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KENYA Aquatica

A Scientific Journal of Kenya Marine and Fisheries Research Institute

KMFRI

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Editorial

Welcome to Volume 11, Issue 1 of Kenya Aquatica, a collection of research contributions advancing our understanding of Kenya's marine and coastal environments and their role in supporting livelihoods, biodiversity conservation, food security and the Blue Economy. Kenya's coastal and marine ecosystems including coral reefs, mangroves, seagrass beds, fisheries and coastal waters, are among the nation's most valuable natural assets. However, these ecosystems face intensifying pressures from climate change, pollution, overexploitation of resources, habitat degradation and rapidly expanding coastal development.

The studies featured in this issue address several of these emerging challenges while offering practical solutions for sustainable management. Climate change and ecosystem resilience emerge as central themes. Research on deep-water seaweed farming demonstrates how innovation can help coastal communities adapt to rising temperatures, disease outbreaks and other environmental stressors that increasingly threaten traditional seaweed production systems. Similarly, studies on coral reefs in Kilifi reveal evidence of reef degradation, declining fish stocks and climate-related bleaching, reinforcing the urgency of strengthening reef conservation and restoration initiatives.

Pollution and environmental quality are also prominent concerns. Reviews of heavy metal contamination in Mombasa's coastal waters highlight the growing influence of industrialization, urbanization and port activities on ecosystem health and human well-being. Complementing this work, oceanographic modelling of marine plastic pollution along the northern Kenyan coast provides new insights into the movement and accumulation of marine debris, offering actionable evidence for waste management and pollution mitigation strategies.

Several papers underscore the role of fisheries and aquaculture in achieving national food security. Investigations into marine crab fisheries reveal untapped opportunities for livelihood diversification and nutritional enhancement, while studies on Artemia production contribute knowledge that could strengthen aquaculture development through improved live-feed availability. Research on fish markets in Western Kenya further highlights how efficient value chains, gender equity and value addition enhance fisheries-based livelihoods.

A notable feature of this volume is the growing of citizen science and community participation. Both the review of citizen science approaches to water quality monitoring and the coral reef assessment conducted by local community members demonstrate how collaborative research can strengthen environmental stewardship, generate cost-effective monitoring data and build local capacity for resource management.

The findings presented in this issue align closely with Kenya's Blue Economy Strategy, Vision 2030, climate adaptation priorities and international commitments under the Sustainable Development Goals, particularly SDGs 2, 5, 13, 14, and 17. They illustrate the critical role of science in informing policy, guiding investment decisions and promoting evidence-based management of aquatic resources.

Looking ahead, there remains a need for long-term environmental monitoring, integrated ecosystem assessments, climate-change impact studies, pollution control research and stronger socio-economic analyses of coastal livelihoods. Continued investment in research capacity, technological innovation and community-based approaches will be essential for safeguarding Kenya's marine and coastal resources for future generations.

We sincerely thank all authors, reviewers, editors, partner institutions and stakeholders whose commitment and expertise have made this issue possible. Their contributions continue to strengthen knowledge generation, support science-based policy development and advance the sustainable management of Kenya's aquatic environments.

As Kenya Aquatica enters its eleventh volume, we invite researchers, practitioners, scholars and policymakers to contribute to strengthening the evidence base for Kenya's Blue Economy.

About
Kenya Aquatica

Kenya Aquatica is the Scientific Journal of the Kenya Marine and Fisheries Research Institute (KMFRI). The aim of the Journal is to provide an avenue for KMFRI researchers and partners to disseminate knowledge generated from research conducted in the aquatic environment of Kenya and resources therein and adjacent to it. This is in line with KMFRI's mandate to undertake research in "marine and freshwater fisheries, aquaculture, environmental and ecological studies, and marine research including chemical and physical oceanography", in order to provide scientific data and information for sustainable development of the Blue Economy.

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Featured front cover picture: Rhythm of Lake Victoria shoreline: boats, fishers and traders united in anticipation of the day's catch.

Featured back cover picture: Diving for knowledge - *KMFRI researcher Dr. Juliet Furaha uncovering the rich coral heritage of the Indian Ocean.*

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Length–weight relationship and condition factor of Nile tilapia (*Oreochromis niloticus*) fingerlings fed diets containing different proportions of water spinach (*Ipomoea aquatica*)

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Abstract

This study examined the effects of dietary water spinach (*Ipomoea aquatica*) on the length–weight relationship and condition factor of Nile tilapia (*Oreochromis niloticus*) fingerlings. Five diets containing 0%, 5%, 10%, 15% and 20% *I. aquatica* were formulated to 30% crude protein and fed to fingerlings reared in a completely randomized design. The regression coefficient *b*, obtained by log-transformation of the length–weight data, was 2.641, 2.832, 2.532, 2.452 and 2.441 for the 0%, 5%, 10%, 15% and 20% diets, respectively, with high coefficients of determination (R^2 = 0.88–0.98) in all treatments. Because *b* was below 3 in every diet, growth was negatively allometric. The condition factor (K) also varied among diets and was highest in fingerlings fed 5% *I. aquatica* (K = 2.55). Water quality remained within the range suitable for *O. niloticus* throughout the trial. These findings indicate that *I. aquatica* can be included in tilapia feeds without compromising fish condition, and that a 5% inclusion gave the best performance. Water spinach therefore shows promise as an affordable, locally available protein source for sustainable aquaculture and for food and nutrition security.

Keywords: *Ipomoea aquatica*, condition factor, length–weight relationship, aquaculture, fish nutrition

Introduction

Aquaculture contributes substantially to food and nutrition security and to economic development worldwide (FAO, 2022). Nile tilapia (*Oreochromis niloticus*) is the most widely farmed finfish after the carps, with global production rising from about 1,001.5 thousand tonnes in 2000 to approximately 4,407.2 thousand tonnes in 2020 (FAO, 2022). Its fast growth, tolerance of high stocking densities, ease of reproduction, consumer acceptability and resistance to disease make it well suited to culture (Eze *et al.*, 2019). Quantitative descriptors such as growth, mortality, recruitment, condition factor and the length–weight relationship are central to fisheries biology and to assessing the performance of farmed stocks (Anani and Nunoo, 2016).

The length–weight relationship and condition factor are widely used to describe growth pattern and to infer the well-being and physiological state of fish (Ridha, 2011; Momi *et al.*, 2020). Growth may be positively allometric (weight increasing faster than length), negatively allometric (length increasing faster than weight) or isometric (length and weight increasing proportionally). The condition factor reflects both abiotic and biotic influences, including season, feed availability, stress, water quality and sex (Degsera *et al.*, 2020; Thiruvengadam and Akinsorotan, 2021). Tools such as FiSAT are commonly used to derive these relationships and to support studies of fish biology (Morey *et al.*, 2002).

Feed is the largest single cost in aquaculture, accounting for up to 60% of total production costs (FAO, 2022), and protein is the most critical and expensive dietary component. Fishmeal is the conventional protein source, but its high cost and limited, increasingly contested supply have driven the search for cheaper and more sustainable alternatives (Chepkirui *et al.*, 2022). Aquatic macrophytes, including water spinach (*Ipomoea aquatica*), have been investigated as partial fishmeal replacements (Odulate *et al.*, 2013; Ramzy *et al.*, 2019; Manuel *et al.*, 2020; Chepkirui *et al.*, 2022). *Ipomoea aquatica* is a fast-growing semi-aquatic macrophyte with trailing stems up to 3 m long that thrives at temperatures above 23.9 °C and pH 5–7 (Austin, 2007). Its leaves are rich in organic acids, polyphenols and amino acids (Doka *et al.*, 2014) as well as fatty acids (Chepkirui *et al.*, 2021), making it a promising plant-based protein source for feed formulation and for the transition toward more sustainable diets (UN Nutrition, 2021).

Although the length–weight relationship and feeding biology of *O. niloticus* have been studied extensively, the influence of feed quality on the condition factor and length–weight relationship of farmed tilapia remains poorly documented, particularly in Kenya. This study therefore assessed the effects of different dietary proportions of *I. aquatica* on the length–weight relationship and condition factor of *O. niloticus* fingerlings.

Materials and methods

Study area

The study was conducted at the Kegati Aquaculture Centre of the Kenya Marine and Fisheries Research Institute (KMFRI), Kisii, Kenya (00°42′ S, 34°47′ E). The site lies in a highland equatorial climate with a mean annual rainfall of about 1,500 mm, the long and short rains falling between March and June and between September and November, respectively.

Experimental diets and set-up

Ipomoea aquatica was collected from irrigation channels, washed to remove debris and transported to the laboratory, where it was sorted, shade-dried, oven-dried at 60 °C and milled into a fine powder for feed formulation. Five diets containing 0% (control), 5%, 10%, 15% and 20% *I. aquatica* were formulated to 30% crude protein together with other ingredients. Monosex *O. niloticus* fingerlings (2.0 ± 1.0 g) were obtained from the KMFRI Sagana station and acclimatised in 1 m^3 circular tanks for three days before stocking. Fingerlings were stocked at 30 fish per tank in a completely randomized design and fed at 5% of body weight twice daily, at 0900 and 1600 h (EAT), throughout the experiment.

Water quality

Dissolved oxygen, pH, salinity, temperature, ammonia and nitrite were monitored daily following Boyd and Tucker (1992), using a YSI multiparameter meter (Yellow Springs, OH, USA). Water samples for nutrient analysis were collected every 14 days and kept in cool boxes with ice packs during transport to the laboratory, where they were analysed following standard procedures.

Fish growth performance

Fish were sampled monthly, early in the morning (0800 h EAT) to minimise handling stress. Fingerlings were scooped from each tank with a net and placed in a bucket half-filled with water. Total length was measured from the snout to the tip of the caudal fin using a measuring board (0.1 cm precision), and body weight was recorded on a sensitive balance. Fingerlings were returned to their tanks immediately after measurement.

Data analysis

The length–weight relationship was determined from $W = aL^b$ (Le Cren, 1951), linearised as $\log W = \log a + b \log L$, where L is the total length (cm), W is the body weight (g), b is the regression coefficient and a is the intercept. The condition factor was calculated as $K = 100W/L^3$ (Froese, 2006).

Results and discussion

Water quality

Water temperature ranged from 22.52 ± 0.16 °C to 23.26 ± 0.15 °C across diets (Table 1). Mean pH ranged from 7.42 ± 0.04 to 7.68 ± 0.03, and dissolved oxygen from 2.68 ± 0.10 to 3.06 ± 0.12 mg/L. Ammonia ranged from 1.38 ± 0.13 to 2.05 ± 0.28 mg/L and nitrite from 1.68 ± 1.76 to 2.26 ± 1.02 mg/L. With the exception of temperature ($P = 0.003$), the parameters did not differ significantly among treatments, and all values remained within the range suitable for the survival and growth of *O. niloticus* (Boyd, 2004).

Length–weight relationship

The regression coefficient *b* obtained by log-transformation was 2.641, 2.832, 2.532, 2.452 and 2.441 for the 0%, 5%, 10%, 15% and 20% *I. aquatica* diets, respectively (Fig. 1). In every treatment *b* was below 3, indicating negative allometric growth, in which length increases faster than weight (Konoyima *et al.*, 2020). This agrees with earlier reports for tilapia and other species (Ogunola and Onada, 2016; Farrag *et al.*, 2018; Ubong *et al.*, 2023; Barman *et al.*, 2023) and falls within the 2.5–3.0 range reported for many fishes (Pauly and Gayanilo, 1997). Although a value of *b* close to 3 is often expected (Allen, 1996), morphological changes associated with food availability, nutrition, water quality, developmental stage, age and health status can shift it (Froese, 2006). The length–weight relationships were strong, with coefficients of determination (R^2) of 0.8806, 0.9346, 0.9597, 0.9711 and 0.9762 for the 0%, 5%, 10%, 15% and 20% diets, respectively (Fig. 1).

Table 1. Water quality parameters (mean ± standard error) for each dietary inclusion of *I. aquatica* during the experimental period. F and P values are from one-way analysis of variance.

Parameter	0%	5%	10%	15%	20%	F	P
Temperature (°C)	22.59 ± 0.16	23.26 ± 0.15	23.12 ± 0.17	23.11 ± 0.14	22.52 ± 0.16	4.019	0.003
Dissolved oxygen (mg/L)	2.79 ± 0.11	2.80 ± 0.11	2.86 ± 0.10	2.68 ± 0.10	3.06 ± 0.12	1.72	0.14
Salinity (g/L)	0.09 ± 0.00	0.08 ± 0.00	0.09 ± 0.00	0.09 ± 0.00	0.09 ± 0.00	0.87	0.48
pH	7.64 ± 0.04	7.42 ± 0.04	7.59 ± 0.03	7.61 ± 0.03	7.68 ± 0.03	8.20	—
Nitrite (mg/L)	1.74 ± 0.42	1.68 ± 1.76	2.17 ± 0.59	2.26 ± 1.02	2.06 ± 0.45	0.98	0.42
Ammonia (mg/L)	2.01 ± 0.24	1.98 ± 1.25	1.85 ± 1.35	1.38 ± 0.13	2.05 ± 0.28	0.41	0.80

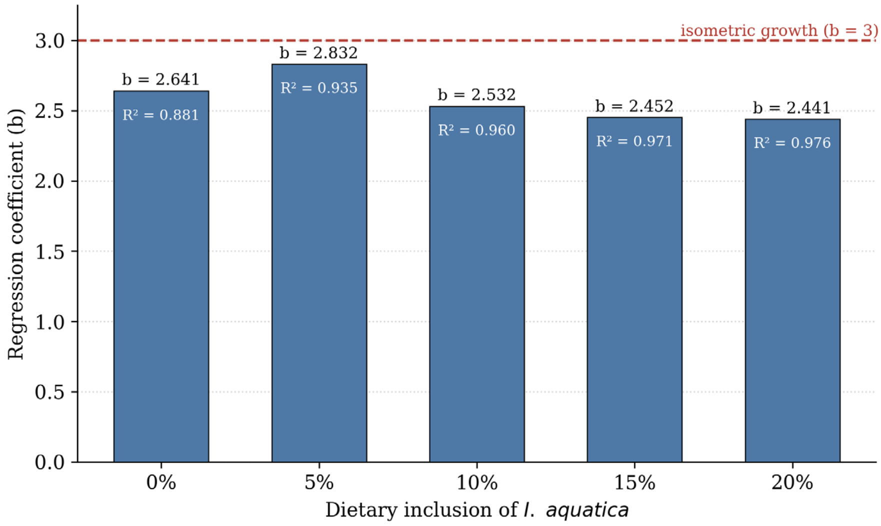

Figure 1. Regression coefficient (*b*) of the length–weight relationship of *O. niloticus* fingerlings fed diets containing 0–20% *I. aquatica*. The dashed line marks isometric growth (b = 3); coefficients of determination (R^2) are shown within each bar.

Condition factor

The condition factor reflects the physical and physiological well-being of fish and responds to feeding, infection, environmental change and physiological state (Ridha, 2011; Momi *et al.*, 2020). Mean K values were 2.36, 2.55, 2.25, 2.09 and 1.93 for the 0%, 5%, 10%, 15% and 20% diets, respectively (Table 2). All values exceeded 1, indicating fish in good condition (Jisr *et al.*, 2018; Abaho *et al.*, 2020), in agreement with earlier studies (Anani and Nunoo, 2016; Farrag *et al.*, 2018; Shija, 2020; Orina *et al.*, 2023). The lowest K, recorded at 20% inclusion, may reflect the higher fibre content of *I. aquatica*, which can form complexes that reduce the digestibility and palatability of the diet (De Silva *et al.*, 2015). Condition factor also tends to decrease as fish length increases (Abowei and Davies, 2009). Such variation in K is known to depend on water quality, diet acceptability, nutrition, season, habitat, sex and maturity (Degsera *et al.*, 2020; Orina *et al.*, 2023).

Table 2. Condition factor (K) of *O. niloticus* fingerlings fed different proportions of *I. aquatica*.

Diet (% *I. aquatica*)	Condition factor (K)
0%	2.36
5%	2.55
10%	2.25
15%	2.09
20%	1.93

Conclusions and recommendations

Dietary inclusion of *I. aquatica* influenced the length–weight relationship and condition factor of *O. niloticus* fingerlings. Although the differences among diets were not statistically significant, fingerlings fed 5% *I. aquatica* showed the best overall performance, and condition factor declined at higher inclusion levels. Water spinach can therefore contribute to lower-cost tilapia feeds and, through this, to fish production and to food and nutrition security in resource-limited communities. On-farm trials in outdoor ponds are recommended to confirm productivity and growth performance under practical conditions. Being affordable, acceptable and locally available, *I. aquatica* has clear potential as a sustainable feed ingredient supporting resilient food systems.

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Habitat Quality and Fish-Based Biotic Integrity Indices of River Kathita and Associated Dams: Implications for Ecosystem Management and Biodiversity Conservation

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Abstract

Anthropogenic activities are increasingly threatening rivers worldwide by altering their biogeochemical processes and compromising their ecological integrity, necessitating robust assessment methods. This study documented biotic integrity, based on fish population dynamics and habitat type in River Kathita and its associated dams, from March to August 2020 at 11 sampling sites. *In-situ* physicochemical parameters were determined using a YSI multiparameter meter, and nutrients were determined using standard methods. Habitat quality was determined based on 9 metrics of Habitat Quality Index. Fish populations were sampled using electrofishing supplemented by gill nets, traps, and long lines where applicable. The collected fish were subsequently identified to species and classified into the following ecological guilds: native, exotic, tolerant, intolerant, and rheophilic. A total of 1,133 individuals and 20 species of fish were sampled and determined for the project. The habitat quality index means score ranges from S1 (21.00 ± 0.73) to S3 (12.17 ± 0.31). The fish-based Index of Biotic Integrity (FIBI) scores from all sites indicate degraded conditions, with FIBI means ranging from S1 (35.33 ±2.72) to S3 (27.00 ±1.13). The results of the HQI and FIBI demonstrated that upper Tana River system and its tributaries are ecologically degraded. These results provide baseline data for continued monitoring and help develop targeted strategies for sustainable ecosystem management and biodiversity conservation

Keywords: Abundance, riparian vegetation, species richness, diversity, substrate type, flow regime

Introduction

The river integrity is achieved when the system's structure and functions are unperturbed and can be defined as its ability to preserve structure and functions via the natural processes and characteristics of its ecoregion. This allows the river to deliver ecological services and operates sustainably (Roseman and Debruyne, 2015; Orina *et al.*, 2018; Li *et al.*, 2021). Initially, assessment methods sorely relied on physical and chemical parameters and have been proven inadequate in determining the ecological condition of surface waters. This resulted to European Water Framework Directive (WFD) recommending incorporation of biological quality elements such as fish to evaluate the ecological health of aquatic environments (Breine *et* *al*., 2018). Assessing the quality and integrity of river ecosystems can be done by applying several approaches. One of them, though costly and time-consuming is analysing physical and chemical parameters to generate a habitat quality index (Raburu and Masese, 2010) while the other technique uses geomorphic features alongside selected biological characteristics that influence habitat structure and energy flow. Lastly, Biomonitoring involves the use of living organisms as indicators of habitat condition and ecosystem integrity. Organisms frequently used for this purpose include macroinvertebrates (Aura, 2008), macrophytes, bryophytes, and fish (Li *et al*., 2010).

According to Karr (1981), the most recommended indicator of ecological integrity across spatial and temporal scales is fish. This is due to its traits such as long lifespan, high mobility and representation at multiple trophic levels. They are reliable makers of biological integrity and ecosystem health as they are conspicuous, sensitive components of freshwater ecosystems. Fish-based indices that combines community structure with physical and chemical data are effective tools for assessing human impacts on aquatic ecosystems as shown by research studies in Europe and Asia. In Kenya, Raburu and Masese (2010) created a Fish Index of Biotic Integrity (FIBI) for the Sondu, Nzoia, and Yala rivers. Similarly, Orina *et al.* (2018) developed a FIBI for River Kuja, which flows into Lake Victoria in south-western Kenya.

Rivers are known to offer diverse ecological and social benefits, including fisheries, water supply, construction materials, habitats, cultural values, recreation, nutrient cycling, soil formation, and overall ecosystem stability (Opperman, 2018). Though these services are very vital for human well-being, they face significant threats from over-exploitation. Thus, the ever-increasing population has intensified pressure on both terrestrial and aquatic resources and in turn resulting to expanded agriculture, deforestation of riparian zones, settlement and river modification. Such human activities contribute to pollution and ecosystem degradation, undermining riverine resources. Understanding the effects of these anthropogenic pressures is therefore crucial. Having a comprehensive knowledge about river ecosystems is vital for crafting effective policies on conservation and sustainable management. The present study evaluated habitat quality and ecosystem integrity in River Kathita and its associated dams through the application of the habitat quality index and the fish index of biotic integrity.

Materials and methods

Study area

River Kathita lies in eastern Kenya between longitudes 037°.56990´E and 038°.00236´E, and latitudes 00°.01329´N and 00°.26667´S. Its basin stretches across altitudes of 472–1982 meters above sea level, covering an estimated catchment area of 1,620 km^2 (Fig. 1).

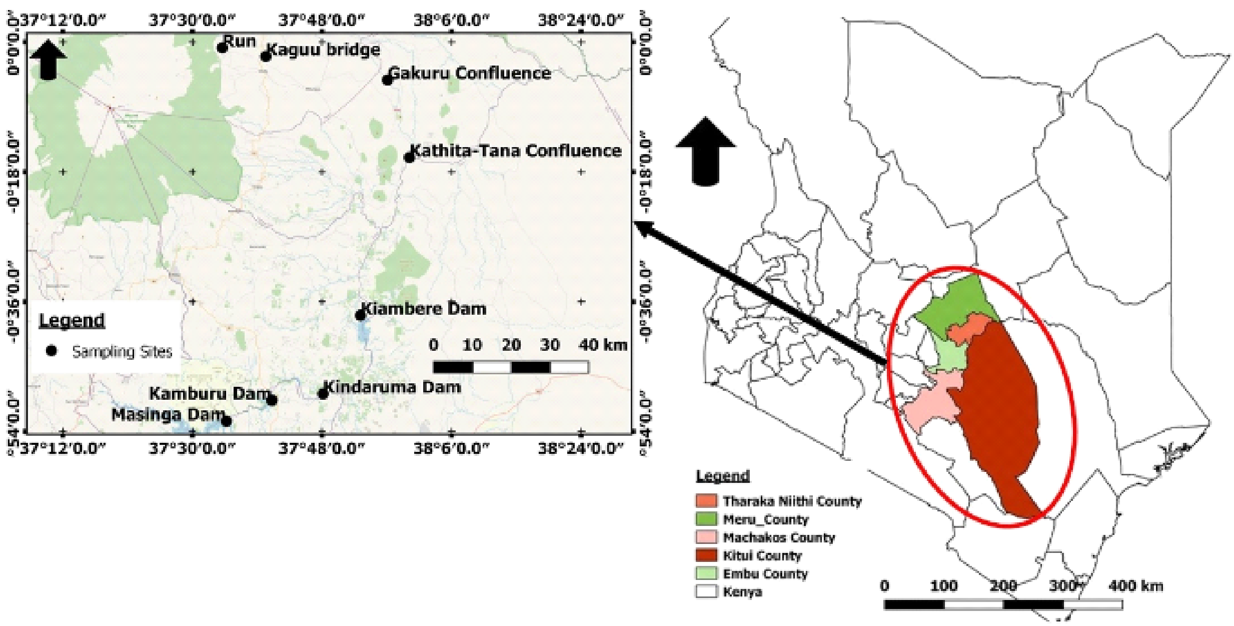

Figure 1. Study area showing sampling sites along Rivers Kathita and Tana, Mt. Kenya region, Kenya.

