

Monitoring the Spatio - Temporal Variations of Nutrients, Physico-Chemical Variables and Influences on the Fish Species within a Shallow Embayment of Winam Gulf (Lake Victoria, Kenya)

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Citation: Mwamburi Job, Aura C. M., Akama E., Mwanchi J., Sharon M., Nyamweya C. S., Achiya J. and Onduso G. (2026). Monitoring the Spatio - Temporal Variations of Nutrients, Physico-Chemical Variables and Influences on the Fish Species within a Shallow Embayment of Winam Gulf (Lake Victoria, Kenya). *Environmental Reports; an International Journal*.

DOI: <https://doi.org/10.51470/ER.2026.8.1.147>

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Received 12 January 2026 | Revised 08 February 2026 | Accepted 14 March 2026 | Available Online April 16 2026

ABSTRACT

Freshwater aquatic ecosystems are continuously exposed to increasing threats from anthropogenic activities, climate change and the expanding urban population. There is no long-term data and information to understand the effects of urbanization and watershed nutrient sources, on the ecological health of the several shallow embayment. This study explored the changes in environmental conditions and influences on fish species composition and distribution, using data collected between 2018 to 2024 (Dry and wet seasons) in Kisumu Bay. Over the seven years, the water temperature, turbidity, salinity, total dissolved solids, conductivity, dissolved oxygen, oxygen reduction potential, pH and Secchi depth revealed significant variations ($p < 0.05$) between the years, stations and sampling months. In 2023, a lower mean turbidity and higher conductivity, increased light transparency was recorded than the previous years in areas which are within the inner zone of the bay. Significant seasonal variations ($p < 0.05$) were evident for the measured turbidity, secchi depth and water temperature. Over the three years (2022-2024) nutrient elements varied significantly but differently across the years, stations and zones; except for dissolved silicon. The mean phytoplankton biomass (Chlorophyll-a) was $104.19 \pm 150.20 \mu\text{g L}^{-1}$ and the lake water exhibited eutrophic to hypereutrophic conditions. The two introduced fish species (*Lates niloticus* and *Oreochromis niloticus*) and the endemic *Haplochromines* and *Synodontis victoriae* were the dominant fish species. Watershed management and efficient waste management are a priority for reduction of soil erosion and control of diffuse pollution sources; and protection of the natural aquatic habitats.

Keywords: Lake Victoria; nutrients; water quality; Winam gulf, eutrophication.

1. INTRODUCTION

Globally, the increasing uncertainty from climate change effects and anthropogenic activities within the watershed areas; and the invasive species are still a concern in conservation of freshwater biodiversity. In Lake Victoria, notable recorded threats include, flooding events and long durations of the water hyacinth presence in the relatively shallow bay (less than 5m), which is also considered a "water hyacinth hotspot". In Kisumu bay, water hyacinth is known to have blocked boat transport and hampered artisanal fishing activities during periods of re-appearance and prolonged [1]. There has never been a clear indication of an ecological balance in invaded lake zones, as remnants of the water hyacinth plant still thrive. Therefore, the lake basin and the benthic habitats are continuously shaped by the living and non-living components. As a result of the increasing eutrophication, nuisance algal blooms are a common occurrence in the lake [2-6]. Recent studies document the presence of cyanotoxin producing phytoplankton species such as *Microcystis spp.*, which can cause impairment of the lake water, which serves as a source of domestic water supply within the Kisumu city and peri-urban areas [4,5; 7-12]; and other regions of the L. Victoria basin [3; 13-16].

In the Winam gulf, recent molecular studies to identify toxin producing cyanobacterial showed that within a relatively small embayment, composition and toxins synthesis potential of cyanobacterial harmful blooms can vary dramatically [8]. These reports confirm the changing limnological conditions, mainly attributed to watershed changes and unconfirmed impacts from the ongoing fish cage culture. However, there is still a need to ensure sustained fish production and utilization of marine and freshwater resources by prioritizing the potential opportunities in the blue economy. Cage fish culture in L. Victoria is a strategy geared towards increased production of Nile tilapia, and a more efficient utilization of the lake water resources, to ensure nutritional food security and improved livelihoods [17-22]. Embayments and riverine wetland areas within the Winam gulf and other bays of Lake Victoria are recognized fish habitats, and critical nursery and breeding areas for the diverse fish species. However, the close proximity to human settlements and expanding urbanization and other related water uses has meant an increasing threat to the resident biodiversity. Kisumu bay is an important coastal zone for the Kisumu County and the linked hinterland; due to its harbour operations and other maritime related activities and infrastructure.

Wild fish production has continued to dwindle, and hence the increasing presence of the new cage aquaculture infrastructure around the Lake Victoria (Kenya), to supplement Nile tilapia production. The endemic *Oreochromis variabilis* and *Labeo victorianus* are among the most threatened fish species, with a declining population size [23]. Similar to many aquatic organisms, the distribution and abundance of early life stages of fish are critical determinants of the population size, but the characteristics of nursery areas and spawning habitats are not clearly mapped and protected. Major changes in fish species in Lake Victoria, especially the diverse community of Haplochromines decline have been attributed to Nile perch [24]. In Lake Victoria, the predator Nile perch diet consists of *Haplochromines* spp., *Rastrineobola argentea*, tilapia, Nile perch juveniles and other small fish species [25-28], but juvenile Nile perch feeds mostly on *Caridina nilotica* [27], macroinvertebrates, copepods and fish fry [26]. Therefore, information to guide demarcation of recruitment habitats is certainly important for efficient aquatic conservation in areas surrounded by heavy anthropogenic impacts. The aquatic micro habitats around the bay are under threats from the increasing impacts from the expanding urban areas; urban runoffs and associated solid and liquid waste discharges into the littoral areas of the lake. Increasing levels of heavy metals, polycyclic aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) among other chemical residues; chronically stress fish. Wild freshwater fish could be susceptible to endocrine disruptor chemicals (EDCs), however only in a few cases a causal link between the presence of EDCs in freshwater fish and altered endocrine functions in exposed fish populations been demonstrated [29], but most of the focus on the effects of EDCs has been on effects at individual level. It is therefore hypothesized that there are unknown scales of threats to aquatic biodiversity, distribution and water quality conditions from complex ecological and biochemical stressors besides the increasing effects from local to regional climatic changes. Addressing these issues is in support of the new blue economy (BE) concept or strategy for safeguarding the world's ocean and water resources. The BE is generally defined as the sustainable use of ocean resources for economic growth, improved livelihoods, and jobs while preserving the health of ocean system [30-31] and is in-line with the United Nations sustainable development goals to reduce unsustainability and promote sustainable development [32].

Kisumu bay waters receives riverine and point discharges, urban runoffs and land sourced materials. Therefore, the study identified Kisumu bay, a more urban impacted zone of Lake Victoria (hereby also considered an "area of concern") to try and document the present ecological status in the wake of increasing cage aquaculture and frequent presence of the transient remnants of the floating water hyacinth within the Winam gulf. There are no detailed monitoring studies on the several bays of Winam gulf, apart from recent molecular studies to identify toxin producing cyanobacterial [8]. Shallow lakeshore areas are important fish habitats and serve as nursery grounds, but they are not clearly demarcated. Poor water quality conditions cause increased fish stress and mortality in cage farms, as evidenced by re-occurrence of fish kills that led to economic losses within the Winam gulf bays (near Kisumu and Homa bay, in November 2022 [33]). There are also concerns on potential effects of cyanotoxins in fish but scientific data and evidence is limited.

We hypothesize that the fish communities and juvenile growth stages are more affected by the poor water quality in relatively more exposed habitats around the urbanized shorelines besides the use of illegal and unregulated fishing gears. Therefore, the main objective of the study is to provide a zonal assessment of the fish communities and key environmental factors as indicators of ecological stress within the bay. The aims were to determine and monitor changes in key water quality parameters and nutrients elements, and their influence on fish species composition, so as to understand changes in aquatic communities; and to inform on the better use of the water resource and responsible blue economy exploitation for ensuring sustainable fisheries.

2. MATERIALS AND METHODS

Study area

Lake Victoria (East Africa) is a large freshwater lake with a lake surface area to catchment area ratio of about 1:2.83. The water surface area, mean and maximum depth, volume and catchment area of Lake Victoria (East Africa) is about 68,800 Km², 40 m, 80 m, 2760 Km³, and 195,000 Km², with about 6% and 43% of the lake water surface area in Kenya and Uganda respectively. This study was conducted in Kisumu Bay (Kisumu County) (Fig.1). The bay forms the extreme eastern end of the Kenyan waters of Lake Victoria (Winam gulf/Nyanza Gulf). Winam gulf is a relatively shallow and narrow basin which widens eastwards forming Homa Bay, Asembo Bay and Nyakach bay. The western section of the Nyanza Gulf deepens and opens to the main Lake Victoria around the Mbita and Rusinga channels. Winam Gulf extends to a length of about 60 km with a variable width of between ca.6 Km (narrow western section) and ca.30 km (Eastern section) and lies between longitude 34° 13' and 34° 52' East and 0° 4' to 0° 32' South of the equator. Kisumu bay is the waterfront area of Kisumu city, which support a large human population and associated infrastructure in the urban and per-urban areas. Kisumu county has a total surface area of about 2,677 Km² water mass of 567 Km² and a population of 1.16 million people [34-35]. Riparian counties support small-scale fishing as a major economic activity [36] with a fish consumption per capita ranging from 2.3 to 71.9 Kg/capita (average for Kenya 3.1 Kg/capita). Major urbanized rivers and streams (Kibos, Kisian and Kisat) flow from the neighboring highland areas in the North Eastern areas. The immediate catchment area is more dominated by human settlements and associated urban developments; cropland and grasslands; rice farms, sugarcane plantations and wetland areas.

Study design

Several past surveys have documented the socio-economic effects and impacts from water hyacinth (after initial invasion in L. Victoria in 1988); and the changing face of the Winam gulf and the bay surface waters between June 1995 and November 1999 [36-39]. There are also increasing concerns on the presence of the algal blooms and associated exposure risks of hepatotoxins (e.g Microcystins) to communities abstracting and using water directly from the lake bays [4-5, 9, 40]. Again, Nile tilapia cage culture is the fastest growing aquaculture production technology in Kenya with 3,696 cages (2018) located along the shores of the five Lake Victoria riparian counties [18-19, 41]. Around the bay the cages are located on the outer areas of the bay near Dunga and Maboko Islands [18, 42-43], to avoid the heavy navigational traffic into the Kisumu bay and due to the bathymetry limitations.

Therefore, the shallow bay is exposed to increasing environmental stressors and require frequent monitoring of the water quality, and development of clear early warning systems to avoid the commonly occurring fish kills.

Physico-chemical parameters and nutrients

Sixteen lake sampling stations were sampled to represent the inner and outer zones of the Kisumu bay which experiences both the long and short rains between the months of March – June and October- December respectively. The inner zone sites are representative of the urban impacted and discharge zones (KSB2 – KSB6, KSB8 – KSB9 and KSB15 – KSB16) and the impact gradient is expected to decrease towards the outer zone sites (KSB-1, KSB7 and KSB10 - KSB14). However, due to logistical challenges, the sampling and setting of fish nets was done within the inner bay zone. In-situ measurements of the vertical profile of water temperature, water depth, pH, dissolved oxygen (D.O), total dissolved solids (TDS), salinity, conductivity and redox potential (ORP) were made using the multi-parameter water quality monitoring meter (YSI – 650 multi-parameter, Professional Series handheld meter; YSI Incorporated, USA). Water samples were collected at 10cm below the water surface at each sampling site and kept in a cooler box for later laboratory analysis of nutrients (Nitrate –nitrogen ($\text{NO}_3\text{-N}$); Nitrite –nitrogen ($\text{NO}_2\text{-N}$), Ammonium- nitrogen ($\text{NH}_4\text{-N}$), total nitrogen (TN), total phosphorus (TP), soluble reactive phosphorus (SRP) and soluble reactive silicon (SRSi)). Dissolved nutrient species were analysed in filtered water samples and the unfiltered portion of the sample was used to analyse TN and TP. The standard methods for the analysis of water and wastewater [44] were employed for the preservation and storage of samples in the field, and laboratory analysis.

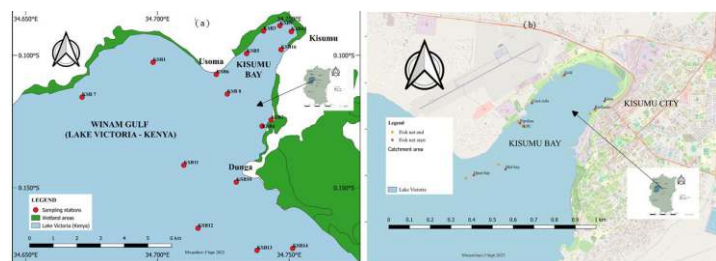


Figure 1: Map of Kisumu Bay showing the sampling stations for water quality measurements (a) and fish sampling (b)

Sampling for fish

Fish surveys were conducted within the bay during different months between 2021 unto August 2023 (Fig. 1b), using different monofilament nets of mesh sizes ranging from 0.5", 0.75", 1.0", 1.25", 1.5", 1.75", 2.0", 2.5", 3.0", and 3.5". The caught fish samples were sorted, and individual fish species identified. The fish length (Total length, fork length and standard length were measured in cm), fresh weights (g), sex ratios, maturity stages were determined according to the standard operating procedures (SOPs) manual [45]. Fish species diversity was determined using the species richness and equitability indices [46-48].

Data analysis

The data exploration and analysis was conducted using R Core Team [49] software package. The assumption of normality distribution of data (homogeneity of variance) was tested and confirmed using the Kolmogorov – Smirnov (KS) and Shapiro - Wilk statistical tests.

The significant differences in water quality variables over the years and between zones was evaluated by analysis of variance (ANOVA) and comparison tests using 0.05 as the level of significance. The non-normal distributed data was analysed using the non-parametric statistical tests. Significant relationships between variables were assessed using Spearman's rho correlation two-tailed test for non-parametric data. The significantly different pairwise variations in the mean values were separated using Wilcoxon test and T - test (for normally distributed variables). Fish Environmental influences on the fish size composition and diversity were determined.

3. RESULTS

The surface waters of Kisumu bay exhibited variations in water quality (Table 1a & 1b) with an overall mean (\pm SD) water depth, transparency, total alkalinity, hardness, and elevation of 3.7 ± 1.2 m, 0.53 ± 0.2 m, 58 ± 8 mgL^{-1} , 50 ± 8 mgL^{-1} and 1142 ± 4 m. a. s. l; whereas the bay water column temperature, conductivity, TDS, turbidity, salinity and chlorophyll-a showed an overall mean (\pm SD) of 27.23 ± 0.99 $^{\circ}\text{C}$, 0.187 ± 0.019 mScm^{-1} , 0.119 ± 0.012 gL^{-1} , 14.10 ± 10.18 NTU, 0.09 ± 0.01 gL^{-1} and 45.88 ± 61.60 μgL^{-1} respectively. The in-situ profiles of water temperature, conductivity, TDS, turbidity, salinity and chlorophyll-a at each site (2023-2024 study) revealed significant variations between the surface and bottom water layers with respect to TDS, salinity and temperature but the variations of turbidity and chlorophyll-a were not significantly different. Seasonal differences were also evident for TDS, salinity and temperature (Table 1a). The water column exhibited significant spatial variations ($p < 0.05$) in the mean water temperature, conductivity, TDS, turbidity and chlorophyll-a except, the Secchi depth and total water depth. Salinity was uniform and varied slightly at sites KSB 15 (Off Kichinjio) and KSB 9 (Off Kisat RM). Similarly, a comparison between the outer and inner bay areas showed a significant difference between all the variables ($p < 0.05$). The water column temperature, turbidity, conductivity, TDS and phytoplankton biomass was much higher within the inner bay waters. The mean (\pm SD) temperature ranged from 25.56 ± 0.08 $^{\circ}\text{C}$ (KSB13) to 28.33 ± 0.64 $^{\circ}\text{C}$ (KSB9). The mean water conductivity and TDS varied from 0.171 ± 0.04 mScm^{-1} (KSB14) to 0.222 ± 0.00 mScm^{-1} (KSB15) and 0.120 ± 0.001 gL^{-1} (KSB13) to 0.137 ± 0.001 gL^{-1} (KSB15) between the stations respectively. The mean chlorophyll-a concentration (proxy for phytoplankton biomass) and water turbidity varied from 18.83 ± 5.83 μgL^{-1} (KSB11) to 94.90 ± 11.1 μgL^{-1} (KSB5) and 8.4 ± 1.9 NTU (KSB11) to 42.3 ± 0.1 NTU (KSB15) respectively. We observed significant relationships between the water quality variables (Table 2a & 2b) except for the sampling depth versus conductivity, salinity chlorophyll-a and turbidity. The secchi depth and total water depth revealed a significant positive relationship, whereas all the rest of the parameters had a variable negative and significant associations with both the total depth and the light transparency.

The trophic state index (TSI) was calculated based on Carlson [50] equations of the total phosphorus, secchi depth and chlorophyll-a concentration. The bay TSI_(Chlorophyll-a), TSI_(Secchi depth) and TSI_(TP) values ranged from 34.60 to 94.97 (mean = 66.96 ± 10.31), 61.50 to 83.19 (mean = 69.80 ± 4.88) and 51.30 to 88.01 (mean = 77.55 ± 11.75) respectively, with a mean TSI for the bay (TSI_(bay)) of 68.91 ± 7.04 (range 53.40 – 89.63). The minimum and maximum mean TSI_(bay) of the bay was 61.41 ± 2.98 (KSB 11) and 73.28 ± 5.30 (KSB 9) respectively.

The inner bay waters showed significantly higher values ($p < 0.05$) than the outer zone, but overall, all the indices reflected a eutrophic to hypereutrophic status of the bay waters based on the total phosphorus, phytoplankton biomass and secchi depth. The concentration of nitrogen species (μM) during the study (2023-2024) period ranged from 0.05 – 0.69 (0.36 ± 0.16), 0.0 – 0.52 (0.14 ± 0.13), 1.38 – 58.16 (10.06 ± 13.62) and 2.21 – 57.01 (15.77 ± 14.33) for $\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$, $\text{NH}_4\text{-N}$ and TN respectively (Table 1b). The levels of ammonium -N were the highest among the dissolved N species then followed by the oxidized forms. Much higher mean (\pm SD) levels of $\text{NH}_4\text{-N}$ were observed at the extreme inner zone sites at KSB15, KSB3 and KSB9. The mean minimum and maximum levels of both the nitrate and nitrite forms of N were highest at inner zone sites KSB13 and KSB15 and KSB6 and KSB2 respectively. The mean TN levels were similarly much higher within the inner sites and ranged from a mean minimum value of $6.57 \mu\text{M}$ (KSB14) to $31.93 \mu\text{M}$ (KSB3). There were no significant zonal variations between the N species, except for the significant temporal variations in nitrate – N and ammonium – N concentrations. The levels of dissolved silicon, soluble P and total phosphorus in surface lake water varied from 0.07 – 0.60 mM (0.35 ± 0.16 mM), 0.03 – 5.79 μM ($0.93 \pm 1.24 \mu\text{M}$) and 0.85 – 13.92 μM ($4.20 \pm 3.56 \mu\text{M}$) respectively. The inner bay areas recorded much higher mean levels of the soluble and total P nutrient elements as compared to the outer zone areas (Fig. 2), but only the zonal mean levels of TP and SRP were significantly different. All the measured mean nutrient elements were significantly different among the months (except for SRP, TP and nitrite – N levels), and also amongst the years (except for TP, nitrite – N, SRP and silica-Si). The total nitrogen and total phosphorus molar ratio ranged from 0.46 to 24.63 with an overall mean (\pm SD) value of 5.28 ± 4.51 . The mean N/P ratio ranged from a minimum of 2.18 ± 1.12 (KSB7) to a maximum value of 10.44 ± 12.38 (KSB8) within the outer and inner zone stations respectively. However, the means values were not significantly different among the two zones. The ratio depicts the established nitrogen deficiency for the phytoplankton production in the waters of Winam gulf of Lake Victoria (Kenya) as reported earlier in different studies.

