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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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ABSTRACT

We assessed 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 potential harmful algal blooming species. Variations in the EQS was 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 rainy period compared to dry season. Additionally, 90.1 % of the sites in 'poor' and 'bad' 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.

## 1. 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 (Neumann et al., 2015). Anthropogenic activities such as land-use for agriculture have altered the hydrologic cycle (Zhang et al., 2022), increased nutrient enrichment in coastal waters (Adams et al., 2020; Malone and 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 (Adams et al., 2020; Anderson et al., 2002; Malone and 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 and Rabalais, 2013) as well as the promotion of infections like fibro-papillomatosis (FP) in sea turtles (Aguirre and 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 and 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

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impaired immune function (Dujon et al., 2021; Jones et al., 2016). 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, bays, 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 (Kiteresi et al., 2012; 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 and Katana, 2022; van de Geer et al., 2022; The Standard Newspaper, 2021). 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.

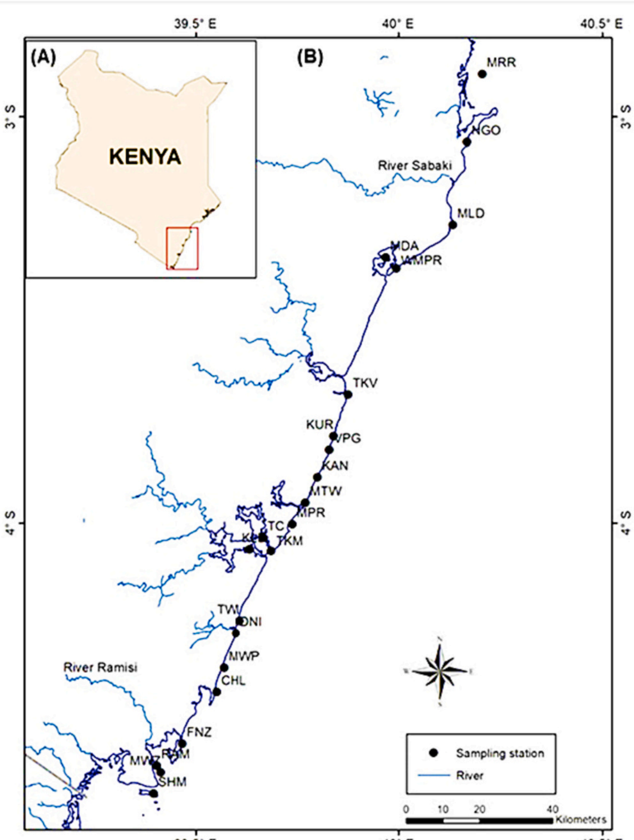
## 2. Materials and methods

### 2.1. 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 600 km (Fig. 1; Kaunda-Arara et al., 2004). The area is primarily affected by the semi-diurnal tides with spring tidal variation of up to 4.0 m. 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 and Majambo, 2020).

### 2.2. Site selection and field sampling

This study focused on sea turtle foraging sites located in coastal areas with possible exposure to anthropogenic pressures. The beach




(C)

| Code | Full Name |
|---|---|
| MRR | Marereni |
| NGO | Ngomeni |
| MLD | Malindi |
| MDA | Mida Creek |
| WMPR | Watamu MNPR |
| TKV | Takaungu Vuma |
| KUR | Kuruwitu |
| VPG | Vipingo |
| KAN | Kanamai |
| MTW | Mtwapa Creek Mouth |
| MPR | Mombasa MNPR |
| TKM | Tudor-Kilindini Mouth |
| TC | Tudor Creek |
| KC | Kilindini Creek |
| TWI | Tiwi |
| DNI | Diani |
| MWP | Mwaepe |
| CHL | Chale |
| FNZ | Funzi |
| RAM | Ramisi Mouth |
| MWZ | Mwazaro |
| SHM | Shimoni |

**Fig. 1.** 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).

management unit (BMU) is a government management framework of fish landing sites and all fisheries activities legally supported by the Fisheries (BMU) Regulations of 2007 (Government of Kenya, 2012). The units were adopted as stations and individual sub-units were treated as sites in this study. 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 < 500 m from the shoreline, and oceanic sites as island sites located ≥500 m from the shoreline with less anthropogenic exposure. See Appendix 1 for a full description of site selection procedures. The ecological surveys were conducted in five tidal creeks (Tudor-TC, Kilindini-KLD, Mtwapa-MTW, Takaungu-Vuma-TKV, and Mida-MDA), four estuarine stations (Marereni-MRR, Ngomeni-NGO, Ramisi-RAM, and Mwazaro-MWZ), 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, Malindi-MLD) and four oceanic sites (Kisite Mpunguti, Whale Island and Magic Island in SHM, WMNPR, and MLD areas, respectively).

A total of 95 sites located in 23 stations 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.

### 2.3. Sampling and laboratory analysis

Surface seawater samples were collected during low tides from each site during the survey. Insitu 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 2–3 times in each site, depending on the spatial coverage of the sites. Field data was collected using the KoboToolbox 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.

