Egyptian Journal of Aquatic Biology & Fisheries
Zoology Department, Faculty of Science,
Ain Shams University, Cairo, Egypt.
ISSN 1110 – 6131
Vol. 29(5): 1347 – 1369 (2025)
www.ejabf.journals.ekb.eg

Seasonal Monitoring of the River Nile in the Area of Greater Cairo Using a Combined Approach of Certain Chemical Criteria and Macroinvertebrates Matrices

Hanaa M.M. El-Khayat[1], Mohamed R. Habib[2], Wafaa A. Mohammed[1], Khaled M. Zayed[2] and Sara S.M. Sayed[1*]

[1] Environmental Research Department, Theodor Bilharz Research Institute, Giza 12411, Egypt
[2] Medical Malacology Department, Theodor Bilharz Research Institute, Giza 12411, Egypt
[*]**Corresponding Author:** eco_gut@yahoo.com

ARTICLE INFO

Article History:
Received: June 8, 2025
Accepted: Sep. 10, 2025
Online: Sep. 26, 2025

Keywords:
The River Nile,
Water quality,
Metals,
Macroinvertebrates,
FFGs,
Indices

ABSTRACT

Macroinvertebrates are valuable surrogates in reflecting the health of river streams through their abundance, taxa richness (TR), and functional feeding groups. The present work aimed at monitoring the water quality (WQ) of the River Nile in the greater Cairo area, the most crowded with multiple human activities, using a combined approach of some physical parameters, metals, and macroinvertebrates matrices during four successive seasons. Results showed that all physical water parameters and metals were approximately at the normal levels (except temperature & Cu). The highest TR was in spring and was consistent with the low average of organic pollution determined by the Hilsenhoff biotic index (HBI), reflecting fair WQ. In contrast, the lowest abundance during winter was consistent with the highest HBI, indicating a poor WQ. Multimetric macroinvertebrates index flanders showed poor WQ during winter, while moderate WQ was observed during the rest three seasons. Filtering collectors and scrapers were the predominant functional feeding groups (FFGs), then predators at lower abundance. Gathering collectors were absent during winter, while shredders were absent in all seasons. The matrix plot showed a strong positive correlation between all FFGs and Fe, Zn, and Ca, while it showed a negative correlation with Cu. Analysis of stream ecological health matrices showed that the Nile stream was autotrophic during spring and winter, while a natural predator-to-prey was balanced during autumn, spring, and summer. Indices of habitat stability and filtering collectors pointed to perfect stable substrates only during autumn, spring, and summer, when most feeding groups were represented.

INTRODUCTION

The River Nile's water quality is influenced by many interventions, such as the hydrodynamic regimes regulated by the Nile barrage and land and water use, including the agricultural return, industrial, municipal, and river ship wastewaters. Due the increase in industrial, agricultural, and recreational activities and poorly constructed drainage and sewerage infrastructure, the quality of the Nile water is a severe concern **(Abdel-Satar *et al.*, 2022)**. Therefore, the regular monitoring of surface water quality from the River Nile is considered the main target of national policy these days in Egypt **(Hussein *et al.*, 2021)**.

Because of their species diversity, bottom-dwelling activity, and different sensitivity to habitat disturbance, macroinvertebrates have been chosen as bioindicators for the biological assessment of streams, and they have thus been used in short- and long-term monitoring of stream environments **(Smith *et al.*, 1999; Resh 2008; Gültekin *et al.*, 2017; Abdel Gawad, 2019)**. Functional feeding classification of aquatic organisms develops the knowledge of trophic dynamics in water systems by simplifying the benthic community into trophic guilds – functional feeding groups (FFGs) **(Cummins, 1995)**, which are related to the metabolic resources that individuals need **(Schäfer *et al.*, 2011)**. Moreover, they have been used to conceptualize community dynamics and assess ecological status **(Vannote *et al.*, 1980)**. Furthermore, knowledge of the functional groups of invertebrates is critical to understanding trophic relationships, organic matter processing, energy flow, and the management actions needed to minimize the deterioration of ecosystem functioning **(Dudgeon, 2010; Ferreira *et al.*, 2012)**. FFGs comprise scrapers that grow on the surfaces of substrates; shredders that feed on the coarse detritus, composed mainly of leaves falling from riparian vegetation, and the fine detritus, either deposited on the substrate used by gatherers or suspended in the water column consumed by filterers; and finally, live animals eaten by predators **(Merritt *et al.*, 2017)**.

The sensitivity of macroinvertebrate assemblages is contributed to their use to assess the ecological implications of heavy metals contamination in the aquatic system. Metal contamination can diminish the number of their species, and density and production may be affected **(Qu *et al.*, 2010; Abdel-Satar *et al.*, 2022)**. The impacts of heavy metals are not only on macroinvertebrate taxa **(Kiffney & Clements, 1996)** but also may affect ecosystem function through accumulated in the environment and consequently contaminating the food chains **(Qu *et al.*, 2010)**. A multimetric index describes the state of an ecosystem utilizing several individual variables (metrics). These metrics each represent a different component of ecosystem quality and are combined into one index value. Multimetric macroinvertebrate index Flanders (MMIF) is based on five indices, allowing for an adaptation of scoring criteria for each river or lake type to reflect the relative distance to reference conditions **(Gabriels *et al.*, 2006)**. Therefore, the present work aimed to evaluate the current water quality of the River Nile in the greater Cairo area using a combined approach of some metals assessment and macroinvertebrate matrices.

MATERIALS AND METHODS

Study area

This study was carried out along the River Nile within Greater Cairo. Samples were collected seasonally from July 2019 to June 2020 from 12 stations extended from El-Saff to El-Kanater cities (Fig. 1). Regarding the habitat nature of stations were as follows (Rocky stations: 1, 7, and 9), (Sandy station: 4), and (Muddy stations: 2, 3, 5, 6, 8, 10, 11, and12).

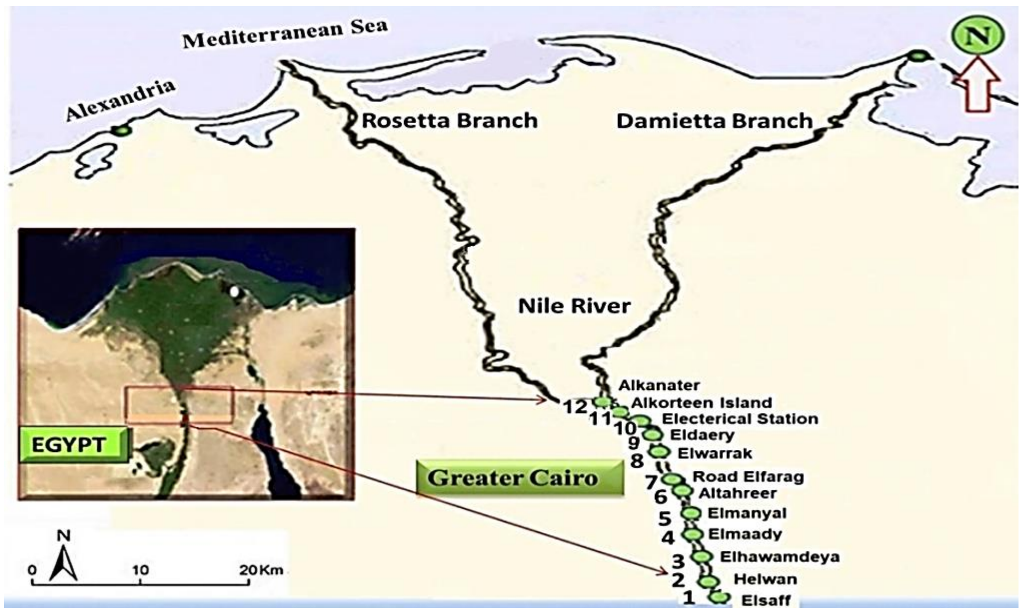

Fig. 1. Map of the study areas and sites (1-12)

Physicochemical parameters and metals

Water temperature and hydrogen ion concentration (pH) were determined using a portable pH meter (Hanna Instruments (HI) 9024). Electrical conductivity (EC) was measured using a portable conductivity meter (HI 9635). Dissolved oxygen was measured using a portable DO meter (HI 98193). These parameters were measured *in situ* at midday at 20cm below the water surface **(Jannat *et al.*, 2019)**. On the other hand, at 50cm below the surface water, samples were collected in 500ml polythene bottles that were previously washed, soaked in 5% nitric acid, and then rinsed with deionized water. About 5ml of nitric acid was added immediately to collect water samples to minimize the adsorption of heavy metals onto the bottle walls **(WHO, 2011)**. In the laboratory, 50ml of each water sample was filtered through Whatman filter paper No. 42, then through a syringe filter (0.45 µm pore size), and samples were kept in the refrigerator until analysis. For each collected water samples, zinc (Zn), iron (Fe), copper (Cu), and calcium (Ca) were analyzed using an atomic absorption spectrophotometer (GBC AVANTA™).

