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Biological Assessment of Mining Pollution in the Lufira River System (Haut-Katanga, Democratic Republic of the Congo) Using Monopisthocotylan Parasites of the Blunt-Toothed African Catfish

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Abstract

This study examined the effects of pollution from the Shituru hydrometallurgic complex on the Upper Lufira Basin, Democratic Republic of the Congo, between September 2015 and September 2017. Physico-chemical water variables and trace metal elements in water and sediment, as well as diversity and infection parameters of monopisthocotylan parasites infesting *Clarias ngamensis*, were assessed at three sites: the Lufira River, Panda River, and Lake Tshangalele. We hypothesised that low pollution would correlate with greater ectoparasite species richness and higher infection parameters. Results indicated severe ecological degradation in the highly polluted Panda River (with high concentrations of TMEs; e.g., 510.830 ± 0.86; 82.470 ± 0.200 µg/L for Co^{2+} and Cu^{2+} in water; 15,771 ± 7068 and 1585 ± 1450 µg/g for Cu^{2+} and Zn^{2+} in the sediment), where neither fish nor parasites were present. Across the other sites, eight parasite species were identified. Seven species occurred on fish from the slightly polluted Lufira River (mean intensity (MI) of 31.28 ± 28.95 parasites per infested fish), while five were found in Lake Tshangalele (MI: 3.23 ± 2.89 parasites per infested fish), confirming the hypothesis. Three species, *Quadriacanthus halajiani*, *Q. domatanai,* and *Macrogyrodactylus clarii*, demonstrated potential as sensitive bioindicators of aquatic pollution in the region.

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1. Introduction

Fish parasites receive particular attention because they can cause diseases, reduce growth, increase mortality, and enhance susceptibility to secondary infections by bacteria, fungi, and other pathogens [1,2]. However, they can also be useful as an interesting scientific tool, for example, as a biological tag for the host's diet, phylogeny, biogeography, or systematics, or they can serve as bioindicators of pollution [3–5]. Regarding pollution, the accumulation of metals in surface water is a major environmental concern worldwide. It limits the quality of drinking water and threatens aquatic ecosystems through the accumulation of metals in organisms, potentially harming their health and leading to biodiversity loss [6,7]. Mining activities in the Katanga Copperbelt, in the Democratic Republic of the Congo (DRC), are often conducted without adherence to environmental protection standards, and mining effluents are often discharged into rivers without prior treatment [6,8,9]. A reported case is the hydrometallurgical complex of Shituru, where metal ores are mined and processed, and untreated effluents from the mining plant, containing Cu^{2+}, Co^{2+}, Zn^{2+}, Pb^{2+}, Cd^{2+}, U^{6+}, V^{5+}, and $As^{3+/5+}$, spill permanently into the Lufira River system [6]. The Shituru hydrometallurgical plant, located in the town of Likasi, was initially constructed in 1929 to produce copper and introduced the production of cobalt in 1947. Over the years, the facility has undergone several upgrades and expansions, producing up to 220,000 tons of copper and 9000 tons of cobalt annually [10,11]. Monitoring methods for the evaluation of the impact of pollution on aquatic systems include usually (i) the classical method consisting of the analysis of physico-chemical parameters and trace metal elements (TMEs) in water and sediment; (ii) the use of biomarkers (any alteration in the biological response of an organism, encompassing cellular, molecular, behavioural or physiological modifications, which is related to exposure to environmental toxicants or their biological effects); and/or (iii) the use of bioindicators (an organism or part of an organism or a community of organisms providing information on the quality of the environment or a part of the environment) [12–15]. The assessment of water pollution using classical or conventional methods is common. However, this approach does not provide sufficient information or comprehensive data on the state of aquatic ecosystems, as it does not directly integrate the full range of anthropogenic disturbances, such as habitat degradation and changes in flow that affect biological life [16,17]. Moreover, physico-chemical analyses of water reflect only a snapshot of conditions at the time of sampling, whereas aquatic organisms represent the effects of their environment as a result of long-term exposure [18]. Thus, the use of bioindicators is recognised as a cost-effective, reliable, simple, and integrated alternative, both spatially and temporally [14,19]. Organisms commonly used as biological indicators for the assessment of water quality include algae, macrophytes, zooplankton, insects, bivalve molluscs, gastropods, fishes, amphibians, parasites, and others [20]. Parasites have also become subjects of research in this field because of the variability of their responses to pollution [21,22]. In the last decades, increasing interest in the ecology of parasites in relation to environmental monitoring has demonstrated that they are sensitive towards the quality of the macroenvironment. This has given rise to a discipline named 'environmental parasitology', which deals with the interactions between parasites and pollutants in the environment [23,24]. Variable responses of parasites to pollutant exposure have been observed at the population and community level of various parasite groups, such as acanthocephalans, cestodes, nematodes, digeneans, and monopisthocoty-

lans [7,23,25,26]. Monopisthocotylans have been mentioned by Sanchez-Ramirez et al. [27] as a model for testing water quality. Blanar et al. [28] have stated that monopisthocotylans are good biological indicators of pollution in aquatic ecosystems due to their sensitivity to any changes in environmental conditions; being mostly ectoparasites, they are directly exposed to pollutants. Unfortunately, in Africa, the bioindication potential of parasites of even relatively well-studied fish is underexplored, although several monopisthocotylans that could serve as promising bioindicators have been identified [29]. Monopisthocotylan parasites are common parasitic flatworms, mostly infecting fish and sporadically aquatic invertebrates, amphibians, and reptiles, with a single species reported from mammals (hippopotamus) [30–36]. For fish, the infection sites of monopisthocotylans are typically gills, fins, and/or skin [37]. They can occasionally be found in the stomach, urinary bladder, intestine, oral or nasal cavity, eyes, and heart [38,39]. They are valuable for biological surveys due to their diversity, wide distribution, high host-specificity, and single-host life cycle [40]. The current study aims to assess water pollution using monopisthocotylan gill parasites of the blunt-toothed African catfish, *Clarias ngamensis* Castelnau, 1861, as bioindicators. Objectives are (i) to characterise the current level of contamination of the Upper Lufira River system by quantifying TMEs in water and sediment, and (ii) to explore the possibility that monopisthocotylans show different occurrence patterns according to environmental conditions. It is hypothesised that (i) parasite communities from a less polluted site are more diverse than those from a more polluted one; and (ii) infection parameters of monopisthocotylans from a less polluted site are higher than those from a more polluted one. In this context, parasite communities from the section of the Lufira River from the source to the confluence with the highly polluted Panda River (considered here as the main source of pollution) are expected to be more diverse, with higher infection parameters than those from Lake Tshangalele, downstream of the confluence. This expectation is based on the observation that pollution negatively affects aquatic ecosystems by decreasing ectoparasite diversity and load [7].

2. Materials and Methods

Study area

This study was conducted between September 2015 and September 2017 in the Lufira River system (Figure 1), one of the major tributaries in the upper section of the Congo Basin, called Lualaba [41–43]. The Lufira River is subdivided into three sections: the Upper Lufira [from the source of the river to Lake Koni (downstream of Lake Tshangalele)], the Middle Lufira (downstream of Lake Koni to the Kyubo Falls), and the Lower Lufira (downstream of the Kyubo Falls to the Kamalondo Depression, at the junction with the Lualaba River) [43,44]. To provide hydroelectric power, two successive dams were built in the Upper Lufira River in 1930 and 1949, which created two artificial lakes, respectively, Lake Tshangalele and Lake Koni [45–47]. This study focuses only on Lake Tshangalele, located about 35 km east of the town of Likasi (Haut-Katanga Province, southern part of the former Katanga Province). It provides a habitat to a variety of fish, and it is also a UNESCO Man and the Biosphere Reserve, rich in birdlife [48,49]. The section of the Lufira River, from the source to the confluence with the Panda River (which drains water containing effluents from the hydrometallurgical complex of Shituru via the rivers Likasi, Buluo and Kiantete), is considered a reference for a slightly polluted system. Lake Tshangalele, receiving water from all these rivers, is considered a more polluted system (Figure 1) [6].

