \pm S.D.	Range	Mean \pm S.D.	Range
NO_3^- ($\mu\text{mol/L}$)	1255 \pm 502.7	59.4 - 4900	71.8 \pm 23.0	1.50 – 495.5	2.07 \pm 0.42	<DL – 82.8
NH_4^+ ($\mu\text{mol/L}$)	78.7 \pm 40.0	0.40 - 907	0.59 \pm 0.10	0.20 - 1.00	2.25 \pm 0.21	0.01 – 23.1
PO_4^- ($\mu\text{mol/L}$)	4.73 \pm 2.00	0.38 -71.3	19.3 \pm 8.7	0.10 – 164.7	1.22 \pm 0.24	<DL – 49.2
SiO_3^{2-} ($\mu\text{mol/L}$)	250 \pm 48.0	86.0 – 597	56.35 \pm 9.5	1.20 -134	33.5 \pm 6.9	4.00 – 163.5
N: P	592.5 \pm 176.8	18.7 – 6997	165.9 \pm 88.8	0.39 -1782	11.8 \pm 4.2	0.02 – 849.5
N: Si	26.39 \pm 14.8	0.16 – 26.3	0.50 \pm 0.09	0.10 - 1.25	0.17 \pm 0.03	0.01 – 0.56

^A Data presented for groundwater were extracted from studies conducted in Mombasa, Diani, and Mida Creek and surface pore water in reefs (Mwaura *et al.*, 2017), estuarine (Ohowa, 1996), and mangroves (Mwashote *et al.*, 2005). The ground and pore water silicates data were only available for Mida Creek and Sabaki estuary while the ammonium data for pore water was only for nearshore sites around Mombasa (Vipingo, Bamburi, and Nyali).

In the groundwater, the highest concentration of NO_3^- , NH_4^+ and PO_4^- was established in the creek waters, while the nearshore urban waters in Nyali were very rich in nitrate but deficient in phosphate. The groundwater nutrient concentrations were also characterized by elevated nitrogen nutrients, especially the nitrate form, relative to phosphate. The review established a strong, insignificant positive correlation ($r = 0.733$, $p = 0.1501$) between nitrate and phosphate concentrations in groundwater in all the sites (Figure 3).

In the pore water, the highest mean nitrate and ammonium concentrations were established in the nearshore reef waters, the site where the lowest phosphate concentrations were established, while the highest phosphate was recorded in the mangrove creek waters. The ammonium levels were $< 0.1 \mu\text{mol/L}$ for the nearshore urban waters in Mombasa, while the silicates ranged between 0.1 and $1.25 \mu\text{mol/L}$ in the pore water (Figure 4).

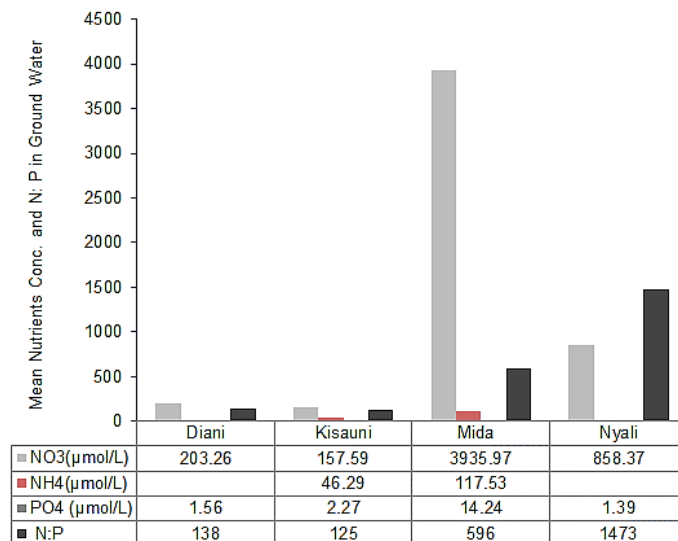


Figure 3. Mean Nitrate-NO₃⁻, Ammonium-NH₄⁺, and Phosphate-PO₄⁻ concentrations and N: P ratios in groundwater across all studies along the coast of Kenya

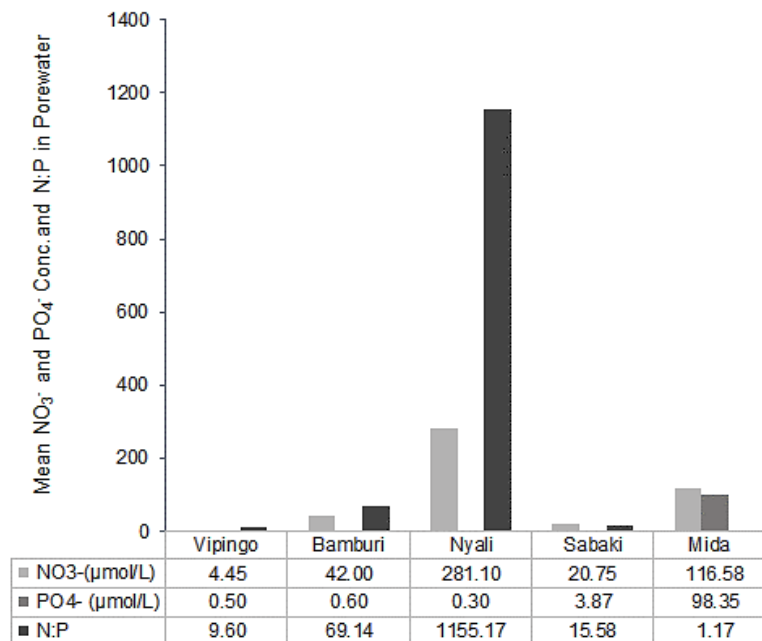


Figure 4. Mean Nitrate-NO₃⁻, Ammonium-NH₄⁺, and Phosphate-PO₄⁻ concentrations and N: P ratios in Pore water across all studies along the coast of Kenya

Nutrient concentrations in seawater showed significant variation across different locations ($p < 0.05$). The review's highest mean nitrate and phosphates seawater concentrations were established in estuarine waters. Ammonium concentrations exceeded nitrate in all locations apart from estuarine waters. The highest value was recorded in the creeks (Figure 5 and Table 2)

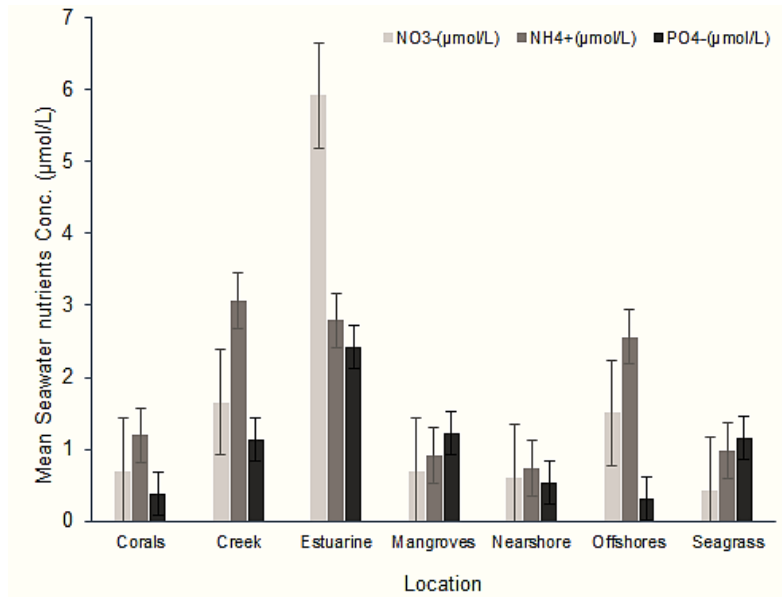


Figure 5. Mean Nitrate-NO₃⁻, Ammonium-NH₄⁺, and Phosphate-PO₄⁻ concentrations in seawater in various locations (study site type) across all studies along the coast of Kenya

Table 2. Minimum and Maximum values of nutrient concentrations (NO₃⁻, NH₄⁺, PO₄⁻ and SiO₃²⁻) and ratios (N: P and N: Si), *Chl-a*, and daily primary production (PP) registered in different ecosystems during the study.

Parameter	Min/Max	Offshore	Corals	Seagrass	Mangroves	Nearshore	Estuarine	Creeks
NO ₃ (μmol/L)	Min	0.10	0.10	0.04	0.69	<DL	0.01	0.002
	Max	5.30	2.50	1.20	0.93	3.09	82.8	18.0
NH ₄ (μmol/L)	Min	0.98	0.07	0.25	0.92	0.01	0.16	0.27
	Max	3.80	3.42	1.80	1.37	2.31	11.1	23.1
PO ₄ (μmol/L)	Min	0.12	0.08	0.38	1.23	<DL	<DL	0.05
	Max	0.56	0.80	2.32	2.47	9.63	49.2	13.8
SiO ₃ ²⁻ (μmol/L)	Min	-	-	-	-	-	5.00	4.00
	Max	-	-	-	-	-	100	164
N: P	Min	0.54	1.61	1.18	0.81	0.02	0.23	0.08
	Max	51.7	47.5	2.18	1.88	111	76.3	30.0
N: Si	Min	-	-	-	-	-	0.03	0.01
	Max	-	-	-	-	-	0.56	0.32
<i>Chl-a</i> (μg/l)	Min	0.20	0.02	0.04	0.75	0.03	0.17	0.06
	Max	1.20	0.90	1.80	3.00	4.71	40.9	21.5
PP(mgC/L-d)	Min	-	100	190	100	-	-	100
	Max	-	356	1050	435	-	-	1600

* Abbreviations: “Min” means minimum, “Max” means maximum

A spatial analysis of nutrient conditions in estuarine waters established the highest mean nitrate concentrations in one of the longest rivers (River Sabaki) located on the north coast of Kenya. The lowest phosphate concentration was established in River Tana, which is also another major river draining through a large catchment area. Almost similar mean nutrient concentrations were established in rivers located on the south coast (Mwena, Ramisi, and Uмба) (Figure 6).

In the creeks, relatively higher ammonium levels were recorded compared to nitrate in all the creeks, apart from Mida and Malindi (Figure 6). The maximum ammonium concentration (23.07 $\mu\text{mol/L}$) encountered in the whole study was reported in the urban site of Mtwapa, while the creeks adjacent to informal settlements in Mombasa (Makupa and Tudor) recorded the highest mean NH_4^+ concentrations (3.85 $\mu\text{mol/L}$) recorded in this study. Both ammonium and phosphates were higher than nitrates in creeks exposed to maritime activities (Kilindini) and informal settlements (Makupa) (Figure 6).

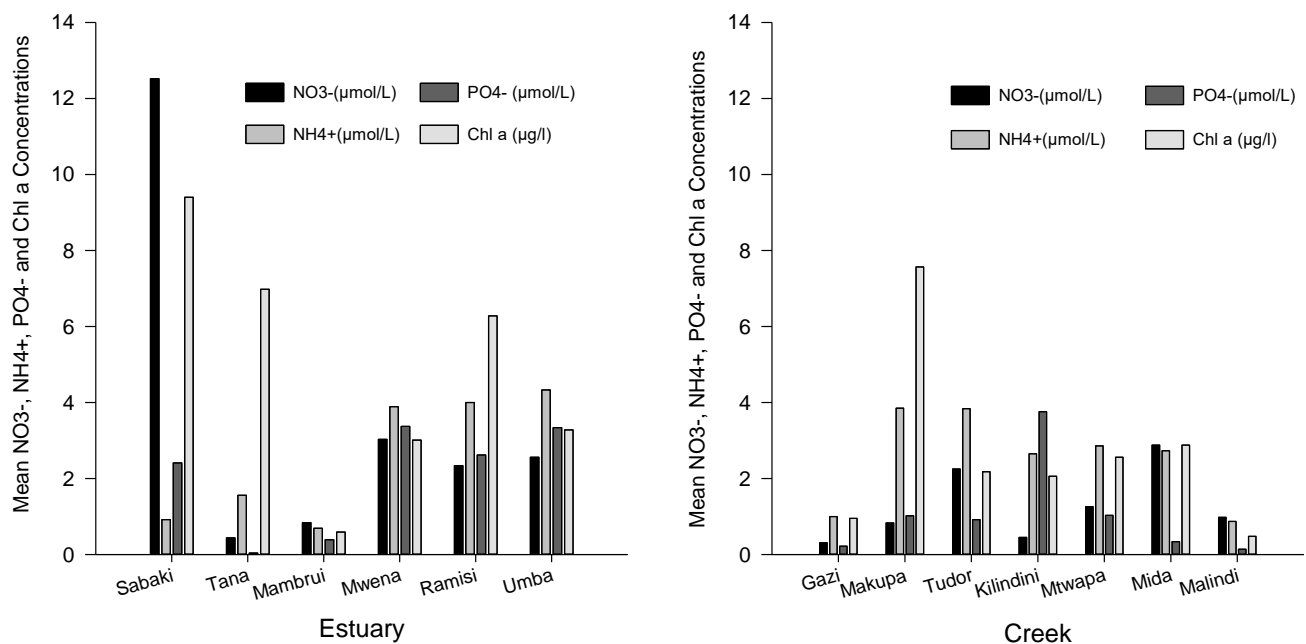


Figure 6. Mean NO_3^- , NH_4^+ and PO_4^- Concentrations averaged across all studies in estuarine (left) and creek (right) waters along the coast of Kenya

A significant seasonal variation was established between dry and wet seasons' nutrient concentrations in all the locations ($p < 0.001$). Higher NO_3^- and NH_4^+ were established during the wet season compared to the dry season in all the sites (Figure 7).

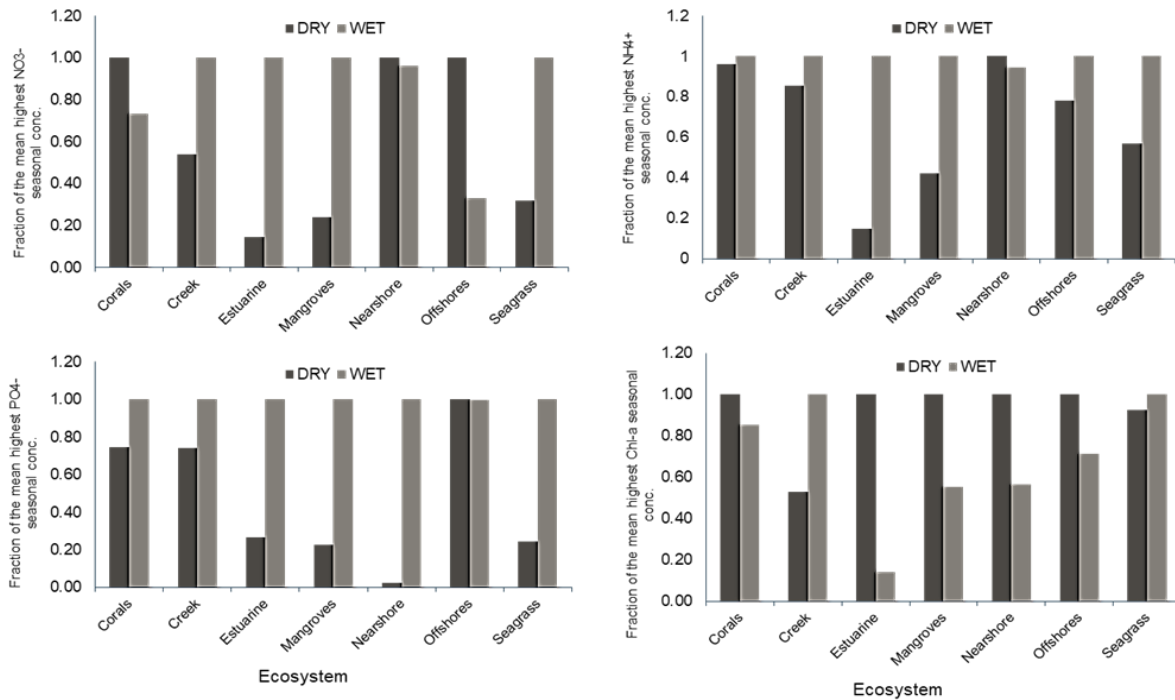


Figure 7. Mean seasonal variation of NO₃⁻ (top left), NH₄⁺ (top right), PO₄²⁻ (bottom left), and *Chl-a* (bottom right) concentrations across all studies in different ecosystems along the coast of Kenya. *Values are shown as a fraction of the average concentration during the season with the highest concentration.

The average seawater N: P ratios were below the Redfield ratio (N: P = 16) in almost all the sites apart from nearshore waters with the lowest values (<2) recorded in mangrove and seagrass waters (Figure 8).

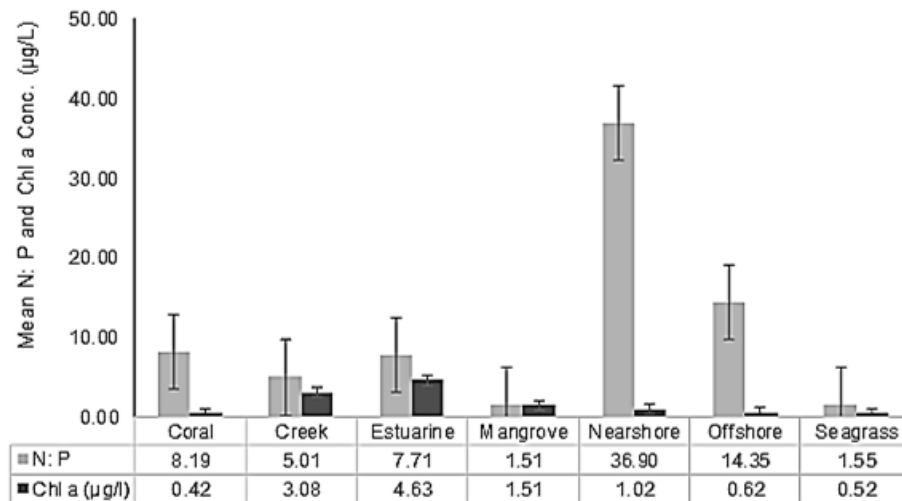


Figure 8. Mean N: P ratios and *Chl-a* in different ecosystems in Kenya's coastal and marine waters

Based on specific locations (water types), the highest N: P ratio, which was above the Redfield ratios ($N: P > 16$) was observed in creek waters in Mida, while the lowest values were in populated urban areas in Tudor and Mtwapa creeks. In the estuarine waters, only Tana had N: P values above the Redfield ratio while the rest had low values < 10 with values < 2.0 recorded in Mambrui and Umba. Greater N: P values were also established during the wet season compared to the dry season in near-shore and offshore waters while a contrasting pattern was observed in coral and estuarine areas for the same (Figure 9).

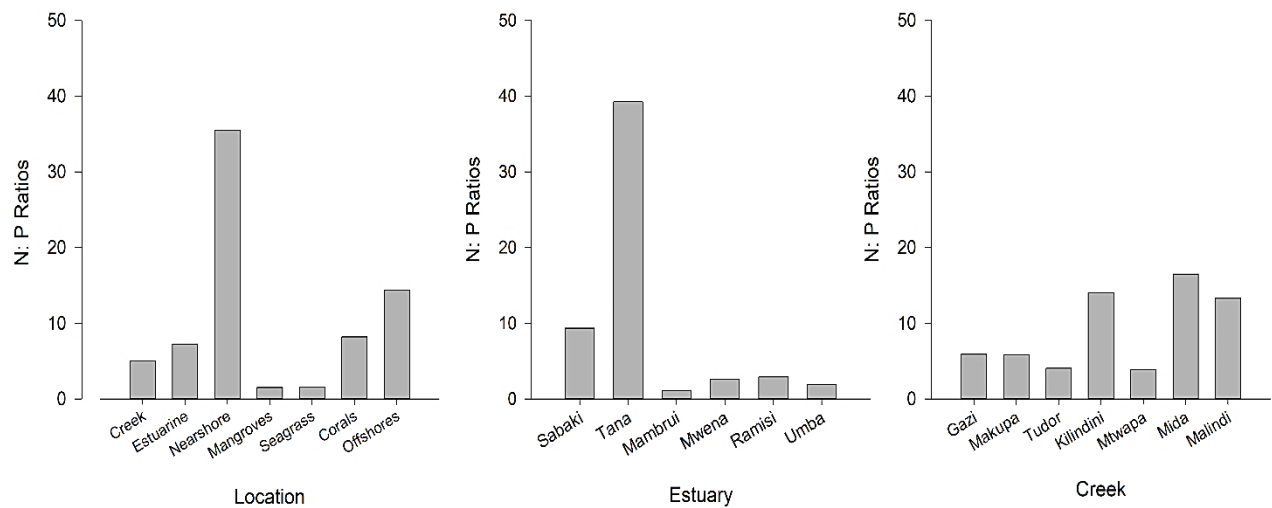


Figure 9: Mean molar N: P Ratios in different locations (left), specific estuaries (centre), and creeks (right) in the study area

2.4.3. Nutrient conditions, primary production, and Phytoplankton community composition

Primary production was analyzed in seawater using *Chl-a* concentrations and daily carbon primary production (Table 2). Very few studies existed on daily primary production ($n = 3$), with the highest value of 1600 and 1050 mgC/L-d established in the creeks and seagrass during the dry (October) and towards the end of the wet season in July, respectively. None of these studies assessed the PP in estuarine environments. On the other hand, *Chl-a* was the most used proxy for examining primary production in the area. The highest *Chl a* concentration of 40.9µg/l, encountered throughout the study, was established in estuarine waters during the dry month of September. The highest and lowest mean *Chl-a* concentrations averaged across all studies were established in estuarine and coral waters, respectively (Figure 8). In the corals, the highest *Chl-a* was also in October, during the dry season, while the lowest value was during the wet season. In the creeks and nearshore waters, a mixed ob-

servation was made in different sites with both high and low *Chl-a* concentrations during both seasons, making it difficult to establish the seasonal influence of primary production in the areas. Productivity in mangroves indicated a higher mean *Chl-a* concentration during the dry season compared to the wet season (0.8 and 1.45µg/l). The study of primary production was, however, limited to those in mangroves only presented in one study, while seagrass and corals were available in three studies. Based on specific sites, the lowest and highest estuarine *Chl-a* concentrations were in Mambrui and Sabaki, respectively. In the creeks, the highest *Chl-a* concentration was in Makupa, which also corresponded to the high NH_4^+ recorded at the site.

A strong positive correlation was established between *Chl-a* and nutrient concentration in seawater among all five locations that provided both data ($r(5) = 0.99$, $p < 0.05$). The correlation was significant with phosphate ($p < 0.001$) and insignificant for nitrate and ammonium concentrations ($p > 0.05$). A significant seasonal variation was established between dry and wet seasons ($p < 0.05$), with higher mean *Chl-a* concentration recorded in estuarine and creek waters during dry and wet seasons respectively (Figure 7). During the dry season, the *Chl-a* concentration was significantly correlated with both ammonium and phosphates in all the stations. This study classified coral and seagrass waters as mesotrophic, mangroves and nearshore stations are higher mesotrophic while estuarine and creek waters are eutrophic based on mean *Chl-a* concentrations following the classification criteria by Simbaura et al., (2005). The trophic classification of different studies in Kenya's waters is also presented in this study (Table 3).

Table 3: Trophic status classification based on individual studies

Site	Classification	Criteria	Source
Kenya's coastal waters	Oligotrophic	<i>Chl-a</i> levels (predominance of re-generated production)	Mengesha et al., 1999
Tudor Creek	Oligotrophic	Nitrate concentrations	Kazungu et al., 1989;
	Higher mesotrophic	<i>Chl-a</i> levels	Okuku et al., 2011
Makupa Creek	Eutrophic	<i>Chl-a</i> levels	Okuku et al., 2011
Mtwapa Creeks	Higher mesotrophic	<i>Chl-a</i> levels	Okuku et al., 2011
	Eutrophic	Bacterial contamination	Mwangi, et al., 2001
	Oligotrophic	Nitrate level	Pole et al., 2016
Gazi Creek	Oligotrophic	<i>Chl-a</i> levels	Okuku et al., 2011
Mida Creek	Mesotrophic	Phytoplankton diversity	Mwashote, Ohowa and Wawiye, 2005

The studies on phytoplankton established and presented in this review were for micro-phytoplankton size classes $< 20 \mu\text{m}$ sampled in the surface waters ($\leq 5\text{m}$). The phytoplankton species richness ranged between 40 and 295 species belonging to 78 genera (Mwashote et al., 2005; Adala et al.,

2007; Kiteresi *et al.*, 2012). The cell density in the area ranged between 1,614 cells l⁻¹ and 68,515 cells L-1 in the creek, estuarine, and nearshore waters (Kitheka *et al.*, 1996; Mwangi *et al.*, 2001; Kiteresi *et al.*, 2012).

Diatoms dominated the microphytoplankton community group in all reviewed studies. Dominance and localized blooms of the nitrogen-fixing cyanobacteria (*Oscillatoria* sp., *Chroococcus limneticus*, and *Trichodesmium* sp.) were reported offshore (Kromkamp *et al.*, 1997) and in coastal waters (Mwangi *et al.*, 2001; Mwashote *et al.*, 2005; Kiteresi *et al.*, 2012). Other dominant species are *Chaetoceros* sp., *Rhizosolenia* sp., *Bacteriastrum* sp., *Nitzschia delicatissima*, *Tabellaria fenestrata* (Lyngbye), *Pseudo-nitzschia* sp, *Coscinodiscus* sp., *Prorocentrum* sp., *Protoperidinium* sp., *Eutrep tiela* sp., *Thalassiosira* sp., *Peridinium* sp., *Actinoptychus* sp., *Tripas* sp., and *Gymnodinium* sp. (Kiteresi *et al.*, 2012) (Table 4).

Table 4: Phytoplankton composition in east African waters based on past studies

Taxonomic Group				Location/Ecosystem	Source
Diatoms (%)	Dinoflagellates (%)	Flagellates (%)	'Others'/ Cyanobacteria (%)		
57.6	41.1	1.3	-	Mombasa (Urban/tourism)	Adala <i>et al.</i> 2007
51.1	22.7	6.8	19.3	Ramisi-Vanga system (Estuarine and Oceanic)	Kiteresi <i>et al.</i> 2012
65.8	32.9	-	1.3	Pemba Channel (Oceanic/Fishing)	Sekadende <i>et al.</i> 2021
70.0	24.0	-	6.0	Zanzibar coast (Urban/tourism)	Moto <i>et al.</i> 2018

2.4.4. HABs in Kenya's coastal and marine waters

This review established 56 species of potentially harmful microalgae existing in Kenya's coastal and marine waters. The species were grouped using two major criteria, i.e. taxonomic groups (dinoflagellates, diatoms, flagellates, and others) and the impact factors (toxin producers, oxygen depleters, and stranglers) (Hansen *et al.*, 2001). About 56 % of the total number of established HABs species was made up of dinoflagellates while flagellates were the least in the group. Based on impact categorization, 67% composition was made up of toxin-producing species (Figure 10). At least 24 species have been reported in Shirazi, Kilindini, Kilifi, and Ngomeni reported (Hansen *et al.*, 2001); 9 species reported off the southern coast of Kenya (ASCLME, 2012); and 24 species identified in Kilindini Creek (Ogongo *et al.*, 2015). Additionally, 26 and 16 potentially harmful microalgae species have been registered in the Ramisi-Vanga system (Kiteresi *et al.*, 2012) and Shirazi creek (Ogongo and Okumu, 2015), respectively. About 67% of the total identified HABs species were

registered in Kilindini Creek. The highest abundance for the blooming species was 215,000 cells/L reported in Kilindini (Hansen *et al.*, 2001) while the toxin-producing species ranged between 3 ± 2 - 407 ± 87 cells/L (Ogongo *et al.*, 2015). In contrast, much higher values were reported for toxic *Pseudo-nitzschia pungens*: up to 60,000 cells/l in Shirazi Creek and 7,000 cells/l in Kilindini and Kilifi Creeks. A full list of all identified potentially harmful microalgae species in Kenya's coastal and marine waters including the nature of effect and site of identification established from this study are provided in Table 12.

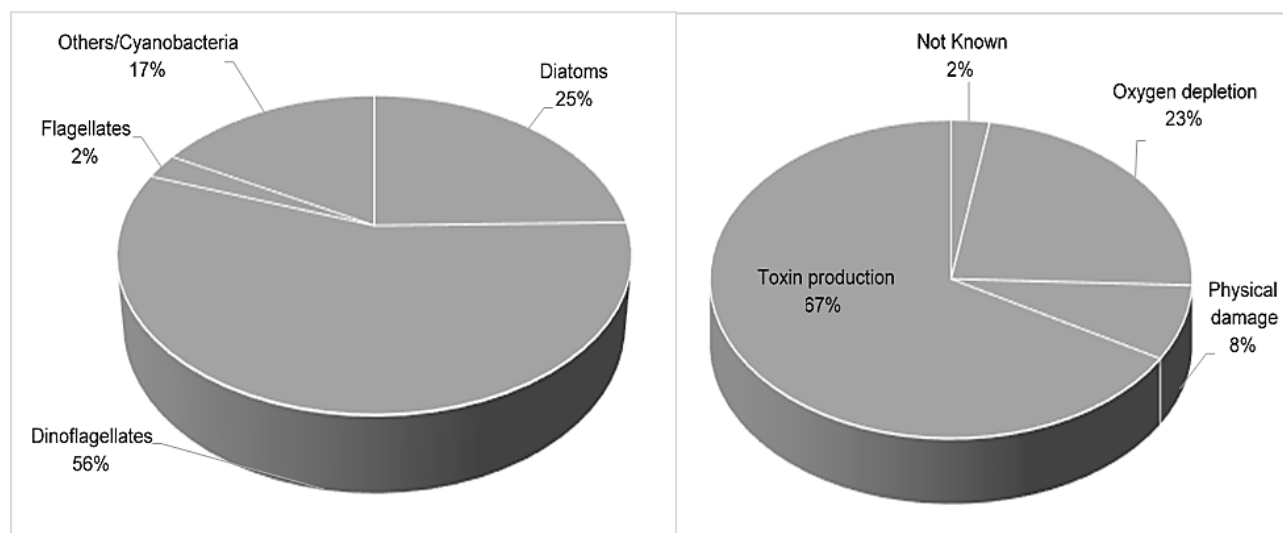


Figure 10. Taxonomic (left) and impact category composition of major groups forming Potential HABs in Kenya's Coastal and Marine waters (n = 56)

2.5. Discussion

Groundwater in the area is primarily used for drinking, while seawater supports fisheries and recreation. However, dissolved inorganic nutrients (NO_3^- , NH_4^+ , and PO_4^-) exceed international standards, with NH_4^+ levels surpassing 0.2 mg/L and NO_3^- levels exceeding 10 mg/L (U.S. EPA, 2000; WHO., 1996). These elevated nutrient levels pose concerns for both drinking water and recreational seawater quality (Mwaguni, 2002; Mwashote *et al.*, 2005; Barongo *et al.*, 2006; Abdulaziz, 2017). Anthropogenic nutrient inputs from agriculture and urbanization account for the major nutrient sources supplied into coastal waters worldwide (Howarth, 2008). About 80% of municipal wastewater is released into the environment untreated globally and human sewage was reported to be the most prevalent urban source of nutrients in coastal waters (Malone & Newton, 2020). High organic matter inputs supplied by untreated wastewater as established in the study, have been reported earlier and linked to high nutrients in pore water due to remineralization processes (Kaiser *et al.*, 2015). Elevated

nutrient concentrations in the creeks around urban areas and estuaries can thus be associated with the supply of untreated sewage, industrial discharges, storm water, and agricultural runoff. Nitrogen supply from nitrogen fixation ([Kromkamp *et al.*, 1997](#)), localized upwelling in nearshore waters ([Obura, 2001](#)), and groundwater ([Mwaura *et al.*, 2017](#)) around Mombasa and Kilifi areas were also reported and can be linked to the elevated nitrate concentration in nearshore waters.

A seasonal comparison indicated higher nutrient levels during the rainy season compared to the dry season. This suggests a greater abundance of nutrients being washed from diffuse land-based sources by surface run-offs during the rainy season. A recent review reported that about 20% of anthropogenic nitrogen used in coastal watersheds is exported into the coastal waters ([Malone & Newton, 2020](#)). Elsewhere, an earlier study by [Glibert and Burkholder \(2006\)](#) established a direct relationship between population development, fertilizer application, and riverine nitrogen and phosphorus fluxes. The elevated nutrient levels reported during the wet seasons in the estuarine waters can thus be a result of agricultural runoffs from the river catchment areas, characterized by synthetic fertilizers. Existing spatial variations established among estuaries and creeks can be a result of differences in the intensity of anthropogenic activities in the watersheds as has been reported in other studies ([Jickells *et al.*, 2017](#)). This is evident in the highest nitrate concentrations established in one of the major rivers (River Sabaki) draining over wide geographical areas in Kenya's highlands with intensive agriculture and the use of artificial fertilizers ([ASCLME, 2012](#)). These variations suggest that the expansion of agriculture and the use of artificial fertilizers in the river catchment as planned under the country's development plans could increase nutrient supply in the estuaries and coastal ocean.

The creeks located around urban populated areas also experience a high generation of municipal and domestic sewage from densely populated residential areas and tourism hotels. These get into the seawater through direct discharges, storm water, and seepage from pit latrines and septic tanks. Open defecation, poor sanitation, and infrastructure facilities in informal settlements were linked to raw sewage discharges into the ocean ([Kithiia & Majambo, 2020](#)). Elevated nutrients in the ground and seawaters around densely populated areas established in this review point toward a sewage management problem. The planned expansion of affordable housing along the coastal region under the country's development plans ([cn.invest.go.ke, 2017](#)) should thus be accompanied by a development in sanitation facilities to avoid sewage contamination of water resources. The problem of anthropogenic nutrient pollution was also reported in other urban areas in the region e.g., Chapwani in Zanzibar

(Kyewalyanga & Lugomela, 2001) and Dar es salam (Hamisi & Mamboya, 2014), and internationally e.g., Indonesia (Adyasari *et al.*, 2018), Long Island, Brazil (Niencheski *et al.*, 2007) and Kaneohe Bay in Hawaii (Hoagland *et al.*, 2002). This indicates that sewage-linked nutrient pollution is a global problem, particularly in developing countries.

The analysis of N-forms revealed elevated levels of NH_4^+ over NO_3^- in almost all the locations in the study. The supply of ammonium in the coastal waters is caused by bacterial remineralization of organic matter under anoxic conditions (Mengesha *et al.*, 1999; Hensen *et al.*, 2006), ballast water, urea fertilizers, aquaculture activities, and increased atmospheric deposition (Glibert, 2017; Jickells *et al.*, 2017). Elevated dissolved ammonium levels may result from high bacterial remineralization of organic matter from sewage and terrestrial runoff, as aquaculture and atmospheric deposition studies in the area remain unverified. High NH_4^+ relative to NO_3^- may also be due to the high consumption of NO_3^- by phytoplankton for primary production. The elevated ammonium levels over nitrates are a threat to the ecosystem's health as they can enhance the development of harmful algal blooms, as discussed in the next section.

The N: P ratios in the dissolved nutrient pool were consistently below the Redfield value (N: P = 16) in all the sites (Figure 9). Low N: P ratios were reported in other coastal areas and linked to increases in the anthropogenic supply of P and high assimilation of N during spring blooms (Malone & Newton, 2020). Low N: P established in the creek, seagrass, and reef waters thus suggests either higher assimilation or lower supply of dissolved inorganic nitrogen (DIN) relative to phosphorus in the area. Sewage discharges released in the coastal waters may have contributed to this due to the high use of detergents containing P nutrients. The nitrogen limitation established in this study agrees with earlier reports about tropical coastal waters (Glibert, 2017). Elevated N: P values established in nearshore and estuarine waters can be due to localized effects of dissolved nitrogen-enriched sewage effluents from hotels and residential areas seeping into the ocean through groundwater (Mwaguni *et al.*, 2017; Mwaura *et al.*, 2017) and agricultural runoff.

Coastal habitats like coral reefs and seagrasses play a critical role in supporting coastal biodiversity and livelihoods. However, elevated nutrient levels in coral reefs can threaten their health by stimulating phytoplankton growth hindering light penetration for the benthic macrophytes and corals (Raj *et al.*, 2020). High levels of DIN from sewage can affect the coral reefs' physiology, making them vulnerable to diseases (Bruno *et al.*, 2003). High nutrient inputs from sewage near coral reefs adjacent to dense tourism hotels require careful management and monitoring to maintain nutrient levels

within limits recommended for healthy coral ecosystems. Acute N limitation ($N:P < 2$) established in seagrass, reef waters, and some estuarine waters is a signal of the vulnerability of these habitats to nitrogen pollution, as was reported in the Gulf of California (Beman *et al.*, 2005). This indicates that many processes in these ecosystems are nitrogen-limited, and any slight supply of nitrogen can lead to an outburst in the ecological processes like primary production, in the system. Very low N:P ratios were reported to promote the growth of N-fixing organisms like *Trichodesmium spp.* in reef waters. *Trichodesmium spp.* blooms are known to be a potential source of ciguatoxins in the ciguatera food chain (Malone & Newton, 2020). Data were missing for Si apart from estuaries and creeks that registered values below normal ($N:Si < 1$) an indication of Si limitation relative to N for phytoplankton (diatoms) growth. Limitation of Si relative to N, and N relative to P can promote the growth of non-diatom nitrogen-fixing phytoplankton species, mainly harmful algal blooms (Glibert *et al.*, 2011; Glibert *et al.*, 2005).

A positive correlation was established between phytoplankton biomass (*Chl-a* concentration), phosphates, and ammonium nitrogen. This may be an indication that phosphates and ammonium are major drivers of primary production in the area. Phosphate and N limitation for primary production has been reported in earlier studies (Turner & Rabalais, 2013). High primary production coincided with high NO_3^- registered in the Sabaki estuary indicating that nitrate could be the source of nitrogen nutrition at the site. All these agree with the claims that human activities are affecting nutrient chemistry and enhancing algal growth in the receiving coastal waters (Anderson *et al.*, 2002; Malone and Newton, 2020).

The review also established lower *Chl-a* during the wet season when nutrient levels were high compared to the dry season in the estuarine and creek waters. This may have been caused by either dilution of the *Chl-a* concentrations by the incoming rainwater or reduced primary production water turbidity caused by sedimentation from runoffs (Kitheka *et al.*, 2005). Land-use changes characterized by intensive agriculture in the river catchment areas have facilitated high erosion and deposition of loose particulate matter and sediment in the estuaries during the early rains, causing high water turbidity (Geeraert *et al.*, 2015; Tamooch *et al.*, 2014).

Although most *Chl-a* concentration in the ecological sites is still below the eutrophication threshold values, e.g., DIN $\sim 1 \mu M$; P- $PO_4 \sim 0.1\text{--}0.2 \mu M$ for corals (Bell, 1992). The higher mean *Chl-a* concentrations ($> 0.2 \mu mol/L$) established in nearshore corals adjacent to urbanized areas are an early

warning to reduce the release of untreated sewage effluents into the ocean and poor use of agricultural fertilizers in river catchment areas. Low *Chl-a* concentrations are ideal for seagrass and corals since high phytoplankton biomass can affect the health of these habitats, as earlier stated in this review (Furnas *et al.*, 2005). These findings compare with the recent report from the neighboring waters, which indicated ammonium and nitrate as the drivers of high *Chl-a* in the coastal waters of Unguja (Peter *et al.*, 2018). The presence of ammonium has been reported to enhance phytoplankton primary production due to the discriminate preference of ammonium compared to other N-forms due to its low energy requirement during assimilation (Glibert, 2017; Lundgren *et al.*, 2016).

Diatoms play a very critical role in the oceans' primary production and biological processes. Their dominance in Kenya's coastal waters is significant to fisheries production since several juvenile fish larvae are herbivores depending on diatoms as the basic source of food (Arshad *et al.*, 2013). Cyanobacteria, chlorophytes, and dinoflagellates have a high affinity for NH_4^+ while diatoms, which are important in the fisheries food web, depend on NO_3^- uptake as a growth N substrate and to sustain cell energy balance (Lundgren *et al.*, 2016). Elevated NH_4^+ levels ($> 1\mu\text{mol/L}$) can therefore repress NO_3^- uptake and favor the blooming species over the non-blooming diatom resulting in a reduction in fisheries (Glibert, 2017). The reported cyanobacteria dominance in the area during the dry season due to their small sizes (Falkowski & Knoll, 2007), nitrogen-fixing ability during N-limiting situations (Sumich & Morrissey, 2004), allelopathic strategies against other phytoplankton species, and ability to use organic forms of phosphorus (Labry *et al.*, 2002). Low flagellate abundance established in Mombasa can be due to their motile nature, enabling them to move to areas with favorable environmental conditions.

Nutrient ratios shape the phytoplankton community structure. Relatively low N: P ratios were reported to change the phytoplankton abundance from a diatom-dominated to a more toxic mixotrophic dinoflagellate (Chia *et al.*, 2018; Glibert, 2017). Nutrients supplied from sewage and agriculture, as demonstrated in the creeks and estuaries, and those likely to occur with the expansion of aquaculture and maritime transport can potentially cause shifts in the ratio N:P or N: Si, resulting in an outbreak of HABs. This was reported in some U.S., European, and Asian coastal waters (Glibert, 2017) and aquaculture lagoons (Hassina *et al.*, 2008). Urbanized coastal areas and estuaries exposed to intensive use of synthetic urea fertilizers in the catchment area are also vulnerable to HABs development (Anderson *et al.*, 2002).

The results from Kilindini concur with those reported high toxic dinoflagellate composition in areas of ballast water discharges (Smayda, 2007). The reported dominance of dinoflagellates can be a threat to fisheries since they compete with juvenile fish larvae for diatoms. *Peridinium sp.* and *Protoperidinium sp.* are heterotrophic, they feed on diatoms to supplement their nutritional requirement during periods of low nutrient supply (Siano & Montresor, 2005). A decline in fish diversity with an increase in dinoflagellates has also been confirmed in other coastal waters (Glibert *et al.*, 2011).

Although the abundance of the HABs species is still below the HABs warning threshold, the presence of saxitoxin-producing species and other potentially toxic forms of dinoflagellates (*Alexandrium spp.*, and *Dinophysis spp.*) (Ogongo *et al.*, 2015) should be treated with caution. Additionally, seven out of twenty species of *Pseudo nitzschia* species in the Western Indian Ocean (WIO) region have been noted as domoic acid producers (Hansen *et al.*, 2001). These HABs can cause oxygen depletion, fish kills, and human intoxication when they bloom resulting in indiscriminate marine wildlife and physical harm through damaging and clogging of gills of fish and invertebrates.

Harmful algal blooms can cause habitat destruction by smothering and killing corals, seagrass, and mangroves with adverse impacts on fisheries and tourism, public health, and local livelihoods. *Heterosigma akashiwo*, *Trichodesmium*, and *Lyngbya majuscula* also threaten recreational activities due to their dermatological effects (skin irritation), with the impacts already reported in neighboring waters of Tanzania (Aziza *et al.*, 2018; Kimambo *et al.*, 2021; Msuya, 2015). The blooming of *Microcystis* species seriously affected salt production in Kenya in 1992 and 1993 (Munga *et al.*, 1992; ASCLME, 2012). Additionally, the bloom reported in Kiunga in 2002 caused dead zones resulting in extensive mortalities of marine life including fish and turtles; a ban on consumption and trade in shellfish, a halt in tourism, and human health problems like eye irritations and headaches, and greatly affected the local livelihoods (ASCLME, 2012; IUCN Eastern Africa Programme, 2004).

The abundance of HABs species recorded in Kenya is much higher than that of Lugomela (2013), who reported low *Pseudo-nitzschia* spp. abundances of <16 cells/l around sheltered inshore waters in Dar es Salaam throughout the 1-year and <18 cells/l reported in the offshore water of the Pemba Channel (Sekadende *et al.*, 2021). Earlier reports indicate HABs along the Somali coast (Obura, 2001) and recurrent *Microcystis* spp. blooms in Kenyan salt ponds. Red tides occur during November's northeast monsoon (ASCLME, 2012), while *Gymnodinium* spp. blooms are linked to strong

upwelling in the Kiunga Marine National Reserve (IUCN Eastern Africa Programme, 2004). Additionally, *Noctiluca scintillans*, *Peridinium quinquecorne*, and *Trichodesmium erythraeum* are commonly found in the Indian Ocean (Hansen *et al.*, 2001).

2.6.Implications to Management

The combination of nutrient conditions and phytoplankton plays a very critical role in determining the integrity and health of the coastal and marine ecosystem. Improved knowledge of nutrient concentration, stoichiometry, and phytoplankton community composition is thus important to the wider interest in the monitoring of these areas. Elevated nutrients reported in the creeks and estuaries (Okuku *et al.*, 2011; Ongore *et al.*, 2013) caused by sewage and terrestrial run-off from agriculture, especially during the rainy season is a clear indication of poor management of sewage and use of agricultural fertilizers in the river catchment areas. The eutrophic conditions established can have major consequences for the existing biodiversity and the benefits they provide to society like fish production. Although the area seems to be experiencing N limitation, which may be linked to high inputs of P from anthropogenic sources. The inputs of both N and P should be managed to ensure a stoichiometric balance of both nutrients, since P limitation was also reported in some areas.

Toxic phytoplankton blooms have harmed fisheries, coastal and marine ecosystems, public health, and coastal economies (Hoagland *et al.*, 2002). This review has summarized several potentially harmful microalgae species occurring in different locations in the area. The reported species like *Trichodesmium* spp., *Pseudo-nitzschia* spp., and *Dinophysis* spp. have the potential to be harmful, however, none of them were highly abundant and calls for alarm with reference HAB's warning systems (Albright *et al.*, 1993). Their presence in low abundances does not mean toxicity in the coastal waters, and therefore their threat to seafood and human health should be deemed minimal pending confirmation. However, a high abundance of *Pseudo-nitzschia* spp. was established in the Ramisi-Vanga system (Kiteresi *et al.*, 2012; Ogongo and Okumu, 2015) and Kilindini creek (Hansen *et al.*, 2001; Ogongo *et al.*, 2015) needs further investigations, as some species are toxic with the capability of producing domoic acid, responsible for ASP. The established nitrogen limitation can enhance the occurrences of N-fixing cyanobacteria, some of which are toxic. Reef fisheries are a major practice supporting many livelihoods in the area. However, blooms of *Trichodesmium* spp., caused by low N: P ratios can produce ciguatoxins causing contamination of seafood in the reef waters. Regular monitoring is thus important not only for the species but also to understand the HABs

dynamics, detect the existence of new species or toxicity of endemic species and manage the effects of naturally occurring blooms, especially if mariculture and offshore fisheries expansion is expected.

A rapidly expanding human population like the one occurring along the coast of Kenya was positively correlated with nutrient enrichment in adjacent coastal waters (Anderson *et al.*, 2002; Glibert *et al.*, 2005) and outbreaks of HABs (Trainer *et al.*, 2003) in other areas. Planned activities (expansion of industries, agriculture, and housing) under the country's economic development agenda (Nyonje, 2018), were also characterized by increased generation of nutrients (N and P) to the adjacent coastal waters if no subsequent sewage, industrial effluent, and agricultural management actions are established. Additionally, the outbreak of harmful dinoflagellate blooms may occur with an increase in ballast water. Improvement of existing sewage management infrastructures, the establishment of new ones where they are lacking and regulation of agricultural activities in the catchment areas are thus necessary to manage the nutrients and reduce their effects on the environment.

2.7. Conclusion

This review provides insight into the status of nutrient pollution and HABs that can guide coastal managers. This is a systematic review of previous studies, including forty original empirical research articles. The results indicate nitrogen limitation and occurrences of localized nutrient pollution in the creeks around major towns and estuarine environments. The review also established elevated *Chl-a* concentration in the estuaries and nearshore coral reefs. The toxic, harmful algal species were identified in all existing studies indicating that they are widespread in Kenya's coastal and marine waters below HABs warning standards

The results from this review call for regular monitoring programs on HABs and precautions on consuming related seafood to ensure public health safety. Elevated *Chl-a* concentrations in the estuaries and nearshore coral reefs also call for mechanisms to reduce sewage discharges and sustainable agricultural practices in the catchment areas to safeguard the reefs and other critical habitats in the area. There was also a spatial and temporal inconsistency in the existing studies making them difficult to compare and inhibiting the establishment of trends in nutrient and HABs dynamics in Kenya's coastal waters. To facilitate effective management decisions, comparative field studies of different sites are necessary especially in potential fisheries, aquaculture, and tourism sites.