Macroinvertebrates samples

Seasonally, five replicates of macroinvertebrates samples were collected from each site of the investigated sites using a sieve of 300mm standard dip-net **(Barbour *et al.*, 1999)**. The samples were placed in labeled aquaria, transferred to the laboratory, sorted out, and identified based on order taxonomic level according to **Hynes (1984)** and **Pescador *et al.* (1995)**, while gastropods were sorted out, counted and identified according to **Ibrahim *et al.* (1999)**.

Assessment of water quality using macroinvertebrates

The collected macroinvertebrates were used as tools to evaluate the status of water quality along the River Nile within Greater Cairo depending on:

Taxa richness (TR)

TR refers to the total number of taxa identified from a stream. If its value is less than 8, this indicates that the stream conditions are poor, while if its value is between 8 and 15 refers that the conditions are moderate, and if it is greater than 15, then the status of the stream is good **(Bode *et al.*, 1996)**.

Hilsenhoff biotic index (HBI)

It estimates the overall tolerance of the community in a sampled area, weighted by the relative abundance of each taxonomic group and their response to organic pollution (OP) according to the equations (Eq.) of **Hilsenhoff (1987)**.

$$\textbf{Eq. (1): Pollution Tolerance index } (PT_i) = n_i * a_i$$

Where, "n" is the number of specimens in a taxon i, and "a" is the pollution tolerance value for a taxon i

$$\textbf{Eq. (2): HBI} = \sum PT_i/N$$

Where, Σ PTi is the total of Pollution Tolerance indices and N is the total number of specimens in a sample. HBI value is ranged from 0 to 10, which reflects the water quality status from excellent to very poor. Excellent: 0.00 - 3.50 (no OP), Very Good: 3.51 - 4.50 (slight OP), Good: 4.51 - 5.50 (some OP), Fair: 5.51 - 6.50 (Fairly significant OP), Fairly Poor: 6.51 - 7.50 (Significant OP), Poor: 7.51 - 8.50 (Very significant OP) & Very Poor: 8.51 - 10.00 (Severe OP) **(Hilsenhoff, 1987)**.

Multimetric macroinvertebrate index Flanders (MMIF)

The MMIF index is based on macroinvertebrate samples that are collected and identified. The index calculation is a type-specific multimetric system (i.e., depends on the type of river or lake a sampling site belongs to). Depending on the five equally weighted metrics, which are taxa richness (Taxa), number of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT), number of other sensitive taxa (NST), the Shannon–Wiener diversity index (SWD) **(Shannon & Weaver, 1949)**, and the mean tolerance score of all present taxa (MTS), all indices were scored from 0 to 4 according to the criteria stated by **Gabriels *et al.* (2010)**. These five metric scores are summed and subsequently divided by 20 to obtain the final index (MMIF), which ranges from zero (bad evaluation quality) to one (high evaluation quality).

Functional feeding groups of macroinvertebrate

The main functional feeding groups of macroinvertebrates were classified as shredders (i.e. Trichoptera and Diptera), scrapers (i.e. Coleoptera, and Gastropoda), gathering collectors (Ephemeroptera and Oligochaeta), filtering collectors (i.e. Decapoda and Bivalva), and predators (i.e Plecoptera. Odonata, Hemiptera, and Archnida) by **Cummins (2018)**. On the other hand, the analysis of ecological stream health was assessed according to the equations of **Masese *et al.* (2014)** and interpretations based on **Merritt *et al.* (2017)** (Table 1).

Table 1. The analysis of stream ecological health equations and interpretations

Index	Equation	Ratio	Indication
Autotrophy/heterotrophy index	SC/(SH + FC+GC)	>0.75	Autotrophy streams
Top-down predators/ total FFGs	P/(SC+SH+FC+GC)	0.1-0.2	Normal predator to prey balance
		>0.2	Overabundance of predators
Habitat stability index	(FC+SC) / (SH+GC)	>0.50	Plentiful stable substrates
Filtering collectors index	FC/GC	>0.50	Enriched, unusual particulate loading of fine particulate food for filters

SH: Shredder, SC: Scrapers, FC: Filtering collectors, GC: Gathering collectors, P: Predators.

Statistical analysis

All physicochemical parameters, metals, and Hilsenhoff biotic index (HBI) were presented as mean ± sd of twelve investigated sites during each season. The matrix plot was presented to evaluate the correlation between functional feeding groups and some metals depending on Spearman correlation. All tests were performed using the statistical programme SPSS version 23 (SPSS, Inc., Chicago, IL) for Windows.

RESULTS

All the examined physicochemical water parameters were at the normal levels of concern, except temperature that exceeded the concern level in the spring and summer seasons (Table 2). Meanwhile, the dissolved oxygen (DO) level was lower than the national recommended water quality criteria in summer (Table 2).

Table 2. Physicochemical parameters during the four seasons of one year (2019- 2020)

Season	Temperature (ºC)	pH	DO (mg/l)	EC (µmoh/cm)
Summer	29.2 ±0.6	7.42 ±0.1	4.86 ±0.12	340.5 ±18
Autumn	20.5±0.89	7.3±0.33	8.3±0.65	423±23.7
Winter	17.6±1.02	7.37±0.4	9.25±0.83	435.4± 132.4
Spring	28.02±0.86	7.43±0.17	9.4±1.43	341.25 ±19.3

DO: Dissolved Oxygen, EC: Electrical Conductivity. National Recommended Water Quality Criteria (25 ºC for temperature, ≥ 800 µmoh/cm for EC, ≥ 5 mg/l for DO), according to **EPA (2009)**.

On the other hand, the level of concern was 13µg/ l for copper (Cu), 1000µg/ l for iron (Fe), 0.12mg/ l for zinc (Zn), and 100mg/ l for calcium (Ca). Similarly, the concentrations of the detected metals were at the normal level during the different seasons, except Cu that slightly exceeded the concern level in autumn and spring (25.3 and 24.7µg/ l, respectively). Cu ranged from 11.17 to 25.26µg/ l, while Fe ranged from 292.23 to 360.70µg/ l; Zn ranged from 11.45 to 30.67mg/ l, while Ca ranged from 24.96 to 36.15mg/ l (Fig. 2).

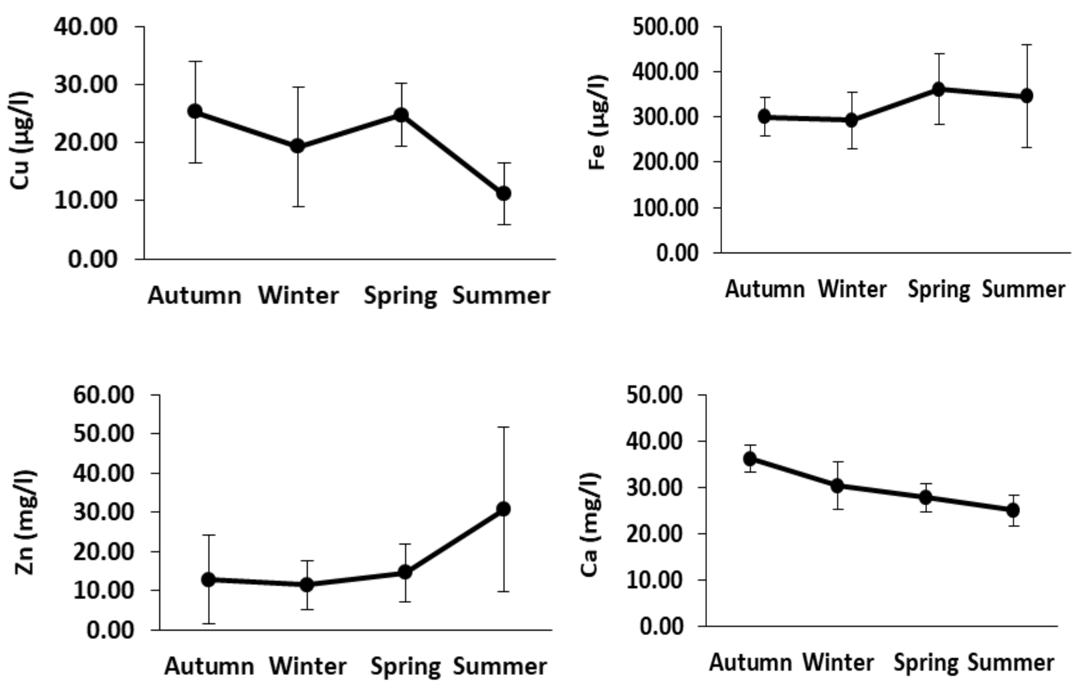

Fig. 2. The concentrations of some metals copper (Cu), iron (Fe), zinc (Zn), and calcium (Ca) determined in the collected water samples during different seasons (2019 -2020)

The sorting of the seasonal macroinvertebrates samples showed ten groups and taxa richness was 18. The highest abundance was noticed during the autumn and spring seasons (449 and 410, respectively), followed by summer and winter (352 and 236, respectively). Groups (Odonata, Decapoda, Gastropoda, and Bivalves) were found during all seasons, while Ephemeroptera and Hemiptera disappeared during winter. On the other hand, Plecoptera and Coleoptera were observed during summer, while Oligocheata and Archnida were observed only during autumn. The sorting of the macroinvertebrate samples showed that Gastropoda was the most distributed organism with an abundance of 538, followed by Decapoda (328), Bivalva (243), Odonata (132), Ephemeroptera (87), Oligochaeta (84), Hemiptera (28) then Hemiptera (3), Archnida (3), and Plecoptera (1) (Table 3).