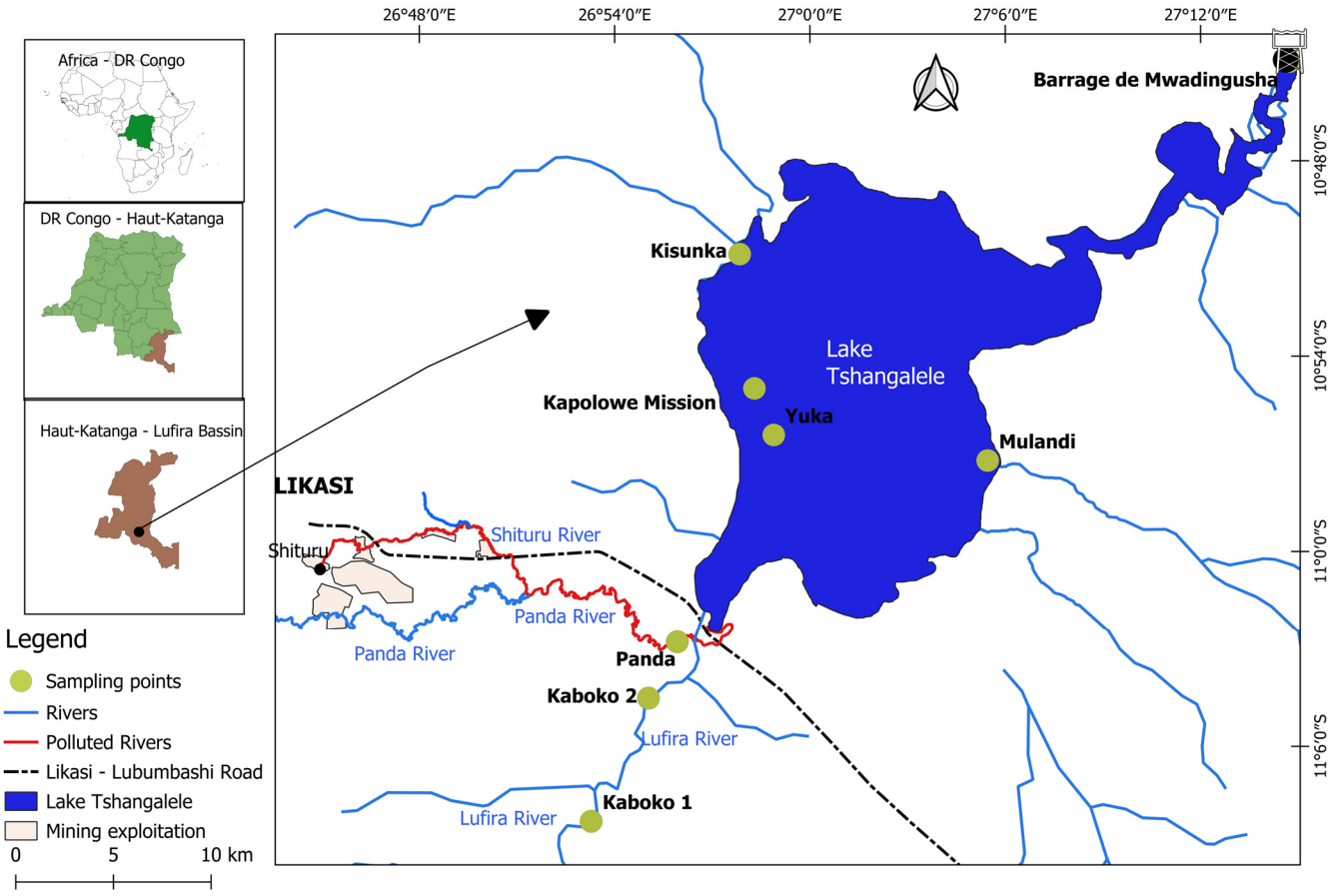


Figure 1. Map of sampling points in the Upper Lufira Basin: Lufira River (Kaboko1: 11°8′17.68″ S, 26°53′16.80″ E and Kaboko2: 11°4′31.60″ S, 26°55′2.40″ E); Lake Tshangalele (Kisunka: 10°50′52.10″ S, 26°57′50.60″ E, Kapolowe Mission: 10°54′59.50″ S, 26°58′17.70″ E, Yuka 10°56′25.30″ S, 26°58′53.40″ E, and Mulandi: 10°57′36.64″ S, 27°6′44.88″ E); and Panda: 11°2′42.99″ S, 26°56′31.8″ E. Polluted rivers in red.

Water and sediment sample collection

The pH and conductivity of the water were measured in the field using a multi-parameter (pH/conductivity/DO) handheld pHenomenal MU 6100 H—Multimeter (Medan, Indonesia) at each site (Figure 1). The electrodes were introduced into the water, and then the measured pH and conductivity were displayed on the screen. Three water samples were collected in 50 mL polypropylene vials that had been thoroughly rinsed with the same water to avoid contamination. Subsequently, in the laboratory, the samples were filtered through a 0.20 µm Millipore membrane to remove particulate matter, ensuring that only the dissolved TMEs in the liquid phase were analysed. From the resulting filtrate, 10 mL of water was acidified with 150 µL of 69% nitric acid (HNO_3) and stored in a freezer until further analysis. Distilled water was used as a reference material to monitor potential contamination during sample handling [50]. Surface sediment samples (±10 cm deep) were collected using a small Ponar grab at the same sites as the water samples. On each site, sediment samples were collected three times, homogenised on a tray, and manually cleaned of atypical elements, such as organisms, plant debris, and gravel. This process yielded three composite samples per site, collected in 50 mL vials after removing the supernatant water. The samples were sieved using a fine-mesh sieve (<2 mm) to discard coarse particles typically characterised by low metal-binding capacity, and to minimise the effect of granulometric variability [51]. The samples were kept at −4 °C and transported to the ECOSPHERE laboratory in Antwerp, Belgium. There, 0.5 g of each sediment sam-

ple was freeze-dried at −54 °C for 48 h. The dried sediment was weighed and digested using 500 µL of nitric acid and 1500 µL of hydrochloric acid (HCl), and then digested in a SP-Discover microwave (CEM, USA) in two steps. The initial step was carried out at 120 °C with a ramp time of five minutes and a hold time of five minutes, under a maximum pressure of 34 bars at 300 W with low stirring. The second step was conducted at 160 °C, maintaining the same ramp and hold times, pressure, power, and stirring conditions. For analysis, the samples were diluted to 5–6% acid concentration for Hg and to 1–2% acid for the remaining elements. Certified reference material (BCR 320R) was used to monitor potential contamination during sample handling and preparation [50].

Quantification of trace metal analysis by High-Resolution Inductively Coupled Plasma Mass Spectrometer

A total of 10 TMEs (Cd^{2+}, Hg^{2+}, Pb^{2+}, U^{6+}, V^{5+}, $Cr^{3+/6+}$, Co^{2+}, Ni^{2+}, Cu^{2+}, and Zn^{2+}) were analysed using a High Resolution Inductively Coupled Plasma-Mass Spectrometer (HR ICP-MS; Element XR, Thermo Fisher Scientific, Bremen, Germany). All measurements were conducted at medium resolution except for Hg due to its analytical sensitivity requirements. All concentration values were compared to the EU quality standards [52,53] and the South African water quality (SAWQ) guidelines [54] for water, and to the consensus-based sediment quality guidelines (SQGs) for freshwater ecosystems for sediment [55], in the present absence of regulations in the DRC.

Fish sampling

Clarias ngamensis (Figure 2), known as 'kilundwe' in Kilamba, a Bantu language spoken in DR Congo, was selected for this study given its economic value in the Upper Lufira Basin [49,56,57]. Fish specimens were collected using gillnets with a mesh size of 20–50 mm knot to knot or purchased from fishermen along the Lufira River and Lake Tshangalele's shores. The sampling was opportunistic, without determining the number of fish to be dissected in advance. Fish were kept alive in an aerated tank and transported to a field laboratory until they were processed. They were killed by severing the spinal cord just posterior to the cranium, immediately prior to examination, following Olivier et al. [58]. Fish were identified up to the species level using the keys compiled by Teugels [59].

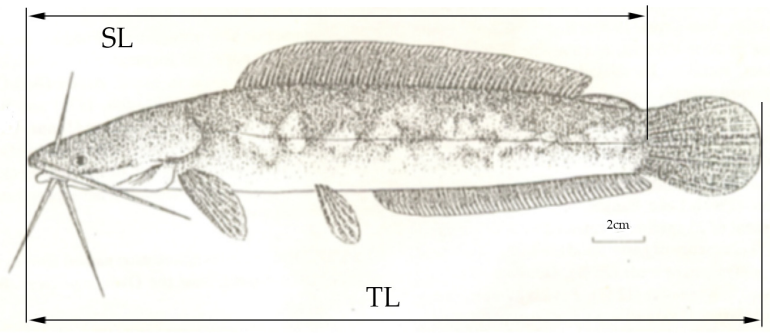


Figure 2. Morphometric picture of *Clarias ngamensis* (lateral view), indicating SL (standard length) and TL (total length), after Teugels [59] (p. 17, Figure 13A).

Parasite sampling

To collect monopisthocotylan parasites, fish were dissected and the right gill arches removed by dorso-ventral section. These were placed in a Petri-dish containing water from the corresponding site for examination using a stereomicroscope Optica 4.0.0 (OPTIKA Srl, Ponteranica, Italy). Parasites were dislodged from the gill filaments using entomological needles and fixed between slide and coverslip into a drop of ammonium picrate-glycerin, according to Nack et al. [60]. Twenty-four hours later, the coverslips were sealed using nail varnish.

Monopisthocotylan community composition, indices of diversity and infection parameters

Parasite identification using a Motic BA310 microscope (Motic, Speed Fair Co., Ltd., Hong Kong) and a phase-contrast microscope (model BX50; Olympus, Tokyo, Japan) was based on the morphology of the sclerotised parts of the posterior attachment organ and the genitals, and on a comparison with congeners [57,61–76]. Parasite diversity was summarised by the species richness index (S), the indices of Shannon (H), and Pielou's evenness index (E) [77].