Chapter Three: Nutrients and phytoplankton variations in Kenya's Coastal Waters



Green turtle Infected with tumors claimed to be amplified by eutrophication. Image credit Daniel Katana (MABICO), Marereni-Kenya

Modified from,

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Anthropogenic Nutrients and Phytoplankton Diversity in Kenya's Coastal Waters: An Ecological Quality Assessment of Sea Turtle Foraging Sites

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3.1. Chapter Summary

We assessed the ecological quality status (EQS) of coastal waters following claims of increasing sea turtle fibro-papillomatosis (FP) infections in Kenya, a disease hypothesized to be associated with 'poor' ecological health. We established widespread phosphate (P) and silicate (Si) limitation, dissolved ammonium contamination, and an increase in potentially harmful microalgae species. Variations in the EQS were established in the sites depending on the indicators used and seasons. Generally, more sites located near hotels, tidal creeks, and estuarine areas showed 'poor' and 'bad' EQS during the rainy period compared to dry season. Additionally, 90.1 % of the sites in 'poor' and 'bad'

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EQS based on dissolved inorganic nitrogen. Low dissolved oxygen, elevated temperature, salinity and ammonium, ‘poor’ EQS based on DIN, and potential bio-toxin-producing phytoplankton species characterized the FP prevalent areas, specifically during the dry season suggesting environmental stress pointing to the hypothesized connection between ecological and sea turtle health.

3.2.Introduction

Coastal ecosystems offer a wide range of services for humans, including supporting ecosystem functions and maintaining water quality. Coastal and marine resources support the livelihoods of over 2.5 billion people living in coastal areas globally (UNEP, 2017). However, expanding coastal populations, economic activities, and settlements have resulted in the rapid degradation of coastal regions (Fuentes *et al.*, 2016). Anthropogenic activities such as land-use for agriculture have altered the hydrologic cycle (Zhang *et al.*, 2022), increased nutrient enrichment in coastal waters (Malone & Newton, 2020), and led to the occurrence of invasive species (Halpern *et al.*, 2008). Increased enrichment of coastal waters with nitrogen-N and phosphorus-P from anthropogenic sources triggers biomass production resulting in eutrophication (Anderson *et al.* 2002; Heil *et al.* 2007). This causes reduced water clarity, oxygen depletion, loss of biodiversity and critical coastal habitats (coral reefs, seagrass meadows, and mangrove forests), increased occurrences of toxic algal blooms, and threats to human health (Anderson *et al.*, 2002; Malone & Newton, 2020). Further, eutrophication has been associated with the disappearance of important native species and the increase of invasive species (Santos *et al.*, 2011; Turner & Rabalais, 2013) as well as the promotion of infections like fibropapillomatosis (FP) in sea turtles (Aguirre & Lutz, 2004; Dujon *et al.*, 2021).

Sea turtles are a group of seven threatened species widely spread throughout the world's tropical and subtropical coastal and marine habitats. These species are vulnerable to anthropogenic pollution due to their high public profile, which increases their exposure to environmental contaminations (Aguirre & Lutz, 2004). Heavily polluted coastal areas impacted by industrial, agricultural, or urban development have been linked with an increase in disease susceptibility and prevalence of FP in sea turtles due to repressed physiology, chronic stress, and impaired immune function (Jones *et al.* 2016; Dujon *et al.* 2021). Owing to this, scientists have advocated for an integrated assessment of marine coastal environments that incorporates the physical, biological, and chemical aspects of the environment (Aguirre and Lutz 2004; Patrício *et al.* 2012) to promote our understanding of the sources of environmental stressors.

The coastal and marine ecosystem in Kenya is characterized by several creeks, estuaries, fringing coral reefs, seagrass beds, and mangrove swamps. The region supports various ecosystem services, abundant biodiversity (ASCLME 2012), and the livelihoods and economies of local coastal communities (Musembi *et al.*, 2019). Increasing anthropogenic nutrient enrichment from direct sewage discharges (Okuku *et al.*, 2011), agricultural run-off (Ongore *et al.*, 2013), and chronic nuisance algal blooms (Okello *et al.*, 2022) have been documented in the area. Additionally, a rise in sea turtle infections and mortality linked to the FP epidemic has also been reported (Jones *et al.*, 2021; Mkare & Katana, 2022; van de Geer *et al.*, 2022). With the current rapid human population growth, urbanization, industrialization, and planned expansions in agriculture in riverine areas, mariculture, and marine transport (cn.invest.go.ke, 2017; Nyonje, 2018), eutrophication problem and general degradation of the coastal environments are likely to intensify. Assessing the quality of coastal and marine ecosystems in Kenya is thus necessary to inform management decisions in this transition.

This study assessed the ecological quality status of sea turtle habitats (foraging sites) located in Kenya's coastal waters through the integration of physical, chemical, and biological indices. Specifically, we (1) explored the spatial variability of environmental (physicochemical) factors and phytoplankton assemblages, (2) determined their relationships, and (3) used selected results from (1) to establish the ecological quality of the assessed sites. Our results provide insights into the environmental and biological factors affecting the ecological health of the coastal ecosystems and provide pathways for the integration into conservation and management strategies.

3.3. Materials and Methods

3.3.1. Study area

The Kenya coastline is located on the western side of the Indian Ocean and extends in the north-south direction from Kiunga (1° 41'S) to Vanga (4° 40'S) along 640 km (Figure 11; Kaunda-Arara *et al.*, 2004). The area is primarily affected by the semi-diurnal tides with spring tidal variation of up to 4.0m. The weather pattern in the region is strongly influenced by the southeast monsoon (SEM) winds occurring from April to October, characterized by low temperatures, high rainfall, and strong currents and winds, and northeast monsoon (NEM) winds from November to March with warmer temperatures, low rainfall, and mild seas. The temperatures range between 25°C and 31°C

(ASCLME, 2012). The major marine habitats in the area are coral reefs, seagrass meadows, mangrove swamps, and sandy beaches, which provide critical foraging, breeding, and nesting grounds for sea turtles besides supporting essential tourism and fishing grounds for the local communities (Musembi *et al.*, 2019). The Kenya coast is characterized by high human population growth (KNBS, 2019), intense industrial, urbanization, and recreational activities, and rivers draining from catchments with intensive agriculture activities (ASCLME, 2012; Kithiia & Majambo, 2020).

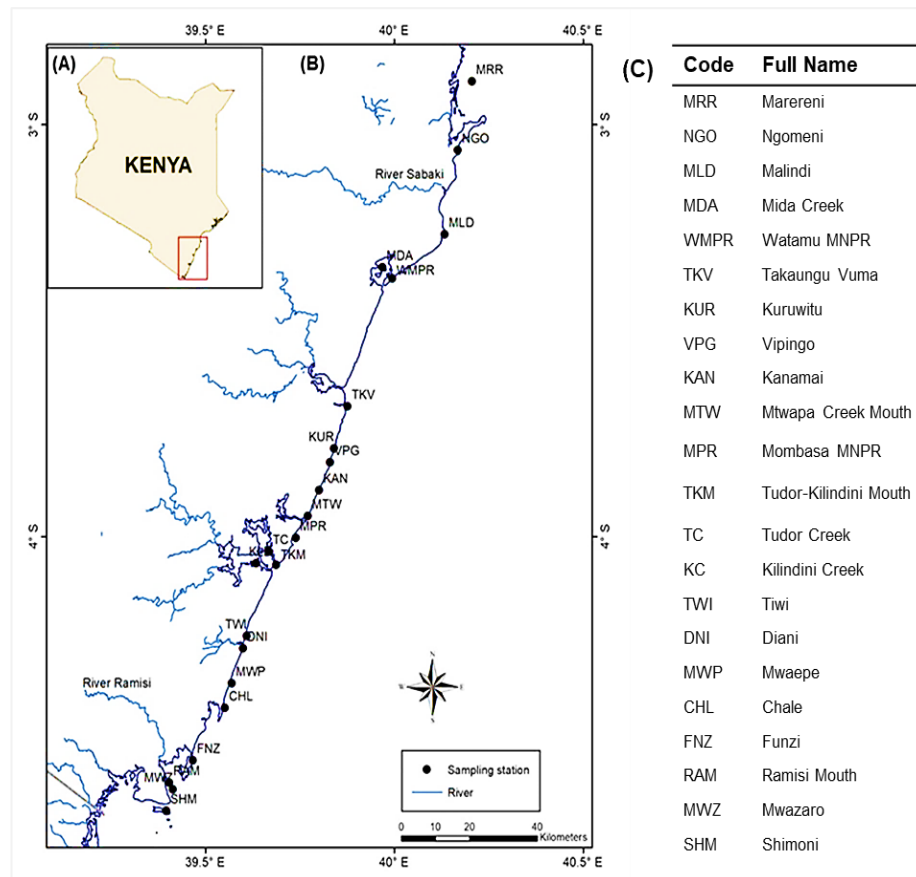


Figure 11: A map of Kenya (A), study areas (B) showing the assessed potential turtle foraging sites, and the sites names (C). (Source of the map – author’s construct)

3.3.2. Site selection and field sampling

This study focused on sea turtle foraging sites located in coastal areas with possible exposure to anthropogenic pressures. This study utilized Kenya's Beach Management Unit (BMU) framework—established under the 2007 Fisheries Regulations (Government of Kenya, 2012) governing fish landing sites—to designate stations and sub-units as sampling sites. The sites were further grouped into four major geomorphological and/or hydrological categories (tidal creeks, estuarine, nearshore, and

oceanic ecosystems) reflecting different coastal ecosystems. According to this study, the tidal creeks were defined as tidal channels mostly containing saline water and low river influence, estuarine as sites located within river mouths, nearshore as sites < 500m from the shoreline, and oceanic sites as island sites located \geq 500m from the shoreline with less anthropogenic exposure. See [Appendix 3](#) for a full description of site selection procedures. The ecological surveys were conducted across 25 locations along Kenya's coast, including five tidal creeks: Tudor (TC), Kilindini (KLD), Mtwapa (MTW), Takaungu-Vuma (TKV), and Mida (MDA). Additionally, four estuarine stations were surveyed: Marereni (MRR), Ngomeni (NGO), Ramisi (RAM), and Mwazaro (MWZ). The study also included twelve nearshore stations: Shimoni (SHM), Funzi (FNZ), Chale (CHL), Mwaepe (MWP), Diani (DNI), Tiwi (TWI), Shelly (SHL), Mombasa Marine National Park and Reserve (MMNPR), Kanamai (KAN), Kuruwitu (KUR), Watamu Marine National Park and Reserve (WMNPR), and Malindi (MLD). Finally, four oceanic sites were examined, specifically Kisite Mpunguti, Whale Island, and Magic Island in the SHM, WMNPR, and MLD areas, respectively.

A total of 95 sites in the above locations were assessed between September 2021 and July 2022 along a stretch of 264 km of Kenya's coastline. Out of the 95 sites, a total of 68 sites located in 18 stations were assessed in September 2021 while 9 sites located in two stations, TKV and NGO, were surveyed in March 2022 due to earlier inaccessibility because of rough sea conditions. A second follow-up survey was conducted in July 2022 in 36 sites in five stations namely Mombasa MNPR, DNI, TWI, MDA, and WMNPR, which were selected based on their ecological importance as marine protected areas (MPAs) and previous reports of the prevalence of FP infections ([Jones et al. 2021](#); [van de Geer et al. 2022](#)). Additional areas assessed in July were the RAM estuary and MWZ in south coast Kenya.

3.3.3. Sampling and laboratory analysis

Surface seawater samples were collected during low tides from each site during the survey. *In situ* measurements for temperature, pH, salinity, conductivity, and dissolved oxygen (DO) were conducted using a portable YSI Professional multi-probe meter, and turbidity was measured using a portable TN100 turbidity meter. All the in-situ measurements and sample collections were done independently two - three times in each site, depending on the spatial coverage of the sites. Field data was collected using the Kobo Toolbox toolkit. All the sampling and laboratory analytical procedures were conducted according to [Helcom and UNEP \(2013\)](#) and [Grasshoff et al. \(2007\)](#). The laboratory analysis of nutrients and Chlorophyll a (*Chl a*) was conducted at the Leibniz Centre for Tropical

Marine Research (ZMT), Germany while phytoplankton analysis was done at Kenya Marine and Fisheries Research Institute (KMFRI), Kenya.

Dissolved inorganic nutrients

Nutrient samples were filtered through disposable syringe filters (Sartorius Minisart ® - 0.45 µm pore size) immediately after sampling, filled into pre-rinsed polyethylene bottles preserved with a mercury chloride solution (0.06mL HgCl₂-solution/10mL sample), and stored frozen. The samples were transported to Germany on dry ice. In the laboratory, nutrient samples were analyzed using TECAN microplate reader-based colorimetric methods. The spectrophotometric method for the sequential determination of nitrate and nitrite at low concentrations in small volumes was used for the analysis of NO_x⁻ (nitrite- NO₂⁻ + nitrates -NO₃⁻) concentrations (García-Robledo *et al.*, 2014). Salicylate method for ammonium-NH₄⁺, PhosVer3 ascorbic acid method for orthophosphate-PO₄³⁻ and silicate-SiO₃²⁻ established using the silicomolybdate method (Ringuet *et al.*, 2011). The detection limit (DL) and limit of quantification (LOQ) of each nutrient were determined according to ISO 11843-2 and the German standard DIN 32645 using blank procedures. The DLs were 0.2 µM, 0.1 µM, and 0.1 µM for NO_x⁻, PO₄³⁻, and NH₄⁺, respectively. In this study, the term dissolved inorganic nitrogen (DIN) refers to the sum of NO_x⁻ and NH₄⁺.

Phytoplankton assemblages

Phytoplankton biomass (*Chl a*) samples were collected by filtering 1L of water onto Whatman GF/F filters under pressure using a handheld pump. At sites with high turbidity, 500ml was taken. The filters were stored frozen until analysis. These were analysed in the laboratory using a non-acidification fluorometric method with a narrow band filter according to Welschmeyer (1994). The accuracy and consistency of the analytical procedures were established by analysing check standards.

For phytoplankton species diversity and abundance analysis, 20 L of seawater was collected using a bucket and filtered through a 20µm phytoplankton net, concentrated up to 50ml fixed in 5% Lugol solution, and kept for species identification and tallying. Phytoplankton species identification and counting were done using an inverted compound microscope (Leica DMIL) following the taxonomic monograph of Botes (2001) and Carmelo and Grethe (1997). The phytoplankton was classified into three taxonomic groupings (class, genus, and species), and characteristic functional groups (FGs). According to this study, the FGs were principally based on the harmful effect of the species on

humans, aquatic biodiversity, and ecosystems according to the IOC-UNESCO List of harmful algal blooms (HABs) (Hansen *et al.* 2001; IOC-UNESCO 2008).

3.3.4. Statistical analysis

All data analysis was conducted using Sigma Plot version 14.0.3.192, R version 4.2.3 (R Core Team, 2024) with the vegan package (Oksanen *et al.*, 2015) and PRIMER-E (v7.0.22) (Clarke & Gorley, 2006). Spatial analyses were conducted using QGIS v3.28.3.

Univariate Analysis of Measures of Diversity

Analyses were conducted on univariate response variables and multivariate (site by phytoplankton abundance matrix). Based on Magurran, (2004) the univariate measures for phytoplankton indices were analyzed:

- i) Margalef's species richness (d), defined as the total number of different species found in a sample:

Equation 1:

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

Where S is the number of species and N is the abundance/total number of individuals, Margal, (1958),

- ii) Abundance (cell density) expressed as the number of cells per liter of seawater, calculated using the formula by Stirling (1985) below:

Equation 2:

$$N = \frac{A \times 1000 \times C}{V \times F \times L}$$

Where: N = Number of phytoplankton cells per liter of the original water; A = Total number of phytoplankton counted; C = Volume of the final concentrate of the sample in mL; V = Volume of the field in mm³; F = Number of fields counted; and L = Volume of the water filtered in liters.

- iii) Shannon-Wiener diversity (H'), which measures the diversity of phytoplankton species in a community, was calculated from the phytoplankton species and abundance data for each site using the formula:

Equation 3:

$$H' = - \sum \rho_i \times \ln(\rho_i)$$

Where \ln is the natural logarithm and ρ_i is the proportion of the entire community made up of species i (Shannon, 1948); and (iv) Pielou's evenness (E) (Pielou, 1996):

Equation 4:

$$E = H / \ln S$$

Spatial patterns in phytoplankton assemblages

To explore spatial changes in phytoplankton assemblage composition, permutational multivariate analysis of variance (PERMANOVA) (Clarke & Warwick, 2001) based on Bray-Curtis similarity of \log_{10} transformed data (Bray & Curtis, 1957) was used to determine whether sites differed significantly from each other. Further, maps showing how the biomass, nutrients, and other physicochemical parameters varied across the sites were explored using QGIS v3.28.3.

Relationship of phytoplankton assemblages to environmental predictors

To examine correlations between environmental variables and phytoplankton assemblages, we first examined multi-collinearity among normalized environmental variables using Spearman's correlation coefficient (ρ) and scatter plot matrices to eliminate co-linear variables and to reduce redundancy. The variables with the largest potential ecological importance were used as surrogates for those variables with which they were highly correlated ($|\rho| \geq 0.85$) (Laliberte & Legendre, 2010). For the multivariate analysis, a distance-based linear model (DistLM) (Clarke and Gorley 2015; Boj, et al. 2016) with stepwise regression as the selection procedure, using the Akaike Information Criterion (AIC) as the selection criterion, was used to derive the most parsimonious models predicting phytoplankton communities, and for the distance-based redundancy analysis (dbRDA) models. The DistLM enabled us to identify predictor variables (on the normalized scale) that contributed significantly to the spatial patterns observed in the assemblage structure, as well as determine how much

variation was explained by each predictor. The dbRDA plot enabled us to visualize the relative contributions of each of the predictor variables on the assemblage structure (Shankar *et al.* 2017; Jupke and Schäfer 2020).

For the univariate analysis, multiple linear regression with stepwise regression as the selection procedure, and models evaluated using the Akaike Information Criterion (AIC) as the selection criterion were used to derive the most parsimonious models predicting each univariate measure. Variance inflation factors (VIFs) were employed to examine how much multicollinearity exists in the multiple regression analysis and none of the VIFs exceeded 2.5. Thus, the partial regression coefficients likely provided reliable estimates of the effects of each predictor variable while holding the effects of all other variables constant (Berk, 2003).

Establishment of the ecological quality status of the study sites

To assess the ecological quality status (EQS) of sea turtle foraging sites, this study adopted modified boundaries [Ecological Quality Ratios (EQR)] from the European Water Framework Directive (WFD - 2000/60/EC). These include inorganic nutrient concentrations (DIN, DIP), ratios (N:P, N:Si), chlorophyll a, and dissolved oxygen (DO) (Newton *et al.*, 2022; Andersen *et al.*, 2010; Simboura *et al.*, 2005; Table 5). The criteria adopted the Redfield ratio (16:1 for N: P, ~1:1 for N: Si) as reference conditions for 'high' ecological status. Deviations were ranked sequentially as 'good', 'moderate', 'poor', and 'bad'. They also used percentile metrics and the EQR values to set the actual deviations (AcDev) and the status boundaries, respectively (Andersen *et al.*, 2010). Additional criteria used were Shannon-Wiener diversity (H') (Karydis & Tsirtsis, 1996), species functional groups (nature of harm to humans and ecosystems) (IOC-UNESCO, 2008), and algal species pollution index according to Palmer, (1969) in Elshobary *et al.* (2020). In the context of this study, the words 'ecological quality' and 'ecological health' are assumed to mean the same thing and are used interchangeably throughout this study.

Table 5: Ecological quality status classification scheme based on nutrients (nitrates, dissolved inorganic nitrogen-DIN, and phosphate –DIP), phytoplankton biomass (Chlorophyll a – Chl a) and dissolved oxygen (DO) modified from Simboura et al. (2005), Elshobary et al. (2020) and Newton et al. (2022). “Ref. con” represents the Reference condition.

Indicator	High	Good	Moderate	Poor	Bad	Source
DIN (μM)	$X < 0.71$	$0.71 < X < 1.61$	$1.61 < X < 7.81$	$7.81 < X < 16.54$	$X > 16.56$	Simboura et al. (2005) and Newton et al. (2022).
DIP (μM)	$X < 0.11$	$0.11 < X < 0.25$	$0.25 < X < 0.93$	$0.93 < X < 1.44$	$X > 1.44$	Simboura et al. (2005) and Newton et al. (2022)
N: P Ref. con = 16	$13.91 < X < 18.34$	$10 < X < 18.34$; $18.34 < X < 23.04$	$2.57 < X < 10$; $23.04 < X < 64.86$	$0.45 < X < 2.57$; $64.86 < X > 128.07$	$X < 0.45$; $X > 128.07$	Simboura et al. (2005) and Newton et al. (2022)
N: Si (Ref. con = 1)	$0.68 < X > 1.15$	$0.47 < X < 0.68$; $1.15 < X < 1.52$	$0.14 < X < 0.47$; $1.52 < X < 4.68$	$0.06 < X < 0.14$; $4.68 < X < 7.87$	$X < 0.06$; $X > 7.87$	Simboura et al. (2005) and Newton et al. (2022)
Chl a (μg/L)	< 0.1	$0.1 < X < 0.6$	$0.6 < X < 2.21$	$2.21 < X > 2.99$	> 2.99	Simboura et al. (2005) and Newton et al. (2022)
DO % Sat.	$98 < X < 104$	$94 < X < 98$; $104 < X < 109$	$70 < X < 94$; $109 < X < 127$	$45 < X < 70$; $127 < X < 140$	$X < 45$; $X > 140$	Newton et al. (2022).
Others						
Shannon-Wiener diversity (H')	$X < 1$ = Heavily polluted (HP); $1 < X < 3$ = Moderately Polluted; $X > 3$ = Clean Water					Karydis & Tsirtsis, (1996)
Algal species pollution index	$X < 10$ = Low organic pollution; $10 < X < 15$ = Moderate pollution; $15 < X < 20$ = Probable high organic pollution; $X \geq 20$ = High organic pollution.					Elshobary et al. (2020)

3.4.Results

3.4.1. Spatial variability in environmental factors and phytoplankton assemblages

Physical Parameters

The seawater was generally oxidic. However, significant spatial variations were observed across the different ecosystems. The mean dissolved oxygen (DO) in the sites ranged from 4.4 to 9.8mg/L and was significantly higher in the nearshore areas compared to the areas in the estuaries and tidal creeks. Significantly higher DO was established in July ($7.6\text{mg/L} \pm 1.2\text{mg/L}$; $P < 0.001$) compared to $6.75 \text{ mg/L} \pm 1.1 \text{ mg/L}$ in September/March. Dissolved oxygen was substantially lower ($< 5\text{mg/L}$) in most sites in NGO during the March survey. The mean seawater temperature in the sites ranged between 24.5 and 29.7°C and was significantly higher in tidal creeks and estuarine waters compared to nearshore and oceanic sites. Temperatures ranged from 25.5 - 29.7°C and 24.7 – 26.9°C during September/March and July surveys, respectively. Similar to temperature, salinity was significantly higher in the tidal creeks compared to estuarine and nearshore waters. The mean

salinity ranges in the sites were 34.1-36.8 and 29.1 – 35.6 PSU during the September/March and July surveys, respectively. Localized low salinity (27 - 28 PSU) was established in TKV and MMNPR areas around Coral Garden in March and July after the rains, respectively, while substantially higher seawater temperatures (28.8°C – 29.7°C) and salinities (36.2 PSU – 37.7 PSU) were observed in MRR and NGO areas during the dry period. On the other hand, the turbidity was significantly higher in estuarine and tidal creek waters compared to nearshore and oceanic sites (Table 6 and Appendix 4). Elevated mean turbidity levels of 28.5 - 43.4 NTU were also observed in the TC waters adjacent to Mombasa City and estuarine areas in MWZ, NGO, and MRR.

Chemical Parameters (Dissolved inorganic nutrients)

Overall, the concentrations of nitrate + nitrite- NO_x^- ranged from < DL to 182.4 μM during the study period. The highest ecosystem's mean NO_x^- of $17.1 \pm 17\mu\text{M}$ was recorded in oceanic waters. Higher NO_x^- concentrations of 117.8 – 182.4 μM were recorded in nearshore areas in VPG, TC (21.5 μM and 42.6 μM), and estuarine waters in MLD and MRR areas (16 μM – 27.4 μM) compared to other sites. However, very low NO_x^- concentrations of 0.41 μM - 1.70 μM were observed in the nearshore waters around the WMNPR station. Significantly higher NO_x^- ($p = < 0.001$) were also observed in September/March compared to July with overall means of $11.5 \pm 9.4 \mu\text{M}$ and $3.0 \pm 6.6 \mu\text{M}$ respectively (Appendix 5).

Ammonium levels in the sites ranged between 0.76 to 63.3 μM during the study and were significantly higher in all the tidal creeks compared to other areas (Table 6). Significantly, higher concentrations of NH_4^+ were also observed in July with a mean value of $26.8 \pm 16.8 \mu\text{M}$ compared to $9.6 \pm 11.8 \mu\text{M}$ in September/March. Unlike NO_x^- , higher NH_4^+ concentrations of 10 μM – 41.4 μM were observed in WMNPR and other sea turtle tumor prevalence areas (TKV, MDA, and NGO) (Appendix 4). The general spatial trend in sites' mean DIN ($\text{NO}_x^- + \text{NH}_4^+$) concentrations was similar to that of ammonium. The DIN concentrations in the sites were 3.9 - 186.0 μM during the study and were significantly higher in tidal creeks and estuarine waters compared to nearshore and oceanic sites (Figure 13A and Table 6).

The mean phosphate concentrations in the sites ranged between 0.06 and 23.5 μM with higher concentrations observed in the nearshore and estuarine sites compared to tidal creeks and oceanic sites. High sites' mean PO_4^{3-} concentrations of 6.7 and 23.5 μM were registered in nearshore sites in the MWZ and SHL stations, respectively. Localized slightly higher PO_4^{3-} concentrations (1 μM

– 1.97 μM) were also observed in areas around TC and Wasini Channel in SHM, and estuarine waters around MLD and MRR. The temporal variations in PO_4^{3-} concentrations were statistically insignificant ($p > 0.05$), with means of $0.36 \pm 0.3 \mu\text{M}$ and $0.38 \pm 0.3 \mu\text{M}$ recorded during September/March and July surveys, respectively.

Table 6: Mean (\pm S.D.), ranges, and *P*-values of physicochemical and biological parameters of water in the different ecosystem types recorded along the Kenya coast. < DL = below detection limit). ***Spatial Variance -Overall P-value*
– *bold = Statistically significant***

	<i>Parameter</i>		<i>Creek</i>	<i>Estuary</i>	<i>Nearshore</i>	<i>Oceanic</i>	<i>Overall</i>	<i>Overall P-value</i>
Physical parameters	DO (mg/L)	$\bar{x} + \text{S.D.}$	6.3 ± 1.1	6.3 ± 0.9	7.5 ± 1.1	7.3 ± 0.7	7.8 ± 1.24	< 0.001
		Range	4.7 - 9.4	4.4 - 9.0	5.6 - 9.8	6.7 - 8.1	4.4 - 9.8	
	Temperature ($^{\circ}\text{C}$)	$\bar{x} + \text{S.D.}$	27.1 ± 0.8	26.6 ± 1.4	26.6 ± 0.8	26.2 ± 0.3	26.7 ± 1.0	0.02
		Range	25.7 - 28.4	24.7 - 29.7	25.1 - 28.7	26.0 - 26.6	24.5 - 29.7	
	Salinity (PSU)	$\bar{x} + \text{S.D.}$	35.3 ± 1.8	34.9 ± 1.4	35.1 ± 0.5	35.1 ± 0.04	35.1 ± 1.1	0.001
		Range	25.5 - 36.8	29.1 - 36.3	33.2 - 35.6	35.1 - 35.2	27.1 - 37.7	
	Turbidity (NTU)	$\bar{x} + \text{S.D.}$	3.3 ± 1.4	7.5 ± 7.9	3.0 ± 5.4	1.8 ± 0.6	4.7 ± 9.4	0.277
		Range	1.2 - 110	0.1 - 34.5	0.2 - 43.4	1.1 - 2.2	0.02 - 110	
Chemical Parameters (Dissolved inorganic nutrients)	pH	$\bar{x} + \text{S.D.}$	7.8 ± 0.4	7.6 ± 0.5	7.8 ± 0.4	7.8 ± 0.4	7.7 ± 0.5	0.003
		Range	7.2 - 8.4	7.0 - 8.4	7.1 - 8.3	7.4 - 8.1	7.4 - 8.4	
	Conductivity ($\mu\text{S/cm}$)	$\bar{x} + \text{S.D.}$	56099 ± 1435	54573 ± 3018	54934 ± 827	54085 ± 1083	54957 ± 1716	0.001
		Range	53658 - 58136	45592 - 59917	53328 - 57670	52854 - 54892	45592 - 59917	
	NO_3^- -($\text{NO}_3^- + \text{NO}_2^-$) (μM)	$\bar{x} + \text{S.D.}$	9.0 ± 10.3	8.7 ± 9.8	10.5 ± 15.6	17.1 ± 17.0	10.0 ± 17.0	0.929
		Range	0.01 - 36.5	< DL - 33.9	< DL - 117.8	3.7 - 36.2	< DL - 182.4	
	NH_4^+ (μM)	$\bar{x} + \text{S.D.}$	20.6 ± 18.8	12.1 ± 12.3	8.5 ± 10.4	3.3 ± 0.8	12.8 ± 20.8	0.004
		Range	2.7 - 63.3	2.9 - 50.3	0.8 - 56.8	2.7 \pm 3.9	0.8 - 168.7	
Biological Parameters (Phytoplankton indices)	DIN (μM)	$\bar{x} + \text{S.D.}$	29.0 ± 16.1	24.8 ± 11.8	18.4 ± 13.9	10.7 ± 4.5	22.8 ± 24.4	<0.001
		Range	7.6 - 58.4	7.6 - 58.4	3.9 - 71.1	7.5 - 13.9	3.9 - 186.0	
	PO_4^{3-} (μM)	$\bar{x} + \text{S.D.}$	0.24 ± 0.1	0.43 ± 0.23	0.80 ± 2.9	0.38 ± 0.3	0.7 ± 4.23	0.057
		Range	0.09 - 0.48	0.1 - 0.93	0.06 - 23.5	0.12 - 0.66	< DL - 69.0	
	SiO_4^{2-} (μM)	$\bar{x} + \text{S.D.}$	5.2 ± 3.2	7.9 ± 8.3	6.8 ± 5.2	4.3 ± 1.5	6.4 ± 7.0	0.505
		Range	1.0 - 14.5	1.0 - 47.7	1.7 - 34.2	3.0 - 5.9	< DL - 74.3	
	N:P	$\bar{x} + \text{S.D.}$	142.3 ± 98.1	90.0 ± 75.1	58.8 ± 38.8	67.9 ± 65.2	103.2 ± 113.9	<0.001
		Range	33.4 - 373.7	15.4 - 312.2	2.4 \pm 21.7	21.7 - 114	0.08 - 801.0	
Biological Parameters (Phytoplankton indices)	N: Si	$\bar{x} + \text{S.D.}$	9.5 ± 8.3	8.1 ± 10.2	3.0 ± 2.1	2.3 ± 1.5	13.0 ± 51.3	<0.001
		Range	1.3 - 27.6	1.4 - 36.6	0.8 - 12.8	1.3 \pm 3.4	0.002 - 483.5	
	Chl a [$\mu\text{g/L}$]	$\bar{x} + \text{S.D.}$	0.56 ± 0.45	0.63 ± 0.56	0.31 ± 0.16	0.25 ± 0.14	0.4 ± 0.4	0.031
		Range	0.1 - 1.56	0.07 - 2.34	0.07 - 0.65	0.16 - 0.41	0.06 - 2.34	
	Abundance (cells/L)	$\bar{x} + \text{S.D.}$	1388 ± 1642	8640 ± 21800	667 ± 412	1676 ± 1819	2341 ± 9960	0.370
		Range	233 - 7669	206 - 86100	113 \pm 2558	450 - 3767	113 - 86100	
	Species Diversity Index (H') (bits/cell)	$\bar{x} + \text{S.D.}$	2.60 ± 0.33	2.30 ± 0.77	2.42 ± 0.30	2.26 ± 0.48	2.43 ± 0.45	0.038
		Range	1.95 - 3.18	0.54 - 3.37	1.70 - 3.15	1.71 - 2.64	0.54 - 3.37	
Biological Parameters (Phytoplankton indices)	Evenness	$\bar{x} + \text{S.D.}$	0.82 ± 0.07	0.77 ± 0.23	0.87 ± 0.10	0.81 ± 0.13	0.85 ± 0.14	0.265
		Range	0.67 - 0.94	0.19 - 0.97	0.62 - 0.99	0.71 - 0.95	0.19 - 0.99	
	Richness	$\bar{x} + \text{S.D.}$	25 ± 9	22 ± 11	18 ± 7	17 ± 7	19 ± 8	0.040
		Range	10 - 43	9 - 53	9 - 37	11 - 25	6 - 53	

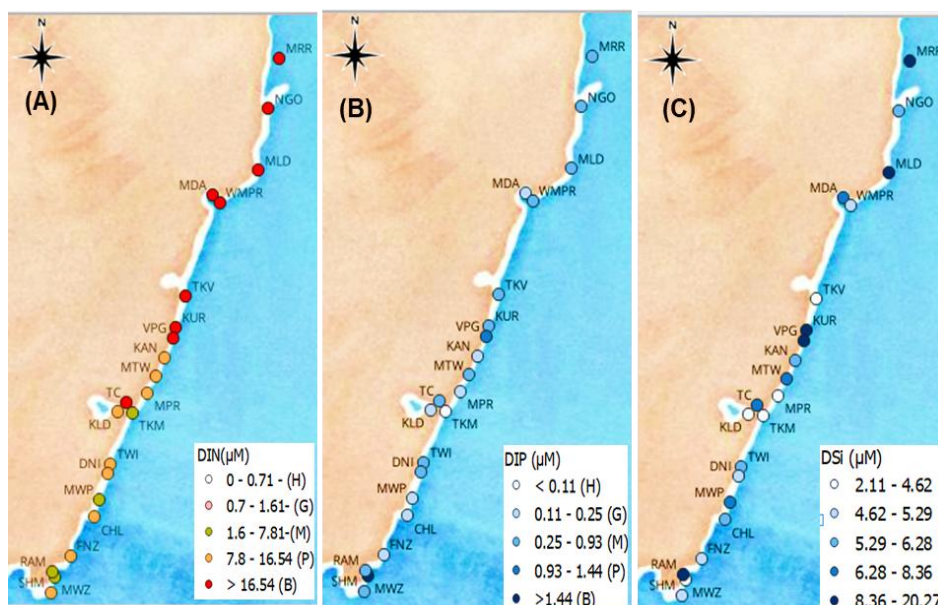


Figure 12: Dissolved inorganic nutrients, nitrogen-DIN ($\text{NO}_x^- + \text{NH}_4^+$), Phosphate-DIP (PO_4^{3-}), and Silicate- DSi (SiO_3^{2-}) in turtle foraging areas. *Refer to Figure 11 for full code names. Abbreviations for EQS ‘H’ – High, ‘G’- Good, ‘M’ -Medium, ‘P’- Poor, and ‘B’ – Bad

The overall ranges of silicate concentrations in the sites were 1.01 - 74.3 μM during the study. The silicate concentrations were slightly higher in the estuarine and nearshore waters compared to the tidal creek and oceanic waters. The concentrations were significantly higher in July after the long rains with ranges of 1.5 - 47.7 μM compared to 1.0 - 34.2 μM recorded in September/March ($P = < 0.001$). Localized higher SiO_3^{2-} concentrations of 29.1 – 34.2 μM were also registered in near-shore sites around VPG and SHL, and tidal creek (TC) waters near Coast General Hospital (Figure 12C). Silicate concentrations were substantially very low in TKV and NGO (1.01 - 2 μM) during the March surveys. Additionally, an increasing trend in SiO_3^{2-} concentrations was observed up-stream at the RAM estuary in July.

The mean N: P ratios in the sites ranged between 0.001 – 177.2 and 15.38 – 373.7 based on NO_x^- and DIN respectively. However, based on DIN, N: P values ranging between 0.001 and 801.0 were registered during the study. Elevated N: P ratios were established in the sites located in the tidal creeks and differed significantly with nearshore sites and insignificantly with estuarine and oceanic sites (Figure 13). The spatial and temporal trends of N: P were comparable to that of ammonium with significantly higher mean N: P (108.3 ± 115.3) established in July compared to 75.3 ± 59.7 in September/March. We also recorded some very low N: P ratios ($\text{N: P} < 2$) in some nearshore sites

in SHL and FNZ, while higher ratios of 179.1 and 500.2 were established in some sites in WMNPR including Coral Garden (Figure 13A).

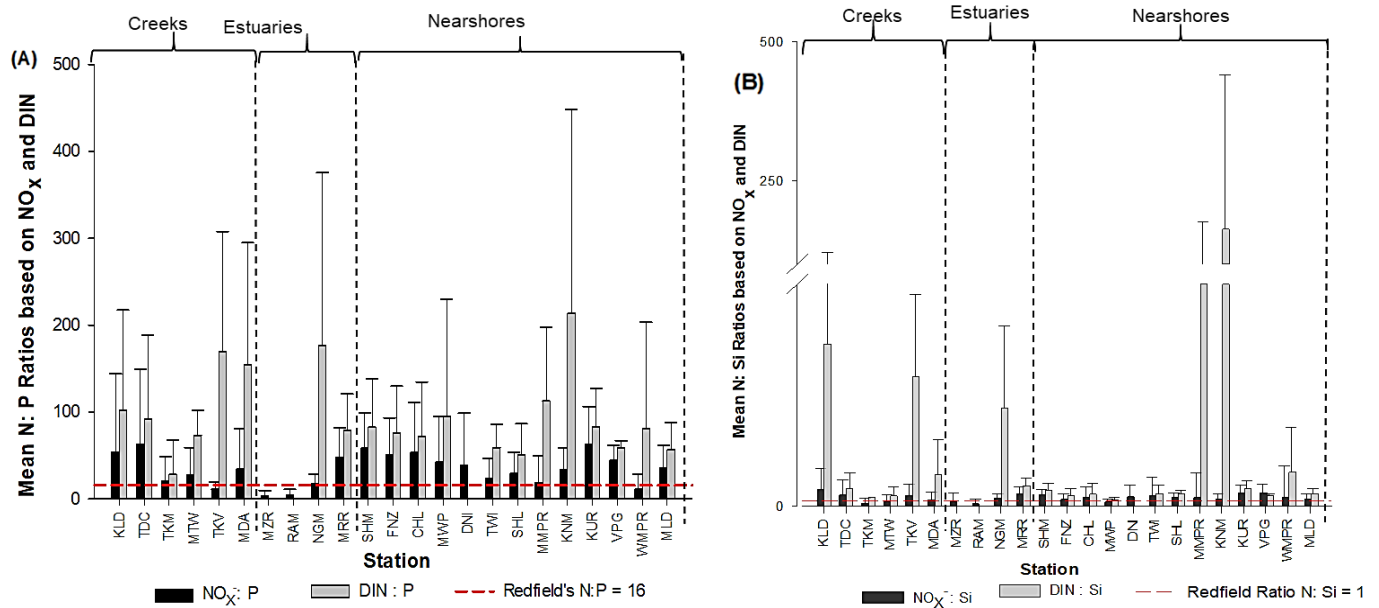


Figure 13: Mean + SDev of Dissolved inorganic nutrient ratios N: P (A) and N: Si (B) based on both NO_x and DIN in sea turtle foraging sites along the Kenya coast recorded over the study period

Similar to N: P, the N: Si ratios varied significantly between the tidal creek and the nearshore waters during the study period. The mean N: Si ratios in the sites ranged from 0.2 - 36.5 and were slightly higher in July, just like ammonium and N: P, with a mean of 6.1 ± 3.6 compared to 4.9 ± 6.8 recorded in September/March. The lowest N:Si ratio of 0.2 was registered in the Coral Garden waters, in WMNPR in July while higher values of 12.5 – 36.6 were recorded in March in the TKV and NGO areas (Figure 13B).

Phytoplankton biomass and community composition

The biomass of phytoplankton (*Chl a*) concentrations ranged between 0.06 and 2.34 $\mu\text{g/L}$ during the study period. Slightly higher *Chl a* were observed in the tidal creeks and estuarine sites compared to nearshore and oceanic sites. Higher *Chl a* concentration of 0.63 - 2.34 $\mu\text{g/L}$ were observed in the RAM estuary compared to other areas. The *Chl a* concentration in the sites located in marine protected areas in MMNPR and WMNPR ranged between 0.06 - 0.8 $\mu\text{g/L}$. Additionally, very low *Chl a* concentrations, with narrow ranges (0.2 - 0.45 $\mu\text{g/L}$) were established in south coast near-shore sites located in TWI, DNI, and CHL (Figure 14A).

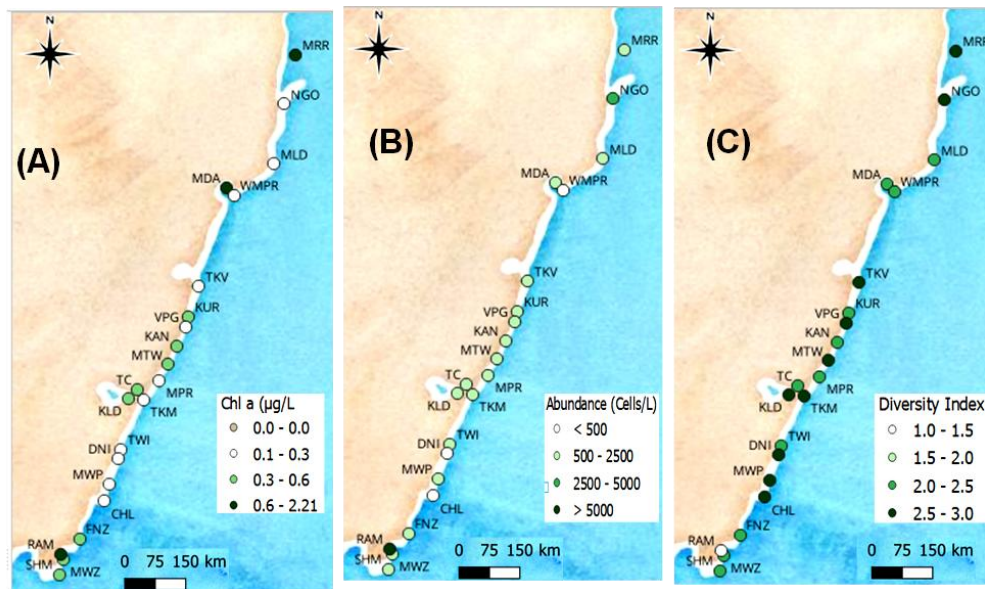


Figure 14: Mean phytoplankton biomass (*Chl a*) abundance and Shannon-Wiener diversity index across sea turtle foraging areas along the Kenya coast over the study period. *Full area names are in [Figure 11](#).

There were also no variations established *Chl a* concentrations between the two sampling surveys with means of $0.46 \pm 0.47 \mu\text{g/L}$ and $0.42 \pm 0.35 \mu\text{g/L}$ recorded for September/March and July, respectively. The phytoplankton abundance was also significantly higher in the estuarine areas compared to other sites with ranges of 113 – 86100 cells/L recorded during the study ([Figure 14B](#)). The abundance was also significantly higher ($P = 0.01$) after the rains registering ranges of 206 - 86100 cells/L in July compared to 113 -13238 cells/L in September/March.

The Shannon-Wiener phytoplankton diversity index ranged from 0.54 to 3.37 bits/cell during the study ([Table 6](#)). A significant spatial variation was established phytoplankton diversity index between the sites located in the tidal creeks and nearshore waters ($P = 0.038$). Both the highest and the lowest phytoplankton diversity index values were registered at the estuarine sites with values 3.05- 3.37 bits/cell registered in NGO in March and values 0.54 - 0.77 bits/cell realized in the RAM estuary after the rains ([Figure 14C](#)). Ranges of 1.71 - 3.37 and 0.54 - 2.90 bits/cell were recorded during September/March and July surveys respectively. The species richness and evenness expressed similar spatial trends to the species diversity index with ranges of 6 – 53, and 0.19 – 0.99 respectively. The species richness was significantly higher ($p < 0.01$) during the dry season in September/March ($n = 140$) compared to July ($n = 69$).

A total of 153 phytoplankton species belonging to 119 genera comprising 90, 35, 14, 5, 4, 3, and 2 diatoms, dinoflagellates, cyanobacteria, chlorophyta, silicoflagellates, flagellate, and euglenophyta

species, respectively, were identified during this study. Significant temporal variations were established in the abundance of cyanobacteria, chlorophyta, and silicoflagellate ($p \leq 0.02$) while the differences in diatoms and dinoflagellates were statistically insignificant ($p > 0.05$). However, the dinoflagellates abundances were slightly higher in July compared to September/March, dominating the phytoplankton composition in the tidal creeks and nearshore areas in MMNPR with 47%, TC (38%), WMNPR (33%), and MTW (23%) (Figure 15).

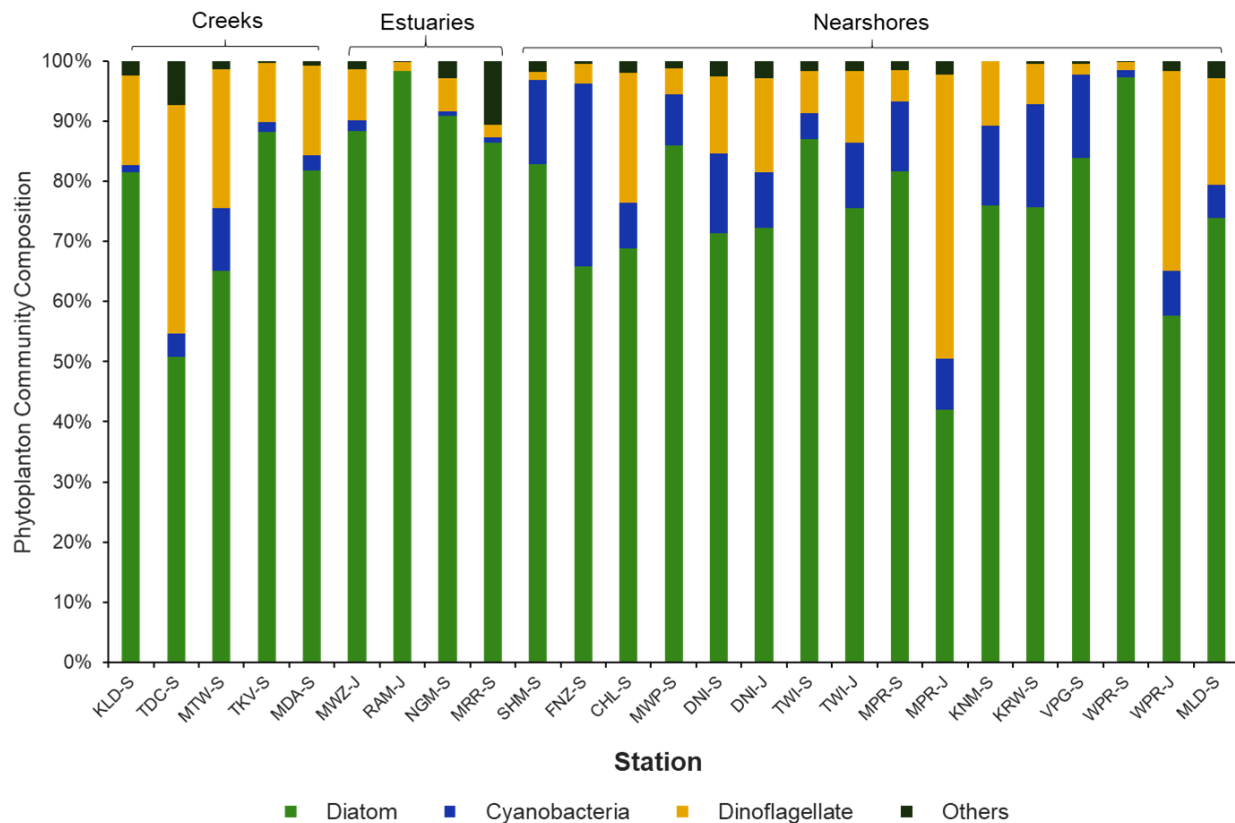


Figure 15: Relative proportion of the phytoplankton community composition based on major phytoplankton groups in the sea turtle foraging sites. *S – September and J- July. Note: MPR – MMNPR; and WPR –WMNPR

We did not establish any specific species that occurred in all 95 sites during the study. However, 32 species were present in 25% of the sites and 11 species were observed in 60% of the sites. No clear spatial cluster was established in phytoplankton species diversity and abundance for the different ecosystem categories using the Bray-Curtis similarity analysis. A dendrogram cluster analysis of species diversity and abundance established $< 5\%$ similarity of the sites indicating that phytoplankton assemblages in the sites were very heterogeneous (Appendix 6).