The surface water concentrations of total P and N species were strongly and moderately correlated ($p < 0.05$) with soluble ammonium nitrogen (Fig 2 & Table 2a & 2b). Similarly, the dissolved oxidized forms of nitrogen were positively and moderately associated, whereas TP was positively correlated ($p < 0.05$) with levels of soluble reactive silica and SRP. All the nutrient elements (except silica-Si) revealed a moderate but significant relationship with turbidity and chlorophyll-a. The turbidity and phytoplankton biomass proxy (chlorophyll- a) in surface water showed a high significant positive (TP-turbidity) and negative (Chlorophyll-a - TP) correlation ($p < 0.05$) with total phosphorus in surface water. Surface water temperature was only significantly and positively correlated with TP and dissolved silica Si ($p < 0.05$). Other significant relationships were observed between the water conductivity and TP and dissolved ammonium-N. Water depth and transparency were mostly negatively associated with measured nutrients ($p < 0.05$) except silica Si. The TSI also exhibited moderate negative (secchi and total depth, water hardness and total depth) and positive relationships ($p < 0.05$) with most of the environmental variables (TN, N/P ratio, nitrate, chlorophyll-a, turbidity, conductivity).

The inter annual spatial and temporal trends in water quality in the past 7 years

We explored the changes in environmental conditions using data collected between 2018 to 2022 (Table 1c) which covered both the dry and wet season months. Over the seven years, the mean annual water temperature, turbidity, salinity, total dissolved solids, conductivity, D.O, ORP, pH and secchi depth revealed variations between the years, sampling months and stations (Fig. 2). Overall, between 2018 – 2024, the highest zonal means were observed for water column salinity, temperature, pH and turbidity within the inner zone, whereas the highest water D.O, ORP and Secchi depth was recorded in the outer zone. There were significant ($p < 0.05$) annual ($n = 588$) and seasonal ($n = 316$) differences in values of the measured parameters except for the D.O levels between the seasons.

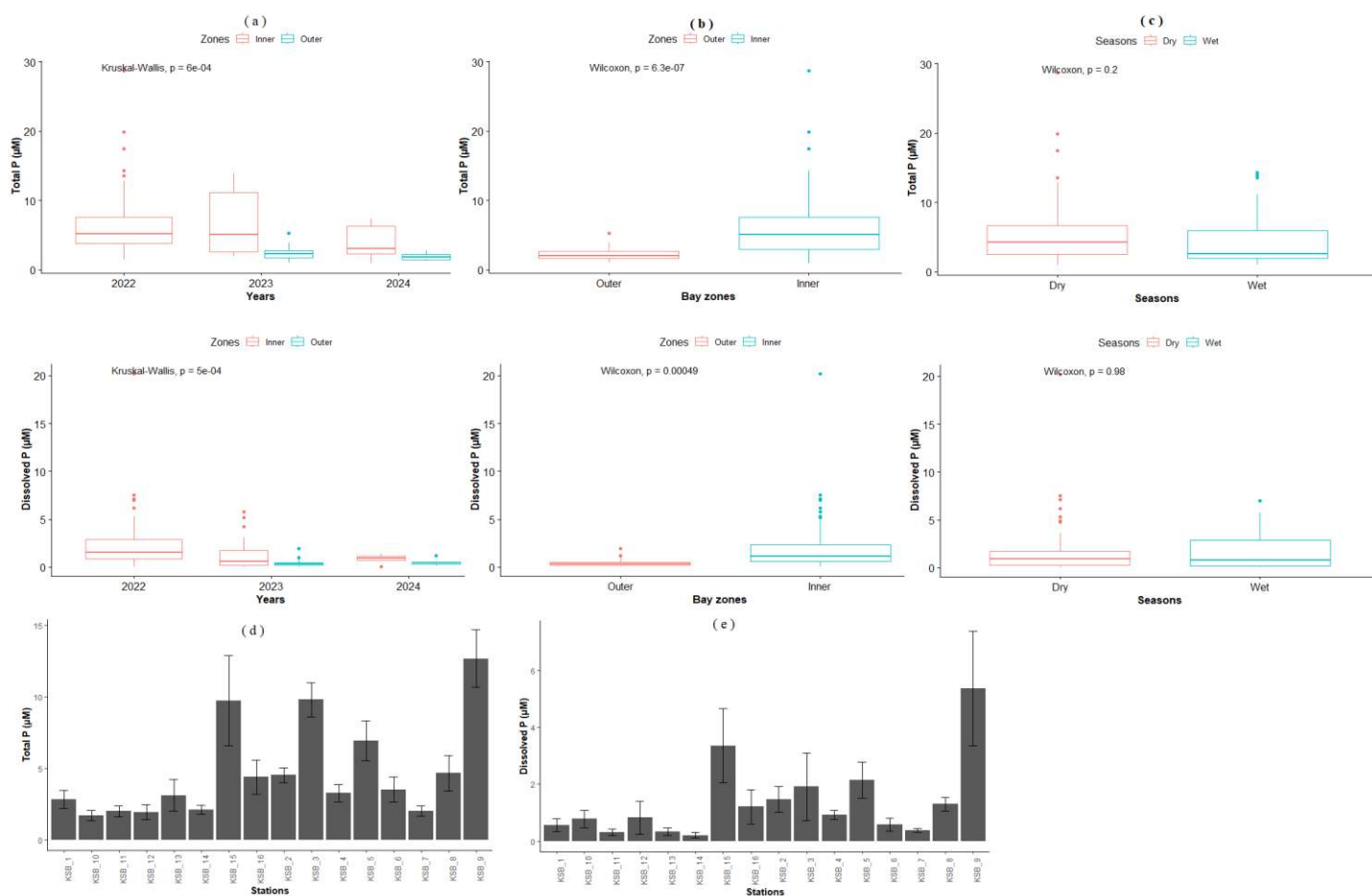
The annual mean water pH and D.O fluctuated from a low value of 7.77 ± 0.48 (2020) and $4.17 \pm 1.37 \text{ mgL}^{-1}$ (2020), to a maxima of 7.99 ± 0.39 (2019) and $5.91 \pm 1.17 \text{ mgL}^{-1}$ (2018) respectively. The overall mean D.O, pH and ORP were 4.60 ± 1.64 , 7.88 ± 0.43 and -164.34 ± 120.63 respectively. Trends of the D.O levels showed a general decline across the years from 2018 to 2020 followed by a wider variation and increasing trend in 2021. Over the seven years significant associations emerged between the surface water quality parameters (total water depth, turbidity, chlorophyll – a and transparency) and nutrient elements in Kisumu Bay (Table 2a and 2b).

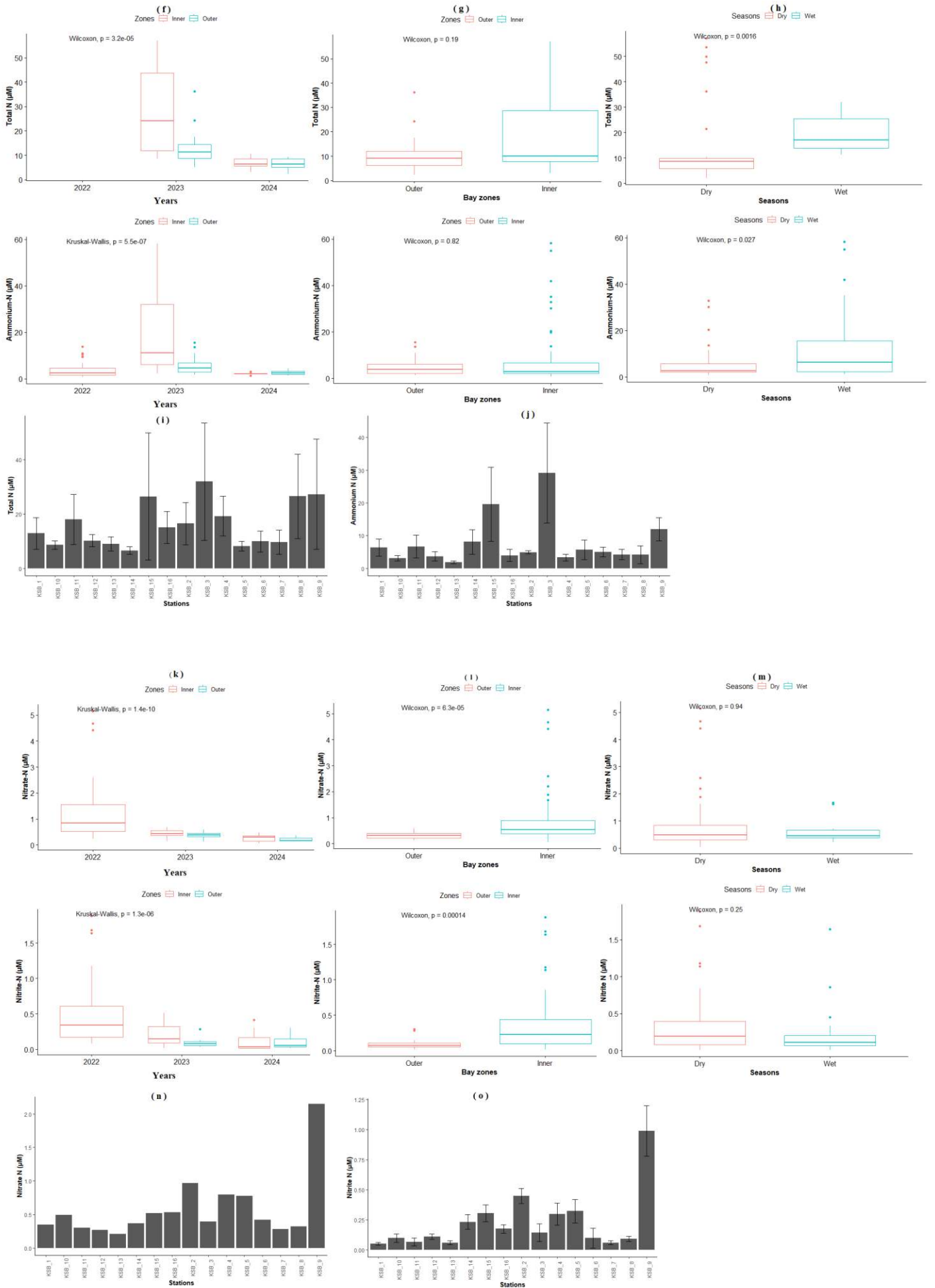
Table 1a: The mean (\pm SD) values of the water quality variables ($n = 195$ for all variables, except turbidity and chlorophyll- a where $n = 193$ during the dry (2023 and 2024 $n = 133$) and wet (2023) season months across the sampling sites (Sample size for nutrient elements (2023 to 2024) in surface water = 32 (Dry) and 16 (Wet); Significantly different values ($p < 0.05$) between the seasons are indicated with a different alphabetical letter)

Parameter	Mean \pm SD	SE	Min	50p	Max	Season
Total depth (m)	3.9 ± 1.3	0.2	1.0	4.1	5.7	Dry
	3.3 ± 1.2	0.3	1.2	3.5	4.4	Wet
Secchi (m)	0.55 ± 0.14	0.02	0.3	0.6	0.9	Dry
	0.49 ± 0.18	0.05	0.2	0.5	0.8	Wet
Temp. (°C)	27.03 ± 0.8	0.08	25.47	26.97	29.93	Dry
	27.70 ± 1.02	0.13	26.54	27.53	29.97	Wet
Cond (mS cm^{-1})	0.181 ± 0.02	0.002	0.100	0.188	0.222	Dry
	0.198 ± 0.009	0.001	0.184	0.195	0.234	Wet
TDS (g L^{-1})	0.114 ± 0.012	0.001	0.090	0.121	0.137	Dry
	0.129 ± 0.006	0.001	0.120	0.126	0.152	Wet
Turb. (NTU)	12.53 ± 7.15	0.62	2.6	10.9	42.4	Dry
	17.42 ± 14.17	1.8	5.4	11.7	75.5	Wet
Chl-a ($\mu\text{g L}^{-1}$)	36.12 ± 25.23	2.20	1.3	24.5	106.3	Dry
	66.50 ± 99.75	12.67	10.3	31.05	453.40	Wet
Sal.	0.08 ± 0.01	0.00	0.07	0.09	0.10	Dry
	0.09 ± 0.00	0.00	0.09	0.09	0.11	Wet
TP (μM)	4.05 ± 3.11	2.29	0.85	2.69	12.35	Dry
	4.49 ± 4.49	1.16	1.01	2.39	13.92	Wet
SRP (μM)	0.67 ± 0.52	0.08	0.06	0.53	1.96	Dry
	1.45 ± 1.96	0.49	0.03	0.47	5.79	Wet
TN (μM)	14.41 ± 15.96	2.87	2.23	8.68	57.01	Dry
	19.59 ± 7.53	2.27	11.25	17.00	31.99	Wet
NO3-N (μM)	0.32 ± 0.17	0.03	0.05	0.31	0.69	Dry
	0.43 ± 0.12	0.03	0.23	0.41	0.67	Wet
NO2-N (μM)	0.15 ± 0.15	0.03	0.01	0.09	0.52	Dry
	0.11 ± 0.09	0.02	0.00	0.09	0.34	Wet
NH4-N (μM)	6.52 ± 7.69	1.36	1.38	4.04	32.79	Dry
	17.13 ± 19.48	4.87	1.60	8.55	58.16	Wet
SiO2-Si (mM)	0.44 ± 0.12	0.02	0.22	0.43	0.60	Dry
	0.17 ± 0.05	0.01	0.07	0.18	0.23	Wet
NP molar ratio	3.77 ± 2.87	0.51	0.46	3.21	15.14	Dry
	7.20 ± 3.76	1.42	1.30	6.49	24.63	Wet
TSI	67.60 ± 6.52	1.15	53.40	66.65	79.00	Dry
	71.54 ± 7.53	1.88	63.90	69.62	89.63	Wet

Table 1b: A comparison of the dissolved nutrient concentrations, total alkalinity, hardness and phytoplankton biomass between 2022 and 2024 ($p = \text{Kruskal-Wallis test } p - \text{value}$)

Parameter		Years						p-values
		2022		2023		2024		
TP (μM)	$\bar{x} \pm \text{SD}$	7.03	± 5.58	4.79	± 4.02	3.05	± 2.10	0.000
	Range	1.40-	28.70	1.01-	13.92	0.85-	7.42	
SRP (μM)	$\bar{x} \pm \text{SD}$	2.63	± 3.48	1.02	± 1.49	0.75	± 0.42	0.000
	Range	1.36-	20.19	0.03-	5.79	0.06-	1.36	
TN (μM)	$\bar{x} \pm \text{SD}$	na		20.97	± 16.10	6.56	± 2.48	0.000
	Range			5.16-	57.01	2.23-	10.35	
$\text{NH}_4\text{-N}$ (μM)	$\bar{x} \pm \text{SD}$	3.65	± 3.18	13.91	± 15.34	2.36	± 0.81	0.000
	Range	0.65-	13.90	1.60-	58.16	1.38-	4.40	
$\text{NO}_3\text{-N}$ (μM)	$\bar{x} \pm \text{SD}$	1.20	± 1.15	0.42	± 0.15	0.24	± 0.12	0.000
	Range	0.23-	5.14	0.13-	0.69	0.05-	0.48	
$\text{NO}_2\text{-N}$ (μM)	$\bar{x} \pm \text{SD}$	0.50	± 0.4	0.15	± 0.13	0.11	± 0.13	0.000
	Range	0.08-	1.88	0-	0.52	0.01-	0.42	
$\text{SiO}_2\text{-Si}$ (mM)	$\bar{x} \pm \text{SD}$	0.38	± 0.23	0.36	± 0.20	0.32	± 0.04	0.650
	Range	0.04-	1.24	0.07-	0.60	0.22-	0.37	
Chl-a (ugL^{-1})	n	41		32		16		0.097
	$\bar{x} \pm \text{SD}$	127.79	± 176.40	52.23	± 72.82	33.36	± 24.92	
	Range	2.32-	715.81	10.3-	453.4	1.30-	97.0	
Alk (mgL^{-1})	n	41		128		65		0.000
	$\bar{x} \pm \text{SD}$	56	± 15	63	± 8	53	± 3	
	Range	42	12	54	82	48	58	
Hard (mgL^{-1})	n	41		16		16		0.000
	$\bar{x} \pm \text{SD}$	40	± 11	47	± 9	53	± 3	
	Range	26	92	32	62	48	58	
	n	41		16		16		





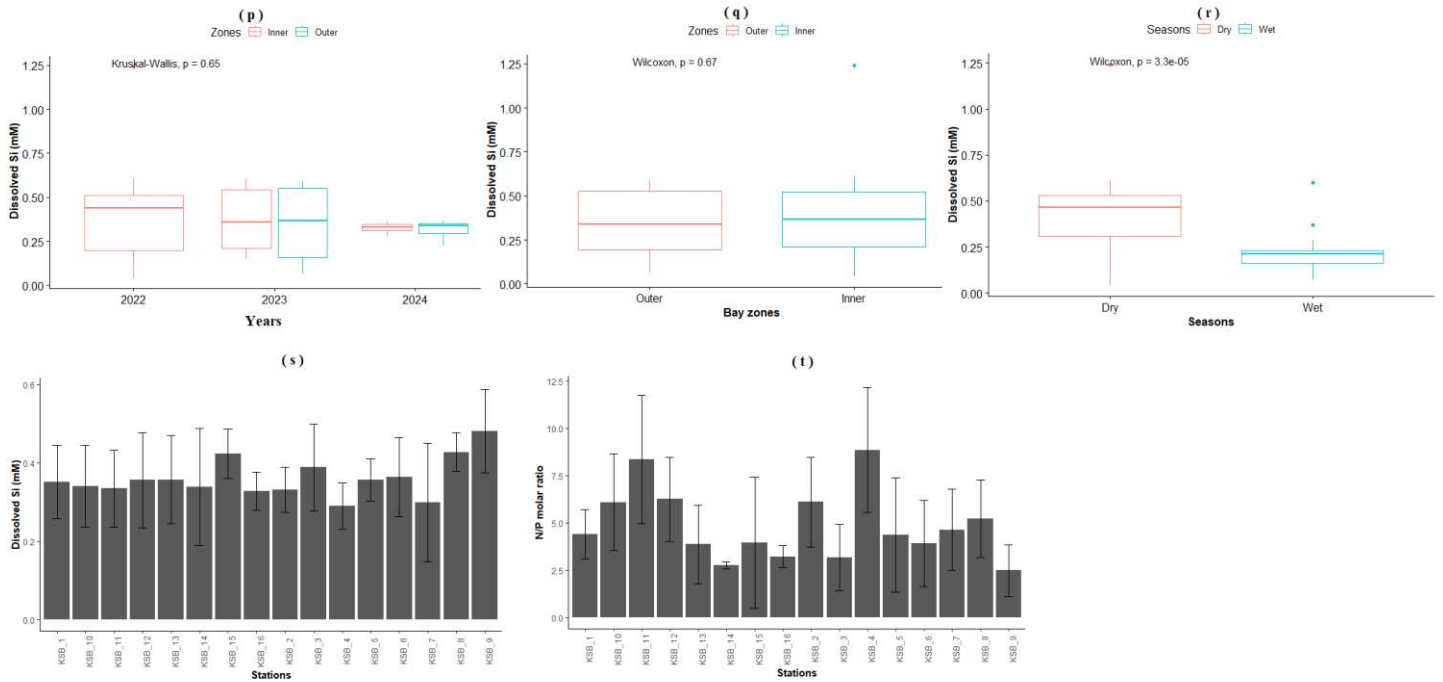


Figure 2 (a-t): The spatial mean (\pm SD) nutrient concentrations (μ M) of the Total P and N, dissolved P, Si (mM), nitrate - N and nitrite - N in lake surface water and the significant variations among the zones, seasons and sampling years from 2022 to 2024.

However, the D.O levels were not significantly different among the seasons. The monthly mean D.O levels in the bay revealed well oxygenated waters, but with relatively lower levels around the months of January - July when compared to September - December. Hypoxic D.O levels (below 2.0 mgL^{-1}) were only observed during the months of April (2019), January, February, November and October (2021). The water pH varied slightly and was near neutral (mean range of $7.77 \pm 0.4 - 7.99 \pm 0.39$) with an ORP value reflecting reducing conditions.