Sampling design

Sampling was carried out once per month between March and August 2020 to collect data for evaluating the ecosystem integrity of River Kathita and four adjacent dams within the upper Tana basin. A total of eleven sites (S1–S11) were selected, with seven (S1–S7) located along River Kathita and four (S8–S11) situated within the constructed dam stretch of the basin. The dams sampled were represented by sites S8, S9, S10, and S11.

Water quality parameters

The *in-situ* physico-chemical variables were measured using YSI Multi probe model. Water samples for nutrients analysis were collected, preserved in cooler boxes prior transportation to Kenya Marine and Fisheries Research Institute (KMFRI) laboratory for nutrients analysis. Analysis of total phosphorus, and total nitrogen were done in the laboratory following standard procedures (ALPHA, 2014).

Habitat quality index

Habitat characteristics for estimating the Habitat Quality Index (HQI) were also recorded. These comprised available instream cover, bottom substrate, largest pool dimensions, number of riffles, water level, channel sinuosity, bank stability, riparian buffer vegetation, and reach aesthetics.

Fish Index of Biotic Integrity (FIBI)

To evaluate the Fish Index of Biotic Integrity (FIBI), several fish community metrics were assessed: number of native species, number of intolerant species, number of rheophilic species, proportion of benthic individuals, proportion of tolerant individuals, proportion of cyprinids, proportion of detritivores, proportion of carnivores, proportion of omnivores, total abundance per 50 m sampling stretch, number of exotic species, a modified index of well-being, and overall fish distribution and abundance.

Habitat quality assessment

The Habitat Quality Index (HQI) was assessed following the methodology outlined by Rodgers (2016). The scoring system applied to the nine HQI metrics is summarized in Table 1. The total HQI for every station was obtained by summing up the rated scores and then characterized as exceptional, high, intermediate, limited and or minimal integrity index for 26 – 31, 20 – 25, 14 – 19, 8 – 13, < 7, respectively.

Table 1. Habitat characteristics scoring criteria for habitat quality assessment.

Habitat parameter	**Scoring criteria**			
Available instream cover	**Abundant** >50% of substrate that favors colonization and fish cover, virtuous mix of several stable cover types such as snags, wood debris, holes in the bank, macrophytes	**Common** 30 – 50% of substrate supports firm habitat; sufficient habitat for maintenance of populations; may be limited in the number of various habitat types.	**Rare** 10 – 29.9% of substrate ropes firm habitat; habitat accessibility less than desirable; substrate frequently troubled or removed.	**Absent** < 10% of substrate supports firm habitat; absence of habitat is obvious; substrate unstable or absent.
Score	**4**	**3**	**2**	**1**
Bottom substrate	**Stable** >50% gravel or larger substrate; gravel, sarsens; dominant substrate type is grit or larger	**Moderately Stable** 30 – 50% gravel or larger substrate; prevailing substrate type is mix of grit with some finer sediments.	**Moderately Unstable** 10 – 29.9% grit or bigger substrate; prevailing substrate type is finer than gravel but may still be a mix of sizes.	**Unstable** < 10% grit or bigger substrate; substrate is uniform sand, silt, clay, or bedrock.
Score	**4**	**3**	**2**	**1**

Habitat parameter	**Scoring criteria**			
Dimension of largest pool	**Large** Pool occupies more than 50% of the channel width; maximum depth is >1 meter	**Moderate** Pool occupies roughly 50% or slightly less of the channel width; maximum depth is 0.5 – 1 meter	**Small** Pool occupies roughly 25% of the channel width; maximum depth is < 0.5 meter	**Absent** Pools are absent, only shallow auxiliary pockets.
Score	4	3	2	1
Number of Riffles To be reckoned, riffles must extend > 50% the width of the channel and be at least as long as the channel width	**Abundant** >5 riffles	**Common** 2 – 4 riffles	**Rare** 1 riffle	**Absent** No riffles
Score	4	3	2	1
Water level	**High** Water reaches the base of both lower banks; <5% of channel substrate is uncovered.	**Moderate** Water fills > 75% of the channel; or < 25% of channel substrate is uncovered.	**Low** Water fills 25-75% of the existing channel or riffle substrates are mostly exposed.	**No flow** Very little water in the channel and mostly extant in standing pools, or stream is dry.
Score	3	2	1	0
Channel sinuosity	**High** ≥ 2 well-defined bends with deep outside areas -cut banks and shallow inside areas-point bars present	**Moderate** 1 well defined bend or ≥ 3 moderately defined bends existing.	**Low** < 3 moderately defined bends or only poorly defined bends existing.	**None** Straight channel; might be channelized
Score	3	2	1	0
Bank stability	**Stable** Little proof (<10%) of erosion or bank letdown; bank angles average <30º	**Moderately Stable** Some proof (10 – 29.9%) of erosion or bank letdown; small areas of erosion mostly healed over, bank angles average 30 – 39.9º	**Moderately Unstable** Proof of erosion or bank letdown is common (30 – 50%); high possibility of erosion during flooding; bank angles average 40 - 60º	**Unstable** Large and frequent proof (> 50%) of erosion or bank letdown; raw areas recurrent along steep banks; bank angles average > 60º
Score	3	2	1	0
Riparian Buffer Vegetation	**Extensive** Width of natural buffer is greater than 20 meters.	**Wide** Width of natural buffer is 10.1 to 20 meters	**Moderate** Width of natural buffer is 5 to 10 meters	**Narrow** Width of natural buffer is less than 5 meters
Score	3	2	1	0
Aesthetics of Reach	**Wilderness** Exceptional natural beauty; usually wooded or unpastured area; no apparent indications of anthropogenic activity.	**Natural Area** Trees or native flora is common; some development evident - from fields, pastures, natural dwellings little proof of human activity	**Common Setting** Not offensive; area is developed but uncluttered such as in an urban park.	**Offensive** Stream does not augment the aesthetics of the area; cluttered; highly developed; may be a dumping area
Score	3	2	1	0

Fish Index of Biotic Integrity (FIBI)

The Fish Index of Biotic Integrity (FIBI) was computed using three major metric categories, comprising a total of twelve sub-metrics categorized into species richness and composition trophic guilds and abundance and condition metrics (Table 2).

Table 2. Fish Index of Biotic Integrity (FIBI) metrics and its scoring criteria.

Category	Metric	Scoring criteria		
		1 (worst)	3 (better)	5 (best)
Species richness and composition	Number of native species	< 3	3 - 5	≥ 6
	Number of intolerant species	< 3	3 - 5	≥6
	Number of rheophilic species	0	1	>1
	Percentage of benthic species	< 7.5	7.5 – 15	>15
	Percentage of tolerant individual	>20	10 – 20	< 10
	Percentage of Cyprinidae individual	< 40	40 - 80	>80
Trophic metric	Percentage of detritivores individual	< 7.5	7.5 – 15	≥ 15
	Percentage of carnivores individual	< 1	1 – 4.4	≥ 4.5
	Percentage of carnivores individual	>45	20 – 45	< 20
Abundance and condition	Number of individuals per 50 m of sampling	< 25	25 – 50	> 50
	Number of exotic species	≥ 2	1	0
	Modified index of well-being	< 1.25	1.25– 2.50	≥ 2.50

Individual species percentage proportion was estimated using the formula.

$$\% = \frac{\text{Number of individual fish population}}{\text{Total number of fish in a community}} \times 100$$

Data analysis

Data on physical-chemical parameters, habitat quality and fish indices was entered in Microsoft excel sheet where means, standard error, minimum and maximum value were obtained from descriptive statistics and presented in tables and graphs. The means were subjected to Analysis of Variance (2-way ANOVA) using SPSS software version 22 to undertake post hoc analyses where there were significant differences of all variables evaluated (p ⊠ 0.05). Tukey test was carried out in cases where there were significant differences in order to obtain the actual pair of stations and months with significant differences. The data was presented using tables and graphs.

Results and discussion

Habitat quality index variation

The Habitat Quality Index (HQI), as illustrated in Figure 2 and Figure 3, showed a distinct pattern of substantial decrease in HQI scores moving downstream through the river system ($F(6,119) = 46.45$, $P < 0.001$). Again, the upstream stations, including S1 and S2, significantly outperformed the downstream stations in terms of HQI scores that were consistently lower at S4, S5, and S6 than at S7, S2, and S1, indicating minimal habitat degradation from upstream to downstream. The HQI scores at S7 were, however, higher than those recorded at S3; however, both stations had significantly lower scores compared with S1. Therefore, whilst S1 had the highest HQI score of all stations, all of the downstream stations had lower scores reflecting deteriorating ecological conditions where habitat quality decreased with increasing distance from the major habitat source. Seasonal effects were also evident, reflected by the variation in mean HQI scores between months; for example, the annual minimum mean HQI score of 14.14 ± 0.94 occurred in May 2020, whilst the annual maximum score occurred in April 2020 with a mean score of 17.43 ± 1.59, which confirm the potential influence of season on habitat quality.

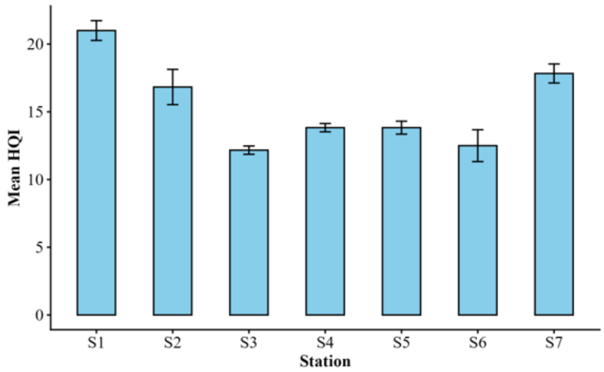

Figure 2. Mean Habitat Quality Index (HQI) values at different sampling stations

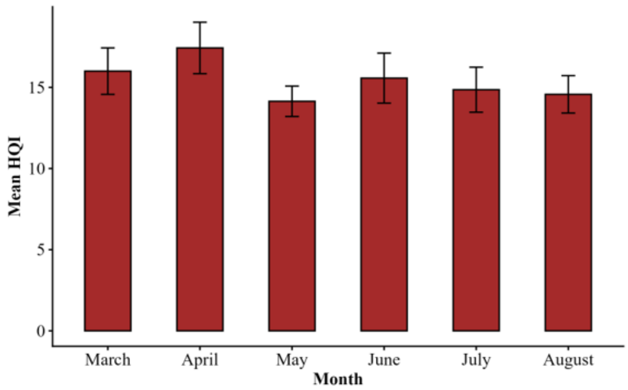

Figure 3. Mean Habitat Quality Index (HQI) in different sampling months.

Fish-based index of biotic integrity (FIBI)

The Fish Index of Biotic Integrity (FIBI) shows a large variation among sampling sites, mean values, from 27.00 ± 1.13 at site S10 (the Dam) to 35.33 ± 2.72 at site S7, with an overall mean of 29.89 ± 0.82. The results of the analysis of variance confirm that there are significant differences between sites ($F_{(7,136)} = 7.56$, $P < 0.001$). Specifically, sites S6 and S7 have consistently shown higher FIBI values than several sites downstream of them (S8, S9, S10, and S11), indicating that they have better ecological conditions than the downstream sites. Site S4 had significantly lower values than sites S6 and S7, indicating that the biotic integrity at site S4 is lower than that of sites S6 and S7. The overall pattern indicates that the fish community structure in the sampling sites is spatially variable. Some sites in the middle of the stream appear to perform better than the downstream and dam-associated sites. Temporal variation is also seen in the mean FIBI values ranging from 29.00 ± 1.98 in April 2020 to 31.33 ± 3.41 in June 2020, indicating that seasons affect the fish-based ecological integrity of the stream (Figure 4 and 5).

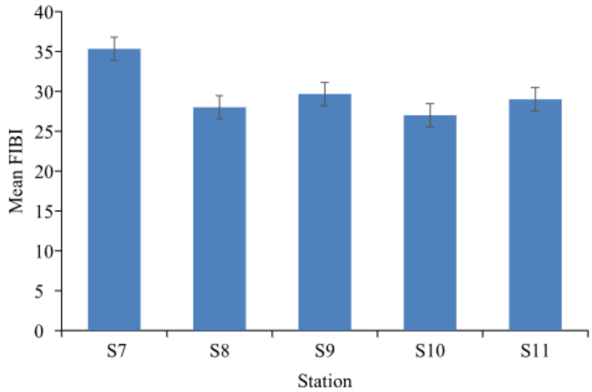

Figure 4. Mean Fish Index of Biotic Integrity (FIBI) at different sampling stations.

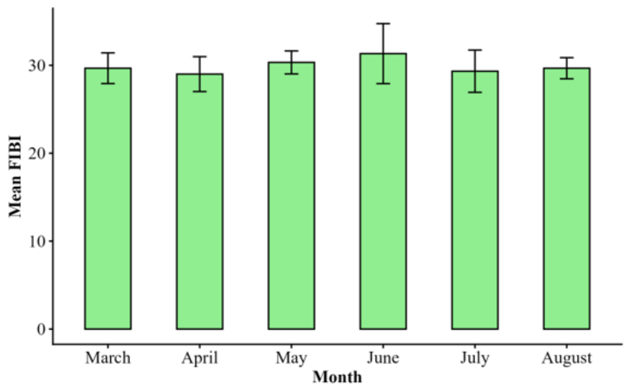

Figure 5. Mean Fish Index of Biotic Integrity (FIBI) in different sampling months.

Discussion

Habitat quality characteristics

Based on the projected Habitat Quality Index (HQI) in this study, six out of seven stations were rated below "good." The observed significant variation in HQI likely reflects changes in the physical attributes of the riverine ecosystem. Raven *et al*. (1998) noted that changes within and between catchments can be driven by agriculture, roads, urbanization, mining, and other anthropogenic activities.

Stations such as S4, S5, and S6 recorded lower integrity indices, which can be attributed to agricultural practices, livestock watering and water abstraction. The riparian vegetation was minimal or absent at some sites thus reducing habitat quality. Riparian vegetation plays a vital role in stabilizing riverbanks, slowing wind speed, and contributing organic matter to the river system. Its reduction downstream can lead to bank erosion, channel changes, sediment-covered substrates and overall habitat impairment (USEPA, 2000). Similar patterns have been reported by Raburu and Masese (2010) and M'Erimba *et al*. (2014) in the Nzoia, Sondu-Miriu, Nyando rivers, and streams in the Mount Kenya region. The significant spatial variation in HQI downstream was largely driven by these anthropogenic activities. Analysis grouped stations such as S5, S7, S3, and S4, as agriculture was the predominant activity around these sampling sites. These findings line up with observations from other Kenyan rivers, including Kuja, Nzoia, Sondu-Miriu, Nyando, and various streams elsewhere.

Comparison of HQI in this study and that of other streams

Habitat Quality Index values in this study were converted to percentages and then results compared to other reported similar studies. The range was between 32.26 – 74.2% showing that the habitat is severely to partially degraded. According to Orina *et al*. (2018), River Kuja (Kegati to River mouth as from November 2016 to August 2017) HQI values ranged from 35.5 - 77.4% similar to the case above. Raburu and Masese (2010) for Rivers Nzoia, Nyando and Sondu-Miriu also reported that Lake Victoria HQI values ranged from 22 – 60.5 translating to severely degraded to degraded. Similarly, Bio-habitat and Century Engineering, 2016 in Anne Arundel county (Magothy, Seven and Salamanders – 2015) got a range of 61.31 – 98.01 indicating it is degraded to minimally degraded sites. Another study by Paul *et al*. (2003) in Maryland wadable streams (piedmont class, coastal plain and Highland class from 1994 – 2000) showed HQI values ranging from 15.43 – 99.35 showing the sites were from severely to minimally degraded. Lastly, Diana, Allan and Diana, 2006 in Southern Michigan (Huron and Raisin basin from 1999 –2000) got HQI values ranging from 33.3 – 79.3 meaning the habitat was severely to partially degraded.

Temporal variation of Habitat Quality Index (HQI)

Over the sampling period, the Habitat Quality Index (HQI) remained relatively stable. A slight increase was observed during the first three months, likely due to the growth of autotrophic organisms coinciding with the rainy season. After this brief rise, the index stabilized over the short monitoring period. Despite this overall stability, the most disturbed sites were readily identifiable and measurable. Similar findings have been reported by Johnson *et al*. (2001) and Roper *et al*. (2002), who noted that short-term sampling can reveal a relatively stable integrity index.

Fish-based Index of Biotic Integrity (FIBI)

The Fish Index of Biotic Integrity (FIBI) indicated that habitat integrity at all six sampled stations was below "good." Two of these stations along River Kathita were also classified as poor habitats based on HQI. A clear trend emerged showing that as land use intensity increased, FIBI values decreased. This was particularly evident at S5 and S10 dam, which recorded the lowest FIBI values within their respective categories.

These findings are consistent with Raburu and Masese (2010), who observed that land use activities significantly influence the integrity of rivers such as Nyando, Sondu-Miriu, and Nzoia within the Lake Victoria basin. Key factors contributing to reduced river integrity include intensive agriculture, urbanization, high population densities, and mining activities. The FIBI values obtained in this study were comparable to those reported in similar ecosystems, likely reflecting analogous land use pressures. These studies included that of Orina *et al*. (2018) for River Kuja (Kegati to River mouth as from November 2016 to August 2017) that obtained FIBI range of 2 – 3.8 that implied that the river integrity is very poor to fair. Raburu and Masese (2010) (R. Nzoia, Nyando and Sondu-Miriu in Feb, March and July 2004) also reported that Lake Victoria FIBI values ranged from 1.7 – 4.6 translating to very poor to good.

Relatively few studies have focused on developing habitat quality and fish-based assessment tools within the Lake Victoria basin and in Kenya. Data on habitat integrity are essential for formulating regulations aimed at restoring and conserving degraded ecosystems. Maintaining and restoring a river's ecological integrity is critical for ensuring the continued provision of sufficient natural resources. According to Sara and Kagan (2014) and UNEP (2016), pristine aquatic ecosystems offer sustainable intrinsic values and benefits to human communities.

Anthropogenic activities, however, can significantly alter biological conditions in rivers by affecting flow regimes, water quality, physical habitat structure, and species interactions (Korsgaard, 2006; USEPA, 2016). Changes in natural vegetation and soil conditions have also been documented (Mutie *et al*., 2006; Stohlgren, 2008), resulting in increased water temperatures, erosion, siltation, and overall habitat degradation (Stevens and Council, 2008; Mbaka, 2010). These alterations negatively impact fish communities due to reduced water quality (Okungu and Opango, 2005; Wetland Consulting Service Ltd, 2014), the use of prohibited fishing gears, and overfishing (Cadwalladr, 1965; Ogutu-Ohwayo, 1990; Mboya *et al*., 2005; Njiru *et al*., 2006; Njiru *et al*., 2008). Raburu, Masese and Aura (2009) further noted that habitat modifications can severely depurate the condition of aquatic environments.

Evidence of these effects was observed in the sampled dams along River Tana, where some fish species were present in certain dams but absent downstream. For example, species found in S11 (Masinga) were missing in lower dams, likely due to migration barriers that limit fish dispersal and reduce diversity and abundance downstream. Previous studies have also highlighted the ecological and economic impacts of introduced species in the Lake Victoria Basin (LVEMP, 1995; Balirwa *et al*., 2003; Aloo *et al*, 2013). Miller and Williams (1989) further noted that non-native fish introductions can contribute to species extinctions. These findings underscore the vulnerability of riverine biota, particularly in the presence of introduced species such as *Clarias gariepinus*, combined with other human-induced pressures.

Conclusion and recommendations

River Kathita supports a diverse fish community, with *Labeobarbus* species being the most abundant. The river serves as a significant source of food for communities along its entire course. However, the habitat is currently degraded and requires urgent restoration to enhance its ecological productivity. Implementing an Ecosystem-Based Management (EBM) approach offers a viable pathway to improve and maintain the ecological integrity of River Kathita.

The findings from this study provide a baseline for monitoring changes in the ecosystem integrity of River Kathita and its associated dams. This information is critical for guiding conservation, restoration, and sustainable management efforts.

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Determinants of Fish Prices and Value Chain Inefficiencies in Kenya: An Econometric Analysis

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Abstract

Fish markets in Western Kenya are vital for food security, nutrition, and livelihoods but face structural inefficiencies that affect trader welfare and price formation. This study analysed trader characteristics, market operations, value chain structure, constraints, and determinants of fish prices. Primary data were collected from 80 fish traders across five counties, generating 97 price observations. Descriptive statistics and a log-linear multiple regression model with HC3 heteroscedasticity-robust standard errors were applied. Results show that fish trading is dominated by female traders (72.5%) and sole proprietorships (85%), with a mean trading experience of 14.6 years. Mean fish price was KES 272.43/kg, varying across markets, species, and processing forms. Econometric results indicate that frying generated a significant 23.2% price premium ($p = 0.0206$), male traders received prices 78.2% higher than female traders ($p = 0.0137$), and fish in Siaya County traded at a 21.5% discount relative to Homabay ($p = 0.0447$). The findings suggest that prices are shaped mainly by value chain structure, processing practices, and spatial factors. Policy should prioritise value addition, gender equity, and improved market infrastructure.

Keywords: Fish prices; market participation; value chain; log-linear regression; post-harvest loss; gender; Western Kenya

Introduction

The fisheries sector plays a vital role in global food systems, contributing significantly to food security, nutrition, and livelihoods, particularly in developing countries. Globally, fish provides a critical source of high-quality animal protein and essential micronutrients for billions of people, while supporting employment across capture fisheries, aquaculture, processing, and trade value chains (FAO, 2020). In many low- and middle-income countries, fish markets are characterised by informal trading systems, fragmented supply chains, and limited infrastructure, conditions that influence price formation and market efficiency. Understanding how these structural factors interact with product characteristics and spatial dynamics is central to improving the performance of fisheries-dependent economies.

Sub-Saharan Africa's fisheries sector is dominated by small-scale fisheries and informal markets, where post-harvest handling, transport logistics, and market access constraints significantly shape economic outcomes. Post-harvest losses remain substantial, estimated at 20 - 30% of landed catch, largely due to inadequate cold-chain systems, poor infrastructure, and limited processing capacity (Akande and Diei-Ouadi, 2010). These inefficiencies contribute to price volatility, reduce incomes for value chain actors, and limit the availability of affordable fish to consumers. Empirical studies across African markets show that spatial fragmentation, weak market integration, and high transaction costs often lead to persistent price disparities across locations, even where supply sources are geographically proximate (Barrett, 1996; Minten and Kyle, 1999; Abdulai, 2000; Mkenda and Van Campenhout, 2011).

In Kenya, the fisheries sector occupies a critical position within the national food system, contributing approximately 0.5% of GDP and supporting direct and indirect livelihoods for an estimated two million people along the value chain (Abila, 2003; GOK, 2022). Inland fisheries - concentrated primarily around Lake Victoria, Lake Turkana, and associated river systems - provide essential nutritional security for millions of low-income households for whom fish represents a principal and affordable source of animal protein (FAO, 2020; Wills *et al*., 2021). Nile tilapia (*Oreochromis niloticus*), Nile perch (*Lates niloticus*), and small pelagic species such as Omena (*Rastrineobola argentea*) dominate production and trade, particularly in Western Kenya (Bokea and Ikiara, 2000).