The Shannon index is widely used to quantify biological diversity. Its calculation accounts for both species richness and the relative abundance of species, including rare taxa.

$$H = -\sum_{i=1}^{s} \frac{P_i \log_2 n_i.}{N}$$

where

i represents a species in the studied system;
S is the total number of species;
n_i is the number of individuals of species i;
N is the total number of individuals sampled (i.e., total abundance);
P_i corresponds to the proportion of individuals of species i (n_i/N).

The value of H is minimal or equal to zero when all individuals in the community belong to a single species, or when each species is represented by only one individual except for a single dominant species comprising all remaining individuals. H increases with increasing species richness and greater evenness in species abundances. The index reaches its maximum when individuals are evenly distributed among all species [77,78].

Pielou's evenness index represents the ratio between observed diversity and maximum possible diversity. It measures the degree of equitability in the distribution of individuals among species by accounting for their relative abundances [79].

$$E = \frac{H}{\log_2 S}$$

where

$\log_2$S represents the maximum diversity;
S is species richness;
H is the Shannon–Weaver diversity index [80].

Pielou's evenness index ranges from 0 to 1. It reaches its maximum when all species have identical abundances within the assemblage, whereas it is minimal when a single species dominates the entire assemblage. Evenness is considered low when $E < 0.6$, moderate when $0.6 \leq E < 0.8$, and high when $E \geq 0.8$. Low E values indicate that a small number of species account for the majority of individuals in the environment. When both H and E are high, the environment is considered non-specialised and isotropic, reflecting a homogeneous distribution of individuals among species. Conversely, low values of H and E indicate a specialised environment [79].

Infection parameters, i.e., prevalence (P), mean intensity (MI), and mean abundance (MA), were determined following definitions given by Margolis et al. [81] and Bush et al. [82] and assessed accordingly.

Prevalence is generally expressed as a percentage and represents the proportion of hosts infected by a given parasite species relative to the total number of hosts examined.

Based on prevalence values, parasite species can be classified as common or core (P > 50%), intermediate or secondary (10% ≤ P ≤ 50%), or rare or satellite (P < 10%) [83,84].

$$P = \frac{number\ of\ hosts\ infested\ by\ a\ given\ species}{total\ number\ of\ hosts\ examined}$$

Mean intensity corresponds to the average total number of parasites of a given species found in all hosts infected by that species and reflects the approximate number of parasite individuals harboured by an infected host. Mean intensity can be categorised as high (MI > 100), moderate (50 ≤ MI ≤ 100), low (10 ≤ MI < 50), or very low (MI < 10) [84].

$$MI = \frac{number\ of\ parasites\ of\ a\ given\ species}{total\ number\ of\ infested\ hosts}$$

Abundance is defined as the total number of parasites of a given species divided by the total number of hosts examined and represents the average number of parasites per host examined.

$$MA = \frac{number\ of\ parasites\ of\ a\ given\ species}{total\ number\ of\ hosts\ examined}$$

Abundance may also be expressed as a function of prevalence and mean intensity:

$$A = P \times MI$$

The Mann–Whitney U-test was used to compare the epidemiological indices between sites, and the Student's *t*-test was carried out to compare monopisthocotylan mean intensities and/or abundances between shared species, using the Past 3.1 software [80]. Voucher specimens were deposited in the collection of the Research Group Zoology: Biodiversity & Toxicology, at Hasselt University (Diepenbeek, Belgium) under accession numbers HU XXV.3.23–XXVI.3.22C.

3. Results

3.1. Level of Contamination in the Lufira River System

Table 1 shows that the Panda River has the lowest pH (7.4), although it remains within the alkaline range like the other sites, and the highest conductivity of 631 ± 98.4 µS/cm.

Table 1. pH and conductivity of water from different sites in the Lufira River system.

Site	pH	Conductivity (µS/cm)
Lufira River	7.97	463 ± 32.6
Panda River	7.40	631 ± 98.4
Lake Tshangalele	7.69	561 ± 55.8

Concentrations of the 10 TMEs determined in water and sediment from the Upper Lufira River system, presented in Tables 2–5, show the following:

(i) For water (Table 2): elevated values, exceeding the EU standards, were observed for U^{6+} and Zn^{2+} exclusively in the Panda River; for Cd^{2+} in both Panda River and Lake Tshangalele; and for Co^{2+} and Cu^{2+} in all sites. When compared to the target value of the SAWQ, all the aforementioned elements exceed in the same way as for the EU standards; however, the concentration of Zn^{2+} exceeds in all the sites.

Table 2. Concentrations (in µg/L) of trace metal elements in water compared to EU standards and to the target and criteria of the South African water quality (SAWQ) guidelines. With TWQR: target water quality range; CEV: chronic effect value; AEV: Acute effect value. In bold: values exceeding the EU standards. Cd (cadmium), Hg (mercury), Pb (lead), U (uranium), V (vanadium), Cr (chromium), Co (cobalt), Ni (nickel), Cu (copper), Zn (zinc). (–): absence of refence value.

TME	Lufira River	Panda River	Lake Tshangalele	EU Standards	SAWQ Guidelines		
					TWQR	CEV	AEV
Cd^{2+}	0.036 ± 0.007	**0.829 ± 0.021**	**0.300 ± 0.054**	0.250	0.150	0.300	3
Hg^{2+}	0.007 ± 0.001	0.006 ± 0.000	0.005 ± 0.000	0.070	0.040	0.080	1.700
Pb^{2+}	0.121 ± 0.005	0.085 ± 0.062	0.130 ± 0.103	7.200	0.200	0.500	4
U^{6+}	0.732 ± 0.005	**2.10 ± 0.022**	0.928 ± 0.328	1.000	–	–	–
V^{5+}	0.609 ± 0.008	2.01 ± 0.031	0.909 ± 0.633	4.000	–	–	–
$Cr^{3+/6+}$	0.061 ± 0.01	0.005 ± 0.00	0.005 ± 0.00	5	12	24	340
Co^{2+}	**16.6 ± 0.09**	**511 ± 0.86**	**22.0 ± 1.03**	0.500	–	–	–
Ni^{2+}	1.14 ± 0.014	1.22 ± 0.038	0.529 ± 0.197	20	–	–	–
Cu^{2+}	**8.32 ± 0.087**	**82.5 ± 0.200**	**9.72 ± 0.272**	7	0.3	0.53	1.6
Zn^{2+}	**2.80 ± 0.182**	**25.2 ± 0.837**	**4.27 ± 1.002**	20	2	3.6	36

Table 3 illustrates the potential relationships between different TMEs, based on their presence across the sampling sites. Mercury (Hg^{2+}), Pb^{2+}, and $Cr^{3+/6+}$ exhibit a negative correlation with Cd^{2+}, while U^{6+}, V^{5+}, Co^{2+}, Cu^{2+}, and Zn^{2+} show a positive correlation with the latter. Furthermore, V^{5+}, Co^{2+}, Cu^{2+}, and Zn^{2+} appear to have a negative correlation with Pb^{2+}. Vanadium, in particular, has a highly significant correlation with U^{6+}, while $Cr^{3+/6+}$ has a negative correlation with U^{6+} and V^{5+}. Cobalt, Cu^{2+}, and Zn^{2+}, as well, showed a negative correlation to $Cr^{3+/6+}$. Finally, Zn^{2+} showed a very high positive correlation with both Co^{2+} and Cu^{2+}.

Table 3. Correlations between trace metal elements in water. With *: significant ($p < 0.05$), **: highly significant ($p < 0.01$), ***: very highly significant ($p < 0.001$); (−) indicating negative correlation. Cd (cadmium), Hg (mercury), Pb (lead), U (uranium), V (vanadium), Cr (chromium), Co (cobalt), Ni (nickel), Cu (copper), Zn (zinc).

ETM	Cd^{2+}	Hg^{2+}	Pb^{2+}	U^{6+}	V^{5+}	$Cr^{3+/6+}$	Co^{2+}	Ni^{2+}	Cu^{2+}	Zn^{2+}
Cd^{2+}	1.00	−0.01	−0.11	0.91 **	0.94 **	−0.25	0.95 **	0.64	0.96 **	0.97 **
Hg^{2+}		1.00	0.00	0.00	0.03	0.93 **	0.10	0.69 *	0.12	0.11
Pb^{2+}			1.00	0.03	−0.02	0.24	−0.23	0.18	−0.16	−0.13
U^{6+}				1.00	0.99 ***	−0.17	0.92 **	0.69	0.93	094
V^{5+}					1.00	−0.14	0.91 **	0.72 **	0.93 **	0.95 **
$Cr^{3+/6+}$						1.00	−0.18	0.56	−0.15	−0.15
Co^{2+}							1.00	0.67	1.00	0.99 ***
Ni^{2+}								1.00	0.70	0.71
Cu^{2+}									1.00	1.00 ***
Zn^{2+}										1.00

(ii) For sediment: Table 4 shows that the Panda River exhibits concentrations of TMEs exceeding regulatory standards for Cd^{2+}, Hg^{2+}, Pb^{2+}, $Cr^{3+/6+}$, Ni^{2+}, Cu^{2+}, and Zn^{2+}. However, no reference values are available for U^{6+}, V^{5+}, and Co^{2+}. For the Lufira River, only Cu^{2+} exceeds the established standard. For Lake Tshangalele, Cd^{2+} and Cu^{2+} are the two TMEs with concentrations above regulatory thresholds.