Zonal water temperature variations were more pronounced than seasonal differences. However, a wider variation appeared within the inner zone and wet season months. Water temperature across the seven years ranged from $22.42 \text{ }^\circ\text{C}$ to $30.0 \text{ }^\circ\text{C}$ (mean range of $26.64 \pm 0.55 - 27.56 \pm 0.88 \text{ }^\circ\text{C}$) with values lower than $24.0 \text{ }^\circ\text{C}$ recorded during February 2019 September 2021 and much higher levels between $27.0 \text{ }^\circ\text{C}$ to $29.0 \text{ }^\circ\text{C}$ recorded from 2021 to 2024. Over the seven years, the overall mean turbidity, secchi depth and temperature were $41.36 \pm 23.7 \text{ NTU}$, $0.37 \pm 0.1 \text{ m}$ and $26.72 \pm 1.1 \text{ }^\circ\text{C}$ respectively. The annual mean water temperature fluctuated from a low value of $26.11 \pm 1.18 \text{ }^\circ\text{C}$ (2021) to a maxima of $28.10 \text{ }^\circ\text{C}$ (2023). Mean inter-annual turbidity peaked at $95.75 \pm 11.6 \text{ NTU}$ and decreased to a low mean value of 16.6 ± 11.4 and 9.17 ± 3.74 in 2023 and 2024 respectively. This was reflected in the unsteady increase in water transparency, which was within a narrow mean range of 0.29 ± 0.07 to $0.53 \pm 0.18 \text{ m}$. However, the water turbidity level was above the 5 Nephelometric turbidity units, a level recommended for drinking water quality (WHO 1998). Monthly light transparency remained above 0.1 m up-to 2020 and increased from May-June 2021 to 2024. Monthly turbidity declined progressively over the years with slight peaks over the wet months. A lower mean turbidity and higher conductivity and light transparency was recorded in 2023 than previous years in areas which are within the inner zone of the bay. (Fig. 4a-b).

Significant spatial and temporal variations ($p < 0.05$) were evident for the measured turbidity, secchi depth and temperature (except for the zonal temperature). The water conductivity and TDS declined over the seven years with overall mean values of $0.177 \pm 0.026 \text{ mScm}^{-1}$ and $0.10 \pm 0.019 \text{ gL}^{-1}$ respectively. The highest values of conductivity, salinity and TDS were associated with areas around the Kisat river mouth (inner bay) and revealed significant variations ($p < 0.05$) between the years, stations, seasons and zones.

Nutrient elements, phytoplankton biomass, total alkalinity and hardness were not measured from 2018 to 2021 (Table 1c). Over the three years (2022-2024) total N, ammonium - N and silica revealed significant variation across the seasons and months, except for SRP, TP and oxidized N forms. The levels of nutrient elements also varied across the years (except for silica), zones (except for TN, ammonium - N and silica) and stations (except for TN and silica). The levels of SRP and silicon varied between the sampling months. Over the three years, the phytoplankton biomass measured in the water column varied significantly ($p < 0.05$) across the stations and bay zones (Figure 6). The chlorophyll - a levels ranged from 2.32 to $715.1 \text{ } \mu\text{gL}^{-1}$ with an overall mean of $91.83 \pm 138.96 \text{ } \mu\text{gL}^{-1}$ (Table 1b) over the three years. The maximum annual mean of $127.79 \pm 176.40 \text{ } \mu\text{gL}^{-1}$ was recorded in 2022 with much lower mean levels from September 2023 to February 2024. Of the total N, the oxidized N forms ranged from 0.39% to 15.88% , with much higher levels of ammonium - N (mean = $7.11 \pm 10.70 \text{ } \mu\text{M}$). The mean (\pm SD) dissolved P, total P, $\text{NO}_3\text{-N}$, $\text{NO}_2\text{-N}$ and $\text{SiO}_2\text{-Si}$ concentration (2022 - 2024) in Kisumu bay was $1.71 \pm 266 \text{ } \mu\text{M}$, $5.52 \pm 4.80 \text{ } \mu\text{M}$, $0.75 \pm 0.89 \text{ } \mu\text{M}$, $0.31 \pm 0.38 \text{ } \mu\text{M}$, $0.36 \pm 0.19 \text{ mM}$ respectively (TN was not measured in 2022). Water hardness and total alkalinity ranged $26 - 92 \text{ mgL}^{-1}$ (mean = $40 \pm 11 \text{ mgL}^{-1}$) and $42 - 128 \text{ mgL}^{-1}$ (mean = $56 \pm 15 \text{ mgL}^{-1}$) with an overall three years mean of $44 \pm 10 \text{ mgL}^{-1}$ and $57 \pm 12 \text{ mgL}^{-1}$ respectively. These concentrations both varied significantly between years, zones (except alkalinity) and months.

Table 1c: The annual mean (\pm SD) values of the water quality variables between 2018 to February 2024 in Kisumu bay

Years		Water quality parameters								
		Temperature (°C)	Conductivity (mScm ⁻¹)	TDS (gL ⁻¹)	Salinity (gL ⁻¹)	Turbidity (NTU)	Secchi depth (m)	D.O (mgL ⁻¹)	pH	ORP (mV)
2018	Mean (\pm SD)	26.81 \pm 1.41	0.172 \pm 0.011	0.090 \pm 0.12	0.08 \pm 0.00	95.75 \pm 11.60	0.30 \pm 0.04	5.91 \pm 1.17	7.90 \pm 0.56	-231 \pm 31.29
	Min	23.60	0.156	0.08	0.07	80.23	0.20	2.89	6.69	-293.700
	Max	27.80	0.187	0.11	0.08	125.75	0.35	7.98	8.56	-202.350
	n	24	24	24	24	24	24	24	24	24
2019	Mean (\pm SD)	26.80 \pm 0.76	0.157 \pm 0.017	0.097 \pm 0.005	0.071 \pm 0.006	59.40 \pm 12.84	0.29 \pm 0.07	4.53 \pm 1.33	7.99 \pm 0.39	-124.99 \pm 31.73
	Min	22.42	0.14	0.09	0.06	44.00	0.10	1.26	7.16	-187.70
	Max	28.40	0.229	0.11	0.08	99.30	0.45	10.28	9.80	111.20
	n	172	172	172	172	157	172	172	172	172
2020	Mean (\pm SD)	26.64 \pm 0.55	0.198 \pm 0.024	0.118 \pm 0.018	0.09 \pm 0.01	52.34 \pm 7.20	0.34 \pm 0.07	4.17 \pm 1.37	7.77 \pm 0.48	-165.76 \pm 51.26
	Min	25.70	0.16	0.08	0.01	38.80	0.20	1.33	6.47	-297.80
	Max	28.60	0.25	0.16	0.12	66.00	0.50	7.49	9.11	117.00
	n	112	112	112	112	112	112	112	112	112
2021	Mean (\pm SD)	27.17 \pm 1.08	0.169 \pm 0.024	0.100 \pm 0.024	0.12 \pm 0.15	43.21 \pm 4.29	0.42 \pm 0.05	4.33 \pm 2.28	7.77 \pm 0.39	- 201.16 \pm 231.30
	Min	23.30	0.12	0.01	0.05	31.00	0.30	0.07	6.63	-2362.20
	Max	30.00	0.32	0.21	0.60	49.70	0.50	10.26	8.53	-51.40
	n	98	98	98	98	77	98	98	90	98
2022	Mean (\pm SD)	27.56 \pm 0.88	0.174 \pm 0.037	0.099 \pm 0.031	0.16 \pm 0.19	34.85 \pm 2.86	0.47 \pm 0.07	5.73 \pm 1.05	7.95 \pm 0.37	-191.28 \pm 91.92
	Min	26.49	0.12	0.06	0.06	30.10	0.40	3.10	7.01	-297.00
	Max	30.00	0.24	0.15	0.60	39.70	0.55	10.26	8.77	259.10
	n	52	52	52	52	38	52	51	52	52
2023	Mean (\pm SD)	27.08 \pm 1.08	0.196 \pm 0.012	0.126 \pm 0.006	0.09 \pm 0.00	16.60 \pm 11.44	0.53 \pm 0.18	na	na	na
	Min	25.47	0.18	0.12	0.09	5.40	0.20			
	Max	29.97	0.15	0.24	0.11	75.50	0.90			
	n	130	130	130	130	128	32			
2024	Mean (\pm SD)	27.53 \pm 0.6	0.169 \pm 0.016	0.104 \pm 0.009	0.07 \pm 0.01	9.17 \pm 3.74	0.53 \pm 0.14	na	na	Na
	Min	26.41	0.150	0.09	0.07	2.60	0.20			
	Max	28.93	0.202	0.124	0.09	16.10	0.70			
	n	65	65	65	65	65	16			
Dry season	Mean (\pm SD)	26.59 \pm 1.21	0.174 \pm 0.025	0.107 \pm 0.016	0.10 \pm 0.10	37.86 \pm 23.35	0.36 \pm 0.12	4.65 \pm 1.53	7.95 \pm 0.41	- 174.84 \pm 153.61
	n	385	385	385	385	363	284	247	246	248
Wet season	Mean (\pm SD)	26.90 \pm 0.90	0.180 \pm 0.027	0.109 \pm 0.023	0.08 \pm 0.02	46.70 \pm 23.29	0.37 \pm 0.11	4.55 \pm 1.77	7.80 \pm 0.45	-151.95 \pm 60.40
	N	268	268	268	268	238	222	210	204	210
Inner zone	Mean (\pm SD)	26.724 \pm 1.16	0.176 \pm 0.028	0.106 \pm 0.020	0.10 \pm 0.09	47.44 \pm 20.98	0.35 \pm 0.10	4.58 \pm 1.66	7.91 \pm 0.42	- 165.26 \pm 123.67
	N	548	548	548	548	498	485	434	427	435
Outer zone	Mean (\pm SD)	26.69 \pm 0.73	0.179 \pm 0.018	0.115 \pm 0.012	0.08 \pm 0.01	11.96 \pm 10.32	0.63 \pm 0.14	5.00 \pm 1.20	7.51 \pm 0.51	-147.03 \pm 15.73
	n	105	105	105	105	103	21	23	23	23
All	Mean (\pm SD)	26.72 \pm 1.10	0.177 \pm 0.026	0.108 \pm 0.019	0.09 \pm 0.08	41.36 \pm 23.70	0.37 \pm 0.11	4.60 \pm 1.64	7.88 \pm 0.43	- 164.34 \pm 120.63
	Min	22.42	0.100	0.01	0.01	2.60	0.10	0.07	6.47	-2362.20
	Max	30.00	0.317	0.21	0.60	125.75	0.90	10.28	9.80	259.10
	n	653	653	653	653	601	506	457	450	458

Table 2a: The Spearman's correlation coefficients (significant = bolded) of the relationships between the water quality parameters in Kisumu Bay (Sample size = 651 for data collected from 2018 to February 2024; except for D.O = 472; pH = 450; ORP = 458; Turbidity = 601 Secchi depth = 504; Alkalinity and hardness = 68; TSI_{TP}, TSI_{Chla}, TSI_{Secchi} & TSI_{Bay} = 83; Chlorophyll-a = 193; total depth = 44 Total nitrogen = 42; N:P ratio = 42). The lower value represents the exact p value)

	T depth (m)	S depth (m)	Temp (°C)	Cond (mScm ⁻¹)	TDS (gL ⁻¹)	Sal (gL ⁻¹)	Turb (NTU)	Secchi (m)	Alk (mgL ⁻¹)	Hard (mgL ⁻¹)	Chla (ugL ⁻¹)	D.O (mgL ⁻¹)	ORP (mv)	pH
T depth (m)	1.000													
S depth (m)	0.187	1.000												
Temp (°C)	-0.436	-0.096	1.000											
Cond (mScm ⁻¹)	-0.578	-0.040	0.288	1.000										
TDS (gL ⁻¹)	-0.502	0.000	0.376	0.856	1.000									
Sal (gL ⁻¹)	-0.506	0.022	0.297	0.829	0.764	1.000								
Turb (NTU)	-0.430	-0.211	0.639	-0.200	0.327	0.148	1.000							
Secchi (m)	0.565	-0.014	0.249	0.203	0.197	0.148	0.579	1.000						
Alk (mgL ⁻¹)	-0.191	-0.285	0.352	0.325	0.321	0.350	0.230	0.014	1.000					
Hard (mgL ⁻¹)	0.059	-0.285	0.033	0.116	0.128	0.062	-0.101	0.155	0.161	1.000				
Chl-a (ugL ⁻¹)	-0.424	-0.208	0.360	0.207	0.133	0.164	0.690	0.519	0.080	-0.074				
D.O (mgL ⁻¹)	0.305	-0.169	0.282	-0.306	0.228	0.206	0.126	0.146	-0.603	-0.500	-0.372	1.000		
ORP (mV)	-0.103	0.025	0.363	0.073	0.234	0.082	0.335	0.294	-0.196	-0.155	0.188	-0.325	1.000	
pH	0.120	-0.023	0.027	-0.220	0.207	0.105	0.104	0.109	0.167	0.150	0.161	0.378	0.050	1.000
	0.007	0.624	0.562	0.000	0.000	0.025	0.037	0.021	0.257	0.307	0.275	0.000	0.288	

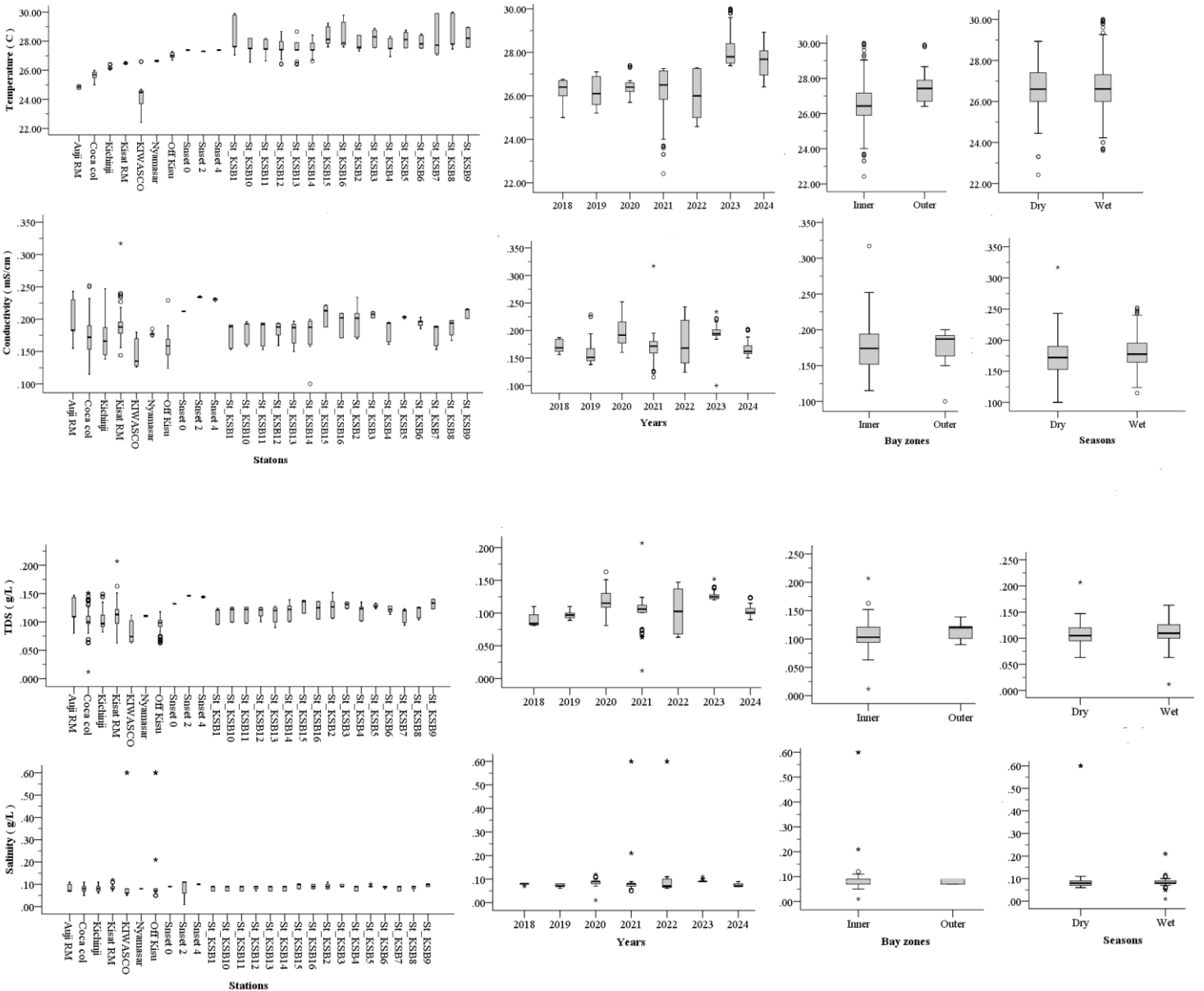
NB: Bolded values of the Spearman's rho correlation coefficients are significant at the 0.01 and 0.05 level of significance (2 tail test)

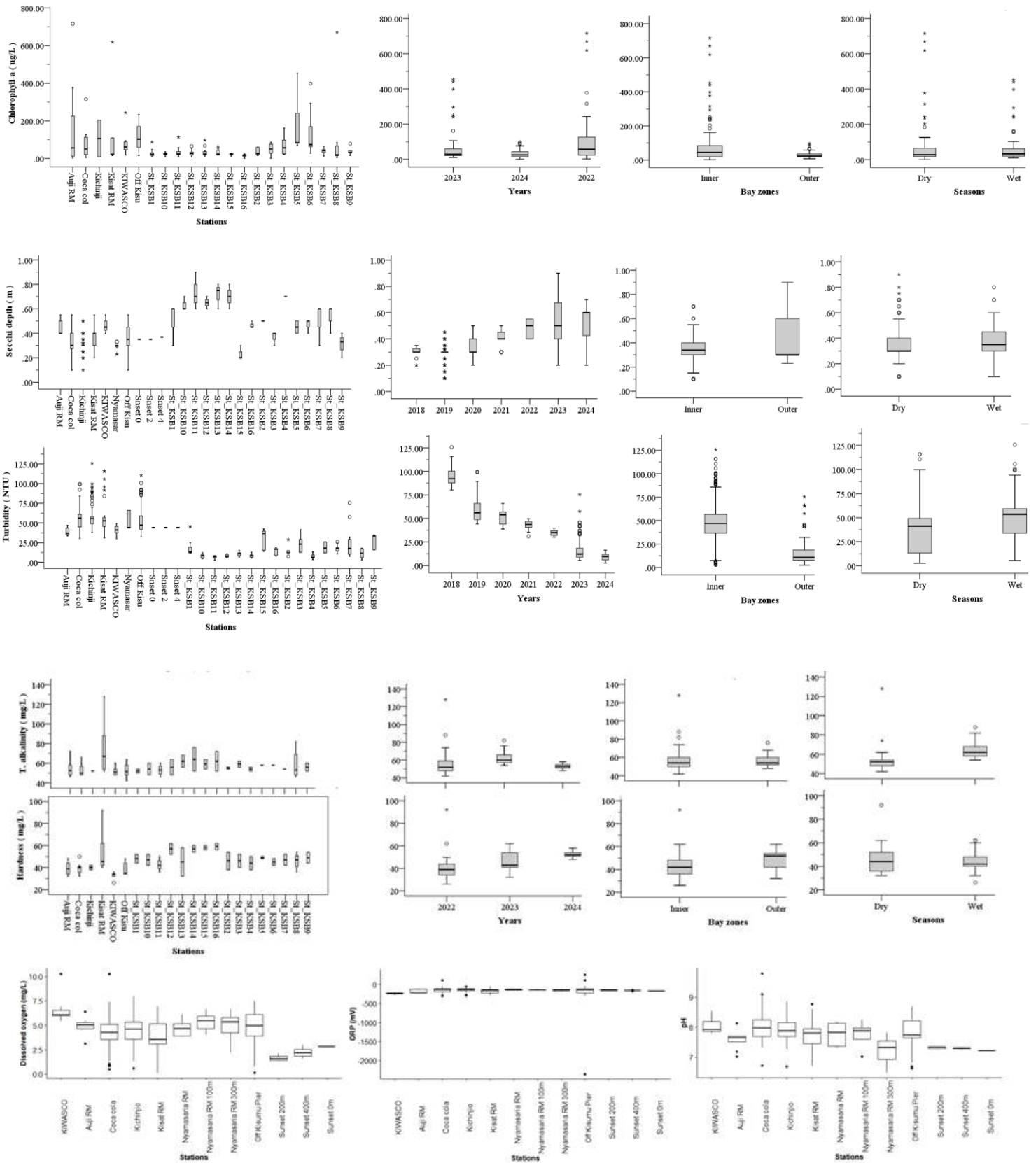
Table 2b: The Spearman's correlation coefficients (significant = bolded) of the relationships between the surface water quality parameters and nutrient elements in Kisumu Bay (Sample size = 73; n = 89 for nutrient elements) The lower value represents the exact p value)