Results of the Hilsenhoff biotic index (HBI) indicated that organic pollution is seasonally varied increasing from fair during summer (5.66) to fairly poor during autumn (7.33) and poor during winter (7.51), then recovering again to fair during spring (6.21) (Table 3).

Table 3. Results of the collected macroinvertebrates samples from the Nile River in the area of Greater Cairo; total seasonal number, total year number, Group %, Taxa richness and Hilsenhoff biotic index (HBI) during 2019-2020

Group	Taxon	Summer	Autumn	Winter	Spring	Total year	Group %
Ephemeroptera	Mayfly larvae	**20**	**40**	**-**	**27**	**87**	**6.0**
Plecoptera	Stonefly larvae	**1**	**-**	**-**	**-**	**1**	**0.1**
Odonata	Dragonfly larvae	**13**	**11**	**3**	**2**	**132**	**9.1**
	Damselfy larvae	**31**	**26**	**10**	**36**		
Decapoda	Shrimps	**83**	**102**	**45**	**98**	**328**	**22.7**
Coleoptera	Riffle beetle	**3**	**-**	**-**	**-**	**3**	**0.2**
Hemiptera	Water bug	**3**	**1**	**-**	**17**	**28**	**1.9**
	Water boatman	-	-	-	**7**		
Gastopoda	(No. of snails species)	**(6 sp.) 63**	**(9 sp.) 136**	**(9 sp.) 174**	**(11 sp.) 165**	**538**	**37.2**
Bivalves		**135**	**46**	**4**	**58**	**243**	**16.8**
Oligochaeta	Aquatic worms	-	**84**	-	-	**84**	**5.8**
Archnida	Fishing spider	-	**3**	-	-	**3**	**0.2**
Total season abundance		**352**	**449**	**236**	**410**	**1447**	**100**
Taxa richness		**14**	**17**	**13**	**18**		
HBI		5.66 $^{\mathbf{b}}$ ±0.63	7.33$^{\mathbf{ac}}$±1.70	7.51$^{\mathbf{c}}$±2.45	6.21$^{\mathbf{ab}}$±0.46		
Water quality criteria		Fair	Fairly poor	Poor	Fair		

The difference HBI values among seasons performed using (one-way ANOVA) test. The same letters refer to insignificant results and the different letters refer to significant results at $P < 0.05$.

The Score criteria of the seasonal water quality evaluation were used to calculate the MMIF index through the indices: Taxa, EPT, NST, SWD, and MTS are summarized in Fig. (3a). The calculated MMIF during different seasons showed moderate water quality during autumn, spring, and summer (0.60, 0.5, and 0.55, respectively), while poor water quality was detected during winter (0.4) (Fig. 3b).

The total sampled macroinvertebrates belong to only four feeding groups (FFG): gathering collectors, predators, filtering collectors, and scrapers, as shredders were absent during all seasons from all stations under study. Data in Table (4) indicate that the highest abundance percentage of FFG was filtering collectors (39.5 %), followed by scrapers (37.4%), then gathering collectors (11.8%), and predators (11.3%).

The matrix plot presented in Fig. (4) shows the relationship between functional feeding groups and some metals. All functional feeding groups showed a strong positive correlation with Ca, Zn, and Fe. On the other hand, all of them showed a negative correlation with Cu; strong in the case of predators (R^2 = -0.704), moderate in the case of filtering collectors (R^2 = -0.532), while a very weak correlation was observed with gathering collectors and scrapers (R^2 = -0.023 & R^2 = -0.028, respectively) (Fig. 4).

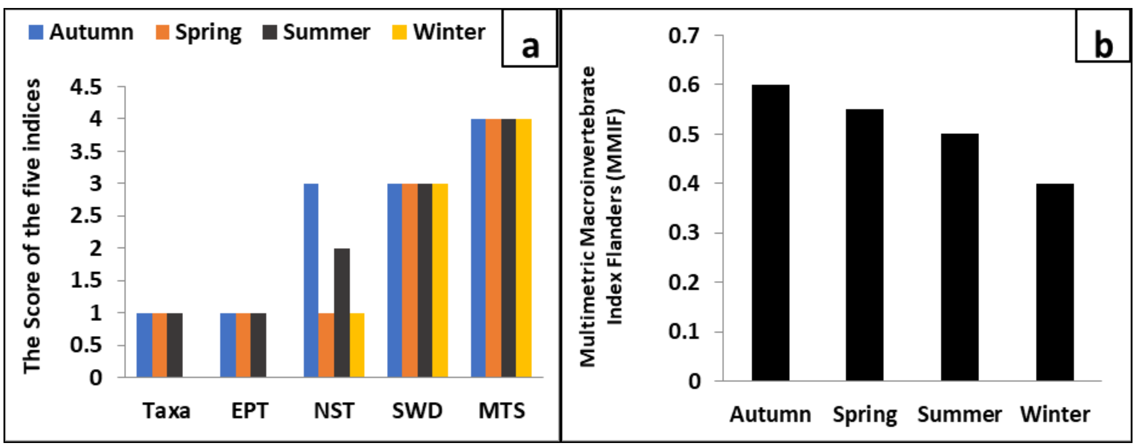

Fig. 3. (**a**): Points score of the five indices (taxa richness: Taxa, number of Ephemeroptera, Plecoptera, and Trichoptera taxa: EPT, number of other sensitive taxa: NST, the Shannon–Wiener diversity index: SWD, the mean tolerance score of all present taxa: MTS) are equally weighted metrics, (**b**) Multimetric Macroinvertbrate index Flanders (MMIF)

Table 4. The relative abundance of functional feeding groups (FFG) of macroinvertebrates collected seasonally along the Nile River stations within Greater Cairo (2019-2020)

FFG	Summer	Autumn	Winter	Spring	Total	Relative abundance (%)
Gathering collectors (GC)	20	124	0	27	171	11.8
Predators (P)	48	41	13	62	164	11.3
Filtering collectors (FC)	218	148	49	156	571	39.5
Scrapers (S)	66	136	174	165	541	37.4
Total	352	449	236	410	**1447**	100

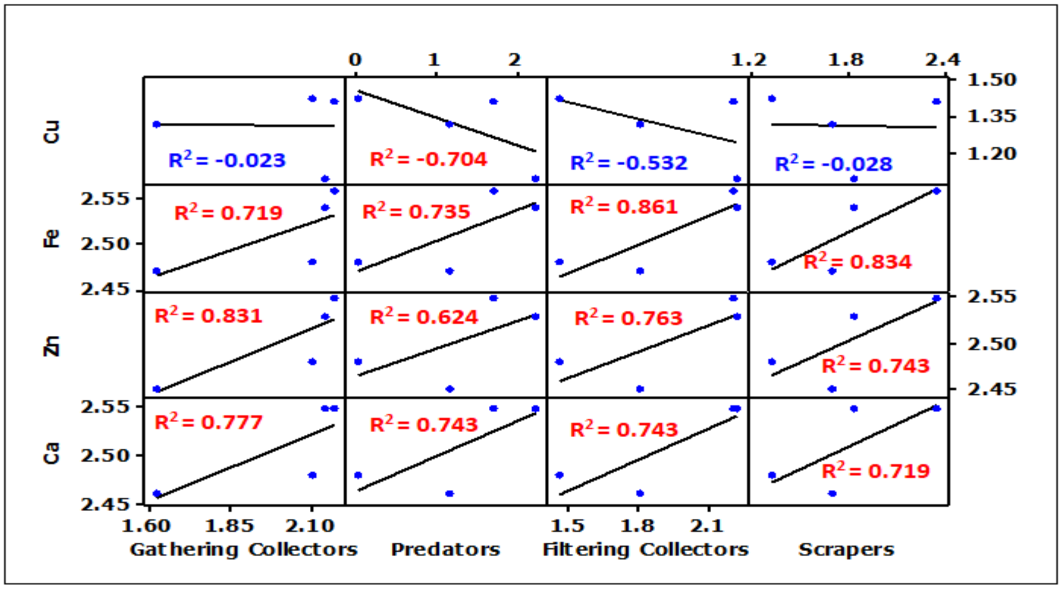

Fig. 4. Matrix plot showed the correlation between functional feeding groups and some metals

Autotrophy/heterotrophy index recorded values > 0.75, indicated autotrophic streams during winter and spring (3.55 and 0.90, respectively), while during autumn and summer it was <0.075, pointing to heterotrophic streams (Fig. 5a). Meanwhile, the top-down predators/total FFGs index ranged from 0.1 to 0.2, indicating natural predator-to-prey balance during autumn, spring, and summer (Fig. 5b).