#### 2.3.1. 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 (0.06 mL $HgCl_2$-solution/10 mL 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- $NO_2^-$ + nitrates -$NO_3^-$) concentrations (García-Robledo et al., 2014). Salicylate method for ammonium-$NH_4^+$, PhosVer3 ascorbic acid method for orthophosphate-$PO_4^{3-}$ and silicate-$SiO_3^{2-}$ 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 $NO_x^-$, $PO_4^{3-}$, and $NH_4^+$, respectively. In this study, the term dissolved inorganic nitrogen (DIN) refers to the sum of $NO_x^-$ and $NH_4^+$.

#### 2.3.2. Phytoplankton assemblages

Phytoplankton biomass (*Chl a)* samples were collected by filtering 1 L of water onto Whatman GF/F filters under pressure using a handheld pump. at sites with high turbidity, 500 mL was taken. The filters were stored frozen until analysis. These were analyzed 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 analyzing check standards.

For phytoplankton species diversity and abundance analysis, 20 L of seawater was collected using a bucket and filtered through a 20 μm phytoplankton net, concentrated up to 50 mL 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 Hasle (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).

### 2.4. Statistical analysis

All data analysis was conducted using SigmaPlot version 14.0.3.192, R version 4.2.3 (R Core Team, 2023) with the vegan package (Oksanen et al., 2015) and PRIMER-E (v7.0.22) (Clarke and Gorley, 2006). Spatial analyses were conducted using QGIS v3.28.3.

#### 2.4.1. 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:

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

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) below:

$$\mathrm{N} = \frac{\mathrm{A} \times 1000 \times \mathrm{C}}{\mathrm{V} \times \mathrm{F} \times \mathrm{L}} \tag{2}$$

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 $mm^3$; F = Number of fields counted; and L = Volume of the water filtered in liters; (iii) Shannon-Wiener diversity (H′), measures the diversity of phytoplankton species in a community, calculated from the phytoplankton species and abundance data for each site using the formula:

$$H^{'} = -\sum \rho i \times In(\rho i) \tag{3}$$

where ln is the natural logarithm and $\rho i$ is the proportion of the entire community made up of species *i* (Shannon, 1948)*;* and (iv) Pielou's evenness (E) (Pielou, 1966):

$$\mathrm{E} = \mathrm{H}/\mathrm{In}\ \mathrm{S} \tag{4}$$

*2.4.2. Spatial patterns in phytoplankton assemblages*

To explore spatial changes in phytoplankton assemblage composition, permutational multivariate analysis of variance (PERMANOVA) (Clarke and Warwick, 2001) based on Bray-Curtis similarity of $Log_{10}$ transformed data (Bray and 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.

*2.4.3. 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 (Appendix 2). 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 and 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 (VIF) were employed to examine how much multicollinearity exists in the multiple regression analysis and none of the VIFs inspected 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).

*2.4.4. Establishment of the ecological quality status of the study sites*

To determine the ecological quality status (EQS) of the sea turtle foraging sites, this study adopted the boundaries [Ecological Quality Ratios(EQR)] for inorganic nutrient concentrations (DIN - $NOx^- + NH_4^+$ and DIP) and ratios N:P and N: Si, *Chl a* and dissolved oxygen (DO) listed for the European Water Framework Directives-WFD – 2000/60/EC (European Parliament and the Council of the European Union, 2000) in Simboura et al., (2005), Andersen et al., (2010) and Newton et al., (2022) with modifications where possible (Table 1). The criteria adopted the Redfield ratio that is, 16:1 for N:P and ~1:1 for N:Si as the reference conditions for 'high' ecological status with deviations ranked in a sequential manner of '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 and 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 1**
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 | | | | Dunn and Karydis, (2022) |
| Algal species pollution index | | X < 10 = Low organic pollution; 10 < X < 15 = Moderate pollution; 15 < X < 20 = Probable high organic pollution; X ≥ 20 = High organic pollution. | | | | Elshobary et al. (2020) |

## 3. Results

*3.1. Spatial variability in environmental factors and phytoplankton assemblages*

*3.1.1. Physical parameters*

The seawater was generally oxic. However, significant spatial variations were observed across the different ecosystems. The mean dissolved oxygen (DO) in the sites ranged from 4.4 to 9.8 mg/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 mg/L ± 1.2 mg/L; $P < 0.001$) compared to 6.75 mg/L ± 1.1 mg/L in September/March. Dissolved oxygen was substantially lower (< 5 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 to 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 2 and Appendix *3*). 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.

### 3.1.2. *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 ± 17 μM 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 $NO_x^-$ ($p \leq 0.001$) were also observed in September/March compared to July with overall means of 11.5 ± 9.4 μM and 3.0 ± 6.6 μM respectively (Appendix *4*).