Overall, 46% of the phytoplankton species composition was made up of harmful algal species. However, the composition of harmful algal species was substantially higher in July compared to September/March with 53 % and 42%, respectively. Substantially higher abundance and diversity of harmful algal species were also observed in the tidal creeks and estuarine areas compared to other areas. The overall mean relative abundance of non-harmful and harmful algae species was 129 ± 409 cells/L and 330 ± 4900 cells/L. Bio-toxin-producing species composed about 17% of the phytoplankton community with a relative abundance of 8 – 828 cells/L and 19 – 75300 cells/L during the September/March and July surveys respectively.

3.4.2. Relationship of phytoplankton indices to the environmental parameters

The multivariate analysis showed that 6 out of the 15 predictor variables explained significant amounts (total of 23.07%) of the variability in phytoplankton community structure (Table 7 and Figure 16). pH was most strongly related (explaining 11.8% of the total variation) to the phytoplankton structure, followed by DO (4.5%) and turbidity (2.9%).

Table 7: Results from a distance-based linear model (DistLM) for all sites sampled over the study period. Variables are listed in order of contribution to explaining variation in the phytoplankton composition. “% variation” represents the explained variation attributable to each variable added to the model.

Variable	F	P-value	% Variation
$R^2 = 23.1\%$; AIC = 934.3			
pH	10.48	< 0.001	11.8
Dissolve Oxygen	6.71	< 0.001	4.45
Turbidity	5.40	< 0.001	2.89
Temperature	4.09	< 0.001	1.94
Conductivity	2.92	< 0.001	1.07
SiO ₃	2.53	< 0.001	0.95

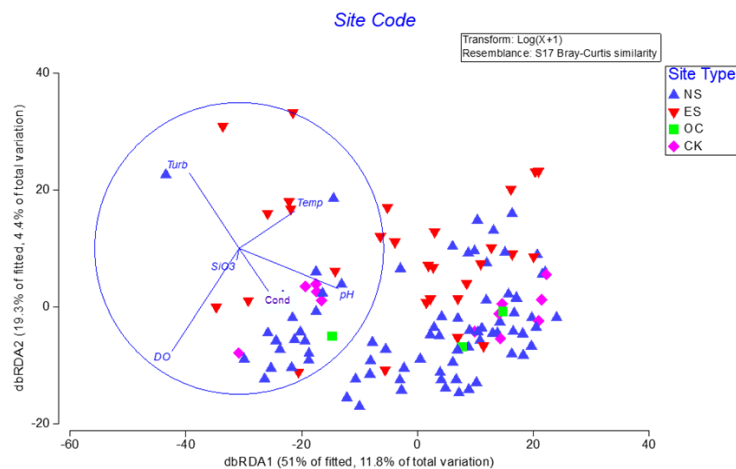


Figure 16: Distance-based redundancy analysis (dbRDA) of phytoplankton samples, overlaid with normalized predictor variables (based on DistLM analysis in Table 8). *NS- Nearshore, ES – Estuarine, OC – Oceanic, and CK – creek

Results from the univariate analysis showed that phytoplankton biomass, richness, and species diversity were highly influenced by a number of the environmental variables considered in this study (Table 8). Increasing Longitude, temperature, and N: Si were significantly associated with a decline in biomass (*Chl a*) while increased latitude and conductivity led to an increase in biomass. Further, increasing salinity and dissolved oxygen were associated with declines in phytoplankton richness. However, the richness increased with an increase in N: Si (Table 8). The association between phytoplankton species diversity and the environmental parameters was only significant with N: Si. The diversity increased with an increasing N: Si. However, an increase in Longitude and dissolved oxygen is significantly associated with an increase in phytoplankton species evenness. The distance-based linear model (DistLM) explained more of the variation in biomass and richness (> 60%) than evenness or diversity (Table 8).

Table 8: Results of multiple linear regression for the relationship of the biodiversity indices to the predictor variables. “S.E” represents the standard error of the coefficient. Bold numbers indicate significant P- values.

Biomass - <i>Chl a</i> ($R^2 = 0.69$)				Richness ($R^2 = 0.63$)			Diversity ($R^2 = 0.34$)			Evenness ($R^2 = 0.45$)		
Variable	Coefficient \pm S.E.	t-value	P-value	Coefficient \pm S.E.	t-value	P-value	Coefficient \pm S.E.	t-value	P-value	Coefficient \pm S.E.	t-value	P-value
Intercept	60.38 \pm 30.9	1.96	0.055	1722.5 \pm 981.2	1.755	0.084	5.55 \pm 50.34	0.11	0.913	-26.18 \pm 12.21	-	0.036
Lat.	0.9 \pm 0.34	2.67	0.009	15.90 \pm 10.66	1.491	0.14	0.03 \pm 0.55	0.061	0.952	-0.24 \pm 0.13	-	0.072
Long.	-1.81 \pm 0.69	-2.61	0.011	-36.67 \pm 22.06	-	0.101	0.12 \pm 1.13	0.106	0.916	0.66 \pm 0.28	2.407	0.019
Temp. °C	-0.18 \pm 0.07	-2.46	0.017	-1.91 \pm 2.37	-	0.423	0.001 \pm 0.12	0.009	0.993	0.03 \pm 0.03	0.901	0.371
Sal.	0.01 \pm 0.02	0.64	0.523	-2.60 \pm 0.74	-	0.001	-0.06 \pm 0.04	-	0.145	0.009 \pm 0.009	1.032	0.306
pH	-0.04 \pm 0.14	-0.26	0.799	-0.02 \pm 4.59	-	0.996	0.23 \pm 0.24	0.992	0.324	0.08 \pm 0.06	1.395	0.168
Turb. (NTU)	0.01 \pm 0.01	1.56	0.95	0.18 \pm 0.19	0.984	0.328	0.01 \pm 0.01	0.525	0.601	0.001 \pm 0.002	0.543	0.589
D.O	-0.03 \pm 0.03	-1.09	0.280	-3.61 \pm 0.98	-	<	-0.05 \pm 0.05	-	0.299	0.04 \pm 0.01	2.904	0.005
Cond.	0.001 \pm 0.0001	3.93	<	-0.002 \pm 0.004	-	0.626	-0.0002 \pm 0.002	-	0.292	-4.8E-05 \pm 5.2E-05	-	0.359
NOx (μ M)	0.01 \pm 0.004	1.39	0.170	0.30 \pm 0.12	2.516	0.014	0.01 \pm 0.01	1.525	0.132	-0.001 \pm 0.001	-	0.495
PO3 (μ M)	-0.19 \pm 0.15	-1.24	0.220	-3.28 \pm 4.86	-	0.502	-0.09 \pm 0.25	-	0.707	0.02 \pm 0.06	0.248	0.805
SiO3 (μ M)	0.01 \pm 0.02	0.75	0.457	0.47 \pm 0.52	0.916	0.363	-0.01 \pm 0.03	-	0.838	-0.007 \pm 0.006	-	0.257
N:P	0.001 \pm 0.001	1.44	0.153	0.01 \pm 0.02	0.558	0.578	-0.0004 \pm 0.001	-	0.734	-0.0002 \pm 0.0003	-	0.423
N: Si	-0.03 \pm 0.01	-3.00	0.004	1.08 \pm 0.31	3.516	0.001	0.04 \pm 0.02	2.541	0.013	0.001 \pm 0.004	0.35	0.727

Spearman correlation analysis revealed significant positive correlations between *Chl a* and both phosphate and silicate ($P \leq 0.001$). A weak but significant relationship was also observed between phytoplankton species abundance and evenness, and silicate concentrations. Based on individual

groups, increasing Cyanobacteria was significantly associated with a decline in ammonium and phosphate concentrations. While dinoflagellates increased significantly with an increase in ammonium and a decrease in silicate concentrations. Flagellates also increased significantly with an increase in phosphate concentrations while increasing ammonium concentrations was associated with a decline in silico-flagellates.

3.4.3. Ecological quality status of the sea turtle foraging sites

The ecological quality of the turtle foraging sites was ‘good’, ‘moderate’, ‘poor’ and ‘bad’ based on the study’s mean *Chl a* and DO, DIP and Shannon-Weiner phytoplankton diversity index, N:Si, and DIN and N:P, respectively. All sites assessed in July, and 90.1% of the sites in September/March were observed with ‘poor’ and ‘bad’ EQS based on DIN (Table 9). Additionally, 3.2 % of the sites were highly polluted based on the Shannon-Weiner diversity index. The overall total score of the Algal Genus Pollution Index in the areas ranged between 11 and 19, indicating ‘moderate’ to ‘probably high’ organic pollution (Table 9). Makonde in CHL was the only site with the highest number of indicators with ‘high’ EQS (n = 4 i.e. DO, DIP, N: Si, and *Chl a*) and ‘moderate’ based on the rest (DIN, N: P and phytoplankton diversity Index) (Appendix 7). The number of sites within the ‘poor’ ecological state based on DIN was significantly higher in July after the long rains compared to the dry September/March period. Most sites with ‘poor’ and ‘bad’ EQS were located in the tidal creeks and estuarine areas. Moreover, a localized ‘bad’ ecological state was established in nearshore areas including MPAs in sites close to hotels in both MMNPR and WMNPR. However, in July, all the sites examined earlier showed an improvement in water quality in terms of DO. Improvements were also established in MMNPR (Mtwapa Mouth, Jumbo Ruins, and Pirates) in terms of DIP and *Chl a* while the EQS in MDA sites (Majaoni and Maji ya Ndani) deteriorated after the rains. There was no change in EQS based on the Shannon-Weiner diversity index (Table 10).

Table 9: Ecological quality status of all the turtle foraging sites based on dissolved oxygen (DO), dissolved inorganic nutrients concentrations (nitrogen-DIN, phosphate-DIP and ratios N:P and N: Si, and Phytoplankton biomass (*Chl a*), Shannon-Weiner diversity index and Algal species Pollution index during September/March and July Months

Indicator	Ecological Quality Status (EQS) Site composition (%)											
	High		Good		Moderate		Poor		Bad		No. of Sites	
	S/M	J	S/M	J	S/M	J	S/M	J	S/M	J	S/M	J
DO	5.6	5.3	11.2	18.4	75.3	73.7	7.9	2.6	0.0	0.0	89	38
DIN	0.0	0.0	0.0	0.0	9.9	0.0	35.2	16.7	54.9	83.3	91	24
DIP	4.3	13.2	34.8	31.6	56.5	47.4	1.1	5.3	3.3	2.6	92	38
N:P	2.4	0.0	2.4	0.0	28.2	16.7	41.2	8.3	25.9	75.0	85	24
N: Si	7.1	0.0	7.1	0.0	48.2	10.0	16.5	15.0	21.2	75.0	85	20
Chl a	39.3	47.4	37.1	26.3	22.5	23.7	1.1	2.6	0.0	0.0	89	38
Shannon-Weiner diversity index			Clean Water		Mod. Pollution		High Pollution				No. of Sites	
			S/M	J	S/M	J	S/M		J		S/M	J
			11.5	0.0	88.5	89.5	0.0		10.5		87	38
Algal species pollution index			Low		Moderate		Prob. High		High		No. of Stations	
			S/M	J	S/M	J	S/M	J	S/M	J	S/M	J
			0	0	100	75	0	25	0	0	19	8

** S/M – September/March, J – July. The Algal Species Pollution Index was done at the Station level.

Table 10: Temporal variations in Ecological quality status in turtle foraging sites

Temporal variations in EQR Status in Sep.(S) and July (J)																
Station	Site	Code	DO		DIN		DIP		N:P		N:Si		Chl a		Diversity	
			Sep	Jul	Sep	Jul	Sep	Jul	Sep	Jul	Sep	Jul	Sep	Jul	Sep	Jul
MDA	Majaoni	MDMJ	M	M	P	B	G	M	P	B	M	B	H	M	MP	MP
MDA	Maji ya ndani	MDMN	M	M	P	B	G	G	P	B	M	B	H	M	MP	MP
MPR	Mtwapa Mouth	M-MT	M	M	P	B	G	H	M	B	G	B	M	H	MP	MP
WPR	Hemmingways	W-HM	M	M	B	P	M	M	M	M	M	P	H	H	MP	MP
MTW	Jumba Ruins	MTJR	P	M	P	-	M	G	P	-	M	-	M	H	MP	MP
MMNPR	Nyali	M-NL	M	G	P	P	G	M	M	M	G	M	G	G	MP	MP
MMNPR	Pirates	M-PR	M	G	B	-	M	H	B	-	B	-	G	H	MP	MP
WMNPR	Richard Bennett	W-RB	M	G	B	P	M	G	-	P	-	B	G	H	MP	MP

***H – High, G – Good, M – Moderate, P – Poor, B – Bad, and MP – Moderate Pollution

3.5.Discussion

The role played by coastal waters in hosting critical marine biodiversity, providing diverse ecosystem services, and supporting local coastal communities' livelihoods is very important in the achievement of sustainable development goals (SDGs) (Landrigan et al., 2020) and addressing the problem of biodiversity loss (Joly, 2022). Assessment and monitoring of the ecological health of coastal waters is thus important for early identification and management of anthropogenic and climatic stresses.

Kenya's coastal waters contain critical coastal habitats including coral reefs and seagrasses, rich marine biodiversity including the critically endangered sea turtles (IUCN, 2005), and supports the livelihood of about 4.3 million people (County Government of Mombasa, 2019). Degradation and especially, eutrophication can thus harm the coastal ecosystems in the area by causing dissolved oxygen depletion, harmful algal blooms development, and loss of critical habitats (Malone & Newton, 2020), and also promotion of infections to existing marine biodiversity (Dujon *et al.*, 2021). All these can affect the normal functioning of the ecosystems and compromise their ability to deliver expected ecosystem services. During the study, we established a high occurrence of P and Si limitation and DIN surplus mainly in the form of ammonium, 'poor', and 'bad' EQS in all the sites in July and 90.1% of the sites in dry season based on DIN. We also established an increase in the number of potential harmful algal blooming species in comparison to those documented in Chapter 2 (Oduor *et al.*, 2023). 'Poor' ecological status established and the increase in the HABs development is a potential threat to the ecosystems and associated ecological and human benefits as discussed below.

3.5.1. Spatial variability in environmental factors and phytoplankton indices

The environmental conditions existing in coastal waters are typically spatially and temporally heterogeneous depending on local physical hydrodynamic processes e.g., (water depth, tidal range, water residence time, sea vivification, upland terrestrial inputs), geomorphological, climatic, and anthropogenic factors (Cartwright & Horn, 2019; Jickells *et al.*, 2017; Li *et al.*, 2023; Tamborski *et al.*, 2015). These driving factors are very diverse along Kenya's coast (ASCLME, 2012; Obura, 2001), and therefore the established spatial variability in physicochemical conditions is not surprising.

Significantly higher turbidity, temperature, and salinity and lower DO were established in the sites located in the tidal creeks and estuaries compared to nearshore and oceanic sites. Tidal creeks and estuaries are widely known for their productivity, which attracts a high human population in the catchment areas, exposing them to anthropogenic threats like high supply of organic matter, nutrients, and sediments (ASCLME, 2012; Kitheka, 1997). These ecosystems are often very shallow, have higher water residence time, and the circulation of seawater is driven by very strong tidal currents compared to neighboring nearshore waters (Kitheka *et al.*, 1996). The high water residence time enhances the accumulation of nutrients and organic matter while their shallow nature facilitates bottom sediment

mixing during tidal changes (Kitheka *et al.*, 1999). High turbidity established in the creeks and estuarine areas can thus be linked to high sediment resuspension during tidal changes, particulate matters discharged from untreated sewage and industrial effluents (Okuku *et al.*, 2011), urban and agricultural run-off (J. Kithiia & Majambo, 2020), and sediments from the river (Tamooch *et al.*, 2012). The seawater temperature is also high due to the high turbidity, which together with the shallow nature of the tidal creeks and estuaries enhances sunlight absorption. We can thus speculate that higher salinities and lower DO observed during the dry season are a result of increased evaporation rate, microbial decomposition of the organic matter, and reduced dissolution of atmospheric oxygen into the seawater caused by the increased temperatures. Nguli *et al.*, (2006) confirmed the higher salinity in the tidal creeks than oceanic waters because of excess evaporation in TC. However, areas around MRR and NGO are characterized by salt pans for salt production. This may have contributed to the substantially higher salinity of 36.2 PSU – 37.7 PSU recorded in the stations during the dry season. In July, the seawater temperature was very low. This occurs due to the relatively cooler water brought from the south by the influence of the East African Coastal Current-EACC. This together with the long rains causes can be associated with the cooling and dilution of seawater causing increased dissolution of atmospheric DO resulting to the higher DO and low salinities. Because the low salinity registered in TKV during the dry season, and the one at the coral garden in July was localized, we can speculate that they were caused by the influx of discharges from ground water. This, however, needs to be verified.

Nutrient concentrations and ratios also portrayed a similar spatial trend to the physical parameters, with significantly higher ammonium, DIN, N: P and N: Si observed in the tidal creeks and estuarine sites. High accumulation of nutrients and organic matter has been reported in the tidal creeks and estuaries, which are released through sediment resuspension during tidal mixing (e.g. Ohowa *et al.*, 1997; Kithika *et al.*, 1996; Mwashote, 1997; Okuku *et al.*, 2011). We established a strong positive relationship between turbidity and silicate and phosphate concentrations that were significantly higher in the nearshore sites, suggesting sediment resuspension and release of the nutrients from pore water during tidal mixing. We also established a negative correlation between DIN and both turbidity and DO, and high ammonium bulk in the DIN, especially in the tidal creeks, estuaries, and localized near-shore sites. This suggests that the DIN was generated by remineralisation processes under high organic matter and low oxygen conditions as reported by Hensen *et al.*, (2006) and Wetzel, (2001). The impact

is more pronounced in creeks and estuaries due to high organic matter input from untreated sewage, industrial effluents, open defecation, and urban runoff (Mwashote, 2005; Okuku *et al.*, 2011; Pole *et al.*, 2016; Kithiia & Majambo, 2020). Additionally, Sabaki and Ramisi river catchments contribute to MRR, NGO, MLD, RAM, and MWZ areas (Ohowa 1996; Ongore *et al.* 2013; Geeraert *et al.* 2015). We also observed the untreated sewage discharged directly into TC waters during the field surveys. Additionally, the tidal creeks and estuaries in Kenya are characterized by mangrove shores, which are globally known for high generation of organic matter and also as nutrient cycling areas (Kristensen *et al.*, 2008). Mutua *et al.*, (2004), also established the importance of benthic fluxes as a source of nutrients in the area as observed during this study in MTW Creek and FNZ Bay.

Higher DIN and phosphate concentrations were also observed in MLD, MRR, and NGO. These areas are located at the Sabaki River estuary, a river that drains its waters from catchment areas characterized by extensive agriculture with intensive use of artificial fertilizers and organic manure (ASCLME, 2012). The observed higher DIN and P in the receiving sea water may thus be a result of fertilizer use in the watershed as reported in other estuarine systems (e.g. Jickells *et al.*, 2017). The phosphate concentrations were also slightly higher in populated residential and tourism areas along TC, SHL, TWI, and Wasini Channels in SHM. From the observations in similar systems in Brazil by Gomes and Wanderley (2016), we can speculate the higher P is from phosphate-containing detergents from domestic sewage and hotels. The observed higher phosphate concentrations in TC were also reported by Okuku *et al.* (2011). Another surprise was that VPG, an area with a low residence population, was established with elevated NO_x^- and silicate concentrations. Whilst this can be speculated to be caused by nutrients seeping into the ground water from leaking septic tanks from tourism hotels and fertilizers used in sisal farms that characterize the area. Our doubts are however cleared by the earlier findings by Mwaura *et al.* (2017). Apart from establishing elevated nitrate concentrations compared to other nearshore sites in MMNPR, the study also observed an increase in the nutrient with a decrease in salinity in VPG, indicating nutrient inputs through groundwater discharges. Ammonium, DIN, and silicate concentrations were also significantly higher during the July surveys. This may have been caused by an increase in the supply of these nutrients from river discharges and surface run-off. Since the primary production was slightly higher, we cannot claim that the increase in these nutrients was due to reduced utilization for primary production by phytoplankton. The significant increase in silicate concentrations after the rains also indicates the dissolution of alumino-silicate minerals during runoff

periods as has been widely documented in the area (e.g. [Kitheka et al., 1999](#); [Ohowa, 1996](#); [Ongore et al., 2013](#)).

The oceanic waters recorded the highest ecosystems' mean NO_x^- during the study. While this may have been contributed by the low number of sites ($n = 4$) sampled, this can be attributed to an influx of nitrogen-rich upwelled offshore waters ([Jebri et al., 2020](#)), high decomposition of macro-algae that characterized the sites, and nitrogen fixation ([Mengesha et al., 1999](#)) as indicated by the widespread nitrogen-fixing cyanobacteria. Elevated DIN in coastal waters during southeast monsoon winds as established in this study were also reported in the neighboring waters in Zanzibar ([Limbu, 2015](#)) and Pemba Channel ([Sekadende et al., 2021](#)). From the above discussion, it is clear that the local hydro-dynamics process, degree of exposure to anthropogenic activities, and primary producers, especially nitrogen-fixing cyanobacteria and seasons had a high influence on the physicochemical parameters at each particular place and time. Analysis of the relationship between nutrients (N, P, and Si) and physical parameters also suggests an origin of nutrients from multiple sources and the importance of benthic fluxes as a source of nutrients in the sea turtle foraging sites during the study.

The spatial and temporal trends in phytoplankton assemblages were almost similar to those of the environmental parameters. Phytoplankton community composition is highly sensitive to environmental changes and their distribution in the aquatic ecosystems is reported to be a result of existing environmental conditions (e.g. [Glibert et al., 2011](#); [IOC-UNESCO, 2008](#); [Lapointe et al., 2015](#)). The spatial distribution of phytoplankton assemblages is discussed in the next paragraphs.

3.5.2. Relationship of phytoplankton indices to the environmental parameters

The multivariate analysis showed that pH, DO, and turbidity were the major environmental factors that contributed to the variability in the sites. The univariate analysis showed that only the influence of DO together with N: Si ratios and salinity was significant and must have influenced the distribution and structure of phytoplankton composition during the study. All these (DO, turbidity, and salinity) can influence due to the role they play in nutrient inputs and water clarity that are important for primary production. The salinity implies freshwater influxes and while turbidity indicates suspended sediments and organic matter that together with D,O links to nutrient supply through remineralisation processes ([Hensen et al., 2006](#)). Higher turbidity and lower DO in the tidal creeks compared to nearshore waters

thus explain why the variations in the Shannon-Wiener phytoplankton diversity index were only significant between these two systems. The species in the creeks are those that can withstand high turbidity and low DO levels. The tidal creeks were also characterized by high ammonium levels, which explains the reason for high dinoflagellates and harmful diatomic species in the creeks and estuaries and the general increase in potential harmful algal blooming species observed in July as later discussed in this study. The selection of pollution (low DO, high turbidity, and ammonium) tolerant species was also demonstrated by high abundance and lower Shannon-Wiener phytoplankton diversity index in July compared to the dry period (1.71 - 3.37 vs 0.54 - 2.90 bits/cell). The phytoplankton richness was significantly higher during the dry season ($n = 140$ vs $n = 69$) due to increased water clarity as established by [Zhang *et al.*, \(2019\)](#). Contrastingly, a high phytoplankton richness and abundance and low *Chl a* concentrations were observed in TKV, and NGO areas during March sampling. This was also accompanied by very low silicate concentrations. March is associated with calm waters ([ASCLME, 2012](#)). The high richness can be attributed to environmental stability and increased light intensity that supported a wide range of species, including those that are less tolerant to turbidity. High species richness and clear waters, however, promote high competition and possible predation ([Wiltshire *et al.*, 2015](#)). This together with a high abundance of low biomass species ([Ghedini *et al.*, 2022](#)), can be linked to the low *Chl a* observed during the study. Additionally, the RAM estuary was established with very low phytoplankton richness and significantly high abundance dominated by *Pseudo-nitzschia*, suggesting a succession ([Wiltshire *et al.*, 2015](#)). It is uncertain if ecological factors like succession, predation, and competition influenced phytoplankton distribution in sea turtle foraging areas. This possibility warrants further investigation.

An increase in longitude and temperature suggests offshore movement to the deeper waters with reduced water column nutrients ([Mengesha *et al.*, 1999](#)), and therefore, the established decline in biomass, richness, and increased evenness is expected. On the other hand, increasing latitude and conductivity simply imply onshore movement, and the positive relationship established with biomass suggests a response to increasing nutrients from terrestrial sources. This response was evident in the RAM estuary and demonstrated in the significant increase in phytoplankton abundance during the long rains. This increase in phytoplankton biomass and abundance as a response to increasing land-based nutrient inputs by river discharges in the area is well documented in the literature ([Kitheka *et al.*, 1996](#);

Mwashote, 1997; Swaleh *et al.*, 2022) and other estuarine systems around the world (e.g. Arhonditsis *et al.*, 2003; Nassar & Gharib, 2014; Sarker *et al.*, 2018; Zhu *et al.*, 2021).

Although nutrients played a major role in the distribution of the phytoplankton assemblages, the only relationship that was significant in biomass production were phosphate and silicate availability. High N explains this: P and N: Si ratios established in about 80% of the sites, implying P and Si limitations. This limitation maybe because of high consumption of P and Si during primary production or lower supply in relation to N from the supplying sources. However, our data points to high consumption rather than supply with the diatoms making the higher composition of the phytoplankton community in the sites. Addition of P and Si is thus expected to cause an increase in primary production. Higher ammonium and phosphate levels were linked to increased *Chl a* and dinoflagellates in tidal creeks, while elevated silicate and ammonium corresponded with greater harmful diatom abundance in estuarine areas, especially RAM (Figure 12, Figure 14A). The connection between high turbidity, ammonium, and phosphate and the development of non-diatom species, especially dinoflagellates in coastal waters is well documented in the literature (Burkholder *et al.*, 2006; Glibert, 2017; Lapointe *et al.*, 2015; Xiao *et al.*, 2018). The dominance of dinoflagellates in the tidal creeks and nearshore areas in July is therefore not surprising. A detailed explanation for this relation is provided in the next section of this study. Cyanobacteria thrived in low ammonium and phosphate conditions due to their nitrogen-fixing capability. This explains their high abundance in nearshore and oceanic waters, which are known to be low in nutrients due to frequent flushing and limited vertical mixing of nutrients in the water column caused by stable thermal stratification as noted by Mengesha *et al.*, (1999). Low nutrients caused by high water exchange and short residence time also explain the low *Chl a* established in nearshore and oceanic sites and the established decline in species richness offshore, which can be speculated to be caused by selective survival of nitrogen-fixing cyanobacteria (Klymiuk *et al.*, 2015).

Comparative analyses in the area have also reported similar findings of a positive correlation between nutrient concentrations and phytoplankton biomass (Okuku *et al.* 2011; Pole *et al.* 2016) and similar coastal ecosystems (Arhonditsis *et al.* 2003; Heil *et al.* 2007; IOC-UNESCO 2008; Glibert 2017; Moreno-Díaz *et al.* 2015). However, this was not true for all the sites located in the creeks. For example, in TC, sites exposed to direct sewage discharges were characterized by high ammonium and phosphate concentrations and very low *Chl a*; this low biomass can be a result of competition and possible predation as demonstrated in the high abundance of dinoflagellates as earlier mentioned. This study

demonstrated that salinity, turbidity, DO, ammonium, phosphate, and silicates were the major physicochemical parameters driving the phytoplankton community composition in the turtle foraging sites during the study. The established dominance of ammonium-nitrogen nutrition for primary production in the area was also reported by (Mengesha *et al.*, 1999). High ammonium and silicates in the estuarine areas promoted the development of harmful diatomic species, while high ammonium and phosphate concentrations in the tidal creeks enhanced dinoflagellates development.

3.5.3. *Ecological quality status and implications in sea turtle foraging sites*

We used five physicochemical and three biological indicators (DO, nutrient concentrations and ratios, and phytoplankton biomass (*Chl a*), Shannon-Wiener diversity (H'), and algal species pollution index) to establish the EQS of the turtle foraging sites using boundaries outlined in Table 5. None of the sites had equal EQS based on all the indicators. However, Makonde in CHL displayed the highest number of indicators with ‘high’ EQS ($n = 4$ i.e. DO, DIP, N: Si, and *Chl a*) and ‘moderate’ based on the rest. We also visually observed coral recruitment and clear waters in the site during the field surveys (Personal field observation), validating that it was in ‘good’ health. The site is less exposed to anthropogenic activities and a higher abundance of nitrogen-fixing cyanobacteria indicates that the DIN was sourced from the atmosphere through nitrogen fixation. The site thus seems to be pristine, and potential for use as a reference site for future ecological quality studies of turtle foraging sites using similar indicators.

Although the sea turtle foraging sites were well-oxygenated (WHO, 2009), we established some localized occurrence of ‘poor’ EQS ($DO < 70\%$) in 20% of the sites located in the creeks and estuarine areas during the dry season. Persistent occurrences of $DO \leq 70\%$ can be harmful to the ecosystem by promoting oxidative stress (Vosloo *et al.* 2013), and enhancing the formation of reduced compounds, like hydrogen sulphide, which can have toxic effects in aquatic animals (Camargo & Alonso, 2006). Besides DO, we also established very low salinity < 28 PSU at the Coral Garden in Mombasa MNPR, although this can be linked to the dilution of the seawater by rainwater, long-term occurrence of such salinities can slow down reef development since most reef-building corals occur within the salinity ranges of 28.7 — 40.4 PSU (Guan *et al.*, 2015).

Based on the nutrient analysis, we established that DIN is a major problem in the area, affecting 90.1% of the sites were in ‘poor’ and ‘bad’ EQS during the dry season with the problem worsening during the long rains suggesting land-based causes (Table 9 and

Table 10 above). The N:P and N:Si ratios also displayed similar trends in EQS, with P and Si limitation established in about 80% of the stations (Figure 13), suggesting either a high supply of DIN in relation to Si and P or higher removal of P and Si in the water column in relation to DIN (Wetzel, 2001). Apart from areas observed with high biomass, it is uncertain what drove the low P and Si in the sea turtle foraging sites during the study. This observation is however in contrast with previous findings of the long-term review in the area (e.g. Oduor *et al.*, 2023), neighbouring waters in Pemba Channel during the same season (e.g. Sekadende *et al.*, 2021), and other tropical oligotrophic waters (e.g. Glibert, 2017; Tyrrell, 1999). Although P limitations have been reported in other tropical coastal areas, especially estuarine waters, we cannot ascertain whether the contrasting results from our study were influenced by the timings of the study or other factors. The cause of the established P and Si limitation in the sea turtle foraging sites is therefore an interesting area that needs to be investigated further.

The productivity of coastal ecosystems highly depends on the quality of essential nutrients N:P and N:Si, with the N:P ratios of most phytoplankton species, reported to range between 5 and 34 (Geider & La Roche, 2002). However, the N: P ratios in the majority of the sites were higher and outside this range. Although the DIN surplus (in comparison to P and Si), occurred in many areas, a nearly complete assimilation of NO_x^- was realized in the RAM estuary and other nearshore sites with values < DL. This suggests that NO_x^- may be an important nutrient that was limiting and ammonium was the main source of N-nutrition for the ongoing primary production in those sites during the study. Substantial evidence shows that persistent P and Si limitation, combined with excessive ammonium-based DIN inputs, poses significant threats to coastal ecosystem health, as discussed earlier in this study. A shift in phytoplankton species composition by such nutrient conditions (DIN contamination/‘bad’ EQS) from diatomic to non-siliceous and harmful dinoflagellates and cyanobacteria is well documented in the literature (e.g. Smayda 1990; Glibert 2017). This was also demonstrated by our findings in TC, MDA Creek, MMNPR, and WMNPR, which were dominated by dinoflagellate and the dominance of harmful diatomic species in the RAM estuary. ‘Poor’ nutrient quality has also been linked with an increase in the production and biomass of macroalgae, affecting DO, light penetration for the benthic biota (Heil *et al.*, 2007), and a switch and disappearance of native macro algae species in turtle

foraging sites as was reported in Brazil ([Santos et al., 2011](#)). We observed the dominance of fleshy algae in many areas, especially sites where turtles are no longer sighted that anglers said were earlier occupied by seagrass ([Personal Communication](#)), which suggests that this problem is already occurring in the area.

The sea turtle foraging sites were very rich in phytoplankton diversity (overall $n = 155$ taxa), which varied significantly among sites suggesting species selection based on existing environmental conditions. Low species richness ($n < 10$) recorded in RAM Mouth, and some sites in WMNPR, and MMNPR indicates that the seawater was under the influence of pollution and thus dominated by a few species that tolerated pollution as demonstrated by [Telesh, \(2004\)](#). The dominance of diatoms in the phytoplankton composition (diversity and abundance) in the area can be due to their tolerance to a wide range of environmental conditions and the availability of silicates needed for primary production. However, the dominance of diatoms is always regarded as less harmful to aquatic ecosystems due to their short and efficient food webs ([Serôdio & Lavaud, 2020](#)). Domination of some species like *Nitzschia* spp. and *Navicula* spp., like those observed in this study, have been associated with pollution and especially eutrophication ([Wetzel, 2001](#)). Pollution-tolerant toxic cyanobacteria *Oscillatoria* spp. and *Nodularia* spp. were also widely spread and significantly higher in oceanic and nearshore sites. This suggests N-limitation and nitrogen nutrition supply through atmospheric N-fixation by these phytoplankton ([Anderson et al., 2002](#)). We also established 7 of 10 of the most pollution-tolerant species, and 19 of 38 pollution-tolerant algal taxa, which resulted in ‘moderate’ to ‘probable high’ pollution status recorded during the study implying that the turtle foraging sites are experiencing organic matter pollution. The ‘probable high’ pollution was established in RAM. River Ramisi discharges contain high amounts of nutrients and organic matter derived from animal waste, decayed vegetation, and run-off from farms ([ASCLME, 2012](#)). This explains the high organic pollution established in the area during the July survey.

Overall, the composition of potential harmful algal blooming species was 46%. The composition was substantially higher in July (53% vs 42%) coinciding with increased P and Si limitation, ammonium-dominated N-nutrition. A shift in dominance of diatomic to harmful dinoflagellates composition was also established in MMNPR, WMNPR, and MDA in July, which supports the claim by [Anderson, \(2008\)](#) that higher ammonium in relation to nitrate nutrition, causes a shift in phytoplankton composition from diatomic to non-diatomic HABs in coastal waters. [Glibert, \(2017\)](#) explains that this shift

occurs because diatoms have a high affinity for nitrates, while other phytoplankton species (dinoflagellates and cyanobacteria) prefer chemically reduced forms of N like ammonium for primary production. A high supply of ammonium can thus enable dinoflagellates and cyanobacteria to outcompete the diatoms and dominate the ecosystems as observed in MMNPR, WMNPR, TC, and MDA creeks. This also explains why 8/10 dominant phytoplankton species were made up of 6 harmful diatom species (*Navicula* spp., *Chaetoceros* spp., *Rhizosolenia* spp., *Thalassionema* spp., *Thalassiosira* spp., and *Nitzschia* spp.), dinoflagellates (*Protoperdinium* spp.) and cyanobacteria (*Oscillatoria* spp.) during the study. The dominant species established in this study are similar to those reported earlier in the area (Oduor *et al.*, 2023; Swaleh *et al.*, 2022) and neighboring waters in Pemba (Sekadende *et al.*, 2021), Dar es Salaam (Hamisi & Mamboya, 2014) and Zanzibar (Limbu & Kyewalyanga, 2015). The blooming of *Pseudo-nitzschia* observed in the RAM estuary was also reported in earlier studies (Mutua *et al.* 2004; Kiteresi *et al.* 2013; Ochieng *et al.* 2015) and the entire Western Indian Ocean region (Hansen *et al.* 2001).

The EQS of areas (MDA, NGO, and MRR) earlier reported with high sea turtle deaths and prevalence of tumors (Jones *et al.*, 2021; Mkare & Katana, 2022) ranged between ‘poor’ and ‘bad’ based on both *Chl a* (Figure 14) and nutrient concentrations and ratios, and ‘moderate’ based on DO, phytoplankton diversity and indicator species. The *Chl a* and nutrient-based ‘poor’ EQS concurs with the claims of Aguirre and Lutz (2004) that FP infections are signals of the ‘poor’ ecological quality in coastal ecosystems. During the dry season in March, we observed very low DO (< 60%), high seawater temperatures (> 28.8 °C), salinities (≥ 36), and ammonium (> 16 μ M) in NGO areas. High temperatures have been linked with the promotion of the disease (Patrício *et al.*, 2012). Additionally, March coincided with the period that high sea turtle mortalities occurred in the MRR-NGO areas in the previous year (Mkare and Katana, 2022), suggesting environmental stress. Apart from the direct cause of environmental stress, high temperatures and salinities can also enhance the development of bio-toxins-producing HABs as established in the sites, which have also been associated with the promotion of the disease (Perrault *et al.*, 2017). High salinity, and low DO as established in the creeks and estuarine areas in NGO, MDA, and TC can promote the mobility of heavy metals from sediments enhancing their ecological risks in the ecosystem (Atkinson *et al.*, 2007; Zhao *et al.*, 2013). Heavy metals have also been linked to tumor development (Aguirre & Lutz, 2004), but this was not examined in the current study. The co-occurrence of ‘poor’ ecological health with areas earlier reported with high turtle

deaths' and FP prevalence supports the widely hypothesized connection between ecosystem health and sea turtle health. This study demonstrated that excessive DIN supply in the form of ammonium, originating from land-based sources as indicated by increases in rains, is a major ecological threat in sea turtle foraging sites. High ammonium together with turbidity and low oxygen can be speculated to be enhancing the development of HABs and contributing to the disappearance of native macro-algae, especially in tidal creeks and estuaries. Additionally, low DO, very high temperature and salinities, and elevated ammonium can be suggested as some of the environmental factors threatening the existence of sea turtles in the area during the dry season as observed in NGO and MRR.

3.6. Management Implications

This study provides information on the locations, magnitude, and variability of ecological quality of the coastal waters that is very important for local coastal managers. It has also demonstrated the possible causes and associated effects of 'poor' ecological health established in some sites, which gives insights to the managers on why action needs to be taken. For example, about 60% of the examined sea turtle foraging sites were located in coral and seagrass habitats ([Unpublished data from the study](#)), which also support local communities' livelihoods through fisheries and tourism ([Musembi *et al.*, 2019](#)). However, 80% of the sites were experiencing DIN-related 'poor' EQS. The origin and effects of the existing nutrient conditions in the foraging sites are also unclear.

We recorded 66 potentially harmful algal blooming species, which represented an additional 8 species of what was reported from a review of long-term data in [Chapter 2 \(Oduor *et al.*, 2023\)](#). The number of biotoxin-producing species in this study was, however, less ($n = 26$ vs $n = 44$ species), but showed an increase in spatial distribution and abundance confirming increased development of HAB species in Kenya's coastal waters. [Kiteresi *et al.* 2013](#) and [Ogongo *et al.* \(2015\)](#) also reported an increasing trend in the development of harmful algal blooming species in the area. These increases in HAB species suggest a continued deterioration of Kenya's coastal waters. We also established the occurrences of bio-toxin-producing species like *Pseudo-nitzschia* spp., *Tripos* spp., *Dinophysis* sp., *Protoperdinium* spp. and *Prorocentrum* spp., *Alexandrium* spp., *Oscillatoria* spp. and, *Trichodesmium* spp., which if not earlier managed can cause potential intoxication of seafood, threatening the health of humans and marine organisms ([Berdalet *et al.*, 2016](#)).

Although we linked increased HABs and fleshy macro-algae in seagrass and coral habitats to DIN pollution, we identified benthic fluxes, agricultural runoff, sewage discharges, and nitrogen-fixation as the main nutrient sources at these sites. These are only speculations and other factors may have influenced the established nutrient conditions and phytoplankton assemblages, especially HABs. The sources, pathways, and ecological effects of the established high DIN are important areas worth investigating if effective management strategies are to be developed.

This is the first ecological quality study that we know of conducted in the turtle foraging sites in Kenya, and the entire Western Indian Ocean-WIO region using both physical-chemical parameters and phytoplankton assemblages. We thus recommend the adoption of the methodology in other areas in the region and the use of our findings as a baseline for comparison with future studies to establish the improvement in the quality of the sites. Additionally, the site established with ‘good’ EQS in CHL Island can be adopted as a reference site for other future ecological quality assessment studies in the area. Our study, however, assumed a bottom-up control of phytoplankton assemblages and presented results on a 2-time survey, with some sites sampled once during the southeast monsoon period that need to be considered when comparing the current results with future findings.

3.7.Conclusion

This is the first attempt to establish the ecological quality of sea turtle foraging sites along the coastal waters in Kenya, and the Western Indian Ocean region by using both physicochemical and biological parameters. We established a heterogenic distribution in environmental parameters and phytoplankton assemblages. This can be speculated to occur due to variations in local hydrodynamics process, degree of exposure to anthropogenic activities, and seasons as documented in the literature. The environmental conditions in shallow areas with long water residence time in the tidal creeks and estuaries were similar and varied significantly with highly flushed sites in the nearshore and oceanic waters. The analysis of the relationship between nutrients and DO and turbidity and the widespread nitrogen-fixing cyanobacteria suggests benthic fluxes and nitrogen fixation as important sources of nutrients in the tidal creeks and estuarine sites, and nearshore and oceanic sites, respectively. Because we conducted only two sampling campaigns during southeast monsoon winds, we cannot make a firm conclusion on the cause of variability of environmental conditions, especially the contrasting finding of P and Si limitation, and phytoplankton assemblages in the turtle foraging sites.

We also established that salinity, turbidity, and DO were the physical parameters that influenced the distribution of phytoplankton assemblages in the sites while the primary production depended on ammonium, phosphate, and silicate nutrients. High turbidity, ammonium, and silicate concentrations in the estuarine areas and ammonium and phosphate concentrations in the tidal creeks co-occurred with a high abundance of harmful diatomic and dinoflagellate species, respectively. This supports the hypothesized connection between increased turbidity, ammonium, and phosphate concentrations and the development of HABs in coastal waters, which is also well established in the literature. However, this study assumed only the bottom-up control of phytoplankton assemblages and did not assess the ecological processes like competition, succession, and predation in the distribution of the phytoplankton composition. This needs to be considered before comparing the results with future studies.

We also established that the ecological quality of the sea turtle foraging sites was ‘poor’ because of an excessive supply of DIN in the ammonium form. This suggests the occurrence of ammonium-driven nitrogen nutrition, silicate, and phosphate-limited primary production in turtle foraging sites, which has been linked with enhancing the development of harmful algal blooms. The increasing harmful algal blooming species development was supported by eight additional species recorded in this study in comparison to the previous review in [Chapter 2 \(Oduor *et al.*, 2023\)](#). Concerning FP infections, the areas with ‘poor’ ecological quality co-occurred with the areas earlier reported with the prevalence of the disease supporting the connection of the disease to the ‘poor’ quality of the ecosystems. The data provided in this study is very important for the local managers since they indicate areas of ecological importance that are in a ‘poor’ ecological state and possible causes and effects. We also established Makonde in CHL with ‘high’ EQS that together with high coral recruitments observed during the field surveys, and location in a less anthropogenic impacted area confirms that it was in ‘good’ ecological health. This makes it a potential reference site for future ecological studies using similar parameters in the area. Additionally, whilst an excessive supply of DIN was evident during the study, we cannot ascertain its origin and channels used to get into the seawater column, which is an important part of the management process. Investigation of the nutrient sources, pathways, and possible effects is thus an area worth consideration in future studies.

Chapter Four: Potentially Harmful Microalgae in Kenya's Coastal Waters



Sampling for analysis of potentially harmful microalgae in Mombasa, Kenya

Diversity and distribution of potentially harmful microalgae in Kenyan Coastal Waters: A systematic survey assessing bloom formation risks

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4.1. Chapter Summary

This study presents the first comprehensive assessment of harmful microalgae along 264 km of Kenya's coastline. Over a yearlong investigation, 59 potentially harmful microalgae species across 40 genera were identified, with 48% of the species capable of producing toxins. Spatial analysis identified tidal creeks and estuaries as hotspots for potential toxin-producing *Alexandrium* spp. and *Pseudo-nitzschia* spp. and high biomass species. Although potential toxin-producers abundances remained below bloom thresholds and no phycotoxins were detected during the survey, their widespread distribution signifies a risk for future harmful algal blooms (HABs), threatening public health, fisheries, and ecosystems. The analysis indicated that temperature, dissolved oxygen, ammonium, and phosphates significantly influence 44% of potential toxin-producing species, suggesting that future changes may promote HAB development in the region. This baseline study indicates the need for identifying drivers of harmful species development, and implementing comprehensive, high-resolution monitoring of HABs to protect Kenya's coastal environments from their impacts.

4.2. Introduction

Marine microalgae play a critical role in aquatic ecosystems as essential components of the food web (Napiórkowska-Krzebietke, 2017). However, harmful algal blooms (HABs) pose significant threats to marine ecosystems, public health, and coastal economies worldwide (Glibert & Burkholder, 2006; Landrigan *et al.*, 2020). These proliferations of toxic or high-biomass algal species can cause mass mortalities of marine organisms, contaminate seafood with phycotoxins, and negatively impact on economic, tourism, recreational and cultural activities (Berdalet *et al.*, 2016, 2022; Glibert & Burkholder, 2006; Lopez *et al.*, 2008; Marshall *et al.*, 2002). The current global increase in HAB events has been linked to factors such as eutrophication, climate change, and anthropogenic disturbances, indicating the need for monitoring and establishment of earlier mitigation strategies (Hallegraeff *et al.*, 2021; Glibert, 2020; Nwankwegu *et al.*, 2019).

Although there is a wealth of literature on HABs in developed countries (e.g. Berdalet *et al.*, 2016; Glibert, 2020; Anderson *et al.*, 2021; Hallegraeff *et al.*, 2021), research on these phenomena in tropical coastal developing regions remains scattered and scarce. Such lack of comprehensive HAB monitoring data poses an increased risk, as tropical developing countries are particularly vulnerable due to existing eutrophic waters, year-round high seawater temperatures favorable for HAB species

proliferation, and high dependence on fisheries and coastal resources (Glibert, 2017; United Nations, 2017). Aside from the studies in Latin America and the Caribbean, global status reports over the past two decades provide virtually no information on harmful algal blooms in African coastal waters (Lan *et al.*, 2024; Hallegraeff, 1993, 1998, 2004, 2021).