During autumn, spring, and summer, the habitat stability index recorded values greater than 0.5, indicating the presence of perfectly stable substrates. Meanwhile, the index was not determined during winter because shredder and gathering collector specimens were absent (Fig. 5c). Similarly, the filtering collectors index recorded values > 0.5 during autumn, spring, and summer, pointing to enriched, unusual particulate loading of fine particulate food for filters (Fig. 5d).

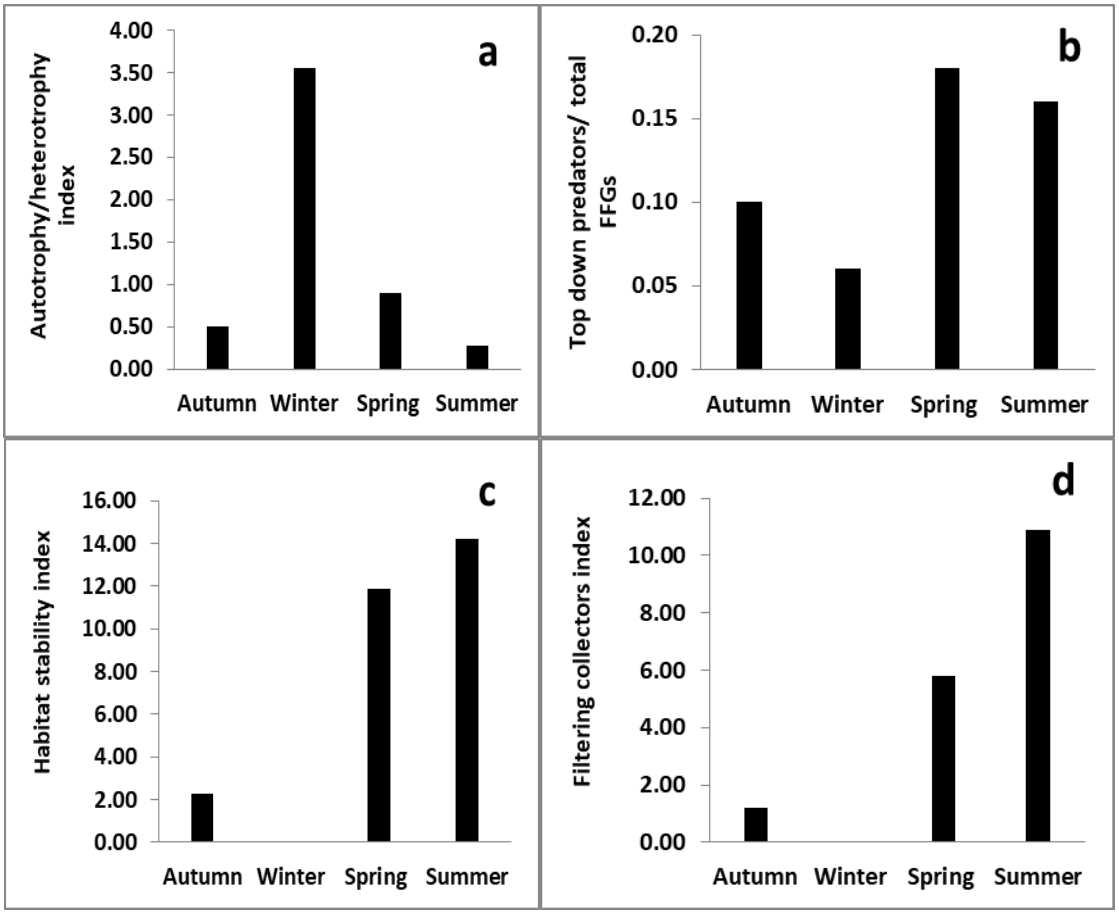

Fig. 5. Analysis of stream ecological health of the River Nile within Greater Cairo according to FFGs indices during the four seasons

DISCUSSION

In the present study, all the examined physicochemical water parameters were at normal levels, except the temperature which slightly exceeded the permissible level (25 ˚C) in spring and summer recording 28.02 and 29.2˚C, respectively. The variations in water temperature depend mainly on the climatic conditions, sampling times, number of sunshine hours, and specific characteristics of water such as turbidity, winds, plant cover, and relative humidity **(Ali *et al.*, 2014; Ghanem *et al.*, 2016; Ghanem, 2022)**. Trace amounts of some metal ions, such as zinc (Zn), copper (Cu), iron (Fe), and calcium (Ca), are required by organisms as cofactors for enzymatic processes **(Kozlowski *et al.*, 2009; Zhang *et al.*, 2014)**. However, an excess of these metal ions will cause severe problems in living organisms due to their higher toxicity, carcinogenicity, and bioaccumulation **(Omraei *et al.*, 2011)**. In the present water samples, Cu levels slightly increased the concern limit during autumn and spring. Although Fe, Zn, and Ca levels were within the normal range, Fe levels were higher in spring and summer than in autumn and winter, and Zn and Ca levels elevated in summer and autumn. These results may be due to seasonal variations in other physicochemical parameters, human activities, and the uptake of organisms.

Seasonality should not be neglected when monitoring and assessing systems. Although this strategy may result in missing information on the overall community at the site **(Linke *et al.*, 1999)**, it can be assumed to be sufficient for water quality assessment purposes. In the present study, the most sensitive collected macroinvertebrate were stoneflies (Plecoptera), followed by riffle beetle (Coleoptera), were only observed during summer, where the HBI average showed the lowest value (5.66), indicating fair water quality. Moreover, the current results showed that the highest taxa richness was (18) during spring and is consistent with the HBI average (6.21), indicating that water quality is fair. According to **Bueno *et al.* (2003)**, the richness and abundance of benthic macroinvertebrates increase in more stable substrates with a plentiful supply of shelter and food resources. In addition, the lowest taxa richness (13) was during winter and compatible with the highest average of HBI (7.51), reflecting the water quality status being very poor. This result might refer to changes in the favorable factors, besides a change in water temperature during the winter season, which in turn affect the ecosystems and biological processes and interfere with growth at the individual level **(Hall & Burns, 2002; Frouz *et al.*, 2002)**. These results are in parallel with the present calculated multimetric macroinvertebrate index Flanders (MMIF), which showed moderate water quality during autumn, spring, and summer, while poor water quality was during winter, reflecting the extreme hydrological regimes and temperatures and other logistical reasons **(Sporka *et al.*, 2006)**.

The occurrence of macroinvertebrates in freshwater streams relies on habitat structure and conditions such as type, composition, and size of the substrate in streams **(Abas *et al.*, 2018; Khudhair *et al.*, 2019)**. In the present work, Odonata, Gastropoda, Decapoda, and Bivalves, were found during all seasons. The structure of odonate communities can shift predictably in response to the disturbance of the riparian vegetation, which in turn makes ambient temperatures influence the abundance of species and their reproductive cycles **(Samways & Steytler, 1996)**. Odonates often have little relevance in standard biotic indices compared to more abundant and pollution-sensitive insect orders, usually present in the benthos of streams, and vegetation structure would likely be a more appropriate environmental characteristic for odonate-related bioassessment **(Smith *et al.*, 2007)**. Meanwhile, gastropods are known to be predominant in aquatic systems and more tolerant of water pollution because they are prone to various pollutants, including metals, industrial chemicals, pesticides, and hydrocarbons **(Alonso & Camarg, 2009; Hellou, 2011; Sayed *et al.*, 2021)**.