Table 4. Concentrations (in µg/g) of trace metal elements in sediment compared to TEC and PEC. TEC = threshold effect concentration, below which adverse effects are not expected to occur; PEC = probable effect concentration, above which adverse effects are expected to occur more often than not [45]. BDL: below the detection limit (0.001). In bold: values exceeding the TEC or PEC. Cd (cadmium), Hg (mercury), Pb (lead), U (uranium), V (vanadium), Cr (chromium), Co (cobalt), Ni (nickel), Cu (copper), Zn (zinc). (–): absence of refence value.

	Sites			Standards	
ETMs	**Lufira River**	**Panda River**	**Lake Tshangalele**	**TEC**	**PEC**
Cd^{2+}	0.12 ± 0.03	**9.38 ± 1.45**	**21.7 ± 9.11**	0.99	4.98
Hg^{2+}	0.11	**1.07 ± 0.01**	BDL	0.18	1.06
Pb^{2+}	5.68 ± 3.33	**133 ± 66.4**	9.10 ± 1.49	35.8	128
U^{6+}	0.67 ± 0.05	26.8 ± 20.3	3.31 ± 0.09	–	–
V^{5+}	36.9 ± 3.79	172 ± 147	23.0 ± 10.4	–	–
$Cr^{3+/6+}$	33.6 ± 8.53	**43.5 ± 13.9**	13.8 ± 7.76	43.4	111
Co^{2+}	15.8 ± 0.13	1690 ± 877	940 ± 85.8	–	–
Ni^{2+}	17.4 ± 2.78	**55.0 ± 26.8**	19.4 ± 0.03	22.7	48.6
Cu^{2+}	**37.4 ± 6.40**	**15,771 ± 7068**	**474 ± 30.5**	31.6	149
Zn^{2+}	20.4 ± 4.36	**1585 ± 1450**	46.7 ± 3.07	121	459

Table 5 highlights the potential relationship among different TMEs, as inferred from their distribution across the different sampling sites. The concentrations of U^{6+}, V^{5+}, Cu^{2+}, and Zn^{2+} are strongly positively correlated with Pb^{2+}. Vanadium, Cu^{2+}, and Zn^{2+} have a very high positive correlation with U^{6+}. Copper and Zn^{2+} are strongly correlated to V^{5+}. Finally, Zn^{2+} has a particularly strong positive correlation with Cu^{2+}.

Table 5. Correlations between trace metal elements in sediment. With *: significant ($p < 0.05$), **: highly significant ($p < 0.01$), ***: very highly significant ($p < 0.001$). Cd (cadmium), Hg (mercury), Pb (lead), U (uranium), V (vanadium), Cr (chromium), Co (cobalt), Ni (nickel), Cu (copper), Zn (zinc).

ETM	**Cd^{2+}**	**Hg^{2+}**	**Pb^{2+}**	**U^{6+}**	**V^{5+}**	**$Cr^{3+/6+}$**	**Co^{2+}**	**Ni^{2+}**	**Cu^{2+}**	**Zn^{2+}**
Cd^{2+}	1.00	0.04	0.00	0.05	0.00	0.04	0.39	0.13	0.03	0.00
Hg^{2+}		1.00	0.93 **	0.87	0.88	0.89 *	0.78 *	0.97 **	0.93 **	0.89 **
Pb^{2+}			1.00	0.99 ***	0.99 ***	0.92 **	0.90 **	0.94 **	1.00 ***	1.00 ***
U^{6+}				1.00	1.00 ***	0.90 **	0.93 **	0.90 **	0.99 ***	1.00 ***
V^{5+}					1.00	0.93 **	0.91 **	0.92 **	0.99 ***	1.00 ***
$Cr^{3+/6+}$						1.00	0.81 *	0.96 **	0.93 **	0.91 **
Co^{2+}							1.00	0.86 *	0.91 **	0.91 **
Ni^{2+}								1.00	0.95 **	0.92 **
Cu^{2+}									1.00	0.99 ***
Zn^{2+}										1.00

3.2. Parasite Community Structure

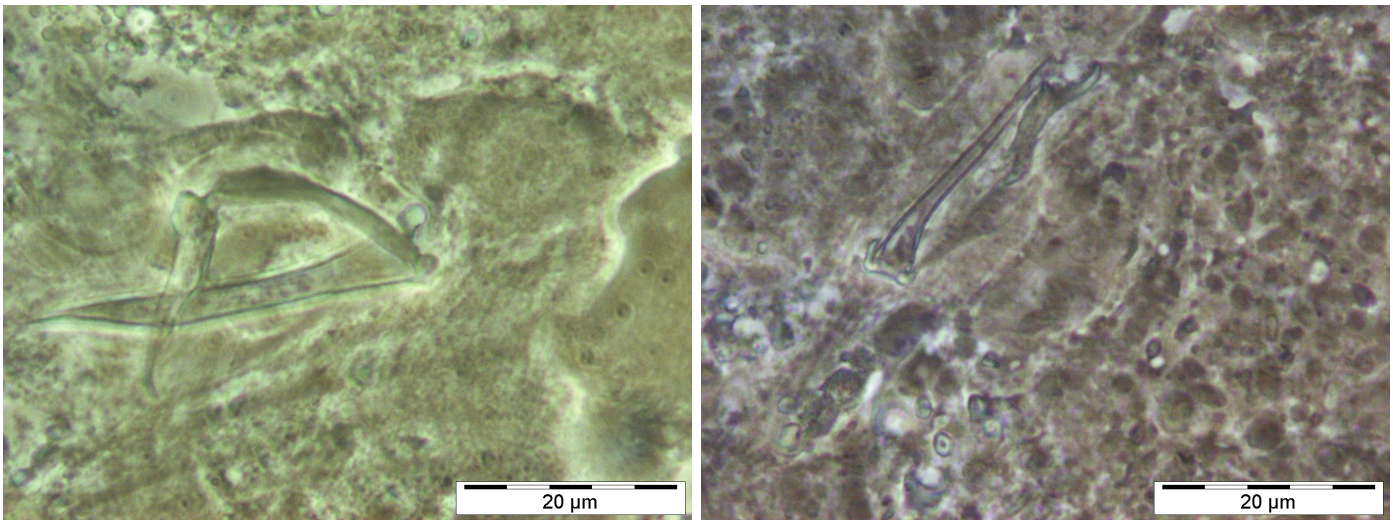
Clarias ngamensis (Figure 3) were collected from both the Lufira River and Lake Tshangalele (number of specimens: n = 36 and 24; the mean (±standard deviation) of total length = 35.5 ± 5.4 and 46.2 ± 11 cm; standard length = 31.3 ± 4.8 and 40.5 ± 9.9 cm; weight 432 ± 226 and 809 ± 522 g). No fish specimens were collected from the Panda River because the pollution has decimated all fish life in the river (personal observations, communication from fishermen, and setting the traps once with a fisherman).

(**A**) (**B**)

Figure 3. *Clarias ngamensis* (**A**,**B**) from the Lufira River collected during the present study.

3.2.1. Monopisthocotylan Community Composition and Its Indices of Diversity

The morphological investigation of monopisthocotylans led to the identification of eight parasite species: five species of *Quadriacanthus* Paperna, 1961 (*Q. halajiani* Kasembele, Bahanak and Vanhove, 2024 [57], *Q. lubandaensis* Kasembele, Bahanak and Vanhove, 2024, *Q. domatanai* Kasembele, Bahanak and Vanhove, 2024, *Q. shigoleyae* Kasembele, Bahanak and Vanhove, 2025 [76], and *Q. aegypticus* El-Naggar and Serag, 1986 [85]); one species of *Gyrodactylus* von Nordmann, 1832 (*G. turkanaensis* Přikrylová, Blažek and Vanhove, 2012 [70]), one species of Gyrodactylidae n. sp. (undescribed; provisionally named as such because it appears to represent a new genus; however, in the absence of molecular data, its taxonomic status cannot be confirmed); and one species of *Macrogyrodactylus* Malmberg, 1957 (*M. clarii* Gusev, 1961) (Figure 4, Tables 6 and 7).