Ammonium levels in the sites ranged between 0.76 and 63.3 μM during the study and were significantly higher in all the tidal creeks compared to other areas (Table 2). Significantly higher concentrations of $NH_4^+$ were also observed in July with a mean value of 26.8 ± 16.8 μM compared to 9.6 ± 11.8 μM in September/March. Unlike $NO_x^-$, higher

**Table 2**

Mean (±SD), 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.

| | Parameter | | Creek | Estuary | Nearshore | Oceanic | Overall | Overall P-value |
|---|---|---|---|---|---|---|---|---|
| Physical parameters | DO (mg/L) | $\bar{x}$ + SD | 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 (°C) | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 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 | |
| | pH | $\bar{x}$ + SD | 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 (μS/cm) | $\bar{x}$ + SD | 56,099 ± 1435 | 54,573 ± 3018 | 54,934 ± 827 | 54,085 ± 1083 | 54,957 ± 1716 | **0.001** |
| | | Range | 53,658–58,136 | 45,592–59,917 | 53,328–57,670 | 52,854–54,892 | 45,592–59,917 | |
| Chemical Parameters (Dissolved inorganic nutrients) | $NO_x^-$-($NO_3^-$ + $NO_2^-$ (μM) | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 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 ± 3.9 | 0.8–168.7 | |
| | DIN (μM) | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 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 | $\bar{x}$ + SD | 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 ± 21.7 | 21.7–114 | 0.08–801.0 | |
| | N: Si | $\bar{x}$ + SD | 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 ± 3.4 | 0.002–483.5 | |
| Biological Parameters (Phytoplankton indices) | Chl a [μg/L] | $\bar{x}$ + SD | 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) | $\bar{x}$ + SD | 1388 ± 1642 | 8640 ± 21,800 | 667 ± 412 | 1676 ± 1819 | 2341 ± 9960 | 0.370 |
| | | Range | 233–7669 | 206–86,100 | 113 ± 2558 | 450–3767 | 113–86,100 | |
| | Species Diversity Index (H′) (bits/cell) | $\bar{x}$ + SD | 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 | |
| | Evenness | $\bar{x}$ + SD | 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 | $\bar{x}$ + SD | 25 ± 9 | 22 ± 11 | 18 ± 7 | 17 ± 7 | 19 ± 8 | **0.040** |
| | | Range | 10–43 | 9–53 | 9–37 | 11–25 | 6–53 | |

$NH_4^+$ concentrations of 10 μM – 41.4 μM were observed in WMNPR and other sea turtle tumor prevalence areas (TKV, MDA, and NGO) (Appendix *5*). The general spatial trend in sites' mean DIN ($NO_x^- + NH_4^+$) 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 (Fig. 3A and Table 2).

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 ± 0.3 μM and 0.38 ± 0.3 μM recorded during September/March and July surveys, respectively.

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 ($P \leq 0.001$). Localized higher $SiO_3^{2-}$ concentrations of 29.1–34.2 μM were also registered in nearshore sites around VPG and SHL, and tidal creek (TC) waters near Coast General Hospital (Fig. 2C). 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 upstream at the RAM estuary in July.

The mean N:P ratios in the sites ranged between 0.001 and 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 rations were established in the sites located in the tidal creeks and differed significantly with nearshore sites and insignificantly with estuarine and oceanic sites (Table 2). 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 (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 (Fig. 3A).

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 to 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 (Fig. 3B).

### 3.1.3. Phytoplankton biomass and community composition

The biomass of phytoplankton (*Chl a)* concentrations ranged between 0.06 and 2.34 μg/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 μg/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 and 0.8 μg/L. Additionally, very low *Chl a* concentrations, with narrow ranges (0.2–0.45 μg/L) were established in south coast nearshore sites located in TWI, DNI, and CHL (Fig. 4A). There were also no variations established *Chl a* concentrations between the two sampling surveys with means of 0.46 ± 0.47 μg/L and 0.42 ± 0.35 μg/L recorded for September/March and July, respectively.

The phytoplankton abundance was significantly higher in the estuarine areas compared to other sites with ranges of 113–86,100 cells/L recorded during the study (Fig. 4B). The abundance was also significantly higher ($P = 0.01$) after the rains registering ranges of 206–86,100 cells/L in July compared to 113–13,238 cells/L in September/March.

The Shannon-Wiener phytoplankton diversity index ranged from 0.54 to 3.37 bits/cell during the study (Table 2). 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 (Fig. 4C). 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)