In the present work, filtering collectors (e.g., decapods and bivalves) and scrapers (e.g., gastropods and Coleoptera) were the most abundant functional feeding groups (FFGs). These results may be attributed to filtering collectors having a broader range of acceptable food materials than specialists **(Merritt *et al.*, 2002)**. Moreover, they are more tolerant of pollution that might alter the availability of specific foods. Filterers feed on suspended fine particulate organic matter, which is transported by the fast current, and provide them with more organic matter in a shorter period **(Buss *et al.*, 2002)**. In this context, **Uwadiae (2010)** found that the filtering collectors were the most abundant FFG occupying 76.64 %, followed by shredders, which constituted 14.76% of the total population. On the other hand, **Wongsanoon *et al.* (2014)** observed that gathering collectors were the most abundant taxa in the upstream and downstream in every month of the study period, followed by scrapers, while shredders and filtering collectors were found in small numbers. In the current work, scrapers occupied almost half the abundance of functional feeding groups (FFGs) and were distributed almost equally during all seasons.

Scrapers are generally most abundant in regions with clear water streams, which may be affected by light conditions and algae abundance **(Otto & Svensson, 1983)**. Additionally, **Addo-Bediako (2021)** attributed the highest numbers of scrapers at some sites of the Dwars and Spekboom Rivers could be due to an increase in algal production. In the current work, the abundance of scrapers was higher during spring, winter, and autumn than during summer. **Alberts *et al.* (2018)** stated that the highest number of scrapers was in spring, and the lowest was in winter. However, benthic algal biomass did not vary between winter and spring, suggesting that the relationship between scrapers and their primary food resource differs by season. Besides, these results might be attributable to scrapers' preference for particular algal taxa that are favored by either shaded or sunlit conditions **(Feminella *et al.*, 1989)**. In the present study, the scraper feeding group is mainly represented by Gastropoda, which included the freshwater snails with taxa richness of 11 species, abundance of 538, and 37.2% of the total macroinvertebrate sample. The gastropods inhabit special microhabitats and summarize environmental changes due to their long maturation period and comparative longevity. They are well known for having high levels of heavy metals in their bodies **(Vukašinović-Pešić *et al.*, 2017)**. Thus, gastropods serve in environmental risk assessments and monitoring as pollution indicators for different compounds **(Tallarico, 2015)**. Additionally, freshwater snails act as the obligatory first intermediate hosts of trematodes, which can cause health hazards in animals and humans. So, they serve not only as a source of food and a place of reproduction for these parasites but also as a means of transport to their next hosts **(Esch *et al.*, 2001; Lockyer *et al.*, 2004; Faltynkova *et al.*, 2008)**. Many studies have been conducted in Egypt to survey larval trematodes in freshwater snails **(Noor El-Din, 2009; Yousif *et al.*, 2010, 2016; Ayoub *et al.*, 2020)** recommending that the presence of trematodes in snails could pose a serious health problem and further studies are necessary to characterize these infections.

On the other hand, shredders were absent during all seasons from all stations in the present work. The shredders digest the coarse particulate organic matter (such as leaves and twigs) to fine particulate organic matter which will be used by other FFGs. This result may be attributed to the reduction of macrophytes **(Merritt *et al.*, 2017)**. The high velocity of water flow may also clear up almost all leaves rapidly, causing the disappearance of shredders **(Wongsanoon *et al.*, 2014)**. Moreover, the decrease in leaf litter hinders the growth and development of shredders, makes the aquatic ecosystem unbalanced, and ultimately affects the structure and function of the ecosystem **(Liu *et al.*, 2019)**. Also, **Barbour *et al.* (1996)** stated that shredders and scrapers are assumed to be more sensitive to disturbance, while collector-gatherers and collector-filterers are more tolerant of the pollution that might alter the availability of specific foods. **Chapman and Chapman (2003)** and **Abdollah *et al.* (2004)** declared that the total number of shredders must be the highest compared to other groups, and the number of filtering collectors is the lowest. Therefore, the widespread distribution of filtering collectors in the present study stretch indicates the unstable status of the study area.

The main routes of trace element uptake in aquatic organisms are directly from the water or indirectly through food **(Wang, 2002; Rainbow, 2007)**. Macroinvertebrates are exposed to metals through their gills and by dietary pathways through water filtration (filterers), grazing on periphyton (scrapers), or preying on other invertebrates (predators) **(Mebane *et al.*, 2020)**.

In the current work, all functional feeding groups (FFGs) showed positive correlations with calcium (Ca). These results partially agree with those of **Park *et al.* (2008)** who observed that, increasing concentrations of Ca had negative impacts on shredders and positive effects on scrapers and collector-filterers. Gathering collectors move into sediment and collect smaller particles of organic matter, coming into contact with metal-polluted substrates **(Santoro *et al.*, 2009)**. Meanwhile, certain scrapers like gastropods may need calcium for shell formation and exoskeletons **(White *et al.*, 2007)**.

Zinc (Zn) may play a key role (e.g., in the carbonic anhydrase enzyme) or is stored in some organs for later excretion **(Rainbow, 2007)**. In the present work, all functional feeding groups (FFGs) showed a strong positive correlation with Zn. These results may be attributed to its low toxicity and high bioavailability by certain organisms **(Clements *et al.*, 2013)**. However, the high concentrations of Zn may cause a reduction in some populations of macroinvertebrates **(Iwasaki *et al.*, 2012)**. Additionally, **Qu *et al.* (2010)** observed that in some species of Ephemeroptera, their abundance decreased with Zn levels.

Also, all FFGs showed a strong positive correlation with iron (Fe). This result was in line with **Noor-Ul-Islam *et al.* (2021)**, who argued the observed positive correlation could indicate that the Fe concentrations were within permissible limits for the survival of many species. In the present study, both filtering collectors and predators showed a strong negative correlation with copper (Cu), while scrapers and gathering collectors showed a very weak negative correlation with Cu. These correlations might be due to Cu dissolving in water as the Cu2+ ion is the most available and toxic form in freshwater **(Blanchard & Grosell, 2006)**. However, Cu concentrations in the aquatic environment can be non-lethal to aquatic organisms in the concentration of around 36μg/ l **(Campana *et al.*, 2012; Clements *et al.*, 2013)**.

The Autotrophic/Heterotrophic index serves instead of Production/Respiration (P/R) **(Vannote *et al.* 1980)**. In the present work, analysis of stream ecological health showed that the River Nile stream was autotrophic during spring and winter, while it was heterotrophic during summer and autumn. These findings may be due to the dominance of scrapers in winter and spring, while the filtering collectors were dominant in summer and autumn. These results are incompatible with **Pereira *et al.* (2021)**, who found that almost all sites were classified as heterotrophic. Also, **Camara *et al.* (2020)** observed that all investigated sites are heterotrophic. **Makaka *et al.* (2018)** reported that the whole of the Tokwe River in Zimbabwe was heterotrophic according to the P/R ratio, indicating the importance of allochthonous resources in the ecology of the whole river system. However, **Masese *et al.* (2014)** suggested the predominance of heterotrophic over autotrophic production could be attributed to extensive pollution by livestock waste that tends to promote the high abundance of collectors over scrappers. **Savić *et al.* (2017)** attributed the dominant staple food chain for aquatic insect communities to non-native detritus. Thus, the present autotrophic state during spring and winter provides macroinvertebrate with suitable habitat through the availability of photosynthetic organisms that produces organic molecules, increases the dissolving oxygen, and removes carbon dioxide and ammonia nitrogen from the water. Meanwhile, the predominance of a heterotrophic state during summer and autumn indicates that the affected water quality may be due to the extensive human activities during these seasons.

In addition, the predator-to-prey index ranged from 0.1 to 0.2, indicating a natural predator-to-prey balance occurred during autumn, spring, and summer. According to **Masese *et al.* (2014)**, the value of the predator-prey ratio must be between 0.1 and 0.2 to achieve the balance between predators and prey. Conversely, **Camara *et al.* (2020)** found that all values of the predator-prey ratio were above 0.2 at all sites, attributing to the presence of an over abundance of predators in Lake Kodjoboue, Côte d'Ivoire. Also, both indices of habitat stability and filtering collectors recorded values of > 0.5, pointing to perfect stable substrates during the autumn, spring, and summer, where gathering collectors were represented. In contrast, habitat stability didn’t achieve during winter due to the absence of two feeding groups, gathering collectors and shredder groups. One explanation for these results would be the intermediate disturbance hypothesis **(Ward & Stanford, 1983; Ward *et al.*, 1999)**. Intermediate sites submitted to a fixed stressor that may generate moderate mortality in the species, not in such numbers that a recovery is impossible, but at the same time, sufficient to limit the growth of competitive species. The same results were obtained by **Pereira *et al.* (2021)**, who stated that the habitat stability index indicated stable substrates that were more plentiful in studied intermediate sites. Also, these results interpret the present observation of the filtering collectors and scrapers as the highest abundant of the trophic group during autumn, spring, and summer. Meanwhile, scrapers (mainly Gastropoda) showed a grateful abundance and diversity in the winter. **Oliveira and Nessimian (2010)** have reported and attributed that the filtering collector was the second most abundant, being more concentrated in rock substrate and litter from rifles.