The absence of established harmful algal bloom monitoring programs in Africa's coastal waters indicates a significant knowledge gap in addressing this rising global environmental issue. Whereas there are information-sharing networks like the North African Network for Harmful Algal Blooms (HANA) in North Africa (Tarchi *et al.*, 2021), the overall monitoring efforts in the region are limited. The Western Indian Ocean Harmful Algal Blooms (WIOHAB) initiative, active between 1998 and 2000, aimed to conduct baseline surveys and enhance research capacity among member states (Hansen *et al.*, 2001). However, only Mauritius and South Africa maintained the monitoring programs since then (Tamele *et al.*, 2019). In East Africa, including Kenya, the absence of organized HABs monitoring programs results in fragmented data from sporadic studies (Chapter 2; Oduor *et al.*, 2023), further complicating the understanding the dynamics of potential HABs and impacts in the region.

Kenya's coastal and marine waters, covering approximately 230,000 km², as part of the Western Indian Ocean region, support diverse and economically vital marine ecosystems including coral reefs, mangrove forests, and seagrass beds (ASCLME, 2012; Ryabinin, 2020). The waters are vital economic hubs, supporting fisheries, mariculture, tourism, and maritime transport, hosting Kilindini Harbor, which is a crucial deep-water seaport in East Africa (Munguti *et al.*, 2020; Musembi *et al.*, 2019; Nyonje, 2018; Runya *et al.*, 2022). The region's mangroves historically hosted Africa's largest oyster farm, highlighting its economic significance (Munguti *et al.*, 2020; Wanjiru, 2009). Currently, coastal fisheries and tourism sustain livelihoods of around 4.3 million Kenyans, with fisheries significantly contributing to coastal communities' income and the country's fish landings (County Government of Mombasa, 2019; Munguti *et al.*, 2020). Additionally, Kenya is strategically investing in blue economy sectors like fisheries, tourism, and maritime transport to drive economic growth and achieve Sustainable Development Goals (SDGs) (Munguti *et al.*, 2020; Nyonje, 2018; United Nations, 2016), while also considering commercial aquaculture and mariculture targeting high seafood safety standards for the European market that have high seafood safety standards (D. Zhang & Tveterås, 2019). Kenya's coastal environments are, however, facing mounting pressures from population growth, urbanization, agricultural runoff, and climate change impacts like ocean

warming and acidification (Haralalka *et al.*, 2012) . These stressors can potentially exacerbate the occurrence and severity of HABs, posing risks to Kenya's fisheries, tourism industry, and public health.

Although very few previous studies have been conducted in the region, the limited existing research has indicated an increasing trend in occurrence of potentially harmful microalgae (Kiteresi *et al.*, 2013; Ochieng *et al.*, 2015). In Kenya, previous studies have primarily focused on specific locations, toxin groups, or isolated events (Chapter 2; Oduor *et al.*, 2023). A comprehensive understanding of the diversity, distribution, and risks of harmful algal bloom formation in Kenya's coastal waters is crucial for developing effective management strategies to protect coastal ecosystems and economies.

This study provides the first comprehensive assessment of potentially harmful microalgae diversity across Kenya's coastal waters over one year, encompassing various coastal habitats and seasons, enabling targeted monitoring, risk evaluation, and proactive strategies to mitigate harmful algal bloom impacts in the region. Specifically, it employs in situ measurements to assess the diversity and distribution of microalgae, and established association between dominant toxin-producing genera and environmental factors. Additionally, the research evaluates potential ecological, social, and public health implications associated with identified potentially harmful microalgae species, drawing insights from available literature to provide a thorough understanding of the phenomenon.

4.3.Methodology

4.3.1. Study area

The study analyzed spatial patterns of potentially harmful microalgae species in three parts along 264 km (41.3%) of Kenya's coastline, focusing on Mombasa and areas with historical HAB species occurrences. Sampling sites were categorized into four hydrological ecosystems: tidal creeks (tidal channels with saline water and low river influence), estuaries (permanent river's mouths), and near-shore (<500m from shore), and oceanic (≥ 500 m from shore). Localized assessments at 22 stations examined HAB species distribution, complemented by monthly surveys in Mombasa to investigate temporal dynamics. Beach management units (G.O.K, 2012) were used as stations and sub-units as sites.

Mombasa (4°03'S 39°40'E), a peri-urban island and port city off the Western Indian Ocean, covers 229.9 km² (Mohamed *et al.*, 2009). It experiences a monsoon climate with semi-diurnal tides and water depths of 2-30 m (Kimeli *et al.*, 2018). Kilindini (KLD) and Tudor (TDC) Tidal creeks, exposed to maritime traffic and sewage sources respectively, surround the city. With no permanent river, Mombasa faces nutrient pollution and human pressures due to its population of 1.3 million and 2.55% annual growth rate (KNBS, 2019). Mombasa was selected for its diverse marine ecosystems, exposure to anthropogenic pressures (marine traffic, urban pollution, agricultural runoff), and accessible sampling sites (Kithiia & Majambo, 2020; Mohamed *et al.*, 2009; Mwaura *et al.*, 2017). The city represents other emerging coastal cities in the region, making it an ideal location for studying HAB dynamics in urban coastal environments.

4.3.2. Site selection

The study conducted an extensive survey of potentially harmful microalgae species along Kenya's coastline, covering 95 sampling sites within 22 stations (Figure 17A). These included five tidal creeks, three estuaries, thirteen nearshore stations, and four integrated oceanic sites. The oceanic sites were Kisite Mpunguti, Whale Island, and Magic Island in Shimoni (SHM), Watamu Marine Park and Reserve (WMPR) on the south coast, and Malindi (MLD), which were only assessed once per season during the study.

In Mombasa, 14 sampling points were strategically selected to reflect various anthropogenic pressures. Kilindini creek sites (S01-S06) were exposed to maritime transport, while Tudor creek sites (S08-S14) were dominated by diverse sewage inputs. An additional reference site (S07) was included located ~200m offshore from the tidal creeks mouths. Mombasa waters were further divided into three hydrological sections based on proximity to open waters: Oceanic (creek mouths), mid and Upper (inland sites with seasonal river mouths) (Figure 17B). This comprehensive site selection aimed to capture the diversity of coastal environments and potential HAB habitats, allowing for a thorough assessment of species distribution and abundance across different ecosystem types and anthropogenic influence gradients. See Appendix 3 and Appendix 8 for detailed description of the selection criteria and sites.

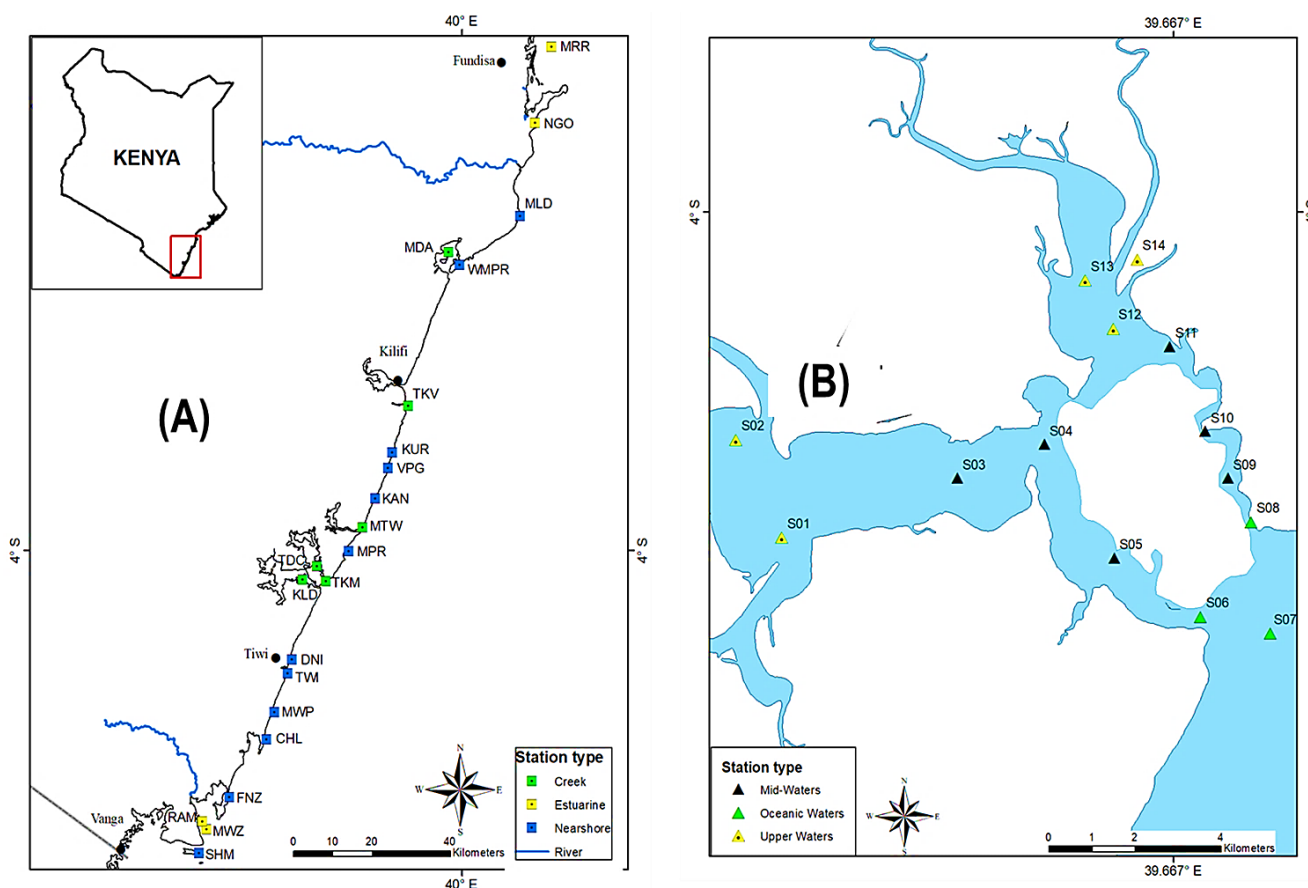


Figure 17: Study area showing study stations (A) and sampling sites in Mombasa (B). Colours distinguish the ecosystem type of the stations (Tidal creeks, Estuarine, and Nearshore), while in the second map, the colours show the water types based on the location in the tidal creeks (Upper, Mid, and Oceanic waters)

4.3.3. Sample collection and analysis

The study conducted monthly surveys of Mombasa's coastal waters from August 2021 to July 2022, with additional surveys in other areas in September 2021 and July 2022 representing dry and wet seasons. Sites that were inaccessible in September were sampled in March 2022. Samples were collected by filtering 20 L of surface seawater through a 20- μ m phytoplankton net, concentrated to 50 mL, and fixed in 5% Lugol solution. Additionally, geographical coordinates and in situ physical parameters data were collected using Kobo Toolbox toolkit with a smartphone GPS toolkit. A total of 463 samples were collected for laboratory analysis.

Identification and quantification of potentially harmful microalgae species were performed using a Sedgewick-Rafter slide and an inverted light microscope, following methods described by Chretiennot-Dinet *et al.*, (1993) and Lundholm *et al.*, (2009) at KMFRI. Taxonomic identification

was based on IOC-UNESCO HAB manuals (Botes, 2003; Chr tiennot-Dinet *et al.*, 1993; Hansen *et al.*, 2001; Lundholm *et al.*, 2009) and acceptance of the names verified using the World Register of Marine Species database and AlgaeBase (<https://www.algaebase.org/>). Recently reported harmful species like *Noctiluca scintillans* and *Tripos* spp. were also considered (Gu *et al.*, 2022; Rolton *et al.*, 2022; Hallegraeff *et al.*, 2021; Karlson *et al.*, 2021; Table 13). Numerical analysis followed APHA (1995) standards, with results expressed as cells per liter (Cells/L).

4.3.4. Data analysis and statistical test

Composition of potentially harmful microalgae

Both univariate and multivariate (site by phytoplankton abundance matrix) analyses were employed to assess harmful microalgae diversity and distribution. Based on Magurran, (2004), univariate measures for phytoplankton indices, (i.e. Margalef's species richness and abundance) (Margalef, 1958) crucial for determining existing species and their blooming status in the study area were utilized: These were defined as follows:

(i) Margalef's species richness (d), defined as the total number of different species found in a sample:

Equation 5:

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

where, S is the number of species and N is the abundance/total number of individuals Margalef, (1958); (ii) abundance (cell density) expressed as the number of cells per liter of seawater calculated using the formula by Stirling (1985):

Equation 6:

$$N = \frac{A \times 1000 \times C}{V \times F \times L}$$

where N = Number of microalgae cells per liter of the original water; A = Total number of phytoplankton counted; C = Volume of the final concentrate of the sample in ml; V = Volume of the field in mm³; F = Number of fields counted; and L = Volume of water filtered in liters (L) (Stirling, 1985).

Distribution of potentially harmful microalgae

The study employed Permutational Multivariate Analysis of Variance (PERMANOVA) to explore spatio-temporal changes in potential harmful microalgal species composition across eco-types, stations, months, and seasons (Clarke & Gorley, 2015). This analysis was based on Bray-Curtis similarity of log-transformed data, which is particularly effective for ecological community data (Bray & Curtis, 1957). The PERMANOVA+ (v1.0.7) software extension to PRIMER-E (v7.0.22) was used for formal testing, providing a robust framework for multivariate analysis (Clarke & Gorley, 2015).

To visualize spatial changes in HAB assemblages and identify hotspot areas, maps were developed using QGIS v. 3.34.4-1-msi. This geographical representation allowed for a clear understanding of the spatial distribution of potentially harmful microalgae species across the study area. Furthermore, to visualize patterns in HAB assemblages across different factors, hybrid multidimensional scaling and dendrograms were employed. Specifically, a semi-strong hybrid MDS with a threshold of 0.9 was used, following the method described by Belbin, (1991). This approach combines the strengths of both multidimensional scaling and hierarchical clustering, providing a comprehensive view of the relationships between potential harmful microalgae assemblages.

To determine associations and contributions to overall variations within the HAB community, non-parametric Spearman's correlation analysis was performed. This analysis focused on species abundances and richness among composition groups, offering insights into the relationships between different potential harmful microalgae species and their relative importance in the ecosystem. Summary statistics and graphs were produced using Sigma Plot version 14.0.3.192 and R version 4.3.3 (R Core Team, 2024).

Determination of risks associated with identified potentially harmful microalgae

We identified potentially harmful microalgal species in Kenyan coastal waters based on IOC-UNESCO criteria, focusing on reported bloom formation and toxin production ability (Botes, 2001; Hansen *et al.*, 2001; Lunholm *et al.*, 2009). The species were classified into three categories: high biomass producers, gill cloggers/physical harm, and toxin producers. Bloom thresholds were established at 10,000 cells/L for physical harm and high biomass producers (Bodeanu, 1992), and at 500 cells/L for toxin producers, as some are known to be toxic even at low concentrations of 500 cells/L (Hansen *et al.*, 2001). Potential toxic effects were evaluated based on the presence of microalgae

taxa linked to seafood toxin syndromes described by Hallegraeff *et al.*, (2021) with modifications (Table 11). These included Paralytic Shellfish Toxins (PST), Diarrheic Shellfish Poisoning (DSP), Amnesic Shellfish Poisoning (ASP), Neurotoxic Shellfish Poisoning (NSP), Ciguatera Fish Poisoning (CFP), Hepatotoxic Shellfish Poisoning (HSP), and Azaspiracid Shellfish Poisoning (AZP). Data on neurotoxins (domoic acid and paralytic shellfish toxins) from Oduor, *et al.*, (2024) were associated with the abundance of potential causative species to assess possible ASP and PSP occurrence threats. Species causing respiratory irritations, water discoloration, and marine aerosol production were also considered (Hallegraeff *et al.*, 2021).

Table 11: Seafood toxic syndromes, toxins and the phytoplankton source of toxins. Adopted from James *et al.*, 2010 with modifications from Lassus *et al.*, (2015), Hallegraeff *et al.*, (2021) and Anderson *et al.*, (2015).

Toxic syndrome	Toxins	Affected sea-food	Toxic algae
Paralytic shellfish poisoning (PSP)	Saxitoxin (STX), neosaxitoxin (NEO), gonyautoxin (GTX) and 18 other analogues	Bivalve shellfish, crustaceans	<i>Alexandrium</i> spp., <i>Gymnodinium</i> spp., <i>Pyrodinium bahamense</i> , <i>Trichodesmium erythraeum</i>
Diarrheic shellfish poisoning (DSP)	Okadaic acid (OA), dinophysistoxins (DTXs), pectenotoxins (PTXs)	Bivalve shellfish	<i>Dinophysis</i> spp., <i>Prorocentrum</i> spp., <i>Phalacroma rotundatum</i> , <i>P. rapa</i> , <i>P. mitra</i> ,
Amnesic shellfish poisoning (ASP)	Domoic acid (DA) and analogues	Bivalve shellfish, finfish	<i>Pseudo-nitzschia</i> spp., <i>Halamphora coffeaeformis</i> , <i>Nitzschia bizertensis</i> , <i>Nitzschia navis-varingica</i>
Neurotoxic shellfish poisoning (NSP)	Brevetoxins (PbTx) Bivalve	Bivalve shellfish	<i>Karenia brevis</i> , <i>K. papilionacea</i>
Ciguatera Fish Poisoning (CFP)	Ciguatoxins (* also maitotoxin)		<i>Gambierdiscus</i> spp., <i>Prorocentrum lima</i> , <i>Amphidinium</i> spp., <i>Coolia</i> spp.
Hepatotoxic Shellfish Poisoning (HSP)	Nodularins	-	<i>Nodularia spumigena</i>
Azaspiracid poisoning (AZP)	Azaspiracids (AZAs) and analogues	Bivalve shellfish	<i>Amphidoma languida</i> , <i>Azadinium poporum</i> , <i>A. spinosum</i> , <i>A. dexteroporum</i> ,
Dermatotoxicity (DTT)	Lyngbiatoxins & antillatoxin	-	<i>Lyngbya majuscula</i>

4.4.Results

4.4.1. Composition of potentially harmful microalgae species in Kenya's coastal waters

Based on the aforementioned IOC-UNESCO criteria, this study identified 59 harmful and potentially harmful microalgae species from 43 genera in Kenya's coastal waters, accounting for 37% of the overall phytoplankton species found in the study area. Among these, 33 species were classified to the species level, while 26 were identified only to the genus level. Notably, 48% of the identified

taxa were potential toxin producers, 35% could generate high biomass leading to anoxic conditions, and 16% posed risks of physical harm through gill clogging. Potential toxin-producing dinoflagellates and cyanobacteria comprised 23% and 13% of the overall species composition, respectively (Figure 18). Potentially harmful diatoms were primarily identified at the genus level, with *Pseudo-nitzschia* spp. and *Nitzschia* spp. noted for potential toxin production. Among the 16 high biomass producers, diatoms and dinoflagellates represented 62.5% and 37.5%, respectively. Additionally, dinoflagellates exhibited the highest diversity among potential gill-clogging species, accounting for 68% (See Table 12 below).

The study revealed diverse cell densities across sites, with *Nitzschia* spp. exhibiting exceptionally high concentrations of 55,650-76,800 cells/L in RAM. Other locations showed lower abundances, ranging from 5 - 12,500 cells/L. The cell densities also varied among potential gill-cloggers, high biomass producers, and toxin producers with maximums of 4450cells/L, 6350cells/L, and 12500cells/, respectively (Table 13 below).

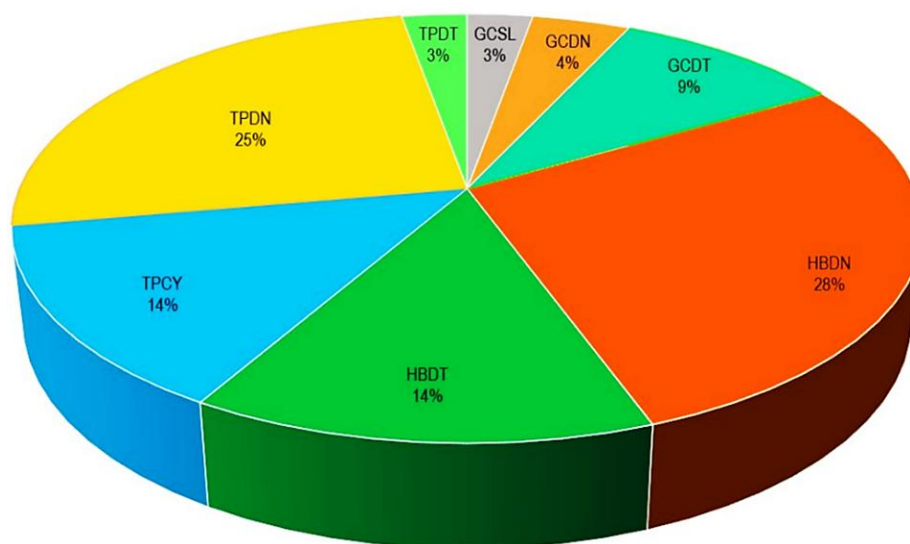


Figure 18: Composition of potentially harmful microalgae species in Kenya's coastal waters (n = 59 species).

*TPDT – toxin producing diatoms, TPDN – Toxin producing dinoflagellates, TPCY – Toxin producing cyanobacteria, GCDT – Gill clogging diatoms, GCDN – gill clogging dinoflagellates, HBDT – high biomass producing diatoms, HBDN – high biomass producing dinoflagellates, GCSL- gill clogging silicoflagellates.

Table 12: List of potentially harmful microalgae and bloom effect in Kenya's coastal waters.

Under the class CYN – Cyanobacteria, DTM – Diatom, DIN – Dinoflagellate, and SIL – Silicoflagellate. Occurrences = no. of observations/samples while under effect TP – Potential Toxin production, PH – Physical Harm, HBP – High Biomass production, GC –Gill clogging, RT – Red Tides. Syndromes ATX – Anatoxin, STX – Saxitoxins, CTX –Ciguatoxins, HSP – Hepatotoxic shellfish poisoning, DA- Domoic acid, ASP – Amnesic shellfish poisoning, PSP –Paralytic Shellfish poisoning, OA – Okadaic acid, DSP – Diarrheic shellfish poisoning, YTX – Yessotoxins, PLTX/OVTV – Palytoxins. Names marked with an asterisk (*) are identified only to the genus level, while those indicated with (X, n = 13) represent species that could belong to the identified genera; therefore, they are not included in the overall reported richness.

	Genus	Name	Class	Mean	Min	Max	Occur.	% Occur.	Effect	Syndrome	Ref/Source
1	<i>Anabaena</i>	<i>Anabaena</i> spp.*	CYN	17 ± 17	8	75	23	5	TP	ATX/HSP	(Pattanaik et al., 2010)
2	<i>Aphanizomenon</i>	* <i>Aphanizomenon</i> spp.	CYN	25 ± 9	19	38	5	1.1	TP	STX, ATX/HSP	(Moustaka-Gouni et al., 2017)
3	<i>Cylindrospermopsis</i>	<i>Cylindrospermopsis curvispora</i>	CYN	31	31	31	1	0.2	TP	Cylindrospermins/HSP	(Moustaka-Gouni et al., 2017)
4	<i>Lyngbya</i>	<i>Lyngbya</i> spp.*	CYN	14 ± 13	5	63	37	8	TP	Lyngbyatoxin	(Lundholm et al., 2009)
5	<i>Nodularia</i>	<i>Nodularia</i> spp.*	CYN	23 ± 8	8	38	20	4.3	TP	Nodularin/HSP	(Lundholm et al., 2009)
6	<i>Oscillatoria</i>	<i>Oscillatoria</i> spp.*	CYN	125 ± 313	8	3615	307	66.3	TP	ATX	(Hansen et al., 2001; Lundholm et al., 2009)
7	<i>Phormidium</i>	<i>Phormidium</i> spp.*	CYN	11 ± 6	8	25	18	3.9	TP	ATX/HSP	(Hansen et al., 2001; Lundholm et al., 2009)
8	<i>Snowella</i>	<i>Snowella</i> spp.*	CYN	20	4	20	3	0.6	TP	Microcystins	(Hansen et al., 2001; Lundholm et al., 2009)
9	<i>Trichodesmium</i>	<i>Trichodesmium erythraeum</i>	CYN	8 ± 5	8	18	3	0.6	TP	STX	(Negri et al., 2004)
10	X	<i>Trichodesmium</i> spp.*	CYN	52 ± 8	8	1332	65	14	TP	STX	(Negri et al., 2004)
11	<i>Cerataulina</i>	<i>Cerataulina pelagica</i>	DTM	263 ± 522	8	4433	393	84.9	PH	GC/Suffocation	(Lorrain et al., 2000; Taylor & Walsby, 1985)
12	X	* <i>Cerataulina</i> spp.*	DTM	144 ± 327	8	3263	299	64.6	PH	GC/Suffocation	(Lorrain et al., 2000; Taylor & Walsby, 1985)
13	<i>Chaetoceros</i>	<i>Chaetoceros</i> spp.*	DTM	263 ± 450	8	4435	393	84.9	PH	GC/Suffocation	(Réveillon et al., 2016)
14	<i>Coscinodiscus</i>	<i>Coscinodiscus</i> spp.*	DTM	7 ± 1	6	8	7	1.5	PH	GC/Suffocation	(Lundholm et al., 2009)
15	<i>Guinardia</i>	<i>Guinardia flaccida</i>	DTM	13 ± 8	8	19	2	0.4	PH	GC/Suffocation	(Lorrain et al., 2000)
16		<i>Guinardia striata</i>	DTM	8	8	8	1	0.2	PH	GC/Suffocation	(Lorrain et al., 2000)
17	X	<i>Guinardia</i> spp.*	DTM	8	8	8	1	0.2	PH	GC/Suffocation	(Lorrain et al., 2000)
18	<i>Leptocylindrus</i>	<i>Leptocylindrus</i> spp.*	DTM	36 ± 130	6	1500	215	46.4	HBP	Anoxia	(Lundholm et al., 2009; Mardones, 2020)
19	<i>Navicula</i>	<i>Navicula</i> spp.*	DTM	69 ± 123	7	1125	396	85.5	HBP	Anoxia	(Lundholm et al., 2009)
20	<i>Nitzschia</i>	<i>Nitzschia</i> spp.*	DTM	113 ± 680	8	1247 3	358	77.3	TP	DA/ASP	(Kotaki et al., 2005; Tan et al., 2016)
21	<i>Pseudo-nitzschia</i>	<i>Pseudo-nitzschia</i> spp.*	DTM	53 ± 138	5	1500	233	50.3	TP	DA/ASP	(Lelong et al., 2012; Lundholm et al., 2009)
22	<i>Rhizosolenia</i>	<i>Rhizosolenia</i> spp.*	DTM	84 ± 354	8	5310	327	70.6	HBP	Anoxia	(IOC-UNESCO, 2001)
23	<i>Skeletonema</i>	<i>Skeletonema costatum</i>	DTM	38 ± 0	38	38	1	0.2	HBP	Anoxia	(IOC-UNESCO, 2001)
24	X	<i>Skeletonema</i> spp.*	DTM	90 ± 164	8	1935	223	48.2	HBP	Anoxia	(IOC-UNESCO, 2001)
25	<i>Synedra</i>	<i>Synedra</i> spp.*	DTM	16 ± 11	8	45	45	9.7	HBP	Anoxia	(Garcés et al., 2001; X. Zhang et al., 2022)
26	<i>Thalassionema</i>	<i>Thalassionema nitzschoides</i>	DTM	38 ± 28	19	150	25	5.4	HBP	Anoxia	(Garcés et al., 2001; X. Zhang et al., 2022)
27	X	<i>Thalassionema</i> spp.*	DTM	82 ± 101	8	803	367	79.3	HBP	Anoxia	(Garcés et al., 2001; X. Zhang et al., 2022)
28	<i>Thalassiosira</i>	<i>Thalassiosira</i> spp.*	DTM	164 ± 168	8	1050	361	78	HBP	Anoxia	(Wang et al., 2021; Lassus et al., 2017)

29	<i>Thalassiothrix</i>	<i>Thalassiothrix</i> spp.*	DTM	20 ± 3	19	25	5	1.1	HBP	Anoxia	(Garcés et al., 2001)
30	<i>Akashiwo</i>	<i>Akashiwo</i> spp.*	DIN	87 ± 338	8	1765	27	5.8	HBP	Anoxia	(Lassus et al., 2017; Lundholm et al., 2009)
31	<i>Alexandrium</i>	<i>Alexandrium</i> spp.*	DIN	31 ± 50	8	595	208	44.9	TP	STX/PSP	(Lassus et al., 2017; Lundholm et al., 2009)
32	<i>Tripes</i>	<i>Tripes azoricus</i>	DIN	53 ± 74	8	750	286	61.8	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
33		<i>Tripes declinatum</i>	DIN	49 ± 230	5	3000	260	56.2	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
34		<i>Tripes extensum</i>	DIN	28 ± 31	8	188	218	47.1	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
35		<i>Tripes furca</i>	DIN	38 ± 46	6	180	174	37.6	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
36		<i>Tripes fusus</i>	DIN	20 ± 24	6	115	8	1.7	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
37		<i>Tripes gibberus</i>	DIN	52 ± 173	8	1333	65	14	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
38		<i>Tripes horridus</i>	DIN	39 ± 52	8	210	39	8.4	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
39		<i>Tripes inflatum</i>	DIN	27 ± 29	8	115	20	4.3	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
40		<i>Tripes lineatum</i>	DIN	10 ± 3	8	15	7	1.5	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
41		<i>Tripes macroceros</i>	DIN	14 ± 23	8	10	6	1.3	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
42		<i>Tripes symmetricum</i>	DIN	44 ± 53	19	189	2	0.4	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
43		<i>Tripes trichoceros</i>	DIN	24 ± 37	8	113	8	1.7	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
44		<i>Tripes muelleri</i>	DIN	10 ± 4	8	15	7	1.5	HBP/R ↓	Anoxia	(Lee et al., 2013; Smayda, 1997)
45	X	<i>Tripes</i> spp.*	DIN	44 ± 46	8	150	31	6.7	HBP/R ↓	Anoxia	(Granéli et al., 1989)
46	<i>Dinophysis</i>	<i>Dinophysis acuminata</i>	DIN	23 ± 18	8	53	9	1.9	TP	OA/DSP	(Lundholm et al., 2009; Suzuki et al., 2009)
47		<i>Dinophysis caudata</i>	DIN	8 ± 0	8	8	1	0.2	TP	OA/DSP	(Lundholm et al., 2009; Suzuki et al., 2009)
48		<i>Dinophysis ovum</i>	DIN	34 ± 24	8	75	9	1.9	TP	OA/DSP	(Lundholm et al., 2009; Suzuki et al., 2009)
49	X	<i>Dinophysis</i> spp.*	DIN	16 ± 14	8	63	29	6.3	TP	OA/DSP	(Lassus et al., 2017; Lundholm et al., 2009)
50	<i>Gambierdiscus</i>	<i>Gambierdiscus</i> spp.	DIN	26 ± 40	8	300	58	12.5	TP	CTX/CFP	(Lassus et al., 2017; Lundholm et al., 2009)
51	<i>Gonyaulax</i>	<i>Gonyaulax</i> spp.*	DIN	40 ± 101	6	660	59	12.7	TP	STX/PSP	(Lassus et al., 2017; Lundholm et al., 2009)
52	<i>Gymnodinium</i>	<i>Gymnodinium</i> spp.*	DIN	15 ± 34	5	255	54	11.7	TP	STX/PSP	(Lassus et al., 2017; Lundholm et al., 2009)
53	<i>Gyrodinium</i>	<i>Gyrodinium</i> spp.*	DIN	10 ± 5	8	19	14	3	HBP	Anoxia	(Lassus et al., 2017; Lundholm et al., 2009)
54	<i>Lingulodinium</i>	<i>Lingulaulax polyedra</i>	DIN	8 ± 0	8	8	2	0.4	TP	YTX	(Lundholm et al., 2009; Paz et al., 2004)
55	X	<i>Lingulodinium</i> spp.*	DIN	8 ± 0	8	8	2	0.4	TP	YTX	(Lundholm et al., 2009; Paz et al., 2004)
56	<i>Noctiluca</i>	<i>Noctiluca scintillans</i>	DIN	6 ± 2	5	8	2	0.4	HBP	Anoxia/NH4	(Gu et al., 2022; Rameshkumar. P et al., 2023)
57	X	<i>Noctiluca</i> spp.*	DIN	11 ± 11	8	60	24	5.2	HBP	Anoxia/NH4	(Gu et al., 2022; Rameshkumar. P et al., 2023)
58	<i>Ostreopsis</i>	<i>Ostreopsis</i> spp.*	DIN	28 ± 33	8	225	67	14.5	TP	PLTX/OVTX	(Lassus et al., 2017; Lundholm et al., 2009)
59	<i>Peridinium</i>	<i>Peridinium quinquecorne</i>	DIN	756 ± 1473	5	6325	29	6.3	HBP	Anoxia	(GEOHAB, 2012; Lundholm et al., 2009)
60	X	<i>Peridinium</i> spp.*	DIN	74 ± 171	7	1855	217	46.9	HBP	Anoxia	(GEOHAB, 2012; Lundholm et al., 2009)
61	<i>Phalacroma</i>	<i>Phalacroma rotundatum</i>	DIN	18 ± 13	8	45	14	3	TP	OA/DSP	(GEOHAB, 2012; Lundholm et al., 2009)
62	X	<i>Phalacroma</i> spp.*	DIN	25 ± 24	8	150	136	29.4	TP	OA/DSP	(GEOHAB, 2012; Lundholm et al., 2009)

63	<i>Prorocentrum</i>	<i>Prorocentrum gracile</i>	DIN	8 ± 0	8	9	3	0.6	TP	OA/DSP	(Lundholm et al., 2009)
64		<i>Prorocentrum lima</i>	DIN	21 ± 50	5	300	35	7.6	TP	OA/DSP	(Lundholm et al., 2009)
65		<i>Prorocentrum micans</i>	DIN	44 ± 61	8	495	199	43	TP	OA/DSP	(Lundholm et al., 2009)
66	X	<i>Prorocentrum spp.*</i>	DIN	55 ± 94	8	525	90	19.4	TP	OA/DSP/YTX	(Lundholm et al., 2009)
67	<i>Protopeiridium</i>	<i>Protopeiridium spp.*</i>	DIN	72 ± 160	8	1650	340	73.4	HBP	Anoxia	(Goswami et al., 2020)
68	<i>Scrippsiella</i>	<i>Scrippsiella trochoidea</i>	DIN	6 ± 0	6	6	1	0.2	PH	GC/Suffocation	(Gustaaf M., 1992)
69		<i>Scrippsiella spp.*</i>	DIN	8 ± 0	8	8	1	0.2	PH	GC/Suffocation	(Gustaaf M., 1992)
70	<i>Dictyocha</i>	<i>Dictyocha fibula</i>	SIL	8 ± 1	8	9	6	1.3	PH	GC/Suffocation	(Garcés et al., 2001; Lundholm et al., 2009)
71	X	<i>Dictyocha spp.*</i>	SIL	8 ± 0	8	8	1	0.2	PH	GC/Suffocation	(Lundholm et al., 2009)

Table 13: Summary statistics of the composition of potentially harmful microalgae in Kenya's coastal waters.

*Effect Composition potential HBP - High Biomass Producers, GC- Gill Cloggers, TP - Toxin Producers; and Potential Shellfish toxin producers abundances AST – Amnesic Shellfish toxins, CFT - Ciguatera Fish toxins, DST - Diarrheic Shellfish toxins, HST – Hepatotoxic shellfish toxins , PST - Paralytic shellfish toxins. **P-Value** represents the significant variations in the HAB-Species composition in the ecosystems.

Category	Overall Study (n =463)				Tidal creeks (n = 258)				Estuarine (n = 104)				Nearshore (n = 97)				Oceanic (n = 8)				P-Value	
	Abundance		Rich		Abundance		Rich		Abundance		Rich		Abundance		Rich		Abundance		Rich			
	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max	Mean	Max		
Effect	Overall	1734 ± 276	82350	16 ± 0.3	37	1342 ± 90	14813	18 ± 0.4	37	3918 ± 1187	82350	16.3 ± 0.6	30	450 ± 28.4	1598	11 ± 0.4	21	1328 ± 825	3788	10 ± 0.7	12	< 0.001
	HBP	542 ± 36	7095	6.4 ± 0.1	14	502 ± 28	3135	7.1 ± 0.1	14	988 ± 135	7095	6.9 ± 0.2	13	193 ± 13.7	833	5 ± 0.2	14	457 ± 298	1350	4 ± 0.3	4	< 0.001
	GC	433 ± 32	5160	4.5 ± 0.1	11	517 ± 42	5160	5.5 ± 0.1	11	606 ± 90	4733	4.3 ± 0.2	9	93 ± 8.9	570	3 ± 0.2	10	90 ± 17	125	3 ± 0.3	4	< 0.001
	TP	761 ± 258	76800	5.2 ± 0.1	13	342 ± 56	12848	5.6 ± 0.2	13	2425 ± 1163	76800	5.6 ± 0.2	11	175 ± 16	1148	4 ± 0.2	10	781 ± 520	2325	3 ± 0.6	5	≤ 0.012
Shellfish Toxicity	AST	604 ± 258	76800	2.1 ± 0.1	7	155 ± 52	12698	2.3 ± 0.1	7	2415 ± 1223	76800	2.6 ± 0.1	5	84.8 ± 10	623	2 ± 0.1	5	773 ± 720	2213	1 ± 0.6	3	< 0.001
	CFT	7 ± 1.1	300	0.3 ± 0.0	2	18 ± 2	56	1.1 ± 0.1	2	37 ± 14	300	1.1 ± 0.04	2	40 ± 6	225	1 ± 0.1	2	19 ± 0	19	1 ± 0.0	1	< 0.001
	DST	41 ± 3.5	581	1.1 ± 0.1	5	63 ± 6	581	1.8 ± 0.1	5	70 ± 9	300	1.8 ± 0.1	5	26 ± 3	75	1.0 ± 0.0	1	438 ± 0	438	1 ± 0.5	1	< 0.001
	HST	86 ± 12	3640	0.9 ± 0.04	4	156 ± 29	3640	1.2 ± 0.03	3	84 ± 25	818	1.3 ± 0.08	4	87 ± 12	525	2 ± 0.1	3	88 ± 27	150	1 ± 0.3	2	< 0.001
	PST	21 ± 2.7	698	0.7 ± 0.04	3	43 ± 7	698	1.4 ± 0.1	1	48 ± 7	319	1.5 ± 0.1	3	10 ± 2	31	1 ± 0.1	2	-	-	-	-	< 0.001
	Others	1 ± 0.2	60	0.1 ± 0.01	1	11 ± 2	60	1.0 ± 0.00	1	8 ± 0.1	8	1.0 ± 0.02	1	-	-	-	-	-	-	-	-	0.002

The most common potential bloom-causing genera were six diatoms (*Navicula*, *Thalassionema*, *Thalassiosira*, *Nitzschia*, *Rhizosolenia*, and *Cerataulina*), two dinoflagellates (*Protoperidinium* and *Triplos*), and the cyanobacterium *Oscillatoria* spp., which were present in 61.8 - 85.5% of the samples (Table 12). At the species level, only *Cerataulina pelagica* occurred in about 85% of the samples, while the rest were observed in $\leq 56\%$ of the samples. *Triplos* was the most diverse dinoflagellate genus (14 species), but only *T. furca* was common (37.4% of samples). Among the potential toxin producing genera, *Dinophysis* showed the highest diversity, while *Alexandrium* was most prevalent, occurring in 45% of samples, albeit in low abundances (mean 31 ± 3.5 cells/L). *Nitzschia* was the dominant and common diatom genus ($\sim 70\%$ of samples), contrasting the potentially toxic *Pseudo-nitzschia* (50.3% of samples, mean 53 ± 138 cells/L). Dominant potential high biomass producers were *Navicula* spp., *Thalassionema* spp., *Thalassiosira* spp., *Protoperidinium* spp., and *Rhizosolenia* spp., while potential gill-cloggers included *Chaetoceros* spp., *Guinardia* spp., *Scrippsiella* spp., *Dictyocha* spp., *Cerataulina* spp., *Triplos* spp., and *Trichodesmium* spp., in that order. Common potential toxin producers were *Nitzschia* spp., *Oscillatoria* spp., *Pseudo-nitzschia* spp., *Alexandrium* spp., *Prorocentrum* spp., *Ostreopsis* spp., *Gonyaulax* spp., *Gambierdiscus* spp., and *Gymnodinium* spp. in that order. Additionally, the study identified 20 rare species, occurring in $\leq 1\%$ of samples. These included nine dinoflagellates, six diatoms, four cyanobacteria, and one silicoflagellate, with a mean density of 20 ± 32 cells/L. Among them, 47% were potential toxin producers, 42% gill-cloggers, and 11% high biomass producers (Table 12).

4.4.2. Spatial Distribution

Significant variations in the distribution of potentially harmful microalgae species were found across the four ecosystems, with less diversity in nearshore waters compared to tidal creeks, estuaries, and oceanic environments ($P < 0.001$). Potentially harmful diatom abundances dominated all the ecosystems, while dinoflagellate richness was significantly lower in oceanic and nearshore areas. Estuaries exhibited lower richness but higher abundances than tidal creeks. Overall richness correlated strongly with the presence of potential harmful dinoflagellates and diatoms in tidal creeks and estuaries, with diatom abundance showing a strong correlation to overall abundance ($r=0.94$; Table 12). In oceanic waters, richness positively associated with potential gill-clogging silicoflagellates and toxin-producing cyanobacteria but negatively with potentially harmful dinoflagellates ($r=-0.89$). Contrastingly, near-shore richness was related more to the potential gill-clogging diatoms and silicoflagellates ($r=0.85$, 0.50).

Species richness of potential toxin producers varied significantly, with 92% found in tidal creeks, while estuarine, nearshore, and oceanic waters hosted 76%, 62%, and 57%, respectively. The potential gill-clogging species richness was also slightly higher in the tidal creeks, while high biomass producers showed consistent richness, higher in estuaries and lower in oceanic waters. The potential gill-cloggers also exhibited slightly higher richness in tidal creeks, while high biomass producers' richness was relatively consistent but higher in estuaries and lower in oceanic waters (Table 13). Notably, the toxin-producing genus *Alexandrium* spp. was prevalent in Mombasa waters, especially Kilindini Creek, throughout the study period. In shallow nearshore waters, *Chaetoceros* spp. was more prevalent, particularly during the dry season, while *Protoperdinium* spp. increased in oceanic waters. *Oscillatoria* spp. was more abundant in nearshore and oceanic areas than in estuarine and creek environments, with rare species primarily found in the tidal creeks.

In the specific stations, higher abundances of potentially harmful microalgae were observed at KLD, TDC, and TKV, and RAM, and NGM, located in tidal creeks and estuaries respectively. However, these estuarine stations exhibited lower species richness as earlier stated. KLD and TKV had the highest richness of potential gill-cloggers and high biomass producers, while nearshore VPG recorded the highest concentration of potential toxin producers (10 species across 8 genera).

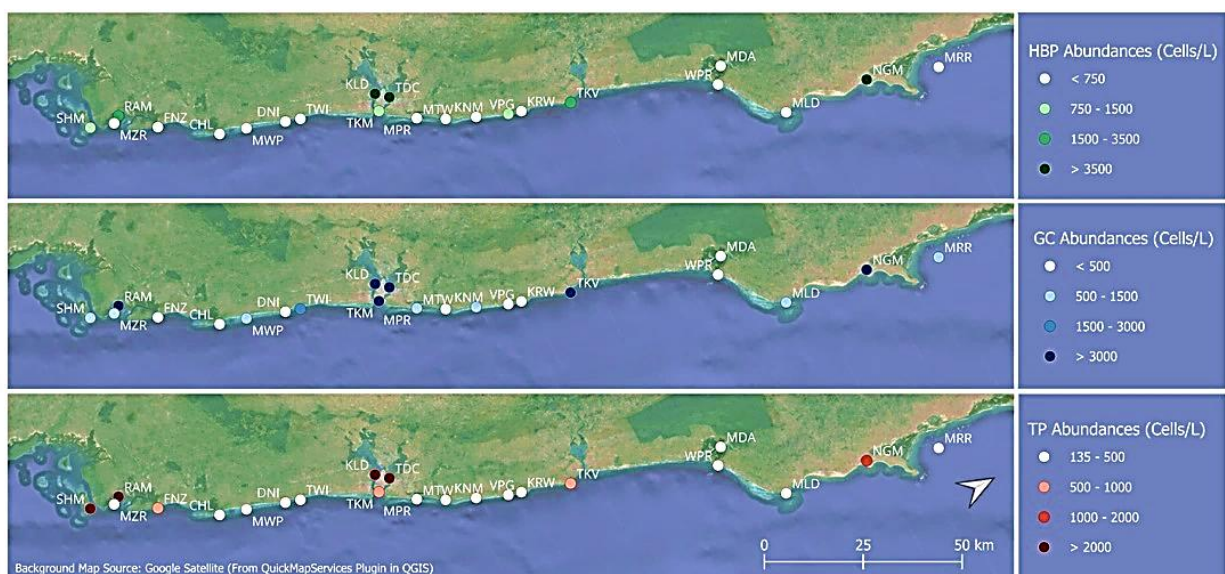


Figure 19: Distribution of the abundances of potentially harmful microalgae species along the Kenya coast.

*Effect categories: HBP - High Biomass Producers, GC - Gill Cloggers, and TP - Toxin Producers

Nearshore stations VPG, WPR, and KNM displayed greater variability and higher toxin producer presence than the others. Elevated abundances of potential toxin producers were also found at RAM river

mouth and nearshore MTW, KNM, FNZ, and SHM stations (Figure 19). Dominant, potential harmful taxa varied across the stations and included *Navicula* spp. (CHL, MWP, VPG, SHM, and MDA), *Nitzschia* spp. (RAM), *Prorocentrum* spp. (MLD), and *Oscillatoria* spp. (FNZ) (Appendix 9).

In Mombasa, monthly surveys over one year identified 58 potentially harmful microalgae species across 40 genera, with 31 taxa classified to species level and 27 to genus level. Significantly, higher abundance and richness were observed in Kilindini Creek, particularly in upstream areas, compared to Tudor Creek and open waters (Figure 20). Potential toxin producers exhibited the greatest diversity at 51% richness, followed by gill-cloggers at 31%. Potentially harmful dinoflagellates also constituted 51% of species richness. Moreover, abundances of potentially harmful cyanobacteria and dinoflagellates were significantly lower in Kilindini Creek and open waters, respectively ($P < 0.05$). The most common taxa encountered were *Nitzschia* spp., *Chaetoceros*, *Guinardia* spp., *Prorocentrum* spp., *Coscinodiscus* spp., *Peridinium* spp., *Protoperidinium* spp., *Rhizosolenia* spp., *Scrippsiella* spp., *Pseudo-nitzschia* spp. and *Oscillatoria* spp. The prevalence of *Alexandrium* spp. was only realized in Kilindini creek, while *Pseudo-nitzschia* spp. showed no spatial variation in occurrence between the two tidal creeks.