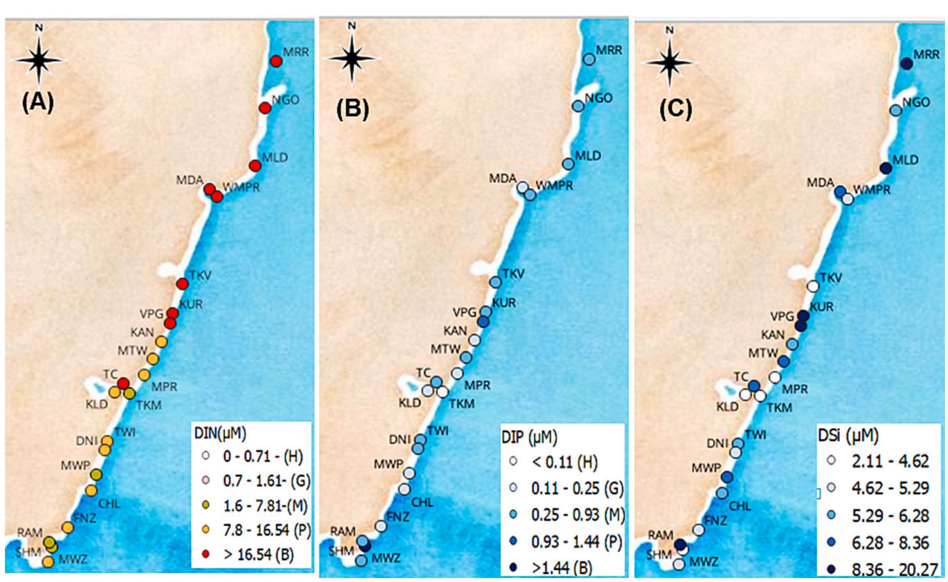



**Fig. 2.** Dissolved inorganic nutrients, nitrogen-DIN ($NO_x^- + NH_4^+$), Phosphate-DIP ($PO_4^{3-}$), and Silicate- DSi ($SiO_3^{2-}$) in turtle foraging areas (Refer to Fig. 1 for full code names). Abbreviations for EQS 'H' – High, 'G'- Good, 'M' -Medium, 'P'- Poor, and 'B' - Bad.

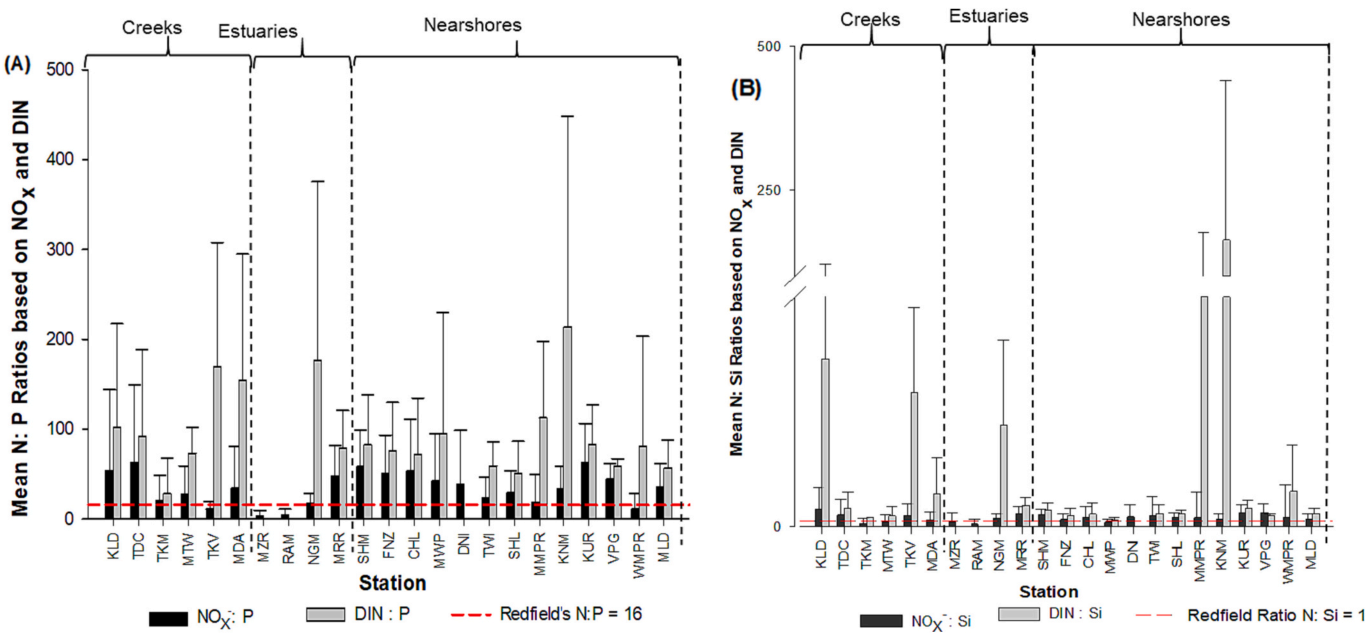



**Fig. 3.** Mean + SDev of Dissolved inorganic nutrient ratios N:P (A) and N:Si (B) based on both $NO_x$ and DIN in sea turtle foraging sites along the Kenya coast recorded over the study period.