Within Kenya's inland fisheries, the Lake Victoria Basin represents the most commercially significant fish production and marketing system, supporting dense networks of landing sites, traders, processors, and urban markets. The basin's proximity to major population centres and cross-border trade routes has fostered relatively active fish markets; however, these markets remain largely informal and characterised by fragmented value chains, multiple intermediaries, and uneven infrastructure development. Variations in road quality, access to cold storage, and proximity to landing sites create spatial differences in transaction costs and market access, which in turn influence fish price formation across locations.

Despite its socio-economic importance, Kenya's fisheries sub-sector continues to operate below its productive and commercial potential. Structural inefficiencies along the value chain - including inadequate cold-chain infrastructure, fragmented market linkages, limited value addition, and information asymmetries between buyers and sellers - erode trader margins, inflate consumer prices, and weaken the sector's overall contribution to food security (Wills *et al*., 2021; Musembi *et al*., 2021). These inefficiencies are reflected in substantial price variability across markets, species, processing forms, and seasons, imposing welfare costs on both traders and consumers (Minten and Kyle, 1999). The persistence of such inefficiencies suggests that improving market performance requires a deeper understanding of the mechanisms underlying price formation.

Price formation in fish markets is inherently multidimensional, shaped by a complex interaction of product characteristics, market structure, processing and preservation methods, geographic conditions, transaction costs, and trader-specific attributes (Rosen, 1974; Tirole, 1988). While a growing body of literature has examined spatial market integration and price transmission in sub-Saharan Africa (Barrett, 1996; Abdulai, 2000; Mkenda and Van Campenhout, 2011), there remains limited empirical evidence on the micro-level determinants of retail fish prices in Kenya, particularly using multivariate econometric approaches that integrate value chain variables. Existing studies have largely focused on farmgate pricing or single-market analyses (Abila, 2003; Kapute *et al*., 2018; Osewe *et al*., 2019), leaving a gap in understanding how value chain structure and trader characteristics jointly influence market-level price outcomes.

This study addresses this gap by integrating descriptive value chain analysis with econometric modelling to examine the determinants of fish prices across selected markets in Western Kenya within the Lake Victoria basin. Using data from a structured survey of fish traders, the analysis captures variations in product attributes, market structures, spatial conditions, and trader characteristics to explain observed price patterns. By linking value chain inefficiencies to price outcomes, the study contributes to ongoing efforts to promote data-driven fisheries development and improve market efficiency within Kenya's Blue Economy framework.

Specifically, the study aims to: (i) characterise the socio-demographic profile of fish traders; (ii) document the structure of fish business operations and value chain linkages; (iii) assess market costs and operational constraints; (iv) analyse fish price dynamics using descriptive statistics; and (v) identify the econometric determinants of fish prices.

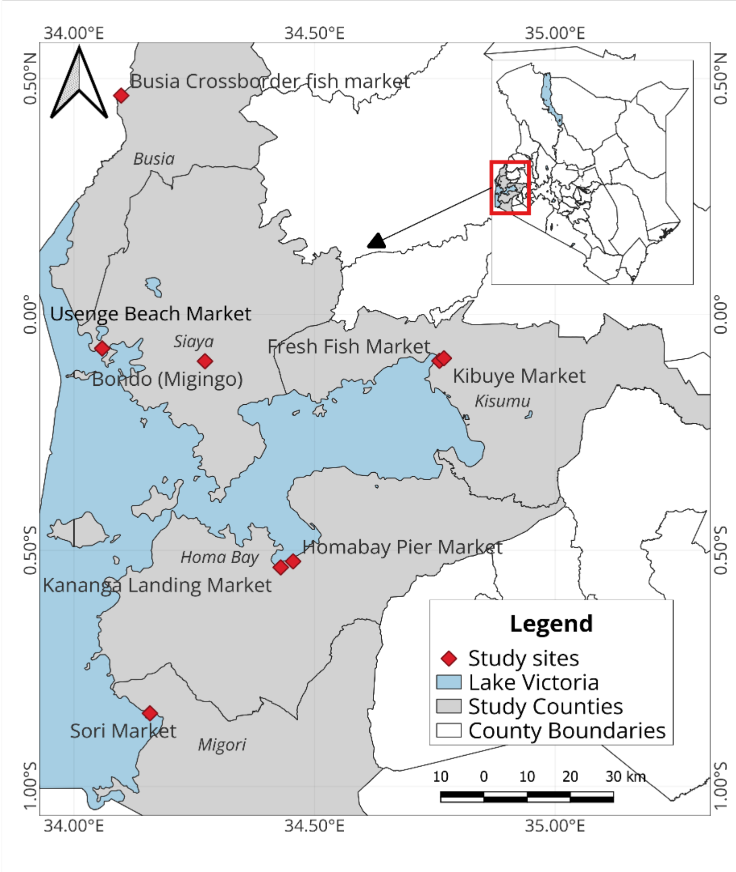

Figure 1. Study Area and Sampled Fish Markets in the Lake Victoria Basin, Western Kenya *(Source: Authors)*.

Materials and methods

Study area

The study was conducted in five counties in Western Kenya - Homabay, Siaya, Kisumu, Migori, and Busia - riparian to Lake Victoria and forming the core of Kenya's freshwater fish production and marketing corridor (Fig. 1**)**. Data were collected from key fish markets including Bondo (Migingo), Busia Crossborder fish market, Fresh Fish Market, Homabay Pier Market, Kananga Landing Market, Kibuye Market, Sori Market and Usenge Beach Market. The study area comprises three market typologies: primary landing-site markets co-located with Beach Management Units (BMUs), secondary urban markets in county headquarters, and border markets along the Busia cross-border corridor with Uganda. Each typology exhibits distinct supply chain structures, competitive conditions, and consumer profiles relevant to fish price formation.

Sampling and data collection

A structured questionnaire survey was administered using KoboCollect to 80 fish traders selected through purposive and systematic random sampling across identified fish markets in the study area (Homabay n=25, Siaya n=24, Kisumu n=11, Migori n=11, Busia n=9), consistent with its demonstrated reliability for real-time, geo-referenced field data collection in fisheries research (Aura *et al.*, 2025). The instrument collected data on: trader socio-demographics; business ownership and function; species, supply sources, and processing forms; market channels, payment modes, and trading frequency; monthly operating costs; market constraints; and fish prices by species and season. Price data were collected via a linked price module in which traders reported the price per kilogram for each species during high and low supply seasons.

Analytical framework

The analysis was conducted in two stages. First, descriptive statistics - frequencies, cross-tabulations, and summary measures - were used to characterise trader profiles, business operations, cost structures, and price distributions. Second, a log-linear multiple regression model was estimated to identify the multivariate determinants of fish prices. The log-linear specification was preferred to the linear OLS model on theoretical grounds (hedonic pricing; Rosen, 1974) and was empirically supported by the Ramsey RESET test ($F = 16.22$, $p < 0.001$), which rejects the linear functional form.

Model specification

The empirical model estimated was:

$$\ln(Price_i) = \beta_0 + \beta_1 Species_i + \beta_2 MarketType_i + \beta_3 Form_i + \beta_4 County_i + \beta_5 Trader_i + \beta_6 \ln(Transport_i) + \varepsilon_i$$

where subscript i indexes observations ($n = 97$). Species is a vector of species dummies (ref: tilapia); MarketType captures market structure (ref: primary/local); Form captures processing form (ref: fresh/whole); County is a vector of geographic fixed effects (ref: Homabay); Trader includes gender (ref: female) and market function (ref: retailer); and ln(Transport) is the log of monthly transport cost (KES). OLS is used. Given heteroscedasticity confirmed by the Breusch-Pagan test ($LM = 32.75$, $p = 0.005$), all standard errors use the HC3 estimator (MacKinnon and White, 1985).

Variable definitions and a-priori expectations

Table 1. Variable definitions and a-priori expected effects on fish price. KES = Kenyan Shillings. Expected signs derived from theoretical priors and prior empirical literature.

Variable	Definition	Type	Expected Sign	Reference
ln(Price)	ln(average price, KES/kg)	Continuous	-	Dependent var.
Nile Perch	=1 if Nile perch; ref: tilapia	Dummy	+	Abila (2003); Osewe *et al.* (2019)
Catfish	=1 if catfish; ref: tilapia	Dummy	+	Osewe *et al.* (2019)
Other Species	=1 if Omena/Dagaa; ref: tilapia	Dummy	-	Musembi *et al.* (2021)
Secondary Market	=1 if town market; ref: primary	Dummy	+	Minten & Kyle (1999)
Border Market	=1 if border market; ref: primary	Dummy	-	Wills *et al.* (2021)
Fried	=1 if fried fish; ref: fresh/whole	Dummy	+	Kapute *et al.* (2018); Rosen (1974)
Smoked	=1 if smoked fish; ref: fresh/whole	Dummy	+/-	Akande & Diei-Ouadi (2010)
Siaya County	=1 if Siaya; ref: Homabay	Dummy	-	Fafchamps & Gabre-Madhin (2006)
Kisumu County	=1 if Kisumu; ref: Homabay	Dummy	+	Osewe *et al.* (2019)
Wholesaler	=1 if wholesaler; ref: retailer	Dummy	-	Tirole (1988); Reardon *et al.* (2009)
Male Trader	=1 if male; ref: female	Dummy	+	Njuki & Sanginga (2013); Fischer & Qaim (2012)
ln(Transport)	ln(monthly transport cost, KES)	Continuous	+	Minten & Kyle (1999); Gollin & Rogerson (2014)

Results and discussion

Socio-demographic profile of fish traders

The socio-demographic profile of fish traders provides essential context for understanding market behaviour, pricing dynamics, and policy targeting. Figure 2 and Table 2 summarise findings across gender, age, experience, geographic distribution, and business ownership.

Gender composition and market roles

Fish trading in the study area is overwhelmingly female dominated: 58 of the 80 sampled traders (72.5%) were female (Fig. 2a; Table 2). This pattern of female dominance in fish retail is consistent with the broad literature on sub-Saharan African fish value chains, where women concentrate in downstream retailing and processing roles while men occupy the harvesting stage (Teh *et al*., 2011; Sanginga, 2013; FAO, 2020; Njuki). Diei-Ouadi and Mgawe (2011) note that women's involvement in post-harvest fish handling and trade across East Africa reflects both cultural norms assigning women responsibility for food marketing and the higher capital requirements of capture fishing that restrict female participation. The feminisation of fish retail is not merely a demographic observation, it has direct implications for the gender price gap that the regression analysis identifies as a statistically significant price determinant (Quisumbing and Pandolfelli, 2010; Fischer and Qaim, 2012).

Age Structure and labour dynamics

Trader age ranged from 22 to 75 years, with a mean of 42.7 years (SD = 11.5; median = 42 years), as shown in Figure 2b. Approximately 60% of traders fell within the 30 - 49-year bracket, indicating fish trading is predominantly an activity of working-age adults with significant productive capacity. Approximately 15% were aged 50 years or above, reflecting long-term occupational attachment. This age profile is broadly consistent with findings by Osewe *et al*. (2019) for fish farming households in Western Kenya and Abila (2003) for Nile perch traders around Lake Victoria. The relatively young median age suggests that many traders are likely to be receptive to productivity-enhancing interventions, including digital technology adoption, skills training, and cold-chain investment. This supports the calls for targeted capacity building among active market participants (Diei-Ouadi and Mgawe, 2011; Republic of Kenya, 2022).

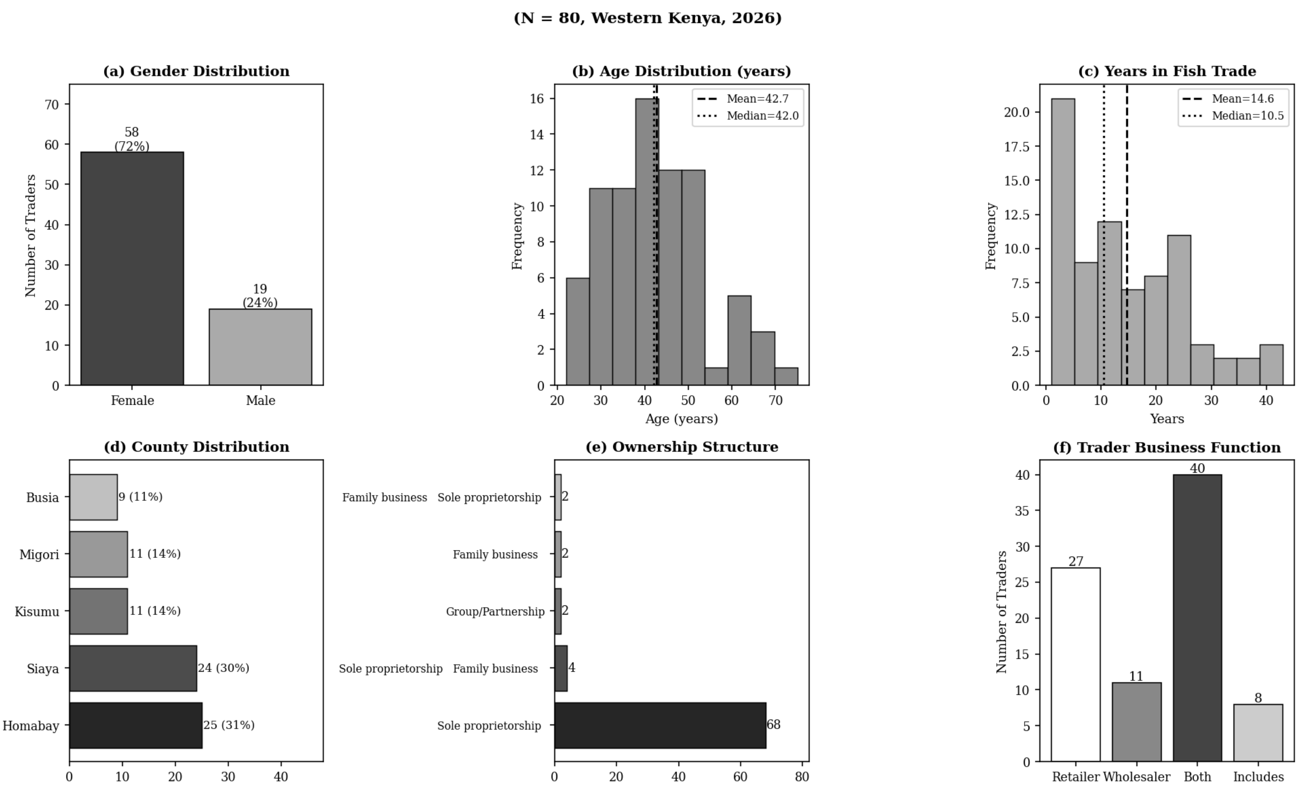

Figure 2. Socio-demographic profile of fish traders (n = 80).

Experience and business tenure

Traders report a mean of 14.6 years in the fish trade (SD = 10.7; median = 10.5 years; Fig. 2c), reflecting deep occupational embeddedness. Approximately 25% of traders had been in the trade for 23 or more years, while another quarter had five or fewer years - pointing to a bimodal cohort combining experienced incumbents and newer entrants. Each trader directly supports a mean of 9 dependents, underscoring the welfare significance of fish retailing beyond the individual trader (Wills *et al*., 2021). Consistent with findings by Fafchamps and Gabre-Madhin (2006) for West African commodity market traders, the regression analysis (Section 4.5) finds that experience does not significantly influence the prices obtained, suggesting that in competitive fish markets, product and market characteristics dominate price formation over trader vintage. This does not diminish the role of experience in market access and supply chain positioning (Makhura *et al*., 2001; Reardon *et al*., 2009).

Table 2. Socio-Demographic characteristics of fish traders in western Kenya (n = 80). Two obs. excluded (missing age/years data). Retailer/wholesaler shares exceed 100% due to dual-role responses.

Variable	n	Mean / %	Std. Dev.	Min	Max
A. Continuous characteristics					
Age (years)	78	42.7	11.5	22	75
Years in fish trade	78	14.6	10.7	1	43
Number of dependents	78	9	6.4	3	35
Number employed	78	2.3	8.3	0	70
B. Categorical (% of traders)					
Female trader	77	72.5%	-	-	-
Sole proprietorship	80	85%	-	-	-
Retailer (incl. dual-role)	80	83.8%	-	-	-
Wholesaler (incl. dual-role)	80	63.7%	-	-	-

Geographic distribution of traders

Homabay County accounts for the largest share of traders that were sampled (25, 31%), followed by Siaya (24, 30%), Kisumu (11, 14%), Migori (11, 14%), and Busia (9, 11%) (Fig. 2d). This distribution reflects the concentration of fish landing infrastructure along the southern shores of Lake Victoria and the Busia border crossing. Preliminary county-level mean prices of Migori (KES 362.5/kg), Homabay (KES 267.8/kg), Kisumu (KES undefined/kg), Siaya (KES 266.5/kg), reveal geographic variation that the regression analysis disaggregates into statistically significant and insignificant components in Section 4.5 (Ravallion, 1986; Barrett, 1996).

Business ownership and functional roles

Sole proprietorship was the dominant ownership structure, accounting for 85% of traders (Fig. 2e). This pattern reflects the informal, micro-enterprise nature of fish retailing in African markets, as documented by Njuki and Sanginga (2013) and Reardon *et al*. (2009). Such ownership arrangements may limit access to credit, reduce opportunities for risk sharing, and constrain investment capacity, particularly among female traders (FAO, 2020; Doss, 2013). Nearly half of the traders (48.8%) operated as both wholesalers and retailers, indicating partial vertical integration within the retail segment of the value chain (Fig. 2f). Dedicated retailers accounted for 33.8%, while pure wholesalers represented 13.8%. This overlap between wholesale and retail functions, also observed in West African fish markets by Fafchamps and Gabre-Madhin (2006), helps explain why the wholesaler coefficient in the regression was not statistically significant (Section 4.5.3e).

Fish business characteristics and market operations

This section characterises the value chain structure of fish trading enterprises, covering capitalisation, species mix, supply sources, processing forms, customer segments, and trading operations (Fig. 3). These characteristics collectively shape the conditions under which prices are formed and several serve as explanatory variables in the econometric model.

Capitalisation and financial structure

Personal savings constitute the primary start-up capital source for 66.2% of traders, followed by microfinance loans (25%), family support (8.8%), and grants (1.3%) (Fig. 3a). The heavy dependence on own savings mirrors findings by Osewe *et al*. (2019) for Western Kenya fish farmers and reflects the limited penetration of formal financial services among small-scale fish traders - a constraint noted across sub-Saharan African agricultural and fish markets (Fafchamps and Gabre-Madhin, 2006; Reardon *et al*., 2009). The low uptake of available microfinance likely reflects high effective interest rates, short repayment cycles, and collateral requirements misaligned with irregular fish trade cash flows (Jack and Suri, 2011; Doss, 2013). This financial constraint limits capital-intensive investments in cold chain infrastructure and volume scaling, perpetuating the small-volume, high-frequency trading cycles typical of informal fish markets (Wills *et al*., 2021).

Species composition and commodity mix

Nile tilapia is the most widely traded species (46.2% of traders), followed by Nile perch (33.8%) and other species including Omena and Dagaa (38.8%) (Fig. 3b). Multiple-species trading is prevalent, reflecting strategies to hedge against species-specific supply seasonality and serve heterogeneous consumer preferences. This is a diversification behaviour consistent with informal food market behaviour in developing economies (Bokea and Ikiara, 2000; Fafchamps and Gabre-Madhin, 2006; Brummett *et al*., 2013). Raw mean prices including tilapia KES 256.6/kg; Nile perch KES 311.2/kg; other species KES 246.5/kg, suggest species differentials that the regression analysis subsequently tests in a multivariate framework (Section 4.5.3a).

Wild-caught lake fish is the primary supply source for 92.5% of traders, with cage-reared aquaculture contributing an emerging share (10.0%; Fig. 3c). A subset of traders (7.5%) source through intermediary fish dealers, reflecting multi-layered intermediation in the supply chain consistent with the intermediary structures doc-

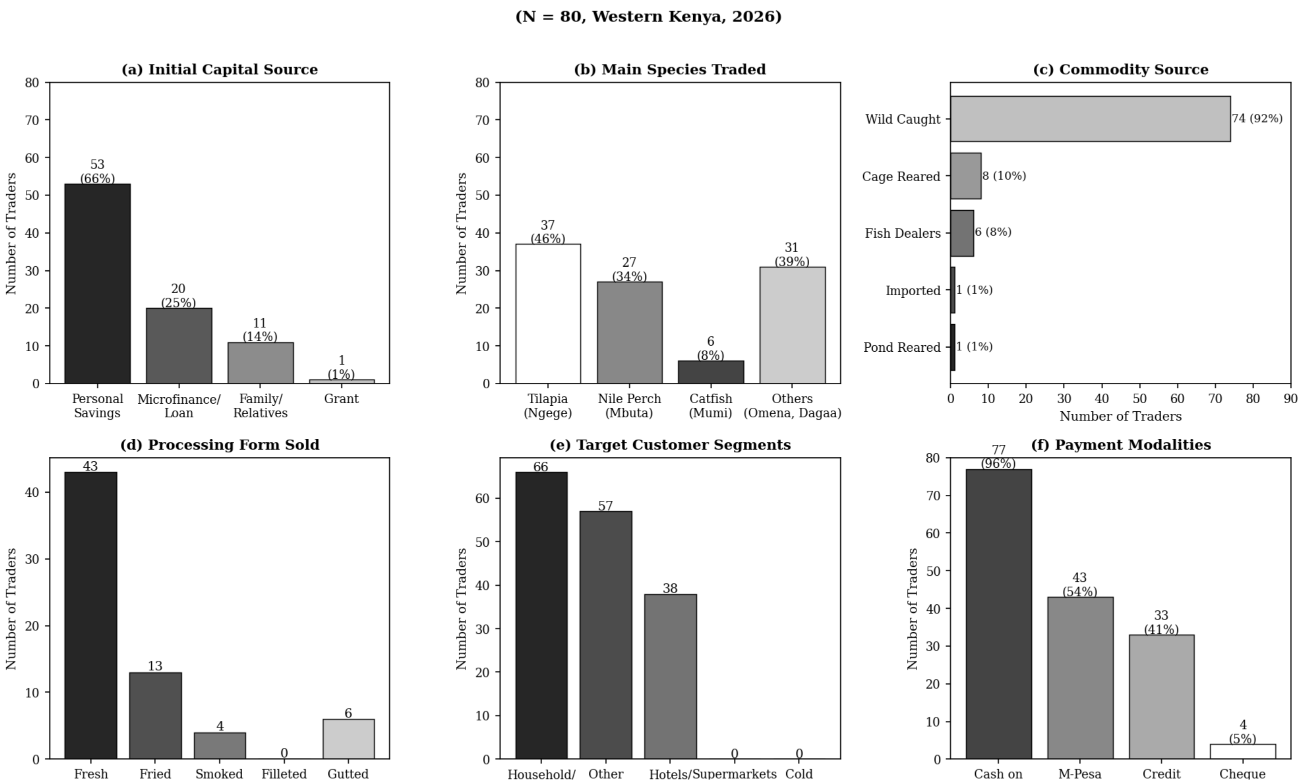

Figure 3. Fish business profile and market characteristics (n = 80).

umented in East African fish markets by Abila (2003) and Wills *et al.* (2021). The dominance of wild-catch supply renders traders highly exposed to seasonal fishing effort variability - the most frequently cited constraint (Section 4.3). The low aquaculture share, despite Kenya's active cage culture development agenda, confirms that aquaculture has yet to substantially penetrate informal retail supply chains (Brummett *et al.*, 2013; GOK, 2022).