(**A**) (**B**)

Figure 4. *Cont.*

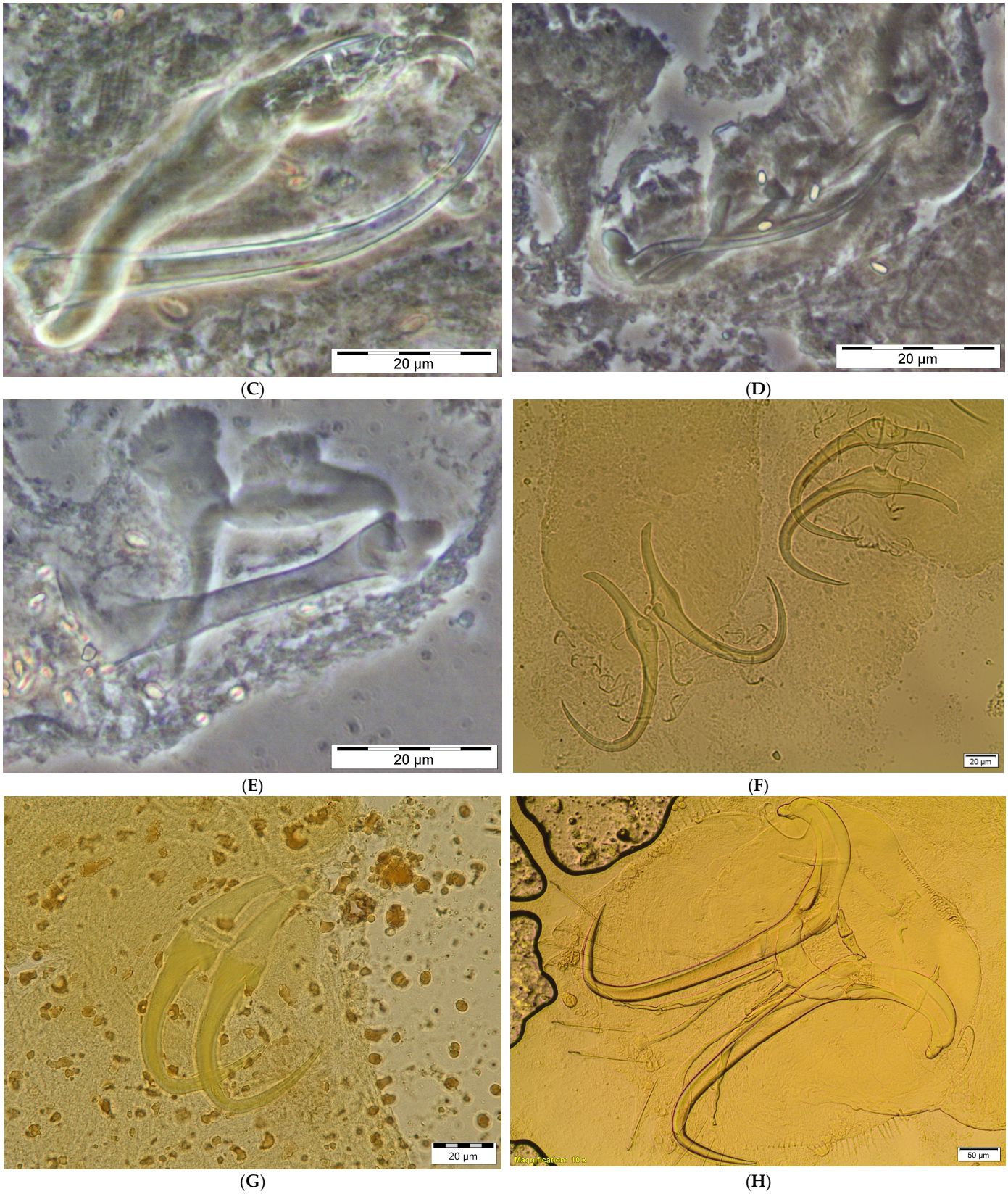


Figure 4. Photomicrographs of the sclerotised structures of the genitals and haptor of parasites of *Clarias ngamensis*: (**A**) *Quadriacanthus halajiani* from Lufira River (male copulatory organ); (**B**) *Quadriacanthus lubandaensis* from Lufira River (genitals); (**C**) *Quadriacanthus domatanai* from Lufira River (genitals); (**D**) *Quadriacanthus shigoleyae* from Lufira River (genitals); (**E**) *Quadriacanthus aegypticus* from Lake Tshangalele (genitals); (**F**) Gyrodactylidae n. sp. from Lufira River (haptor); (**G**) *Gyrodactylus turkanaensis* from Lake Tshangalele (haptor); (**H**) *Macrogyrodactylus clarii* from Lake Tshangalele (haptor). Scale bar 20 µm for (**A**–**G**); 50 µm for (**H**).

Table 6. Identification of monopisthocotylans recovered on the gills of *Clarias ngamensis* per site. With "X": species recorded; "–": species not recorded.

Parasite Classification				Site	
Order	**Family**	**Genus**	**Species**	**Lufira River**	**Lake Tshangalele**
Dactylogyridea Bychowsky, 1937	Dactylogyridae Bychowsky, 1933	*Quadriacanthus*	*Q. halajiani*	X	X
			Q. lubandaensis	X	–
			Q. domatanai	X	X
			Q. shigoleyae	X	–
			Q. aegypticus	X	X
Gyrodactylidea Bychowsky, 1937	Gyrodactylidae Cobbold, 1864	*Gyrodactylus*	*G. turkanaensis*	–	X
		Gyrodactylidae n. gen.	Gyrodactylidae n. gen. n. sp.	X	–
		Macrogyrodactylus	*M. clarii*	X	X

Table 7. Indices of diversity of monopisthocotylans recovered on the gills of *Clarias ngamensis* per site.

Indices of diversity	Site	
	Lufira River	**Lake Tshangalele**
Species richness (S)	7	5
Index of Shannon (H)	1.69	1.47
Evenness (J)	0.87	0.91

Quadriacanthus halajiani is recognised by the tube-shaped male copulatory organ (MCO), which is slightly curved. Its accessory piece (AP) is simple, thicker in its distal part and with a bulge and depression, respectively, on the external and internal sides of its median part and ending in a well-developed point. *Quadriacanthus lubandaensis* is identified based on the simple and straight copulatory tube of the MCO, with a thin margin, and the AP with a hook-like ending, not well developed and less curved, articulating with the copulatory tube. *Quadriacanthus domatanai* is diagnosed using the long, slightly curved copulatory tube, which is wide at its proximal end and tapered at the distal extremity. Its AP is large, robust and thickening in its median part, with a bulge on the external and internal faces, ending in a hook. *Quadriacanthus shigoleyae* is identified based on the femoral tube-shaped MCO, with a thickened AP ending in a tail-shaped extremity with a pointed ending [57,76]. *Quadriacanthus aegypticus* is recognised based on its tubular and straight copulatory tube, widest at its base and narrowing towards its distal extremity. Its AP ends in two distinctive lateral, club-shaped outgrowths projecting from its posterior half [57,85]. Gyrodactylidae n. sp. has a haptor armed with two large anchors, strongly curved, each consisting of a long shaft and a blade separated from the point by a small notch located on the inner side of the curvature. It has a proportionally short marginal hook handle and a long filamentous ventral bar membrane, like in *Gyrodactylus alberti* Paperna, 1973 and *Gyrodactylus nyongensis* Nack, Bilong Bilong and Euzet, 2005 [86]. The ventral bar is composed of a median part bearing two reniform lateral expansions, and it is posteriorly extended by a weakly sclerotised stoloniform process. This flatworm has sixteen marginal hooks and a rod-shaped dorsal bar. It is assigned to Gyrodactylidae van Beneden and Hesse, 1863, with a haptor circular to ellipsoid in outline, ventrally armed with 16 hinged (gyrodactylid) marginal hooks, a pair of ventral anchors supported by superficial, and deep bars with the deep root of ventral anchor knob-like; a superficial (ventral) bar usually with shield, double ribbons, and/or accessory sclerites; and a deep (dorsal) bar inserted into the deep roots of the anchors [87]. *Gyrodactylus turkanaensis* is recognised because of the slender

anchors, with a flattened area on the inner part of the root, which narrows substantially after joining the shaft. Marginal hook sickles are distinctive, thinner, and rise from the sickle foot at a forward angle. The sickle foot has a triangular profile and joins smoothly with the sickle proper [70]. *Macrogyrodactylus clarii* is identified based on the haptor bearing a single pair of robust anchors positioned centrally within the haptor. Eight pairs of marginal hooks are present. The dorsal bar is comparatively small and composed of two sclerites that appear to articulate along their inner margins, aligned in a straight configuration. The ventral bar exhibits a complex morphology, consisting of a Y-shaped sclerite accompanied by two pairs of relatively long, posteriorly directed, rod-like sclerites. In addition to the dorsal and ventral bars, two slightly curved accessory sclerites are present, each located in the anterolateral region of the haptor, in close association with the anterolateral marginal hooklet [88].

3.2.2. Infection Parameters of Monopisthocotylans

Prevalence, mean intensity, and abundance are presented in Figures 5–7.