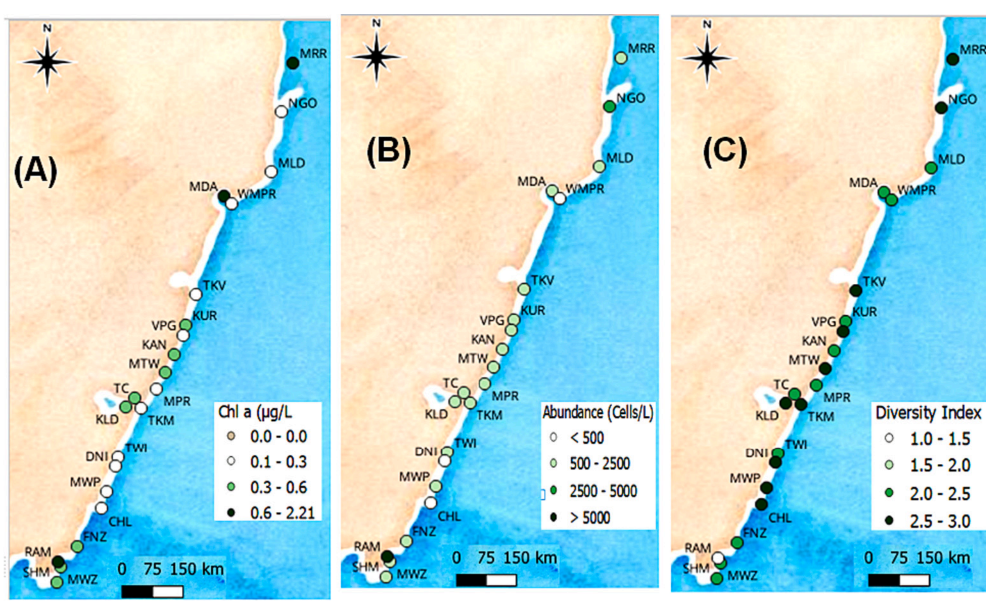



**Fig. 4.** Mean phytoplankton biomass (*Chl a*) abundance and Shannon-Wiener diversity index across sea turtle foraging areas along the Kenya coast over the study period. Full area names are in Fig. 1.

compared to July (n = 69).

A total of 154 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 %) (Fig. 5).

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 heterogenous (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

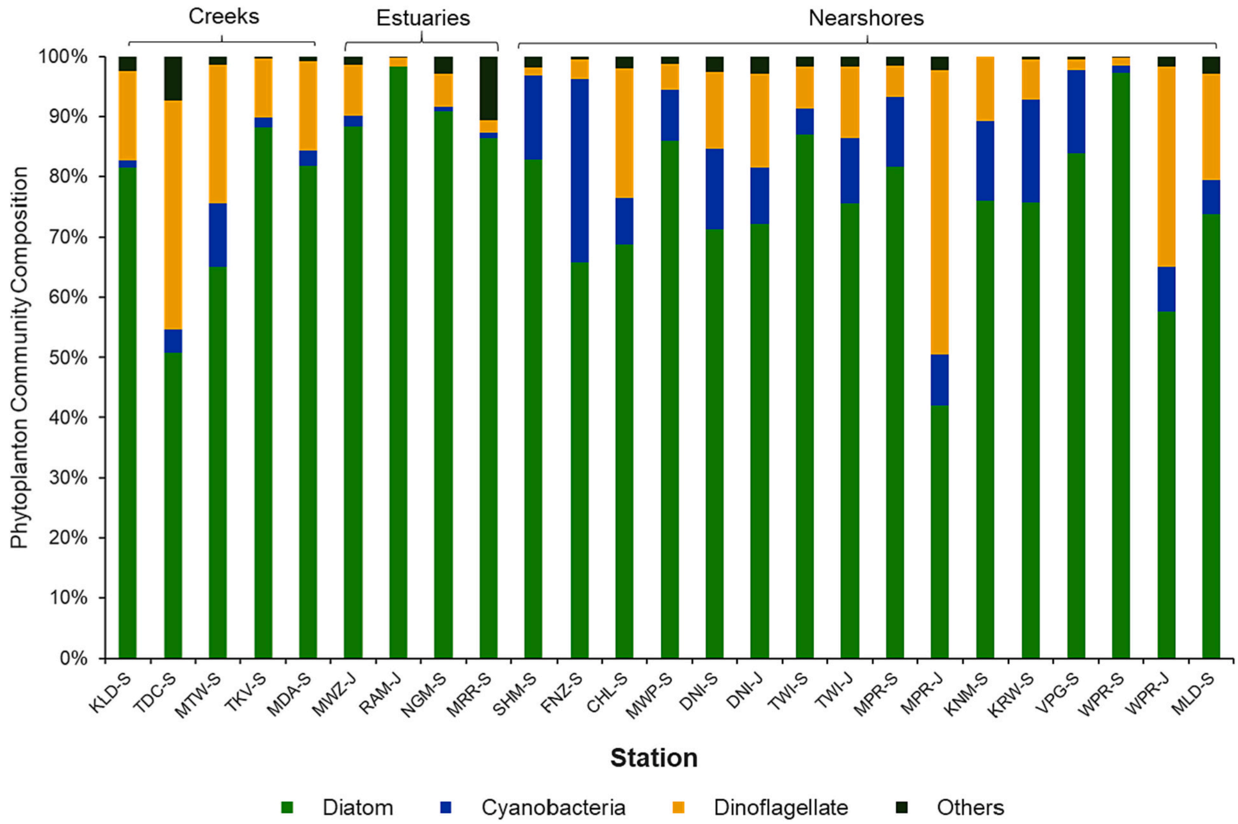



**Fig. 5.** Relative proportion of the phytoplankton community composition based on major phytoplankton groups in the sea turtle foraging sites. S – September and J- July. Note: MPR – MMNPR; and WPR -WMNPR.

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 19–75,300 cells/L during the September/March and July surveys respectively.

### 3.2. *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 3 and Fig. 6). 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 %).

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 4). 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 4). 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 4).