	TP (μM)	SRP (μM)	TN (μM)	NO3-N (μM)	NO2-N (μM)	NH4-N (μM)	SiO2-Si (mM)	N:P ratio	TSI _(Bay)
TP (μM)	1.000								
SRP (μM)	0.562	1.000							
TN (μM)	0.333	-0.124	1.000						
NO3-N (μM)	0.424	0.290	0.251	1.000					
NO2-N (μM)	0.437	0.259	0.122	0.738	1.000				
NH4-N (μM)	0.317	0.062	0.684	-0.091	0.218	1.000			
SiO2-Si (mM)	0.302	0.317	0.002	-0.125	0.109	0.168	1.000		
N:P ratio	0.130	0.135	0.701	0.203	0.563	-0.001	-0.452	1.000	
TSI _(Bay)	0.481	0.006	0.398	0.136	0.235	-0.117	-0.183	0.054	1.000
TSI _(TP)	0.448	0.058	0.379	-0.266	-0.060	0.484	-0.174	0.049	0.591
TSI _(Chla)	0.314	-0.069	0.314	0.241	0.349	0.137	-0.333	0.172	0.792
TSI _(Secchi)	0.135	0.005	0.176	0.002	0.020	-0.070	-0.163	0.074	0.723
	0.292	0.969	0.165	0.989	0.873	0.581	0.199	0.560	0.000

Chlorophyll - (ugL-1)	0.600	0.344	0.184	0.350	0.340	0.255	0.013	0.173	0.542
	0.000	0.001	0.244	0.001	0.001	0.016	0.902	0.173	0.000
Alkalinity (mgL ⁻¹)	0.020	0.270	-0.082	0.026	0.009	0.316	-0.086	0.015	0.061
	0.864	0.005	0.580	0.827	0.936	0.006	0.467	0.921	0.683
Hardness (mgL ⁻¹)	-0.084	0.030	-0.524	-0.442	-0.201	0.181	0.102	-0.683	-0.209
	0.484	0.840	0.000	0.000	0.087	0.269	0.390	0.000	0.154
Total depth (m)	-0.444	-0.376	-0.293	-0.405	-0.332	0.412	0.138	-0.184	-0.399
	0.000	0.004	0.024	0.001	0.010	0.001	0.297	0.163	0.002
Secchi depth (m)	-0.578	-0.455	-0.388	-0.340	-0.189	-0.205	0.083	-0.115	-0.408
	0.000	0.000	0.002	0.006	0.135	0.104	0.515	0.367	0.001
Temperature (°C)	0.266	0.108	0.176	0.093	0.084	-0.058	-0.432	0.125	0.239
	0.035	0.398	0.163	0.465	0.507	0.647	0.000	0.325	0.052
Conductivity (mScm ⁻¹)	0.336	0.101	0.133	0.060	0.003	0.533	0.050	-0.110	0.269
	0.007	0.429	0.293	0.639	0.980	0.000	0.693	0.386	0.032
TDS (g/L)	0.204	0.018	0.130	0.020	-0.075	0.513	-0.095	-0.087	0.222
	0.109	0.887	0.308	0.874	0.555	0.000	0.457	0.493	0.077
Salinity (g/L)	0.277	0.063	0.152	0.058	-0.004	0.519	0.043	-0.049	0.171
	0.028	0.626	0.232	0.650	0.975	0.000	0.733	0.699	0.176
Turbidity (NTU)	0.693	0.391	0.667	0.649	0.175	0.545	0.162	0.436	0.406
	0.000	0.006	0.000	0.000	0.235	0.000	0.270	0.000	0.001
D.O (mgL-1)	0.626	-0.583	NA	-0.277	-0.512	-0.259	-0.334	NA	-0.381
	0.013	0.022	NA	0.318	0.051	0.351	0.223	NA	0.162
ORP (mV)	0.238	0.288	NA	0.406	0.35	0.182	0.088	NA	-0.068
	0.374	0.279	NA	0.119	0.184	0.499	0.745	NA	0.803
pH	0.399	-0.32	NA	0.146	-0.153	-0.387	-0.212	NA	-0.054
	0.188	0.227	NA	0.590	0.573	0.138	0.43	NA	0.841

NB: Bolded values of the Spearman's rho correlation coefficients are significant at the 0.01 and 0.05 level of significance (2 tail test)





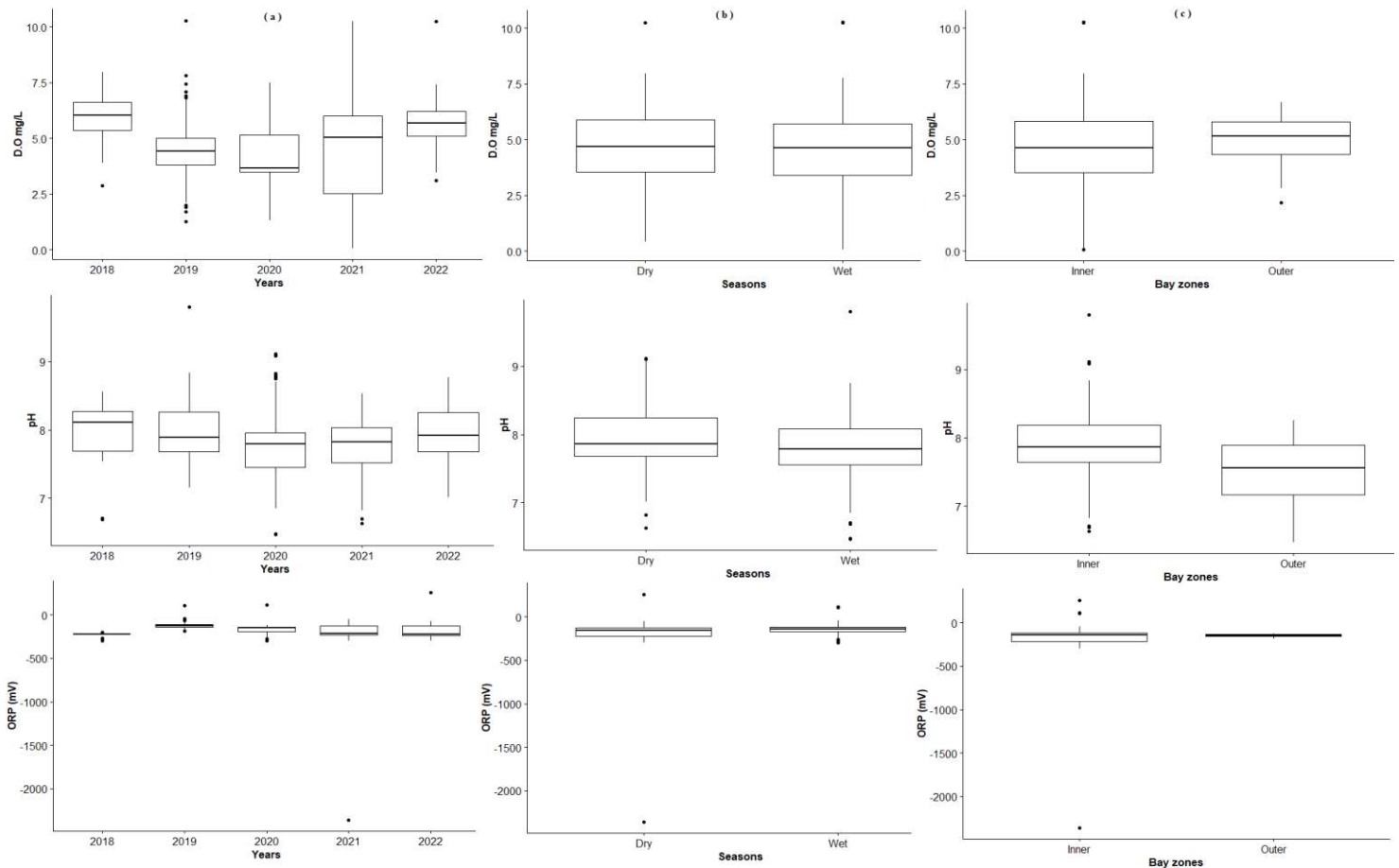


Figure 3: Boxplots showing the water column variations of the D.O, pH, ORP, temperature, conductivity, TDS, salinity, chlorophyll-a, turbidity and water transparency (surface), between the stations, years (a), seasons (b) and bay zones (c) within Kisumu bay between 2018 to 2024. (NB: The station names for the measurements of the D.O, pH and ORP were retained as these parameters were not determined in 2023 and 2024 sampling. Similarly, names for the extra stations sampled between 2018 to 2022 were also retained in the graphical representation.)

Fish species

A total of 18 fish species among the Teleostei class were recorded from seven stations around the inner and outer areas of Kisumu bay during the wet and dry months of 2021 - 2023, and consisted of 12 genera, 8 families and 6 orders. The species were dominated by *Haplochromines*, *Oreochromis niloticus*, *Lates niloticus* and *Synodontis victoriae* (Table 3-4). There were significant differences in the mean sizes of the dominant individual fish species caught. Kisat, Pier and Kichinjio recorded a higher species diversity although only 2 sites (Kisat and Pier) were sampled in 2021. The stations maximum Shannon Weiner fish diversity was 1.380, 1.133 and 1.305 at Kichinjio, Kisat and Pier stations (Fig. 4). The samples of *Lates niloticus* (2021 survey) were dominated by immature fish (> 93%). Sexed *Lates niloticus* ratio was 1:0.5. The regression coefficient or slope from the log transformed fish body weight and total length was used to understand the growth performance of fish species and general wellbeing of the population. The coefficient of determination (*L. niloticus*) was above 0.97 for both seasons, but the slope (b) of the relationship between TL and weight was found to be significantly lower ($p < 0.05$) than the isometric value for both seasons, indicating allometric fish regrowth (Fig. 5a-b).

The relative condition factor [51] ranged from 0.93 – 1.18 (dry) and 0.28-1.12 (wet) with a mean (\pm SD) of 1.0 ± 0.07 and $0.99(\pm 0.10)$ respectively.

The size frequency distribution of the *Haplochromines* shows a modal class length of 4.6 – 5.8 cm (TL) with the spatial mean range of 6.4 ± 1.26 to 8.7 ± 1.42 cm. *Lates niloticus* modal class length was 5.0 – 8.6 cm (TL) with the spatial mean range of 8.56 ± 4.41 to 14.75 ± 3.4 cm (TL) and an individual body weight ranging from 12.14 ± 21.71 g to 38.95 ± 23.18 g. The modal class total length of *Oreochromis niloticus* and *Synodontis victoriae* were 8 – 12 cm and 14.5 to 15.9 cm respectively. The spatial total length mean ranged from 7.01 ± 2.37 to 16 cm (*Oreochromis niloticus*) and 13.72 ± 1.03 cm to 15.2 ± 2.08 cm (*Synodontis victoriae*). The male: female sex ratio ranged from 1:0.73 (*Haplochromines*); 1:0.36 (*Oreochromis niloticus*); 1:0.5 (*Lates niloticus*); 1:0.64 (*Synodontis victoriae*); 1:0.56 (*Clarias gariepinus*); 1:2.33 (*E. profundus* and 1:0.6 (*Brycinus sadleri*); 1:1.22 (*Brycinus jacksonii*). A much higher number of individual *Haplochromines* *Lates niloticus* and *Synodontis victoriae* were caught in the 0.5"-1.75" 0.5"-1.5" and 0.75"-1.5" mesh size nets during the 2021 to 2023 fish gill netting surveys. (Table 5).

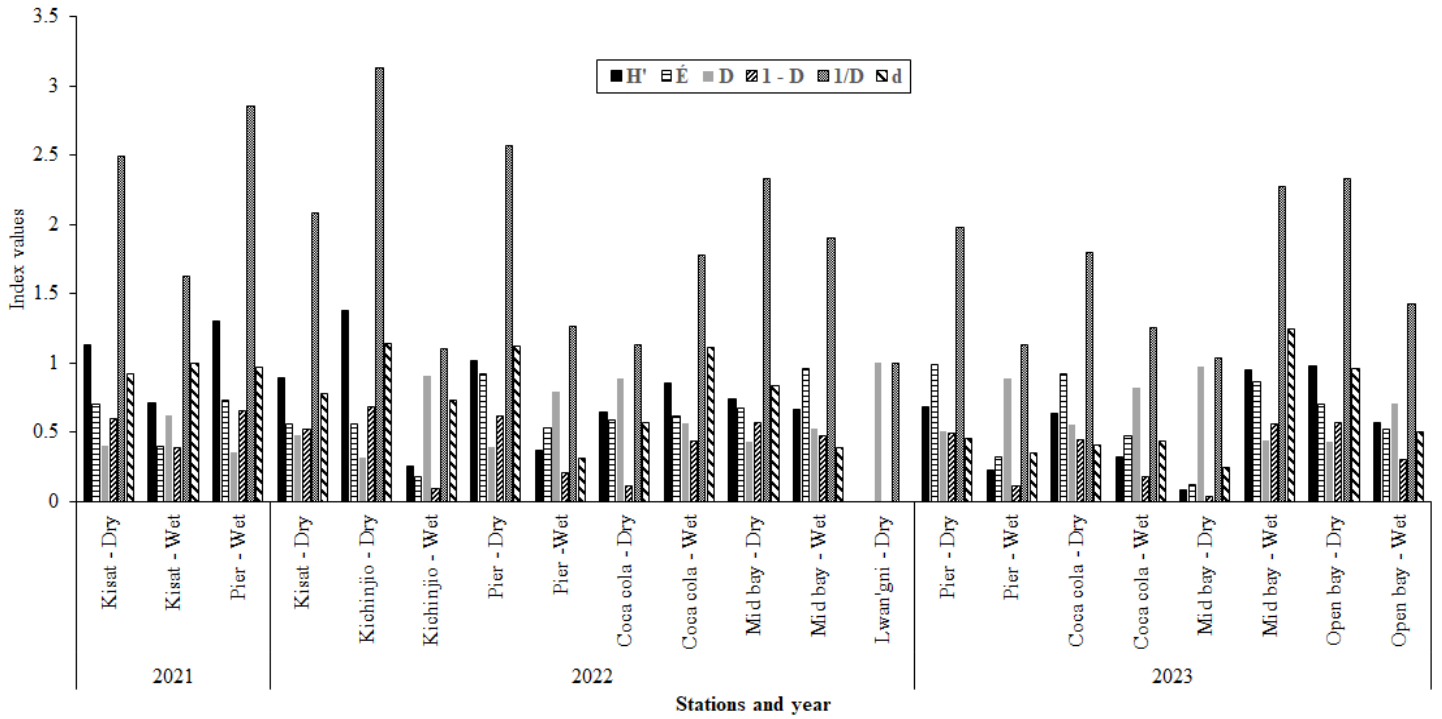
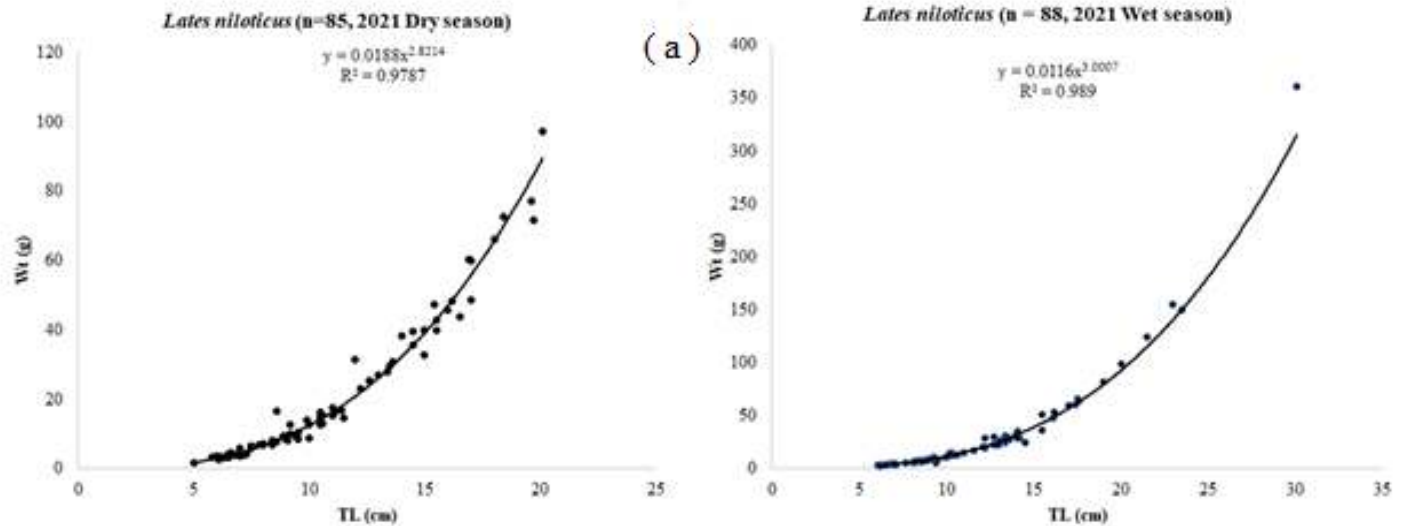


Figure 4: The fish diversity indices (Shannon-Weiner diversity H', Pielou's evenness E, Simpson dominance D, Simpson index of diversity 1-D, Simpson reciprocal 1/D and Margalef richness d) according to seasons (Wet and dry) for each station surveyed between 2021 - 2023

Table 3: Fish species recorded and overall percentage by numbers and weight during the 2021 - 2023 monthly surveys in Kisumu bay

Class	Order	Family	Species	Numbers (%)	Weight (%)
Telestoi	Cyprinidiformes	Cyprinidae	<i>Enteromius profundus</i>	1.3	0.19
			<i>Labeobarbus victoriana</i>	0.4	0.72
			<i>Labeo victorianus</i>	0.3	0.36
			<i>E.neumayeri</i>	0.1	0.04
			<i>Brycinus jacksonii</i>	2.8	0.59
	Perciformes	Centropomidae	<i>B.sadleri</i>	0.8	0.23
			<i>Lates niloticus</i>	16.9	17.2
	Cichliformes	Cichlidae	<i>Oreochromis niloticus</i>	21.3	17.3
			<i>O. leucostictus</i>	0.5	0.65
			<i>Cptodon zillii</i>	0.2	0.98
			<i>O. esculentus</i>	0.1	0.27
			<i>Haplochromines spp.</i>	43.7	13.11
	Ceratodontiformes	Protopteridae	<i>Protopterus aethiopicus</i>	0.2	6.31
	Siluriformes	Claridae	<i>Clarias gariepinus</i>	1.4	29.3
		Mochokidae	<i>Synodontis victoriana</i>	10.3	12.9
Anabantiformes	Anabantidae	<i>S. afrofishcheri</i>	0.1	0.13	
		<i>Ctenopoma murei</i>	0.1	0.07	