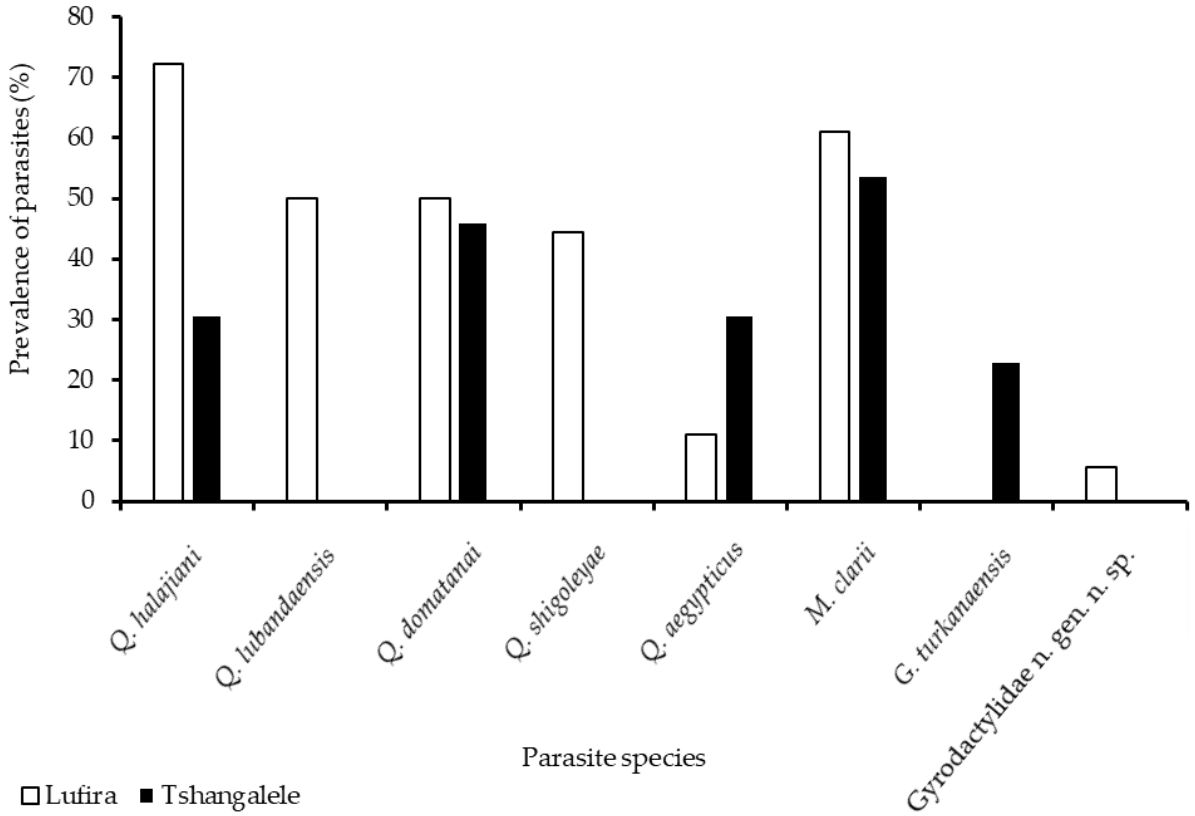


Figure 5. Parasite prevalence per monopisthocotylan species recovered on the gills of *Clarias ngamensis* in the Upper Lufira Basin, per site.

Figure 5 shows that four species (*Q. halajiani*, *Q. lubandaensis*, *Q. domatanai*, and *M. clarii*) were more prevalent in the Lufira River, each with P > 50%. In contrast, at Lake Tshangalele, only *M. clarii* reached a P > 50%. When all species are considered together, the total prevalence of monopisthocolylans in fish from both the Lufira River and Lake Tshangalele was 100%.

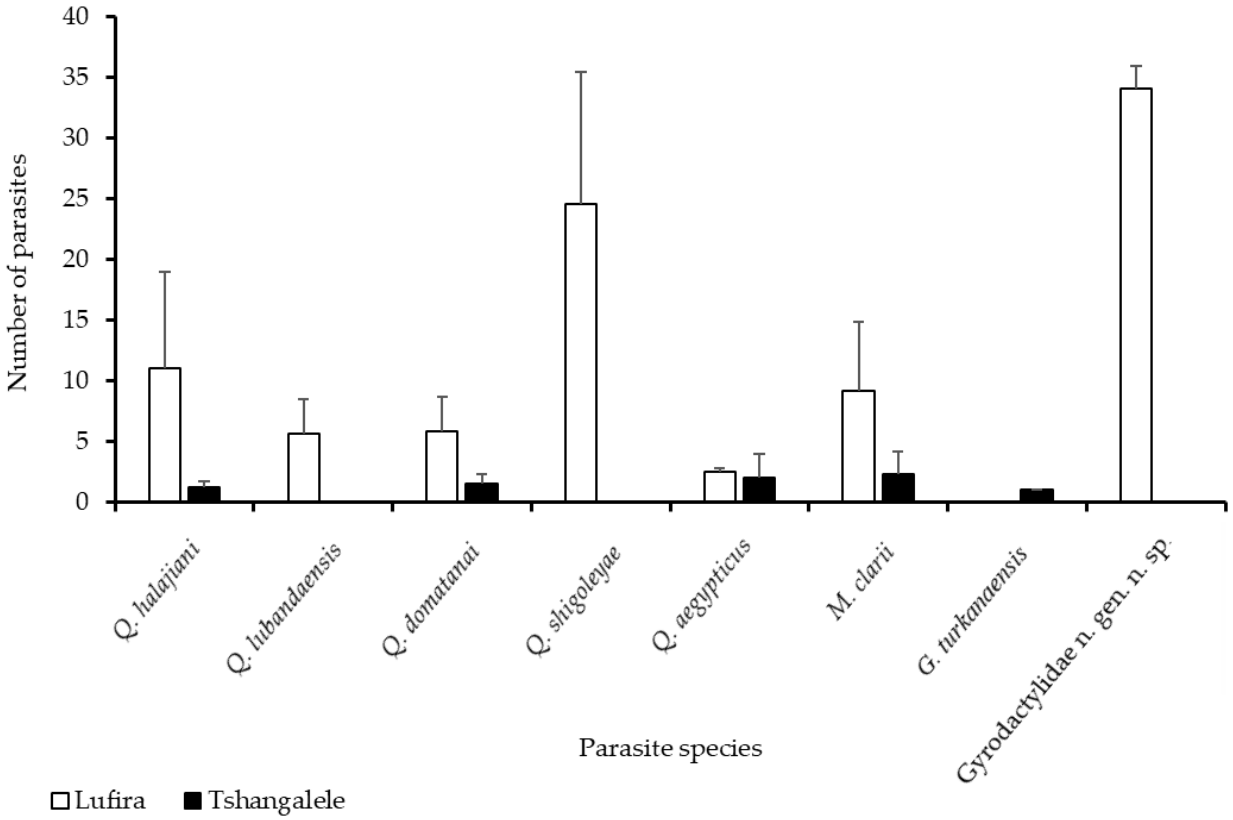


Figure 6. Mean intensity per monopisthocotylan species recovered from the gills of *Clarias ngamensis* in the Upper Lufira Basin, per site. Whisker above the mean indicates the standard deviation.

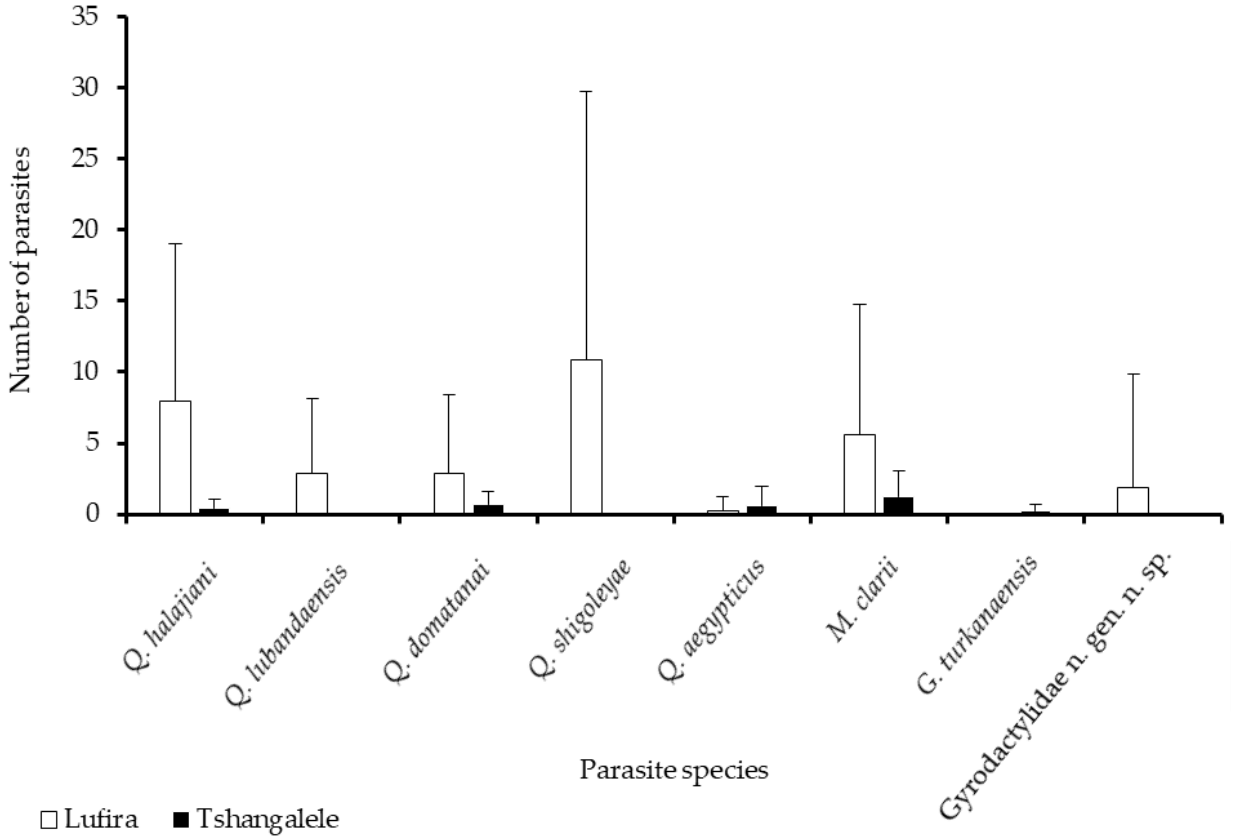


Figure 7. Abundance per monopisthocotylan species recovered on the gills of *Clarias ngamensis* in the Upper Lufira Basin, per site, with standard deviation.