In the current work, chemical assessment cooperated with macroinvertebrate matrices have successfully differentiated between the River Nile seasonal water qualities. In the same consequence, **El-Khayat *et al.* (2011)** made a comparison between chemical examination (physical parameters and metal levels) and biological assessment (depending on macroinvertebrate data) in a total of 643 sites in the Egyptian River Nile branches, main canals over eight successive seasons and they concluded that the biological assessment was more able to define the contaminated stations than the chemical examination, where the percentage of polluted sites determined using biological assessment was higher than those detected by chemical assessment. Furthermore, the studies of **Bartram and Balance (1996)**, **Karr and Chu (2000)**, **Mandaville (2002)**, and **EPA (2011)** indicated that the biological assessment was more able to define the polluted sites than a chemical assessment. Hence, the biological assessment provides direct measures of the cumulative response of the biological community to all sources of stress to which the organisms were exposed over a period, while chemicals are designed to protect the biological community of a water body from different toxic levels of pollutants, which is valid only for the time when the sample was collected.

CONCLUSION

The chemical assessment showed that all physical water parameters and metals were around the permissible levels during all seasons (except temperature and Cu), while macroinvertebrate matrices successfully distinguished between the River Nile water qualities. The HBI showed water quality varying from fair during spring to poor during winter. Also, the MMIF showed water quality ranging from poor during winter to moderate during the rest seasons. Filtering collectors and scrapers were the predominant functional feeding groups, followed by predators during all seasons. Although gathering collectors were only observed during autumn, spring, and summer, shredders were absent during all seasons. Hence, the analysis of stream ecological health using matrices showed that the River Nile was heterotrophic during summer and autumn and autotrophic during spring and winter. The stable habitat and substrates were observed during autumn, spring, and summer, where most feeding groups were represented.

FUNDING

This work was supported by an internal research project (Grant No. 104C) from Theodor Bilharz Research Institute (TBRI), Egypt.

REFERENCES

Abas, A.; Rak, A.E.; Omar, S.A.S. and Kumaran, J.V. (2018). Correlation between benthic macroinvertebrate distribution and substrate composition in selected recreational rivers in Kelantan, Malaysia. *J. Sustain. Sci. Manag.*, 13(1): 39–48.

Abdel Gawad, S.S. (2019). Using benthic macroinvertebrates as indicators for assessment the water quality in River Nile, Egypt. *Egypt. J. Basic Appl. Sci.*, 6(1): 206-219.

Abdel-Satar, A.M.; Belal, D.M.; Salem, S.G.; Abdelmageed, A.A.; Abdo, M.H.; Gawad, S.S.A. and Al-Afify, A.D. (2022). Benthic diatoms and macroinvertebrates status with relevant to sediment quality of islands shores in the Nile River, Egypt. *Rendiconti Lincei. Scienze Fisiche e Naturali*, 1-19.

Abdollah, A.; de Mazancourt, C.; Elinge, M.M.; Graw, B.; Grzesuik, M.; Henson, K.; Kamoga, M.; Leonhartsberger, P.; Matemba, R.B.; Merl, M.; Moss, B.; Nuhu, D.; Oduro, D.J.; Provvedi, S.; Rasoma, R.V.; Ratsoavina, K.; Trevelyan, R.; Tumanye, N.; Ujoh, V.N.; van de Wiel, G.; Wagner, K.; Waylen, K. and Yonas, M. (2004). Comparative studies on the structure of an upland African stream ecosystem. *Fresh. For.*, 21: 27-47.

Addo-Bediako, A. (2021). Spatial distribution patterns of benthic macroinvertebrate functional feeding groups in two rivers of the olifants river system, South Africa. *J. Fresh. Ecol.*, 36(1): 97-109.

Alberts, J.M.; Fritz, K.M. and Buffam, I. (2018). Response to basal resources by stream macroinvertebrates is shaped by watershed urbanization, riparian canopy cover, and season. *Fresh. Sci.*, 37(3): 640-652.

Ali, E.M.; Shabaan-Dessouki, S.A.; Soliman, A.R.I. and El Shenawy, A.S. (2014). Characterization of chemical water quality in the Nile River, Egypt. *Int. J. Pure and Appl. Biosci.*, 2(3): 35-53.

Alonso, A. and Camargo, J.A. (2009). Long-term effects of ammonia on the behavioral activity of the aquatic snail Potamopyrgus antipodarum (Hydrobiidae, Mollusca). *Arch. Environ. Cont. Toxicol.*, 56: 796-802.

Ayoub, M.; Abdel-Motleb, A. and Tadros, M. (2020). First Egyptian record of the eye fluke (Philophthalamus palpebrarum) transmitted by Melanoides tuberculata snail and its experimental life cycle. *Egypt. J. Aqua. Biol. Fish.*, 24(2): 147-162.

Barbour, M.; Gerritsen, J.; Griffith, G.; Frydenborg, R.; Mccarron, E.; White, J. and Bastian, M. (1996). A framework for biological criteria for Florida streams using benthic macroinvertebrates. *J. North Ameri. Benthol. Soci.*, 15(2): 185–211.

Barbour, M.T.; Gerritsen, J.; Snyder, B.D. and Stribling, J.B. (1999). *Bioassessment protocols for use in streams and wadeable rivers: Periphyton, benthic macroinvertebrates and fish*, 2nd Ed, EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

Bartram, J. and Balance, R. (1996). *Water quality monitoring-A practical guide to the design and implementation of freshwater quality studies and monitoring programmes*. NEP/WHO, Geneva.

Blanchard, J. and Grosell, M. (2006). Copper toxicity across salinities from freshwater to seawater in the euryhaline fish Fundulus heteroclitus: Is copper an ionoregulatory toxicant in high salinities? *Aqua. Toxicol.*, 80: 131–139.

Bode, R.W.; Novak, M.A. and Abele, L.E. (1996). *Quality Assurance Work Plan for Biological Stream Monitoring in New York State*. NYS Department of Environmental Conservation, Albany.

Bueno, A.A.P.; Buckup, G.B. and Ferreira, B.D.P. (2003). Estrutura da comunidade deinvertebrados bentônicos em dois cursos d'água do Rio Grande do Sul, Brazil. *Revista brasileira de Zoologia*, 20: 115–125.

Buss, D.F.; Baptista, D.F.; Silveira, M.P.; Nessimian, J.L. and Dorville, L.F.M. (2002). Influence of water chemistry and environmental degradation on macroinvertebrate assemblages in a river basin in south-east Brazil. *Hydrobiol.*, 481: 125–136.

Camara, I.A.; Kra, M.K.; Kouadio, N.K.; Konan, M.K.; Edia, E.O.; Doumbia, L.; Ouattara, A. and Diomande, D. (2020). Composition, structure and functional feeding of aquatic entomofauna in Kodjoboué Lake: water quality assessment. *Open J. Ecol.*, 10(4): 160-176.

Campana, O.; Simpson, S.L.; Spadaro, D.A. and Blasco, J. (2012). Sub-lethal effects of copper to benthic invertebrates explained by sediment properties and dietary exposure. *Environ. Sci. Technol.*, 46: 6835–6842.

Chapman, A. and Chapman, L.J. (2003). Deforestation in Tropical Africa: Impacts on aquatic ecosystems. In: Crisman, L.T.; Chapman, L.J.; Chapman, C.A. and Kaufman, Les. *Conservation, Ecology, and Management of African Fresh Waters*. University Press of Florida, Gainsville, pp. 229-247.

Clements, W.H.; Cadmus, P. and Brinkman, P.S.F. (2013). Responses of aquatic insects to Cu and Zn in stream microcosms: understanding differences between single species tests and field responses. *Environ. Sci. Technol.*, 47: 7506–7513.

Cummins, K.W. (1995). Invertebrates. In: Calow, P. and Petts, G.E. (eds). *The Rivers Handbook*. Blackwell Scientific, Oxford, pp. 234–250.

Cummins, K.W. (2018). Functional analysis of stream macroinvertebrates. In: Gokce, D. (ed). *Limnology: Some New Aspects of Inland Water Ecology*. United kingdom, London, pp. 63-78.