Fish is predominantly sold fresh and whole (53.8%; Fig. 3d), with fried fish being the most commercially significant processed form (16.2%) and smoked fish a minor category (5%). The limited prevalence of processed forms reflects constrained value-addition capacity: lack of cold storage, basic processing equipment, and market linkages for differentiated products limits most traders to minimally processed fresh fish. This is a structural constraint documented across sub-Saharan African fish markets by Akande and Diei-Ouadi (2010), Diei-Ouadi and Mgawe (2011), and Musembi *et al.* (2021). Yet the gross price differential between fried (KES 302.2/kg) and fresh whole fish (KES 271/kg) of KES 31.19/kg premium, constitutes a compelling business case for expanded frying operations, confirmed as statistically robust in the econometric analysis (Section 4.5.3c).

Household and individual consumers are the primary target market (73.8% of traders; Fig. 2e), followed by other traders and middlemen (66.3%) and hotels/restaurants (61.3%). Supermarkets and cold storage facilities remain marginal buyers (1–3%), underscoring the dominance of informal supply chains and limited organised retail penetration. This is consistent with patterns described for sub-Saharan African agri-food systems by Reardon *et al.* (2009) and Wills *et al.* (2021). The prevalence of inter-trader sales reflects layered intermediation in the fish value chain, where retail traders often simultaneously perform pass-through wholesaling and direct consumer retail - a structural feature complicating wholesale-retail price margin estimation.

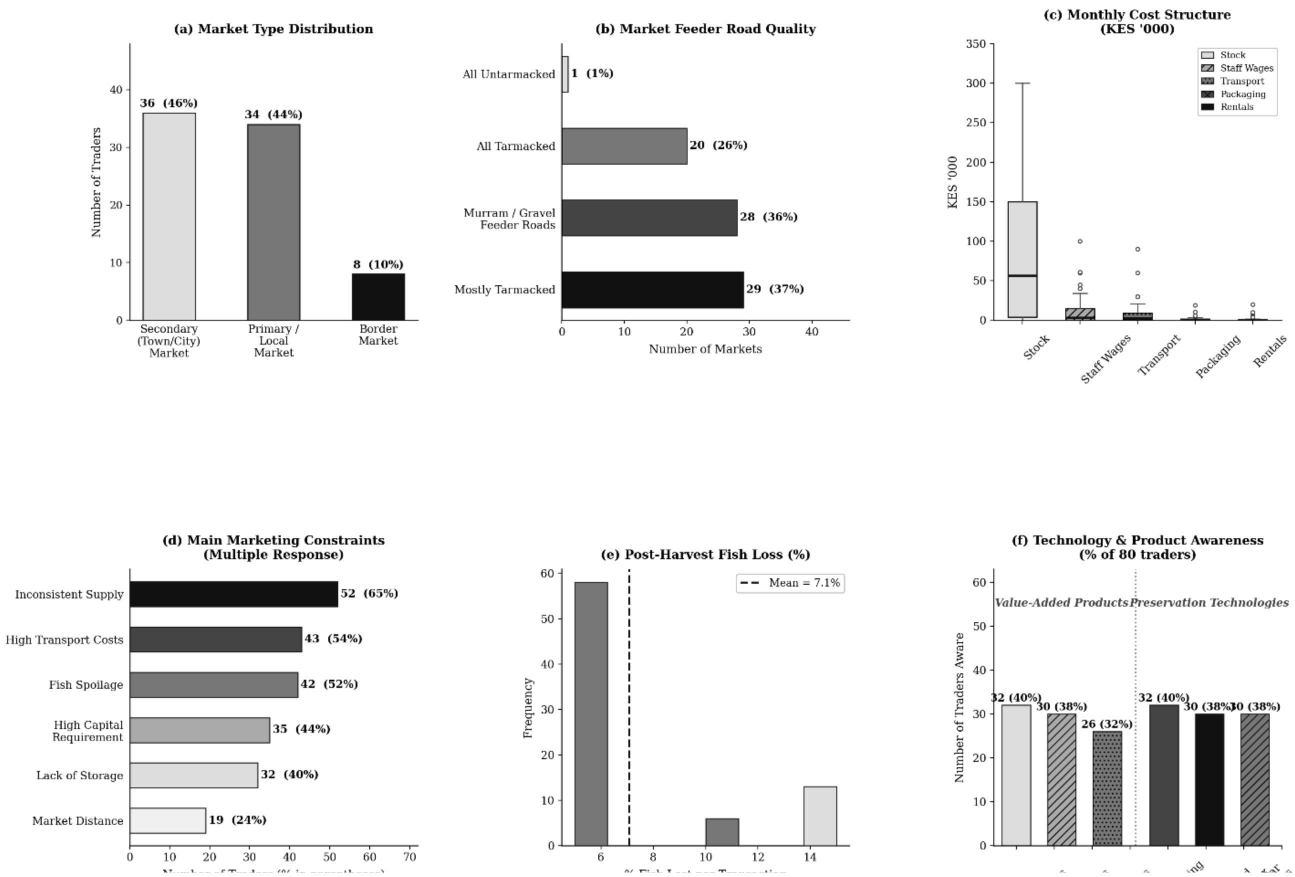

Figure 4. Market structure, costs, and operational constraints (n = 80).

Most traders operate at high frequency: 47.5% sell every day, with a further 35% trading four to five days per week (Fig. 3f). This trading intensity is consistent with the perishability imperative of fresh fish, which compels rapid stock liquidation regardless of prevailing prices - a constraint that also limits traders' pricing power, as noted by Minten and Kyle (1999) and Fafchamps and Gabre-Madhin (2006). Cash on delivery dominates (96.2%), consistent with the spot-market character of informal fish retailing. Mobile banking through M-Pesa is used by 53.8% of traders, representing significant adoption of digital payment infrastructure - consistent with Kenya's documented leadership in mobile money and the findings of Jack and Suri (2011) on M-Pesa adoption among low-income traders.

Market structure, costs, and operational constraints

This section analyses the structural conditions of fish markets, the cost architecture of enterprises, and the operational constraints impeding efficiency - the proximate causes of market dysfunction and contextual backdrop for interpreting price determination results. Fig.ure 4 provides an overview.

The survey covers three market typologies: secondary town/city markets (45%), primary/local markets (42.5%), and border markets in Busia (10%) (Fig. 4a). Raw price differentials are directionally consistent with theoretical expectations: secondary markets record the highest mean prices (KES 298.9/kg), primary markets follow (KES 256.6/kg), and border markets record the lowest (KES 226/kg). The urban price premium of KES 42.30/kg over primary markets reflects higher consumer purchasing power, longer supply chains, and stronger hotel/restaurant demand - findings consistent with Minten and Kyle (1999) and Osewe *et al.* (2019). The below-average border market price is consistent with competitive pressure from Ugandan fish imports at the Busia crossing (Wills *et al.*, 2021; Mkenda and Van Campenhout, 2011). Formal econometric testing of these differentials is presented in Section 4.5.3b.

Market feeder road quality is critical for supply chain efficiency and post-harvest loss rates (Fig. 4b). Approximately 25.6% of traders operate in fully tarmacked corridors, 37.2% on mostly tarmacked roads, and 35.9% on murram (gravel) feeder roads. The prevalence of unpaved feeder roads - particularly in primary and border markets - generates measurable vehicle operating costs, transit delays, and perishable-goods losses consistent with evidence from Africa reviewed by Fafchamps and Gabre-Madhin (2006) and Gollin and Rogerson (2014). Poor road infrastructure represents a systemic spatial friction with direct implications for the geographic price differentials documented in Section 4.5.3d. The infrastructure assessment is consistent with the road-price relationship established by Faminow and Benson (1990) and Minten and Kyle (1999) for other developing country contexts.

Table 3. Monthly business operating costs of fish traders (KES/month, n = 80). KES = Kenyan Shillings. "% Zero Cost" = share reporting no expenditure.

Cost Component	n	Mean (KES)	Median (KES)	Std. Dev.	% Zero Cost
Stock of fish (purchase)	77	109,883	56,000	175,240	0.0
Staff and own wages	64	22,177	3,000	38,600	34.4
Transport	76	7,902	2,500	15,600	0.0
Packaging materials	74	1,590	500	4,100	32.4
Municipal fees (cess)	77	1,230	900	1,040	19.5
Rentals	67	1,329	0	3,200	53.7
Electricity	57	249	0	600	75.4

Fish stock purchase dominates monthly operating costs (mean KES 109,883; median KES 56,000; Fig. 4c and Table 3). The large mean-median gap confirms heavy right skewness driven by a small number of high-volume traders (Fafchamps and Gabre-Madhin, 2006). Staff wages average KES 22,177/month but

Table 4. Main constraints to fish trade, multiple response (n = 80). Multiple responses permitted; percentages sum to >100%.

Constraint	Frequency	% of Respondents	Rank
Inconsistent supply and availability	52	65.0	1
High transportation costs	43	53.8	2
Fish spoilage	42	52.5	3
High capital requirements	35	43.8	4
Lack of storage facilities	32	40.0	5
Time to reach the market	19	23.8	6

with a median of only KES 3,000, confirming most businesses are sole-operator enterprises. Transport costs average KES 7,902/month (median KES 2,500). This is a significant recurrent expense that the regression model treats as a price determinant via the ln(Transport) variable. Municipal cess fees (KES 1,230), packaging (KES 1,590), and rentals (KES 1,329) represent minor but persistent cost items, with high zero-cost shares for rentals (53.7%) and electricity (75.4%) confirming the dominance of informal open-market trading (Reardon *et al.*, 2009).

Traders face multiple co-occurring constraints (Fig. 4d; Table 4). Inconsistent supply and availability is the most pervasive challenge (52 traders, 65.0%), underscoring vulnerability to seasonal fishing effort variability and the absence of aquaculture-based supply buffering - consistent with the dominant constraint profile reported by Wills et al. (2021) for the broader Kenyan fish value chain and by Brummett et al. (2013) for Lake Victoria fisheries. High transportation costs rank second (43, 53.8%), reflecting poor infrastructure and long distances from landing sites - corroborating the infrastructure-cost-price nexus established by Minten and Kyle (1999) and Fafchamps and Gabre-Madhin (2006). Fish spoilage ranks third (42, 52.5%), directly attributable to cold chain absence - consistent with Akande and Diei-Ouadi (2010) who estimate regional post-harvest losses at 20–30% of landed catch. High capital requirements (35, 43.8%) and storage limitations (32, 40.0%) complete the top-five profile.

The mean estimated in-market fish loss is 7.1% of stock (range: 5–15%; Fig. 4e). While below the regional estimates of 20 - 30% reported by Akande and Diei-Ouadi (2010), this loss translates to a direct monthly financial loss of approximately KES 5,500 - 7,700 per trader on mean stock purchases of KES 109,883 translating to a substantial drain on thin margins. Losses arise from spoilage without refrigeration, mechanical damage on unpaved feeder roads, and unsold end-of-day stock. Awareness of post-harvest handling technologies is moderate: the smoking kiln is recognised by approximately 40% of traders, the improved fish display box by 35%, and solar driers by 30% (Fig. 4f). Value-added products including fish sausages (46%), fish samosas (44%), fish fingers (36%), are known to a minority but remain outside current trading repertoires, reflecting gaps in processing skills, equipment access, and institutional market linkages identified in the post-harvest literature (Diei-Ouadi and Mgawe, 2011; Musembi *et al.*, 2021).

4.4 Fish price dynamics and descriptive analysis

Before presenting the econometric analysis, this section characterises fish price variation across key categorical dimensions using Figure 5 and Table 5. This descriptive decomposition establishes the empirical regularities that the regression analysis subsequently tests in a multivariate framework - providing an important bridge between descriptive and inferential components and improving overall scientific flow (Wooldridge, 2013; Musembi *et al.*, 2021).

The pooled mean fish price is KES 272.43/kg (median KES 275/kg; SD = KES 115.2/kg; Figure 4a). The positive skewness (mean > median) reflects a right-tailed distribution driven by a small number of high-value transactions, consistent with price distributions documented in other African fish market studies (Kapute *et al.*, 2018; Musembi *et al.*, 2021). The log-transformation of the price variable employed in the regression model normalises this distribution and improves model fit, confirmed by the superior Adjusted R^2 of the log-linear model (0.3676) relative to the linear OLS specification (0.2494) and by the Ramsey RESET test rejecting the linear functional form ($F = 16.22$, $p < 0.001$) (Wooldridge, 2013).

Nile perch commands the highest mean price (KES 311.2/kg; Fig. 5b), followed by catfish (KES 281.7/kg), tilapia (KES 256.6/kg), and small pelagics (KES 246.5/kg). The Nile perch premium is consistent with evidence from Abila (2003) and Bokea and Ikiara (2000) that larger average size, higher flesh yield, and positioning in hotel/export markets translate into retail price premiums. The low price of small pelagics reflects their market position as a lower-cost protein source for price-sensitive consumers, consistent with Musembi *et al.* (2021). However, Section 4.5.3a demonstrates that these raw differentials lose statistical significance in the multivariate regression once market type, processing form, and geographic controls are included (Osewe *et al.*, 2019).

Mean fish prices are highest in secondary urban markets (KES 298.9/kg), followed by primary/local markets (KES 256.6/kg), and lowest in border markets (KES 226/kg; Fig. 5c). The urban premium of KES 42.30/kg is consistent with theoretical predictions and empirical findings by Minten and Kyle (1999) and Osewe *et al.* (2019). The below-average border price reflects competitive pressure from Ugandan fish imports at Busia (Wills *et al.*, 2021; Mkenda and Van Campenhout, 2011).

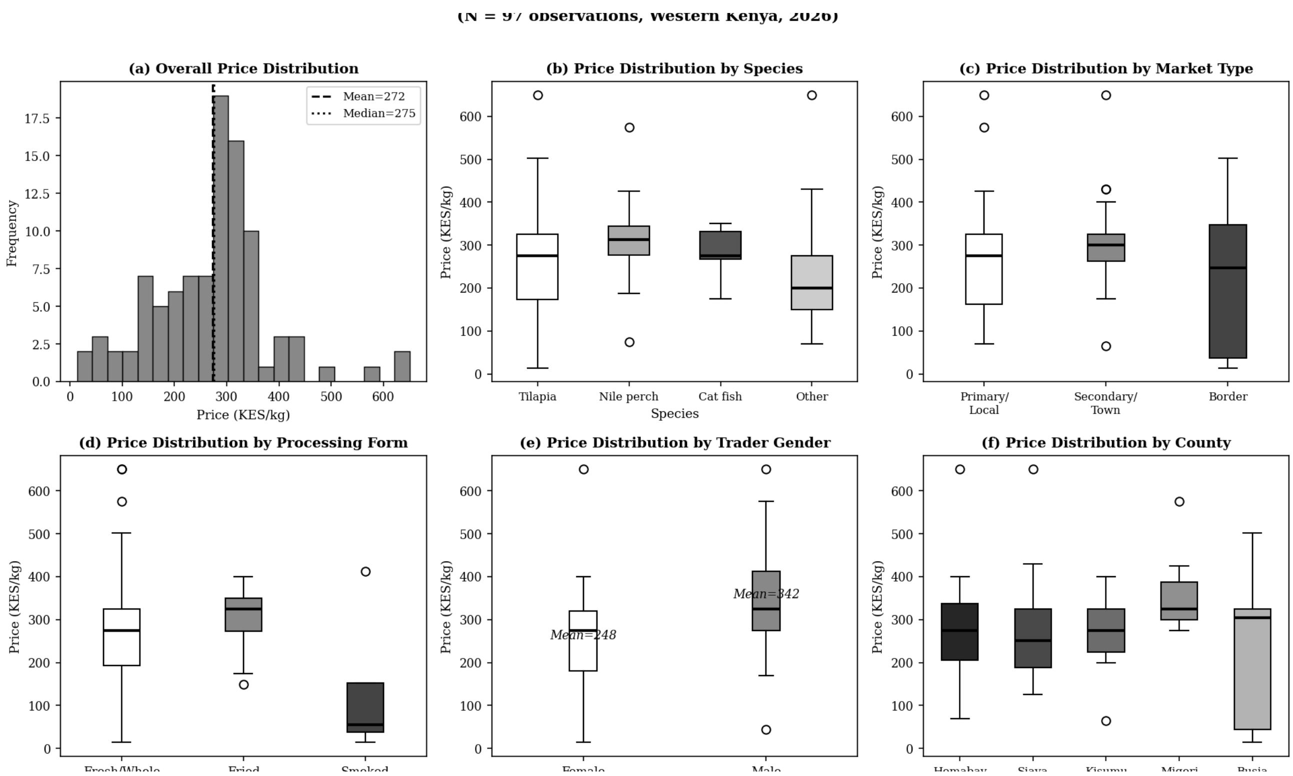

Figure 5. Fish price dynamics and descriptive analysis (n = 97 observations).

Table 5. Summary of fish prices by key categorical variables (KES/kg, n = 97 observations). Sample restricted to non-outlier price observations (≤ Q3 + 3×IQR). Prices in KES/kg.

Category	n	Mean	Median	Std. Dev.	Range (KES)
A. By Species					
Tilapia (Ngege)	36	256.6	275	131	14–650
Nile Perch (Mbuta)	30	311.2	313	80	90–575
Catfish (Mumi)	6	281.7	275	65	200–350
Other species (Omena, Dagaa)	25	246.5	200	128	55–650
B. By Market Type					
Primary/Local Market	47	256.6	275	118	14–650
Secondary/Town Market	42	298.9	300	88	125–600
Border Market	8	226	248	191	14–575
C. By Processing Form					
Fresh/Whole	71	271	275	119	14–650
Fried	22	302.2	325	63	200–450
D. By Trader Gender					
Female traders	-	247.6	275	-	-
Male traders	-	341.9	300	-	-
E. By County					
Homabay	39	267.8	275	108	80–600
Siaya	28	266.5	251	113	55–570
Migori	8	362.5	325	98	250–575

A clear processing premium is evident: fried fish records a mean of KES 302.2/kg compared to KES 271/kg for fresh whole fish representing a 12% gross premium (Fig. 5d). This premium reflects the value added through processing (cooking labour and energy costs, extended shelf life, improved presentation, and consumer convenience) corresponding to the hedonic pricing hypothesis (Rosen, 1974; Diei-Ouadi and Mgawe, 2011; Kapute *et al.*, 2018). This relationship is confirmed as statistically significant and constitutes the study's most policy-actionable finding (Section 4.5.3c).

A notable gender price differential is evident: male traders obtain a mean of KES 341.9/kg compared to KES 247.6/kg for female traders representing a gap of KES 94.30/kg (Fig. 5e). This differential is consistent across market types and species, and as the regression analysis demonstrates (Section 4.5.3e), persists and remains statistically significant after controlling for all observable covariates. This suggests a structural rather than compositional origin (Quisumbing and Pandolfelli, 2010; Njuki and Sanginga, 2013; Doss, 2013). Geographic price variation by county is also visible (Fig. 5f), with Siaya recording lower mean prices which is a finding formally tested in Section 4.5.3d.

Econometric analysis of fish price determinants

This section presents the main scientific contribution of the study - a multivariate econometric analysis of fish price determinants that simultaneously controls for product characteristics, value chain and market structure, processing form, spatial effects, and trader characteristics (Wooldridge, 2013; Rosen, 1974).

Table 6 presents the full results of the parsimonious log-linear model while Figure 6 illustrates key coefficients graphically. The model is jointly significant ($F = 4.534$, $p < 0.001$) and explains 45% of variation in log fish prices ($R^2 = 0.4467$; Adjusted $R^2 = 0.3676$; AIC = 149.91; BIC = 183.38). Three variables are statistically significant at $p < 0.05$: the fried fish dummy, trader gender (male), and Siaya County. All other variables carry theoretically expected signs but do not reach conventional significance thresholds, discussed in detail in Section 4.5.3.

Interpretation of key determinants

Product characteristics - species effects

None of the species dummies attain statistical significance ($p > 0.75$ in all cases), despite raw price differentials suggesting Nile perch commands higher prices than tilapia. Once market type, pro-

Table 6. Log-linear regression results: determinants of fish prices (n = 97). *** p < 0.01; ** p < 0.05; * p < 0.10. HC3 robust SE in parentheses (MacKinnon and White, 1985). Dependent variable: ln(average fish price, KES/kg). % Effect = (e^β - 1) × 100 for dummies; ln(Transport) is a price elasticity.

Variable	Coef. (β)	Robust SE	z-stat	p-value	% Effect on Price
Constant	5.009	(0.450)	11.132	0.000***	-
SPECIES (ref. tilapia)					
Nile Perch	-0.021	(0.128)	-0.166	0.868	-2.1%
Catfish	0.039	(0.210)	0.188	0.851	+4.0%
Other species (Omena/Dagaa)	0.039	(0.137)	0.286	0.775	+4.0%
MARKET TYPE (ref. primary/local)					
Secondary/Town Market	0.120	(0.106)	1.136	0.256	+12.8%
Border Market	-0.674	(0.643)	-1.049	0.294	-49%
PROCESSING FORM (ref. fresh/whole)					
Fried	0.2086**	(0.090)	2.316	0.0206	+23.2%
Smoked	-1.0092	(0.812)	-1.244	0.214	-63.6%
COUNTY (ref. Homabay)					
Siaya County	-0.242**	(0.121)	-2.008	0.0447	-21.5%
Kisumu County	-0.1818	(0.134)	-1.358	0.174	-16.6%
TRADER CHARACTERISTICS					
Wholesaler (ref: retailer)	-0.1638	(0.112)	-1.458	0.145	-15.1%
Male trader	0.5775**	(0.234)	2.465	0.0137	+78.2%
ln (Transport cost, KES/month)	0.0598	(0.055)	1.093	0.274	Elasticity
N = 97 | R² = 0.4467 | Adj. R² = 0.3676 | F = 4.534 (p < 0.001) | AIC = 149.91 | BIC = 183.38					

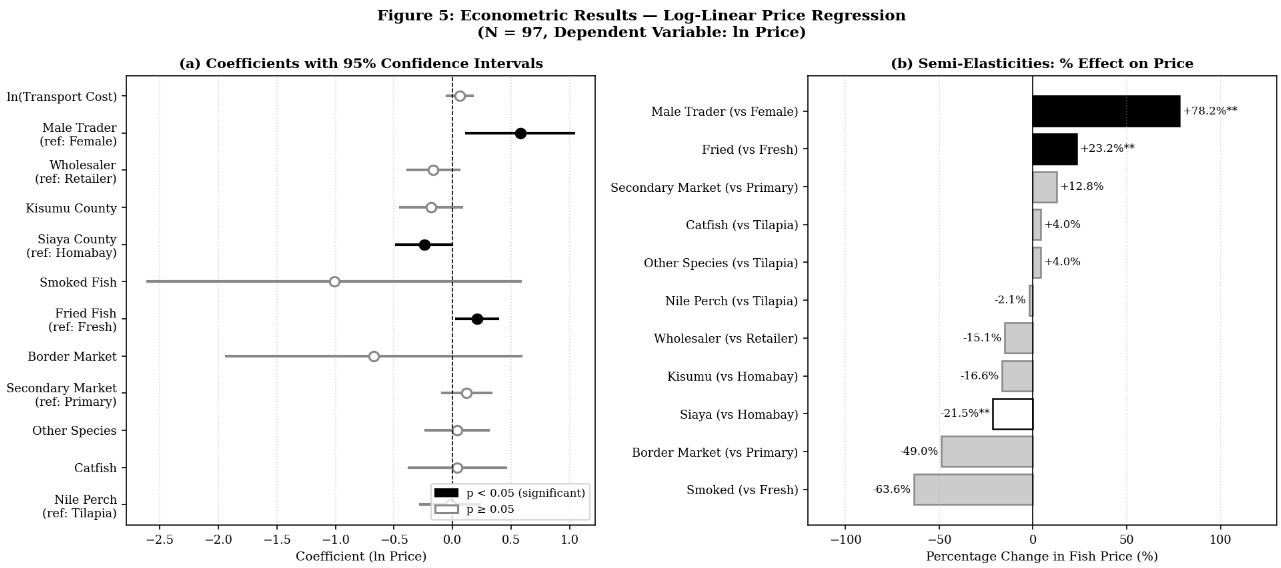

Figure 6. Econometric results of log-linear price regression. (a) Coefficient plot with 95% confidence intervals; (b) semi-elasticities (% effect on price).

cessing form, geographic location, and trader characteristics are controlled for, species identity does not independently predict retail price (Table 6). This finding implies high consumer substitutability between species in informal retail markets and that observed raw species price differentials are largely mediated by other covariates particularly market type and location. This contrasts with Abila (2003) who documented a significant Nile perch premium in formal export-oriented channels, suggesting the premium is more pronounced in formal trade than in the informal retail markets covered here. Osewe *et al*. (2019) similarly find that species identity has declining marginal influence on farmgate prices once market structure variables are included. The result also corroborates Brummett *et al*. (2013), who note that in informal African fish markets, competitive substitution between species constrains species-specific price premiums over time.