Figure 6 shows that the mean intensities of individual species are low (MI < 10) in both systems, according to the threshold of Valtonen et al. [83], except for *Q. halajiani*, *Q. shigoleyae*, and Gyrodactylidae n. gen. n. sp. in the Lufira River, with mean intensities of 11 ± 7.94, 24.5 ± 10.9, and 34.0 ± 1.89 worms per infested fish, respectively. However, considering all parasite species together, the mean intensities are 31.3 ± 29.0 ($10 \leq MI < 50$) and 3.23 ± 2.89 (IM < 10), respectively, for the Lufira River and Lake Tshangalele.

Only *Q. shigoleyae* in the Lufira River reached more than 10 parasites per examined fish (Figure 7), while all other monopisthocotylans had abundances below 10 in both sites. For all parasite species together, the abundances are 31.3 ± 29.0 and 3.23 ± 2.89 for, respectively, the Lufira River and Lake Tshangalele.

The Mann–Whitney test applied to the mean intensities and abundances of monopisthocotylan species from the Lufira River and the Lake Tshangalele, reveals significant differences when comparing all species together (Mean Intensity: Mann–Whitney U = 0; $z = -3.037$; $p = 0.002 < 0.05$; Abundance: Mann–Whitney U = 4; $z = -2.517$; $p = 0.012 < 0.05$). When we compare the species shared by both sites, the test indicates significant differences only for *Q. halajiani* (t = 2.74; $p = 0.007$), *Q. domatanai* (t = 4.38; $p = 4.87 \times 10^5$), and *M. clarii* (t = 4.85; $p = 4.28 \times 10^6$) at the 95% threshold for mean intensities. In terms of abundance, the comparison between different shared species shows no significant difference. No difference was found in prevalence when comparing either all species or only the shared species.

4. Discussion

Assessing water quality is crucial. To understand the current status of the selected aquatic ecosystem in the Upper Lufira Basin in the DR Congo, we measured and quantified the physico-chemical characteristics of water, as well as TMEs in water and sediment. The monopisthocotylan community structure was also studied as a potential indicator for the impact of pollution on parasites in this aquatic environment.

4.1. Level of Contamination in the Lufira River System

The pH values obtained in this study ranged from slightly neutral to alkaline (7.40–7.97). Under this pH range, in this study, it is too early to predict the influence of water parameters on aquatic organisms since it has been indicated that more toxic effects of metals happen in acidic conditions, barring a few exceptions where an increase in pH towards the alkaline range increases metal toxicity [89–91]. Nevertheless, aquatic systems behave unpredictably, since their response to metals is governed by a multitude of complex interlocking factors [89]. Among these factors, the water conductivity could also influence the underlying toxicity of pollutants. The conductivity values measured in this study ranged from 463 ± 32.6 to 631 ± 98.4 µS/cm, the highest value recorded being from the Panda River. This is because the Panda River receives effluents from the Shituru hydrometallurgic complex in the Likasi region via the Likasi, Buluo, and Kiantete rivers. The discharge of contaminants from mining activities into streams leads to the deterioration of water quality and poses a substantial threat to aquatic ecosystems [92]. The 10 selected trace elements Cd^{2+}, Hg^{2+}, Pb^{2+}, U^{6+}, V^{5+}, $Cr^{3+/6+}$, Co^{2+}, Ni^{2+}, Cu^{2+}, and Zn^{2+} were recovered at various concentrations from water and sediment. They are often discharged from mine extractions and operations in the Upper Congo Basin, particularly around the town of Likasi [6,9]. Likasi is historically and contemporarily an important mining site where Cu–Co ores (comprising both sulfidic and non-sulfidic types), originating from sediment-hosted stratiform Cu–Co deposits and their supergene alteration products from across the Katanga Copperbelt, have been processed since 1929 [10]. The Shituru hydrometallurgical plant generates tailings that are deposited in two basins, the northern-

most of which remains active and discharges untreated drainage water and suspended solids into the Lufira River system. Although the Shituru hydrometallurgic complex is taken as the main source of pollution in the Upper Congo Basin, several other sources of pollution should be considered, even if their contribution can unfortunately not be properly assessed: artisanal mining pollution (which is illegal and uncontrolled) and "unreported" mining plants around Likasi. Our results indicate that in the Panda River, Cd^{2+}, U^{6+}, and Zn^{2+} concentrations exceed EU standards by a factor of three for Cd^{2+}, two for U^{6+} and slightly more than the standard for Zn^{2+} (Table 2). This situation is not new, as Katemo et al. [6] had also measured exceedances up to 22 µg/L, 15 µg/L, and 1410 µg/L, respectively, for Cd^{2+}, U^{6+}, and Zn^{2+} in the Panda River (which is well above the values found in the present work). It is also important to pay attention to Co^{2+} and Cu^{2+}, the concentrations of which are above the standards at all sampling sites. Concentrations of Co^{2+} exceed the EU standards at the Lufira River, the Panda River, and Lake Tshangalele by 30, 1000, and 40 times, respectively (Table 2). As for Cu^{2+}, concentrations in these three sites exceed the EU standards slightly for the Lufira River and Lake Tshangalele and 11 times for the Panda River. The Lufira River, which is assumed to be less polluted or only slightly polluted, curiously contains high values of Co^{2+} and Cu^{2+}. Among other factors, this could be due to the existence of another source of pollution upstream of the sampling site or a diffuse pollution evoked earlier by artisanal mining, or an unknown or understudied source, e.g., wind pollution from tailings. For the other elements (Hg^{2+}, Pb^{2+}, V^{5+}, $Cr^{3+/6+}$, Ni^{2+}), concentrations at all sites are below EU standards. Despite their low concentrations, these elements may still represent an environmental risk to aquatic organisms due to their potential for bioaccumulation (accumulation of substances, e.g., pollutants within organisms) and biomagnification (increasing pollutant concentrations along food webs with higher concentrations in organisms of higher trophic levels) within the trophic chain [93,94]. Furthermore, the results show that the zone of influence of mining pollution extends as far as Lake Tshangalele, although the values of certain elements (Hg^{2+}, Pb^{2+}, U^{6+}, V^{5+}, $Cr^{3+/6+}$, Co^{2+}, Ni^{2+}, Zn^{2+}) analysed in the sediment are low, suggesting that Lake Tshangalele plays the role of a decanter. This role should normally be played by the Shituru tailings disposal facility, which has been saturated since 1986 [11]. Sediment plays a substantial role in assessing environmental quality, as pollutants that may exist in low concentrations in the water column can be trapped and accumulated in large quantities in the sediment [50,95]. The results of this study are compared to the consensus-based SQGs for freshwater ecosystems [55] and other results from the Upper Congo Basin. In the Lufira River, the concentration of Cu^{2+} in the sediment exceeds the threshold effect concentration (TEC) but not the probable effect concentration (PEC) (Table 4). This could be interpreted as the river not undergoing increased anthropogenic pressure over the long term, as sediment studies can also integrate the temporal variability of the aquatic environment and help trace the contamination history of a site through the study of the different sedimentary strata deposited [96]. The Lufira River could be perceived as slightly polluted, instead of unpolluted. However, the Panda River has several metals in concentrations higher than the TEC and PEC, e.g., Cd^{2+}, Hg^{2+}, Pb^{2+}, Ni^{2+}, Cu^{2+}, and Zn^{2+}. There are no references reported for the SGQs for U, V, and Co, although they were measured in quite high concentrations in the present study. Observing the concentrations of metals in Lake Tshangalele, we noticed lower concentrations for almost all the TMEs compared to the Panda River, except for Cd^{2+}, which was surprisingly higher. Once again, this indicates that Lake Tshangalele could be considered a decanter. Zooming out, mining is conducted in the whole Katangese Copperbelt area, which boasts a vast wealth of mineral resources (copper, cobalt, uranium, zinc, etc.), and the former Katanga Province is facing pollution from industrial and artisanal mining activities, as well as from abandoned mines [97,98]. Very

high concentrations found in the sediment illustrate how impacted rivers are in the Upper Congo Basin: Atibu et al. [99] in Lubumbashi (Bangweulu-Mweru ecoregion) (µg/g) Cu^{2+}: 40,152, Zn^{2+}: 5463, Pb^{2+}: 10,321; Atibu et al. [8,97] in Kolwezi (Upper Lualaba ecoregion) Cu^{2+}: 209,827; 47,468; Pb^{2+}: 1165; 851.9; Mutombo et al. [100] in Likasi (Upper Lufira Basin) Cu^{2+}: 9975. These are double to a hundred times the values found in the present study.