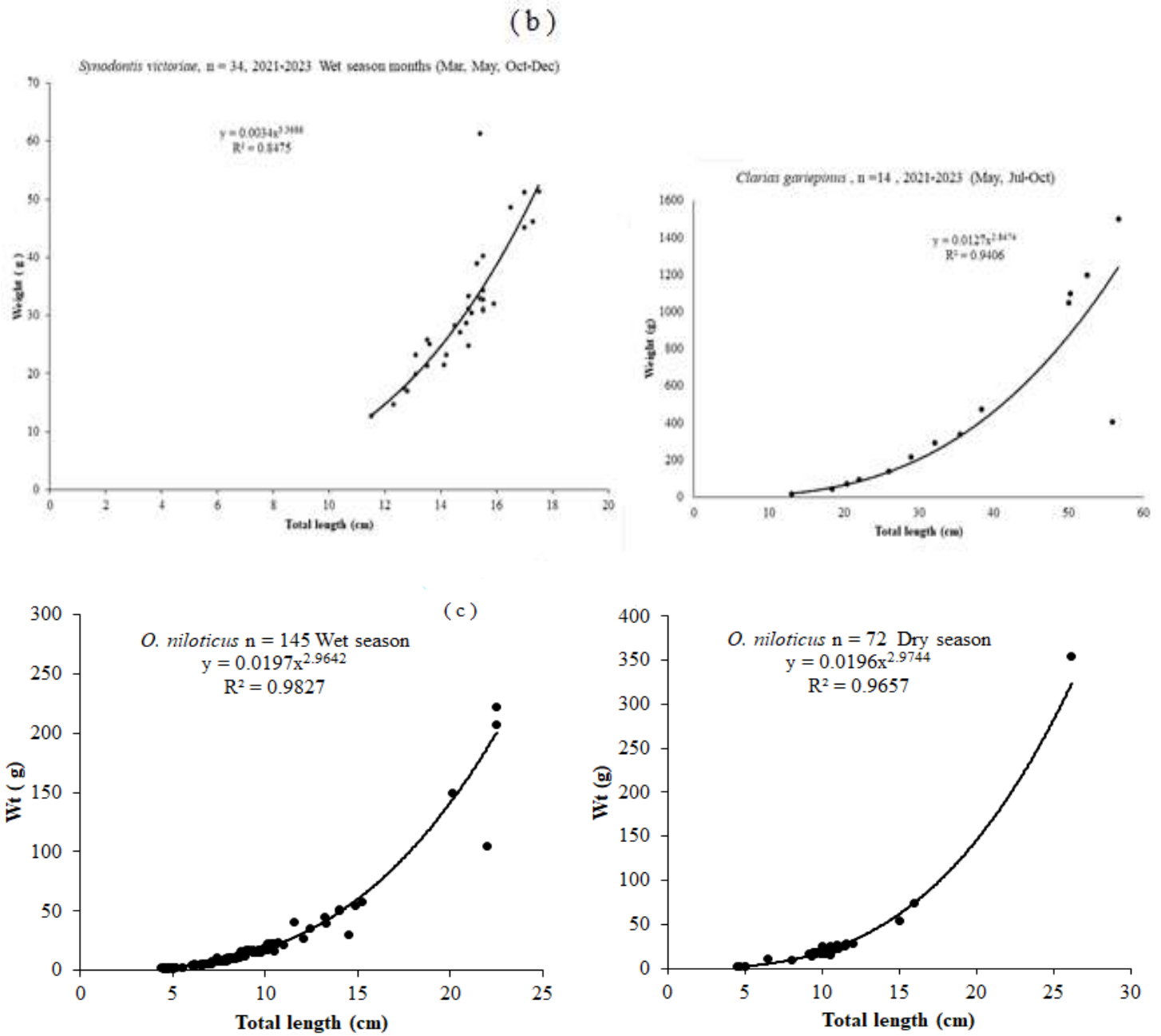


Figure 5(a-b): The relationship between total length (cm) and weight (g) of *Lates niloticus* caught in 2021 (a) *Synodontis victoriae* and *Clarias gariepinus* (b) and *Oreochromis niloticus* (c) caught in 2021 to 2023 during the dry and wet seasons.

Table 4: The mean total length (cm) and weight (g) of the dominant fish species (a different superscript letter indicates significantly different means across the stations)

	<i>O. niloticus</i>		<i>L. niloticus</i>		<i>S. victoriae</i>		<i>Haplochromine spp.</i>	
Stations	Mean TL (cm)	Mean wt. (g)	Mean TL (cm)	Mean wt. (g)	Mean TL (cm)	Mean wt. (g)	Mean TL (cm)	Mean wt. (g)
Kichinjio	16	73.89	11.21 ± 3.15 ^a	21.29 ± 14.08 ^a	15.2 ± 2.08 ^a	33.14 ± 15.85 ^a	7.83 ± 1.66 ^a	7.92 ± 5.30 ^a
Kisat	7.14 ± 2.84 ^a	11.82 ± 37.28 ^a	9.90 ± 3.68 ^{ab}	16.78 ± 20.15 ^{ab}	14.64 ± 1.39 ^a	30.37 ± 10.04 ^a	7.20 ± 1.75 ^a	6.83 ± 4.92 ^b
Mid bay	10.43 ± 1.75 ^a	23.53 ± 25.67 ^b	14.75 ± 3.40 ^{abc}	38.95 ± 23.18 ^{abc}	13.72 ± 1.03 ^b	23.85 ± 6.15 ^b	8.7 ± 1.42 ^b	10.42 ± 4.54 ^c
Open bay	10.27 ± 0.21 ^a	16.92 ± 1.11 ^{cb}	10.05 ± 4.44 ^{ab}	20.29 ± 39.45 ^{acb}	16	44.32	6.4 ± 1.26 ^c	4.11 ± 2.90 ^d
Pier	10.9 ± 4.38 ^a	29.13 ± 29.45 ^{adebc}						
Coca cola	7.01 ± 2.37 ^a	8.21 ± 11.21 ^{ae}	8.56 ± 4.41 ^{cd}	12.14 ± 21.71 ^{cd}			7.73 ± 1.88 ^a	7.73 ± 5.86 ^a
Pipeline / KPC	11.02 ± 3.92 ^a	31.88 ± 40.56 ^{ecb}	12.55 ± 4.91 ^{acdb}	36.28 ± 56.24 ^{ecd}	13.82 ± 1.83 ^a	26.98 ± 10.13 ^a	7.60 ± 1.48 ^{ac}	7.25 ± 5.41 ^{ad}
Lwang'ni							7.6 ± 0.29 ^a	6.12 ± 1.03 ^a

Table 5: Percentage composition of *Haplochromine spp.* (a) *Synodontis victoriae* (b) *Lates niloticus* (c) and *Oreochromis niloticus* (d) according to the mesh sizes for the weight and total length classes

Haplochromine_TL									
TL (cm)	0.5"	0.75"	1.0"	1.25"	1.5"	1.75"	2.0"	2.5"	
< 3.4	-	-	-	-	-	-	-	-	-
3.4 - 4.6	-	1.1	-	-	-	-	-	-	-
4.6 - 5.8	2.7	20	0.2	-	-	2	-	-	-
5.8 - 7.0	2.7	8.8	2	-	0.7	2.9	-	-	-
7.0 - 8.2	-	1.6	18.9	-	2.5	-	0.2	-	-
8.2 - 9.4	-	1.8	10.6	7.6	2.2	-	0.7	0.2	-
> 9.4	-	0.9	0.7	1.3	7	-	0.7	-	-
Haplochromine_Wt									
Wt (g)	0.5"	0.75"	1.0"	1.25"	1.5"	1.75"	2.0"	2.5"	
< 1	-	-	-	-	-	-	-	-	-
1 - 7	5.4	31.2	17.3	-	2.9	4.9	0.2	-	-
7 - 13	-	2	14.4	8.1	3.1	-	0.9	0.2	-
13 - 19	-	0.7	0.7	0.9	3.6	-	-	-	-
19 - 25	-	-	-	-	2.7	-	0.2	-	-
25 - 31	-	0.2	-	-	-	-	-	-	-
Synodontis_TL									
TL (cm)	0.75"	1.0"	1.25"	1.5"	2.0"				
< 9.2	-	-	-	-	-				
9.2 - 10.6	-	0.96	-	-	-				
10.6 - 12.0	-	4.81	-	0.96	-				
12.0 - 13.4	0.96	8.65	-	5.77	-				
13.4 - 14.5	0.96	9.61	-	18.27	1.92				
14.5 - 15.9	2.88	8.65	2.88	19.23	2.88				
> 15.9	1.92	1.92	-	4.81	2.88				
Synodontis_Wt									
Wt (g)	0.75"	1.0"	1.25"	1.5"	2.0"				
< 10.0	-	0.95	-	-	-				
10.0 - 17.5	0.95	7.62	-	2.86	-				
17.5 - 25.0	-	11.43	-	9.52	1.9				
25.0 - 32.5	2.86	8.57	0.95	20.95	-				
32.5 - 40.0	0.95	2.86	1.9	10.48	2.86				
40.0 - 47.5	0.95	0.95	-	3.81	-				
> 47.5	0.95	1.9	-	0.95	2.86				
Lates_TL									
TL (cm)	0.5"	0.75"	1.0"	1.25"	1.5"	1.75"	2.0"	2.5"	3.0"
< 5.0	-	-	-	-	-	-	-	-	-
5.0 - 8.6	4.07	26.16	7.56	-	0.58	-	-	-	-
8.6 - 12.2	-	3.49	17.44	3.49	1.74	1.16	-	1.74	-
12.2 - 15.8	-	1.16	5.81	1.16	10.47	-	0.58	-	-
15.8 - 19.4	-	-	0.58	-	3.48	-	4.65	-	-
19.4 - 23.0	-	-	0.58	-	0.58	-	2.33	-	-
> 23.0	-	-	-	-	-	-	0.58	-	0.58
Lates_Wt									
Wt (g)	0.5"	0.75"	1.0"	1.25"	1.5"	1.75"	2.0"	2.5"	3.0"
< 1.64	0.58	30.81	30.81	5.23	14.53	1.16	1.16	1.74	-
1.64 - 52.84	-	-	0.58	0.58	1.74	-	5.81	-	-
52.84 - 104.0	-	-	-	-	0.58	-	1.16	-	-
104.0 - 155.2	-	-	-	-	-	-	-	-	-
155.2 - 206.4	-	-	-	-	-	-	-	-	-
206.4 - 257.6	-	-	-	-	-	-	-	-	-
> 257.6	-	-	-	-	-	-	-	-	0.5
Oreochromis_TL									
TL (cm)	0.5"	0.75"	1.0"	1.25"	1.5"	2.0"	3.0"	3.5"	
< 4.0	-	-	-	-	-	-	-	-	-
4.0 - 8.0	1.38	19.35	12.9	4.61	0.46	0.46	-	-	-
8.0 - 12.0	-	-	28.11	8.29	16.59	0.46	-	-	-
12.0 - 16.0	-	-	-	-	1.38	3.69	-	-	-
16.0 - 20.0	-	-	-	-	-	-	-	-	-
20.0 - 24.0	-	-	-	-	-	-	1.38	0.46	-
> 24.0	-	-	-	-	0.46	-	-	-	-
Oreochromis_Wt									
Wt (g)	0.5"	0.75"	1.0"	1.25"	1.5"	2.0"	3.0"	3.5"	
< 1.0	-	-	-	-	-	-	-	-	-
1.0 - 50.0	1.38	19.35	41.01	12.9	17.97	2.76	-	-	-
50.0 - 100.0	-	-	-	-	0.46	1.84	-	-	-
100.0 - 150.0	-	-	-	-	-	-	0.92	-	-
150.0 - 200.0	-	-	-	-	-	-	-	-	-
200.0 - 250.0	-	-	-	-	-	-	0.46	0.46	-
> 250	-	-	-	-	-	-	0.46	-	-

4. DISCUSSIONS

4.1 Water quality

Studies on specific ecologies of the bay areas are few and are mostly part of the larger gulf ecological studies. However, there is significant evidence that most of the sheltered bays are "areas of concern" due to their close proximity to increasing urbanization and maritime activities.

The limnological conditions influence to a greater extent the normal functioning and productivity of lake ecosystems. Within the shallow lake bays of Lake Victoria, water conditions reflect the high littoral exchanges and the influx of different materials from surrounding areas (including both the diffuse and point sources of polluting substances). The addition of untreated effluents may alter the ambient water quality and ecological communities in receiving waters. Increased nutrients and elevated temperatures increase primary producer growth rates and biomass [52], and in turn result in an increased BOD and turbidity. These changes can cause increased fish mortality and blooms of toxic cyanobacteria [53]. High total ammonia - N concentrations (greater than 2mg/L chronic exposure and 17 mg/L acute exposure at pH = 7 and at 20°C [54] are particularly influential in reducing growth rates and reproductive success or causing direct mortality in aquatic invertebrates and fish [55]. In rivers and streams sensitive invertebrate taxa are less favored in such conditions than the more tolerant worms and true flies (Chironomus), and this may be similar to the littoral shoreline areas. The lake bays have been shown to suffer from frequent periods of excessive growth of phytoplankton, especially the cyanobacteria (blue green algae) [4,14, 56-57]. This study focused on the inner and outer bay areas as water surfaces for potential utilization for the blue economy. Within the bay, water quality was found to differ significantly between the inner shallow zones and the outer exposed areas, which define a general increase in water depth outwards the bay area. In particular, a higher algal productivity, higher temperatures and increased water turbidity characterized the inner areas of the bay. This is consistent with previous gradients along the whole gulf [58] when moving away from the littoral zones. In the gulf, from the Ndere Islands the euphotic depth ranges from 2.1 m to 5.8m, with a much higher value of 8.5m in the main lake.

The most recent data on the chlorophyll-a concentrations depicts increasing levels with a range of in the open waters of the Winam Gulf. Urban storm drainage and watershed contribute to the increased loading of suspended materials around the bay; however this may become apparent when seasonal effects are also considered. Such effects of urban runoffs can override the true seasonal effects around the shores and discharge areas of urban fed streams Kisian, Kisat and Kibos are all urban related streams and transport and discharge suspended loads nearby the bay waters. Water quality changes noted in urban fed streams includes high TSS and nutrients [59-60] and the downstream reaches are characterized by depleted DO levels, but may vary depending on the flow volumes which can be highly variable.

The lake water was found to exhibit eutrophic to hypereutrophic conditions (**Table 7**) based on the algal biomass and light transparency values. Environmental conditions associated with a high potential for cyanobacterial biomass are TP (>50 - 100 µg/L⁻¹), high water residence time, pH (>6), secchi depth (<1m) and temperature of ≥ 25 °C. The shallow bay is restricted to the hydrological exchanges and hence the water quality conditions exhibited of high turbidity and algal biomass. Whole gulf trends (from 2016 - 2022) of phytoplankton biomass maps from Sentinel -2 imagery [2] depicts increasing monthly average levels of chlorophyll- a towards the shore and bays (18 - 362 µg/L⁻¹) from the Rusinga channel areas. Similar high concentrations of chlorophyll-a and total phytoplankton biovolume were observed between 2011 and January 2012 at a site in the bay.

Table 7: Carlson's trophic state index (TSI) and categories of lake productivity

TSI Values	Trophic State	TP ($\mu\text{g L}^{-1}$)	Chl-a ($\mu\text{g L}^{-1}$)	Secchi (m)	
0–40	Oligotrophic (O)	0–12	0–2.6	>8–4	
40–50	Mesotrophic (ME)	12–24	2.6–7.3	4–2	
50–70	Eutrophic (E)	24–96	7.3–56	2–0.5	
70–100+	Hypereutrophic (HE)	96–384+	56–155+	0.5–0.25	
This Study (Mean Values \pm 95% Confidence Interval)					
Parameter	TSI Value	Trophic State	TP ($\mu\text{g L}^{-1}$)	Chl-a ($\mu\text{g L}^{-1}$)	Secchi (m)
Mean	69.49	E–HE	157.04	52.23	0.53
Lower	66.16	E–HE	96.61	39.5	0.46
Upper	72.82	HE	217.47	64.97	0.59
Sample Size (n)	16	—	130	32	—

[4]. Concentrations ranged from 19-50 μgL^{-1} (surface water), 10-30 μgL^{-1} (integrated water samples), 274 to 1038 μgL^{-1} (patch water samples) and 18 to 1737 μgL^{-1} (shore water samples) with a corresponding high biovolume of phytoplankton cells in patch and shore areas [4]. A lake-wide beach sites study (September–November 2015) by Orina [61] found significant variations in chlorophyll-a levels among the sites and the distance from the lakeshore (0m to 100m distance), with a high concentration at Kichinjio site (of $66.9\pm 0.97 \mu\text{gL}^{-1}$). This study recorded lower chlorophyll-a levels, possibly due to the temporal variations reported earlier, but still there were signs of algal bloom development during the sampling time. Periodic influxes of suspended solids from adjoining terrestrial surfaces and urban rivers discharge contribute particulate material loading to the surface water, effectively reducing the light availability, but increasing available nutrients for primary production. This selectively favours specialized algal species such as Microcystis which are capable of remaining near the water surface through use of gas vesicles [62]. Buoyant cyanobacteria such as Anabaena and Microcystis may float upward when mixing is weak and accumulate in dense surface blooms, while others such as Cyndrospermopsis and Planktothrix stay dispersed, but can reach very high cell densities, resulting in high turbidity. Toxic cyanobacterial species can result in production of high levels of intracellular and extracellular cyanotoxins such as microcystins [3,9,63, 64–66] in water which is a potential health risk.

The localized nature of such blooms around the bays may mean a high exposure risk to microcystins to resident fish and caged tilapia.

Environmental gradients were assessed using PCA of the measured physical and chemical parameters (Fig. 8 a-b). The surface water quality variations were explained by more than two dimensions (over 70% of the variations). Within the water column, PC1 explained about 50.2% of the variations and was positively and well correlated with all the parameters except the sampling depth, whereas PC2 explained about 21.3 % of the variation and was positively correlated with the sampling depth. Similarly, when only the surface water quality variations were analysed, both PC1 (37.8 – 34.5%) and PC2 (13.4 - 15.8 %) explained about 51.2% of the variation. All the variables were well correlated with PC1. The total depth and light transparency were all negatively correlated with PC1, whereas water temperature, turbidity, conductivity and TDS were more positively correlated with PC1 (except for the sampling depth) compared to salinity and chlorophyll-a. Principal component 2 separated the outer zone stations from the inner zone; and appeared to represent the effects of bathymetry on the microhabitat. PC1 represents the effects of the increasing littoral or shoreline exchanges within the inner zone; and contribution of the frequent re-suspension and redistribution of the unsettled externally derived particulate and dissolved materials influx due to the different anthropogenic impacts.

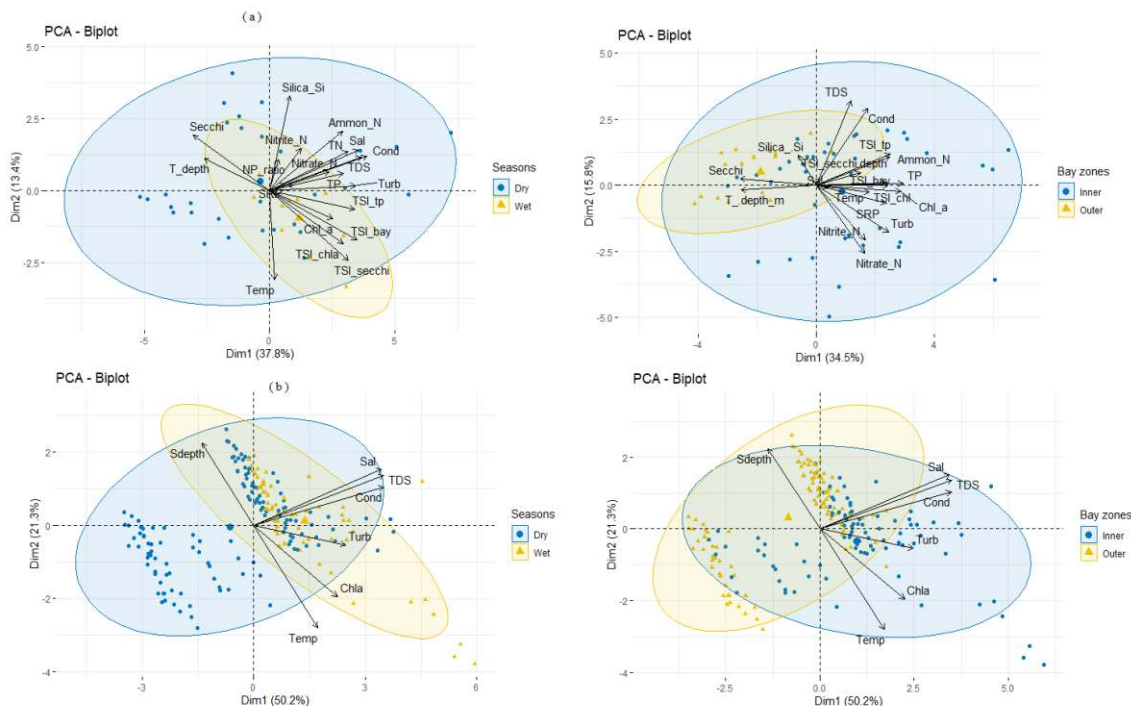


Figure 6a – b: PCA ordination plot of the sampling stations and measured water quality variables in the surface water (a) and the water column (b) during 2023 and 2024 surveys grouped by seasons and bay zones

Shallow lakes have a pronounced impact from wind driven mixing and bottom sediments resuspension. Surface mixing dynamics affect the transport of both dissolved substances, such as nutrients, and of particulates, such as phytoplankton, near the lake surface. There is limited littoral vegetation which is a critical component along the bay as it buffers the lake from particulate materials influx. Rooted macrophytes (vascular plants), once established, can reduce turbulent mixing, increase sedimentation rates and reduce sediment resuspension in shallow water bodies [67].