In the Spearman correlation analysis, *Chl a* was also observed with a significant positive correlation with phosphate and silicate, respectively ($P \leq 0.001$). 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 (Appendix *2*).

**Table 3**
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 |

### 3.3. *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 5). Additionally, 3.2 % of the sites were highly polluted based on the Shannon-Weiner diversity index. The overall total score of

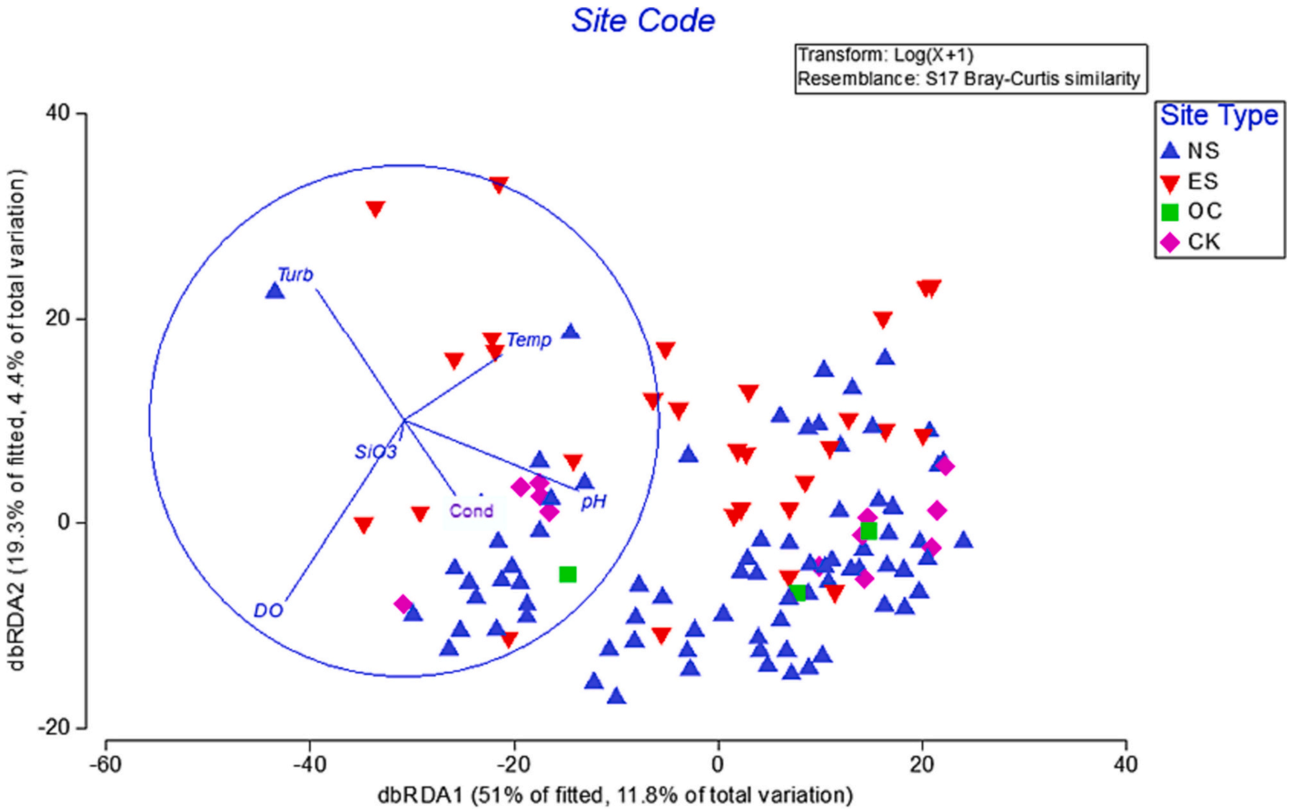



**Fig. 6.** Distance-based redundancy analysis (dbRDA) of phytoplankton samples, overlaid with normalized predictor variables (based on DistLM analysis in Table 4). NS- Nearshore, ES – Estuarine, OC – Oceanic, and CK – creek.