Value chain and market structure effects

Secondary urban markets show a positive but non-significant price premium of 12.8% over primary markets ($p = 0.256$). The direction is consistent with Minten and Kyle (1999) and Osewe *et al*. (2019) but insignificance likely reflects high within-category price dispersion. Border markets exhibit a large negative coefficient ($\beta = -0.6739$; 49% discount), consistent with Wills *et al*. (2021) and the competitive pressure of Ugandan imports at the Busia corridor, though non-significant ($p = 0.2943$) due to the small subsample ($n = 8$). The non-significance of the wholesaler dummy (-15.1% discount; $p = 0.1447$) may partly reflect the large share of traders performing both retail and wholesale functions simultaneously, attenuating the margin differential expected by Tirole (1988) and Reardon *et al*. (2009). These results highlight the importance of distinguishing between directionally consistent but underpowered estimates and evidence against theoretical predictions (Wooldridge, 2013).

Value addition - the frying premium

Fried fish commands a statistically significant price premium of 23.2% over fresh whole fish ($\beta = 0.2086$, HC3-robust $z = 2.316$, $p = 0.0206$). This semi-elasticity implies frying multiplies the per-unit price by approximately 1.23, ceteris paribus. The result strongly corroborates the value addition hypothesis advanced by the hedonic pricing framework (Rosen, 1974): processing transforms the physical form of the product, extends shelf life, improves consumer convenience, and thereby commands a higher implicit price (Diei-Ouadi and Mgawe, 2011; Kapute *et al*., 2018; Musembi *et al*., 2021). The magnitude of the premium is comparable to that reported by Kapute *et al*. (2018) for Malawian fish markets and by Musembi *et al*. (2021) for tilapia in Nairobi. Smoked fish yields a large negative but insignificant coefficient ($\beta = -1.0092$, $p = 0.2137$), interpreted with caution given $n = 4$ for this category and the distinct consumer market served by smoked fish (price-sensitive buyers and inter-regional traders; Akande and Diei-Ouadi, 2010).

Spatial effects - county-level price differentials

Siaya County records a statistically significant price discount of 21.5% relative to Homabay ($\beta = -0.242$, $p = 0.0447$), surviving after controlling for all observable covariates. Based on spatial market integration theory (Ravallion, 1986; Faminow and Benson, 1990; Barrett, 1996), persistent statistically significant price gaps between proximate counties sharing common lake supply sources signal market segmentation or spatial arbitrage failures. The most probable mechanisms are inferior feeder road quality along Siaya supply routes and information asymmetries between Siaya landing sites and retail markets consistent with the infrastructure patterns in Section 4.3.2 and with Fafchamps and Gabre-Madhin (2006) and Mkenda and Van Campenhout (2011). The finding parallels Abdulai (2000) who identifies persistent geographic price gaps in Ghanaian markets as evidence of incomplete spatial market integration. Kisumu County shows a negative but non-significant coefficient ($\beta = -0.1818$, $p = 0.1743$), suggesting a 16.6% discount that does not reach statistical significance possibly reflecting Kisumu's competitive urban pricing environment (Goletti and Babu, 1994).

Trader characteristics - gender price gap and intermediation

Male traders command prices 78.2% higher than female traders, all else equal (β = 0.5775, p = 0.0137). This substantial gender premium persisting after controlling for all observable covariates strongly suggests a structural rather than compositional origin. Multiple mechanisms are operative: male traders may access capital-intensive transport and bulk sourcing enabling greater market selectivity; maintain stronger social networks with high-value institutional buyers (Makhura *et al.*, 2001; Fafchamps and Gabre-Madhin, 2006); hold more established trading relationships commanding trust-based premiums; and face fewer time constraints due to domestic labour burdens disproportionately falling on female traders (Njuki and Sanginga, 2013; Doss, 2013; FAO, 2020). The magnitude of the estimated gap is consistent with Quisumbing and Pandolfelli (2010), who demonstrate across multiple African contexts that gender market outcome differentials persist even after controlling for observable differences in human capital and assets. Fischer and Qaim (2012) document comparable gender price gaps in Kenyan horticultural markets, while FAO (2020) identifies the gender price gap as a priority equity concern across sub-Saharan African food systems.

Wholesalers receive prices 15.1% lower than retailers (β = -0.1638, p = 0.1447), consistent with the theory of trade intermediation (Tirole, 1988; Reardon *et al.*, 2009), though statistical non-significance likely reflects the high prevalence of dual-role traders (48.8%) blurring the wholesale-retail boundary and attenuating the estimated margin differential.

Transaction costs - transport elasticity and cost pass-through

The transport cost elasticity is positive (β = 0.0598) but not statistically significant (p = 0.2742). While the direction conforms to the a priori expectation that higher transport costs raise consumer prices (Minten and Kyle, 1999; Gollin and Rogerson, 2014), the insignificance suggests incomplete cost pass-through. Two mechanisms can explain this, first, competitive market pressures combined with the perishability constraint limit sellers' pricing power, in line with Fafchamps and Gabre-Madhin (2006) and two, transport costs are measured as monthly aggregates rather than per-unit or per-trip figures, which introduces noise that weakens the observed relationship. Incomplete pass-through is consistent with findings by Abdulai (2000) and Goletti and Babu (1994) on asymmetric price transmission in developing country agricultural markets, where competitive pressure compresses the share of cost increases passed through to final prices.

Model diagnostics and robustness checks

Table 7 presents the full diagnostic test suite (Figure 6 provides diagnostic plots). The Breusch-Pagan test (LM = 32.75, p = 0.005) confirms heteroscedasticity (Fig. 7), validating the HC3 robust standard errors (MacKinnon and White, 1985; Davidson and MacKinnon, 1993). Non-normal residuals (Jarque-Bera, $p < 0.001$) are expected with price data and do not invalidate large-sample OLS inference under HC3 (Wooldridge, 2013). The Ramsey RESET test (F = 16.22, $p < 0.001$) confirms non-linearity in the linear OLS specification, providing additional empirical justification for the log transformation. The Durbin-Watson statistic (DW = 2.077) provides no evidence of first-order serial autocorrelation, as expected for cross-sectional data. All VIF values are below 3.0, confirming no problematic multicollinearity (Hair *et al.*, 2010). The log-linear model outperforms the OLS linear specification on all information criteria (AIC: 149.91 vs 200; Adj. R^2: 0.3676 vs 0.2494). Collectively, these diagnostics confirm the preferred specification is appropriate and that the key inferences i.e. the frying premium, gender price gap, and Siaya discount are robust to alternative specifications (Wooldridge, 2013).

Table 7. Model diagnostic and specification test results. HC3 = heteroscedasticity-consistent estimator (MacKinnon and White, 1985). VIF = variance inflation factor (Hair *et al.*, 2010). Source: Authors' computation.

Test	Statistic	p-value	Conclusion
Breusch-Pagan (heteroscedasticity)	LM = 32.75	0.005	Heteroscedasticity present → HC3 robust SE applied
Jarque-Bera (residual normality)	JB = 103.96	< 0.001	Non-normal; large-sample inference valid under HC3
Ramsey RESET (functional form)	F = 16.22	< 0.001	Non-linearity in OLS linear → log transformation justified
Durbin-Watson (autocorrelation)	DW = 2.077	-	No evidence of serial autocorrelation
VIF (multicollinearity)	Max VIF = 2.71	-	All VIF < 3.0; no severe multicollinearity
OLS linear model (robustness)	R^2 = 0.3432	Adj R^2 = 0.2494	Inferior fit confirms log-linear is preferred specification

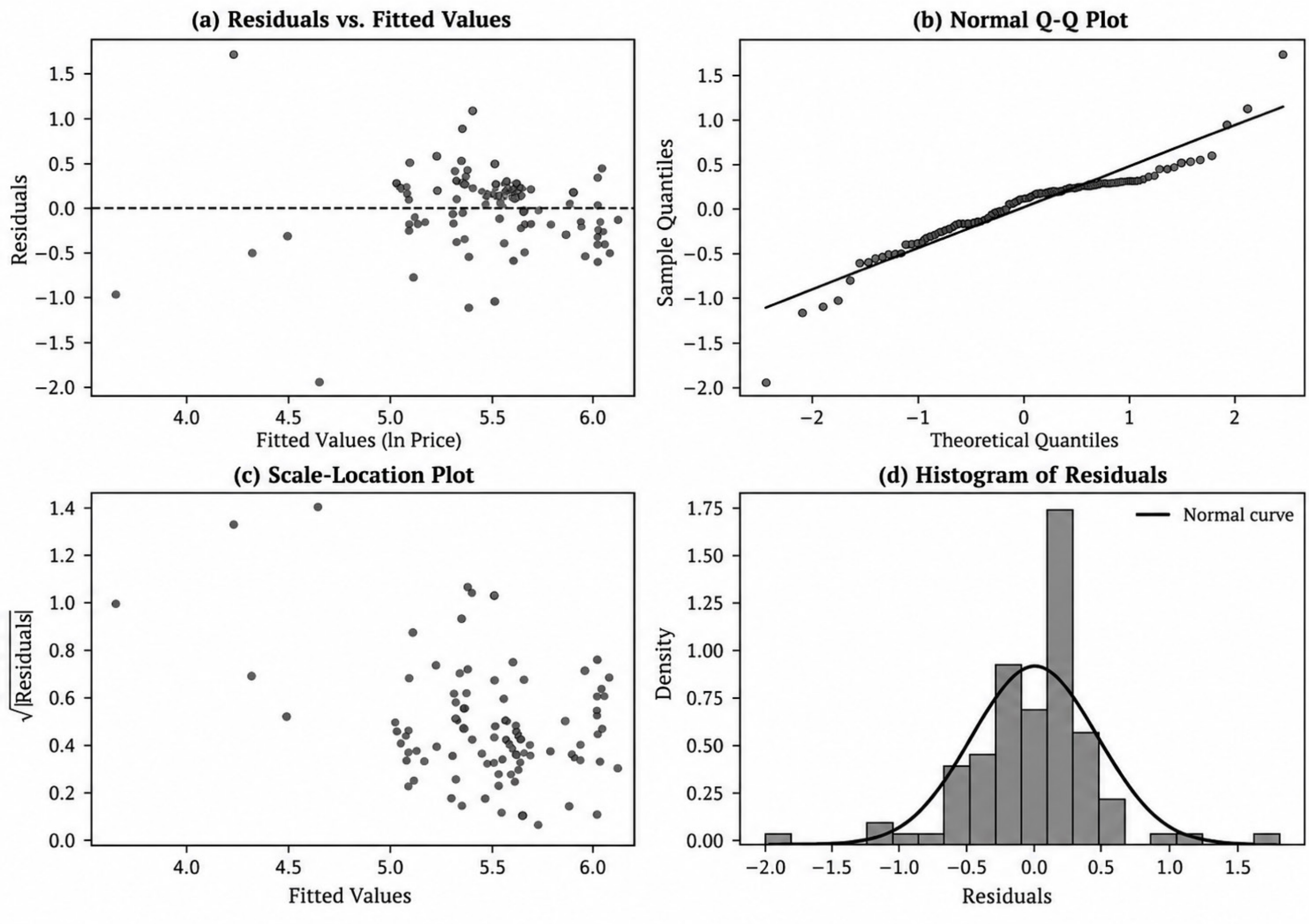

Figure 7. Model diagnostic plots of log-linear regression.

Conclusion and recommendations

This study provides an integrated analysis of fish marketing systems in Western Kenya, showing that the sector is dominated by small-scale, informal, and female-led enterprises operating under significant structural constraints, including inconsistent fish supply, high transport costs, inadequate storage infrastructure, and post-harvest losses. Econometric analysis identified value addition through frying, trader gender, and spatial location as the key determinants of fish prices, while species type and market category were not significant once value chain and trader-level factors were controlled for. The findings demonstrate that fish price formation is shaped more by structural and institutional conditions than by intrinsic product characteristics, highlighting the central role of value chain inefficiencies in determi

ning trader welfare and market outcomes. To improve efficiency and equity, policy interventions should prioritise post-harvest value addition, women's access to capital and high-value markets, improved road and market infrastructure, expansion of cold chain systems, and access to tailored financial services to strengthen market integration, reduce losses, and enhance the contribution of fisheries to livelihoods and food security in Kenya.

Acknowledgement

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Epilithic diatoms as bioindicators of water quality in the Njoro and Kamweti rivers, Kenya

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Abstract

Diatoms were used as bioindicators to assess the water quality and ecological status of the Njoro and Kamweti rivers in Kenya. The physicochemical characteristics of the two rivers differed significantly ($P < 0.05$). Twenty-seven diatom species across eight classes were identified, of which 25 were common to both rivers, while *Eunotia exigua* and *Amphora libyca* occurred only in the Kamweti River. Pollution-tolerant genera, particularly *Gomphonema* and *Nitzschia*, were abundant in the Njoro River, whereas the Kamweti River supported a higher diversity of diatoms, indicating better ecological conditions. Diatom abundance increased downstream in both rivers, reflecting the cumulative effects of catchment activities along the river continuum. The Trophic Diatom Index, Diatom Pollution Sensitivity and Saprobic Index likewise increased downstream and were consistently higher in the Njoro River, consistent with greater nutrient loading and organic pollution and a shift toward assemblages that are less sensitive to pollution. The study provides baseline information to support the sustainable management and conservation of freshwater resources in Kenya.

Keywords: Epilithic diatoms, bioindicators, water quality, ecological assessment, biomonitoring, tropical rivers

Introduction

Freshwater ecosystems are increasingly threatened by rapid land-use change driven by population growth, agricultural expansion and urban development (Vorosmarty *et al.*, 2010). These pressures alter catchment processes and, in turn, hydrological and biogeochemical cycles (Meybeck, 2003), so that rivers receive increasing loads of nutrients, sediments and organic pollutants from the surrounding landscape (Allan, 2004). The result is a progressive decline in water quality and a disruption of aquatic ecosystem functions (Dudgeon *et al.*, 2006). In many tropical regions these impacts are exacerbated by poor land management and limited wastewater treatment, and most tropical rivers are consequently experiencing rapid, human-driven ecological deterioration (Dudgeon *et al.*, 2006).

River systems are highly sensitive to catchment disturbance because they integrate upstream inputs along a longitudinal continuum (Allan, 2004). Owing to catchment connectivity, pollutants and nutrients move downstream and accumulate, producing spatial gradients in water quality in which concentrations are generally higher towards the bottom of the catchment (Dudgeon *et al.*, 2006). These gradients are reflected in the composition of the biological community and can be used to assess river health across the landscape. Benthic diatoms are widely used as bioindicators of the overall condition of freshwater ecosystems: their well-documented sensitivity to environmental change (Stevenson *et al.*, 2010) gives a rapid response to nutrient enrichment, organic pollution, and shifts in pH and conductivity (Smol and Stoermer, 2010). The high ecological specificity and diversity of benthic diatom assemblages provide fine resolution on ecosystem water quality (Bellinger and Sigee, 2015), and diatoms therefore play a central role in biomonitoring programmes globally.

Several biological communities, including macroinvertebrates and fish, have been used to assess and monitor freshwater quality (Lobo *et al.*, 2016). However, few studies have used diatoms as bioindicators in river ecosystems in Kenya. Diatoms are unicellular microalgae that are sensitive to environmental change in lakes and rivers (Bere and Mangadze, 2014; Flower and Williams, 2023) and can adapt to varied conditions, making them effective indicators of river ecosystem quality (Stenert *et al.*, 2018; Mbao *et al.*, 2022). The objectives of this study were to (i) analyse the distribution of benthic diatoms in the Njoro and Kamweti rivers; (ii) determine the Trophic Diatom Index, Saprobic Index and Diatom Pollution Sensitivity and compare nutrient levels and organic pollution between the two rivers; and (iii) evaluate diatom biodiversity using the Shannon index and assess species richness in the two rivers. Integrating biological and physicochemical data provides insight into how each river has responded to changes in land use and offers important baseline information for the sustainable management and conservation of tropical riverine ecosystems in Kenya.

Materials and methods

Description of study sites

The Kamweti River, the reference site, is located at approximately 0.413° S, 37.346° E. Its catchment lies on the southern side of Mount Kenya within Mount Kenya National Park and is dominated by Afromontane forest. The upstream reaches are characterised by minimal anthropogenic activity, dense riparian vegetation and well-oxygenated water, making the river an ideal reference system. The Njoro River is a second-order stream that rises on the eastern slopes of the Mau Escarpment (0.3978° S, 35.8961° E) and flows for about 55 km before discharging into Lake Nakuru, a Ramsar Wetland of International Importance. Along its course the Njoro River receives inputs from agriculture, settlements and urban areas, which affect its water quality to varying degrees. A total of 10 sampling sites was selected — five on each river.

Sampling procedure and laboratory analysis

Dissolved oxygen (DO), electrical conductivity (EC), total dissolved solids (TDS), pH, turbidity and temperature were measured *in situ* at each sampling site using a multi-probe water quality meter (OAKTON 650 LCD). Water samples for nutrient analysis were collected from flowing water in plastic bottles, stored in a cool box at about 0–5 °C and transported to the laboratory within five to six hours for filtration. In the laboratory, soluble reactive phosphorus (SRP), total nitrogen (TN) and total phosphorus (TP) were determined using standard colorimetric methods following APHA (1995).

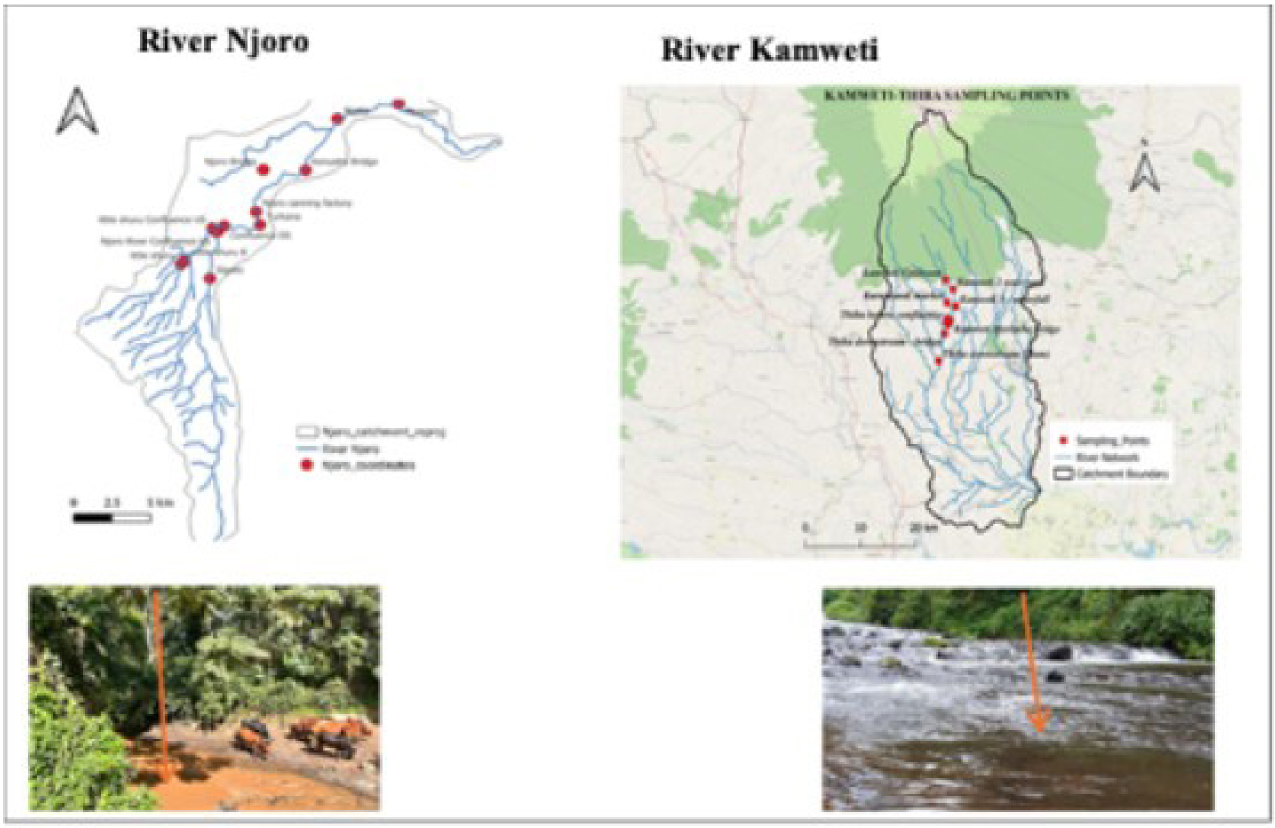

Figure 1. Schematic location map of the Njoro and Kamweti rivers, Kenya, showing the sampling sites along each river from upstream to downstream.

Diatom collection

At each site, eight rocks of similar size (gravel to cobble) were collected at random following standard protocols. Each rock was fitted with a plastic O-ring, and a surface area of about 40 mm diameter was scraped with a toothbrush; the material was rinsed into a 50 mL plastic bottle and fixed with 99% ethanol. In the laboratory, the samples from each site were pooled into a composite sample in a 100 mL bottle. A 4 mL subsample was placed in a boiling tube and heated for about 2 minutes, and the diatoms were cleaned using concentrated acids (2 mL each of HCl, HNO_3 and H_2SO_4) following Kahlert *et al.* (2012). Samples were centrifuged at 3,500 rpm for 4 minutes, and 1 mL was mounted in Naphrax following Weihoefer and Pan (2006). Diatoms were identified under immersion oil at 1,200× magnification as described by Tan *et al.* (2013), following Krammer and Lange-Bertalot (1986, 1988, 1991a, b) and the guidelines of Barber and Haworth (1981). The identified taxa, with photographs, are documented in the Supplementary file.