Sediments resuspended during dredging operations pose a variety of water quality and ecological concerns. The sediment plume in the immediate vicinity of the dredging could influence the behavior of fish and impact the health of less mobile aquatic vertebrates and invertebrates. Resettling of suspended particulates could also impact bottom-dwelling organisms. Resuspension can also result in higher concentrations of particulate-associated contaminants in the water column, drive increases of dissolved contaminant concentrations (a contaminant "release" pathway), and disperse contaminants wider in the aquatic environment. Upon settling, resuspended sediments become generated residuals [68]. Recent dredging operations in the bay may have influenced water column nutrients. This may also contribute to increased contaminant exposure and risk to biota [68] due to post dredging residuals.

Spatio-temporal trends in water quality within Kisumu bay

Gulf-wide gradients in the total ionic concentrations depict a decreasing electrical conductivity towards the mainlake. Within the bay area, the variations in conductivity emerged between the inner and outer zones and between seasons. The bay is a narrow zone consisting of submerged, floating and emergent shore vegetation compared to the open waters moving outwards into the Winam gulf. This reflects increased effects from onland washout and waste water discharge into the bay zone as the surrounding area is mainly an urban and industrial zone with agricultural activities located in the peri-urban areas. The open bay sediments mainly consist of soft silty mud but more sandy bottoms appear on the river inlet areas in the southern and northern zones. Previous studies show lack of continuous published data within the bay and other lake embayments of Lake Victoria (Kenya) from 2003 to 2008 and 2011 to 2016; but the Kisumu bay area has been sampled as a single site in many of the past lake surveys (Table 8). More recently, investigations conducted by Guya [69] in November 2017 using littoral and river mouth sites within the southern Winam gulf revealed the persistent eutrophication of the relatively shallow gulf.

The eastern section of the Winam gulf recorded much higher levels of SRP (mean range = 1.9 - 2.7 μM), TP (mean range = 4.1 - 6.0 μM), TN (mean range = 29.5 - 39.6 μM) and SiO_2 (0.2 - 0.6 mM) than the western section [69] as most of the sites were more represented by river mouth areas. Five littoral North-South transects across the Winam gulf [70], but located outside the Kisumu bay, recorded surface water SRP and TP concentrations ranging from 0.74 - 4.61 μM and 1.84 - 8.95 μM respectively. Mean and median levels of TP (5.52 \pm 4.80 and 3.88 μM) and SRP (1.71 \pm 2.66 and 0.94 μM) found in this study (2022 - 2024 data) were much lower than the range values reported for the transect sites located near the urbanised areas of Kisumu and Homabay [70] which receive treated wastewaters. The difference between TP and SRP in water above the sediment - water interface showed a decreasing pattern from the inner to the outer transects across the Winam gulf [71] with a concentration range of 3.50 - 6.91 μM (SRP) and 6.45 - 17.21 μM (TP) during July 2007 sampling. River Nyando is among the inflowing rivers into the Eastern section of Winam gulf found to exhibit apparently high SRP (range = 1.11 - 3.90 μM) and TP (2.21 - 27.95 μM) in surface water [72] during both the low and high flow regimes between July 2015 and March 2016.

The total concentration of phosphorus found in lake waters (overall mean of 5.52 \pm 4.80 μM and an annual mean range of 7.03 \pm 5.58 μM to 3.05 \pm 2.10 μM) within the extreme eastern edge of the Winam gulf (Kenya) is way above the levels reported for mesotrophic and oligotrophic basins of a large lake [73] which ranged from 5.93 - 59.33 $\mu\text{g L}^{-1}$ (43 years monitoring data 1970-2013) and 2.73 \pm 2.05 to 7.96 \pm 6.55 $\mu\text{g L}^{-1}$ (18 years monitoring data 1996-2014) of total phosphorus and an accompanying low range of the phytoplankton biomass (0.67 - 16.7 $\mu\text{g L}^{-1}$) and a very high secchi depth of 1.17m to 10.84 m. These lakes are characterized as P - limited as compared to the N limitation of phytoplankton growth within most of the Lake Victoria (Kenya) bays. The percentage of the dissolved P in surface water ranged from 2.11 - 66.45% (outer bay) and 1.34 - 89.65 % (Inner bay); implying readily available soluble P within the inner bay and increasing availability of organic and particulate phosphorus towards the outer bay. The appreciably high levels of N sources and available P with optimal temperatures and light conditions are important drivers of the common harmful cyanobacterial blooms frequently encountered in many of the bays within Winam gulf which exposes the water users to associated risks of exposure to cyanotoxins.

Table 8: A comparison of the long-term water quality data showing the mean (\pm SD) and range values of some of the physico-chemical variables determined within Winam gulf and Kisumu bay between 2003 and 2024

Parameters	Winam gulf (2003)		Kisumu bay (2009 to 2010)	Kisumu bay (2023 to 2024)	
	Inner gulf (S) (10 sites)	Outer gulf (S) (9 sites)	Surface water (S) (D - W) (5 sites) ($\bar{x} \pm \text{SE}$)	Water column (16 Sites) n	
Total depth (m)	4.9 \pm 2.2	7.0 \pm 3.3 (2.5-13)	-	3.7 \pm 1.2 (1.0-5.7) 48	
Secchi (m)	0.5 \pm 0.1	0.8 \pm 0.4 (0.35-1.3)	0.22 \pm 0.01	0.53 \pm 0.2 (0.2-0.9) 48	
Temp ($^{\circ}\text{C}$)	25.5 \pm 1.1	26.0 \pm 1.2 (24.2-27.8)	28.1 \pm 0.2-26.7 \pm 0.1	27.08 \pm 1.08 (25.47-29.97) 195	
Alk. (mgCaCO ₃ L ⁻¹)	96 \pm 38	65 \pm 9 (4-130)	-	63 \pm 9 (56-82)* 32	
Hard. (mgL ⁻¹)	31 \pm 2	27 \pm 4 (20-34)	-	47 \pm 9 (38-62)* 32	
Turbidity (NTU)	-	-	227.4 \pm 8.8-268.6 \pm 8.9	16.6 \pm 11.4 (5.4-75.5) 193	
TDS (g/L-1)	-	-	0.288 \pm 0.002-0.287 \pm 0.002	0.126 \pm 0.006 (0.120-0.152) 195	
Salinity (g/L-1)	-	-	-	0.09 \pm 0.0 (0.09-0.11) 195	
Cond (mScm-1)	0.182 \pm 0.021	0.176 \pm 0.02 (0.146-0.204)	0.165 \pm 0.0036-0.167 \pm 0.0037	0.187 \pm 0.019 (0.100-0.234) 195	
pH	8.0 \pm 0.2	8.0 \pm 0.3 (7.5-8.3)	7.7 \pm 0.1-7.8 \pm 0.0	7.88 \pm 0.43 (6.47-9.8)* 450	
D.O (mgL-1)	8.4 \pm 1.4	8.4 \pm 1.7 (6.5-11.2)	5.9 \pm 0.2-6.3 \pm 0.2	4.60 \pm 1.64 (0.07-10.28)* 457	

ORP (mV)	193±35	194±39 (159-255)	-	-164.34±120.63 (-297-259)*	458
Molar N/P ratio	-	-	5.4±0.1-5.3±0.1**	5.28±4.51 (0.46-24.63)*	48
Chl-a (ugL-1)	27.7±12.3	20.0±3.8 (9.7-34.7)	28.8±2.4-18.4±1.1&&	45.88±61.60 (1.3-453.4)	193
SiO ₂ -Si (mgL-1)	14.2±6.1	8.8±4.8 (0.15-19.4)	23.9±1.3-28.1±1.2	9.79±4.55 (1.86-16.98)*	48
SRP (ugL-1)	206±84	155±64 (62-302)	140.2±4.8-139.4±5.8	28.76±38.50 (0.86-179.43)*	48
TP (ugL-1)	-	-	280.8±4.1-289.6±3.8	129.95±110.36 (26.29-431.14)*	48
TN (ugL-1)	-	-	1501.2±8-1518.5±7	220.90±200.79 (31.26-798.63)*	42
NO ₂ -N (ugL-1)	-	-	20.8±0.9-25.5±1.5	1.91±1.84 (0.0-7.22) *	48
NO ₃ -N (ugL-1)	111±51	105.6±74.8 (49.8-247.5)	41.7±1.6-44.5±1.6	5.02±2.24 (0.73-9.65)*	48
NH ₄ -N (ugL-1)	17.4±11.4	20.0±3.8 (2.8-36.6)	107.6±5.4-109.7±6.3	140.88±190.84 (19.31-814.69)*	48

NB: Chla-a= Chlorophyll-a; Al=alkalinity; Hard=Hardness; Cond = electrical conductivity; Temp = temperature; *values=Surface water (S); ^aD.O ^bORP and ^cpH values=2018-2022 data; (-) = Not determined; Dry(D) + wet (W) seasons months of August -September and March to May respectively; **=N/P mass ratio; &&=mgL⁻¹; n = 2023/24 sample size except, for ^aD.O, ^bORP and ^cpH where the n values represent the 2018-2022 data.

Water quality of urban streams and rivers draining the bay were investigated by Omondi et al. [59] and Kobingi et al. [60] who reported the increasing impacts in the relatively small river channels and waterways from the urban human activities. Treist et al. [74] also used epilithic diatoms assemblage and observed a decline downstream of the algal species in rivers with conductivity, alkalinity, turbidity, dissolved oxygen and silicate as the most important water quality variables which influenced the species distribution in studied rivers. The high nutrient loading within the small waterways into the bay are of significance in management of the bay eutrophication. Therefore, there is a need to consider rehabilitation plans for the impacted urban rivers and streams into the bay.

4.2 Fish composition and diversity

Three fish species (*Haplochromines spp.*, *Lates niloticus* and *Synodontis victoriae*) dominated (percentage by numbers in 2021) the experimental catch survey at KPC/Pipeline pier and Kisat, between March and December 2021 with the rest of the species present in very low numbers. Spatially, more sites were sampled during 2022 and 2023 compared to 2021 monthly surveys. During the survey, the lowest number of species were recorded during the month of July (2021). In 2022, a high species richness was observed at Kichinjio and Kisat, compared to the rest of the sites, which recorded between 1 and 3 fish species. Very low number of fish species (*Lates niloticus*, *Haplochromine spp.*, *Oreochromis leucostictus* and *Enteromius profundus*) and abundances were recorded in 2023 from 3 sites. The sex ratio (M:F) of *Haplochromine spp.*, *Synodontis victoriae* and *Oreochromis niloticus* was 1:0.73, 1: 0.64 and 1:0.36 respectively. Stage 3 and 4 (*O. niloticus*) and stage 3 -5 (*S. victoriae*). *Haplochromine spp.* were dominated by stages 3 – 6 (Males and females). Fish survey sites were more concentrated in the inner zone compared to the outer zone and this may not allow making detailed comparisons in diversity results and spatial variations. Shallow littoral waters are ecologically important aquatic habitats and support a number of fish species and other aquatic organisms. These fishing areas are often exploited by artisanal fishermen who often are capable of using illegal gears. Around the lake, the management of small -scale inland fisheries has outlawed the use of small size fishing gears in Lake Victoria [75]; which eventually contribute to undersize catches of commercially regulated and exploited fish species. However, from the experimental gears employed in the survey; low strict enforcement of this regulation can result in exploitation of juveniles. The few rocky lakeshores on the southern and sandy bottoms in the north are suitable spawning areas for Nile tilapia within the gulf, but are under threat from increasing human disturbances. The sampled Nile tilapia fish caught were all below the length at 50% maturity for Nile tilapia population.

Previously, the recorded size at 50% maturity for Nile tilapia in Lake Victoria (Kenya) was 31cm TL (females) and 35 cm TL (males) [76] which was higher than 27.5 cm TL (females) and 31.5 cm TL (males) recorded by Ogari and Asila [77]. Most of the Nile perch were below 30 cm TL, a sign of good recruitment which usually follows the rainy months. The age groups entering the fishery [78] in Tanzanian waters coincides with the months of November and December; and July to September in Uganda. The length at which 50% of the caught individuals mature in Nyanza gulf decreased from 74 cm TL (males) and 102 cm (females) (in 1978 – 1984) to 55 cm TL (males) and 70 cm TL (females) in 1990s. [77,79]. The shallow nearshore areas are important spawning and nursery areas for small Nile perch.

Nile perch introduction into Lake Victoria has played a key role in shaping the trophic relationships which saw the decline (1980s) and suspect recovery of the haplochromine fish species. *Lates niloticus* prefers three key species (*Haplochromines*, *Rastrineobola argentea* *Caridina nilotica*) but changes its diet depending on the availability of the food items and growth stage. In Mwanza gulf [26] shrimps were the main diet for Nile perch of between 5 and 30 cm TL. At a size of 3 to 4cm Nile perch shifted from size-selective predation on the largest cyclopoids to predation on the largest, less abundant, calanoids and zooplanktivory ended at a size of ca. 5cm.

The study area is an important habitat providing refugia zones for prey species and juvenile fish species. Larval studies show that major nursery grounds for Nile perch, Nile tilapia and daga are the shallow inshore areas [76, 80]. The zones are highly productive compared to the outer areas, hence delivering a rich food source from the remaining patchy macrophytes along the vegetated littoral areas, river mouths and a wetland zone; with frequent influx of scattered water hyacinth and other floating macrophytes in the bay. In the study area, benthic macroinvertebrates were found dominated by the tolerant groups. The Chaoborid and Chironomid larvae are major food source of insectivorous and zoo-planktivorous haplochromines and Odonata nymphs are the food of *Lates niloticus* [81-83]; and therefore can influence the fish distribution besides the seasonal rainfall influences on the fish species reproduction cycles. Nile tilapia spawns throughout the year and in Kenya, the breeding peaks are in March to May and October to December [76, 84]. Besides this, juvenile *Clarias sp.*, *Haplochromine spp.*, *Synodontis* and *Shilbe* are among the indigenous bait fish species used in the hook and line fishery in Lake Victoria. Therefore, continued use of small mesh size gear (illegal gear) in fish breeding and nursery areas may result in risks to biodiversity conservation. More awareness and actions to protect the diverse fish microhabitats is key to a more sustainable blue economy.

The mean TL for *O. niloticus*, caught by all experimental gears was less than 16 cm. Nile tilapia are known to be herbivorous, and feed mostly on algae but were also found to consume diverse food items which included insects, algae and plant materials. The major diet of fish <5 cm TL was zooplankton whereas bigger fish included a wider range of food items in their diet. Mean total length (\pm SD) at first maturity from trawl and beach seine data in Nyanza gulf (1998 to 2000) was 30.81 ± 0.09 cm for females and 34.5 ± 0.48 cm for males [84]. Several recent investigations on the biology of major species have reported changes in Nile perch, Nile tilapia and *Rastrineobola argentea* (2014-2015) fish sizes [85-88] in the Kenyan part of L. Victoria compare to previous study data and recommend enforcement of the fishery regulations (Nile perch slot size of 50-85 cm TL; and illegal gears for *R. argentea* and *O. niloticus* fishery). The declining fish abundance has been reported previously and was mainly attributed to over-exploitation, use of illegal fishing gears and deteriorating water quality from increasing pollution. Within the Winam gulf during the period 2012 -2016 [89] there was a difference between the turbid and eutrophic waters and clear open waters; and 13 fish species were recorded from the bottom and mid-water trawl surveys. The bottom-dwelling fish (*Clarias gariepinus*, *Synodontis victoriae*, *Bagrus docmak*, *Protopterus aethiopicus* and *Schilbe intermedius* were prevalent [89] in zones generally characterized by a relatively low dissolved oxygen; and high TN and TP, turbidity and fecal coliforms levels. The type of fish gear and areas sampled can influence the fish catch composition greatly. This is evident from the study as more untrawlable littoral and shallower waters (less than 5m) were fished. The bay area consists of diverse fish habitat types (wetlands and river mouth areas) which are considered refugia and breeding areas for fish species and are recognized as important critical areas for fish breeding and nursery grounds (hotspot areas for biodiversity). This explains the high number of fish species encountered during the survey. Within the Winam gulf, there was observed an increasing light transparency and decreasing water turbidity, which was also evident in the study area over the past six years. This could be explained by increased water exchange after the causeway at Mbita was opened from 2017. The shallow bays are flushed more frequently and even the water hyacinth does not permanently reside in the bay. Scattered water hyacinth mats are seen flushed in and out of the bay faster than before allowing uptake of nutrients; and not causing much effects on the underwater environmental conditions. The relatively clear water facilitates co-existence of many fish species, with great influences on the catch rates of haplochromines reported in northern Lake Victoria [90-91]. The bay and other shallow water zones are critical fish habitats and need to be protected from threats of expanding urban and industrial establishments and uncontrolled artisanal and commercial fisheries activities. The loss of biodiversity of fish in Lake Victoria is mainly due to habitat degradation and loss, eutrophication, predation and competition from introduced non-native fish species (Nile perch and Nile tilapia) and, in some cases, the unsustainable use of the lake from overfishing or the use of improper fishing gears [88]. As cage aquaculture expands within the Lake bays, good water quality is central towards achieving a sustainable tilapia cage aquaculture in Lake Victoria, and hence the need for such continuous long-term monitoring data.

A number of fish health management practices and biosecurity measures have been recommended [92-93] as fish are susceptible to a variety of infections and diseases caused by microbial pathogens and parasites [93]. Cage sites outside the bay were initially prone to fish kills due to poor cage sitting and sudden water deoxygenation events. This explains the relocation of cages to deeper lake areas (> 4 m deep) where natural water mixing occurs and farmers are assured of sustainable investments with application of good aquaculture practices and enforcement of developed guidelines.