**Table 4**
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 - *Chl a* ($R^2$ = 0.69) | | | Richness ($R^2$ = 0.63) | | | Diversity ($R^2$ = 0.34) | | | Evenness ($R^2$ = 0.45) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Variable | Coefficient ± S.E. | t-value | *P*-value | Coefficient ± S.E. | t-value | *P*-value | Coefficient ± S.E. | t-value | *P*-value | Coefficient ± S. E. | t-value | *P*-value |
| Intercept | 60.38 ± 30.9 | 1.96 | 0.055 | 1722.5 ± 981.2 | 1.755 | 0.084 | 5.55 ± 50.34 | .11 | 0.913 | −26.18 ± 12.21 | −2.144 | 0.036 |
| Lat. | 0.9 ± 0.34 | 2.67 | **0.009** | 15.90 ± 10.66 | 1.491 | 0.14 | 0.03 ± 0.55 | 0.061 | 0.952 | −0.24 ± 0.13 | −1.825 | 0.072 |
| Long. | −1.81 ± 0.69 | −2.61 | **0.011** | −36.67 ± 22.06 | −1.662 | 0.101 | 0.12 ± 1.13 | 0.106 | 0.916 | 0.66 ± 0.28 | 2.407 | **0.019** |
| Temp. °C | −0.18 ± 0.07 | −2.46 | **0.017** | −1.91 ± 2.37 | −0.807 | 0.423 | 0.001 ± 0.12 | 0.009 | 0.993 | 0.03 ± 0.03 | 0.901 | 0.371 |
| Sal. | 0.01 ± 0.02 | 0.64 | 0.523 | −2.60 ± 0.74 | −3.530 | **0.001** | −0.06 ± 0.04 | −1.474 | 0.145 | 0.009 ± 0.009 | 1.032 | 0.306 |
| pH | −0.04 ± 0.14 | −0.26 | 0.799 | −0.02 ± 4.59 | −0.005 | 0.996 | 0.23 ± 0.24 | 0.992 | 0.324 | 0.08 ± 0.06 | 1.395 | 0.168 |
| Turb. (NTU) | 0.01 ± 0.01 | 1.56 | 0.95 | 0.18 ± 0.19 | 0.984 | 0.328 | 0.01 ± 0.01 | 0.525 | 0.601 | 0.001 ± 0.002 | 0.543 | 0.589 |
| D·O | −0.03 ± 0.03 | −1.09 | 0.280 | −3.61 ± 0.98 | −3.679 | **<0.001** | −0.05 ± 0.05 | −1.047 | 0.299 | 0.04 ± 0.01 | 2.904 | **0.005** |
| Cond. | 0.001 ± 0.0001 | 3.93 | **<0.001** | −0.002 ± 004 | −0.490 | 0.626 | −0.0002 ± 0002 | −1.062 | 0.292 | −4.8E-05 ± 5.2E-05 | −0.924 | 0.359 |
| NOx (μM) | 0.01 ± 0.004 | 1.39 | 0.170 | 0.30 ± 0.12 | 2.516 | 0.014 | 0.01 ± 0.01 | 1.525 | 0.132 | −0.001 ± 0.001 | −0.686 | 0.495 |
| PO3 (μM) | −0.19 ± 0.15 | −1.24 | 0.220 | −3.28 ± 4.86 | −0.674 | 0.502 | −0.09 ± 0.25 | −0.377 | 0.707 | 0.02 ± 0.06 | 0.248 | 0.805 |
| SiO3 (μM) | 0.01 ± 0.02 | 0.75 | 0.457 | 0.47 ± 0.52 | 0.916 | 0.363 | −0.01 ± 0.03 | −0.205 | 0.838 | −0.007 ± 0.006 | −1.144 | 0.257 |
| N:P | 0.001 ± 0.001 | 1.44 | 0.153 | 0.01 ± 0.02 | 0.558 | 0.578 | −0.0004 ± 0.001 | −0.342 | 0.734 | −0.0002 ± 0.0003 | −0.806 | 0.423 |
| N: Si | −0.03 ± 0.01 | −3.00 | **0.004** | 1.08 ± 0.31 | 3.516 | **0.001** | 0.04 ± 0.02 | 2.541 | 0.013 | 0.001 ± 0.004 | 0.35 | 0.727 |

the Algal Genus Pollution Index in the areas ranged between 11 and 19, indicating 'moderate' to 'probably high' organic pollution (Table 5). 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-

**Table 5**
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[a] | J[a] | 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[b] | | | 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 |

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

Weiner diversity index (Table 6).

## 4. 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, 2021), and supports the livelihood of about 4.3 million people (Treasury, 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 and 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 the literature (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.

### 4.1. 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 Jickells et al., 2017; Tamborski et al., 2015; Cartwright and Horn, 2019; Li et al., 2023). 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 et al., 1996). 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 runoff (Kithiia and Majambo, 2020), and sediments from the river (Tamooh et al., 2012, 2014). The seawater temperature is also high due to the

**Table 6**
Temporal variations in ecological quality status in turtle foraging sites.

| Temporal variations in EQR[a] Status in September (Sep) and July (Jul) | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 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 |

[a] H – High, G – Good, M – Moderate, P – Poor, B – Bad, and MP – Moderate Pollution.

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 sea water caused by the increased temperatures. The higher salinity in the tidal creeks than oceanic waters as a result of excess evaporation has been confirmed in TC by Nguli and Ryderg, (2007). 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 (Jacobs et al., 2020). 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 also 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 have been reported in the tidal creeks and estuaries, which are released through sediment resuspension during tidal mixing (e.g. Kitheka et al., 1996; Ohowa, 1996; Okuku et al., 2011; Mwashote, 1997). 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 porewater 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 nearshore sites. This suggests that the DIN was generated by remineralization processes under high organic matter and low oxygen conditions as reported by Hensen et al., (2006) and Wetzel, (2001). The effect is higher in the creeks and estuaries due to high organic matter supplied through direct discharges of untreated sewage and industrial effluents, open defecation, and urban run-off in the creeks (Kithiia and Majambo, 2020; Mwashote et al., 2005; Okuku et al., 2011; Tunje et al., 2016) and Sabaki and Ramisi rivers catchment areas that drain in MRR, NGO, and MLD, and RAM and MWZ (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). The importance of benthic fluxes as a source of nutrients in the area as observed during this study was also established in MTW Creek and FNZ Bay by Mutua et al. (2004).