Data analysis

Descriptive statistics (means and standard deviations) were calculated for each physicochemical parameter, and one-way analysis of variance (ANOVA) was used to test for differences among sampling sites at the 5% significance level (in R and Python). Three diatom-based indices were used to assess the ecological health of the rivers (Table 1): the Trophic Diatom Index (TDI), based on the sensitivity and indicator values of each species (Kelly and Whitton, 1995); the Diatom Pollution Sensitivity (DPS), based on the formula of Kelly and Whitton (1995); and the Saprobic Index (SI), derived from the saprobic values and frequencies of occurrence of each species following Rott *et al.* (1997). Differences in diatom community composition among sites were tested using a permutational multivariate analysis of variance (PERMANOVA) based on the Bray–Curtis dissimilarity. The relationship between sampling sites and environmental variables was explored using principal component analysis (PCA) of the standardised environmental data. Species richness, the Shannon–Wiener index (H′), Simpson's index (D) and evenness were calculated in PAST software as measures of diatom diversity.

Table 1. Diatom indices used to assess the ecological status of the rivers.

Diatom index	Formula and notation
Trophic Diatom Index (TDI)	TDI = (WMS × 25) - 25, where WMS is the weighted mean sensitivity; for species *j*, A*j* = relative abundance, S*j* = pollution sensitivity (1 ≤ S ≤ 5) and V*j* = indicator value (1 ≤ V ≤ 3) (Kelly and Whitton, 1995)
Diatom Pollution Sensitivity (DPS)	Based on the relative abundance (A*j*) and pollution-sensitivity value (S*j*) of each species *j* (Kelly and Whitton, 1995)
Saprobic Index (SI)	Derived from the saprobic value (*si*) and frequency of occurrence (*hi*; rare = 1, frequent = 3, abundant = 5) of each species *i* (Rott *et al.*, 1997)

TDI interpretation: 0–20, very low; 20–40, low; 40–60, moderate; 60–80, high; >80, very high nutrient loading.

Results

Physicochemical parameters

Mean values of the environmental variables across all sampling sites are summarised in Table 2. The variables differed significantly among sites (one-way ANOVA, P < 0.05). The Kamweti River consistently had lower ionic strength and nutrient concentrations than the Njoro River; at the Kamweti reference site (Kamweti-1), EC (21.67 ± 1.15 µS/cm), SRP (18.33 ± 1.15 µg/L), TDS (10.33 ± 1.15 mg/L), TN (0.80 ± 0.00 mg/L), TP (49.67 ± 0.58 µg/L) and turbidity (5.00 ± 1.00 NTU) were all low, while DO (8.83 ± 0.06 mg/L) was high and temperature low (13.90 ± 0.26 °C). The Njoro River had significantly higher EC, TDS, TN, TP, turbidity and pH than the Kamweti River (P ≤ 0.05; Table 2).

Diatom taxa and abundance

The study recorded 27 diatom species from eight classes in the two rivers (Table 3). Twenty-five species were shared between the rivers, while *Eunotia exigua* and *Amphora libyca* (Ehr.) occurred only in the Kamweti River. *Gomphonema* was the most abundant genus (187 individuals), most of them in the Njoro River (133), followed by *Nitzschia* (152 individuals; 84 in the Njoro River). Several species were comparatively abundant in the Kamweti River, including *Achnanthes inflata*, *Achnanthes lanceolata*, *Cocconeis placentula*, *Amphora libyca*, *Amphora pediculus*, *Epithemia sorex* and *Epithemia sorex* var. *gracilis*. A PERMANOVA based on the Bray–Curtis distance indicated a significant difference in diatom abundance between the two rivers (F = 3.268; P = 0.008). A post-hoc Mann–Whitney U test showed that, although community structure differed overall, no individual species differed significantly between the rivers at P = 0.05.

Table 2. Environmental variables (mean ± standard error) at the 10 sampling sites. Sigotik to Mogoon are on the Njoro River; Kamweti-1 to Thiba are on the Kamweti River.

Parameter	Sigotik	Little Shuru	Kenyatta	Ngata	Mogoon	Kamweti-1	Kamweti-2	Kamweti-3	Kamweti-5	Thiba
DO (mg/L)	6.64 ± 0.77	7.40 ± 0.53	7.24 ± 0.23	7.56 ± 0.30	7.93 ± 0.34	8.83 ± 0.06	8.37 ± 0.15	8.53 ± 0.40	8.20 ± 0.20	6.90 ± 0.53
EC (µS/cm)	31.67 ± 3.06	44.00 ± 2.00	41.00 ± 1.00	51.33 ± 2.08	51.00 ± 4.58	21.67 ± 1.15	22.67 ± 1.53	23.00 ± 0.00	24.67 ± 0.58	25.33 ± 2.08
pH	7.87 ± 0.06	7.53 ± 0.06	7.50 ± 0.10	8.07 ± 0.57	8.43 ± 0.42	7.10 ± 0.10	7.27 ± 0.12	7.17 ± 0.31	7.07 ± 0.15	7.03 ± 0.15
SRP (µg/L)	16.33 ± 2.31	21.67 ± 0.58	13.67 ± 1.15	14.67 ± 0.58	42.67 ± 2.52	18.33 ± 1.15	12.00 ± 0.00	16.67 ± 0.58	12.67 ± 1.53	18.33 ± 1.53
TDS (mg/L)	29.33 ± 0.58	40.67 ± 1.15	48.33 ± 1.15	45.67 ± 1.15	45.67 ± 1.15	10.33 ± 1.15	11.00 ± 1.00	10.67 ± 0.58	8.00 ± 0.00	8.00 ± 0.00
Temperature (°C)	19.00 ± 0.56	19.27 ± 0.91	18.30 ± 0.85	16.83 ± 0.35	16.93 ± 0.23	13.90 ± 0.26	15.53 ± 0.60	16.47 ± 0.21	16.90 ± 0.60	18.83 ± 0.42
TN (mg/L)	0.60 ± 0.00	0.84 ± 0.01	0.81 ± 0.01	0.72 ± 0.03	0.84 ± 0.01	0.80 ± 0.00	0.83 ± 0.01	0.76 ± 0.09	0.62 ± 0.01	0.70 ± 0.01
TP (µg/L)	41.33 ± 1.15	44.33 ± 0.58	80.67 ± 1.53	122.00 ± 2.65	143.67 ± 4.04	49.67 ± 0.58	35.33 ± 0.58	36.33 ± 0.58	38.33 ± 1.15	44.33 ± 2.31
Turbidity (NTU)	29.67 ± 0.58	32.33 ± 2.08	41.67 ± 1.53	46.67 ± 1.15	51.67 ± 2.31	5.00 ± 1.00	9.00 ± 0.00	13.00 ± 1.73	14.00 ± 0.00	15.67 ± 1.15

Table 3. Distribution of benthic diatoms across the sampling sites of the Njoro and Kamweti rivers. SIG, Sigotik; LSH, Little Shuru; KEN, Kenyatta; NGA, Ngata; MOG, Mogoon; KAM-1 to KAM-5, Kamweti 1–5; THI, Thiba. (+) present; (-) absent.

Species	SIG	LSH	KEN	NGA	MOG	KAM-1	KAM-2	KAM-3	KAM-5	THI
Cyclotella meneghiniana	+	+	-	-	-	+	+	-	+	+
Fragilaria brevistriata	+	+	+	-	-	+	-	+	+	+
Fragilaria ulna	+	-	-	-	-	-	+	+	+	+
Eunotia exigua	-	-	-	-	-	+	+	+	+	+
Achnanthes inflata	+	+	+	-	-	+	+	+	+	+
Achnanthes lanceolata	-	-	-	+	-	+	+	+	+	+
Cocconeis placentula	+	+	+	-	+	-	-	-	+	+
Amphora libyca	-	-	-	-	-	+	+	+	-	+
Amphora pediculus	+	+	+	-	-	+	+	+	+	+
Anomoeoneis sphaerophora	+	+	+	-	-	+	+	+	+	+
Gomphonema gracile	+	+	+	+	+	+	+	-	-	+
Gomphonema parvulum	+	+	-	+	+	-	-	+	+	+
Gomphonema subclavatum	+	+	+	+	+	-	+	+	-	-
Navicula cincta	-	-	+	-	-	-	+	+	-	-
Navicula cryptocephala	+	+	-	+	-	+	-	-	+	+
Pinnularia subcapitata	+	+	+	-	-	-	-	-	+	+
Pinnularia viridis	+	+	-	-	-	+	-	+	+	+
Sellaphora pupula	+	+	-	-	-	-	+	+	-	-
Cymbella spp.	+	-	-	-	-	+	-	+	+	+
Stauroneis anceps	-	+	-	-	-	-	+	+	+	+
Stauroneis phoenicenteron	+	+	+	-	+	+	+	-	-	+
Hantzschia amphioxys	+	+	-	-	-	-	+	+	-	+
Nitzschia amphibia	+	+	+	+	-	+	+	+	+	+
Nitzschia palea	-	-	+	+	+	-	-	+	-	-
Epithemia sorex	+	-	+	-	-	+	+	-	+	+
Epithemia sorex var. *gracilis*	-	-	+	-	+	+	+	+	-	+
Surirella angusta	+	-	+	-	+	-	-	+	+	-
Surirella brebissonii	+	-	-	-	-	+	+	+	-	+

Principal component analysis of the environmental data separated the sites clearly along the first two axes, which together explained about 82% of the variation (Fig. 2). The Njoro sites grouped on the side associated with high EC, TDS, turbidity, TP, SRP and pH, whereas the Kamweti sites grouped with high dissolved oxygen, indicating that nutrient and ionic gradients were the main environmental drivers separating the two rivers.

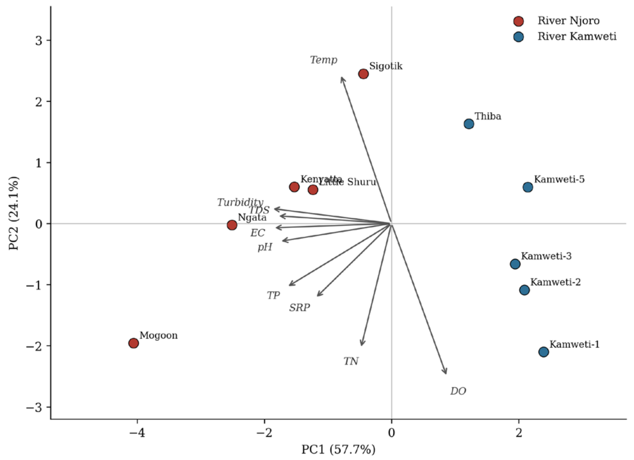

Figure 2. Principal component analysis (PCA) ordination of the sampling sites based on the standardised environmental variables. Arrows indicate the direction and relative strength of each environmental gradient. Sites of the Njoro River (red) separate from those of the Kamweti River (blue) along the nutrient and ionic gradient.

Diatom diversity, evenness and richness

Species richness, Shannon–Wiener diversity, Simpson's index and evenness for the 10 sites are given in Table 4. Overall, the Kamweti River had higher species richness and diversity than the Njoro River. In the Njoro River, both species richness (20 to 8) and Shannon–Wiener diversity (2.89 to 1.94) decreased from upstream to downstream, with a corresponding decline in Simpson's index and evenness. In contrast, the Kamweti River showed a longitudinal increase in species richness (17 to 22), Shannon–Wiener diversity (2.69 to 3.03), Simpson's index (0.93 to 0.95) and evenness from upstream to downstream.

Table 4. Diatom diversity indices at the sampling sites of the Njoro River (Sigotik–Mogoon) and Kamweti River (Kamweti-1–Thiba).

Diversity index	Sigotik	Little Shuru	Kenyatta	Ngata	Mogoon	Kamweti-1	Kamweti-2	Kamweti-3	Kamweti-5	Thiba
Shannon–Wiener (H')	2.89	2.73	2.56	1.84	1.94	2.69	2.91	2.92	2.78	3.03
Simpson's index (D)	0.94	0.93	0.91	0.83	0.84	0.93	0.94	0.94	0.93	0.95
Species richness	20	17	15	7	8	17	20	20	17	22
Evenness	0.96	0.96	0.94	0.94	0.93	0.95	0.97	0.98	0.98	0.98

Diatom indices for water quality assessment

The diatom indices were broadly consistent across sites and were generally higher in the Njoro River than in the Kamweti River (Table 5). Within the Njoro River, DPS ranged from 3.69 to 5.35, TDI from 45.90 to 60.98 and SI from 2.84 to 3.44. DPS was notably higher at the mid- and downstream sites — Mogoon (5.35), Ngata (4.25) and Kenyatta (4.04) — reflecting increasing ecological pressure along the river continuum. All three indices were higher in the Njoro River, indicating greater nutrient enrichment and organic pollution. All indices also increased from upstream to downstream in both rivers: the highest TDI in the Njoro River (60.98, at Mogoon) exceeded the highest TDI in the Kamweti River (46.44, at Thiba).

Table 5. Diatom indices for water quality assessment at the sampling sites.

Sampling point	DPS	TDI	SI	River
Sigotik	3.69	45.90	2.84	Njoro
Little Shuru	4.10	47.94	2.92	Njoro
Kenyatta	4.04	52.78	3.11	Njoro
Ngata	4.25	55.60	3.22	Njoro
Mogoon	5.35	60.98	3.44	Njoro
Kamweti-1	3.36	37.28	2.30	Kamweti
Kamweti-2	3.48	39.05	2.60	Kamweti
Kamweti-3	3.50	37.12	2.68	Kamweti
Kamweti-5	3.88	38.78	2.68	Kamweti
Thiba	4.37	46.44	2.98	Kamweti

Pollution-tolerant and oligotrophic indicator species

Indicator-species analysis showed a clear association between the ecological separation of sites and the DPS and TDI metrics. The 10 species with the highest pollution-tolerance (DPS) indicator values (Table 6) were mostly taxa typically associated with nutrient enrichment and organic pollution. *Gomphonema gracile* had the highest value (79.6), followed by *Stauroneis phoenicenteron* (66.2), *Gomphonema parvulum* (55.8), *Gomphonema subclavatum* (52.2), *Cocconeis placentula* (50.9) and *Nitzschia palea* (46.8); collectively these taxa indicate increased ecological stress and were strongly associated with the impacted sites.

The oligotrophic indicator group (species with low TDI values; Table 7) was representative of low-nutrient, less-disturbed conditions. *Amphora pediculus* had the highest oligotrophic indicator value (64.29), followed by *Anomoeoneis sphaerophora* (62.50) and *Fragilaria ulna* (62.22). Other notable taxa included *Achnanthes inflata* (60.71), *Epithemia sorex* (56.73), *Achnanthes lanceolata* (56.00), *Cyclotella meneghiniana* (55.85), *Eunotia exigua* (51.93), *Pinnularia viridis* (51.76) and *Surirella brebissonii* (50.53). These taxa were strongly associated with oligotrophic conditions and indicative of the relatively high water quality of the upstream sites.

Table 6. Top 10 pollution-tolerant indicator species based on Diatom Pollution Sensitivity (DPS).

Species	Indicator value
Gomphonema gracile	79.6
Stauroneis phoenicenteron	66.2
Gomphonema parvulum	55.8
Gomphonema subclavatum	52.2
Cocconeis placentula	50.9
Nitzschia palea	46.8
Nitzschia amphibia	44.7
Pinnularia subcapitata	30.0
Stauroneis anceps	27.7
Navicula cryptocephala	26.2

Table 7. Top 10 oligotrophic indicator species based on low Trophic Diatom Index (TDI) values.

Species	Indicator value
Amphora pediculus	64.29
Anomoeoneis sphaerophora	62.50
Fragilaria ulna	62.22
Achnanthes inflata	60.71
Epithemia sorex	56.73
Achnanthes lanceolata	56.00
Cyclotella meneghiniana	55.85
Eunotia exigua	51.93
Pinnularia viridis	51.76
Surirella brebissonii	50.53

Discussion

In many rural communities, water for domestic use is drawn from surface freshwaters such as streams and rivers, which also support fisheries, irrigation, and municipal and industrial supply as well as recreation (Koskey *et al.*, 2021; Siddha and Sahu, 2022). Over recent decades, however, many riverine ecosystems have been degraded by catchment deforestation, agricultural intensification and industrial development. In East Africa, rivers are also frequently used as sinks for waste disposal, which lowers water quality, and poor river management is a major cause of nutrient enrichment and siltation in African lakes.

Water quality parameters

Physicochemical variables are major drivers of diatom species richness in rivers (Dalu *et al.*, 2017; Jyrkankallio-Mikkola *et al.*, 2018). The two rivers differed significantly, with poorer water quality in the Njoro River associated with elevated TP, TN, turbidity and TDS and lower DO, which together shape the composition and distribution of diatoms (Triest *et al.*, 2012; Bere and Mangadze, 2014). Conductivity reflects the ionic content that influences nutrient availability and uptake by diatoms during primary production (Shibabaw *et al.*, 2021). The higher turbidity in the Njoro River may also reduce the light reaching the substrate and affect diatom development (Dalu *et al.*, 2017), and the high nutrient sites

(Kenyatta, Ngata and Mogoon) supported lower species richness. Water pH was an important factor, affecting diatom growth both directly, through physiological stress, and indirectly, by modifying other physicochemical variables (Shibabaw *et al.*, 2021).

Distribution of epilithic diatoms in the Kamweti and Njoro rivers

The distribution of epilithic diatoms in the two rivers was spatially distinct and reflected the environmental quality and degree of human disturbance in each. The upper reach of the Kamweti River (Kamweti-1), widely regarded as a reference site because of its near-pristine condition (Koskey *et al.*, 2021), supported a consistently high abundance of *Eunotia exigua*, a taxon characteristic of oligotrophic, low-conductivity and undisturbed freshwaters (Van Dam *et al.*, 1994; Stevenson *et al.*, 2010). This indicates high ecological integrity and limited human influence. A distinct change in assemblage composition occurred in the lower reaches of the Kamweti River (Kamweti-5 and Thiba), where pollution-tolerant taxa appeared, most likely owing to increased agricultural activity along the corridor that raises nutrient and organic-matter loads and favours eutrophic assemblages — a longitudinal pattern consistent with the river continuum concept (Vannote *et al.*, 1980).

The Njoro River was dominated by pollution-tolerant species such as *Gomphonema parvulum*, *Gomphonema subclavatum*, *Nitzschia palea*, *Navicula cryptocephala* and *Pinnularia subcapitata*, which are widely used as indicators of eutrophic and organically polluted waters with high concentrations of nutrients, organic matter and ions from anthropogenic sources (Lange-Bertalot, 1979; Kelly *et al.*, 2008). *Gomphonema parvulum* and *Nitzschia palea*, in particular, are commonly reported from river systems degraded by industrial effluent, agricultural runoff and domestic waste, and their dominance in the Njoro River reflects the cumulative effects of these activities in the catchment (Mathooko, 2001; Merimba, 2020). Such anthropogenic pressure is known to restructure diatom communities in favour of opportunistic, tolerant taxa at the expense of sensitive ones, impairing ecological integrity (Bere and Tundisi, 2010; Stevenson *et al.*, 2010). The contrast between the relatively pristine assemblages of the Kamweti River and the pollution-tolerant communities of the Njoro River highlights the sensitivity of epilithic diatoms to environmental gradients and their value as reliable indicators of water quality in lotic ecosystems.

Diatom diversity and richness

Diatom diversity and richness in the two rivers were shaped by the longitudinal gradient and by anthropogenic disturbance. Assemblages in the Kamweti River were significantly more diverse than those in the Njoro River, reflecting its higher ecological quality. The comparatively low species number at the pristine upstream site (Kamweti-1) is consistent with the tendency of oligotrophic headwaters to support fewer but more specialised taxa, owing to limited nutrients and habitat heterogeneity (Passy, 2007; Van Dam *et al.*, 1994). The increase in diatom diversity and richness from mid- to downstream sites in the Kamweti River accords with the river continuum concept, which predicts that diversity increases as resources and habitat complexity increase downstream (Vannote *et al.*, 1980; Allan and Castillo, 2007).

In the Njoro River, diversity was highest at the upstream sites (Sigotik, Little Shuru and Kenyatta) and declined markedly towards the downstream sites (Ngata and Mogoon), reflecting the cumulative stress of anthropogenic inputs. Such loss of diversity and richness is typical of polluted systems, where eutrophication and organic pollution promote a few pollution-tolerant species (Lange-Bertalot, 1979; Kelly *et al.*, 2008). Comparable patterns have been reported in impacted tropical rivers, where downstream reaches become dominated by tolerant diatoms with reduced richness and evenness (Bere and Tundisi, 2010), and a clear inverse relationship between diatom diversity and pollution has been demonstrated in rivers under agricultural and urban pressure (Wang *et al.*, 2014). Thus the two rivers illustrate how moderate nutrient enrichment can sustain near-natural diversity in the Kamweti River, whereas the excessive nutrient and organic loading of the Njoro River drives ecological degradation and loss of diatom diversity.

Diatom indices for water quality assessment

The DPS, TDI and SI were all higher in the Njoro River than in the Kamweti River, indicating greater nutrient enrichment and organic pollution in the former. High index values reflect a shift in assemblage composition towards pollution-tolerant species under increasing eutrophication. The literature consistently reports that rising nutrient loads increase diatom indices as tolerant species outcompete sensitive ones (Kelly *et al.*, 2008). Both rivers showed higher index values downstream, reflecting the cumulative effect of agricultural, domestic and industrial inputs, and previous studies have similarly reported increases in TDI and saprobity in nutrient-enriched rivers (Triest *et al.*, 2012; Bere and Tundisi, 2010). High saprobity has also been linked to the dominance of taxa such as *Nitzschia*, a known indicator of organic pollution (Lange-Bertalot, 1979).

Conclusions and recommendations

The two rivers differed markedly in ecological condition: the Kamweti River was generally of good quality, whereas the Njoro River was heavily impacted, mainly by nutrient enrichment and organic waste. Pollution-tolerant diatoms increased in the downstream reaches of both rivers, indicating a direct link with human activity and confirming the reliability of diatoms as indicators of river health. Improved management is recommended, including the control of agricultural runoff and wastewater discharge, the restoration of riparian buffers and the adoption of sustainable land-management practices, prioritising the Njoro River and the downstream reaches of both rivers. Future work should incorporate seasonal sampling and multi-metric indices to capture temporal variability and improve the accuracy of diatom-based biomonitoring.

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Temporal changes in physicochemical water quality parameters in the Ferguson's Gulf, Lake Turkana, Kenya

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Abstract

This study assessed temporal changes in selected physicochemical water quality parameters in Ferguson's Gulf, Lake Turkana, between 2021 and 2026. Temperature, dissolved oxygen and conductivity were measured at six sampling stations previously established during a 2021 survey, and the two periods were compared. Marked changes were recorded across all stations. Mean temperature declined slightly, from 30.5 °C in 2021 to 29.3 °C in 2026, whereas dissolved oxygen rose substantially, from 4.45 to 7.73 mg/L. Over the same period, conductivity fell from 3652 to 2511 µS/cm. These shifts coincide with the recent rise in the level of Lake Turkana and the associated increase in freshwater inflow, which appear to have enhanced mixing and diluted the historically saline waters of the Gulf. The findings highlight the ecological sensitivity of Ferguson's Gulf to hydrological and climatic change and underline the need for continued water quality monitoring to support sustainable fisheries management and climate-resilient conservation of the Gulf.