4.4 The Blue economy concept and Kisumu bay ecology

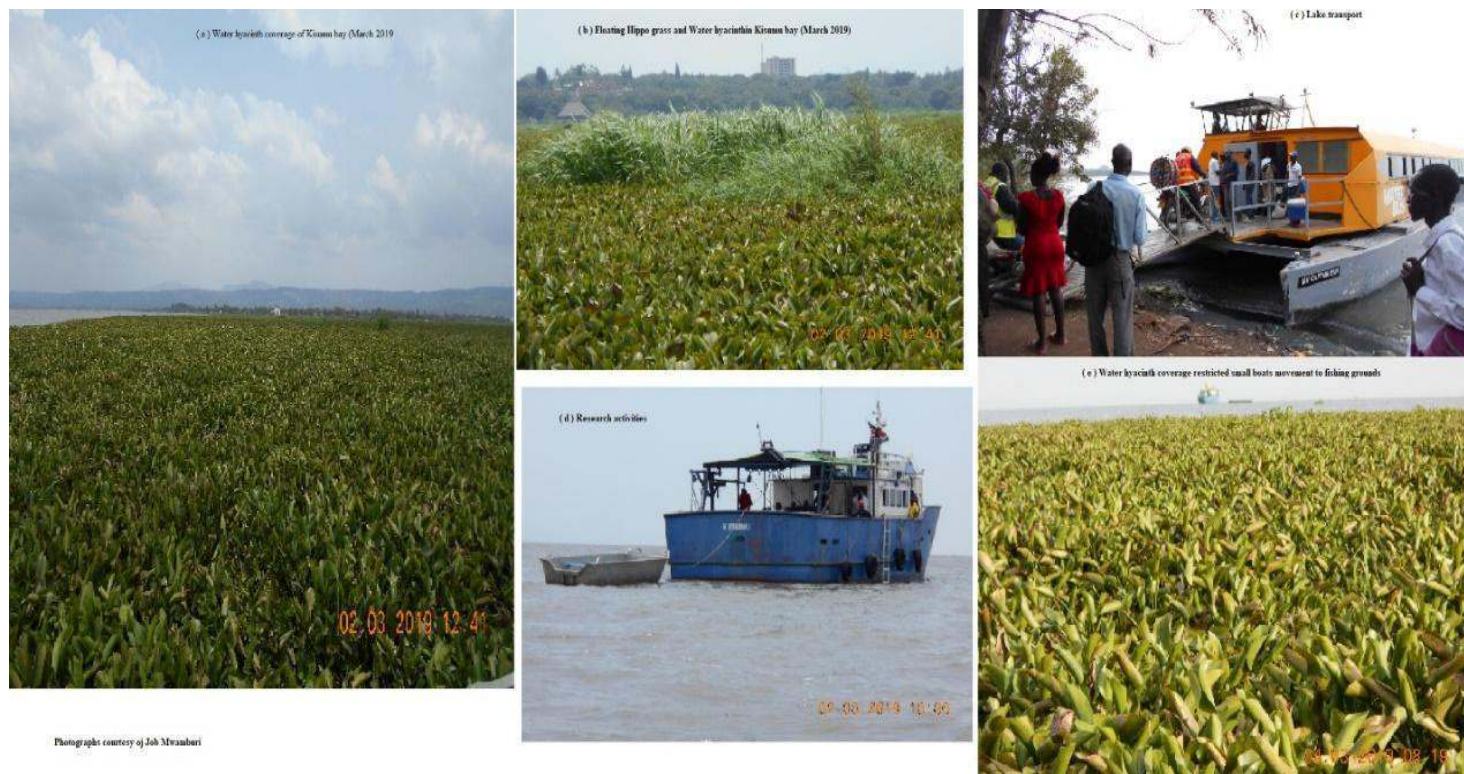
Maritime, fisheries, freshwater aquaculture, mariculture and bio-prospecting and mining activities are among the key components of the blue economic growth. The blue economy concept promotes economic growth, social inclusion and improved livelihoods at the same time as ensuring the environmental sustainability of oceans and seas [94]. The FAO flagship initiative (blue growth initiative) aims at supporting more productive, responsible and sustainable fisheries and aquaculture sectors by improving the governance and management of the aquatic ecosystems by conserving biodiversity and habitats and by empowering communities [95]. However human activities and types of land use have an increasing impact on the water resources. They affect both the quality and quantity of available safe water for the different uses; and hence the need for integrated approaches in the exploitation of both the marine and freshwater resources.

In Kenya, Lake Victoria dominates the inland waters capture fishery production (94,349 metric tonnes in 2021) which is about 71% of the country's total fish production [96], but globally there is still unmet demand for fish. Lake fish production is both of nutritional and commercial benefits to the riparian countries communities. The unique combination of high quality protein and important micronutrients (Iodine, calcium, zinc, iron, vitamin A, D and B12, polyunsaturated fatty acids such as eicosapentaenoic acid and docosahexaenoic acid) in fish plays a significant role in combating the triple burden of hunger micronutrient deficiencies and non-communicable diseases [97-98].

Fisheries and aquaculture production is projected to increase (by over 60%) by 2050 in order to feed the world population [99]. Aquaculture development is therefore set to provide the shortfall from capture production. Nile tilapia fish cage culture in Lake Victoria (Kenya) has attracted rapid investments since 2015 [41;43,100], with different types of designs and cage sizes (ca. 3,696 to 4,357 total cages during the 2018 survey). Fish cages in Lake Victoria are normally deployed in areas of suitable depth (inshore and offshore) and away from navigational routes, to allow free water circulation and to ensure they meet the acceptable water quality criteria. Using a profitability analysis model, it is suggested that improvement in economic profitability of Nile tilapia (*Oreochromis niloticus*) cage culture by changing from current small volume cages to large cages is more likely to attract formal financial institutions to cage culture financing and hence advancing the blue economy concept [101]. Hence the implementation of a spatial plan on the use of lake water for other integrated new developments juxtaposing the normal fisheries and maritime activities.

There is a growing body of evidence showing that biodiversity is important for generating and stabilizing ecosystem functions, and thus ensures the provisioning of numerous ecosystem services to society [102].

The inner and outer zones of the Kisumu Bay area surveyed has an approximate water surface area of about 26 Km² and is mainly dominated by urban and maritime related human activities. However, it is hypothesized that invasion and presence of water hyacinth caused a significant change to underwater biodiversity especially periphyton, microbial and macroinvertebrates communities although there is less matching scientific evidence of data due to a lack of long term investigation during periods on extensive coverage and difficulties in accessing areas under water hyacinth (Fig. 7). Again, urbanization or expansion of cities to meet the demands of the population and economic growth has a significant local impact on water resources. Habitat alteration due to urbanization can be both drastic and increasingly widespread. Large parcels of land are devegetated, paved and dramatically modified in ways that often greatly exceed habitat changes that occur from logging, traditional farming and many other land uses [103]. Also, land modifications during urban growth are usually long-term and indeed often intensify with time so that there is no opportunity for successional recovery [104]. Therefore, the preservation and restoration of local indigenous species biodiversity must be emphasized. Restoration ecologists, land managers, and urban planners can help maintain native birds in fragmented landscapes by a combination of short- and long-term actions designed to restore ecological function [103] (e. g the increase coverage of urban parks; preservation of wetland areas and restoration of vegetated riparian lands islands and shorelines). Macrophytes and vegetated littoral areas constitute important aquatic biodiversity refugia zones and; fish nursery and breeding areas.



Photograph courtesy of Job Mwamburi

Figure 9: The changing cover of Kisumu bay water surface during water hyacinth coverage and other maritime activities

Persistent pollution problems and ecological threats

Plastic pollution and associated ecological threats to both the freshwater and marine ecosystems is still perceived as a serious challenge to the blue economy developments due to the worldwide production and utilization of plastic materials and products. Already this has elicited a series of studies to quantify amounts in the different ecosystems and create the required awareness amongst the various stakeholders to ensure eradication of many types of plastics materials in surface waters. Rainfall variability has been observed over Africa between the years 1900-2000. Seasonal rainfall has also been established to be sensitive to El Niño climate variability [105]. An extremely wet category of rainfall which was recorded between 1997 and 1998 coincided with the 1997 El Niño episode experienced in Eastern Africa. The extremely dry season experienced between the years 2002 and 2006 would be attributed to the teleconnection of El Niño and La Niña in the region [106]. According to recorded water level trends [107] the Lake Victoria water levels rose steadily from 2008 to 2021 causing backflow flooding. The 2020 floods displaced hundreds to thousands of people and disrupted transportation, drinking water, sanitation, and power systems.

The trends (1987-2016) of Kenyan lake basin wide precipitation record high rainfall events in March April and May (long rains) and October November and December (short rains) [108]. The high variability and intensity of precipitation may exacerbate such problems due to increased flushing of uncollected and undisposed or poorly disposed solid wastes around the urbanized bay catchments especially after flooding events around the low-lying riparian areas. Conversion of forest lands and human settlements on riparian areas, including the growth of towns, is changing water quality and quantity, organic matter and nutrient inputs, and the diversity and composition of aquatic communities [109]. Natural buffer vegetation and wetland areas contribute to the maintenance of the aquatic habitat integrity and hence there is need to ensure their protection around the bay. Therefore, it is important for the Blue economy investments to ensure they adopt clear actions on reducing plastic pollution sources in surface waters (both marine and freshwater) juxtaposing the fast urban areas expansion and rapid population growth, especially around the major surface water bodies (lakes, rivers and reservoirs). Lake Victoria basin has one of the highest population densities in Africa [14].

The demand for land and its associated resources and ecosystem services, such as agriculture, grazing land, urbanization, rural and urban settlements is still increasing in much of East Africa. This has driven human-related change over short and long-term scales [110].

Cage Aquaculture

The national and county governments and multilateral development projects and programs have supported cage aquaculture development in the lake (the space of the Kenyan lake water surface suitable for cage aquaculture is about 362 Km² or 9 percent); with a multiplier effect on the associated infrastructure for feed and seed production countywide. International donors and development agencies' role should not be limited to technology transfer, and capacity building, but they should collaborate with the national and county governments in developing innovative financial models that favor sustainable aquaculture enterprises [111]. More collaborative research should be devoted to design and construction of climate smart culture systems, developing new species to guarantee supply of high-quality products; develop and scale low cost and highly nutritious fish feeds based on novel ingredients; and to enhance resilient livelihoods through innovative aquaculture practices and market linkages to create employment opportunities for youths and women [111]. Even though cage culture holds great promise for boosting output, generating employment opportunities, and enhancing the economic wellbeing of rural communities, site suitability for installing cages is still poorly regulated [111]. Over 45% of cage installations are made within 200 meters of fish breeding grounds, which may put other lake users in conflict [42]. To sustainably utilize the blue economy potential on cage aquaculture the National and County governments should provide cage culture investment and management guidelines to cater for any water use conflicts with other maritime activities; wild capture fisheries and demarcated fish protection areas. There is also a need to develop a long-term inter county lake surface spatial plan for a sustainable lake basin resources utilization.

Most of the tropical running waters studied, commonly exhibit longitudinal variations in water quality. Some of the medium-sized rivers drainage basins show downstream decrease in compositional loads and concentrations of solutes due to the different landuse characteristics. The downstream total N and P concentrations decreased (1740 ± 417 to 654 ± 144 μgL^{-1}) and increased (223 ± 35 to 51 ± 2 μgL^{-1}) respectively, along the Awach-Kibuon river [112]. However, the water quality of the more urbanized but small streams was evidently of poor quality based on some of the measured physico-chemical variables and total N and P (s). This is attributed to the use of urbanized streams as dilutants and conduits of treated effluents and other solid and liquid surface waste loads before entry into the bay waters.

Climatic changes can cause small to large variations in key environmental variables of the freshwater aquatic ecosystems, and influence the productivity and distribution patterns of the different aquatic communities and at varying scale. The study area is a hotspot area and is exposed to several anthropogenic impacts compared to other bay areas with less urbanized catchments. We therefore support the continuous monitoring of such key embayment areas, which are under increasing urbanization, human settlements and population, for the long-term effective strategy for habitat protection and ecosystem management.

CONCLUSIONS

The dominant fish species were represented by the two introduced species (*Lates niloticus* and *Oreochromis niloticus*) and the endemic *Haplochromines* and *Synodontis victoriae*. Other rare endemic fish species caught were *O. esculentus* and *O. variabilis* and the introduced *Coptodon zilli* and *O. leucostictus*. The bay area is among the suitable shallow sheltered zones for spawning and nursery ground for most fish species. *Haplochromines* are omnivores exploiting a wide range of food resources (algae molluscs, insects, plant material detritus and zooplankton) and this explains their dominance and wide occurrences within the bay. Conditions encountered within the bay favour zooplankton growth and hence the zooplanktivores exploit the available food item. Also pronounced seasonality effects on available nutrients ensures sustained phytoplankton communities commonly found dominated by cyanobacteria within major bays of the Winam gulf. The decline in fish production mainly results from the use of illegal undersize nets that harvest the juvenile fish species; unprotected breeding and nursery areas; beach seining and increasing fishing effort. Therefore, the fisheries managers and community BMUs need to ensure measures to reduce the fish decline are enforced. Embayments and beach sites around urbanizing lake areas need to be supported with improved sanitation facilities, waste collection and infrastructure to reduce the direct influx of both surface waste discharges and solid wastes into the surrounding aquatic habitats. Also the rehabilitation of urban impacted streams and maintenance of the littoral wetland areas and vegetation will ensure the sustenance of the natural buffer zones and reduce silt deposition and sedimentation into the bay, this study provided a more localised and specific investigation of the ecology of an "area of concern", experiencing increasing population and economic growth, as the focus on the blue economy opportunities continue to emerge and the gulf-wide need to accommodate cage aquaculture in the same surface water space. Overall, there appears to be ecologically important effects of the spatial extent of the different urban land uses on adjoining water quality. The bay water was found to exhibit eutrophic (based on algal biomass) to hypereutrophic conditions (based on the and light transparency values and total phosphorus concentration). However, based on the mean TSI of these water quality variables (Secchi depth, TP and chlorophyll-a), the lake is still of a high ecological productivity (eutrophic state).

Recommendations to inform management

- Lake riparian counties attract different investments and developments and should therefore incorporate strategies to minimize destruction of wetland areas; improve sanitation services, enhance watershed management and reduce eutrophication sources; and improve solid waste management plans.
- Investments in eco-tourisms activities can create more awareness on the need to protect biodiversity and critical aquatic habitats.

There is need to reduce any potential conflicts between water users and maritime activities as cage aquaculture expands; and more importantly improve capability and preparedness for emergencies such as the recent cage-related fish kills in Lake Victoria, through strengthening of research, training, extension, and advisory services.

DECLARATION

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

FUNDING

There was no funding.

AUTHOR CONTRIBUTIONS

JM: Investigation; Writing -original draft; Writing-review and editing; conceptualization; Formal analysis; Project administration; supervision; methodology visualisation and validation; **CAM and CSN:** Investigation; Writing-review and editing; Project administration; supervision; methodology visualisation and validation; **EA; SM; JM; JA; GO:** Investigation Writing -original draft; Writing-review and editing; conceptualization; Formal analysis; methodology visualisation and validation.

ACKNOWLEDGEMENTS

This work was supported by Kenya Marine and Fisheries Research Institute (KMFRI). We would like to acknowledge the support of KMFRI in conducting the field activities and development of this report. Much appreciation to the scientific, mercantile staff and technical teams at KMFRI who were involved in the data collection and analysis.

REFERENCES

- Ongore, C. O., Aura, C.M., Ogari, Z., Njiru, J. M., & Nyamweya, C. S. (2018). Spatial – temporal dynamics of water hyacinth, *Eichhornia crassipes* (Mart.), other macrophyte and their impact on fisheries in Lake Victoria, Kenya. *J. of Great Lakes Research*,
- Tebbs, E., Byrne, A., Lomeo, D., Thompson, H., Owoko, W., Nyaga, J., Ongore, C., Last, J., Migeni, Z., & Everitt, L. (2023). Satellite earth observations of the sustainable management of the African great lakes. *Kings College London*.
- Olokotum, M., Mitroi, V., Troussellier, M., Semayalo, R., Bernard, C., Montuelle, B., Okello, W., Quiblier, C., & Humbert, J. F. (2022). A review of the socioecological causes and consequences of cyanobacterial blooms in Lake Victoria. *Harmful Algae*, 96: 10129
- Simiyu, B. M., Oduor, S. O., Rohrlack, T., Sitoki, L., & Kurmayer, R. (2018). Microcystin content in phytoplankton and in small fish from eutrophic Nyanza Gulf Lake Victoria Kenya. *Toxins*, 10:275,
- Sitoki, L., Kurmayer, R., & Rott, E. (2012). Spatial variation of phytoplankton composition, biovolume and resulting microcystin concentrations in the Nyanza Gulf (Lake Victoria, Kenya). *Hydrobiologia*, 691: 109–122.
- Ochumba, P. B. O., & Kibaara, D. I. (1989). Observations on blue-green algal blooms in the open waters of Lake Victoria, Kenya. *Afr. J. Ecol.*, 27(1):23–34.
- Obuya, J.A., Onyango, H. O., Olokotum, M., Zepernick, B., Natwora, K., Otieno, D., Owoko, W., Achieng, D., Otuo, P. W., & Keyombe, J. L. (2024). Socio-economic consequences of cyanobacterial harmful algal blooms in small-scale fishing communities of Winam gulf, Lake Victoria. *J. Great Lakes Research*, 50(5):102236,
- Brown, K.M., Barker, K.B., Wagner, R.S., Ward, C.S., Sitoki, L., Njiru, J., Omondi, R., Achiya, J., Getabu, A., McKay, R. M., Bullerjahn, G. S., & NSF-IRES Lake Victoria Research Consortium. (2024). Bacterial community and cyanotoxin gene distribution of the Winam gulf, Lake Victoria, Kenya. *Environmental Microbiology*, 16(3):e13297,
- Roegner, F. A., Corman, J. R., Sitoki, L. M., Kwena, Z. A., Ogari, Z., Miruka, J. B., Xong, A., Weirich, C., Aura, C. M., & Miller, T. R. (2023). Impacts of algal blooms and microcystins in fish on small-scale fishers in Winam Gulf Lake Victoria: implications for health and livelihood. *Ecology and Society*, 28: 49,
- Roegner, A., Sitoki, L., Weirich, C., Corman, J., Owage, D., Umami, M., Odada, E., Miruka, J., Ogari, Z., Smith, W., Rejmankova, E., & Miller, T. R. (2020). Harmful algal blooms threaten the health of peri-urban fisher communities: A case study in Kisumu bay Lake Victoria, Kenya. *Exposure and Health*, 12: 835-848
- Onyango, D. M., Orina, P. S., Ramkat, R. C., Kowenje, C., Githukia, C. M., Lusweti, D., & Lung'anya H. B. O. (2020). Review of current state of knowledge of microcystin and its impacts on fish in Lake Victoria. *Lakes & Reservoirs: Science Policy and Management for Sustainable Use*, 25: 350–361.
- Simiyu, B. M., Amukhuma, H. S., Sitoki, L., Okello, W., & Kurmayer, R. (2022). Interannual variability of water quality conditions in the Nyanza gulf of Lake Victoria, Kenya. *J. Great Lakes Research*, 48: 97–109,
- Omara, T., Nagawa, C. B., Kyarimpa, C., Bohmdorfer, S., Rosenau, T., Lugasi, S. O., Matovu, H., Odongo, S. & Ssebugere, P. (2023). Lacustrine cyanobacteria, algal blooms and cyanotoxins in East Africa: Implications for human and ecological health protection. *Phycology*, 3:147–167.
- Olokotum, M., Humbert, J. F., Quiblier, C., Okello, W., Semyalo, M., Troussellier, M., Marie, B., Baumann, K., Kurmayer, R., & Bernard, C. (2020). Characterization of potential threats from cyanobacterial toxins in Lake Victoria embayments and during water treatment, *Toxins (Basel)*, 14(10): 664.
- Mchau, G. J., Makule, E., Machunda, R., Gong, Y. Y., & Kimanya, M. (2019). Harmful algal bloom and associated health risks among users of Lake Victoria freshwater: Ukerewe Island, Tanzania. *Journal of Water and Health*, 17: 826–836.
- Sekadiende, B. C., Lyimo, T. J., & Kurmayer, R. (2005). Microcystin production by cyanobacteria in the Mwanza gulf (Lake Victoria, Tanzania). *Hydrobiologia*, 543: 299–304. DOI:10.1007/s10750-004-6949-6
- Aura, C.M., Mwarabu, L. R., Nyamweya, C. S., Ongore, C. O., Musa, S., Keyombe, J. L., Guya, F., Awuor, J. F., Owili, M., & Njiru, J. M., (2024). Unbundling sustainable community-based cage aquaculture in an afro-tropical lake for blue growth. *J. Great Lakes Research*, 50(5):102410.
- Okechi, J. K., Peoples, N., Nyamweya, C. S., Glaser, S., & Kaufman, L. (2022). The ecological health of Lake Victoria (Kenya) in the face of growing cage aquaculture. *Conservation Science and Practice*, 4(11): e12826,
- Hamilton, S. E., Gallo, S. M., Krach, N., Nyamweya, C. S., Okechi, J. K., Aura, C., Ogari, Z., Roberts, P., & Kaufman, L. (2019). The use of unmanned aircraft systems and high-resolution satellite imagery to monitor tilapia fish cage expansion in Lake Victoria Kenya. *Bull. Mar. Sci.*, 96: 71–93
- Ogello, E. O., & Munguti, J. M. (2016). Aquaculture: a promising solution for food security, poverty and malnutrition in Kenya. *Afri. J. Food, Agric., Nutr. Dev.*, 16(4): 11332-11351
- Opiyo, M. A., Marijani, E., Muendo, P., Odede, R., Leschen, W., & Charo-Karisa, H. (2018). A review of aquaculture production and health management practices of farmed fish in Kenya. *Int. J. Veterinary Sciences and Medicine*,
- FAO (2022). The state of the world fisheries and aquaculture 2022: towards blue transformation. Rome; FAO PP. 1-11.
- Maithya, J., Charo, H., Wangila, B. C., Ouma, H., & Orinda, C. (2003). Aquaculture strategy for restoration of threatened Lake Victoria fishes: The case for *Oreochromis variabilis* (Boulenger, 1906) and *Labeo victorianus* (Boulenger, 1901). *J. Fish Biol.*, 15: 359-412.
- Marshall, B. E. (2018). Guilty as charged: Nile perch was the cause of the haplochromine decline in Lake Victoria. *Can. J. Fish. Aquat. Sci.*, 75, 1542–1559.