Dudgeon, D. (2010). Prospects for sustaining freshwater biodiversity in the 21st century: linking ecosystem structure and function. *Curr. Opin. Environ. Sust.*, 2: 422–430.

El-Khayat, H.M.; Mahmoud, K.M.; Mostafa, B.B.; El-Deeb, F.A.; Tantawy, A.A.; Ragb, F.M.; Ismail, N.M.; El-Said, K.M. and Abu Taleb, H.M. (2011). Habitat characteristics for different freshwater snail species as determined biologically through macroinvertebrate information. *J. Egypt. Soci. Parasitol.*, 41(3): 651-664.

EPA (United States Environmental protection Agency). (2009). *National recommended water quality criteria*. Office of Water, Office of Science and Technology.

EPA (United States Environmental protection Agency). (2011). *Basics: What are Biocriteria and Bioassessment Data?*. http://water.epa.gov/scitech/swguidance/stan-dards./basics.cfm.

Esch, G.W.; Curtis, L.A. and Barger, M.A. (2001). A perspective on the ecology of trematode communities in snails. *Parasitol.*, 123: 57-75.

Faltynkova, A.; Nasincova, V. and Kablaskova, L. (2008). Larval trematodes (Digenea) of planorbid snails (Gastropoda: Pulmonata) in Central Europe: a survey of species and key to their identification. *Syst. Parasitol.*, 69: 155-178.

Feminella, J.W.; Power, M.E. and Resh, V.H. (1989). Periphyton responses to invertebrate grazing and riparian canopy in three northern California coastal streams. *Fresh. Biol.*, 22(3): 445-457.

Ferreira, V.; Encalada, A.C. and Graça, M.A.S. (2012). Effects of litter diversity on decomposition and biological colonization of submerged litter in temperate and tropical streams. *Fresh. Sci.*, 31: 945–962.

Frouz, J.; Ali, A. and Lobinske, R.J. (2002). Influence of temperature on developmental rate, wing length, and larval head capsule size of pestiferous midge Chironomus crassicaudatus (Diptera: Chironomidae). *J. Econ. Entomol.*, 95: 699-705.

Gabriels, W.; Goethals, P.L.M. and De Pauw, N. (2006). Development of a multimetric assessment system based on macroinvertebrates for rivers in Flanders (Belgium) according to the European Water Framework Directive. *Internationale Vereinigung Für Theoretische und Angewandte Limnologie: Verhandlungen*, 29: 2279–2282.

Gabriels, W.; Lock, K.; De Pauw, N. and Goethals, P.L. (2010). Multimetric Macroinvertebrate Index Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). *Limnol.*, 40(3): 199-207.

Ghanem, M.; Shehata, S. and Ghanem, E. (2016). Influences of some pollutants on water quality of El-Bagouria canal at Kafr El-Zayat region, El-Gharbia Governorate, Egypt. *Egypt. J. Aqua. Biol. Fish.*, 20(2): 89-111.

Ghanem, M.H. (2022). Study of some Physicochemical Parameters in the Water of El-Bagouria Canal at El-Menoufia Governorate, Egypt. *Egypt. J. Hospital Medici.*, 86(1): 258-265.

Gueltekin, Z.; Aydin, R. and Winkelmann, C. (2017). Macroinvertebrate composition in the metarhithral zones of the Munzur and Pülümür rivers: a preliminary study. *Turkish J. Zool.*, 41(6): 1100-1104.

Hall, C.J. and Burns, C.W. (2002). Mortality and growth responses of Daphnia of carinata to temperature and salinity. *Fresh. Biol.*, 47: 451-459.

Hellou, J. (2011). Behavioral ecotoxicology. An "early warning" signal to assess environmental quality. *Environ. Sci. Poll. Res.*, 18: 1–11. http://doi:10.1007/s11356-010-0367-2.

Hilsenhoff, W.L. (1987). An improved biotic index of organic stream pollution. *Great Lakes Entomolog.*, 20: 31–39.

Hussein, A.M.; Mahmoud, R.K.; Sillanpää, M. and Wahed, M.S.A. (2021). Impacts alum DWTPs sludge discharge and changes in flow regime of the Nile River on the quality of surface water and cultivated soils in Fayoum watershed, Egypt. *Sci. Total Environ.*, 766: 144333.

Hynes, H.B.N. (1984). The relationships between the taxonomy and ecology of aquatic insects. In: Resh, V.H. and Rosenberg, D.M. (eds). *The Ecology of Aquatic Insects*. Praeger, New York, pp. 9–23.

Ibrahim, A.; Bishai, H. and Khalil, M. (1999). *Freshwater Molluscs of Egypt: Publication of National Biodiversity Unit, No. 10*. Egyptian Environmental Affairs Agency, Dept. Nature Protection.

Iwasaki, Y.; Kagaya, T.; Miyamoto, K. and Matsuda, H. (2012). Responses of riverine macroinvertebrates to zinc in natural streams: implications for the japanese water quality standard. *Water, Air, and Soil Poll.*, 223: 145–158.

Jannat, N.; Mottalib, M.A. and Alam, M.N. (2019). Assessment of physico-chemical propertiesof surface water of Mokeshbeel, Gazipur. Bangladesh. *J. Environ. Sci.*, 2(3): 1–6. http://doi: 10.24966/ESCR-5020/100014.

Karr, J.R. and Chu, E.W. (2000). Sustaining living rivers. *Hydrobiol.*, 422: 1–14. https://doi.org/10.1023/A:1017097611303.

Khudhair, N.; Yan, C.; Liu, M. and Yu, H. (2019). Effects of habitat types on macroinvertebrates assemblages structure: Case study of Sun Island Bund Wetland. *Biomed Res. Int.*, 1–13. http://doi:10.1155/2019/2650678.

Kiffney, P.M. and Clements, W.H. (1996). Effects of metals on stream macroinvertebrate assemblages from different altitudes. *Ecol. Appl.*, 6: 472–481.

Kozlowski, H.; Janicka-Klos, A.; Brasun, J.; Gaggelli, E.; Valensin, D. and Valensin, G. (2009). Copper, iron, and zinc ions homeostasis and their role in neurodegenerative disorders (metal uptake, transport, distribution and regulation). *Coor. Chem. Rev.*, 253(21-22): 2665-2685.

Linke, S.; Bailey, R.C. and Schwindt, J. (1999). Temporal variability of stream bioassessments using benthic macroinvertebrates. *Fresh. Biol.*, 42: 575–584.

Liu, M.H.; Meng, Y.; Cao, J.J.; Cui, X.B. and Al, M.N. (2019). Functional traits of Zoobenthos in Naolihe Wetland. *J. North. For. Univ.*, 47(1): 76-82.

Lockyer, A.E.; Jones, C.S.; Noble, L.R. and Rollinson, D. (2004). Trematodes and snails: an intimate association. *Canadian J. Zool.*, 82: 351-269.

Makaka, C.; Muteveri, T.; Makoni, P.; Phiri, C. and Dube, T. (2018). Longitudinal distribution of the functional feeding groups (FFGs) of aquatic macroinvertebrates and ecosystem integrity of Tokwe River, Zimbabwe. *J. Biodiver. Environ. Sci.*, 13(1): 16-33.

Mandaville, S.M. (2002). *Benthic Macroinvertebrates in Freshwaters-Taxa Tolerance Values, Metrics, and Protocols*. Soil & Water Conservation Society of Metro Halifax, Nova Scotia. http://lakes.chebucto.org/H-1/tolerance.pdf.

Masese, F.O.; Kitaka, N.; Kipkemboi, J.; Gettel, G.M.; Irvine, K. and McClain, M.E. (2014). Macroinvertebrate functional feeding groups in Kenyan highland streams: evidence for a diverse shredder guild. *Fresh. Sci.*, 33(2): 435-450.

Mebane, C.A.; Chowdhury, M.J.; De Schamphelaere, K.A.; Lofts, S.; Paquin, P.R.; Santore, R.C. and Wood, C.M. (2020). Metal bioavailability models: Current status, lessons learned, considerations for regulatory use, and the path forward. *Environ. Toxicol. Chem.*, 39: 60–84.

Merritt, R.W.; Cummins, K.W. and Berg, M.B. (2017). Trophic relationships of macroinvertebrates. In: *Methods in Stream Ecology*, *Volume 1*. Academic Press, pp. 413-433. https://doi.org/10.1016/B978-0-12-416558-8.00020-2.

Merritt, R.W.; Cummins, K.W.; Berg, M.B.; Novak, J.A.; Higgins, M.J.; Wessell, K.J. and Lessard, J.L. (2002). Development and application of a macroinvertebrate functional-group approach in the bioassessment of remnant river oxbows in southwest Florida. *J. North Ameri. Benthol. Soci.*, (2): 290-310.