Keywords: Lake turkana, ferguson's gulf, water quality, temporal changes

Introduction

Lake Turkana lies in the Kenyan Rift Valley, predominantly in the arid northern part of the country, with its northern end extending into Ethiopia, making it a transboundary resource shared between the two nations. Approximately 260 km long and covering a surface area of about 6405 km² within a catchment of roughly 130,860 km², it is the world's largest permanent desert lake (Gownaris *et al.*, 2017). The lake occupies a closed basin with no surface outlet, so that inflowing water is lost almost entirely through evaporation (Avery, 2012). Its hydrology, ecology and fisheries are driven largely by the Omo River, which rises in the Ethiopian highlands and supplies more than 90% of the total inflow, with the Turkwel and Kerio rivers and several seasonal streams contributing the remainder (Gownaris *et al.*, 2017; Avery, 2012).

Ferguson's Gulf, located along the western shore of Lake Turkana, is a shallow, enclosed bay separated from the main lake by the Longech/Namukuse spit. The spit shelters the Gulf from the strong wave action of the open lake and restricts water exchange, so that the Gulf exhibits ecological characteristics that differ markedly from those of the open lake. Earlier limnological work showed that the Gulf supports plankton communities and primary productivity several-fold higher than the open lake (Källqvist *et al.*, 1988), and it has long been recognised as the most productive fishing ground and an important fish breeding area in Lake Turkana (Hopson, 1982; Kolding, 1993; Gownaris *et al.*, 2017). During the productive period of the mid-1970s the Gulf supported exceptionally high tilapia catches, contributing to a lake-wide yield of the order of 16 000 t in 1976 (Kolding, 1993).

Despite its ecological and economic significance, Ferguson's Gulf remains highly sensitive to hydrological and climatic change. Because it is shallow and only narrowly connected to the main lake, the Gulf has dried out completely on several occasions in recent decades during periods of low lake level, and it is among the first parts of the lake to be exposed when water levels fall (Keyombe *et al.*, 2021; Avery, 2012; Gownaris *et al.*, 2017). Changes in water quality variables such as dissolved oxygen, conductivity and temperature may therefore have important consequences for fisheries productivity and ecosystem stability in the Gulf.

In recent years, however, the water levels of Lake Turkana and other Rift Valley lakes in Kenya have risen markedly, producing environmental and ecological changes within the lake basins and affecting local communities (Herrnegger *et al.*, 2021; Byrne *et al.*, 2024). The rise has been attributed mainly to increased inflow driven by higher rainfall in the Ethiopian highlands and across the wider East African region, linked to anomalously warm sea-surface temperatures in the western Indian Ocean (Herrnegger *et al.*, 2021; Liao *et al.*, 2025; Byrne *et al.*, 2024). At Lake Turkana the rising waters have disrupted nearshore habitats, altered the shoreline and inundated settlements and fishing infrastructure, particularly around Ferguson's Gulf. Such hydrological change is likely to alter the physicochemical characteristics of the lake, especially dissolved oxygen and conductivity, yet recent measurements from the Gulf remain scarce.

Although several studies have been undertaken to examine the limnology and fisheries aspects of Lake Turkana, there exists limited information on temporal changes in water quality parameters in the Ferguson's Gulf. This study aimed to compare selected aspects of physicochemical parameters between 2021 and 2026 in order to evaluate temporal trends and provide insights into the ongoing environmental changes within the Gulf ecosystem in a span of five years.

Materials and methods

Study area

The study was carried out in Ferguson's Gulf, a shallow embayment on the western shore of Lake Turkana (Fig. 1). The Gulf is separated from the main lake by the Longech/Namukuse spit to the east and adjoins the town of Kalokol to the west. Characterised by shallow water, limited circulation and high biological productivity, its semi-enclosed nature makes it highly responsive to environmental fluctuations and to changes in lake hydrology.

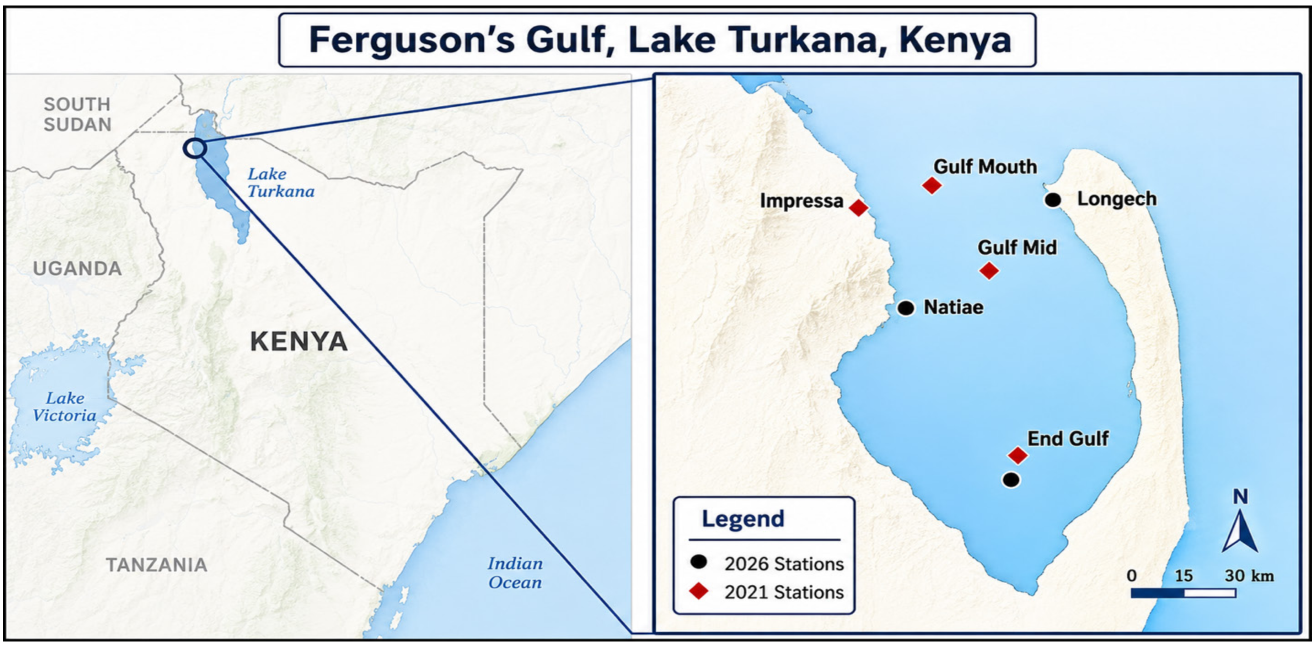

Figure 1. Map of Lake Turkana and the sampling stations in Ferguson's Gulf.

Data collection and analysis

Sampling was undertaken at the six stations established during the 2021 survey of the Gulf, namely the Gulf mouth, Natirae, Longech, the Impressa docking site, the middle of the Gulf and its southern end (Keyombe *et al.*, 2021). Temperature, dissolved oxygen and conductivity were measured in situ with a SAIV CTD profiler (model 208), whereas the 2021 survey had used a YSI Professional Plus handheld multiparameter meter. Water samples for nutrient analysis were collected in sample bottles and kept in a cool box for transport to the KMFRI laboratory.

Logged data were retrieved from the instrument using the Minisoft SD200W software and analysed in Pasgear2 and Microsoft Excel.

Results

Considerable temporal variation was observed in temperature, dissolved oxygen and conductivity across all six sampling stations between 2021 and 2026 (Table 1).

Temperature

Water temperature declined slightly at most stations (Fig. 2). The largest changes occurred in the middle of the Gulf (-2.32 °C) and at Natirae (-3.38 °C), and the smallest at Impressa (-0.28 °C) and the Gulf mouth (+0.15 °C). Overall, the mean temperature fell from 30.53 °C in 2021 to 29.28 °C in 2026.

Table 1. Temporal changes in selected water quality parameters in Ferguson's Gulf between 2021 and 2026. 2021 values are from Keyombe *et al.* (2021). DO, dissolved oxygen (mg/L); EC, electrical conductivity (µS/cm); T, temperature (°C).

Station	2021			2026			Change		
	DO	EC	T	DO	EC	T	DO	EC	T
Impressa	3.63	4045	30.1	8.38	2240	29.82	+4.75	-1805	-0.28
Gulf mouth	2.2	3354	29.1	7.05	2602	29.25	+4.85	-752	+0.15
Longech	4.33	3389	29.5	7.09	2497	29.1	+2.76	-892	-0.4
Gulf mid	5.16	3560	31.6	8.1	2609	29.28	+2.94	-951	-2.32
Natirae	5.03	3998	32.8	9.06	2467	29.42	+4.03	-1531	-3.38
Southern end	6.37	3563	30.1	6.72	2653	28.79	+0.35	-910	-1.31
Total	**4.45**	**3652**	**30.53**	**7.73**	**2511**	**29.28**	**+3.28**	**-1141**	**-1.25**

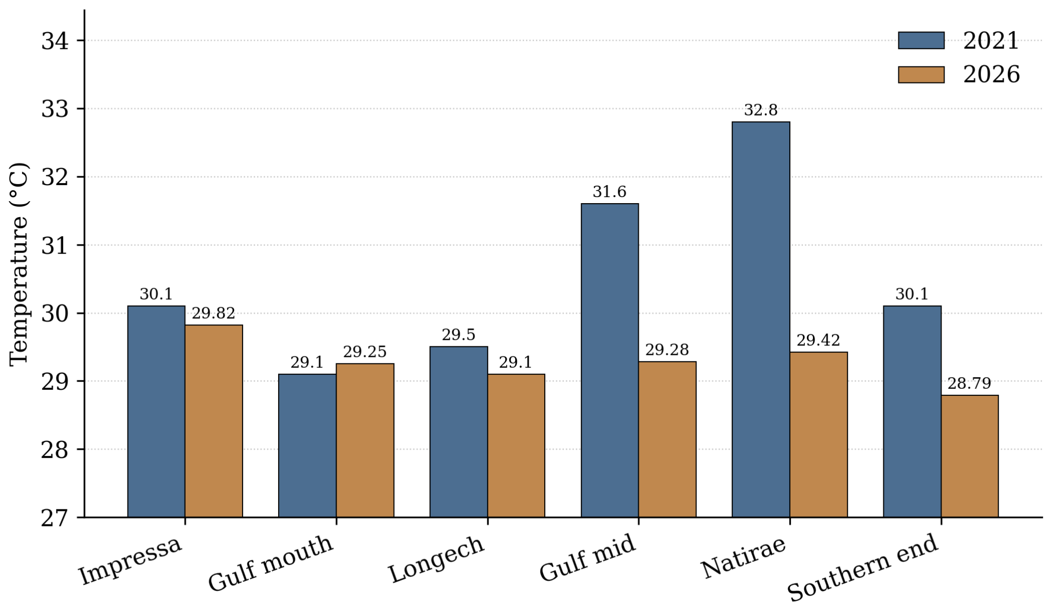

Figure 2. Temperature variation across sampling stations in Ferguson's Gulf between 2021 and 2026.

Dissolved oxygen

Dissolved oxygen increased at every station between 2021 and 2026 (Fig. 3). The overall mean rose from 4.45 mg/L in 2021 to 7.73 mg/L in 2026, an increase of 3.28 mg/L in half a decade. The greatest increases were recorded at the Gulf mouth (+4.85 mg/L) and Impressa (+4.75 mg/L), and the smallest at the southern end of the Gulf (+0.5 mg/L).

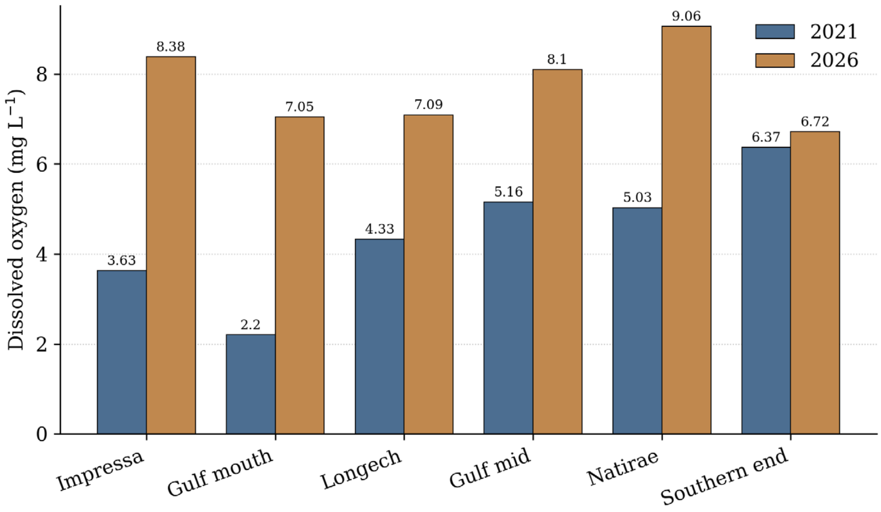

Figure 3. Dissolved oxygen variation across sampling stations in Ferguson's Gulf between 2021 and 2026.

Conductivity

Conductivity decreased substantially at all stations between 2021 and 2026 (Fig. 4). The overall mean fell from 3652 µS/cm in 2021 to 2511 µS/cm in 2026, a decline of 1141 µS/cm. The largest decreases occurred at Impressa (-1805 µS/cm) and Natirae (-1531 µS/cm), and the smallest at the Gulf mouth (-752 µS/cm).

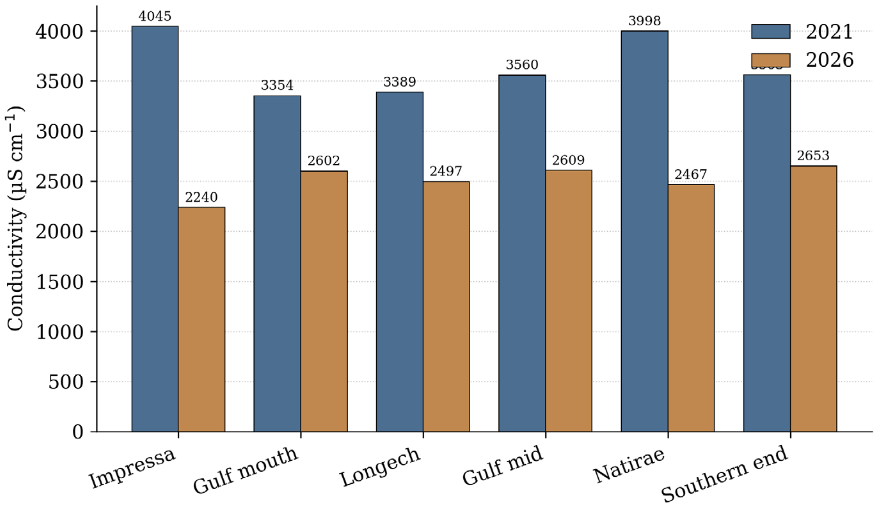

Figure 4. Variation in conductivity across sampling stations in Ferguson's Gulf between 2021 and 2026.

Discussion

The present study revealed considerable temporal change in the physicochemical characteristics of Ferguson's Gulf between 2021 and 2026. These changes are consistent with the ongoing rise in the water level of Lake Turkana and other Rift Valley lakes (Herrnegger *et al.*, 2021; Byrne *et al.*, 2024). Increased inflow from the Omo River is the principal driver of the rising water levels and the associated hydrological changes within the lake (Liao *et al.*, 2025), while strong winds, wave action and the shallowness of the Gulf promote water mixing and dilution (Liao *et al.*, 2025).

Water temperature showed a slight overall decline between the two study periods, decreasing from a mean of 30.53 °C in 2021 to 29.28 °C in 2026. The reduction was more pronounced at stations in the middle section of the Gulf and may be associated with the increase in depth and volume that accompanied the rising lake levels. Greater mixing between the Gulf and the open lake is also likely to have moderated temperatures relative to the period before flooding, when the Gulf was shallower. Temperature is an important ecological factor that influences dissolved oxygen solubility, metabolic rate and primary productivity in tropical lake ecosystems.

A substantial increase in dissolved oxygen across all sampling stations was one of the most notable findings of this study, with the overall mean rising by about 3.28 mg/L between 2021 and 2026. This increase is most plausibly linked to enhanced water mixing and improved connectivity between the Gulf and the open lake as water levels rose, which would promote circulation and the replenishment of oxygen within the previously shallower and more isolated Gulf. Greater dilution of organic matter and reduced stagnation associated with higher Omo River inflow may also have lowered oxygen demand. Because dissolved oxygen governs the distribution, survival and productivity of fish and other aquatic organisms, the higher concentrations recorded in 2026 are likely to be ecologically beneficial for the Gulf fishery, at least in the short term.

Conductivity declined markedly, from an overall mean of 3652 µS/cm in 2021 to 2511 µS/cm in 2026. Lake Turkana is naturally alkaline and saline, with characteristically high conductivity maintained by intense evaporation within its closed basin (Källqvist *et al.*, 1988). The decline recorded here is therefore consistent with dilution of the lake water by increased freshwater inflow, which lowers the concentration of dissolved salts and ions. A sustained reduction in conductivity could influence the distribution and survival of plankton and fish species adapted to the lake's alkaline conditions, and warrants continued monitoring.

Conclusions and recommendations

This study demonstrates clear temporal change in the physicochemical conditions of Ferguson's Gulf, driven largely by the ongoing rise in the level of Lake Turkana. The variation among stations highlights the spatial heterogeneity of the Gulf: the largest changes in conductivity and dissolved oxygen occurred at stations such as Impressa and Natirae, most likely reflecting localised shoreline flooding and freshwater intrusion as water levels rose. The higher oxygen and lower conductivity recorded in 2026 may currently favour aquatic productivity and the fishery, but continued hydrological fluctuation could alter habitat structure, the shoreline ecosystem and fish breeding grounds. Continued water quality monitoring is therefore needed to clarify the long-term ecological consequences of rising lake levels and to support climate-resilient conservation of the Gulf. Seasonal sampling is also recommended, to separate short-term variability from longer-term environmental trends.

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Citizen science in water quality monitoring: a critical review of data quality, validation frameworks, and community engagement

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Abstract

Freshwater ecosystems are increasingly threatened by anthropogenic pressures and climate change, necessitating monitoring approaches that provide timely, spatially extensive, and cost-effective environmental data. Citizen science has emerged as a promising complementary approach to conventional water quality monitoring by engaging volunteers in data collection while enhancing public participation in environmental stewardship. This review critically synthesizes current knowledge on the application of citizen science in water quality monitoring, with particular emphasis on data quality, validation frameworks, technological innovations, and participant engagement. The review examines the evolution of citizen science from traditional observational programmes to technology-enabled monitoring platforms incorporating low-cost Internet of Things (IoT) sensors, smartphone applications, and GPS-enabled data collection. Despite persistent concerns regarding data accuracy, observer bias, and variable participant expertise, growing evidence demonstrates that well-designed citizen science programmes can generate data comparable in quality to professionally collected datasets when supported by standardized protocols, training, and robust quality assurance procedures. Emerging validation approaches, including statistical anomaly detection, peer verification, automated sensor calibration, and hybrid expert–citizen assessment frameworks, further enhance data reliability and scientific credibility. Beyond data generation, citizen science contributes to environmental awareness, community empowerment, and improved local resource management, although sustained participation depends on socio-economic incentives, stakeholder collaboration, institutional support, and integration into formal environmental governance. The review concludes that citizen science represents a valuable component of integrated water quality monitoring systems and recommends greater investment in digital technologies, standardized validation frameworks, and long-term programme evaluation to strengthen its contribution to sustainable freshwater management and evidence-based environmental decision-making.

Keywords: Citizen science, validation frameworks, participatory monitoring, data quality

Introduction

Freshwater ecosystems play an important ecological role through the provision of essential services, including the supply of water, habitat for aquatic animals and plants, and the cycling of nutrients (Macklin and Lewin 2015). However, they are facing ever-increasing risks due to rising land-use changes, population pressures, urbanisation and economic growth (Moshi *et al*., 2022; Kitaka *et al*., 2024). All these variables significantly change the hydrologic processes of freshwater ecosystems impacting the quality of freshwater as experienced through modified flow regimes, increased sedimentation, and additional nutrients which negatively impacting both human and ecosystem health (Njue, 2023). In many regions of the world, particularly throughout Eastern Africa, there exist weak regulatory frameworks and limited water quality monitoring capabilities; this creates major barriers to the implementation of evidence-based water resource management (Kagalou *et al*., 2012). In order to make sound, evidence-based decisions, aquatic resource managers require reliable data. However, traditional sampling methods that utilize expert-based sampling and laboratory analysis are often too costly, spatially limited, and impossible to sustain over extended time periods resulting in large gaps of critical data especially in regions experiencing resource constraints (Aura *et al*., 2021). This infers that there is need for alternative and complementary monitoring methods to be adopted to fill in the large voids of data. The development and use of citizen science via community-based programs to monitor ecosystem's status is gaining traction in most parts of the world (Requier *et al*., 2020).

Citizen science is a method of using non-professionals to aid scientists in collecting, analysing and creating knowledge through participation in the process (Hecker *et al*., 2109). This has recently become an increasingly attractive volunteer option for monitoring freshwater ecosystems. This is in line with co-management approaches granting stakeholders an opportunity to contribute towards sustainable management of freshwater resources (Obiero *et al*., 2015). This therefore allow for a greater degree of public involvement with environment stewardship (Conrad and Hilchey, 2011). Participants in citizen science projects can collect samples from areas that are difficult for researchers to reach or access, producing larger datasets than could otherwise be generated by scientific researchers alone (Welvaert and Caley, 2016). Due to this, the use of citizen science in monitoring freshwater systems has been expanded on a global scale as a viable method for increasing geographical coverage and data availability to detect environmental change (Dominguez-Rendon *et al*., 2024).

Additionally, citizen science increases statistical power and allows long-term monitoring of environmental changes, creates environmental awareness, and encourages locals to participate in resource management (Schmeller *et al*., 2009; Fraisl *et al*., 2022; Phillips *et al*., 2022). However, empirical studies based on public interaction with data collection are still under-reported in the scientific literature, with concerns about data quality being the most stated barrier (Catlin-Groves, 2012). Citizen inexperience on appropriately scientific data collection can actually bias or skew data, hampering data quality and reliability (Flanagin and Metzger, 2008). This is closely related to the issue of the validation process, which determines the reliability and usefulness of citizen-generated data. While some studies highlight the successful incorporation of validation methods such as calibration, cross-checking, and protocol standardisation, others point out the lack of standardised frameworks for assuring data dependability (Welvaert and Caley, 2016).

The extent to which individuals have participated in citizen science programs is a critical factor influencing the success of these types of programs, with all aspects of citizen science having a direct impact on the quality of data generated and the overall sustainability of the project. Greater participation may extend the spatial and temporal scope of the data collection. However, this does not guarantee the accuracy of the data being produced. Furthermore, there is still a lack of research regarding how citizen science will contribute to long-term environmental management and policy (Phillips *et al.*, 2022; Dominguez-Rendon *et al.*, 2024). Research to-date has typically examined data quality, data validation processes and drivers of participation separately, rarely examining them in conjunction, and especially true in the contexts of monitoring of freshwater and lakes (Jollymore *et al.*, 2017). A clear example of the above statement is in East Africa, where standardised protocols for citizen-based monitoring are still not fully developed (Requier *et al.*, 2020). Thus, this paper provides a critical review of the literature regarding citizen science in relation to the monitoring of water quality through the examination of three dimensions: data quality and reliability, validation processes and involvement of participants to provide a comprehensive synthesis of the strengths and weaknesses and future opportunities for the integration of citizen science into the environmental monitoring and management framework.