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 De Quevedo et al., (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. While 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. 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 (Jacobs 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 and Kyewalyanga, 2015) and Pemba Channel (Sekadende et al., 2021). From the above discussion, it is clear that the local hydrodynamics process, degree of exposure to anthropogenic activities, and primary producers especially nitrogen-fixing cyanobacteria and seasons had a high influence on the physico-chemical 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; Wang et al., 2021). The spatial distribution of phytoplankton assemblages is discussed in the next paragraphs.

### *4.2. 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 DO links to nutrient supply through remineralization 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, this 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). Whether these ecological factors (succession, predation, and competition) also contributed to the distribution of phytoplankton assemblages in the sea turtle foraging sites is unclear, however, it remains an interesting likelihood that needs to be investigated.

An increase in longitude and temperature suggests offshore movement to the deeper waters with reduced water column nutrients (Mengesha et al., 1999; Tamooh et al., 2014), 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 (Lemley et al., 2019). This response was evident in the RAM estuary and also 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; Mariam et al., 2022; Mwashote, 1997) and other estuarine systems around the world (e.g. Arhonditsis et al., 2003; Nassar and Gharib, 2014; Sarker et al., 2018; Wang et al., 2021; 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. This is explained by high N:P and N:Si ratios established in about 80 % of the sites, implying P and Si limitations. This limitation maybe as a result of high consumption of P and Si during primary production or lower supply in relation to N from the supplying sources (Conley et al., 2009). However, our data points to high consumption rather than supply with the diatoms making the higher composition of phytoplankton community in the sites. Addition of P and Si is thus expected to cause an increase in primary production. Higher ammonium and phosphate concentrations coincided with higher *Chl a* and dinoflagellates in the tidal creeks while high silicate and ammonium concentrations corresponded with high biomass and abundance of harmful diatomic species in the estuarine areas, especially in RAM (Figs. 2 and 4A). The connection between high turbidity, ammonium, and phosphate and the development of non-diatomic 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 were, however, favored by a reduction in ammonium and phosphate concentrations due to their ability to generate nitrogen through fixation. 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 offshores, 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; Tunje et al., 2016) and similar coastal ecosystems (Arhonditsis et al., 2003; Glibert, 2017; Heil et al., 2007; IOC-UNESCO, 2008; 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.

### *4.3. 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 1: 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. Table 1. 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. Persistence occurrences of DO ≤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 and Álvaro, 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 (Gagliano et al., 2010; 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 (Tables 5 and 6). 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 (Fig. 3), 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), neighboring 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 and Julie, 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. There is huge evidence of potential threats of persistent occurrence of P and Si limitation with excessive inputs of DIN in the form of ammonium on the health of coastal ecosystems, many of which have been mentioned in earlier discussions of 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, MNPR, 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 fishermen 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 = 154 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 MNPR 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 and Lavaud, 2020). Domination of some species like *Nitzchia* 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 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 Heisler et al., (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 (*Protoperidinium* 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 and Mamboya, 2014) and Zanzibar (Limbu and 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 and Katana, 2022; van de Geer et al., 2022; The Standard Newspaper, 2021) ranged between 'poor' and 'bad' based on both *Chl a* (Fig. 4) 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 μM) 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 and 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.

### 4.4. Management implications

This study provides information on the locations, magnitude, and variability of ecological quality of the coastal waters which 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 are also critical fisheries and tourism sites that support the livelihoods of local communities (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 a total of 64 potentially harmful algal blooming species, which represented an additional seven (7) species of what was reported from a review of long-term data by 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 suggesting increased development of HAB species in Kenya's coastal waters. Kiteresi et al., (2013) and Ochieng 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, Ceratium* spp., *Dinophysis* sp., *Protoperidinium* 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 mentioned the increase in HABs and development of fleshy macro-algae in seagrass and coral habitats in the sites exposed to sewage discharges to be associated with the established DIN pollution, and benthic fluxes, agricultural run-offs, sewage discharges, and nitrogen-fixation as the major sources of nutrients the 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 which need to be considered when comparing the current results with future findings.

## 5. 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 flashed 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' as a result 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 enhanced development of harmful algal blooms. The increasing harmful algal blooming species development was supported by seven additional species recorded in this study in comparison to the previous studies (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, while 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.

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## CRediT authorship contribution statement

Conceptualization - N.A.O and N. M., Methodology - N. A. O., E. O. N., N. A. M., and C. M. M.; Investigation - N. A. O., E. O. N., N. A. M., and C. M. M.; Laboratory work - N. A. O. and E. O. N.; Data analysis - N. A. O., L. K. I, and P. K. B.; Writing - original draft preparation - N. A. O.; Writing - review and editing - N. M., C. N. M., P. K. B., L. K. I., E. O. N., N. A. M., and C. M. M.; Supervision - N. M., and C. N. M.; Funding acquisition - N.

A. O., and N. M. All authors have read and agreed to the published version of the manusript.

## Declaration of competing interest

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## Data availability

Data will be made available on request.

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## Appendix A. Supplementary data

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