Noor El-Din, S.N.A.; EL-Sheikh, H.E. and Taima, A.A. (2009). The First Record of a Monostome trioculate Cercaria (Trematoda: Notocotylidae) From Bithynia goryi Snail In Egypt. *Egypt. J. Experi. Biol. Zool.*, 5: 307–312.

Noor-Ul-Islam, H.; Khan, K.; Zia, S.A.; Naeem, M. and Shams, W.A. (2021). Heavy metals accumulation in dragonflies (Odonata) and their habitats in district Swabi, Khyber Pakhtunkhwa, Pakistan: assessing dragonfly bionomics in the region. *Bull. Environ. Contamin. Toxicol.*, 107(5): 838-847.

Oliveira, A.L.H.D. and Nessimian, J.L. (2010). Spatial distribution and functional feeding groups of aquatic insect communities in Serra da Bocaina streams, southeastern Brazil. *Acta Limnol. Brasil.*, 424-441.

Omraei, M.; Esfandian, H.; Katal, R. and Ghorbani, M. (2011). Study of the removal of Zn (II) from aqueous solution using polypyrrole nanocomposite. *Desalination*, 271(1-3): 248-256.

Otto, C. and Svenson, B. (1983). Properties of acid brown water streams in South Sweden. *Archiv. für. Hydrobiologie*, 99: 15-36.

Park, Y.S.; Lek, S.; Chon, T.S. and Verdonschot, P.F. (2008). Evaluation of environmental factors to determine the distribution of functional feeding groups of benthic macroinvertebrates using an artificial neural network. *J. Ecol. Environ.*, 31(3): 233-241.

Pereira, P.D.S.; Souza, N.F.D.; Baptista, D.F.; Ribeiro-Alves, M.; Santos, H.L.C. and Buss, D.F. (2021). Functional Feeding Group composition and attributes: evaluation of freshwater ecosystems in Atlantic Forest, Brazil. *Biota Neotropica*, 21(2): e20201016. https://doi.org/10.1590/1676-0611-BN-2020-1016.

Pescador, M.L.; Rasmussen, A.K. and Harris, S.C. (1995). *Identification Manual for the Caddisfly (Trichoptera) Larvae of Florida*. Division of Water Resource Management, Florida Department of Environmental Protection, Tallahassee.

Qu, X.; Wu, N.; Tang, T.; Cai, Q. and Park, Y.S. (2010). Effects of heavy metals on benthic macroinvertebrate communities in high mountain streams. *Annales de Limnologie*, 46(4): 291-302.

Rainbow, P.S. (2007). Trace metal bioaccumulation: Models, metabolic availability and toxicity. *Environ. Int.*, 33: 576–582.

Resh, V.H. (2008). Which group is best? Attributes of different biological assemblages used in freshwater biomonitoring programs. *Environ. Monit. Ass.*, 138(1-3): 131–138.

Samways, M.J. and Steytler, N.S. (1996). Dragonfly (Odonata) distribution patterns in urban and forest landscapes, and recommendations for riparian management. *Biol. Conserv.*, 78(3): 279–288.

Santoro, A.; Blo, G.; Mastrolitti, S. and Fagioli, F. (2009). Bioaccumulation of heavy metals by aquatic macroinvertebrates along the Basento River in the south of Italy. *Water, Air, and Soil Poll.*, 201: 19–31.

Savić, A.; Đorđević, M.; Jušković, M. and Pešić, V. (2017). Ecological Analysis of Macroinvertebrate Communities Based on Functional Feeding Groups: A Case Study in Southeastern Serbia. *Biologica Nyssana*, 8: 159-166.

Sayed, S.S.M.; Abdel-Motleb, A.; Saleh, H.A.; El-Hamid, R.M.A.; Kader, A.A. and Abdel-Wareth, M.T. (2021). Pollution by organochlorine and organophosphorus pesticides residues in watercourses of some Egyptian governorates with reference to the distribution of macroinvertebrates. *Int. J. Environ. Stu.*, 78(6): 914-939.

Schäfer, R.B.; Kefford, B.J.; Metzeling, L.; Liess, M.; Burgert, S.; Marchant, R.; Pettigrove, V.; Goonan, P. and Nugegoda, D.A. (2011). Trait database of stream invertebrates for the ecological risk assessment of single and combined effects of salinity and pesticides in South-East Australia. *Sci. Total Environ.*, 409: 2055–2063.

Shannon, C.E. and Weaver, W. (1949). *The Mathematical Theory of Communication*. University of Illinois Press, Urbana, Illinois.

Smith, J.; Samways, M.J. and Taylor, S. (2007). Assessing riparian quality using two complementary sets of bioindicators. *Biodiv. Conserv.*, 16(9): 2695-2713.

Smith, M.J.; Kay, W.R.; Edward, D.H.D.; Papas, P.J.; Richardson, K.S.J.; Simpson, J.C.; Pinder, A.M.; Cale, D.J.; Horwitz, P.H.J.; Davis, J.A.; Yung, F.H.; Norris, R.H. and Halse, S.A. (1999). AusRivAS: using macroinvertebrates to assess ecological condition of rivers in Western Australia. *Fresh. Biol.*, 41(2): 269-282.

Sporka, F.; Vlek, H.E.; Bulankova, E. and Krno, I. (2006). Influence of seasonal variation on bioassessment of streams using macroinvertebrates. *Hydrobiologia*, 566: 543–555.

Tallarico, L.D.F. (2015). Freshwater Gastropods as a Tool for Ecotoxicology Assessments in Latin America. *Ameri. malacol. bull.*, 33: 330–336.

Uwadiae, R.E. (2010). Macroinvertebrates functional feeding groups as indices of biological assessment in a tropical aquatic ecosystem: implications for ecosystem functions. *New York Sci. J.*, 3(8): 6-15.

Vannote, R.L.; Minshall, G.W.; Cummins, K.W.; Sedell, J.R. and Cushing, C.E. (1980). The River Continuum Concept. *Canadian J. Fish. Aqua. Sci.*, 37(1): 130–137.

Vukašinović-Pešić, V.; Blagojević, N.; Vukanović, S.; Savić, A. and Pešić, V. (2017). Heavy metal concentrations in different tissues of the snail Viviparus mamillatus (Küster, 1852) from lacustrine and riverine environments in Montenegro. *Turkish J. Fish. Aqua. Sci.*, 17(3): 557-563.

Wang, W.X. (2002). Interactions of trace metals and different marine food chains. *Marine Ecol. Progress Ser.*, 243: 295–309.

Ward, J.V. and Stanford, J.A. (1983). The intermediate disturbance hypothesis: an explanation for biotic diversity in lotic ecosystems. In: Fontaine, T.D. III and Bartell, S.M. (eds). *Dynamics of lotic ecosystems*. Ann Arbor Science, Michigan, USA, pp. 347-356.

Ward, J.V.; Tockner, K. and Schiemer, F. (1999). Biodiversity of foodplain river ecosystems: ecotones and connectivity. *Regulated Rivers Res. Manag.*, 15: 125-139.

White, M.M.; Chejlava, M.; Fried, B. and Sherma, J. (2007). The concentration of calcium carbonate in shells of freshwater snails. *Ameri. malacol. bull.*, 22(1): 139-142.

WHO (World Health Organization). (2011). *The use of DDT in malaria vector control: WHO position statement* (No.WHO/HTM/GMP/2011).

Wongsanoon, J.; Jatisatienr, A.; Mungkornasawakul, P. and Phalaraksh, C. (2014). Macroinvertebrate functional feeding groups of the Rajjaprabha dam, Surat Thani, Thailand. *Scholarly J. Biol. Sci.*, 3(1): 7-17.

Yousif, F.; Ayoub, M.; Tadros, M. and El bardicy, S. (2016). The first record of Centrocestus formosanus (Nishigori, 1924) (Digenea: Heterophyidae) in Egypt. *Experi. Parasitol.*, 168: 56-61.

Yousif, S.; Ibrahim, A.; El-Bardicy, S.; Sleem, S. and Ayoub, M. (2010). Morphology of new eleven cercariae produced from Melanoides tuberculata snails in Egypt. *Australian J. Basic Appl. Sci.*, 4(6): 1482-1494.

Zhang, Y.; Chi, H.; Zhang, W.; Sun, Y.; Liang, Q.; Gu, Y. and Jing, R. (2014). Highly efficient adsorption of copper ions by a PVP-reduced graphene oxide based on a new adsorptions mechanism. *Nano-Micro Lett.*, 6(1): 80-87.