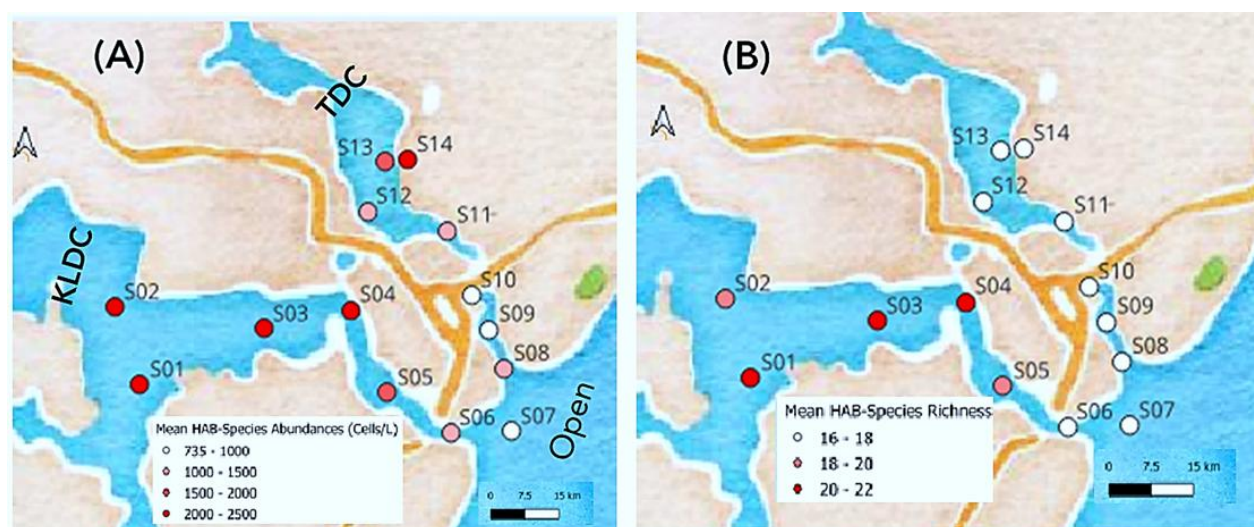


Figure 20: Spatial distribution of potentially harmful microalgae species' abundances (A) and richness (B) in Mombasa coastal waters

4.4.3. Temporal Distribution

Temporal analysis of potentially harmful microalgae in Kenya's coastal waters revealed distinct patterns in abundance and richness across Kilindini Creek, Tudor Creek, and open waters (Figure 21). In

Kilindini Creek, species richness and abundance trends generally aligned, except in February, when increased species richness coincided with a decrease in overall abundance. Peak abundances of potential gill-clogging diatoms and silicoflagellates occurred in February, mirroring overall abundance trends. Across all water bodies, the highest overall abundances were recorded in April, coinciding with peaks in high biomass-producing diatoms and toxin-producing cyanobacteria, and dinoflagellates. In contrast, Tudor Creek exhibited diverse temporal patterns. Mean abundance peaked in April, aligning with trends for potential toxin-producing diatoms, dinoflagellates, and cyanobacteria. Species richness, however, peaked in March, corresponding with slight increases in gill-clogging diatom and silicoflagellate abundances. The lowest abundance and richness of potentially harmful microalgae species were observed in July and August in both Tudor Creek and open waters (Appendix 10).

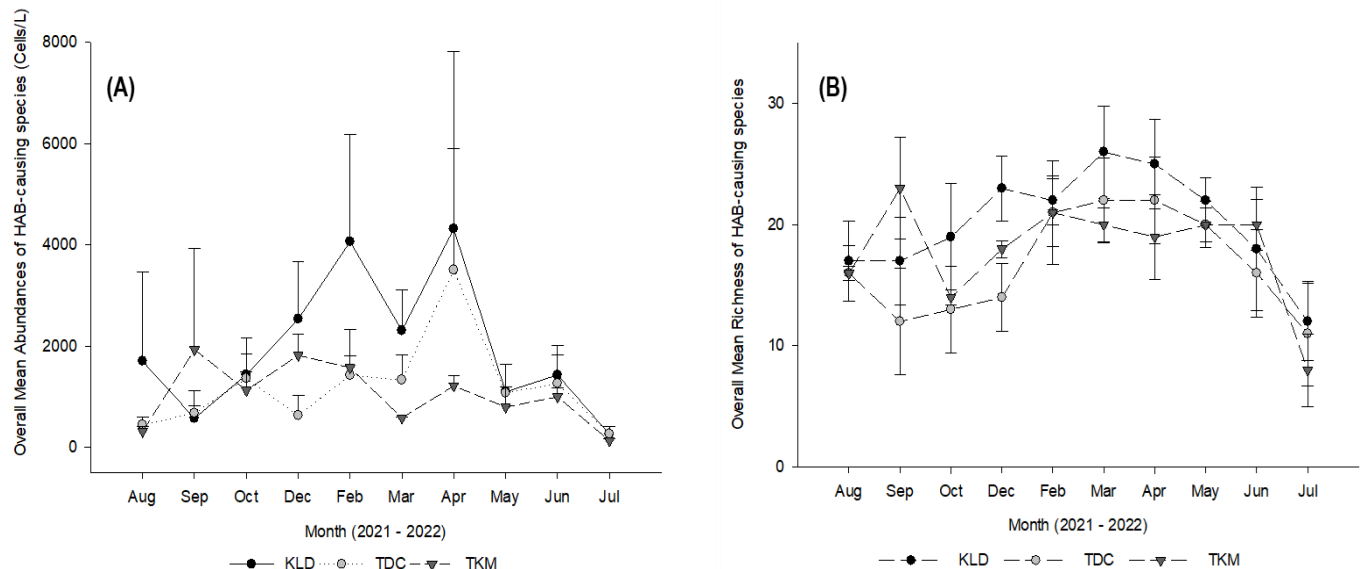


Figure 21: Mean monthly cell abundances and species richness of potentially harmful microalgae species in Mombasa coastal waters. *KLD – Kilindini Creek, TDC – Tudor Creek, and TKM – Tudor- Kilindini Mouth (Open Waters)

The study noted similarities in the overall composition of potentially harmful microalgae between dry and wet seasons; however, species richness was slightly higher during the wet season (Table 14). Significant differences in abundances were observed at the individual species level across seasons and locations. For instance, *Aphanizomenon* spp. and *Lingulodinium* spp. were exclusive to the dry season, while *Snowella* spp. was specific to the wet season. *Noctiluca* spp. and *Peridinium* spp. were more abundant during the wet season, contrasting with *Cerataulina* spp., *Chaetoceros* spp., *Coscinodiscus* spp., *Trichodesmium* spp., and *Guinardia* spp., which peaked in the dry season (Figure 22).

Spatially, *Trichodesmium* spp. was found exclusively in middle Mombasa waters during the dry season, whereas *Tripes* spp. Showed higher abundances in oceanic waters during the dry season and mid/upper waters during the wet season. Among potential toxin producers, only *Gambierdiscus* spp. exhibited seasonal fluctuations, while *Alexandrium* spp. remained consistently present in Kilindini Creek throughout the study period. Other taxa, such as *Cylindrospermopsis* spp., *Lingulodinium* spp., *Phalacroma* spp., and *Ostreopsis* spp., were common in the dry season, while *Nitzschia* spp., *Noctiluca scintillans*, *Gymnodinium* spp., *Alexandrium* spp., *Anabaena* spp., and *Akashiwo* spp. were more abundant during the wet season. Overall, potential toxin-producing species varied between seasons, with significantly higher composition observed during the wet season ($p < 0.001$).

Table 14: Seasonal variation in potentially harmful microalgae in Kenya's coastal waters.

*Effect Composition potential HBP - High Biomass Producers, GC- Gill Cloggers, TP - Toxin Producers; and Potential Shellfish toxin producers abundances AST – Amnesic Shellfish toxins, CFT - Ciguatera Fish toxins, DST - Diarrhetic Shellfish toxins, HST – Hepatotoxic shellfish toxins, PST - Paralytic shellfish toxins.

Category		Dry Season (n = 244)				Wet Season (n = 216)			
		Abundance		Rich		Abundance		Rich	
		Mean	Max	Mean	Max	Mean	Max	Mean	Max
Effect	Overall	1222 ± 84	7560	16 ± 0.4	37	1333 ± 119	14813	16 ± 0.4	30
	HBP	193 ± 14	5918	6.4 ± 0.16	14	626 ± 62	7095	6 ± 0.2	14
	GC	93 ± 9	5160	4.7 ± 0.15	10	369 ± 35	3555	4 ± 0.2	11
	TP	175 ± 16	2325	5.3 ± 0.14	12	385 ± 72	12848	5 ± 0.2	13
Shellfish Toxicity	AST	85 ± 10	2213	2.2 ± 0.07	5	223 ± 67	12698	2 ± 0.1	7
	CFT	40 ± 6	156	0.3 ± 0.03	2	8 ± 2.0	300	0.3 ± 0.0	2
	DST	26 ± 3	581	1.1 ± 0.06	4	34 ± 4.3	495	1.2 ± 0.1	5
	HST	87 ± 12	1669	1.0 ± 0.05	3	92 ± 21	3640	0.8 ± 0.1	4
	PST	10 ± 2	319	0.7 ± 0.05	3	26 ± 5.1	698	0.8 ± 0.1	3
	Others	-	8	0.01 ± 0.01	1	1.3 ± 0.3	60	0.1 ± 0.02	1

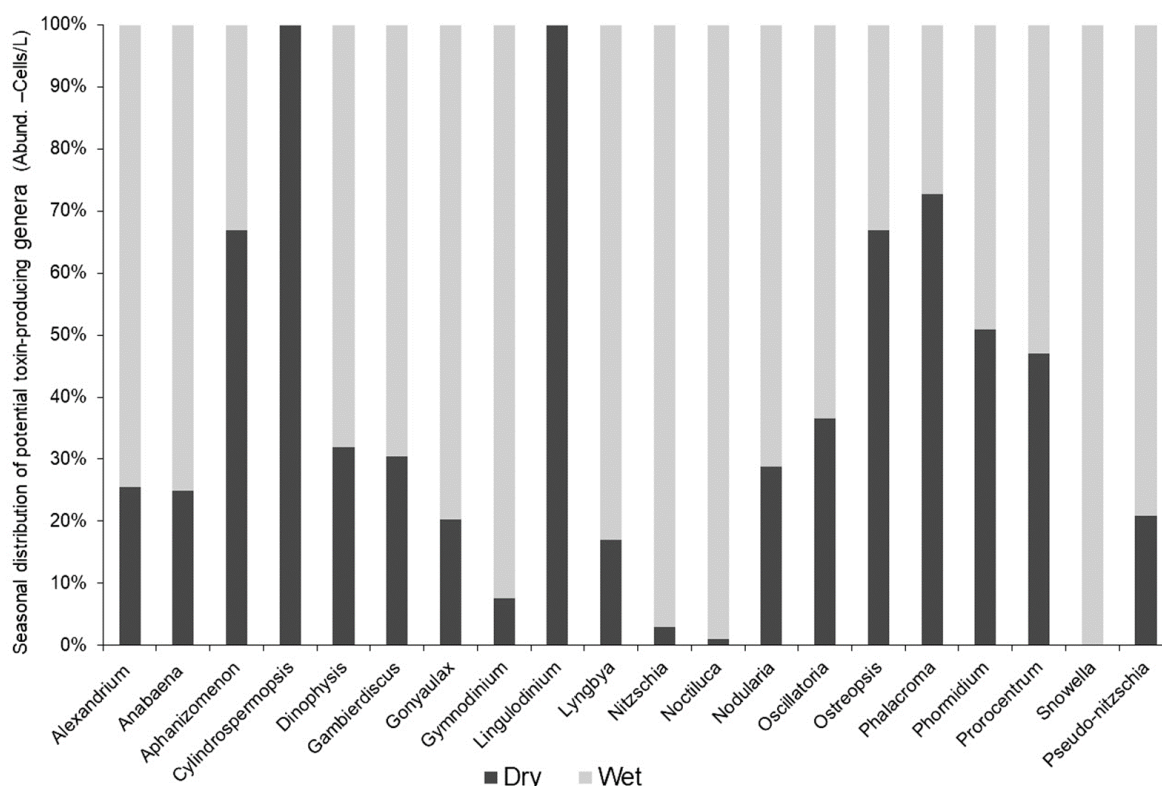


Figure 22: Seasonal Distribution of potentially toxigenic genera in Kenya's coastal waters.

*The percentage represents cell abundance compared to the sum of both seasons.

4.4.4. Potential Impacts of HAB-causing species

The potential gill-clogging species *Chaetoceros* spp. and *Scrippsiella* spp. dominated TKV NGO, MLD, DNI, MWP, and Mombasa waters. *Chaetoceros* spp. was more prevalent in Kilindini Creek in February, while *Scrippsiella* spp. showed significant elevation in Mombasa tidal creeks, particularly in upper Tudor Creek in April. The potential high biomass producers included diatoms, *Navicula* spp. (100-2550 cells/L) and *Rhizosolenia* spp., along with the dinoflagellate *Peridinium quinquecorne*. *Navicula* spp. showed elevated populations in SHM, MDA, MPR, and RAM, while *Rhizosolenia* spp. was notably higher in Mombasa waters (415±43 cells/L). *Thalassiosira* spp. was also significantly higher in Kilindini Creek, and *Peridinium* spp. was elevated in KLD and KNM (225-325 cells/L) compared to other stations (8-95 cells/L).

The number of potential toxin-producing plankton genera was ≤ 5 , with varying abundances across the samples. The cell abundances of potential genera associated with AST and PST were notably

higher in estuarine waters during the wet season. In contrast, DST was significantly elevated during the dry season in oceanic waters compared to tidal creeks; while HST-associated genera, cell abundances were higher in the creeks, with no seasonal differences observed (Table 13). In Mombasa, potential PST, DST, and HST causative genera were elevated in Kilindini and Tudor Tidal creeks. Additional elevated abundances of potential DST and HST causative genera were in MLD and TKM, SHM, and FNZ. However, potential future occurrence hotspots for CFT included coral reef areas in RAM, CHL, WPR, and MTW, while AST hotspots were identified in Kilindini Creek, NGM, RAM, and TKV (Figure 23). The tidal creek water in Mombasa showed potential for PST at specific sites between October and December.

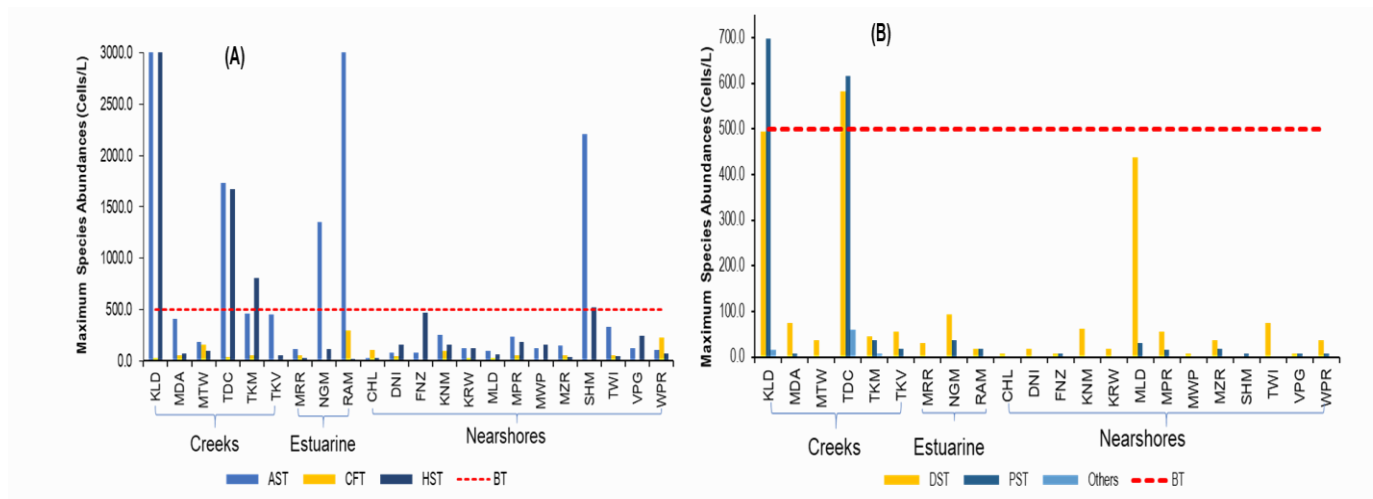


Figure 23: Distribution of seafood toxin-producing species in Kenya's coastal waters.

*AST – Amnesic Shellfish Poisoning Toxins; CFT – Ciguatera Fish Poisoning toxins; HST - Hepatotoxic Shellfish Poisoning toxins; DST – Diarrhetic Shellfish Poisoning toxins; PST – Paralytic Shellfish Poisoning toxins; Others – NH_4 Intoxication and BT – Study's assumed bloom threshold for toxin-producers of 500 cells/L.

4.5. Discussion

4.5.1. Composition of potentially harmful microalgae

This study revealed a significant presence of potentially harmful microalgae in Kenya's coastal waters, indicating the region's vulnerability to future harmful algal blooms (HABs) and associated risks. We identified 66 toxic and potentially harmful species (Figure 18), suggesting that a threat to seafood safety and public health exists in Kenya's coastal waters. Dinoflagellates dominated harmful microalgae communities in Kenya's coastal waters, albeit in low abundances. Prevalence of dinoflagellates in HAB communities aligns with global trends, where this phytoplankton group dominates the harmful

bloom events (Hallegraeff *et al.*, 2021; Glibert *et al.*, 2018). Dinoflagellates contribute up to 38% of marine food poisoning reported worldwide (Lassus *et al.*, 2017; Moore *et al.*, 2008). This is attributed to their adaptability in eutrophic environments and their ability to heterotrophic nutrient (Berdalet *et al.*, 2016; Hallegraeff *et al.*, 2021). However, diatoms dominated in abundance due to their rapid growth and affinity for high concentrations of dissolved inorganic nutrients (Glibert *et al.*, 2018). Additionally, diatoms thrive in mixed water columns to avoid sinking, while dinoflagellates prefer stratified waters, leading to variations in their composition, especially in coastal embayments with high freshwater input (Glibert, 2017). Ammonium-dominated eutrophic conditions are known to favor the development of potentially toxin-producing dinoflagellates and diatoms (Glibert *et al.*, 2018; Hillebrand *et al.*, 2014). This is supported by the prevalence of *Pseudo-nitzschia* spp. Found, which are known to thrive in environments rich in reduced nitrogen forms (Kudela *et al.*, 2008). This study identified previously unreported potentially harmful species in the area, including *Triplos furca*, *T. gibberus*, and *Cylindrospermopsis curvispora*, significantly expanding the known diversity compared to previous reports in Chapter 2 (Oduor *et al.*, 2023). However, using a 20-µm phytoplankton net likely underestimated species richness by excluding smaller organisms. The actual count of potentially harmful microalgae may be higher than reported, as many harmful dinoflagellates and haptophytes smaller than 20 µM were not captured in the samples.

4.5.2. Distribution of potentially harmful microalgae in Kenya's coastal waters

The distribution of potentially harmful microalgae in Kenya's coastal waters varied significantly across different ecosystems. This distribution pattern reflects the interaction of environmental factors, including nutrients, water stability, light, residence time, and grazing pressure, along with species-specific habitat preferences (Anderson *et al.*, 2002; Glibert, 2017; Heisler *et al.*, 2008). higher presence in closed systems indicates significant terrestrial influences, including nutrient imbalances and eutrophication (Wang *et al.*, 2021; Malone and Newton, 2020; Glibert, 2017). Conversely, potentially harmful microalgae composition was lower in oceanic and nearshore waters compared to tidal creeks and estuaries. This difference stems from frequent tidal flushing, water mixing, and short residence times in open systems, creating nutrient-poor environments with limited microalgal development (Anderson *et al.*, 2002).

The diatoms and silicoflagellates were relatively more abundant in oceanic and nearshore waters, likely due to silicates suspended from sediments during tidal mixing, which favored their growth compared to other taxonomic classes (Heisler *et al.*, 2008; Tantanasarit *et al.*, 2013). Oceanic waters exhibited positive correlations between overall abundance and gill-clogging silicoflagellates, toxin-producing diatoms, and cyanobacteria, but a negative correlation with dinoflagellates. This suggests favorable conditions for these classes, while heterotrophic dinoflagellates may control their populations as reported in similar studies (Glibert *et al.*, 2018; Wiltshire *et al.*, 2015). However, the nearshore waters exhibited strong associations between cyanobacteria and toxin-producers, as well as diatoms and gill-cloggers, which indicates potential risks of the development of toxin-producing cyanobacteria and gill-clogging diatoms blooms. Additionally, the prevalence of potential toxin-producing cyanobacteria (e.g., *Oscillatoria* spp.) and toxic diatoms (e.g., *Pseudo-nitzschia* spp. and *Nitzschia* spp.) in both oceanic and nearshore waters indicates potential hotspots for bloom formation and associated ecological impacts (Hyytiäinen *et al.*, 2021). Despite comparable hydrodynamic and nutrient profiles described in Chapter 3 and other studies in the region (Kyewalyanga and Lugomela, 2001; Oduor *et al.*, 2024), *Nitzschia* spp. was more prevalent in oceanic waters (89%) than nearshore waters (56%). This difference may be due to the benthic nature of most *Nitzschia* species (Lassus *et al.*, 2017), though further investigation is needed.

Closed systems, like tidal creeks and estuaries, receive nutrient inputs from diverse terrestrial and oceanic sources, making them nutrient-rich environments capable of supporting higher microalgae biomass (Righetti *et al.*, 2020). The dominance of potentially harmful dinoflagellates and diatoms in these areas can be linked to the high nutrient levels, water stability, long water residence times, which allow for microalgae growth and accumulation (Huete-Ortega *et al.*, 2010). In tidal creeks, overall species richness strongly correlated with abundance, suggesting that the increasing number of cells were as a result of new emerging species (Siano & Montresor, 2005). The stronger association between overall species richness and potential toxin-producing diatoms and high-biomass producing dinoflagellates in the estuaries suggests hotspots for these harmful algal outbreaks.

In Mombasa, dinoflagellates comprised 51% of species richness, likely due to their ability to thrive in eutrophic waters, as previously discussed. The composition of potentially harmful microalgae was also significantly higher in Kilindini Creek compared to Tudor Creek and open waters (Figure 20). Kilindini Creek's higher diversity may be due to longer water residence time, exposure to port operations, ballast water, and agricultural runoff (Adala *et al.*, 2007; Kitheka, 2000). These factors provide

diverse nutrient sources and potentially introduce cysts of toxic dinoflagellates, such as *Alexandrium* spp. (Smayda, 2007), which was prevalent throughout the study.

Temporal variations in abundance and diversity of potentially harmful microalgae occurred across all waters, reflecting changes in environmental conditions due to seasonal variations and both natural and terrestrial/anthropogenic influences. Potentially harmful dinoflagellates and cyanobacteria were more prevalent during dry conditions in February and March, characterized by high temperatures and low nutrients as shown in Chapter 3 (Oduor *et al.*, 2024; Obura, 2001). These conditions favor their development over diatoms due to their adaptability through mixotrophy, heterotrophy, and nitrogen fixation (Glibert *et al.*, 2006; Glibert & Burkholder, 2011; Heil *et al.*, 2007). However, high abundances of gill-clogging *Chaetoceros* spp. and *Dictyocha* spp. in Kilindini Creek during February's dry season suggested silicate availability that favored their production or reduced grazing pressure (Wiltshire *et al.*, 2015). Similar temporal trends in Mombasa's waters from March to July indicate that seasonal environmental changes during the long rainy season influence the ecological composition and functions of Mombasa coastal waters. The potentially harmful microalgae exhibited peak abundances across all waters in April, coinciding with the onset of long rains (Appendix 11). This timing suggests that land-based nutrient influx promotes the proliferation of high-biomass, toxin-producing cyanobacteria, dinoflagellates, and diatoms (Figure 21/Appendix 10). The initial rains, characterized by increased turbidity, ammonium, and silica concentrations (Chapter 5), likely favored potentially toxic diatoms and high-biomass dinoflagellates in Tudor Creek and open waters (Glibert *et al.*, 2018; Heisler *et al.*, 2008). This environmental shift altered the composition from toxic dinoflagellates and cyanobacteria to diatoms, high-biomass dinoflagellates, and silicoflagellates, demonstrating the influence of environmental factors on species composition and coastal ecosystem productivity (Lu *et al.*, 2022; Nishimura *et al.*, 2020).

4.5.3. Association between potential toxin producers and environmental factors

Significant spatial and seasonal variations in the distribution of potentially toxin-producing genera indicate that environmental factors, seasonal changes, and spatial characteristics collectively influence harmful microalgae abundance in Kenya's coastal waters. Redundancy Analysis revealed correlations between taxonomic composition and environmental parameters, explaining 37% of community variations (adjusted R-squared: 0.29). A refined model accounted for 23.8% of the variance (adjusted R-squared: 0.214).

The variance partitioning showed that ecosystem type (5%), seasonal changes (1%), and environmental variables (11%), with pH and NH_4^+ each contributing 4%, collectively explained the observed variation. These findings align with global observations regarding harmful algal bloom species' environmental preferences (Glibert *et al.*, 2018; Heil *et al.*, 2007; Heisler *et al.*, 2008; Telesh, 2004). The positive correlation between *Alexandrium* spp. and ammonium aligns with findings from Ogongo *et al.*, (2015) in Kilindini Creek and (Bouchouicha Smida *et al.*, 2012) in Tunisian lagoon waters, indicating a potential risk of *Alexandrium* blooms with persistent ammonium enrichment.

The potential risks of harmful algal blooms (HABs) in Kenya's coastal waters are significant, despite overall abundances remaining below established bloom thresholds (Bodeanu, 1992; Smayda, 1997). This study detected the presence of potential ecological impacts by identifying and mapping the distribution of microalgae species associated with seafood toxin syndromes and other harmful effects. The presence of higher *Nitzschia* spp. cell densities in Kilindini Creek and Ramisi Estuary, coupled with a decrease in species richness in Ramisi Estuary. This change may lead to reduced microalgal diversity and could affect the coastal food web (Berdalet *et al.*, 2016). Additionally, rise of heterotrophic dinoflagellates e.g. members of *Protoperidinium* spp. in oceanic waters indicates predation competition with zooplankton, which are crucial for fisheries productivity (Jeong *et al.*, 2010; Siano & Montresor, 2005). Some heterotrophs species like *Stephanopyxis* spp., which occurred among rare species and is linked to reduced egg production in copepods, potentially affecting fisheries (Poulet *et al.*, 2006). Furthermore, increased abundances of gill-cloggers such as *Chaetoceros* spp. in Kilindini Creek may threaten future fisheries production, necessitating investigations into their causes to protect local fisheries.

While most potential toxin producers were identified up to the genus level, their toxic effects are attributed to the presence of known toxin-producing species in the region (e.g. Ochieng *et al.*, 2015; Lugomela, 2006, 2013; Hansen *et al.*, 2001; Kyewalyanga and Lugomela, 2001). The presence of toxin-producing genera like *Dinophysis*, *Alexandrium*, *Pyrodinium*, *Gambierdiscus*, *Nitzschia*, and *Pseudo-nitzschia* poses risks of shellfish contamination, even at low cell concentrations. This threatens artisanal fisheries that coastal communities heavily rely on in the area (Musembi *et al.*, 2019). Significantly, potentially toxic dinoflagellates accounted for 25% of the observed species richness, with these organisms being linked to 38% of food poisoning cases worldwide (Lassus *et al.*, 2015). *Alexandrium* spp. and *Pyrodinium* spp. are particularly concerning due to their links to potentially fatal paralytic

shellfish poisoning (Eisnor & Wiegand, 2024). The spread of *Gambierdiscus* spp. during the dry season in Mombasa's upper creek and nearshore waters raises risks of ciguatera fish poisoning for reef fishing and tourism activities. Additionally, the presence of *Gymnodinium* spp., prevalent during the wet season in Kilindini Creek—especially during December's tourism peak—poses threats of neurotoxic shellfish poisoning and skin irritations for anglers and recreational users. However, heterotrophic dinoflagellates, such as *Phalacroma* spp. and *Protoperidinium* spp., previously disputed as toxigenic, derive their toxicity from grazing on smaller toxic species (Krock *et al.*, 2009); thus, act as vectors and their harmful effects depend on the availability of these prey in the ecosystem. A parallel study in July found no domoic acid or paralytic shellfish toxins at the sites (Oduor *et al.*, 2024), despite the presence of potential toxin-producing species. This absence may be due to low species abundances and suboptimal assessment timing, as these species typically peak in May with the onset of the rainy season (Kiteresi *et al.*, 2012; Kiteresi *et al.*, 2013; Ochieng *et al.*, 2015).

Positive correlations between temperature, ammonium levels, and harmful algal blooms were identified in this study, aligning with Ogongo *et al.*, (2015) in Mombasa. Ocean warming and ammonium dominated silicate and phosphate limited eutrophic waters, along with high turbidity and acidification significantly influence the occurrence of harmful algal species and enhance the development and toxicity of potential toxin producers like *Alexandrium*, *Pseudo-nitzschia*, and *Dinophysis* spp. (Glibert *et al.*, 2018; Likumahua *et al.*, 2022; Xiao *et al.*, 2019). Future changes in these conditions may thus lead to increased proliferation of toxin-producing dinoflagellates and heightened toxicity, as observed in similar ecosystems (Glibert *et al.*, 2018; IOC-UNESCO, 2008). While this study identified several notable potentially toxin-producing microalgae species throughout Kenya's coastal waters at low abundances below bloom thresholds—indicating no immediate contamination risks—the spatial spread of species like *Alexandrium* spp. and *Pseudo-nitzschia* spp. suggests potential for future blooms under favorable environmental conditions. Additionally, the study focused primarily on larger microalgae species, but did not assess smaller phototrophic groups (< 20µm) that may produce highly toxic compounds (Krock *et al.*, 2009), suggesting that the composition of the potential toxic species, maybe more than reported. Proactive monitoring of potentially harmful microalgae is thus essential for timely risk identification and management due to the ecological and economic importance of Kenya's coastal waters.

4.6. Conclusion

This study provides the first comprehensive assessment of potentially harmful microalgae in Kenya's coastal waters, identifying 59 taxa across 43 genera, with 48% classified as potential toxin producers (microplankton mentioned as producing or carrying toxins produced by others as listed in [Table 12](#) above) besides high biomass producers and gill cloggers. Dinoflagellates dominated the diversity of potentially harmful microalgae, with toxin-producing species comprising 23% of the total composition, consistent with global trends that report a higher prevalence of dinoflagellate-associated harmful algal bloom events ([Lassus et al., 2015](#)). The distribution of potentially harmful algal bloom species varies significantly across ecosystems, with tidal creeks exhibiting the highest diversity and abundance, while nearshore waters show lower richness. Additionally, specific taxa such as *Alexandrium* spp. and *Chaetoceros* spp. demonstrate distinct spatial preferences, emphasizing the ecological variability in harmful microalgae distribution. *Alexandrium* spp. was notably abundant in Kilindini Creek, likely due to ballast water ([Smayda, 2007](#)). The dry season showed lower species abundances with prevalent harmful diatoms, while potentially harmful dinoflagellates and toxin producers, indicating terrestrial influence, dominated the wet season.

Although abundances of potentially harmful microalgae remained below bloom thresholds, the widespread presence of toxin producers like *Alexandrium* spp., *Pseudo-nitzschia* spp., and *Gambierdiscus* spp. poses risks for future outbreaks and seafood contamination, threatening public health. The study showed the association and possible influence of temperature, ammonium, and dissolved oxygen on potentially harmful microalgae occurrences across diverse ecosystems in the study area. We have also identified potential hotspots for these risks, necessitating continuous monitoring and proactive management strategies. Future research should extend to the northern Kenya Banks, where harmful algal blooms have been reported, and focus on smaller microalgae species and precise species-level identification of toxic groups, as not all species under the reported genera are harmful. Additionally, understanding the factors influencing potentially harmful microalgae occurrence in the area is essential, given the weak observed correlations between environmental variables and species. While Kenya's coastal waters harbor numerous potentially harmful algal species at currently low densities, their established associations with environmental parameters suggest that future changes could trigger toxic blooms, posing ecological, social, and public health risks.

Chapter Five: Nutrient sources and pathways in the coastal waters of Mombasa, Kenya

Distribution and driving factors of nutrient and organic matter dynamics in Mombasa, Kenya

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5.1.Chapter Summary

This study investigated the distribution, sources, and pathways of anthropogenic nutrients and organic matter in the coastal waters of Mombasa, Kenya's second-largest city and a highly urbanized island port. Utilizing dissolved inorganic nutrient measurements, isotopic signatures ($\delta^{13}\text{C}$ and $\delta^{15}\text{N}$), and radon, the research revealed significant spatiotemporal variations in nutrient levels. Tudor tidal creek exhibited substantially higher nutrient concentrations than surrounding areas, primarily due to sewage discharges, wastewater, and surface runoff. Seasonal variations revealed significant nutrient concentration increases during wet periods, highlighting the role of surface runoff in terrestrial nutrient supply. Isotopic analysis revealed that nutrients primarily originated from terrestrial wastewater, with hospital sewage identified as a major contributor in Tudor tidal creek, and atmospheric nitrogen as primary nutrient sources nearshore areas. Additionally, submarine groundwater discharge emerged as a significant pathway for delivering dissolved inorganic nitrogen, phosphate, and silicate to Mombasa's coastal waters. Mombasa's ecological quality status was '*Bad*' for mean DIN and N: P and N: Si ratios, while DIP was '*Poor*' in Tudor tidal creek and '*Moderate*' in Kilindini tidal creek and adjacent oceanic waters. This study highlights that coastal cities in developing regions face significant pollution risks from inadequate wastewater treatment and sanitation, with sewage entering the ocean

via direct discharges, surface runoff, and submarine groundwater. Given the area's ecological importance, we recommend improved wastewater management and adaptive conservation strategies that account for local and seasonal dynamics to mitigate nutrient pollution and preserve the ecosystem's health. This research provides a foundation for future studies on nutrient dynamics in rapidly urbanizing coastal regions.

5.2.Introduction

Coastal and marine ecosystems are vital harboring rich biodiversity and provide numerous ecosystem services, supporting local economies through fisheries, tourism, and maritime transport (GEF-8, 2023; Malone & Newton, 2020; Masselink et al., 2021; Ocean & Coasts, 2024). However, these ecosystems face growing threats from excessive organic matter and nutrient (nitrogen and phosphorus) inputs from anthropogenic sources like sewage, and industrial discharges, and agricultural runoff (Malone & Newton, 2020; Saito *et al.*, 2018). As at now, about 70% of untreated wastewater discharge generated from coastal regions goes into the oceans globally (Ocean & Coasts, 2024). Anthropogenic activities introduce an imbalanced supply of nutrients to coastal waters, with excess nitrogen and phosphorus but low silicates, intensifying eutrophication in coastal ecosystems. This can result in harmful algal blooms, hypoxia, biodiversity loss, and ecosystem degradation, severely impacting coastal productivity and services (Glibert, 2020; Malone & Newton, 2020; Nwankwegu *et al.*, 2019).

Nutrient pollution is currently affecting nearly 245,000 km² of coastal and marine ecosystems globally, with 34% of coastlines at high risk of degradation due to coastal development (GEF-8, 2023; Ocean & Coasts, 2024). This problem has been intensified by the global population growth, which has caused a tenfold rise in environmental nitrogen over the past century (Saito *et al.*, 2018). Oceanic nitrogen and phosphorus enrichment have dramatically increased, with nitrogen nearly doubling and phosphorus tripling over the past century (Millennium Ecosystem Assessment, 2005; Saito *et al.*, 2018). Additionally, reduced forms of nitrogen such as ammonium and urea fosters harmful algal growth, particularly in eutrophic coastal areas with limited phosphorus and silicates, significantly affecting marine ecosystems (Glibert, 2020; Glibert *et al.*, 2018; Heil *et al.*, 2007). Coastal zones naturally filter pollutants, including nutrients and organic matter, from terrestrial sources (Caballero-Alfonso *et al.*, 2015). However, developed shorelines and restricted lagoons with prolonged water retention are particularly vulnerable to chronic nutrient loading (Davis *et al.*, 2009). Shallow coastal ecosystems, such as estuaries and bays, are typically eutrophic due to tidal fluctuations, freshwater

inflow, and diverse nutrient inputs. These sources include agriculture, sewage, atmospheric deposition, and submarine groundwater discharge (SGD) (Adyasari *et al.*, 2018; Jickells *et al.*, 2017).

The Western Indian Ocean (WIO) region is highly susceptible to nutrient pollution due to inadequate wastewater treatment in major coastal cities, resulting in direct discharge of sewage and industrial effluents into coastal waters (Hamisi & Mamboya, 2014; Kithiia & Majambo, 2020; Okuku *et al.*, 2011). While localized reports indicate elevated nutrient levels and reduced water quality in critical habitats in Chapter 2 (Oduor *et al.*, 2023), quantitative assessments of organic matter and nutrient fluxes in Kenyan coastal waters remain limited. The identification and relative contribution of nutrient sources are largely unknown. Recent meta-analyses have highlighted the scarcity of data on coastal biogeochemistry and groundwater in Kenya and Africa as a whole (Davis *et al.*, 2021; Richardson *et al.*, 2024; Wilson *et al.*, 2024), indicating the need for comprehensive studies in this region.

Understanding anthropogenic nutrient and organic matter dynamics is crucial for protecting coastal ecosystems and developing effective environmental management strategies. Stable isotope analysis of nitrogen and carbon in benthic macro algae reveals nutrient sources, pathways, and utilization by primary producers, providing insights into the origin and reactivity of nutrients in coastal ecosystems (Orlandi *et al.*, 2014; Sarma *et al.*, 2020). Additionally, radon (^{222}Rn), a uranium decay product, serves as an effective tracer for submarine groundwater discharge (SGD) due to its short half-life, non-reactivity, and high groundwater concentration (Dulaiova *et al.*, 2008). It quantifies both fresh and saline SGD, revealing important pathways for nutrient input and carbon export in coastal waters (Knapp *et al.*, 2020; Sospedra *et al.*, 2018). These techniques are essential for establishing the sources, pathways, and utilization of nutrients by primary producers, which is necessary for developing ecosystem-based management strategies to mitigate pollution-related issues (Wurtsbaugh *et al.*, 2019). These approaches are potentially applicable in the present study of coastal nutrient dynamics.

This study examined the distribution, sources, and pathways of macronutrients (N, P and Si) and organic matter around Mombasa, a highly urbanized and industrialized coastal area to provide a foundation for developing ecosystem-based management strategies for mitigating pollution and protecting coastal ecosystems in the region. Specifically, this study: (1) evaluated the spatiotemporal patterns and seasonal variations of dissolved inorganic nutrients and particulate organic matter in Mombasa's coastal waters; (2) examined potential sources contributing to macronutrient enrichment in the Mombasa coastal ecosystem; and (3) evaluated the potential pathways for nutrient transport and distribution within these coastal waters, including SGD.

5.3. Methodology

5.3.1. Study Area

This study was conducted in the coastal waters of Mombasa, a peri-urban island and port city located in the Western Indian Ocean ($3^{\circ}80'-4^{\circ}10'S$, $39^{\circ}60'-39^{\circ}80'E$; [Figure 25](#)), with a land area of 14.7 km^2 ([Mohamed *et al.*, 2009](#)). The region experiences a hot tropical climate influenced by monsoon winds, with temperatures ranging from $19-34^{\circ}\text{C}$ and a mean annual rainfall of $1,144 \text{ mm}$. The climate is characterized by semi-annual inter-tropical convergence zone passages, resulting in short rains during the northeast monsoon (November - March) and long rains during the southeast monsoon (April – October). Mombasa experiences semi-diurnal tides, with spring tidal amplitudes reaching up to 4.0 m ([Nguli *et al.*, 2006](#)) and water depths range from $2-30 \text{ m}$ from shallow upper tidal creek waters to deeper oceanic waters at the tidal creek mouths ([Kimeli *et al.*, 2018](#)).

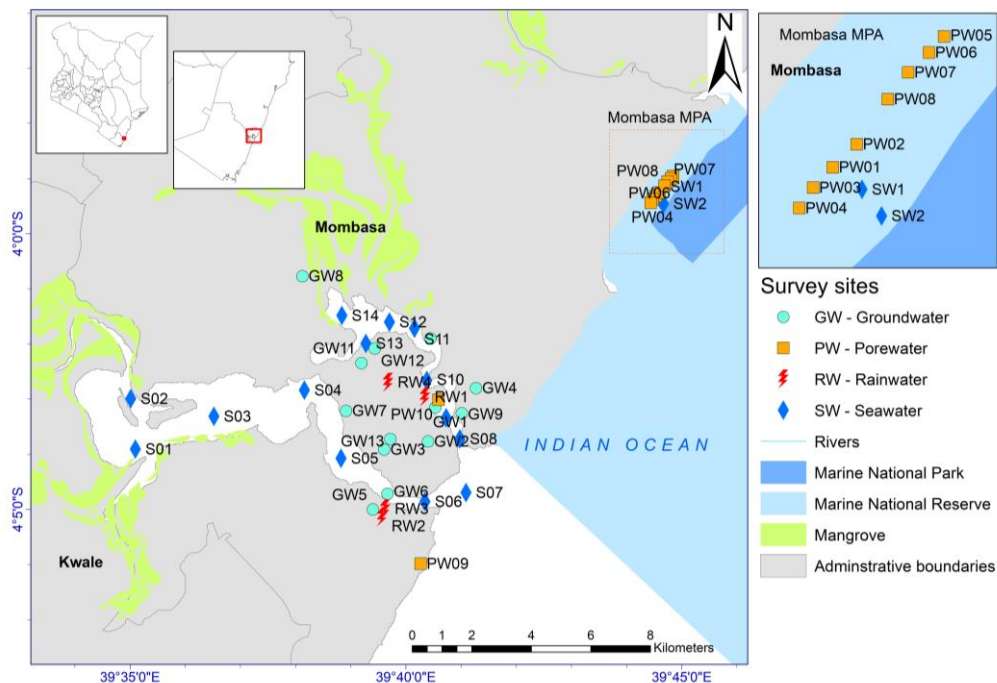


Figure 25: Map of Mombasa City and sampling sites. Location of Mombasa on Kenya's south-eastern coast (Top left). Creeks and sampling locations (Bottom) with colours and symbols representing different water compartments i.e., Groundwater, pore water, rainwater and seawater.

5.3.2. Site Selection

A total of 14 sampling sites were selected to represent different anthropogenic nutrient sources and environmental conditions in Mombasa's coastal waters. We chose these sites strategically based on proximity to potential nutrient sources, such as sewage outfalls, industrial effluents, and riverine inputs, as well as their accessibility during low tides. The sampling sites were distributed across the Tudor and Kilindini tidal creeks, which contain diverse ecological habitats, including saline and freshwater marshlands, seagrass beds, mangrove forests, coral reef ecosystems, and intermittent riverine systems (Adala *et al.*, 2007).

Tudor tidal creek, which is approximately 20 km long, is subjected to urban and industrial activities, while Kilindini tidal creek is primarily impacted by port activities and shipping operations. Sites S01 – S06 are located in Kilindini tidal creek while S08 – S14 are located in Tudor tidal creek. Additional site S07, located ~100m offshore from creek mouths, was included to assess the oceanic influence. Furthermore, the seawater in the tidal creeks were divided into three hydrological sections namely oceanic-waters (tidal creek mouths, S06 and S08), mid-waters (between oceanic and upper), and upper-waters (inland sites with seasonal river mouths, S01-S02, S12-S14; Appendix 8).

5.3.3. Sample Collection and Analysis

Dissolved Inorganic Nutrients

A total of 430 samples were collected, including surface water (340), groundwater from wells (62), pore water (20), and rainwater (8). From August 2021 to July 2022, surface water sampling was conducted monthly at 14 sites by collecting 500 ml of surface seawater from a depth of 50 cm during low tides. Groundwater samples were obtained from 24 wells in September 2021 (during the dry season) and July 2022 (rainy season). Additionally, pore water samples were collected at three beaches in Mombasa Marine Park and Reserve (MMPR, PW01-PW08), Shelly Beach (PW09), and Madobini (PW10) using a Retractable-Tip piezometer system (, PW09, PW10), with each bore purged three times for 2-3 minutes to eliminate stagnant groundwater (Beck *et al.*, 2008). Rainwater samples were collected from four urban locations in Mombasa, including busy traffic areas (RW1 and RW2) and crowded residential areas (RW3 and RW4; Figure 25) during September-October 2021. The samples were collected in clean, elevated plastic containers, capturing approximately 500mL of the initial raindrops. Seawater samples were collected in triplicates, while all other samples were collected in duplicates. The seawater and pore water samples were filtered immediately in the field while well

water and rainwater samples were collected in pre-rinsed 500 mL bottles, stored in cooler boxes with ice, and filtered in the laboratory within 24 hours after collection. All the samples were filtered using disposable syringe filters (Sartorius Minisart ® - 0.45 µM pore size), 25 mL aliquots of the filtrates were filled into pre-rinsed polyethylene bottles preserved with a mercury chloride solution (0.06 mL HgCl₂-solution/10 mL sample), and stored frozen. All samples were transported to Germany on dry ice and stored at -20°C in the laboratory until the time of analysis for nutrient.

In the laboratory, nutrient concentrations were analyzed using a TECAN microplate reader with colorimetric methods with triplicate instrument precision of 0.4, 1.8, 0.9, and 0.2 µM for NO_x⁻, NH₄⁺, PO₄³⁻, and SiO₃²⁻, respectively). These included spectrophotometric determination of nitrate and nitrite (NO_x⁻) (García-Robledo *et al.*, 2014), salicylate method for ammonium (NH₄⁺), PhosVer3 ascorbic acid method for orthophosphate (PO₄³⁻), and silicomolybdate method for silicate (SiO₃²⁻) (Ringuet *et al.*, 2011). Detection limits were 0.2 µM for NO_x⁻, 0.1 µM for PO₄³⁻, and 0.1 µM for NH₄⁺, determined according to ISO 11843-2 and German standard DIN 32645. All sampling and analytical procedures followed HELCOM (2014) and Grasshoff *et al.* (2007) guidelines.

Organic Matter Measurements

The organic matter (OM), including organic carbon (C_{org}) and nitrogen (N_{org}), was quantified in seston, macro algae, and sediments to evaluate nitrogen sources. Seston samples were obtained by filtering 1L of surface seawater through pre-combusted 0.7 µM Whatman GF/F filters, which were frozen until analysis. These filters were later vacuum-dried at 60°C for 24 hours and subsampled for organic carbon and nitrogen analysis. Sediment cores were collected from three sites at depths under 30 cm due to obstacles posed by rocky shores, sub-sampled at 5 cm intervals, freeze-dried for transport, and then dried at 60°C for 48 hours. The sediment samples were grinded and homogenized using a ball mill before organic carbon and nitrogen analysis.

Macroalgae – Isotopic composition

No single benthic macro algae species was found across all the sampling sites, therefore, we collected tissue samples of living macro algae and seagrasses along shorelines in replicates, depending on availability. After cleaning and drying the samples at 40°C for 48 hours at Kenya Marine and Fisheries Research Institute (KMFRI) laboratory, we transported them to the Leibniz Centre for Tropical Marine Research (ZMT), for organic and isotopic carbon and nitrogen analysis. Samples were then ground into fine powder, stored in plastic vials, weighed 2-5 mg into silver capsules, acidified them

with diluted hydrochloric acid (5% HCl) to remove inorganic carbon, and re-dried for 24 hours. These were then flash-combusted in a CN elemental analyzer to determine organic and total C and N contents using Euro EA3000 mass spectrometry. Pre-weighed tissue samples underwent isotope ratio mass spectrometry using Delta plus mass spectrometer connected to the Flash EA 1112 via a Finnigan ConFloIII for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ analysis, with the latter measured using the denitrifier method to ensure precise and clear isotopic characterization of the primary producers in the study area (Derse *et al.*, 2007; Sigman *et al.*, 2001).

All the sample preparations and storage including cleaning and drying of macro algae and sediments in the field was done at KMFRI laboratory, Kenya while the laboratory analysis of nutrients, organic and isotopic carbon and nitrogen analysis were conducted at the ZMT, Germany.

5.3.4. Submarine Groundwater Discharge (SGD)

Submarine groundwater discharge was investigated as a nutrient source and pathway in Mombasa coastal waters using a 51-hour ^{222}Rn time-series study in June 2022. The study, conducted in Mombasa Marine Park near a coral garden, employed a DurrIDGE RAD7 radon monitor alongside measurements of water parameters and meteorological data. During the time series study, hourly surface water samples were collected for the first 27 hours for analysis of biogeochemical parameters. Additionally, beach pore water (8 sites) and Mombasa groundwater (12 wells) were also analyzed for biogeochemical parameters. Samples were processed for ^{222}Rn , dissolved inorganic carbon (DIC), total alkalinity (TA), and nutrients following Burnett and Dulaiova, (2003) procedures. DIC was analyzed using standardized system with Dickson's reference material (Batch 193), while TA was determined by Gran titration. Duplicate TA analyses yielded a standard deviation of $9.8 \mu\text{mol kg}^{-1}$. Nutrient processing followed previously described methods already mentioned. All samples were filtered and preserved according to established protocols in Burnett and Dulaiova, (2003).