4.2. Parasite Community Structure

The release of metals into freshwater ecosystems is a major concern, threatening organisms and strongly challenging sustainable development. To study the impact of pollution on aquatic organisms, *Clarias ngamensis* was selected in view of its presence and economic value in the region [56]. It plays a considerable role in supplying animal protein to the local population around the Upper Lufira Basin, including large towns such as Likasi (<20 km distance), but also Lubumbashi (>100 km). The investigation of its monopisthocotylan parasites indicated eight species (Figure 4, Table 6). These results increase the known number of monopisthocotylan parasites infesting *C. ngamensis* to 11, which now shares *M. clarii* and *G. turkanaensis* with *Clarias gariepinus* (Burchell, 1822) [57,70,76,101].

The two overall systems have four parasite species in common; three species are characteristic of the Lufira River, and one species was found only in Lake Tshangalele. While *Quadriacanthus* and *Gyrodactylus* species have previously been studied in the region, these findings represent the first record of *Macrogyrodactylus* and the Gyrodactylidae n. gen. n. sp. in the Upper Lufira Basin [76,102]. Representatives of *Macrogyrodactylus*, *Gyrodactylus,* and the unknown gyrodactylid are also recorded for the first time on *C. ngamensis*, from which only representatives of *Quadriacanthus* have been recorded to date [57,76,103]. Comparing the parasite diversity of the two sites, the Lufira River, representing the less-polluted environment, is more species-rich (S = 7) than Lake Tshangalele, the reference for the more polluted environment (S = 5). In terms of species distribution, the Shannon diversity index was higher in the Lufira River, with an evenness > 0.8, i.e., also high [77]. The first hypothesis, that the less-polluted or slightly polluted site is more diverse in species than the more polluted one, is confirmed. This is in line with the results and theories of Lafferty [104], Lafferty and Kuris [105], Sanchez-Ramirez et al. [27], and Kouadio et al. [106], who found that polluted sites are less diverse in ectoparasite species than unpolluted sites. The second hypothesis pertained to infestation levels: fish in slightly or less polluted sites are more parasitised than those in polluted sites. For the prevalence, this second hypothesis cannot be immediately confirmed when considering all parasites together, since all fish sampled in the Lufira River and Lake Tshangalele were infested with at least one parasite, bringing prevalence to 100% in both sites. Comparing the prevalence of individual species shared between the two sites, the prevalence of the parasite species in the Lufira is higher than that of the monopisthocotylans in Lake Tshangalele, except for *Q. aegypticus*. This is in line with the second hypothesis. Similar situations were observed on *C. gariepinus* in the upper Manyame catchment, a subtropical African river system in Zimbabwe, by Madanire-Moyo and Barson [107] and Barson et al. [108], who observed low prevalences of monopisthocotylans in polluted environments compared to unpolluted environments. Kouadio et al. [106] also found a low infestation rate of monopisthocotylans on *C. gariepinus* in the Bagoué River in the Ivory Coast, with the heavily impacted zone (affected by gold panning) showing a lower infestation rate compared to the non-impacted reference zone. Blanar et al. [28] recognise in a quantitative meta-analysis that responses of monopisthocotylans to pollutants differ between different genera: *Dactylogyrus* sp. increase in number and prevalence, *Paradiplozoon* sp. decrease, and *Gyrodactylus* sp. tend to hold steady. Further, Cavalcanti et al. [109] assert that on the same host, different monopisthocotylan species exhibit either an increase or a decrease

in their populations in response to changes in physico-chemical conditions. A positive correlation was observed for the abundance of *Cichlidogyrus tilapiae* Paperna, 1960, with dissolved oxygen, while the abundance of *C. sclerosus* Paperna and Thurston, 1969, *C. thurstonae* Ergens, 1981, and *Scutogyrus longicornis* (Paperna and Thurston, 1969) was negatively correlated with dissolved oxygen. Our results also confirm that responses of monopisthocotylans differ between parasite species. Another way of analysing pollution is to focus on species that are present in less-polluted sites, but absent in polluted sites, as in the case of *Q. lubandaensis* (P = 50%), *Q. shigoleyae* (P = 44%), and Gyrodactylidae n. gen. n. sp. (P = 6%), which are present in the Lufira and not in Lake Tshangalele. Their absence from the lake may reflect a sensitivity to pollutants and environmental conditions, and as a result, may afford them the status of pollution indicator species or good candidates for pollution assessment. An in-depth study of their ecology is needed to consolidate this proposition. However, it is also important to understand why *G. turkanaensis* (P = 23%) is present in Lake Tshangalele and not in the Lufira River. Referring to Barson et al. [108], we find almost similar situations where three species from the same three genera (*Gyrodactylus rysavyi* Ergens, 1973, *Macrogyrodactylus karibae* Douëllou and Chisawa, 1995, and *Quadriacanthus clariadis* Paperna, 1961) behaved differently in a host congeneric to the fish studied here, *C. gariepinus,* in three distinct sites: a first site under industrial pollution (I), a second site under urban pollution (II), and the third one an unpolluted site (III). On the one hand, *G. rysavyi* is totally absent in site II with urban pollution, whereas it is present in site III, the unpolluted site, at a lower (P = 33.3%) prevalence than in site I, with industrial pollution, where prevalence is 40%. On the other hand, *M. karibae* has a lower prevalence (13.3%) in site III compared to site I with P = 50% and site II with P = 40%. Finally, in site I, *Q. clariadis* is absent, whereas in sites II and III, prevalences are equal, 25%. Barson et al. [108] grouped all fish parasite species and obtained high prevalences and species richness in the unpolluted environment and low values in polluted environments. Madanire-Moyo and Barson [107] found prevalences of 18.2% and 23.5% for *M. clarii* on *C. gariepinus* in unpolluted sites, while it was absent in the site polluted by industrial discharges. They concluded that the decrease in prevalence is a consequence of pollution. For mean intensities and abundances, in our study, parasite infestation is low in Lake Tshangalele compared to the Lufira River, corroborating the hypothesis that these indices are low in polluted sites. Barson et al. [108] and Madanire-Moyo and Barson [107] came to the same results and conclusions. In the present study, three species (*Q. halajiani*, *Q. domatanai*, and *M. clarii*) showed significant differences in mean intensities and abundances and, therefore, are here proposed as potential indicator species in pollution assessment studies. Indeed, these species responded as pollution impact indicators in line with our second hypothesis.

5. Conclusions

The results show the Lufira River itself is less polluted among the studied sites (pH: 7.97; conductivity: 463 ± 32.6 µS/cm; Co^{2+} and Cu^{2+} concentrations in water: 16.6 ± 0.09 and 8.32 ± 0.087 µg/L, respectively; and in sediment: 15.8 ± 0.13 and 37.4 ± 6.40 µg/g, respectively). It converges with the Panda River (carrier of TMEs; pH: 7.40; conductivity: 631 ± 98.4 µS/cm; Co^{2+} and Cu^{2+} in water: 511 ± 0.86 and 82.5 ± 0.200 µg/L; and in sediment: 1690 ± 877 and 15,771 ± 7068 µg/g, respectively). The combined waters flow downstream into Lake Tshangalele, which is more polluted than the Lufira River (pH: 7.69; conductivity: 561 ± 55.8 µS/cm; Co^{2+} and Cu^{2+} in water: 22.0 ± 1.03 and 9.72 ± 0.272 µg/L; and in sediment: 940 ± 85.8 and 474 ± 30.5 µg/g, respectively). Three monopisthocotylan species, *Q. halajiani* (P: 72%; MI: 11 ± 7.94; A: 7.94 ± 11.06 vs. P: 31%; MI: 1.25 ± 0.5; A: 0.38 ± 0.65), *Q. domatanai* (P: 50%; MI: 5.78 ± 2.89; A: 2.89 ± 5.51 vs.

P: 46%; MI: 1.5 ± 0.84; A: 0.69 ± 0.95), and *M. clarii* (P: 61%; MI: 9.18 ± 5.61; A: 5.61 ± 9.16 vs. P: 54%; MI: 2.29 ± 1.89; A: 1.23 ± 1.79) differ in infection parameters between the Lufira River and Lake Tshangalele, respectively, which we suspect is due to the influence of pollution. Changes in diversity were observed, such as lower species richness and epidemiological indices in Lake Tshangalele (more polluted environment; S= five parasite species; MI: 3.23 ± 2.89 parasites per infected fish) compared to the Lufira River (less polluted environment; S= seven parasite species; 31.3 ± 29.0 parasites per infected fish). We, therefore, suggest that these monopisthocotylan parasites could be used as tools for monitoring ecosystem degradation. As the choice of host species remains dependent on their presence and interest (economical or ecological value), we suggest future studies to be carried out on *C. ngamensis*, as well as other fish species, in the Upper Lufira Basin or elsewhere in the former Katanga Province and the DR Congo as a whole, which is permanently under threat of pollution given its mining richness. It would be advisable to also carry out in vitro experiments to draw generalising conclusions on the impacts of pollutants on fish parasite diversity and infection parameters, as in the laboratory, several factors could be controlled. Finally, we recommend the use of other parasite groups as bioindicators of pollution for future studies because different groups of parasites behave differently towards pollution.

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