25. Nkalubo, W., Chapman, L. & Muyodi, F. (2014). Feeding ecology of the intensively fished Nile perch, *Lates niloticus* in Lake Victoria Uganda. *Aquat. Ecosyst. Health Manag.*, 17:62-69.
26. Katunzi, E. F. B., Van Densen, W. L. T., Wanink, J. H., & Witte, F. (2006). Spatial and seasonal patterns in the feeding habits of juvenile *Lates niloticus* (L.) in Mwanza gulf of Lake Victoria. *Hydrobiologia*, 568: 121-123.
27. Outa, N. O., Yongo, E., & Keyombe, J. L. A. (2017). Ontogenic changes in prey by prey ingested by Nile perch (*Lates niloticus*) caught in Nyanza gulf of Lake Victoria Kenya. *Lakes Reserv. Res. Manage.*, 20:1-5.
28. Kische-Machumu M. A., Witte, F., Wanink, J. H., & Katunzi, E. F. (2012). The diet of Nile perch, *Lates niloticus* (L.) after resurgence of haplochromine cichlids in the Mwanza gulf of Lake Victoria. *Hydrobiologia*, 682: 111-119.
29. Jobling, S & Tyler, C. R. (2003). Endocrine disruption in wild freshwater fish. *Pure and Applied Chem.*, 75: 2219-2234.
30. UN (2014). United nations Blue economy concept paper.
31. World Bank and United Nations (2017). Blue economy: Increasing long-term benefits of the sustainable use of marine resources for small island developing state and coastal least developed countries.
32. UN (2018). Working group on the issue of human rights and transnational corporations and other businesses enterprises. Note A/73/163 by secretary general UN General assembly, New York.
33. Aura, C. M., Awandu, H., Mziri, V., Awuor, F. J., Nyamweya, C. S., Musa, S., & Osore, M., (2025). Unbundling fish kills associated losses and compliance aspects in lacustrine cage aquaculture farms. *Aquaculture, Fish and Fisheries*, 5(4): e70101,
34. Kenya National Bureau of Statistics (KNBS). (2019). The 2019 Kenya Population and Housing Census. Analytical Report on Population Dynamics. Volume VI, Ministry of Treasury and Planning. Nairobi.
35. Kenya National Bureau of Statistics (KNBS). (2014-2018). Statistical abstract 2014-2018, ISBN 9966-767-49-5 Nairobi Kenya.
36. Owiti, H. O., Ochiewo, J., Aura, C. M., Kayanda, R., Sunil, S. S., Otuo, P. W., Obuya, J. A., & Njiru, J. M. (2021). The lost coin: Redefining the economic and financial value of small-scale fisheries, the case of Lake Victoria, Kenya. *Social Sciences & Humanities Open*, 4(1): 100221
37. Opande, G. O., Onyango, J. C., & Wagai, S. O. (2004). Lake Victoria: The water hyacinth (*Eichhornia crassipes* [Mart.] Solms), its socio-economic effects, control measures and resurgence in the Winam gulf. *Limnologia*, 34:105-109.
38. May, L., Dobel, A. J., & Ongore, C. (2022). Controlling water hyacinth (*Eichhornia crassipes* (Mart.) Solms): a proposed framework for preventive management. *Inland Waters*, 12(1):163-172.
39. Fusilli, L., Collins, M. O., Laneve, G., Palombo, A., Pignatti, S., & Santini, F., (2013). Assessment of the abnormal growth of floating macrophytes in Winam Gulf (Kenya) by using MODIS imagery time series. *Int. J. Applied Earth Observation and Geoinformation*, 20: 33-41.
40. Onyango, D. O., & Opiyo, S. B., (2021). Detection of historical landscape changes in Lake Victoria Basin, Kenya using remote sensing multi-spectral indices. *Watershed Ecol. Environ.*, 4:1-11
41. Orina, P., Ogello, E., Kembenya, E., Muthoni, C., Musa S., Ombwa, V., Mwainge, V., Abwao, J., Ondiba, R., Kengere, J., & Karoza, S. (2021). The state of cage culture in Lake Victoria: A focus on sustainability, rural economic empowerment, and food security. *Aquatic Ecosystem Health & Management*, 24(1): 56-63.
42. Njiru, J. M., Aura, C. M., & Okechi, J. K. (2019). Cage fish culture in Lake Victoria: A boon or a disaster in waiting? *Fish. Manag. Ecol.*, 26: 426-434,.
43. Aura C. M., Musa S., Yongo, E., Okechi, J., Njiru, J. M., Ogari, Z., Wanyama, R., Charo-Karisa, H., Mbugua, H., Kidera, S., Ombwa, V., & Oucho, A. J., (2018). Integration of mapping and socioeconomic status of cage culture: Towards balancing lake-use and culture fisheries in Lake Victoria, Kenya. *Aquaculture Research*, 49: 532-545.
44. APHA (2005). American Public Health Association (APHA), Standard methods for the examination of water and wastewater, 21st Edn. APHA-AWWA-WEF, Washington, DC. ISBN:0875530478S
45. LVFO (2007). Standard operating procedures for collecting biological and ecological information from the fishes of Lake Victoria. Lake Victoria Fisheries Organization, SOP No. 1. Jinja Uganda 45pp.
46. Shannon, C. E., & Wiener, W. (1949). *The mathematical theory of communication*. University of Illinois Press, Urbana, IL.
47. Simpson, E. H. (1949). Measurement of diversity. *Nature*, 163: 688
48. Pielou, E. C. (1975). *Ecological diversity*. Wiley and Sons New York
49. R Core Team (2020). R: A language and environment for statistical computing. R Foundation for Statistical Computing.
50. Carlson, R., (1977). A trophic state index for lakes. *Limnol. Oceanogr.*, 22(2): 361-369.
51. Le' Cren E. D. (1951). The length-weight relationship and seasonal cycle in gonad weight and condition in the Perch (*Perca fluviatilis*). *J. Animal Ecology*, 20:201-219
52. Murdock, J., Roelke, D., & Gelwick, F. (2004). Interactions between flow, periphyton and nutrients in a heavily impacted urban stream: Implications for stream restoration effectiveness. *Ecological Engineering*, 22: 197-207
53. Carey, R. O., & Migliaccio, K. W., (2009). Contribution of wastewater treatment effluent to nutrient dynamics in aquatic ecosystems: A review. *Environ. Manage.*, 44:205-217.
54. USEPA (2013). Aquatic life ambient water quality criteria for ammonia-Freshwater 2013. U.S Environmental Protection Agency Office of Water. EPA-822-R-13-001.
55. Constable, M., Charlton, M., Jensen, F., McDonald, K., Craig, G., & Taylor, K. W., (2003). An ecological risk assessment of ammonia in the aquatic environment. *Human and Ecol. Risk Assess.*, 9: 527-546.
56. Miruka, J. B., Getabu, A., Sitoki, L., James, O., Mwamburi, J., George, O., Chrispine, N., & Odoli, C. (2021). Water quality, phytoplankton composition and microcystin concentration in Kisumu Bay (Kenya) of Lake Victoria after a prolonged water hyacinth infestation period. *Lake & Reservoirs: Research & Management*, 26, e12380
57. Gikuma-Njuru, P., Hecky, R. E., MacIntyre, S., & Guildford, S. J. (2013). Strong spatial differentiation of N and P deficiency, primary productivity and community composition between Nyanza gulf and Lake Victoria (Kenya, East Africa). *Freshw. Bio.*, 58(11): 2237-2252.
58. Omondi, C. A., Omondi R., Onyango, D., & Onyangore H. D. (2023). Assessment of the current status of water quality and nutrients enrichment of rivers Kisat and Auji and Nyalenda wastewater sedimentation ponds. *Pan Afr. Sci. J.*, 2(3).
59. Kobingi, N., Raburu, P. O., Masese, F. O., & Gichuki, J. (2009). Assessment of pollution impacts on the ecological integrity of the Kisian and Kisat rivers in Lake Victoria drainage basin Kenya. *Afr. J. Environ. Sci. Technol.*, 3: 97-107
60. Orina, B., Onyango, D. M., Lungayia, H. B., Oduor, A., Sifuna, A. W., & Otuya, P. (2020). Water quality assessment of fishing beaches in Lake Victoria Kenya basin. *Open J. Ecology*, 10(1): 22-35.
61. Gikuma-Njuru, P., & Hecky, R. E. (2005). Nutrient concentrations in Nyanza gulf Lake Victoria, Kenya: Light limits alga demand and abundances. *Hydrobiologia*, 534: 131-140.

62. Chorus, I., & Bartram, J. (Eds.), (1999). *Toxic cyanobacteria in water*. London: E & FN Spon, on behalf of UNESCO WHO and UNEP.
63. Olokutum, M., Troussellier, M., Escalas, A., Hamlaoui, S., Okello, W., Semyalo, M., Humbert, J. F., & Bernard, R. (2021). High nutrient loading and climate parameters influence the dominance and dissimilarity of toxigenic cyanobacteria in northern bays of Lake Victoria. *J. Great Lakes Res.*, 47: 985–996.
64. Okello, W., & Kurmayer, R. (2011). Seasonal development of cyanobacteria and microcystin production in Ugandan freshwater lakes. *Lakes & Reservoirs: Research and Management*, DOI: 10.1111/J.1440-1770.2011.00450.x
65. Chen, H., Burke, E., & Prepas, E. E., (2011). Cyanobacterial toxins in freshwater. In: Nragu J. O. (Ed.) *Encyclopedia of environmental health*.
66. Barko, J. W., & James, W. J., (1998). Effects of submerged aquatic macrophytes on nutrient dynamics, sedimentation, and resuspension. *The structuring role of submerged macrophytes in Lakes*, Springer, New York.
67. Bridges, T. S., Gustavson, K. E., Schroeder, P., Ells, S. J., Hayes, D., Nadeau, S. C., Palermo, M. R., & Patmont, C. R., (2010). Dredging processes and remedy effectiveness. *Integr. Environ. Assess. Manag.*, 6(4): 619–630.
68. Guya, F. J. (2024). Biotic and abiotic factors driving water hyacinth, *Pontederia crassipes*, phenology in Nyanza gulf, Lake Victoria, Kenya. *J. Great Lakes Res.*, 102355
69. Guya, F. J. (2020). Biogeochemical characterization, phosphorus sources and intrinsic drivers to its speciation within the Nyanza gulf of Lake Victoria. *Lakes & Reservoirs: Science Policy and Management for sustainable Use*, 25(1): 31–43.
70. Guya, F. J. (2013). Bioavailability of particle-associated nutrients as affected by internal regeneration processes in the Nyanza gulf region of Lake Victoria. *Lakes & Reservoirs: Research and Management*, 18: 129–143.
71. Guya, F. J. (2019). Intrinsic and extrinsic sources of phosphorus loading into the Nyando River, Kenya. *Lakes & Reservoirs: Science Policy and Management for Sustainable Use*, 24(4): 362–371; DOI:10.1111/lre.12289
72. Dove, A., & Chapra, S. C., (2015). Long-term trends of nutrient and trophic response variables for the Great Lakes. *Limnol. Oceanogr.*, 60:696–721.
73. Treist, L., Lung'ayia, H. Ndiritu, G., & Beyene, A. (2012). Epilithic diatoms as indicators in tropical African rivers. *Hydrobiologia*, 695:343-360.
74. Nyamweya, C., Lawrence, T., Ajode, M. Z., Smith, S., Achieng, A. O., Barasa, J. E. et al. (2023). Lake Victoria: Overview of research needs and the way forward. *J. Great Lakes Res.*, 49(6): 102211.
75. Njiru, J. M. (2003). Ecology and population characteristics of Nile tilapia *Oreochromis niloticus* (L.) in Lake Victoria, Kenya. Ph.D Thesis, Moi University.
76. Ogari, J., & Asila, A. A. (1992). Status of fisheries of Lake Victoria Kenya sector. *FAO Fisheries Report 475*, FAO Rome.
77. Mkumbo, O. C. (2002). Assessment and management of Nile perch (*Lates niloticus*, L.) stocks in the Tanzanian waters of Lake Victoria. Ph.D Thesis, University of Hull.
78. Ogari, J. & Asila, A. A. (1988). The food of the Nile perch *Lates niloticus* (L.) after the disappearance of the haplochromines cichlids. *J. Fish Biol.*, 32: 571-577.
79. Manyala, J. O., (1995). Growth, mortality and yield of dagaa (*Rastrineobola argentea*) Pellagrini 1904 in the Winam gulf. *Scientia Marina*.
80. Hughes, N. F. (1986). Changes in feeding biology of the Nile perch *Lates niloticus* in Lake Victoria since 1960. *J. Fish Biol.*, 29:541–54.
81. Goldschmidt, T., Witte, F., & Wanink, J. H., (1993). Cascading effects of the introduced Nile perch in Lake Victoria. *Conserv. Biol.*, 7: 686-706.
82. Ogutu-Ohwayo, R. E. (1990). The decline of the native fishes of Lake Victoria and Kyoga and the impact of introduced species. *Environ. Biol. Fish.*, 27: 81-96.
83. Njiru, M., Okeyo-Owuor, J.B., Muchiri, M., & Cowx, I. G. (2004). Shifts in the food of Nile tilapia *Oreochromis niloticus* (L.) in Lake Victoria, Kenya. *Afr. J. Ecol.*, 42: 163-170.
84. Yongo, E., & Outa, N. (2016). Growth and population of Nile tilapia *Oreochromis niloticus* (L.) in the open waters of Lake Victoria, Kenya. *Lakes & Reservoirs*, 21: 375–379.
85. Yongo, E., Manyala, J., & Agembe, S. (2017). Growth, mortality and recruitment of silver cyprinid (*Rastrineobola argentea*). *Lakes & Reservoirs*, 23: 244–249.
86. Yongo, E., Agembe S., Outa, N., & Owili, M. (2018). Growth, mortality and recruitment of Nile perch (*Lates niloticus*) in Lake Victoria, Kenya. *Lakes & Reservoirs*, 23: 17–23.
87. Outa, N., Yongo, E. O., Keyombe, J. L. A., Ogello, E. O., & Wanjala, D. N. (2020). A review on the status of some major fish species in Lake Victoria. *Lakes & Reservoirs*, 25: 105–111.
88. Kundu, R., Aura, C.M., Nyamweya, C., Agembe, S., Sitoki, L., Lungayia, H. B. O., Ongore, C., Ogari, Z., & Werimo, K. (2017). Changes in pollution indicators in Lake Victoria. *Lakes & Reservoirs*, 22(3): 199-214.
89. Seehausen, O., van Alphen, J. J. M., & Witte, F. (1997a). Cichlid fish diversity threatened by eutrophication. *Science*, 277:1808-1811.
90. Seehausen, O., Witte, F., Katunzi, E. F., Smits, J., & Bouton, N. (1997b). Pattern of the remnant Cichlid fauna in southern Lake Victoria. *Conserv. Biol.*, 11:890-904.
91. Kyule-Muendo, D., Otachi, E., Awour, F., Ogello, E., Obiero, K., Abwao, J., Muthoni, C., & Munguti, J. (2022). Status of fish health management and biosecurity measures in Western Kenya. *CABI Agriculture and Bioscience*. DOI 10.1186/s43170-022-00086-7
92. Mwainge, V. M., Ogwai, C., Aura, C. M., Mutie, A., Ombwa, V., Nyaboke, H., Oyier, K. N., & Nyaundi, J. (2021). An overview of fish disease and parasite occurrence in cage culture. *Aquat. Ecosyst. Health Manage.*, 24(1): 43-55.
93. World Bank (2017). The potential of the blue economy. World Bank, Washington DC.
94. FAO (2018). The FAO blue growth initiative: strategy for the development of fisheries and aquaculture in Eastern Africa. *FAO circular FAA/C1161*.
95. FASB (2021). Fisheries annual statistical bulletin. Ministry of Agriculture Livestock Fisheries & Cooperatives, Kenya.
96. Wessels, L., Kjelleevold, M., Kolding, J., Odoli, C., Aakre, I., Reich, F., & Pucher, J. (2023). Putting small fish on the table. *Food Security*, 15: 1025–1039.
97. Kolding, J., van Zwieten, P., Maritim, F., Funge-Smith, S., & Poulain, F. (2023). Freshwater small pelagic fish and fisheries in major African lakes. *FAO Technical Paper No. 642*.
98. FAO (2014). The state of food and agriculture: Innovation in family farming. FAO Rome Italy.
99. Hamilton, S. E., et al. (2022). High-resolution bathymetries and shorelines for the Great Lakes of the White Nile basin. *Sci Data*, 9, 642.
100. Musa, S., Aura, C. M., & Okechi, J. K. (2021). Economic analysis of tilapia cage culture in Lake Victoria using different cage volumes. *J. Appl. Aquacult.*, 34(3):674–692.
101. Socolar, J. B., Glory, J. J., Kunin, W. E., & Edwards, D. P. (2016). How should beta-diversity inform biodiversity conservation? *Trends Ecol. Evol.*, 31: 67–80.
102. Marzluff, J. M. & Ewing, K. (2001). Restoration of fragmented landscapes for the conservation of birds. *Restoration Ecol.*, 9(3):739-755.
103. McKinney M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biol. Conserv.*, 127: 247-260.
104. Hulme, M. (1992). Rainfall changes in Africa: 1931-1960 to 1961-1990. *Int. J. Climatology*, 12: 665–699.

105. Anyamba, A., Tucker, C. J., & Mahoney, R., (2002). From El Niño to La Niña: Vegetation response patterns. *J. Climate*, 15, 3096-3103.
106. NASA Earth Observatory (2020). Lake Victoria's rising water levels. Accessed 17 April 2024.
107. Wabire, O. E., Mukhovi, S. N., & Nyandega, I. A. (2016). The spatial and temporal characteristics of rainfall over the Lake Victoria basin of Kenya in 1987-2016. *Atmospheric and Climate Sciences*, 10(2): 240-257.
108. Odada, E. O., Olago, D. O., & Ochola, W. O. (Eds.), (2006). Environment for development: An ecosystems assessment of Lake Victoria basin. *UNEP/PASS*, Nairobi, Kenya.
109. Mugo, R., et al. (2020). Quantifying Land Use Land Cover Changes in the Lake Victoria Basin Using Satellite Remote Sensing. *Remote Sensing*, 12, 17: 2829.
110. Munguti, J. M., et al. (2023). Role of multilateral development organizations, public and private investments in aquaculture subsector in Kenya. *Front. Sustain. Food Syst.*, 7:1208918.
111. Osure, G. O.; Getabu, A.; Aura, C. M.; Omondi, R.; & Basweti, E. (2022). Longitudinal fish species diversity and distribution in a medium-sized Afro-tropical river. *Pan Africa Sci. J.*, 2(2).