5.3.5. Statistical Analysis

Data analysis was performed using R 4.2.3 with vegan package (Oksanen *et al.*, 2024; R Core Team, 2024), PRIMER-E v7.0.22 (Clarke & Gorley, 2006), Sigma Plot 14.0.3.192, and QGIS v3.28.3. Statistical analyses assessed spatio-temporal variations in nutrient concentrations and POM levels. The Kruskal-Wallis tests assessed differences levels across sites and months while Paired t-tests compared these parameters between seasons at different sites. Additionally, Principal Component Analysis

(PCA) was used to extract principal components based on variable correlations, while Spearman's rank correlation coefficients evaluated relationships among physicochemical parameters, particulate organic matter, and isotope values ($p < 0.05$). Isotopic analysis of carbon and nitrogen was conducted at various sites affected by different human activities to identify the sources of nutrients following classifications in Umezawa *et al.*, (2008) while the ecological quality status were classified using criteria listed in Chapter 3 (Oduor *et al.*, 2024).

Mass balance models for SGD estimations

A non-steady state radon mass balance model was developed to estimate SGD into the study area. The model incorporated tidal inputs/outputs, atmospheric evasion, radioactive decay, molecular diffusion, and groundwater concentrations following established protocols (Equation 7) (Burnett *et al.*, 2006; Taniguchi *et al.*, 2019)

Equation 7

$$SGD\ Flux = \frac{\int_{t=0}^{t=25} ({}^{222}Rn_{sw} \Delta V) - {}^{226}Ra \lambda V + {}^{222}Rn \lambda V - (J_{diff} A) + J_{atm} A}{{}^{222}Rn_{gw}}$$

Where ${}^{222}Rn_{sw}$ (dpm m^{-3}) represents Surface water radon concentration. ΔV Is the change in tidal volume ($m^3\ s^{-1}$). ${}^{226}Ra$ is the ${}^{226}Rn$ activity (dpm m^{-3}). λ Is the ${}^{222}Rn$ decay constant ($0.18\ d^{-1}$), V is the canal volume, and A is the canal area. Molecular diffusion (J_{diff}) from sediments ($16 \pm 6\ dpm\ m^{-2}\ h^{-1}$) was estimated from data presented in Makings *et al.* (2014) for a nearby estuary with similar geology. ${}^{222}Rn_{gw}$ (dpm m^{-3}) represents the average groundwater radon endmember concentration. (J_{atm}) represents atmospheric evasion of ${}^{222}Rn$ as a function of piston velocities and the concentration difference between water and air calculated using wind speed data obtained at 30-min intervals (Burnett and Dulaiova, 2003 and references herein).

The SGD-derived carbon and nutrient fluxes were calculated by multiplying SGD estimates with average groundwater concentrations. Multiple models were prepared, including beach seeps only, beach seeps with Mombasa bores, and potentially coastal Mombasa bores. Uncertainties in calculations were determined using Monte Carlo simulations. Limitations included hourly wind speed data affecting evasion calculations, tidal flux uncertainties, and diffusion uncertainties.

5.4.Results

5.4.1. *Variations of nutrients and organic matter in Mombasa coastal waters*

Physical parameters

Mombasa's coastal waters exhibited distinct spatial and temporal patterns in physical parameters and nutrient concentrations across enclosed tidal creek systems (Tudor and Kilindini) and open waters (Table 15). Tidal creeks, particularly Tudor, showed higher temperatures, salinity, and turbidity, with lower dissolved oxygen levels compared to open waters. Both Tudor and Kilindini creeks shared a common trend of decreasing temperature and increasing dissolved oxygen levels as water moved downstream. However, turbidity patterns differed between the two creeks. In Tudor, turbidity reached its peak in mid-water areas, gradually decreasing both upstream and downstream. In contrast, Kilindini Creek displayed a consistent decline in turbidity as water progressed downstream. Temporally, the peak turbidity at Tudor creek was in March at Tudor Creek mouth, coinciding with lower dissolved oxygen and conductivity levels, while Kilindini Creek's turbidity peaked in upper waters in April, aligning with the onset of rains (Appendix 11).

Open waters registered peak turbidity in May. Kilindini consistently displayed year-round downstream decreases in turbidity, whereas Tudor only showed this trend during the wet season. Dissolved oxygen decreased upstream across all seasons, while salinity trends varied seasonally. Conductivity patterns were complex, with Tudor Creek consistently showing significantly higher conductivity levels compared to oceanic waters. Seasonal differences were also observed, with both tidal creeks displaying increasing downstream conductivity during wet seasons. However, dry seasons revealed contrasting patterns between the two creeks. Kilindini Creek showed a clear decreasing trend in conductivity downstream, while Tudor Creek lacked a distinct trend, characterized by slightly lower conductivity levels in mid-waters compared to upper and oceanic waters.

5.4.2. *Dissolved inorganic nutrients*

Similar to turbidity, Tudor Creek consistently exhibited higher overall mean nutrient concentrations than Kilindini Creek, particularly in mid-waters, although the DIN lacked clear spatial trends in both creeks. The NO_x^- concentrations was also higher in mid and upper waters of Tudor creek, while only slight elevations were registered in NO_x^- in mid-waters of Kilindini Creek (Figure 26A). Notably,

ammonium concentrations were markedly higher ($\geq 95.4 \mu\text{M}$) in mid-waters of Tudor Creek and upper waters of both creeks (Figure 26B). Silicate concentrations were significantly higher in tidal creeks compared to open waters, with Tudor Creek showing the highest variability, particularly in mid and upper-waters (Figure 26C). Kilindini Creek displayed a consistent downstream decrease in silicate concentrations throughout the study. However, phosphate concentrations displayed no clear spatial patterns, but were generally higher in Tudor Creek, especially at S10, and S12 (Figure 26D).

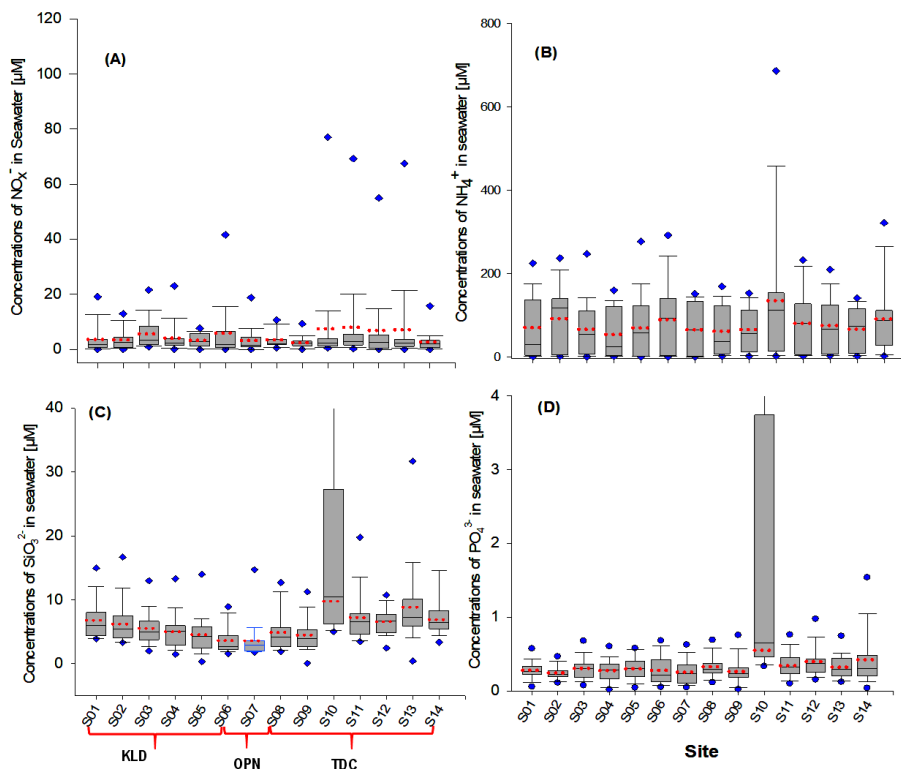


Figure 26: Distribution of dissolved inorganic nutrients - NO_x^- (Nitrates + Nitrites) (A), Ammonium (B), Silicates (C) and phosphates (D) in the surface seawater in Mombasa. KLD – Kilindini creek, OPN – Open Oceanic waters, and TDC – Tudor creek. The Upper waters are S01-S03, and S12 - S14, Mid-waters – S04 –S05, and S09 –S11, Oceanic waters – Creek Mouths S06 and S08, and Open waters – S07.

Nutrient concentrations in Mombasa's coastal waters exhibited distinct spatiotemporal variations across Tudor Creek, Kilindini Creek, and Open Waters (Figure 28). Tudor Creek experienced significant increases in DIN and PO_4^{3-} between August and September, with a tenfold and six fold increase respectively, particularly in mid and upper waters. Kilindini Creek also experienced a notable increase in NO_x^- during this period (Figure 28A). October was characterized by fluctuating nutrient levels, with elevated phosphate concentrations at direct sewage-discharge sites in Tudor Creek (S08, S10, and S12) and high DIN concentrations in upper waters of both tidal creeks. The SiO_3^{2-} concentrations

remained consistently low in Tudor Creek until April, while in Kilindini Creek, both SiO_3^{2-} and PO_4^{3-} stayed low, during the same period, from August to March. In Open Waters, SiO_3^{2-} , PO_4^{3-} , and NH_4^+ concentrations remained low until December, and PO_4^{3-} peaked in May. Actually, the onset of rains in April triggered a general increase in all nutrients in the tidal creeks while only SiO_3^{2-} showing more pronounced increases in the open waters (Appendix 11 and Figure 28). From April to July, PO_4^{3-} mirrored NO_x^- trends, while SiO_3^{2-} paralleled NH_4^+ with the two groups displaying inverse trends in all the waters (Figure 28). Based on overall seasonal means, the analysis revealed distinct nutrient trends across locations. Kilindini Creek experienced an upstream increase in SiO_3^{2-} levels throughout the study period. In contrast, Tudor Creek showed increases in nutrient concentrations only at its oceanic site during the wet season. Additionally, Tudor Creek consistently showed higher SiO_3^{2-} levels in mid and upper waters, with similar NO_x^- trends across both seasons. Overall, NO_x^- concentrations were higher in tidal creek waters during the dry season, while NH_4^+ concentrations were significantly higher during the wet season in all the waters (Table 15).

The nutrient ratios (N: P and N: Si) generally exceeded Redfield ratios (Figure 27). However, August showed potential nitrogen scarcity, while September had localized nitrogen limitations, particularly in Kilindini Creek, open waters and Tudor Creek mid-waters. N: P ratios increased in February and were higher in the wet season. N:Si ratios were typically higher in the dry season, with some exceptions in mid and oceanic waters of Kilindini Creek and upper waters of Tudor Creek (Figure 27A and B). Based on the nutrient ratios (Table 5), the ecological stations composition of Mombasa can be defined as 'Bad'

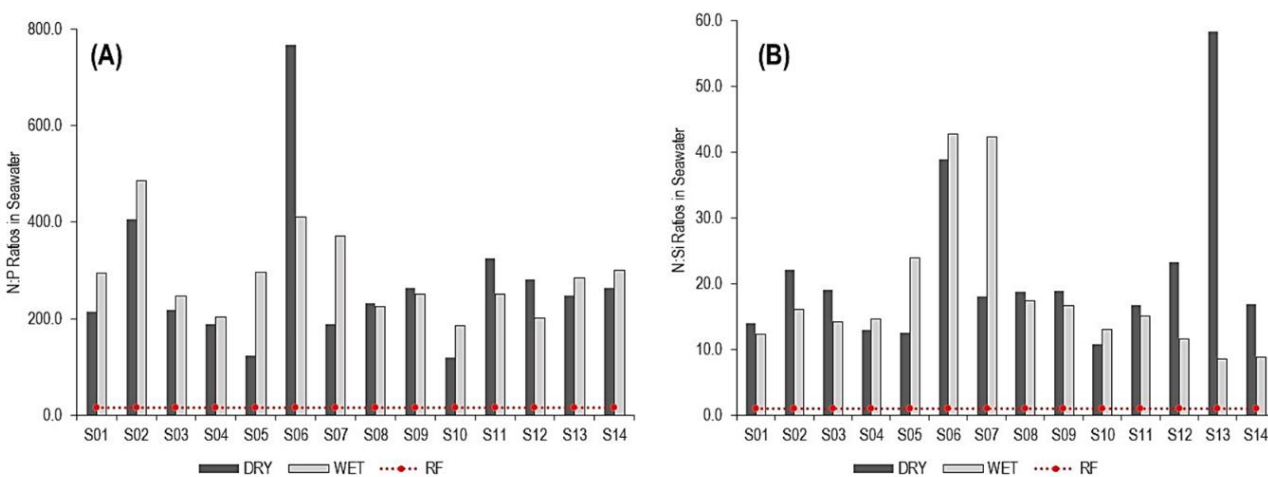


Figure 27: Seasonal variations in nutrient ratios in Mombasa coastal waters: (A) N: P and (B) N: Si. Red dotted lines indicate Redfield ratios.

Table 15: Summary statistics of physical-chemical parameters in Mombasa seawater. Temp. – Temperature, Sal. – Salinity, Turb. – Turbidity, DO – Dissolved Oxygen, Cond. – Conductivity, DIN – Dissolved inorganic nitrogen, C_{tot} – Total carbon, POC –particulate organic carbon, PON – Particulate organic nitrogen. *P-values* indicate variations across Tudor, Kilindini, and Open Waters. **Bold** values are statistically significant.

Parameter	Overall Seawater (n = 340)			Tudor Creek (n = 168)			Kilindini Creek (n = 145)			Open Waters (n = 23)			Groundwater (n = 62)			Pore water (n = 19)			Rainwater (n = 8)			P-Value
	Mean	Min	Max	Mean ± s.d.	Min	Max	Mean ± s.d.	Min	Max	Mean ± SD	Min	Max	Mean	Min	Max	Mean	Min	Max	Mean	Min	Max	
Temp. °C	28.0 ± 0.0	25.4	30.8	28.2 ± 1.2	25.7	30.8	27.9 ± 1.3	25.6	30.7	27.1 ± 1.3	25.4	29.1	29.7 ± 0.3	25.5	33.3	27.3 ± 0.3	25.1	29.2	28.4 ± 0.01	28.4	28.6	0.001
Sal. (PSU)	35.0 ± 0.1	28	37.8	35.0 ± 1.2	28.6	37.6	34.9 ± 1.3	28	37.8	35.0 ± 0.5	33.6	35.4	1.7 ± 0.3	0.1	8.9	34.7 ± 0.3	33.6	35.7	-	-	-	0.234
pH	7.9 ± 0.02	7.1	8.8	7.9 ± 0.4	7.1	8.5	8.0 ± 0.4	7.2	8.8	8.0 ± 0.5	7.2	8.6	6.4 ± 0.1	5.1	7.6	7.6 ± 0.1	6.5	7.9	-	-	-	0.184
Turb. (NTU)	6.8 ± 0.6	0.01	110	8.3 ± 12.5	1.3	110	5.6 ± 0.6	0.1	48.2	2.9 ± 3.7	0.01	17.5	1.1 ± 0.1	0.01	2.8	-	-	-	-	-	-	<0.001
DO (%)	69.0 ± 0.6	19.2	113.13	66.7 ± 11.5	32.4	113.1	70.2 ± 1.3	27.9	105.5	76.7 ± 7.6	66.2	97.8	49.5 ± 2.4	19.7	91.7	12.0 ± 6.2	2.2	22.3	-	-	-	<0.001
Cond. (mS/cm)	55658 ± 288	5819	61834	55447 ± 7112	581	60796	55975 ± 176	4640	61834	55213 ± 1240	5377	5719	3474 ± 522	132	20824	-	-	-	-	-	-	0.006
NO _x ⁻ [μM]	4.9 ± 0.6	0.01	102.6	5.5 ± 1.0	0.01	102.6	4.4 ± 0.6	0.01	60.84	3.2 ± 4.6	0.01	21.6	906.8 ± 107	0.01	3550	3.2 ± 1.2	0.0	19.2	10.3 ± 2.3	5.3	18.1	0.443
NH ₄ ⁺ [μM]	76.4 ± 5.0	0.01	705.2	83.2 ± 93.5	1.53	705.2	72.0 ± 6.7	0.14	298	64.9 ± 59.2	0.01	151	30.8 ± 8.9	0	325	39.7 ± 11.9	6.5	99.5	5.8 ± 1.5	2.9	10.6	0.328
DIN [μM]	81.4 ± 4.9	0.02	706	88.9 ± 92.0	1.6	706	76.5 ± 6.5	0.34	304.5	68.1 ± 59.1	0.02	152.3	903 ± 104	0	3569.8	46.3 ± 10.4	14.3	100.2	16.1 ± 3.2	8.9	25.7	0.400
PO ₄ ³⁻ [μM]	0.8 ± 0.3	0.01	56.4	1.3 ± 6.7	0.02	56.4	0.28 ± 0.01	0.01	0.72	0.3 ± 0.2	0.04	0.7	2.0 ± 0.3	0.1	13.1	2.8 ± 1.1	0.0	21.7	3.0 ± 1.4	0.5	8	<0.001
SiO ₃ ²⁻ [μM]	7.6 ± 0.8	0.01	160.7	10.3 ± 20.8	0.05	160.7	5.1 ± 0.3	0.01	18.26	3.6 ± 3.3.1	1.78	16.8	152.9 ± 11.5	1.5	342.6	19.3 ± 4.9	5.2	103.2	4.2 ± 0.7	2.5	6.4	<0.001
N/P	274.8 ± 26.2	0.18	5613.9	242 ± 432.5	4	1213	321.1 ± 55.0	4.5	5614	260 ± 61.3	0.18	959.3	1852 ± 449	63.8	16196	22.4 ± 5.4	4.6	45.8	12.2 ± 4.3	1.1	22	0.862
N/Si	20.2 ± 2.3	0.01	505.7	18.8 ± 4.0	0.28	505.7	21.2 ± 2.4	0.2	158.7	27.4 ± 6.2	0.01	70.2	7.0 ± 0.9	0.6	32.7	2.3 ± 0.5	0.6	4.2	4.2 ± 1.1	1.9	8.4	0.443
Seston (mg/L)	36.9 ± 0.6	28.5	74.1	38.4 ± 8.2	30.6	66.1	37.8 ± 5.1	30.8	51.7	35.5 ± 3.5	30.8	40.9										0.175
C _{tot}	0.8 ± 0.03	0.3	2.1	0.9 ± 0.4	0.4	2.1	0.7 ± 0.2	0.4	1.3	0.9 ± 0.5	0.4	1.9										0.169
POC (%)	0.6 ± 0.03	0.2	1.9	0.7 ± 0.3	0.3	1.9	0.6 ± 0.2	0.3	1.3	0.8 ± 0.5	0.3	1.8										0.057
PON (%)	0.08 ± 0.003	0.03	0.2	0.1 ± 0.04	0.04	0.2	0.1 ± 0.02	0.04	0.1	0.1 ± 0.03	0.1	0.1										0.123
POC/PON	7.4 ± 0.2	3.1	20	7.4 ± 2.9	4.4	20.3	7.1 ± 2.4	3.8	18.7	10.0 ± 6.1	5.1	20.4										<0.001

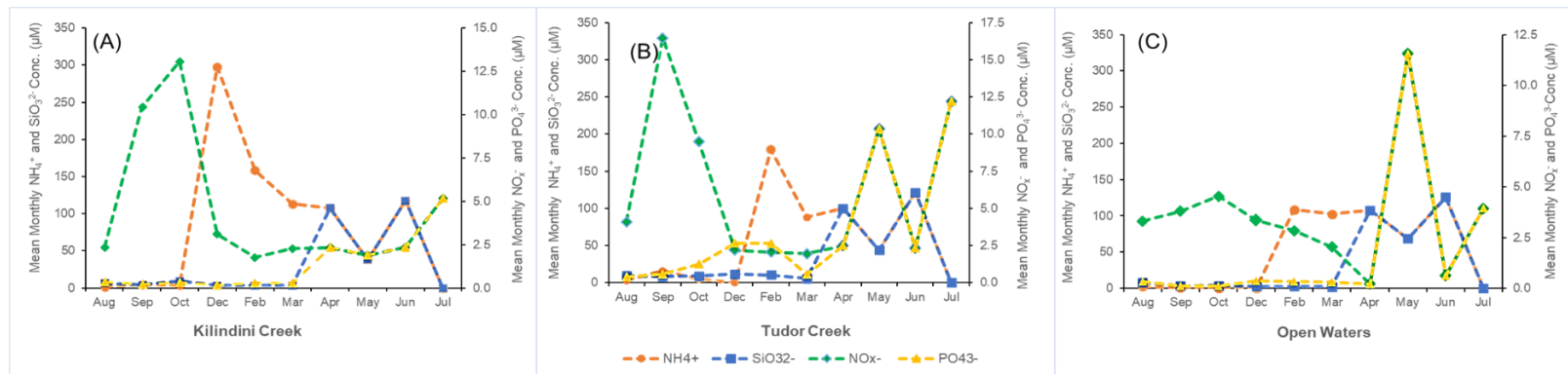


Figure 28: Spatiotemporal variation in dissolved inorganic nutrients in Mombasa Coastal waters in Kilindini Creek (A), Tudor Creek (B) and Open Oceanic waters (C). Dotted lines, NH₄⁺ – Ammonium, – SiO₃²⁻ – Silicates, Nitrates +Nitrites (NO_x⁻) and PO₄³⁻ – Orthophosphates.

Organic matter

The distribution of organic matter varied significantly among seston, macro algae, and sediments. Macroalgae had higher organic nitrogen (N_{org}) and carbon (C_{org}) content, averaging $2.2 \pm 0.7\%$ and $31.1 \pm 8.5\%$, compared to sediments ($0.1 \pm 0.03\%$ and $1.0 \pm 0.5\%$) and seston ($0.1 \pm 0.04\%$ and $0.6 \pm 0.3\%$). The sediments N_{org} , C_{org} , total carbon (C_{tot}), and C_{org}/N_{org} ratios compositions ranged from 0.01-0.1%, 0.2-2%, 0.3-2.0%, and 11.39-35.9%, respectively. In the sediment cores, vertical distribution of C_{org}/N_{org} in Tudor Creek midwaters (S10) peaked in the top 0-5 cm, with nitrogen remaining stable while carbon decreased with depth. In contrast, upper waters at S11 showed N_{org} highest at the surface and C_{org} peaking at 5-10 cm layer, with C_{org}/N_{org} ratios increasing with depth. At S12A, both N_{org} and C_{org} were greatest at 20-22 cm, while in S12B, they peaked at 15-19 cm, with nitrogen increasing and carbon decreasing with depth (Table 16). Overall, significant spatial variation in C_{org} content was observed, but no significant differences in C_{org}/N_{org} ratios were found within the top 15 cm across the sites.

Table 16: Distribution of organic matter in the sediments. Sites = S10 – S12-B, C_{tot} - total carbon, C_{org} - organic carbon, N_{org} – Organic nitrogen and C_{org}/N_{org} – Ratios of organic carbon and nitrogen

Depth (cm)	C_{tot} [%]				C_{org} [%]				N_{org} [%]				C_{org}/N_{org}			
	S10	S11	S12-A	S12-B	S10	S11	S12-A	S12-B	S10	S11	S12-A	S12-B	S10	S11	S12-A	S12-B
0-5	1.54	0.49	1.29	1.36	0.87	0.48	1.12	1.34	0.06	0.03	0.06	0.06	15.45	14.09	18.97	21.19
5-10	1.52	0.62	1.19	1.35	0.75	0.50	1.11	1.31	0.06	0.03	0.06	0.07	13.62	15.53	17.34	18.93
10-15	1.55	0.28	0.71	1.51	0.64	0.24	0.67	1.43	0.06	0.01	0.04	0.08	11.39	21.55	17.10	17.83
15-20	-	0.44	0.92	2.03	-	0.36	0.78	1.98	-	0.01	0.04	0.11	-	35.90	20.45	18.52
> 20	-	-	1.60	-	-	-	1.59	-	-	-	0.09	-	-	-	17.06	-

In the surface waters, particulate organic matter displayed similar spatiotemporal patterns to nutrients. Tudor Creek consistently exhibited the highest seston concentrations, with both tidal creeks showing a progressive decrease in seston and particulate organic carbon downstream (Table 15 and Figure 29A). The highest mean POC composition (1.2%) was recorded in the upper waters (S14) in Tudor Creek (Appendix 14). Spatially, total carbon (C_{tot}) concentrations were generally higher upstream in both tidal creeks. Kilindini Creek showed notably higher C_{tot} and POC in upper waters (at S02), with lower concentrations in mid-waters (Appendix 14B). POC: PON ratios were higher in open waters compared to creek waters, suggesting terrestrial influence. This was further supported by a weak but statistically significant correlation was also established between C_{tot} and latitude ($r = 0.38$, $p < 0.01$). POC: PON ratios varied significantly spatio-temporally ($p < 0.001$), with highest

values in October for Tudor Creek (10.3 ± 5.1) and open waters (19.4 ± 1.5), while consistently lower in mid-waters of both tidal creeks (Figure 29D). A weak but significant negative correlation was found between temperature and seston, C_{tot} , and POC ($r = -0.37$, $p < 0.01$), with the relationship being stronger for POC.

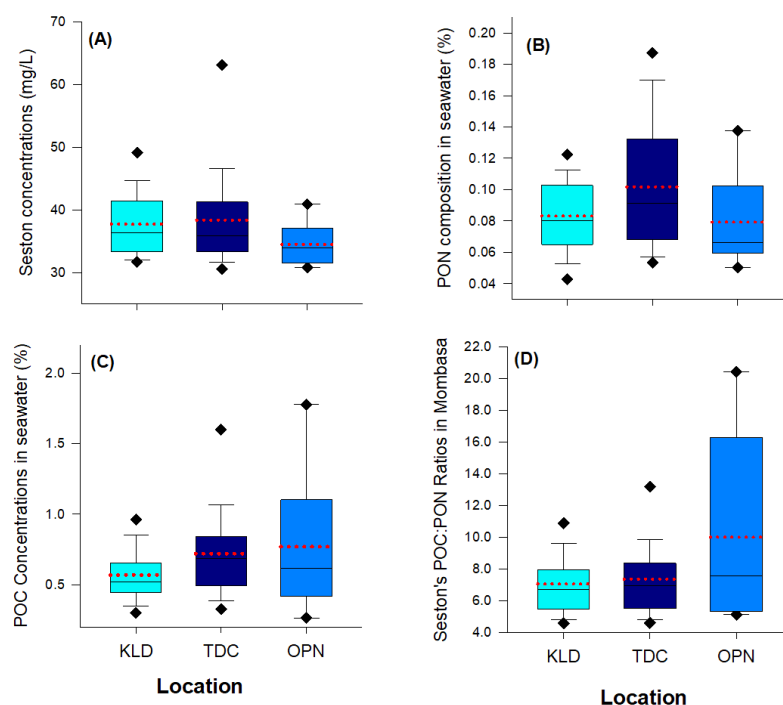


Figure 29: Box and whisker plots showing concentration of seston (A) and the composition of particulate organic nitrogen (PON) (B), particulate organic carbon (POC) (C) and POC: PON ratios (D) across Tudor Creek, Kilindini Creek, and open waters. The horizontal bar inside the box indicates the median value of each cluster, while the red dotted lines denotes the mean value.

Temporally, seston concentrations peaked in October for Kilindini Creek (47.3 ± 2.2 mg/L) and open waters (41.4 ± 1.4 mg/L), while Tudor Creek reached its maximum in May (39.3 ± 2.4 mg/L). C_{tot} and POC in Kilindini Creek were highest in October ($1.6 \pm 0.4\%$ and $1.5 \pm 0.4\%$, respectively), while Tudor Creek's highest C_{tot} occurred in May ($1.1 \pm 0.3\%$). Tudor Creek showed a seston spike in the upper waters (at S13) in May with decreased C_{tot} and POC, while Kilindini mouth exhibited an inverse seston-carbon relationship in July. The POC concentrations showed seasonal variations, being slightly higher in the dry season ($0.9 \pm 0.4\%$) compared to the wet season ($0.6 \pm 0.2\%$), though not statistically significant. During the wet season, open waters exhibited higher POC levels than creek waters. However, both POC and PON decreased downstream in both seasons, with this trend being more pronounced in Tudor Creek. Seasonal correlations were observed between POM components

and environmental parameters. In the dry season, dissolved oxygen positively correlated with organic nitrogen ($r = 0.41$, $p = 0.05$), while silicate concentrations negatively correlated with POC ($r = -0.39$, $p = 0.04$). During the wet season, both seston and organic nitrogen negatively correlated with salinity ($r = -0.33$, $p = 0.029$; $r = -0.50$, $p < 0.001$).

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5.4.2. Potential sources contributing to nutrient enrichment in Mombasa coastal waters

Nutrients from Groundwater

The analysis of potential nutrient sources in Mombasa's coastal waters revealed distinct contributions from groundwater, rainwater, and sewage inputs. Groundwater exhibited varying nutrient profiles between well water and pore water. Well water was characterized by high concentrations of NO_x^- and silicate, while pore water showed elevated levels of silicates and ammonium. Mainland wells, particularly those in informal settlements, contained higher DIN concentrations compared to island wells, with NO_x^- accounting for 89.3% of DIN. Nutrient concentrations in the wells exhibited significant seasonal variations, with DIN levels markedly higher in the wet season (1870 ± 1020 μM) compared to the dry season (677 ± 487 μM). Significant seasonal variations were also observed in dissolved silicate concentrations, while phosphate concentrations remained consistently low. Salinity in wells ranged from 0.6 to 6.0 PSU and negatively correlated with nutrient levels, particularly

NO_x^- and silicate. Porewater exhibited distinct nutrient profiles, with salinity ranging between 33.6 and 35.6 PSU. Elevated NO_x^- and SiO_3^{2-} concentrations were observed near the MMPR ($6.7 \pm 7.6 \mu\text{M}$, and $20.5 \pm 3.6 \mu\text{M}$), while higher PO_4^{3-} concentrations were found in MDB in Tudor Creek ($5.2 \pm 8.3 \mu\text{M}$; Figure 30A). Low salinity, high dissolved oxygen, high NH_4^+ ($99.5 \mu\text{M}$), and very low NO_x^- ($0.7 \mu\text{M}$) concentrations were also observed in porewater near MMPR's coral garden (Figure 30A and Appendix 13). Both well and pore waters consistently displayed elevated N: P and N: Si ratios throughout the study period.

Nutrients from Rainwater

Rainwater samples indicated elevated DIN concentrations, primarily in the form of NO_x^- ($5.3 - 14.5 \mu\text{M}$), compared to other nutrients, with elevated levels registered inland at RW4 ($23.6 \pm 3.0 \mu\text{M}$). The PO_4^{3-} concentration was highest at RW2 ($8.0 \pm 0.3 \mu\text{M}$), a high-traffic area associated with numerous garages and open tire combustion. Statistical analysis revealed significant differences in phosphate concentrations ($p = 0.03$) between busy traffic and residential areas (Figure 30B and Appendix 13), with Mombasa Island sites (RW1 and RW2) showing notably low N: P ratios ($1.1 - 3.6$).

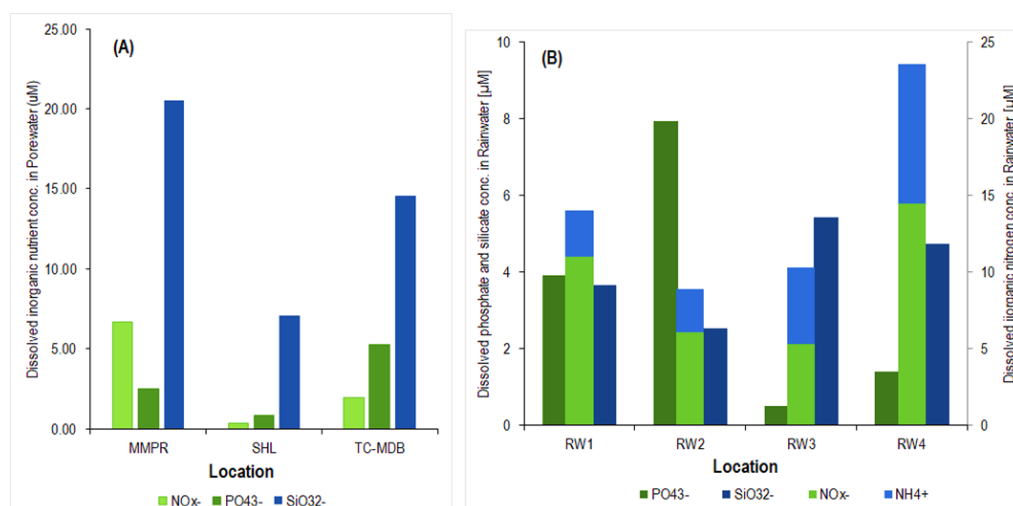


Figure 30: Dissolved inorganic nutrient concentrations in pore water (A) and rainwater (B) in Mombasa, Kenya. Locations MMPR – Mombasa Marine Park, SHL – Shelly, TC-MDB –Tudor Creek, Madobini. RW1 and RW2 represents busy traffic areas in Mombasa Island while RW3 and RW4 are residential areas.

Nutrient source identification via isotopic analysis of macroalgae

The analysis of nutrient sources in Mombasa using macro algae isotopic composition revealed distinct patterns across locations with each area exhibiting distinct macro algae compositions. MMPR primarily contained seagrasses with some brown and green macro algae, Shelly Beach featured a mix of seagrasses and green algae while Tudor Creek exclusively had macro algae. Sewage influence

was evident in the isotopic composition of macro algae. Open nearshore sites at MMPR had the lightest nitrogen isotope values, while macro algae tissues from sewage-affected areas exhibited heavier $\delta^{15}\text{N}$ values ($\sim +14.4 \pm 0.7\text{‰}$), indicating a clear influence of sewage inputs (Figure 32). Sites along Tudor Creek recorded the highest organic nitrogen content in macro algae with 3.4% and 2.5% at S09 (MDB) and S08 (MEP), respectively (Table 17). The $\delta^{15}\text{N}$ values ranged from 2.6‰ to 14.9‰, while $\delta^{13}\text{C}$ values ranged from -29.8‰ to -6.4‰, with mean values of +7.8‰ ($\pm 3.2\text{‰}$) and -13.1‰ ($\pm 4.9\text{‰}$) respectively. Seasonal variations showed significantly lighter $\delta^{15}\text{N}$ values during the rainy season, while organic nitrogen composition positively correlated with $\delta^{15}\text{N}$ in macro algae.

Table 17: Mean (\pm s.d.) chemical composition (elemental C/N ratios and carbon and nitrogen stable isotope ($\delta^{13}\text{C}$, $\delta^{15}\text{N}$) of macro-algae in Mombasa Coastal waters, Kenya. n = number of samples.

Location	Collection site	n	N [%]	C _{tot} [%]	C _{org} [%]	$\delta^{15}\text{N}$ [‰]	$\delta^{13}\text{C}_{\text{org}}$ [‰]
Mombasa MP&R	Coral Garden	5	2.3 ± 0.4	36.0 ± 5.4	34.0 ± 6.4	6.4 ± 1.4	-12.5 ± 4.7
	Nyali Beach	1	2.7 ± 0.1	37.3 ± 0.1	36.3 ± 0.1	8.7 ± 0.1	-12.0 ± 0.1
	Pirates	3	2.1 ± 0.3	37.4 ± 3.8	37.3 ± 4.0	3.2 ± 0.6	-10.9 ± 2.2
	Serena	3	1.4 ± 0.9	29.2 ± 12.7	24.6 ± 18.1	4.8 ± 1.6	-19.0 ± 9.6
	Star Fish	4	1.6 ± 0.3	32.3 ± 3.9	31.1 ± 5.6	6.6 ± 1.9	-13.6 ± 4.6
	Wras Watini	1	2.3 ± 0.1	35.4 ± 0.1	34.3 ± 0.1	4.5 ± 0.1	-6.4 ± 0.1
Shelly Beach	Shelly beach	5	2.3 ± 0.4	32.5 ± 4.0	30.5 ± 5.2	11.0 ± 2.3	-10.3 ± 2.1
	Simba Villa	3	2.2 ± 0.3	36.2 ± 4.4	35.5 ± 4.5	9.4 ± 1.8	-9.0 ± 0.3
Tudor Creek	Madobini	2	3.4 ± 0.2	28.0 ± 4.0	27.6 ± 4.7	10.1 ± 0.6	-14.5 ± 0.4
	Marina English Point	3	2.5 ± 1.2	26.9 ± 8.2	22.5 ± 11.7	11.4 ± 3.3	-18.2 ± 1.4

5.4.3. Pathways of nutrient transport and distribution in Mombasa coastal waters

Submarine ground water discharge

Results revealed distinct chemical signatures across the three water types (Table 18). Groundwater samples exhibited consistently lower salinities and elevated concentrations of ^{222}Rn , DIN, and silicate compared to pore water and oceanic waters. Phosphate concentrations were significantly higher in both pore and well waters relative to adjacent oceanic waters. Time series analysis revealed congruent patterns in NO_x^- , SiO_3 , and ^{222}Rn concentrations, with low tides exhibiting decreased salinity and elevated nutrients, indicating SGD's coastal nutrient dynamics influence (Figure 31). The PCA also revealed ^{222}Rn in seawater positively correlated with temperature, dissolved oxygen, pH, phosphate, and silicate, while negatively correlating with salinity, TA, DIC, and DIN, with only salinity showing significant correlation. Combining the salinity-mixing model with our ^{222}Rn , mass balance results in a best-estimate flux SGD to the MMPR of $147.15 \text{ m}^2\text{day}^{-1}$ (freshwater groundwater influx ranged between 23.35 and $398.87 \text{ Lm}^{-2} \text{ d}^{-1}$, Table 19). This discharge supplied substantial amounts

of nutrients and carbon to the coastal waters contributing 8.85, 0.36 and 4.22 mmol/m²/day of DIN, phosphate, and silicate, respectively (Table 19).

Table 18: Water quality and biogeochemical data from Mombasa Marine Park and Reserve (MMPR) time series site, beach pore water and Mombasa wells in June 2022. ²²²Rn – Radon, DO – Dissolved Oxygen, TA – total alkalinity, DIC – Dissolved inorganic carbon

Parameter	Seawater (²²² Rn, TA and Sal. = 25, all Others = 27)			Pore water (n = 8)			Groundwater (n = 12)		
	Mean + s.d.	Min	Max	Mean + s.d.	Min	Max	Mean + s.d.	Min	Max
²²² Rn(dpm/L)	0.33 ± 0.30	0.00	1.46	0.28 ± 0.10	0.16	0.45	12.10 ± 14.32	1.02	48.46
Temperature (°C)	27.06 ± 0.95	25.30	28.50	28.08 ± 0.67	27.30	29.30	28.27 ± 1.68	26.20	31.50
pH	8.26 ± 0.14	8.00	8.53	7.43 ± 0.42	6.46	7.77	7.05 ± 0.32	6.30	7.45
DO (mg/L)	7.11 ± 1.39	5.06	10.50	0.69 ± 0.45	0.18	1.71	4.76 ± 1.48	1.83	6.71
TDS (g/L)	52.95 ± 0.23	52.10	53.30	52.90 ± 0.14	52.80	53.10	756.8 ± 811	1.60	1722
Salinity (ppt)	34.5 ± 0.69	32.58	35.72	34.77±0.80	33.64	35.65	1.27 ± 1.56	0.01	6.12
TA (μmol/kg)	2275 ± 29	2224	2354.5	3425 ± 791	2689	4575	8457 ± 1169	6003.7	10137
DIC (μmol/kg)	1943 ± 106	1715	2159	2627 ± 6937	2551	2708	9470 ± 947	8047.9	10535
NOx [μmol/kg]	2.53 ± 1.4	0.60	6.41	5.94 ± 7.35	0.73	19.2	1871 ± 1020	603.4	3551
NO2 [μmol/kg]	0.28 ± 0.07	0.16	0.50	0.21±0.06	0.14	0.3	12.0 ± 32.4	0.19	103.7
NH4 [μmol/kg]	6.07 ± 10.77	0.00	48.67	38.74±29.28	6.51	99.5	3.05 ± 6.0	0.0	19.2
PO4 [μmol/kg]	0.19 ± 0.07	0.06	0.41	2.38±1.11	1.24	4.28	2.24 ± 3.9	0.11	13.1
SiO3 [μmol/kg]	3.59 ± 1.53	2.36	10.78	20.20±3.42	15.84	24.6	218.1 ± 102	35.3	342.6
TA:DIC	1.17 ± 0.06	1.05	1.30	1.05±0.01	1.05	1.1	0.92 ± 0.1	0.8	1.0
N:P	46.4 ± 154.7	15.82	122.82	22.04±13.27	4.65	45.9	4924 ± 5965	133.8	16198
N:Si	2.45 ± 7.1	0.38	4.69	2.28±1.20	0.59	4.22	11.5 ± 9.9	2.7	32.7

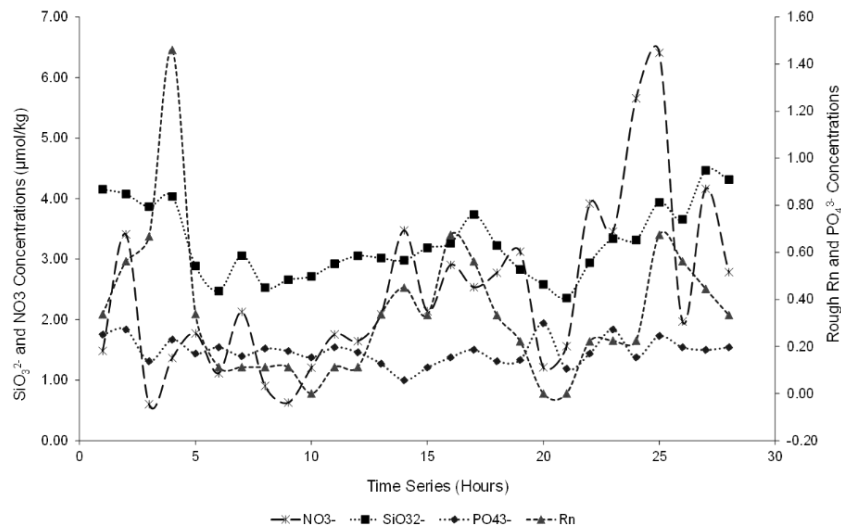


Figure 31: Variations in Rn-estimated SGD and SGD nutrient fluxes DIN, PO₄³⁻ and SiO₃²⁻ in Mombasa Marine Park in June 2022

Table 19: Submarine groundwater discharge and associated biogeochemical fluxes at MMPR in June of 2022.

Parameter	Mixed fSGD Average	Min fSGD	Max fSGD
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Groundwater discharge (L/day)	1471.47	3928.72	233.52
Groundwater (cm/day)	147.15	392.87	23.35
Groundwater flow(L/m ² / day)	147.15	392.87	23.35
TA Flux (mmol/m ² /day)	448.95	515.05	132.71
DIC Flux (mmol/m ² /day)	342.22	395.23	133.38
NO _x ⁻ Flux (mmol/m ² /day)	2.13	0.90	3.00
NO ₂ Flux (mmol/m ² /day)	0.11	0.03	0.12
NH ₄ ⁺ Flux (mmol/m ² /day)	6.61	5.83	8.11
PO ₄ ³⁻ Flux (mmol/m ² /day)	0.36	0.36	0.06
SiO ₃ Flux (mmol/m ² /day)	4.22	3.04	2.25
CO ₂ Flux	11.47	5.70	9.58

***Mixed Average fSGD**: Represents the estimated composition and delivery of SGD, derived from a salinity mixing model, indicating that it consists of approximately **4.73% fresh groundwater** and **92.3% saline porewater**; **Minimum fSGD**: Indicates the maximum potential fluxes from saline beach pore water, assuming that **100% of the SGD is saline**, based on the highest surface water salinities. **Maximum fSGD**: Reflects the highest possible fluxes of fresh groundwater, calculated from the salinity-mixing model using the lowest surface water salinities, showing a composition of **44.8% fresh groundwater** and **55.2% saline groundwater**. All biogeochemical fluxes listed in the table are expressed as proportions relative to their concentrations in both fresh and saline SGD.

5.5. Discussion

5.5.1. Distribution of nutrients and organic matter in Mombasa coastal waters

Dissolved inorganic nutrients

This study examined nutrient and organic matter variations in Mombasa's coastal waters, revealing significant heterogeneity. Tudor Creek consistently showed higher turbidity and nutrient levels compared to Kilindini Creek and open waters, reflecting localized pollution, tidal effects, and river discharges. Tudor Creek's nutrient patterns can be associated with sewage inputs from hospitals and municipalities, sediment resuspension during tidal exchanges due to its shallow nature, and seasonal river inputs during rainy periods. These findings align with previous research on the area's complex coastal dynamics and anthropogenic influences (Kithiia & Majambo, 2020; Nguli *et al.*, 2006; Okuku *et al.*, 2011), indicating interaction between human and environmental factors in shaping coastal water quality.

Distinct spatiotemporal nutrient patterns was also evident in Mombasa's coastal waters. Tudor Creek exhibited distinct nutrient dynamics in March, characterized by elevated turbidity and reduced dissolved oxygen, conductivity, and salinity during the non-rainfall period (Appendix 11). This suggests an increased freshwater supply resulting from higher sewage discharge. Conversely, Kilindini's upper waters showed peak turbidity in April from sediment erosion (Kitheka, 2000), while open waters

peaked in May due to gradual settling of suspended materials supplied by terrestrial run-offs (McCandliss *et al.*, 2002; Mutia *et al.*, 2021). Nutrient levels rose during the wet season due to increased runoff, evidenced by lower salinities and higher turbidities in upper tidal creeks (Appendix 12). In Tudor Creek, silicate and ammonium concentrations decreased downstream, negatively correlating with salinity but positively with turbidity, reflecting runoff effects and sewage discharges (Kithiia & Majambo, 2020; Ongore *et al.*, 2013). Kilindini Creek exhibited consistent downstream decreases in SiO_3^{2-} concentrations, while seasonal variations in NO_x^- and dissolved oxygen levels showed inverse patterns, increasing downstream during dry season and shifting during wet season suggesting nutrient-enhanced productivity and terrestrial influx influences (Kitheka, 2000; Li *et al.*, 2014). These findings reveal complex spatiotemporal nutrient patterns in Mombasa's coastal waters, with Tudor Creek exhibiting higher pollution levels due to various inputs, while overall nutrient concentrations increase during wet seasons, highlighting the significant impact of terrestrial runoff on water quality.

The nutrient ratios (N:P and N:Si) in Mombasa's coastal waters were consistently higher than the recommended Redfield ratios, indicating imbalanced nutrient supply for primary production enriched with DIN in relation to P and Si (Figure 27). This pattern is similar to observations in the region during the south-east monsoon (Sekadende *et al.*, 2021) and some Mediterranean coastal areas affected by anthropogenic inputs (Pujo-Pay *et al.*, 2011). However, unlike the previous studies which indicated nitrogen limitation across Kenya's coastal waters (Chapter 2; Oduor *et al.*, 2023), Mombasa's waters experienced phosphorus and silicate limitations, more pronounced during wet seasons. This suggests runoff rich in DIN but deficient in phosphorus and silicates, a pattern observed in other areas in the region (Chapter 3; Oduor *et al.*, 2024), and some tropical coastal systems (Choudhury & Bhadury, 2015). However, nitrogen limitation occurred in August and localized in September in oceanic waters, coinciding with offshore dredging activities (Personal Observation). This suggests higher N and Si inputs from suspended sediments, while lower N:P ratios at sewage-exposed Tudor Creek sites during dry seasons highlighted sewage's role in disproportionate phosphate enrichment (Xu *et al.*, 2008). This variation emphasizes the importance of site-specific assessments in understanding nutrient dynamics in coastal ecosystems.