Materials and methods

The study utilised a systematic review using the PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses) framework as a guide to provide transparency, reproducibility, and rigor in the identification and synthesis of literature pertaining to citizen science and water quality monitoring (Fig. 1). To ensure a comprehensive synthesis of citizen science (CS) applications in Lake Victoria, Kisumu, a systematic literature search was conducted across multiple electronic databases, including Google Scholar, Scopus and Web of Science (WoS), with a timeframe of 2008 to 2026 to capture the evolution from early community-based monitoring to modern technology-enabled citizen sensor systems. To guarantee thorough coverage, more research was conducted by tracking the backward and forward citations of major papers. Search queries utilised keyword combinations such as "water quality" and "citizen science", "water quality" and "community monitoring", "data validation" and "citizen science", "data accuracy" and "citizen science", "water quality" and "participatory monitoring". These concepts are combined using Boolean operators and wildcards

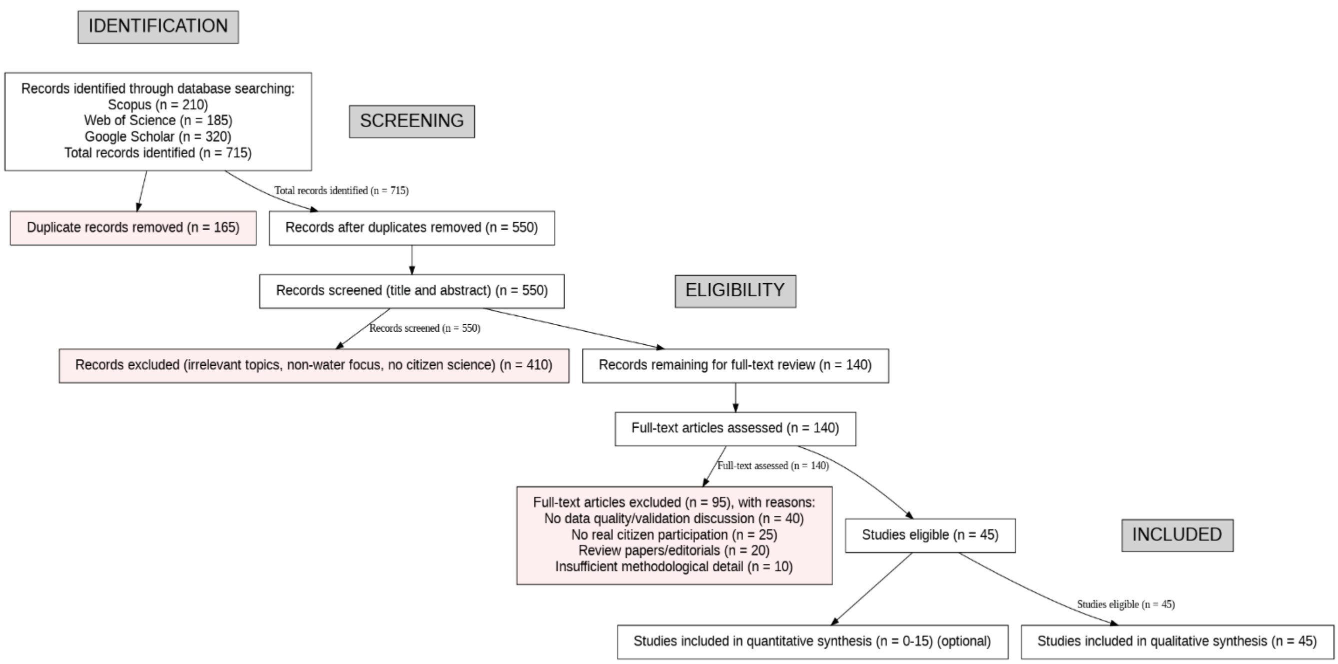

Figure 1. PRISMA flow diagram for the review of data quality, validation frameworks, and community engagement.

Results and discussion

This paper systematically reviewed the application of citizen science in water quality monitoring by accessing data quality, validation processes and participant involvement (Table 1 and 2). The results indicate that most citizen science projects for water quality monitoring follow a contributory model in which citizens participate primarily in data collection rather than research design or decision-making (Shirk *et al.*, 2012).

The findings further emphasize Citizen Science's expanding importance as a complementing technique to traditional freshwater monitoring, particularly in resource-constrained contexts. By combining information from multiple research, this review reveals that the usefulness of citizen science in lake monitoring varies and is influenced by the combination of data quality, validation processes, participant involvement, and environmental impact (Table 2).

Table 1. Key summary of studies on Kenyan water catchments' monitoring challenges.

Study area	Parameters/Methods	Findings	Reference
Afrotropical riverine ecosystems	Community-Based Index of Ecosystem Integrity (CIEI)	The CIEI originality shown herein could be utilized to identify regions where conservation should be prioritized, with a broader application for snapshot studies in marginalized areas and little costs.	(Aura *et al.*, 2021)
Tropical mountainous landscapes of Kenya	Participatory Monitoring & Valuation of hydrological systems	Citizen science can effectively enhance hydrological monitoring and research in distant tropical mountainous locations.	(Njue, 2023)
East African rivers	Community-based collaboration for environmental monitoring	Improving citizen science in system health monitoring is the future management method for raising awareness, closing the knowledge gap between users, policymakers, and scientific results, and interpreting social-ecological data.	(Kitaka *et al.*, 2024)
General catchments	Empowering Citizen Scientists to improve water quality	We are all citizen scientists and change agents," yet the true potential of CS data for water resource management has yet to be realized.	(Warner *et al.*, 2024)

Table 2. Synthesis of citizen science studies in water quality monitoring (n = 45).

Category	Sub-category	No. of Studies (n)	%	Key Findings
Geographic Distribution	Europe & North America	24	53%	Dominant research regions with well-established frameworks
	Asia	9	20%	Growing adoption, variable validation rigor
	Africa	6	13%	Limited studies, weak standardisation
	Latin America	6	13%	Emerging applications, strong community engagement

Category	Sub-category	No. of Studies (n)	%	Key Findings
Water Body Type	Rivers/Streams	30	67%	Most monitored systems
	Lakes/Wetlands	15	33%	Underrepresented, especially in developing regions
Citizen Science Approach	Contributory	28	62%	Participants mainly collect data
	Collaborative	11	24%	Partial involvement in analysis/ design
	Co-created	6	13%	Full engagement, but rare
Parameters Measured	Physical (Temp, Turbidity)	38	84%	High reliability across studies
	Chemical (pH, DO)	32	71%	Moderate to high accuracy
	Nutrients (N, P)	18	40%	High variability, lower reliability
	Biological (Macroinvertebrates, Diatoms)	14	31%	Requires expertise, prone to bias
Data Quality Level	High (comparable to professional)	19	42%	Strong protocols and training are present
	Moderate	17	38%	Some inconsistencies observed
	Low	9	20%	Weak protocols, minimal validation
Validation Methods Used	Calibration	25	56%	Improves measurement accuracy
	Cross-checking with experts	21	47%	Enhances credibility
	Replicate sampling	18	40%	Reduces random error
	Standardized protocols	29	64%	Most effective validation approach
	No clear validation	8	18%	Associated with low data quality
Participant Training	Structured training provided	27	60%	Strongly linked to high-quality data
	Minimal/no training	18	40%	Associated with higher variability
Participant Engagement	Short-term projects	20	44%	Limited consistency
	Long-term engagement	25	56%	Improved data reliability over time

Citizen science conceptual foundations for environmental monitoring

Citizen science refers to the involvement of non-professional citizens in scientific research activities, such as data collecting, analysis, and dissemination (Bonney *et al.*, 2009). Progressively, it has moved from the simple citizen scientist observations to a more complicated, technology-enabled systems including citizen sensors and crowdsourced data (Catlin-Groves, 2012; Newman *et al.*, 2012). Conventionally, freshwater monitoring was a centralized effort, limited to the government agencies and the academic institutions. However, the twenty-first century has seen a paradigm shift from contributing models to co-developed models, where the local communities select research subjects (Warner *et al.*, 2024). This process has been accelerated in Lake Victoria Basin by the deficiency of centralized systems to give the real-time data on localized hotspots.

The literature exhibits a steady development from passive observation to technology-enabled systems. The acceptance of citizen sensors, low-cost internet of things (IoT) devices that give continuous streams of temperature and conductivity data effectively changes the lake's coastline into a dispersed digital laboratory(Kitaka *et al.*, 2024). It is important to note that this transition is not merely technological but also institutional, it entails the formal inclusion of Water Resource Users Associations (WRUAs) into the scientific process, essentially transforming the local people into guardians of the ecosystem (Aura *et al.*, 2021; Kitaka *et al.*, 2024). Citizen science is increasingly being used in environmental monitoring to conserve biodiversity, analyze water quality, and monitor ecosystems. Studies have showed that citizen-based monitoring can greatly increase spatial and temporal data coverage while encouraging environmental awareness and stewardship (Fraisl *et al.*, 2022; Schmeller *et al.*, 2009). In freshwater systems, including rivers and lakes, citizen science has been utilized to monitor characteristic citizen science such as water quality, biodiversity, and ecosystem integrity, and its significance in assisting adaptive management and policy processes (Aura *et al.*, 2021; Fraisl *et al.*, 2022; Moshi *et al.*, 2022). However, despite the growing popularity, citizen science remains an antagonistic topic, notably in terms of scientific validity of its outputs and its inclusion into the official environmental management.

Data quality and validation frameworks in citizen science

The Internet, Web 2.0, and Social Networking technologies enable citizens to actively participate in citizen science initiatives by submitting data to scientific programs via the internet. However, contributors' inadequate training, experience, and skill might result in low-quality, inaccurate, or even malicious data being shared (Foster-Smith and Evans, 2003). As a result, the scientific community frequently considers citizen science data unworthy of use in real scientific study, which contributes to low volunteer retention rates (Hunter *et al.*, 2013). The lack of scientific method and the use of non-standardized, poorly conceived data collection methods (Paulos, 2009; Silvertown, 2009), this frequently leads to incomplete or inaccurate data. Also, volunteers' lack of dedication to collecting field data can result in gaps in data throughout time and space. as a result of these concerns, many members of the scientific community believe that citizen science data is unworthy of real scientific investigation (Foster-Smith and Evans, 2003; Galloway *et al.*, 2006; Delaney *et al.*, 2008). While scientists are frequently suspicious of unpaid volunteers' capacity to provide correct datasets, a growing body of articles clearly demonstrates that various forms of citizen-science programs can produce data with accuracy comparable to or greater than that of professionals (Kosmala *et al.*, 2016).

However, recent empirical research contradicts this criticism of data incredibility, revealing that volunteer-led monitoring can attain high levels of accuracy, especially when compared to professional laboratory baselines. Data cleansing and quality improvement technologies can increase data quality to a certain extent. For example, simple techniques can be used to validate data input (such as grammar, format, and values) by comparing it against schemas. Comparing data sets to alternate sources or historical trends allows for a more sophisticated assessment of data quality (Fig. 2). However, these methods are limited if there is no comparable or longitudinal data for trend analysis (Hunter *et al*., 2013). An increasing amount of research demonstrates that citizen scientists can create high-quality data under suitable settings, particularly when backed by defined methods, training, and quality control mechanisms (Dickinson *et al*., 2010; Fraisl *et al*., 2022). Empirical investigations strongly support this possibility. For example Moshi *et al*. (2022) showed that citizen scientists were able to properly detect nutrient pollution patterns in Lake Tanganyika, demonstrating that non-expert contributions can provide scientifically reliable results. Similarly, (Aura *et al*., 2021) created a community-based index for ecosystem integrity in Afrotropical riverine systems, demonstrating the potential of citizen-generated data to inform ecological studies. This dependability is not accidental, it is the product of established protocols and multi-layered validation systems. The integrity of citizen science data is dependent on a mix of expert verification, rigorous training, and the use of statistical bias correction variables As (Conrad and Hilchey, 2011; Jollymore *et al*., 2017).

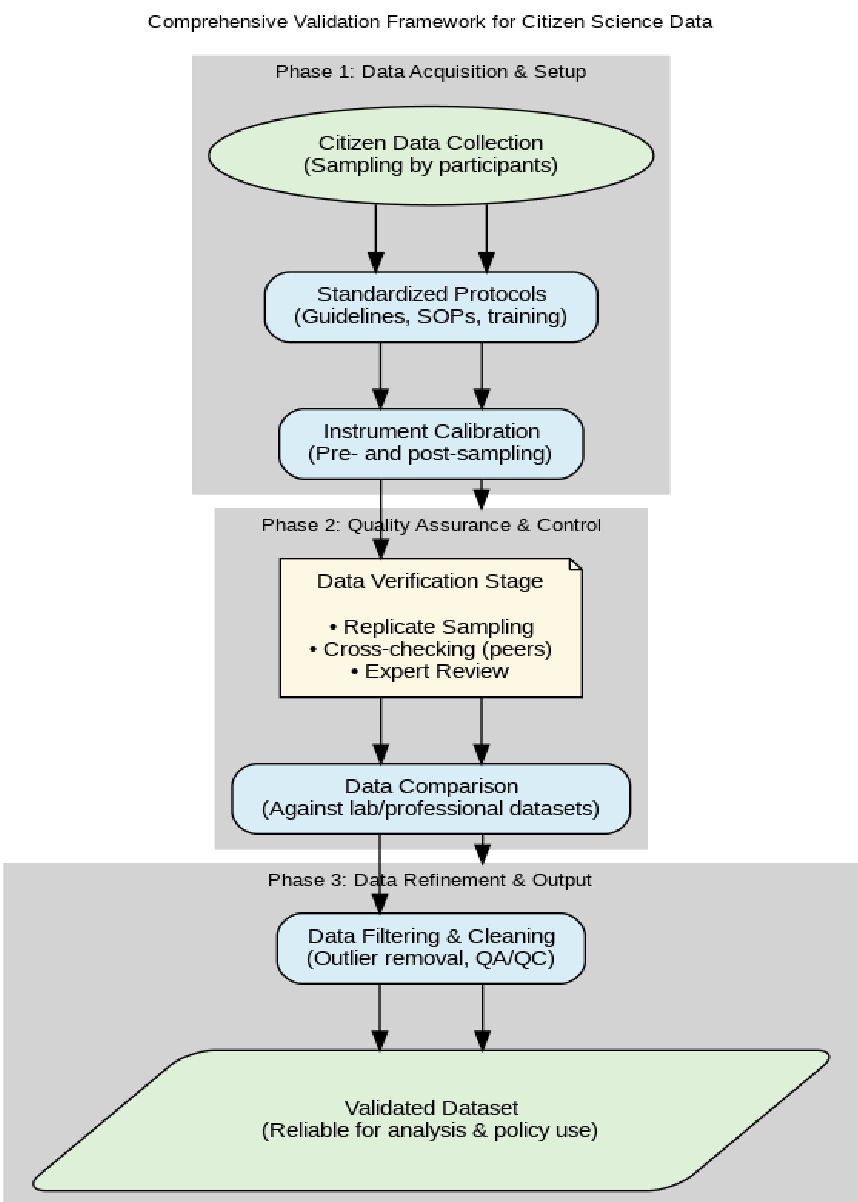

Figure 2. Comprehensive validation frameworks for citizen science in data quality and validation frameworks.

Warner *et al*. (2024) argued that technology-enabled validation, such as GPS-tagged photography for species identification or automated colorimetry, eliminates much of the human subjectivity that previously hampered citizen data integration into formal regulatory frameworks. Furthermore, as Fraisl *et al*. (2022) pointed out, statistical anomaly detection and peer-to-peer verification enable the fast identification of inaccuracies, ensuring that the generated datasets are suitable for academic analysis and policy action. Several approaches for assessing the quality of citizen-collected data have been proposed, including employing a majority decision or control group, using reputation systems, and assessing the trustworthiness of contributed data by analyzing user mobility patterns and past quality (Meek *et al*., 2014) (Table 3). Study by Storey *et al*. (2016) and inputs from community groups identified many sorts of forms that improved data quality, including quality assurance (data checking and discussion with the groups as data were submitted). These types of support have been shown to improve the quality of volunteer data. Recent evaluations underline the importance of organized validation systems that include technology tools, established protocols, and institutional support (Kitaka *et al*., 2024).

Furthermore, DPSIR (Drivers-Pressures-State-Impact-Response) model have been developed to contextualize citizen-generated data within the larger environmental management systems (Kagalou *et al*., 2012). Without such integrative techniques, citizen science's full potential for informing the environmental decision-making may be missed. When volunteers are given tools that are not standardized, errors definitely increase. However, the success attainment of macroinvertebrate-based indices, such as the SASS5 and the CIEI tools, demonstrates that when biological indicators are translated into more simplified graphical keys, the results are scientifically proven. As a result, the focus must now move from whether citizen scientists can collect data to how scientists can improve the interfaces they use. Storey *et al*. (2016) suggested that community monitoring could benefit from a website for data sharing among monitoring groups. This could be through training and sharing ideas, questions, and observations.

Table 3. Relationship between key factors and data quality.

Factor	Impact on Data Quality	Evidence from Review
Training	Strong positive	Trained participants consistently produced accurate data
Validation protocols	Critical	Absence leads to unreliable datasets
Parameter type	Variable	Simple parameters more reliable than complex ones
Participant experience	Positive	Long-term involvement improves consistency
Equipment quality	Moderate	Low-cost tools increase variability

Participant involvement in citizen science

Citizen scientist participation is an essential foundation of research in determining its success and sustainability and (Weeser *et al*., 2018) highlights that Citizen science programs might have encouraged local people to form opinions on local policies and provide remedies to environmental concerns, which they could then transmit to the government (Table 4). Given enough professional support, community-based water monitoring can give data that is credible enough to supplement professionally obtained data while also increasing community members' opportunities, confidence, and abilities in making freshwater decisions (Storey *et al*., 2016).

Table 4. Synthesis of citizen science studies in water quality monitoring (n = 45).

Category	Sub-category	No. of Studies (n)	%	Key Findings
Geographic Distribution	Europe & North America	24	53%	Dominant research regions with well-established frameworks
	Asia	9	20%	Growing adoption, variable validation rigor
	Africa	6	13%	Limited studies, weak standardisation
	Latin America	6	13%	Emerging applications, strong community engagement
Water Body Type	Rivers/Streams	30	67%	Most monitored systems
	Lakes/Wetlands	15	33%	Underrepresented, especially in developing regions

Category	Sub-category	No. of Studies (n)	%	Key Findings
Citizen Science Approach	Contributory	28	62%	Participants mainly collect data
	Collaborative	11	24%	Partial involvement in analysis/design
	Co-created	6	13%	Full engagement, but rare
Parameters Measured	Physical (Temp, Turbidity)	38	84%	High reliability across studies
	Chemical (pH, DO)	32	71%	Moderate to high accuracy
	Nutrients (N, P)	18	40%	High variability, lower reliability
	Biological (Macroinvertebrates, Diatoms)	14	31%	Requires expertise, prone to bias
Data Quality Level	High (comparable to professional)	19	42%	Strong protocols and training are present
	Moderate	17	38%	Some inconsistencies observed
	Low	9	20%	Weak protocols, minimal validation
Validation Methods Used	Calibration	25	56%	Improves measurement accuracy
	Cross-checking with experts	21	47%	Enhances credibility
	Replicate sampling	18	40%	Reduces random error
	Standardized protocols	29	64%	Most effective validation approach
	No clear validation	8	18%	Associated with low data quality
Participant Training	Structured training provided	27	60%	Strongly linked to high-quality data
	Minimal/no training	18	40%	Associated with higher variability
Participant Engagement	Short-term projects	20	44%	Limited consistency
	Long-term engagement	25	56%	Improved data reliability over time

Engagement has several components, for instance motivation, participation levels, retention rate, and diversity (Vasiliades *et al*., 2021). The assumption from Capdevila *et al*. (2020) is that citizens have enormous potential to expand temporal and spatial data availability and so bridge the data gap that the international community faces. However, citizen-derived data may be selective and biased, highlighting the need to develop design principles for successful citizen science projects. To offer a focus for this exploration, Kim *et al*. (2011) created and launched Creek Watch, an iPhone application and website that allows volunteers without high level of education to report information about streams to assist water management initiatives. Nonetheless, the locals' past abilities and expertise, together with sufficient training and support, enabled them to collect meaningful data on water quality and changes (Christofalos *et al*., 2022). Many authors also underlined that water quality and monitoring projects improved residents' understanding and appreciation for the environment (Capdevila *et al*., 2020). Individuals participate in citizen research for many reasons, including scientific curiosity, environmental concern, learning and social connection (Bonney *et al*., 2009). According to this research, involvement often goes down over time due to lack of consistent incentive, feedbacks, and institutional support (Phillips *et al*., 2022).

When citizens see that their data is neglected by institutional authorities, participation fatigue sets in, and the monitoring program do not succeed (Conrad and Daoust, 2008). Furthermore, (Capdevila *et al.*, 2020) believes that social characteristics may have played an important role in encouraging citizens to participate in citizen scientific programs. First, societal commitment may have motivated citizens to act as scientists. Storey *et al.* (2016) discovered that volunteers felt more confident in their data, which enhanced their motivation to continue, interactions with their local government, especially when they believed the council genuinely cared about their results since they were strongly motivated by the desire to make a difference. Another important feature is inclusivity, factors such as education, resource availability, and cultural environment that influence citizen's participation. Participatory monitoring research projects in underdeveloped countries, particularly Kenya, have shown the ability to empower the community and raise environmental awareness (Njue, 2023; Warner *et al.* 2024). In another study, the literature highlighted the importance of explicit knowledge and expertise in the field of study, for example Capdevila *et al.*, (2020) illustrates how highly educated volunteers with competence in research and management can boost a project's production.

Sierra Leone and Zambia has also included citizen-generated data in their most recent SDG indicator 6.3.2 report to the United Nations Environment Programme (Warner *et al.*, 2024). The viability of these monitoring projects is highly dependent on the dynamic citizen scientist of participant involvement and the socio-psychological factors of volunteer retention. According to Fraisl *et al.*, (2022), while initial engagement may be fuelled by altruistic environmental concerns, long-term commitment requires a livelihood-centric incentive however, this long-term engagement remains difficult. The fact that there are such severe water quality data gaps worldwide (Organization, 2021) and that national governments struggle to gather enough data, gives an opportunity for citizen scientist-generated data to fill this void (Warner *et al.*, 2024).

Conclusion and recommendations

This review demonstrates that citizen science has emerged as a valuable and increasingly reliable approach for complementing conventional water quality monitoring. Evidence from the literature indicates that most citizen science programmes adopt a contributory model, whereby scientists design monitoring frameworks while volunteers participate primarily in data collection. When supported by standardized protocols, appropriate training, and robust quality assurance and validation procedures, citizen-generated data can provide scientifically credible information for environmental assessment and management. Beyond improving spatial and temporal monitoring coverage, citizen science enhances environmental awareness, promotes stakeholder participation, and strengthens community stewardship of freshwater resources, thereby contributing to more inclusive and sustainable water governance.

To maximize the potential of citizen science, future initiatives should move beyond contributory approaches towards more collaborative and co-created monitoring frameworks that actively involve communities in project design, data interpretation, and decision-making. Greater investment is needed in digital innovations, including low-cost Internet of Things (IoT) sensors, remote sensing technologies, GPS-enabled mobile applications, and biological assessment tools such as the South African Scoring System version 5 (SASS5) macroinvertebrate index, to improve data accuracy, expand monitoring capabilities, and enable near real-time environmental surveillance. Additionally, harmonized validation frameworks, long-term participant training, institutional support, and integration of citizen-generated data into national water quality monitoring programmes are essential to enhance data credibility, strengthen evidence-based decision-making, and ensure the long-term sustainability of citizen science as a component of integrated freshwater resource management.

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