Distribution of Organic Matter in Mombasa Coastal Waters

The distribution of organic matter in Mombasa's coastal waters varied significantly, with Tudor Creek exhibiting higher and more variable PON levels, while open waters showed slightly elevated

C_{tot} and C_{org} in seston, indicating potential terrestrial influence. Elevated POC levels in upper and open waters can be attributed to mangrove systems and phytoplankton, respectively, known for their high carbon content (Kimeli *et al.*, 2021; Umezawa *et al.*, 2008). Tidal creeks exhibited decreasing POC downstream from seasonal river mouths and mangroves, aligning with observations in similar ecosystems (Kimeli *et al.*, 2021). During the dry season, higher POC in oceanic waters coincided with elevated NO_x^- levels (Reference herein), suggesting increased primary production and carbon-enriched phytoplankton biomass (Oduor *et al.*, 2024). Seston and carbon composition spatial trends were consistent across May, June, and October, suggesting links to phytoplankton growth or sediment resuspension (Nyunja *et al.*, 2009). An inverse relationship between in July indicated soil-enriched seston input during low tides (Okuku *et al.*, 2011). Negative correlations were established between seston and salinity, and temperature and seston/carbon composition, particularly during the wet season, reflecting freshwater influx, bacterial degradation, and enhanced biological activity (Boëchat & Giani, 2008; Fileto *et al.*, 2010).

This study revealed significant variations in organic matter distribution across Mombasa's coastal ecosystem components. Macroalgae exhibited higher carbon and nitrogen levels compared to sediments and seston, reflecting their role as primary producers (Queirós *et al.*, 2019). Sediments contained more organic carbon than seston, likely due to decomposition and benthic fauna activity (Huguet, 2017), while seston showed higher organic nitrogen content, possibly from phytoplankton and fresh terrestrial inputs (Parparov *et al.*, 2014).

In sediments exposed to direct sewage inputs (S10-S12), organic matter composition varied vertically, influenced by hydrodynamic conditions and anthropogenic exposures (Kaymak *et al.*, 2023). Site-specific trends in organic carbon and nitrogen distribution indicated distinct interactions among deposition, degradation, and bioturbation processes (Kerner & Edelkraut, 1995; Leipe *et al.*, 2000). For example, higher organic carbon and nitrogen in surface sediments at Site S12 (Table 16) were attributed to open defecation, proximity to mangroves, and low tidal mixing (Kithiia & Wambua, 2010; Mohamed *et al.*, 2009). The study suggests that terrestrial inputs, biological processes, and human activities significantly influence organic matter distribution in Mombasa's coastal waters, with findings comparable to similar regional ecosystems (Kimeli *et al.*, 2021; Tamooch *et al.*, 2012). Overall, these results suggest that terrestrial inputs, biological processes, and human activities influence the distribution of organic matter in Mombasa's coastal waters.

5.5.2. *Potential sources and pathways of nutrients in Mombasa coastal waters*

Nutrients in coastal and marine waters originate from various natural and anthropogenic sources. This study established a complex network of nutrient sources and pathways, significantly affecting the ecosystem's health and dynamics in Mombasa's coastal waters reveals. Sewage emerges as a primary nutrient source, particularly evident in Tudor Creek's midwaters (Figure 26, Figure 27). Significantly higher concentrations of dissolved inorganic nutrients and organic matter in this area align with findings from previous studies in the area and similar urbanized coastal environments (Okuku *et al.*, 2011; Saito *et al.*, 2018). Elevated phosphate levels, often correlated with silicates and turbidity, which indicates both sediment sources and sewage inputs, potentially linked to detergent use (Gomes De Quevedo & Wanderley, 2016; Kithika *et al.*, 1999). Literature has shown that sewage is a major source of nutrients in coastal waters (Malone & Newton, 2020). In Mombasa, spatial and temporal correlations between known sewage inputs and nutrient concentrations have been observed, particularly in Tudor Creek. Direct discharge of sewage into coastal waters is the primary pathway for nutrient input, as confirmed by several studies (Kithika & Majambo, 2020; Okuku *et al.*, 2011). Elevated dissolved inorganic nitrogen and phosphate concentrations in Tudor Creek's midwaters during the dry season further substantiate sewage-derived nutrient imbalances (Xu *et al.*, 2008). These findings collectively demonstrate that sewage is a significant contributor to nutrient inputs in Mombasa's coastal waters.

Terrestrial runoff significantly contributes nutrients and organic matter to Mombasa's coastal waters, especially during wet seasons (Figure 28A). Elevated DIN and silicate concentrations, coupled with decreased salinity in Kilindini Creek during rainy periods, indicate intensified surface runoff and freshwater influx from terrestrial sources (Alex *et al.*, 2021; Barongo *et al.*, 2006; Kithika & Majambo, 2020). The seasonal variations in nutrient transport highlight the significance of runoff as the primary pathway for terrestrial nutrients to reach coastal waters, complementing sewage inputs aligning with the findings of Kithika, (2000) and Kazungu *et al.*, (1989) in the creeks.

Apart from terrestrial sources, a significant contribution of mangrove ecosystems was also evident during the study with high silicate concentrations in upper tidal creeks near mangroves indicate their role in dissolved silicate exports (Wang *et al.*, 2023). These ecosystems act as both source and pathway, releasing nutrients through leaf litter decomposition and root exudates, which are then transported to coastal waters via tidal flushing (Kithika, 1998). Additionally, rainwater analysis indicated that atmospheric deposition could be a significant nutrient source, particularly for nitrogen. Elevated

DIN levels, predominantly NO_x^- , were observed, likely due to urban industrial activities, vehicle emissions, and industrial pollutants (Berthold *et al.*, 2019; Mahowald *et al.*, 2008). The atmosphere serves as a direct pathway for these nutrients to enter coastal waters through precipitation and dry deposition; however, the contribution of dry deposition was not assessed during the current study. The contribution of sediment resuspension to nutrient enrichment was also evidenced by the correlation between phosphate levels, silicates, and turbidity. This process is particularly important in areas with high tidal activity and anthropogenic disturbances like Tudor creek, serving as a pathway for nutrients stored in sediments to re-enter the water column. This is enhanced by tidal influences, with soil-enriched seston inputs. The influence of biological activities in nutrient inputs and distribution in Mombasa's coastal waters was also suggested, with downstream NO_x^- increases (Figure 26), inversely related to chlorophyll-a (Chapter 3; Oduor *et al.*, 2024), suggesting phytoplankton uptake upstream and potential nitrification or nitrogen fixation downstream, acting as both nutrient sources and sinks as reported by Kromkamp *et al.*, (1997).

Groundwater serves as both a source and a pathway for nutrients to enter coastal waters in Mombasa through seepage and submarine discharge. The analysis of groundwater in wells near informal settlements (GW4 and GW8) also revealed significant nutrient contributions, likely from sewage leaching through septic tanks. Higher DIN concentrations, especially NO_x^- , were observed in these areas, attributed to poor sanitation infrastructure (Barongo *et al.*, 2006; Mwaguni, 2002). Notably high silicate levels in mainland areas (GW4) suggest silicate-rich soil composition (Kazungu *et al.*, 1989). These nutrient concentration patterns align with previous findings in Mombasa's informal settlements and reflect similar environmental challenges in urbanized areas of developing countries globally (Adyasari *et al.*, 2018; Barongo *et al.*, 2006; Mwaguni *et al.*, 2017; Olonga *et al.*, 2014). The results highlight the impact of urbanization and inadequate sanitation on groundwater quality in coastal regions.

The porewater analysis also revealed diverse nutrient dynamics closely linked to groundwater interactions with high NH_4^+ concentrations relative to NO_x^- in benthic sediments indicate organic matter demineralization under anoxic conditions (Hensen *et al.*, 2006; Wetzel, 2001). Phosphate concentrations were high in the pore water in Tudor Creek's MDB, primarily due to untreated domestic sewage discharge rich in detergent-derived phosphates (Baawain *et al.*, 2015). In the MMPR, distinct nutrient profiles emerged, with evidence of groundwater leaching and possible hotel septic tank leakage near Coral Garden, supporting the earlier claims of Mwaura *et al.*, (2017), of sewage seepage in the

area. Porewater acts as a source of nutrients, which is transported to the water column through diffusion and advection processes. This study found lower nutrient concentrations than previous Mombasa reports, but comparable to regional averages (Kitheka *et al.*, 1999; Ohowa, 1996). Porewater nutrient composition varied across sites, reflecting diverse exposures to local environmental factors such as tidal influences, sediment types, hydrology, and human impacts (Palmer-Felgate, *et al.*, 2010), which indicates the need for further research on porewater-groundwater interactions in coastal nutrient dynamics in Mombasa.

Lastly, the submarine groundwater discharge also emerged as a significant nutrient source and pathway in Mombasa's coastal waters. Distinct chemical signatures of groundwater, characterized by lower salinities and elevated ^{222}Rn , DIN, and silicate concentrations, indicate its potential impact on coastal nutrient dynamics (Table 19). The substantial SGD flux to the MMPR contributes significant amounts of DIN, phosphate, and silicate to coastal waters, with flux rates ranging from 23.35 to 392.87 L/m²/day. This contribution is especially important during low tides, where elevated nutrients were observed (Figure 31). This study found slightly higher SGD rates at Kenyatta Beach compared to earlier findings by Kamermans *et al.*, (2002). The nutrient contributions from SGD align with recent global coastal water medians reported by Santos *et al.*, (2021) for DIN, phosphate, and silicate concentrations. However, the assessment's limited scope—conducted in the park north of Mombasa Island and using static point measurements—may not fully represent SGD's nutrient supply to coastal areas, highlighting the need for more comprehensive spatial and temporal evaluations in the region.

Nutrient source identification via isotopic analysis of macro algae

The results from stable isotope analysis of macro algae indicated diverse origins of organic matter, revealing distinct mixing patterns across various locations (Figure 32). Macroalgae from Tudor Creek predominantly derived nutrients from marine wastewater, while Shelly Beach showed a greater influence from terrestrial wastewater and organic soil contributions (Figure 32).

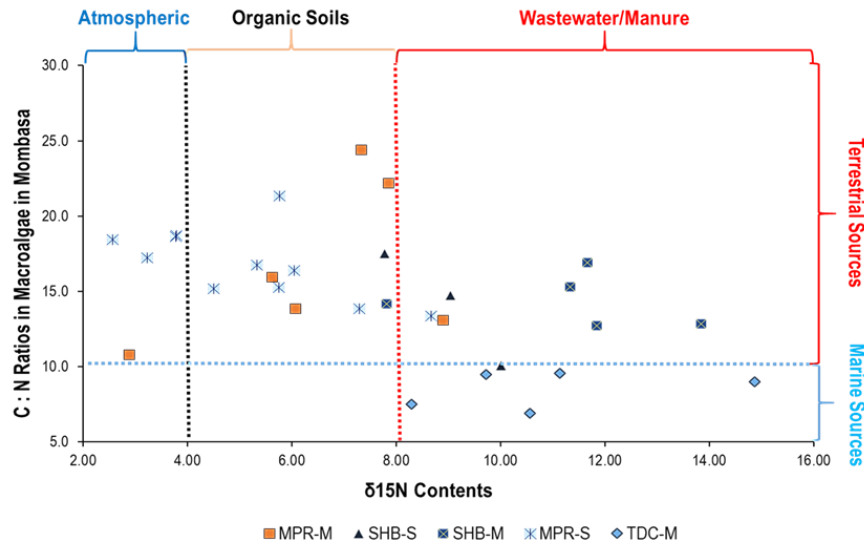


Figure 32: N-Source Determination using $C_{org}:N_{org}$ and $\delta^{15}N$ contents in macro algae. SHB – Shelly Beach, TDC – Tudor Creek, and MPR – Mombasa Marine Park and Reserve, S – September, M – May.

In the MPR, organic soil and atmospheric sources were the primary contributors, with minor influences from wastewater near Nyali Beach, complementing the nutrients supplied through SGD established earlier. Seasonal variations were evident, as macro algae collected in September, during the dry season exhibited higher proportions of wastewater-derived organic matter compared to July, wet season ($15.9 \pm 4.3\%$ and $6.1 \pm 2.3\%$, respectively), which were more closely associated with terrestrial organic soils. The MPR area demonstrated lower nitrogen concentrations than creek waters, likely due to dilution with nutrient-deficient oceanic waters during high tides, necessitating reliance on atmospheric nitrogen sources for nutritional needs (Goosen *et al.*, 1997; Mwaure *et al.*, 2017). This study reveals that Mombasa's coastal waters receive significant nutrient inputs from various sources, primarily wastewater through terrestrial runoff, direct sewage discharges (e.g., Tudor Creek), and submarine groundwater discharge (e.g., marine park). Atmospheric deposition and sediment resuspension also contribute to the nutrient budget. The dominant nutrient sources and pathways vary by location, highlighting the need for localized assessments. While Mombasa shares similarities with other coastal cities in the region (Machiwa, 2010), it exhibits unique characteristics, such as the significant role of hospital sewage in nutrient inputs. These findings indicate the importance of site-specific studies and comprehensive approaches when developing effective nutrient management strategies for coastal ecosystems.

5.5.3. *Implication of nutrients and organic matter distribution in Mombasa*

Groundwater contamination poses a significant challenge in Mombasa, with nutrient levels in wells, particularly near slum areas, exceeding international standards ($\text{NH}_4^+ = 1.5 \text{ mg/L}$ ($83 \mu\text{M}$); and $\text{NO}_x^- = 50 \text{ mg/L}$ ($806 \mu\text{M}$) (U.S. EPA, 2000; WHO., 1996). This indicates potential septic tank contamination and an increased risk of waterborne diseases, highlighting the need for improved nutrient management strategies. Tudor Creek exemplifies the region's nutrient flux dynamics, with substantial fluctuations in DIN and phosphate concentrations, driven by increased runoff and anthropogenic influences. These fluctuations can enhance primary productivity, potentially leading to ecological shifts such as declines in coral reefs and seagrasses due to macroalgal proliferation (Lapointe *et al.*, 2010; Mwaura *et al.*, 2017; Okello *et al.*, 2022). High turbidity, especially like those established in Tudor creek, are also known with reduction of light penetration, limiting photosynthesis and affecting community structures and aquatic food webs (Berdalet *et al.*, 2022). The ecological impacts of SGD can significantly impact coastal ecosystems by altering nutrient dynamics, potentially promoting algal blooms and influencing species diversity through salinity changes (Cheng *et al.*, 2020; Lecher & Mackey, 2018). Previous studies in the region have shown that SGD reduces seagrass diversity due to continuous salinity reductions (Kamermans *et al.*, 2002). Elevated ^{222}Ra activities correlated with increased chlorophyll *a* suggest SGD enhances nutrient supply in oligotrophic waters (Honda *et al.*, 2018). However, further long-term studies are needed to verify these correlations in Mombasa and assess potential threats to critical habitats like corals and seagrasses.

The impact of sewage discharge on nutrient levels in Mombasa aligns with findings from other developing coastal cities, such as Mumbai, India, where untreated sewage significantly contributes to coastal eutrophication (Machado & Imberger, 2014). The ecological quality status of Mombasa's coastal waters is concerning, with DIN, N: P, and N: Si ratios classified as '*Bad*' and DIP as '*Moderate*' in both tidal creeks and adjacent oceanic waters. Wastewater, primarily from sewage, is a major nutrient source, highlighting the need for improved management, especially in slum areas. Implementing targeted, area-specific conservation efforts and adaptive management is crucial to address varied terrestrial nutrient inputs during wet seasons. Enhanced monitoring programs and comprehensive stakeholder collaboration, involving both catchment communities and coastal managers, are essential for sustainable nutrient management in Mombasa's coastal waters, as inputs originate from both inland sources and direct discharges.

5.6. Conclusion

This study investigated the distribution, sources, and pathways of nutrients and organic matter in Mombasa's coastal waters, revealing significant spatial and temporal patterns that affect ecosystem health and pose management challenges. The analysis highlighted marked spatial heterogeneity in nutrient patterns, with Tudor Creek exhibiting higher nutrient loads compared to Kilindini Creek and open waters, indicating localized anthropogenic pressures primarily from sewage discharge. Nutrient concentrations in Mombasa's waters were notably elevated compared to those reported in the western Indian Ocean's open waters, suggesting substantial local inputs.

This study identified multiple nutrient sources in Mombasa's coastal waters, including groundwater, sewage discharge, and atmospheric deposition. Groundwater significantly contributed to DIN levels, while sewage discharge elevated ammonium and phosphate concentrations. Atmospheric deposition, especially in high-traffic areas, increased phosphate and nitrate inputs. Macroalgae N and C isotopic results support the dominance of terrestrial wastewater in nutrient supply, with contributions from organic soils and atmospheric inputs in open nearshore areas. This pioneering study on submarine groundwater discharge in Kenya's coastal waters highlights its significant role in nutrient fluxes within Mombasa Marine Park and Reserve, emphasizing the need for comprehensive nutrient management strategies.

Mombasa's coastal waters experiences nutrient imbalances from terrestrial runoff, with seasonal increases in DIN during wet seasons, alongside silicate and phosphate limitations, likely due to enhanced runoff and anthropogenic inputs. The ecological quality status of Mombasa's coastal waters is classified as '*Bad*' based on DIN, N: P, and N: Si ratios, while dissolved inorganic phosphorus levels are rated as '*Moderate*' in tidal creeks and adjacent oceanic areas. Additionally, nutrient concentrations in Mombasa's groundwater often exceeded WHO standards, especially in human-affected areas, posing potential risks to human health.

These findings highlight the interconnectedness of terrestrial and marine systems in nutrient dynamics within Mombasa's coastal ecosystem, revealing that spatial and temporal variability reflects a complex interplay between natural processes and anthropogenic influences. This comprehensive understanding of nutrient dynamics provides crucial insights on distribution, sources and pathways of nutrients, which are critical for developing targeted management strategies to mitigate eutrophication risks and preserve the ecological integrity of Mombasa's coastal waters. Future research should prioritize long-term monitoring of nutrient dynamics, including top-down control mechanisms, across

seasons. This approach will inform site-specific management plans to address pollution and mitigate eutrophication risks, ultimately safeguarding the ecological integrity of marine protected areas. Enhancing monitoring programs and fostering stakeholder collaboration will support targeted conservation efforts against nutrient inputs from the diverse sources.

Chapter Six: General Discussion

6.1. Chapter Summary

This chapter synthesizes findings from [Chapter 2](#) to [Chapter 5](#) addressing objectives from [Chapter 1](#). It compares study outcomes with similar research, highlighting unique contributions and confirming existing knowledge. The chapter explores broader implications, examining theoretical and practical consequences of the findings. Based on comprehensive results, it provides actionable management recommendations and implementation strategies. Concluding with key takeaways, it emphasizes the research's significance in its field. This synthesis bridges detailed chapter analyses and the study's overarching importance, offering a holistic view of the research's contributions. It informs future management practices and research directions, serving as a crucial link between individual findings and their collective impact.

6.2. Introduction

Essential nutrients are pivotal in shaping aquatic food webs, influencing coastal biodiversity and phytoplankton communities, while fundamentally determining marine ecosystem composition, dynamics, productivity, and resilience ([Glibert, 2017](#); [Glibert *et al.*, 2018](#)). One of the aims for studying nutrient dynamics is to enhance our understanding on ecosystem functions under current environmental conditions and predict coastal habitats' vulnerability to increasing anthropogenic activities and associated nutrient inputs ([Adams *et al.*, 2020](#); [Glibert, 2020](#)). Such a study examines the effects of enrichment on ecosystem health, including coastal eutrophication, habitat losses, phytoplankton community shifts, and harmful algal bloom development ([Heisler *et al.*, 2008](#); [Hoegh-Guldberg *et al.*, 2017](#); [Malone & Newton, 2020](#)). It also assesses economic impacts on public health, fisheries and aquaculture, providing a comprehensive understanding of coastal ecosystem changes ([Berdalet *et al.*, 2016](#); [Hoagland *et al.*, 2002](#); [Landrigan *et al.*, 2020](#)). These studies on nutrient dynamics and associated effects provide critical insights for developing water quality management strategies and informing coastal conservation decision-making. This thesis research aimed to assess anthropogenic nutrient dynamics and their ecological impacts in Kenya's coastal waters, focusing on phytoplankton community structure and the development of potentially harmful microalgae. It aimed to achieve four specific objectives. Firstly, it sought to assess the spatio-temporal distribution of anthropogenic nutrients and phytoplankton assemblages in Kenya's coastal waters. Secondly, the research focused on investigating the distribution, sources, and pathways of anthropogenic nutrients in these coastal waters, with a particular emphasis

on the Mombasa region. Thirdly, the study aimed to determine the composition and distribution of potentially harmful microalgae species in Kenya's coastal waters, as well as evaluate the risks associated with bloom formation. Lastly, it examined the relationship between the compositions of potentially harmful microalgae species with environmental factors in Kenya's coastal waters. This comprehensive one-year study along Kenya's coast off the Western Indian Ocean (WIO) region achieved its objectives through a structured approach. This thesis dissertation study, divided into four chapters, encompassed a literature review ([Chapter 2](#)), *in situ* sampling, field surveys, and laboratory analyses of nutrients and phytoplankton communities ([Chapter 3](#)). It also included identification and quantification of potentially harmful microalgal species and phycotoxins (domoic acid and saxitoxin) in [Chapter 4](#). Whilst [Chapter 5](#) focused on analyzing organic carbon and nitrogen isotopic ratios, and biogeochemical parameters (^{222}Rn , TA and, DIC) to determine nutrient sources and pathways. Each chapter addressed specific aspects of coastal ecosystem dynamics, providing a thorough examination of the region's ecological processes.

6.3.Nutrient distribution in Kenya’s coastal and marine waters

Uncontrolled coastal development and catchment activities significantly influence nutrient dynamics in adjacent coastal waters and estuaries at river discharge points due to urbanization and agricultural intensification ([Glibert, Harrison, *et al.*, 2006](#); [Glibert & Burkholder, 2006](#)). This PhD thesis revealed peak nutrient concentrations in estuaries and tidal creeks near urban areas and nearshore sites adjacent to hotels ([Chapter 3](#) and [5](#)), demonstrating how activities in catchment areas and adjacent coastal zones significantly influence coastal water nutrient composition. Creeks around Mombasa, particularly Tudor Creek, exhibited elevated turbidity and organic matter levels ([Chapter 5](#)), potentially degrading coastal ecosystem quality. High turbidity reduces light penetration and increases water temperature, while excess organic matter from sewage can deplete dissolved oxygen during decomposition. These conditions stress marine organisms, potentially leading to biodiversity loss and damage to benthic habitats like seagrass and corals ([Jones *et al.*, 2021](#)).

Nutrient conditions varied seasonally. Dry season saw higher phosphate levels at sewage discharge points, while wet season brought increased dissolved inorganic nitrogen, primarily as ammonium. The overall nutrient levels peaked in upper tidal creek waters, negatively correlating with downstream salinity ([Chapters and 5](#)) indicating terrestrial sources. The study detected dissolved ammonium contamination, suggesting anthropogenic inputs or altered nitrogen-cycling processes by supplying re-

duced forms of nitrogen (IOC-UNESCO, 2008). Analysis of different ecosystems revealed higher pollution in creeks and estuaries compared to nearshore and oceanic waters with coral reefs and seagrass habitats; however, these habitats were established with 'Bad' ecological state, especially during wet season. Additionally, Porewater in sewage-impacted sites and groundwater in informal settlements showed ammonium and DIN contamination, respectively, posing potential health risks and necessitating improved nutrient management. Contradictory results in nutrient stoichiometry were observed, with in situ surveys revealing phosphate and silicate limitations, contrasting with literature-reported nitrogen limitation (Chapter 2, 3, and 5). This suggests that anthropogenic nutrient inputs are changing nutrient composition, with anthropogenic enrichment causing DIN-rich eutrophic coastal waters, consistent with similar studies' conclusions (Glibert, 2017; Kemp *et al.*, 2005; Murrell *et al.*, 2007). The widespread phosphate and silicate limitation identified across the examined sites aligns with recent regional studies (Sekadende *et al.*, 2021). This limitation potentially constrains primary productivity in the coastal waters. Mombasa exhibited nutrient co-limitation, with P and Si limiting during the wet season and N limiting in August-September. This pattern aligns with studies on macronutrient imbalances in urbanized coastal waters caused by sewage input (Oduor *et al.*, 2023). Mombasa's coastal waters exhibit elevated nutrient concentrations, mirroring issues in other rapidly urbanizing coastal cities. In the Western Indian Ocean, Dar es Salaam and Maputo face similar challenges with high dissolved inorganic nitrogen and phosphorus levels (Mangion *et al.*, 2016; Langa *et al.*, 2021). Globally, Manila Bay and Guanabara Bay show comparable or higher nutrient concentrations (Jacinto *et al.*, 2011; de Souza Barros *et al.*, 2017). This widespread problem in developing coastal areas with inadequate wastewater management underscores the urgent need for improved sewage treatment and adaptive conservation strategies to protect coastal ecosystem health

6.4. Association between nutrients and phytoplankton assemblages

Phytoplankton, as the foundation of most marine food webs, plays a critical role in oceanic biogeochemical cycles and serves as a key indicator of coastal water quality and ecosystem health (Napiórkowska-Krzebietke, 2017). The species diversity and *Chl-a* concentrations were higher during the dry season across all ecosystems, including coral reefs, characterized by low nutrient concentrations and turbidity, indicating good water quality supportive of diverse phytoplankton species (Chapter 2 and Chapter 3). However, estuaries exhibited higher species abundance and productivity but lower species richness compared to other ecosystems, suggesting deteriorated water quality and environmental stress favoring short-lived, stress-tolerant species (Clark *et al.*, 2015). Shannon-Wiener indices

ranged from 0.54 to 3.37 bits/cell, comparable to regional upwelling waters (Sekadende *et al.*, 2021), with lower values in estuarine environments. Estuaries globally exhibit low species richness, often due to salinity stress rather than nutrient availability. The fluctuating salinity, turbidity, and nutrients in these environments favor the productivity of a few resilient species adapted to these challenging conditions (Jones *et al.*, 2021).

The composition of nutrients available significantly influence phytoplankton community dynamics, particularly regarding harmful algal species, as certain species thrive under specific nutrient conditions. It is known to promote the development of harmful algal blooms (Glibert, 2020; Glibert *et al.*, 2007; IOC-UNESCO, 2008). During this study, a shift from natural nitrogen-limited conditions reported in literature in Chapter 2 (Oduor *et al.*, 2023) to a DIN-enriched state, with phosphorus and silica becoming limiting factors Chapter 3 (Oduor *et al.*, 2024). Additionally, nutrient concentrations correlated with increased blooms of potentially harmful species like *Karenia brevis* and *Dinophysis* spp., peaking during rainy seasons due to increased runoff and terrestrial nutrient supply in anthropogenically impacted areas. A shift from natural nitrogen-limited conditions to a DIN-enriched state with phosphorus and silica limitation was observed, aligning with global trends of nutrient enrichment effects on phytoplankton communities (Murrell *et al.*, 2007). This change in nutrient composition significantly influenced phytoplankton community dynamics, particularly regarding potentially harmful microalgae species, which increased during wet season (Chapter 4). The study found widespread low abundance of potential toxin-producing HAB species in Kenya's coastal waters, indicating vulnerability to harmful algal blooms under ammonium-dominated conditions limited by phosphorus and silicate (Glibert *et al.*, 2004). These findings reveal nutrient contamination and potential shifts in phytoplankton composition towards harmful microalgae, driven by high ammonium supply and P and Si limitation. This alteration may impact ecosystem health and coastal productivity, suggesting the necessity for continued water quality monitoring and management to mitigate ecological and socioeconomic consequences.

6.5.Sources and Pathways of nutrients in Kenya's Coastal Waters

Assessing nutrient sources and pathways in coastal waters is crucial for developing effective management strategies to mitigate eutrophication and protect marine ecosystems (Saito *et al.*, 2018). Kenya's coastal waters face significant nutrient enrichment from diverse anthropogenic sources, with urban areas contributing disproportionately through untreated sewage discharge and industrial effluents

(Chapter 2, 3 and 5). Tudor tidal creek exhibited substantially higher nutrient concentrations than surrounding areas, primarily due to sewage discharges and wastewater (Chapter 5). Similar to previous observations in the area (Ohowa, 1996; Ongore et al., 2013), seasonal variations showed significant increases in nutrient concentrations during wet periods, highlighting the role of surface runoff in transporting terrestrial nutrients. Agricultural runoff, especially during rainy seasons, contributed to elevated nitrogen and phosphorus levels (Chapter 2, 3 and 5). The terrestrial influence was further indicated by decreasing POC: PON ratios downstream in the tidal creeks (detailed in Chapter 5). Groundwater in informal settlements also showed high DIN concentrations, serving as both a source and pathway through SGD, consistent with studies in similar settings (Adyasari et al., 2018; Lemley & Adams, 2020). Additionally, submarine groundwater discharge emerged as a significant nutrient pathway, with flux rates of 23.35 to 392.87 L/m²/day for DIN, phosphate, and silicate (Chapter 5), comparable to global coastal areas (Santos et al., 2021). However, limitations in wind speed data, tidal flux, and diffusion calculations suggest the need for further study to verify SGD's role and nutrient supply in the area. Atmospheric deposition and nitrogen fixation were also established as potential nutrient sources in Mombasa's coastal areas. High-traffic zones contribute to nitrogen deposition, while nearshore areas benefit from atmospheric nitrogen fixation (Chapter 2 and Chapter 5), supporting earlier findings on nutrient dynamics in coastal regions (Elser et al., 2009; Herbert, 1999). This thesis study reveals complex nutrient dynamics in Kenya's coastal waters, with both anthropogenic and natural sources contributing through direct discharges and diffuse pathways, including SGD. This complexity, particularly evident in Mombasa, necessitates a multi-faceted management approach. Future strategies should address individual sources and their interconnections while also considering nutrient utilization by primary producers, which was not considered in the present study. This comprehensive approach is crucial for effectively reducing nutrient pollution and protecting valuable marine ecosystems in Mombasa and similar coastal regions.

6.6. Ecological Implication of Nutrient Conditions in Kenya's Coastal Waters

Kenya's coastal waters face significant ecological challenges due to nutrient pollution, particularly during wet seasons (Chapters 2, 3 & 5). This nutrient enrichment, characterized by elevated DIN in the form of ammonium in silicon- and phosphorus-limited waters, creates eutrophic conditions that are known to foster the growth of potentially harmful microalgae (Glibert, 2017; Glibert et al., 2018). Seasonal variations in nutrient levels coincide with increased abundance of potentially harmful microalgae, especially in nutrient-rich areas like creeks and estuaries (Chapter 2 and Chapter 3). While

abundances remained below bloom thresholds during the study period, the widespread distribution of toxin-producing species signals a risk for future HAB events (Chapter 4; Oduor *et al.*, 2024). Notably, some taxa like *Alexandrium* spp. and *Pseudo-nitzschia pungens* have reached densities of 60,000 cells/L in previous studies (Hansen *et al.*, 2001; Chapter 2). The study identified 59 taxa of potentially harmful microalgae from 40 genera, with 47% capable of toxin production, indicating a substantial risk to marine ecosystems and human health (Chapter 3 and 4).

This thesis also revealed occurrence of DIN contamination in Kenya's coastal marine waters with 90.1% of sites exhibiting 'poor' to 'bad' ecological quality status (EQS). Sites near in nearshore areas close to hotels, tidal creeks, and estuarine areas showed marked EQS deterioration during rainy periods (Chapter 3), indicating the strong influence of terrestrial runoff and anthropogenic activities on coastal water quality. The 'poor' EQS, marked by low oxygen, high temperature, salinity, and ammonium, plus potential biotoxin-producing phytoplankton, indicates multiple environmental stressors. This highlights the intricate link between water quality deterioration and marine ecosystem health. This situation aligns with global trends where increasing anthropogenic pressures and changing environmental conditions raise concerns about HAB events and the survival of vulnerable coastal species like sea turtles (Dai *et al.*, 2023; Glibert, 2017; Van Houtan *et al.*, 2014). Nutrient pollution through submarine groundwater discharge in Mombasa Marine Park & Reserve also poses significant ecological risks. The SGD-supplied nutrients and reduced salinity can alter nutrient conditions, potentially fostering harmful algal growth. This may negatively impact coral and seagrass development, jeopardizing the protected area's ecosystem integrity (Kirkman *et al.*, 2020; Mwaura *et al.*, 2017; Obura *et al.*, 2017).

6.7.Limitations, Gaps, and Future Directions

This thesis on nutrient conditions and ecological implications in Kenya's coastal waters offers valuable insights despite several limitations. While the study's focused approach and comprehensive spatial coverage of 44% of Kenya's coastline provide significant data, certain constraints warrant consideration. The exclusion of smaller microalgae (<20 µm) and macro algae may underestimate the full range of toxic and harmful algal species, potentially leading to an incomplete understanding of ecosystem health and risks. Monthly sampling might have missed short-lived bloom events (Cloern, 1996), and the one-year duration limits the capture of long-term trends and seasonal variations in bloom dynamics and nutrient fluxes. Methodologically, the focus on bottom-up control without considering top-down influences may oversimplify ecosystem dynamics, potentially misrepresenting the complex ecological

processes governing nutrient cycles and phytoplankton composition. Geographical gaps, including un-assessed HAB-prone areas like the North Kenya Banks, prevent a comprehensive evaluation of HAB prevalence across Kenya's entire coastal waters limiting the study's ability to inform coastal management strategies for all areas. Regulatory challenges under NAGOYA protocols limited phytotoxic analysis to water samples, excluding fish despite suggested contamination.

These limitations have implications for the study's outcomes, potentially underestimating HAB risks and restricting generalizability. However, they also present opportunities for future research. Addressing these constraints in subsequent studies could enhance our understanding of coastal marine ecosystems, leading to more robust environmental management strategies. Future directions may include longer-term studies, more frequent sampling, broader geographical coverage, and comprehensive analysis of algal species and ecosystem interactions. To address the identified limitations and enhance future research, this thesis proposes the following approaches:

6.7.1. Modelling and predictions of future bloom formation risks

Modelling offers a promising approach to forecast future harmful algal blooms (HABs) in Kenya's coastal waters, addressing data deficiency in resource-limited regions (Glibert, 2020). By leveraging baseline data on potentially harmful microalgae and environmental drivers, researchers can develop predictive models for future blooms. This approach has proven effective in various locations, utilizing environmental variables and nutrient inputs to compensate for limited spatio-temporal data collection capabilities. For example, the Harmful Algal Bloom Forecast System has successfully predicted *Karenia brevis* blooms in the Gulf of Mexico (Stumpf *et al.*, 2003; Tomlinson *et al.*, 2009). Additionally, machine-learning approaches have forecasted microcystin concentrations in Lake Erie (Zhou *et al.*, 2023), while artificial neural networks have modelled *Alexandrium minutum* blooms in the Mediterranean Sea (Zingone *et al.*, 2021). Beyond HABs, models have also demonstrated efficacy in addressing nutrient hazards, as seen in the Tampa Bay Coastal Ocean Model's simulation of nutrient-rich wastewater plumes from the Piney Point discharge (Liu *et al.*, 2024). These examples highlight the potential for similar methodologies to be applied in Kenya's coastal waters, enhancing our ability to predict and manage future blooms since several potential species were established during the study.

The integration of newly acquired in-situ data from the Kenyan coast into predictive models is crucial. Advanced modelling techniques such as structural equation modelling (SEM) can be used to explore complex interactions between environmental factors, phytoplankton species and potentially harmful

microalgae. Additionally, machine learning can be employed for real-time predictions of eutrophication and harmful algal bloom occurrences, utilizing established relationships between environmental variables and phytoplankton assemblages that align with global trends (Chapters 3 and 4; Glibert *et al.*, 2018). Furthermore, incorporating data on atmospheric deposition and groundwater inflow—significant nutrient sources for harmful algal blooms—into these models will enhance their accuracy. Implementing advanced modelling techniques can enhance Kenya's ability to predict and mitigate harmful algal blooms, improving coastal water management and early warning systems despite resource limitations, ultimately benefiting marine ecosystems and coastal communities.

6.7.2. *Nutrient budget in the Tidal Creeks, Estuaries and other critical habitats*

Conducting a comprehensive nutrient budget is crucial for effectively managing nutrient pollution and related issues in Kenya's coastal waters. This approach addresses current study limitations and provides extensive data for informed decision-making. While this thesis offered valuable insights into nutrient distribution from various land-based sources, including direct sewage discharges, submarine groundwater discharge, porewater, rainwater, and groundwater, its snapshot nature and limited sampling frequency are insufficient for deriving conclusive management strategies. Given the dynamic nature of coastal ecosystems, a more comprehensive and long-term nutrient budget analysis is necessary to develop effective pollution management strategies.

Nutrient budgets provide comprehensive ecosystem insights, enabling targeted pollution mitigation strategies through detailed analysis of nutrient sources, sinks, and fluxes, aiding its application globally. For instance, in the southern North Sea, researchers employed an interdisciplinary approach to study pollution pathways and develop abatement strategies, analyzing hydrodynamic processes and using numerical models to examine particle transport (Meyerjürgens *et al.*, 2023). In southeastern China, researchers characterized nitrogen and phosphorus budgets in coastal aquaculture ponds, providing valuable insights into nutrient dynamics and environmental impacts (Yang *et al.*, 2021). Nutrient budgets are essential for Kenya's coastal waters, especially in urban tidal creeks, hotel-adjacent nearshore areas, and estuaries, which contain vital habitats including coral and seagrass ecosystems that need thorough evaluation of nutrient sources and sinks. Nutrient budgets in Kenya's coastal waters will enable managers to identify major contributors and develop targeted mitigation strategies. By quantifying inputs from various sources, understanding seasonal variability, and assessing outputs and nutrient balance, this approach informs site-specific interventions and supports the implementation of continuous water quality monitoring systems.

A comprehensive nutrient budget for Kenya's coastal waters should account for various sources, sinks, and fluxes within the ecosystem. It must consider temporal variability, spatial heterogeneity, system hydrodynamics, and internal cycling. The budget should also incorporate complex interactions between different nutrient species and biological components for a holistic understanding. Key nutrient inputs include river discharges, urban sewage, agricultural runoff, atmospheric deposition, and groundwater seepage (Ohowa *et al.*, 1997; Okello *et al.*, 2020; Okuku *et al.*, 2011). Nutrient sinks comprise sediment burial, limited denitrification, and uptake by marine organisms (Kitheka *et al.*, 2005). Internal cycling involves remineralization of organic matter in sediments and upwelling of nutrient-rich bottom waters (Jacobs *et al.*, 2020; Nguli *et al.*, 2006). Current research has revealed elevated nutrient levels in urban creeks and estuaries, seasonal variations in concentrations, and spatial variability across different coastal ecosystems, indicating areas of need for intervention. Furthermore, this thesis has identified major nutrient sources, including rivers and sewage, atmospheric deposition and groundwater, while sediments are recognized as both sinks and potential sources in Mombasa (Chapters 2 and 5). Future Mombasa research should quantify sewage and seasonal river discharges, along with nutrient utilization and sinking rates, to establish comprehensive water column nutrient budgets in the critical coastal habitats.

6.7.3. *Management interventions; Integrated Coastal Zone Management*

Integrated Coastal Zone Management (ICZM) offers a comprehensive approach to address nutrient pollution, eutrophication and harmful algal blooms (HABs) in Kenya's coastal waters (Boesch, 2019, Elmgren *et al.*, 2015). The current study lacks data for formulating a holistic management strategy that accounts for the spatio-temporal variability of nutrient contamination along Kenya's coast. ICZM can effectively tackle this issue by implementing site-specific interventions and recognizing the diverse challenges faced by different coastal ecosystems.

While Kenya has already adopted an ICZM Policy, which provides a framework for sustainable management of coastal and marine resources (NEMA-Kenya, 2007), its application seems to be limited based on the findings of this thesis and other related studies (Mwaura *et al.*, 2017). This approach is particularly appropriate for Kenya's waters due to the varying sources of nutrient pollution, including river discharges, urban sewage, and agricultural runoff (Okello *et al.*, 2022). By zoning coastal areas, ICZM allows for independent evaluation and implementation of mitigation strategies tailored to each ecosystem's unique environmental exposures given the variability of nutrients sources and pathways.

The substantial nutrient influx into coral reef areas within Mombasa Park ([Chapter 3](#) and [4](#)) underscores the importance of ICZM in effectively addressing and mitigating nutrient pollution to protect coastal ecosystems. ICZM has been successfully applied in other regions, such as the southern North Sea, where researchers employed an interdisciplinary approach to study pollution pathways and develop abatement strategies ([Meyerjürgens et al., 2023](#)). In Kenya, this approach can help address the localized nutrient pollution observed in estuaries and creeks adjacent to urban areas ([Oduor et al., 2024](#); [Oduor et al., 2023](#)). ICZM integrates community engagement, continuous monitoring systems, and public awareness campaigns to foster long-term environmental stewardship and provide valuable data for effective decision-making on management of coastal areas. This will facilitate designation and implementation of comprehensive policies to address nutrient pollution from all sources, effectively protecting critical coastal habitats in the region.

6.8. Conclusion

The conservation and management of anthropogenic nutrient enrichment in Kenya's coastal waters are critical, influenced by varying hydrological and human activities affecting these habitats. This comprehensive study of Kenya's coastal waters reveals critical insights into nutrient dynamics, phytoplankton assemblages, and the distribution of potentially harmful microalgae species. Elevated nitrogen and phosphorus levels were observed, especially during wet seasons, attributed to sewage from domestic, municipal, and hotel sources, particularly around Mombasa and nearshore hotel sites, suggesting urbanization's impact on coastal ecosystems.

The study documented 59 potentially harmful microalgae species across 40 genera, with 48% capable of producing toxins. Tidal creeks and estuaries emerged as hotspots for potential toxin-producing genera like *Alexandrium* and *Pseudo-nitzschia*. Although abundances remained below bloom thresholds, the widespread distribution indicates a risk for future harmful algal blooms. Environmental factors, particularly temperature, dissolved oxygen, ammonium, and phosphates, significantly influenced occurrences of 44% of potential toxin-producing species in Kenya's coastal waters, indicating that future changes in these parameters are likely to promote HAB development. Increasing temperature, ammonium, and phosphates, coupled with reduced dissolved oxygen, may foster the growth of these potentially toxic harmful microalgae species, posing risks to marine ecosystems, public health, and coastal economies.

This thesis reveals that Kenya's coastal waters, home to critical habitats and endangered species, face threats from nutrient pollution. The study demonstrates a correlation between nutrient enrichment and the proliferation of potentially harmful microalgae, highlighting the need for effective management strategies to avoid future blooms to safeguard coastal ecosystems and economies. By providing comprehensive data on nutrient variability, sources, pathways, and associated phytoplankton assemblages, this research contributes significantly to understanding the complex interactions within these ecosystems. These insights form a solid foundation for future studies in rapidly urbanizing coastal regions and offer valuable information for conservation efforts. The findings can guide the development of appropriate methods to manage nutrients and mitigate harmful algal blooms, thereby protecting these ecologically significant areas and their biodiversity.

6.9.Recommendations

Based on the findings of this thesis, and successful interventions in similar coastal ecosystems, the following recommendations are proposed to address and mitigate the impacts of nutrient pollution impacts in Kenya's coastal waters:

1. *Enhance Monitoring Systems:* Implement continuous water quality monitoring, particularly in tidal creeks, estuaries, and near hotels where elevated nutrient levels were observed. This will provide valuable data for decision-making and help track seasonal variations in nutrient loading. This can also be integrated with remote sensing technology to enable efficient identification of nutrient enrichment hotspots and potential harmful algal blooms across larger spatial scales (Klemas, 2011).
2. *Improve Wastewater Management:* Develop targeted strategies focusing on areas with high nutrient concentrations, such as Tudor tidal creek and informal settlements. Upgrade and expand wastewater treatment facilities, implementing tertiary treatment to remove nutrients before discharge (Wurtsbaugh *et al.*, 2019). This addresses the major contributors to nutrient pollution from hospital identified in the study.
3. *Implement Green Infrastructure:* Besides the improvement of wastewater management, this thesis recommends installation of bioswales and rain gardens to filter nutrients from urban runoff (Davis *et al.*, 2009). This approach will help mitigate nutrient supply through surface runoff, a significant pathway identified in the study during the wet season.

4. **Address Submarine Groundwater Discharge:** I recommend monitoring and managing groundwater quality, especially in areas with high SGD. The government should develop and enforce regulations on septic system maintenance and implement constructed wetlands to intercept and treat groundwater before it reaches coastal waters ([Knapp et al., 2020](#)). I also recommend conducting further studies to accurately quantify the contribution of SGD in sensitive habitats like coral reefs and seagrass beds, using seepage meters and geochemical tracers such as radium and radon isotope.
5. *Engage Local Communities:* Initiate citizen science programs for reporting new direct sewage and wastewater discharge areas and nutrient-associated habitat changes. This will enhance monitoring efforts, raise awareness, and foster community involvement in environmental stewardship ([Conrad and Hilchey, 2011](#)).
6. *Launch public awareness campaigns* addressing open defecation in mangroves and untreated sewage discharge, particularly in urban areas and informal settlements near Tudor Creek. Educate communities on sustainable sewage management practices and the threats of nutrient and wastewater pollution. These efforts can foster community involvement in environmental stewardship and reduce nutrient inputs to coastal waters.
7. This thesis recommends adopting the DPSIR (Drivers, Pressures, State, Impact, Response) framework to comprehensively assess and manage nutrient enrichment and harmful algal blooms in Kenya's coastal waters ([Surridge & Watson, 2012](#)). This multidisciplinary approach is particularly suitable for addressing complex environmental issues in coastal areas, including urbanized Marine Protected Areas like Mombasa, Malindi, and Watamu. The DPSIR framework can effectively link large-scale human drivers of change with their impacts on coastal systems and guide management responses ([Newton et al., 2020](#)). The implementation requires collaboration among coastal managers, local communities, hotel owners, researchers, and policymakers to identify and address nutrient sources from various origins, including river catchments, nearshore areas, and adjacent hotels. This approach will support blue economy development and sustainable development goals while addressing planned expansions in agriculture, maritime activities, and aquaculture ([Nyonje, 2018](#)).

List of Appendices

7.1. Chapter 2: Appendices

Appendix 1: Literature Review Search String_Process

Nutrients and harmful algal blooms in Kenya's coastal and marine waters: A Review	
Protocol for the systematic review	
Reviewer: Nancy A. Oduor	
Purpose	To examine the status and drivers of nutrient pollution and harmful algal blooms (HABs) development in Kenya's coastal and marine waters
Keywords and Synonyms	Nutrient* Phytoplankton Primary production HABs or Harmful algal blooms Coast* OR marine Kenya
Sources/ Digital libraries	Web of science
Search String	(((((TS=(Nutrient*)) OR TS=(phytoplankton)) OR TS=(primary production))) OR TS=(HABs)) OR TS=(Harmful algal blooms)) AND TS=(coast*) OR TS=(marine)) AND TS=(Kenya)
Inclusion criteria	<ul style="list-style-type: none"> ✓ Journal articles, book chapters, and technical reports with empirical data. ✓ Published in 2021 or earlier, ✓ Study focus on Kenya's coastal and marine waters
Exclusion criteria	<ul style="list-style-type: none"> ✗ Editorial and meeting proceeding articles ✗ Conference abstracts ✗ Lack of information on nutrients pollution, phytoplankton, and HABs

Search link: <https://www.webofscience.com/wos/woscc/summary/d83f84a3-0432-4836-a1b0-304f03aa9aa7-4c14e292/relevance/1>

Appendix 2: Sources of Primary Research Data for Systematic Literature Review

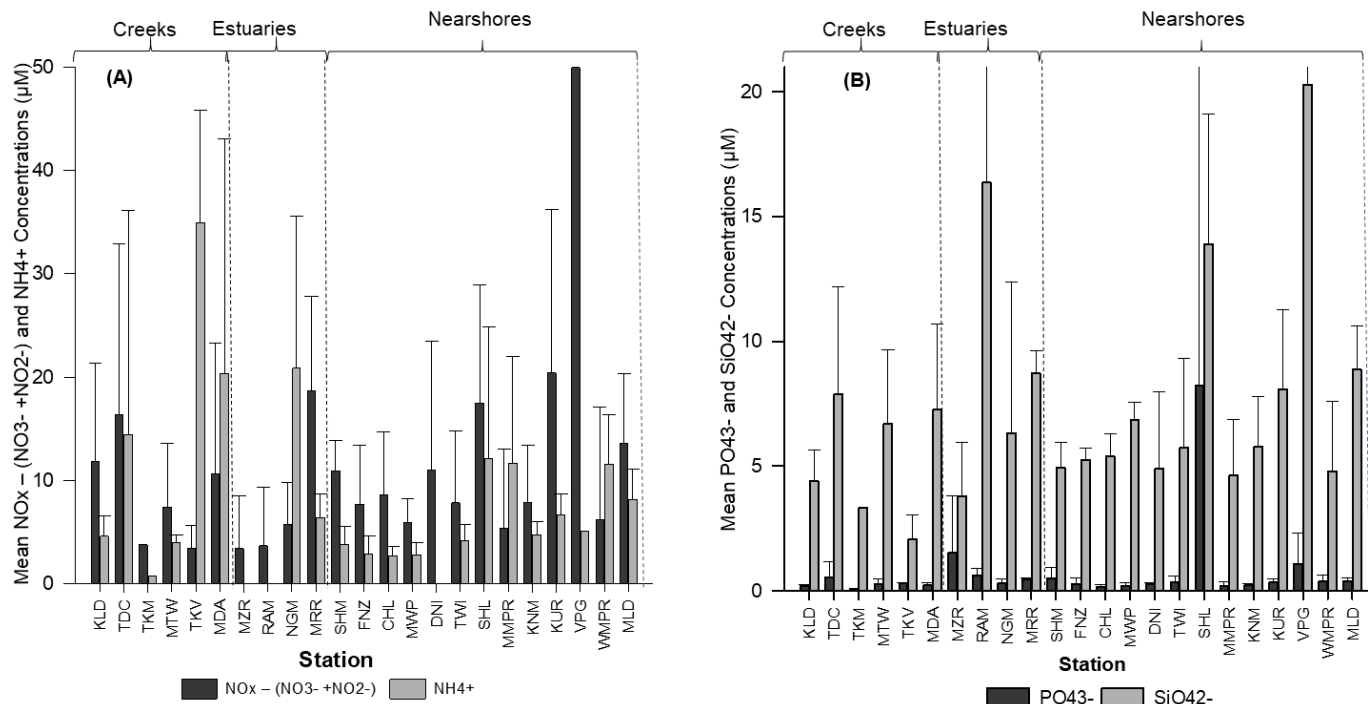
Publication	Year	Decade	Theme	Site	Parameters	Kenyan C	Tudor Cr	Makupa C	Kilindini	Shelly	Mtwapa	Gazi	Shirazi	Mida	Ngomeni	Sabaki	Tana	Kiwayu	Ramsi-Va	Kilifi	Mombasa	North Kei	Nyali	Bamburi	Vipingo	Nyali	Diani	Kisumu
1 Ochumba 1983	1983	1980s	Fisheries	Kenyan coast	NUT/PP	✓																						
2 Kazungu et al., 1989	1989	1980s	Pollution survey	Tudor Creek	NUT		✓																					
3 Okenwa and Rume	1990	1990s	Ecological/Quality survey	Tudor Creek	planktons	✓																						
4 Munga et al., 1992	1992	1990s	Aquaculture	Ngomeni	HABs										✓													
5 Osore 1992	1992	1990s	Ecological/Quality survey	Makupa creek	planktons			✓																				
6 Kihelika et al., 1994	1996	1990s	Pollution survey Coral reefs, Seagrass, Mangroves	Gazi-Bay, Estuary	NUT/Planktons						✓																	
7 Ohova et al., 1996	1996	1990s	Pollution survey	Sabaki Estuary	NUT											✓												
8 Osore et al., 1997	1997	1990s	Ecological survey	Kilindini	planktons				✓																			
9 Magoni 1997	1997	1990s	Bathymetry	Mtwapa Creep	NUT						✓																	
10 Mwangi et al., 1998	1998	1990s	Pollution survey	Mombasa Marine Park	NUT															✓								
11 Wawire et al., 1999	1999	1990s	Public health	Kenyan waters	HABs	✓																						
12 Mwashole et al., 1999	1999	1990s	Groundwater pollution	Nyali and Diani Beaches	NUT											✓	✓								✓	✓		
13 Obura 2001	2001	2000s	Coral reefs	Kenya coast-Mombasa, Ali	NUT/Planktons	✓																						
14 Hansen et al., 2001	2001	2000s	Public health	Ngomeni, Kilifi, Kilindini Cr	HABs				✓						✓				✓									
15 Mwalume et al., 2003	2003	2000s	Fisheries	Mida creek	NUT/Planktons									✓														
16 Munga et al., 1993	2003	2000s	Pollution survey	Mombasa	NUT				✓																			
17 IUCN 2004b	2004	2000s	Public health	Kunga	HABs																	✓						
18 Barono et al., 2006	2006	2000s	Groundwater pollution	Kisumu-Mombasa	NUT																						✓	
19 Okuku et al., 2011	2011	2010s	Pollution Survey	Makupa, Tudor, Mtwapa an	NUT/PP		✓	✓			✓																	
20 Kiteresi et al., 2012	2012	2010s	Ecological/Quality survey	Ramisi-Vanga systems alor	NUT/Phyts														✓									
21 ASCLIME, 2012	2012	2010s	Ecoaytmen analysis	Kenyan waters	NUT/PP	✓																						
22 Onopore et al., 2013	2013	2010s	Pollution survey	R. Tana, Sabaki, Umta an	NUT/PP											✓	✓		✓									
23 Ogongo et al., 2015	2015	2010s	Aquaculture	Shirazi Creek	HABs/NUT								✓															
24 Ogongo et al., 2015	2015	2010s	Public health	Kilindini Creek	HABs/NUT				✓																			
25 Pole et al., 2016	2016	2010s	Pollution survey	Mtwapa Creek	NUT/PP						✓																	
26 Mwaura et al., 2017	2017	2010s	Coral reefs	Nyali, Bamburi, Vipingo, M	NUT																	✓	✓	✓	✓			
27 Okuku et al., 2019	2019	2010s	Pollution survey	Kilindini Port, Makupa, Kilin	NUT/PP		✓	✓	✓		✓					✓	✓		✓									
28 Jacobs et al., 2020	2020	2020s	Fisheries	North Kenya Banks	NUT/Phyts																	✓						
29 Okelo et al., 2022	2022	2020s	Coral reefs	Mtwapa Creek, MMPR	NUT/PP						✓										✓							
30 Mwashole et al., 2005	2005	2000s	Mangroves	Mida creek	NUT/Phyts									✓														
31 Mwaiguni et al., 2017	2017	2010s	Public health	Kisumu-Mombasa	NUT																						✓	
32 Hatiman 2017	2017	2010s	Pollution Survey	Madubaha, Shelly, Jommo	NUT		✓			✓											✓							
33 Mwaiguni 2002	2002	2000s			NUT																					✓		✓
34 Mengesha et al., 1999	1999	1990s	Phyts PP	Gazi, Sabaki, Kiwayu	NUT/PP																							
35 Kromkamp et al., 1997	1997	1990s	NUT and PP	Gazi, Sabaki, Kiwayu	NUT/PP							✓				✓		✓										
36 Bergman, 2001	2001	2000s		EACC	Phyts/Cyano	✓																						
37 Painter, 2020	2020	2020s		EACC	NUT	✓																						
38 Okuku et al., 2016	2016	2010s		Tana River	Planktons												✓											
39 Kiteresi, 2013	2013	2010s		Sabaki, Ramisi, Umta, Mw	HABs											✓			✓									
40 Kiteresi, 2013	2013	2010s	NUT	Sabaki, Ramisi, Umta, Mw	NUT											✓			✓									
						0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

7.2.Chapter 3: Appendices

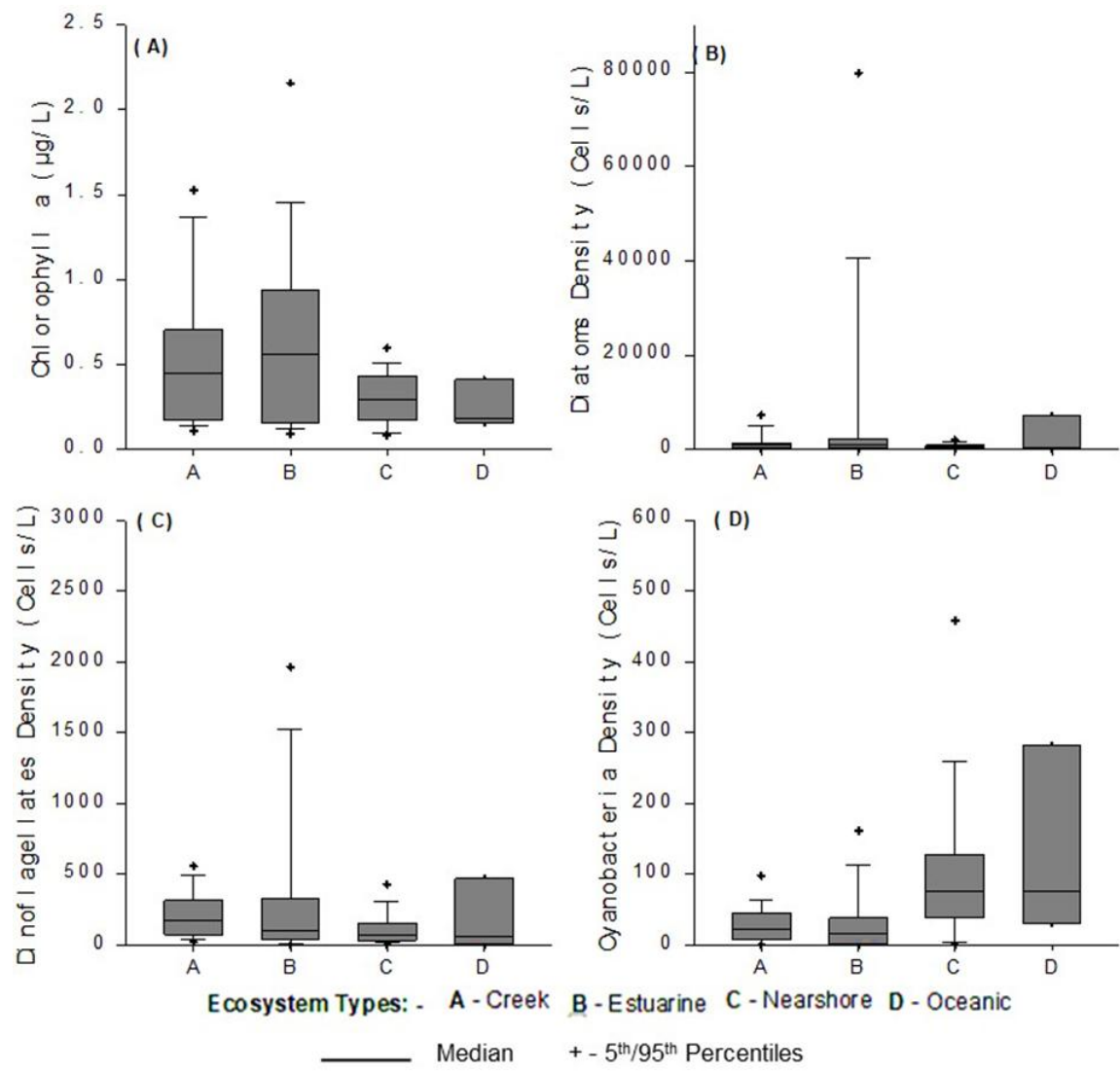
Appendix 3: Establishment of sampling sites

This study focused on sea turtle foraging sites, which were identified through Focus Group Discussions (FGDs) with local fishermen, community turtle conservation groups, and coastal resource managers. Participants were first introduced to sea turtle biology, population status, and management issues. They then identified areas where turtles were frequently or occasionally sighted, or no longer seen. These sites were ranked by sighting frequency and located on printed Google maps using markers. Sites were categorized into four hydrological types: creeks (saline tidal channels), estuarine (river mouths), near-shore (within 500m of the shoreline), and oceanic ($\geq 500\text{m}$ offshore). Sites exposed to anthropogenic pressures and accessible for ecological surveys were prioritized. Two sites where turtles were no longer sighted and three oceanic sites with high sightings were also included. Ground truthing was conducted to verify site coordinates, followed by ecological surveys.

Appendix 4: Dissolved inorganic nutrients (NO_x-Nitrate + Nitrite, Ammonium, Phosphate, and Silicates)



Appendix 5: Phytoplankton Composition in Turtle Foraging Sites Based on Ecosystem Types



Transform: Log($X+1$)
Resemblance: S17 Bray-Curtis similarity

The dendrogram illustrates the hierarchical clustering of 60 sites. The y-axis shows the similarity level from 0 at the top to 100 at the bottom. The x-axis lists the site abbreviations. Red dashed lines are overlaid on the dendrogram to highlight several distinct clusters, indicating groups of sites that are highly similar to each other.

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Appendix 7: Ecological health status of the turtle foraging Sites

List of sites and the ecological quality status (EQS) based on different parameters Dissolved Oxygen – DO, Dissolved inorganic nitrogen-DIN, phosphate-DIP, DIN: DIP ratios (N: P), DIN to silicate ratios (N: Si), phytoplankton biomass (*Chl a*) and Shannon-Weiner diversity (Div). The EQS- H – High, G – Good, M – Moderate, P – Poor, and B – Bad. For Diversity – CW – Clean Water, MP – Moderate Pollution, and HP – Highly Polluted

Month	Station	Stn. Code	Site	Site Code	DO	DIN	DIP	N: P	N:Si	Chl a	DIV
S/M	Shimoni	SHM	Jetty	SM-JT	M	B	P	M	P	G	MP
S/M	Vipingo	VPG	Bandari ya Vipingo	VP-BV	G	B	B	B	B	H	MP
S/M	Kanamai	KNM	Barawa	KN-BR	M	B	M	B	B	H	MP
S/M	Tiwi	TWI	BAT	T-BAT	M	P	M	B	B	G	MP
S/M	Funzi	FNZ	Black Marine	FZ-BM	M	B	G	B	M	G	MP
S/M	Mtwapa	MTW	Chamba cha adam	MTCC	M	B	G	P	M	H	MP
S/M	Malindi	MLD	Chamba Cha Kambani	ML-CH	G	B	M	P	M	G	MP
S/M	Funzi	FNZ	Chemchem	FZ-CC	M	P	M	M	M	G	MP
S/M	Chale	CHL	Shimo la Mlango	CH-SM	M	B	G	B	P	H	MP
S/M	Mombasa MP & R	MPR	Coral beach	M-CB	M	B	G	P	M	G	MP
S/M	Tudor Creek	TDC	Fort Jesus	T-FJ	M	P	G	P	M	H	MP
S/M	Marereni	MRR	Fungu	MR-FN	M	B	M	P	M	M	CW
S/M	Mtwapa	MTW	Fungu la Mvinjeni	MTFM	M	B	M	-	-	M	MP
S/M	Malindi	MLD	Fungu ya juu	ML-FJ	M	B	M	P	M	G	MP
S/M	Tiwi	TWI	Fuzo	T-FZ	M	B	G	P	M	H	MP
S/M	Watamu MP & R	WPR	Hemmingways	W-HMS	M	P	G	P	M	H	MP
S/M	Watamu MP & R	WPR	Jacaranda	W-JC	M	P	M	M	M	H	MP
S/M	Mtwapa	MTW	Jumba Ruins	MTJR-S	M	P	G	P	M	H	MP
S/M	Kanamai	KNM	Kack Land	KN-KL	M	B	M	-	-	G	MP
S/M	Watamu MP & R	WPR	Kadaini	W-KD	P	P	M	M	G	M	CW
S/M	Ngomeni	NGM	Kafeni	N-KF	M	B	M	P	M	G	MP
S/M	Tudor Creek	TDC	Kenya Meat Comm.	T-KMC	M	B	M	P	P	M	MP
S/M	Kanamai	KNM	Kibuyuni	KN-KY	H	P	G	P	M	G	MP
S/M	Mwaepe	MWP	Kigomeni	MW-KG	G	P	G	P	G	H	MP
S/M	Kuruwitu	KRW	Kijangwani	KR-KJ	G	P	M	M	M	G	MP
S/M	Kilindini Creek	KLD	Kilindini Mouth	K-KM	G	B	H	B	B	G	CW
S/M	Kilindini Creek	KLD	Kipevu	K-KP	M	P	G	P	P	G	MP
S/M	Ngomeni	NGM	Kivunjeni	N-KV	M	B	M	M	M	G	MP
S/M	Tudor Creek	TDC	Kombani RM	T-KRM	M	B	M	B	P	M	MP
S/M	Mombasa MP & R	MPR	Kota manange	M-KO	M	P	G	P	B	M	CW
S/M	Kuruwitu	KRW	Bandarini	KRKB	H	B	M	P	M	G	MP
S/M	Marereni	MRR	Kwa Adiadi	MR-KA	M	B	M	M	M	M	MP
S/M	Marereni	MRR	Kwa Mtsere	MR-KM	M	P	M	-	-	P	MP
S/M	Funzi	FNZ	Langoni	FZ-LN	M	M	G	M	H	M	MP
S/M	Chale	CHL	Ligagana	CH-LG	M	P	M	P	M	H	MP
S/M	Kilindini Creek	KLD	Liwatoni	K-LW	M	M	G	M	M	G	MP
S/M	Shimoni	SHM	Lower Mpunguti	SM-LM	M	P	G	B	P	G	MP
S/M	Tudor Creek	TDC	Madobini	T-MD	M	M	G	M	H	H	MP
S/M	Malindi	MLD	Magic Island	ML-MI	M	M	M	M	G	H	MP
S/M	Kilindini Creek	KLD	K-Navy	K-NV	M	P	G	P	P	G	MP
S/M	Mida	MDA	Majaoni	MDMJS	M	P	G	M	G	M	MP
S/M	Chale	CHL	Makonde	CH-MK	H	M	H	M	H	H	MP
S/M	Marereni	MRR	Maluwayu	MR-ML	M	B	M	P	M	M	MP
S/M	Kilindini Creek	KLD	Mambome RM	K-MBR	M	B	M	P	M	M	MP
S/M	Funzi	FNZ	Matandiko ya Chini	FZ-MC	M	M	G	M	H	G	MP

S/M	Ngomeni	NGM	Mbwanyongo	NMB	M	B	M	-	-	G	MP
S/M	Tiwi	TWI	Mkwanju	T-MJ	M	P	M	M	G	G	-
S/M	Mwaepe	MWP	Mlango wa Galu	MW-MG	G	M	G	M	H	H	MP
S/M	Tiwi	TWI	Mlango wa Tiwi	T-MTS	M	B	M	M	M	H	MP
S/M	Mida	MDA	Maji ya ndani	MDMNS	P	P	M	P	M	M	MP
S/M	Mida	MDA	Mvinje moja	MDMV	M	B	M	B	B	M	MP
S/M	Kilindini Creek	KLD	Mwache RM	K-MRM	M	P	G	P	M	M	MP
S/M	Diani	DNI	Mwachema	DN-MW	M	B	M	-	-	H	MP
S/M	Marereni	MRR	Mwamba Bakari	MR-MB	M	B	M	P	P	M	MP
S/M	Kuruwitu	KRW	Mwanamia	KRMW	H	B	M	P	P	G	MP
S/M	Diani	DNI	Mwanamochi	DN-MM	M	B	M	B	B	H	MP
S/M	Mwaepe	MWP	Mwanyaza	MW-MZ	M	P	M	M	M	G	CW
S/M	Tiwi	TWI	Mwapadi	T-MP	M	P	G	P	M	H	MP
S/M	Malindi	MLD	M-Navy	ML-NV	G	B	M	M	M	H	MP
S/M	Mida	MDA	Sita	MDST	M	B	M	B	P	M	MP
S/M	Mombasa MP & R	MPR	Nyali	M-NLS	M	P	G	M	G	G	MP
S/M	Kuruwitu	KRW	Nyamamba	KRNY	G	B	G	P	P	H	MP
S/M	Tudor-Kilindini Mouth	TKM	T-K Mouth	T-KM	M	M	H	M	M	H	MP
S/M	Kanamai	KNM	Paradise	KN-PD	H	-	G	-	-	G	MP
S/M	Vipingo	VP	Pilani	VP-PL	M	P	G	P	M	G	CW
S/M	Mombasa MP & R	MPR	Pirates	M-PRS	M	B	M	B	B	G	MP
S/M	Shelly	SHL	Pungu Villa	SHPV	-	B	B	M	M	-	-
S/M	Watu MP & R	WPR	Richard Bennett	SHSH	M	P	M	M	M	H	MP
S/M	Shelly	SHL	Sacred heart	SHSH	-	P	M	H	M	-	-
S/M	Ngomeni	NGM	ShaliShali Komeoni	NSS	M	B	M	P	M	G	MP
S/M	Mida	MDA	Thonga	MD-TH	M	P	G	B	P	M	MP
S/M	Shimoni	SHM	Shimoni garden	SM-SG	G	P	G	P	M	H	MP
S/M	Malindi	MLD	Simitini	ML-SM	G	B	M	P	M	G	MP
S/M	Mida	MDA	Yava	MDYV	M	P	M	P	M	M	MP
S/M	Kanamai	KNM	Sun and Sand	KN-SS	M	P	G	P	M	G	MP
S/M	Tiwi	TWI	Tangi la Meli	T-TM	M	P	M	G	H	H	MP
S/M	Shelly	SHL	Timbwani	SHTB	-	B	M	P	M	-	-
S/M	Tudor Creek	TDC	Tsalu RM	T-TRM	M	M	M	H	M	M	MP
S/M	Shimoni	SHM	Wasini Channel	SM-WC	M	P	M	G	M	G	MP
S/M	Mombasa MP & R	MPR	Mtwapa Mouth	M-MTS	M	B	M	-	-	G	-
S/M	Ngomeni	NGM	Imbo Nasoro	N-IN	P	B	H	B	B	H	CW
S/M	Ngomeni	NGM	Maliwai	NML	P	B	G	B	B	H	MP
S/M	Takaungu/Vuma	TKV	Bekombo	TV-BK	M	B	M	B	B	H	MP
S/M	Ngomeni	NGM	Ras Kwa Mwalimu	N-RM	P	B	G	B	B	H	MP
S/M	Takaungu/Vuma	TKV	Dzambani	TV-DZ	P	B	M	B	B	H	MP
S/M	Ngomeni	NGM	Vitangani	N-VT	M	B	M	P	B	H	MP
S/M	Takaungu/Vuma	TKV	Kajutoni	TV_KJ	P	B	G	B	B	H	CW
S/M	Takaungu/Vuma	TKV	Park	TV-PK	M	B	M	B	B	H	CW
S/M	Takaungu/Vuma	TKV	Vitanga Viwili	TV-VV	M	B	M	P	B	H	CW
S/M	Tudor Creek	TDC	Coast General	T-CG	M	B	B	M	P	M	MP
S/M	Tudor Creek	TDC	Tudor Water Sports	T-WS	M	B	M	B	P	H	MP
S/M	Malindi MP & R	MLD	Coral Garden	ML-CG	M	B	M	B	B	G	MP
Jul	Mida	MDA	Kirepwe	MDKE	M	-	G	-	-	G	MP
Jul	Mida	MDA	Majaoni	MDMJJ	M	B	M	B	B	M	MP
Jul	Mida	MDA	Maji ya ndani	MDMNJ	M	B	G	B	B	M	MP
Jul	Mida	MDA	Saidi Ndau	MDSN	M	-	H	-	-	G	MP
Jul	Mombasa MP & R	MPR	Mtwapa Mouth	M-MTJ	M	B	H	B	B	H	MP

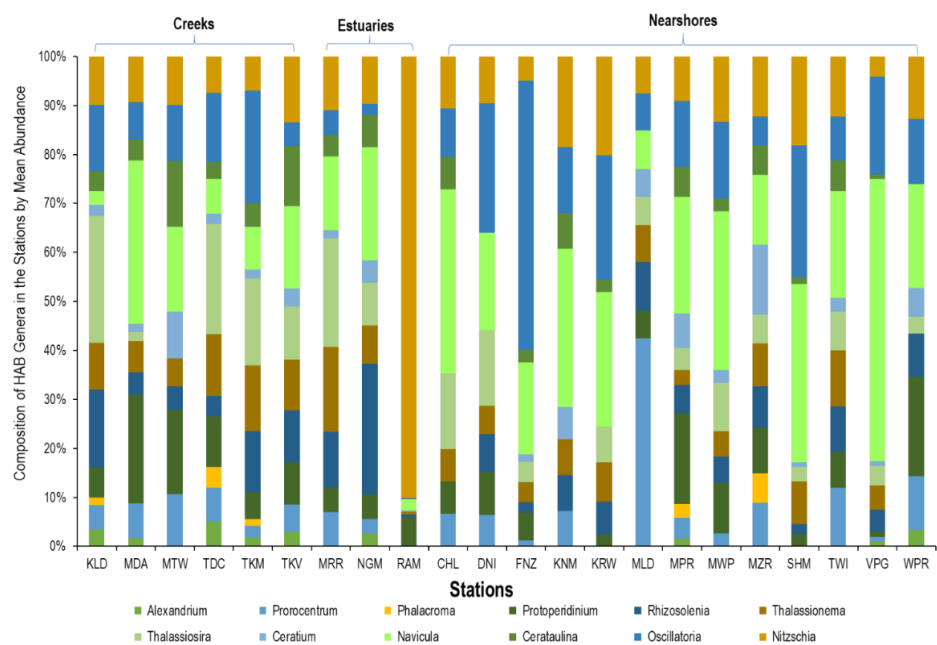
Jul	Tiwi	TWI	Amani Tiwi	TAT	M	-	G	-	-	H	MP
Jul	Watamu MP & R	WPR	Captain Andys	W-CA	M	B	M	M	P	G	MP
Jul	Ramisi	RAM	Chikondo	RA-CH	H	B	M	B	B	M	MP
Jul	Watamu MP & R	WPR	Coral garden	W-CG	M	B	G	B	B	H	MP
Jul	Mombasa MP & R	MPR	Cowrieshell	M-CS	G	-	H	-	-	H	MP
Jul	Mwazaro	MZR	Gamba la kasa	MZ-GK	M	B	B	B	B	G	MP
Jul	Watamu MP & R	WPR	Hemmingways	W-HMJ	M	P	M	M	P	H	MP
Jul	Diani	DNI	Jacaranda	DN-JD	M	-	G	-	-	H	MP
Jul	Mtwapa	MTW	Jumba Ruins	MTJR-J	M	-	G	-	-	H	MP
Jul	Mwazaro	MZR	Kifungu	MZ_KF	G	B	M	B	B	M	MP
Jul	Mwazaro	MZR	Kombeni	MZ-KB	M	B	M	B		G	MP
Jul	Tiwi	TWI	Kongo River	T-KR	M	-	G	-	-	H	MP
Jul	Mwazaro	MZR	Mundu	MZ-MU	M	B	P	B	B	G	MP
Jul	Mwazaro	MZR	Mwamba mkuu	MZ-MM	M	B	M	B	B	G	MP
Jul	Diani	DNI	Mwanamochi	DN-MO	M	-	M	-	-	H	MP
Jul	Mwazaro	MZR	Mwazaro	MZ-RO	M	B	M	B	B	G	MP
Jul	Mwazaro	MZR	Ndua	MZ-ND	M	B	M	B	B	M	MP
Jul	Mombasa MP & R	MPR	Nyali	M-NLJ	G	P	M	M	M	G	MP
Jul	Tiwi	TWI	Panga point	T-PO	M	-	G	-	-	H	MP
Jul	Mombasa MP & R	MPR	Pirates	M-PRJ	G	-	H	-	-	H	MP
Jul	Ramisi	RAM	Ramisi Mouth-A	RA-RA	M	B	M	B	B	M	MP
Jul	Ramisi	RAM	Ramisi Mouth-B	RA-RB	M	B	P	B	B	M	HP
Jul	Ramisi	RAM	Ramisi Mouth-C	RA-RC	M	B	M	B	-	M	HP
Jul	Ramisi	RAM	Ramisi Mouth-D	RA-RD	P	B	M	B	-	M	HP
Jul	Ramisi	RAM	Ramisi Mouth-E	RA-RE	M	B	M	B	-	P	HP
Jul	Mombasa MP & R	MPR	Ras Iwatine	M-RI	G	B	M	M	M	H	MP
Jul	Watamu MP & R	WPR	Richard Bennett	W-RBJ	G	P	G	P	B	H	MP
Jul	Mombasa MP & R	MPR	Serena	M-SR	M	-	G	-	-	H	MP
Jul	Diani	DNI	Southern palms	DN-SP	M	-	M	B	B	H	MP
Jul	Mombasa MP & R	MPR	Star Fish	M-SF	G	-	G	-	-	G	MP
Jul	Watamu MP & R	WPR	Turtle Bay	W-TB	M	P	G	P	P	H	MP
Jul	Watamu MP & R	WPR	Whale Island	W-WL	H	B	M	-	-	H	MP
Jul	Mombasa MP & R	MPR	Coral garden	M-CG	M	-	H	-	-	H	MP

7.3.Chapter 4: Appendices

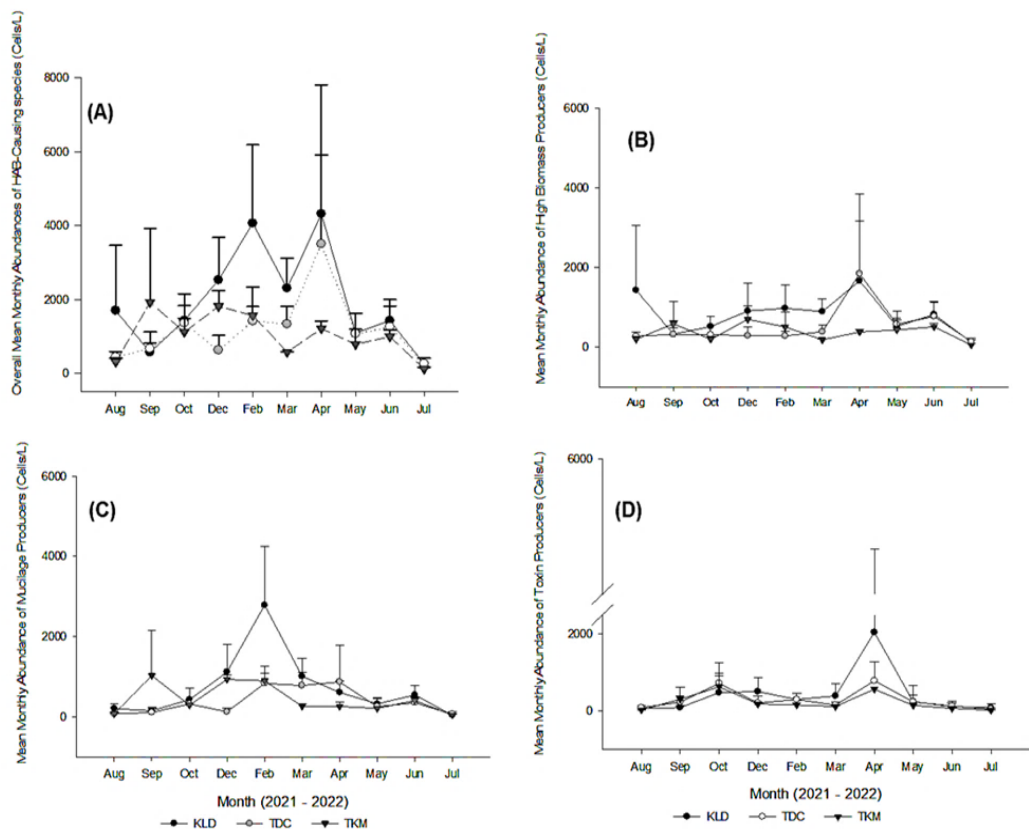
Appendix 8: Description of study sites in Mombasa

Location	Site Name	Code	Lat.	Long.	Site Description
Kilindini Creek	R. Mwache	S01	-4.063054	39.591835	Seasonal river Mouth draining agricultural watersheds
	R. Mambone	S02	-4.043175	39.583285	Seasonal river Mouth draining residential and airport watersheds.
	Kenya Navy	S03	-4.050679	39.626409	Dockyard and Sewage
	Kipevu	S04	-4.048472	39.638056	Port docking area adjacent to Kipevu sewage treatment plant
	Liwatoni Bridge	S05	-4.073306	39.657778	Port docking area -Ballast water
	Kilindini Creek Mouth	S06	-4.076925	39.679345	ballast water an sea water mixtures
Open Waters	Open Water	S07	-4.080664	39.692236	open waters, about 500m from the creeks mouths
Tudor Creek	Fort Jesus (Tudor Creek Mouth)	S08	-4.067986	39.685038	Untreated Municipal sewage and seawater mixtures
	Madobini	S09	-4.049823	39.675704	Raw domestic sewage discharges
	Coast General Hospital	S10	-4.046830	39.673025	Sewage from hospitals
	Tudor Water Sports	S11	-4.027889	39.665833	Recreational activities and hotel sewage
	Kenya Meat Commission (KMC)	S12	-4.029722	39.653333	Industrial and raw sewage and open defecation from informal settlements
	R. Kombeni	S13	-4.009449	39.647291	Seasonal river Mouth draining agriculture and informal settlements watersheds
	River Tsalu	S14	-4.015183	39.654673	Seasonal river Mouth draining upcoming settlements watersheds
Kanamai	Kanamai	S15	-3.932417	39.782778	Control site - community managed coral reefs, less populated and no human development

Appendix 9: Composition of HAB-species in the stations. Proportion of potentially harmful microalgae genera in Kenya's coastal waters compared to overall abundance across stations

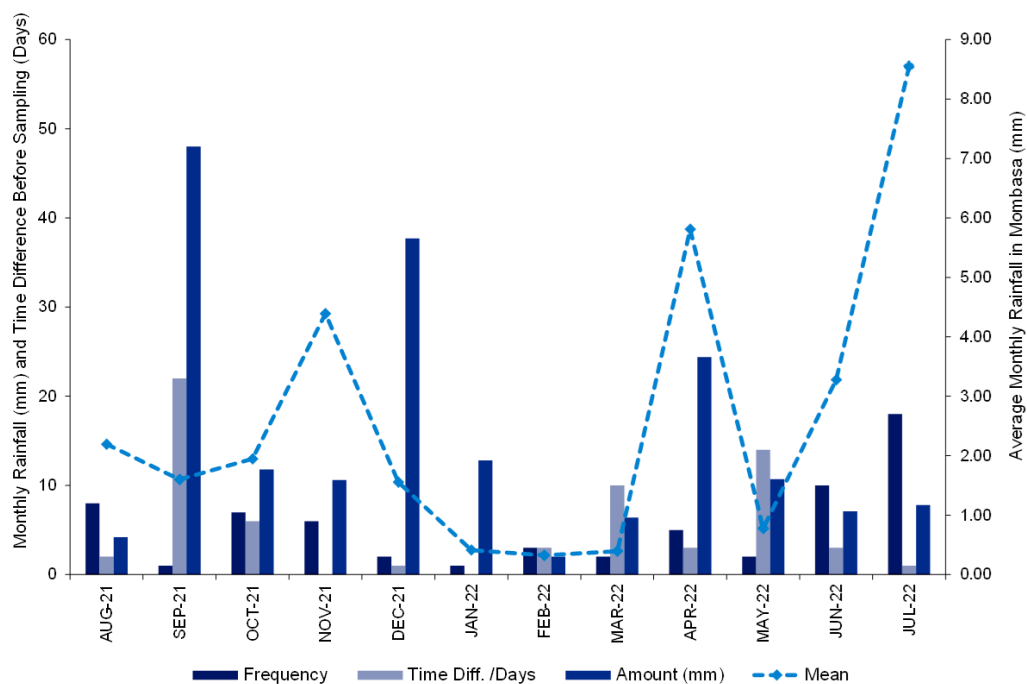


Appendix 10: Monthly mean species abundances Overall (A), potential High Biomass Producers (B), Mucilage Producers (gill-clogging species)(C) and Toxin Producers (D).

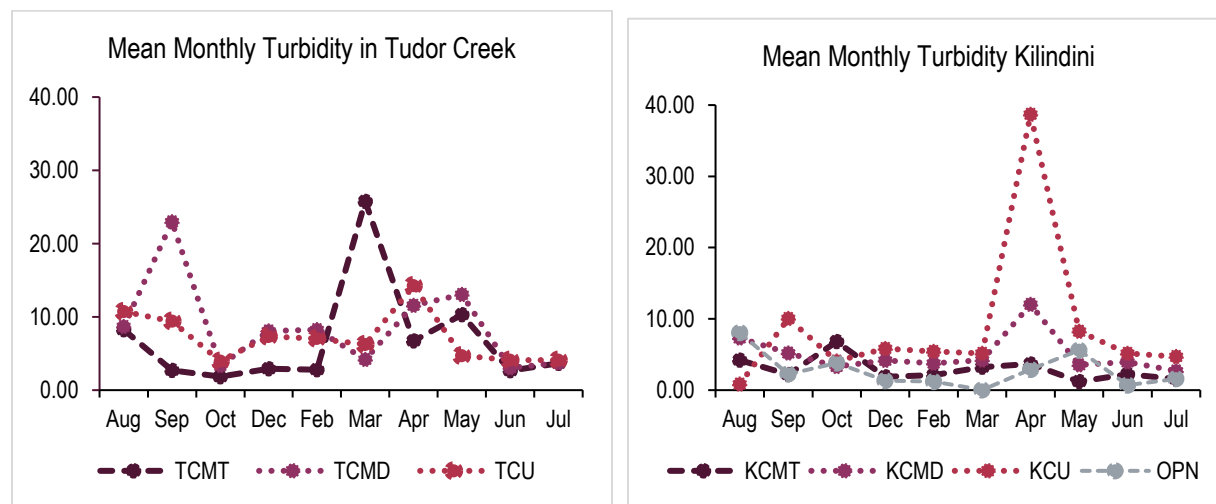


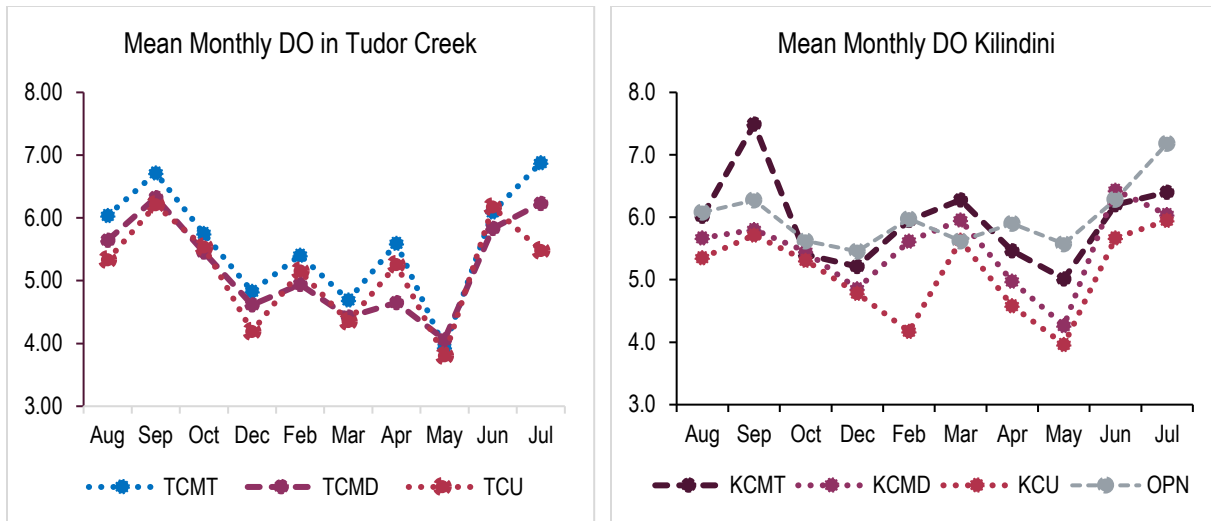
7.4.Chapter 5: Appendices

Appendix 11: Rainfall in Mombasa during the study



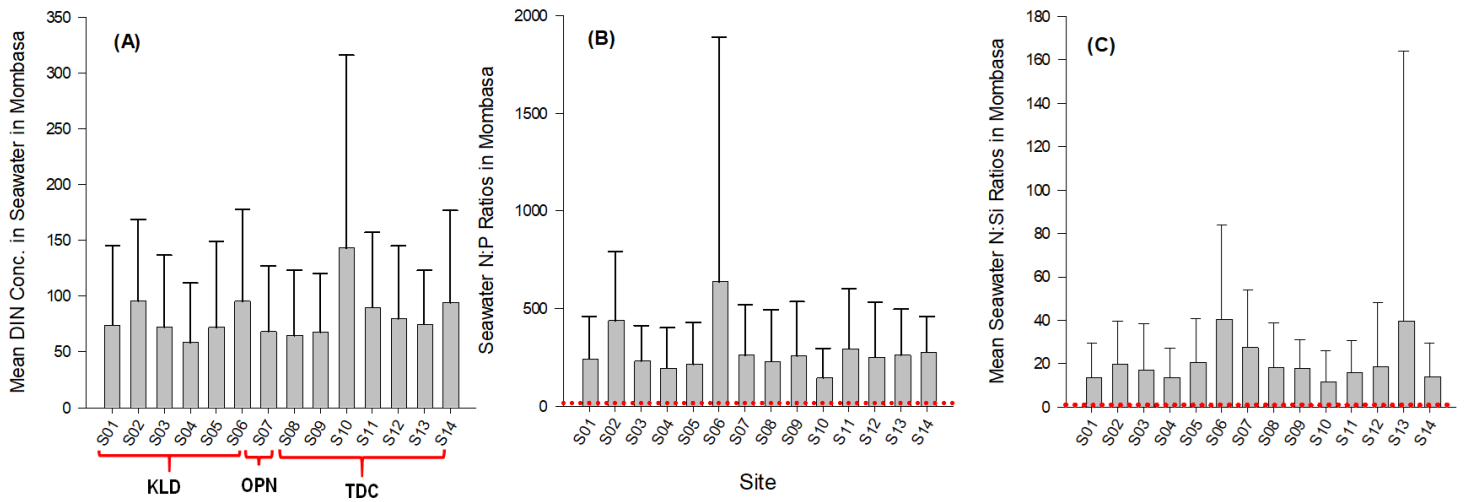
Appendix 12: Spatial and temporal distribution in Turbidity and DO



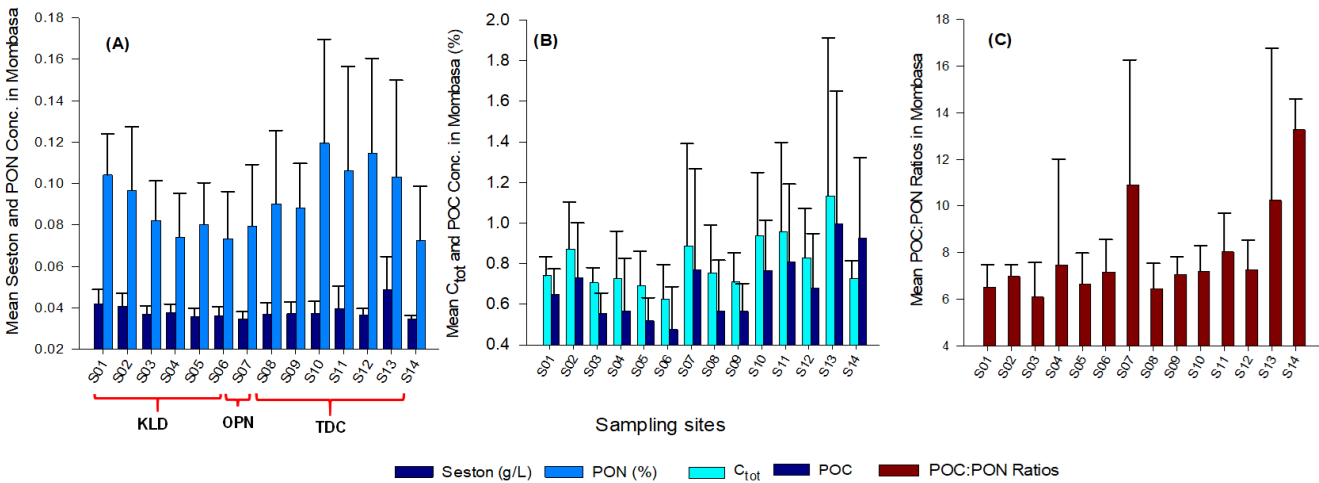


Spatial and temporal distribution of Turbidity (Top) and Dissolved Oxygen- DO in Mombasa coastal waters. TCMT – Tudor creek Mouth, TCMD – Tudor Creek Midwaters, TCU – Tudor creek Upper waters, KCMT – Kilindini Creek Mouth, KCMD – Kilindini midwaters, KCU – Kilindini creek upper waters, and OPN - Open waters

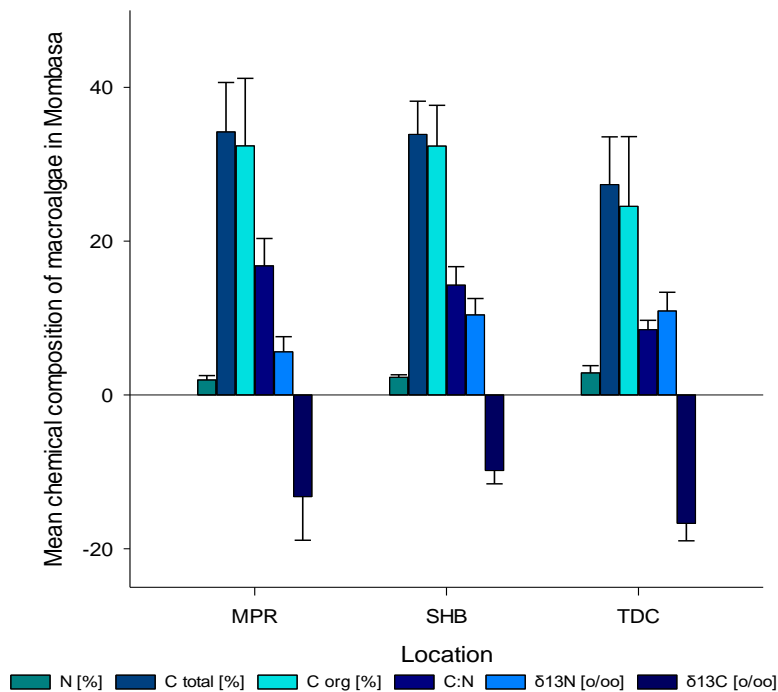
Appendix 13: Spatial distribution in DIN, N/P and N/Si



Appendix 14: Spatial distribution in mean seston and components - mean and S.d.



Appendix 15: Mean chemical composition of macro-algae in Mombasa coastal waters



Appendix 16: Macroalgae composition for isotopic analysis

MMPR primarily contained seagrasses with some brown and green macro algae, including *Cymodocea serrulata*, *Holodule* spp., *Syringodium isoetifolium*, *Thalassodendron ciliatum*, *Halmeda macroloba*, *Sargassum* spp., and *Ulva* spp. Shelly Beach featured a mix of seagrasses (*Cymodecia rotundata*, *Halodule wright*) and green algae (*Ulva* spp.) while Tudor Creek exclusively had macro algae, including *Ulva* spp., *Halimeda renchii*, and *Hyponaea musciformis* at MEP (adjacent to S08) and at S09 (MDB).

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