ORIGINAL ARTICLE

Impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River Basin in Ethiopia

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ABSTRACT

The Koka hydropower dam is one of the oldest large dams in Ethiopia. Damming is one of the anthropogenic activities impacting the distribution of aquatic life forms. However, to date, little attention has been focused on the dam's impacts on the river macroinvertebrate assemblages in Ethiopia. The objective of this study was, therefore, to assess the impacts of the Koka hydropower dam on macroinvertebrate assemblages in the Awash River basin in Ethiopia. In the three river reaches on the Awash River (upstream near the source of the river, midstream above the dam, and downstream below the dam), a total of 15 sites were selected for sampling. The statistical analysis tested the null hypothesis that there are no differences in macroinvertebrate assemblage patterns or a range of univariate metrics between the three river reaches. Additional analyses involved the identification of taxa responsible for significant differences in macroinvertebrate structure (*e.g.*, percentage similarity) and an exploration of the variables that structure macroinvertebrates (*e.g.*, canonical correspondence analysis). In the upstream, midstream, and downstream reaches of the Awash River, we recorded a total of 73 taxa belonging to 43 families and 12 orders. Trichoptera was the dominant order in the upstream river reach, whereas Diptera dominated the midstream and downstream river reaches. The diversity of macroinvertebrates decreased from upstream to midstream and downstream. The three river reaches differed significantly in Shannon and Simpson diversity indices, % EPT, EPT taxa abundance, total taxa richness, evenness index, % collectors, and % scrapers. In this study, we observed that macroinvertebrate assemblage differences and spatial patterns were significantly associated with values of river flow changes (velocity), phosphate concentration, and substrate index. The findings of this study have broad implications for the assessment of the impacts of dam construction on the rivers of the studied region in the future.

INTRODUCTION

Dams are structures that restrict water flow and create reservoirs to provide specific human needs (Schmutz and Moog, 2018; Winton *et al.*, 2019) such as easy navigation, lessen flooding, and water supply (hydropower generation and irrigation, and potable water) (Schmutz and Moog, 2018; Zhang and Gu, 2023). Hydropower dams are among the most detrimental anthropologic activities in

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river basins by altering the physiography of watersheds and flow regimes, presenting barriers to fish migration and sediment movement, and impacting other aquatic species and their habitats (Schmutz and Moog, 2018; Englmaier *et al.*, 2020). Zhang and Gu (2023) noted that dams are among the earliest types of anthropogenic infrastructure and have made significant contributions to economic progress throughout human history. According to Schmutz and Moog (2018), there are currently more than 58,400 large dams being constructed, operated, or planned worldwide. The World Commission on Dams (2000) defined large dams as those that are "at least 15 meters high from the lowest foundation to the crest, or a dam between 5 and 15 meters high impounding more than 3 million cubic meters".

Since the 1970s, dam construction has decreased in the vast majority of developed countries such as North America, Europe, and Oceania, whereas it has increased in developing countries in Africa, Asia, and South America (Zhang and Gu, 2023). Emerging nations, such as China and India, have tremendous hydropower potential as well (Gernaat *et al.*, 2017). The World Commission on Dams (2000) estimated that 160–320 new large dams are constructed annually around the world. Ethiopia, located in Africa, has the second-largest population after Nigeria (Hagos *et al.*, 2022) and depends on rain-fed agriculture with limited coverage of electricity for many communities (Hagos *et al.*, 2022). To address electricity and water demands, the Ethiopian government is building dams on major rivers (Hagos *et al.*, 2022). The construction of the Grand Ethiopian Renaissance Dam, Africa's largest, on the Nile River Basin is one of approximately 200 places in the country where hydropower development is feasible (Degefu *et al.*, 2015; Hagos *et al.*, 2022). The Koka hydropower dam, abbreviated KHD, was built in the 1960s on the Awash River and is one of Ethiopia's oldest dams (Degefu *et al.*, 2015; Bussi *et al.*, 2021).

Natural variations in river flow regimes related to climatic conditions, geology, and geography of the watershed are important for the long-term ecological integrity of the river ecosystems (Poff *et al.*, 1997). However, alterations to natural flow regimes have far-reaching ecological as well as social and geopolitical consequences (World Commission on Dams, 2000). According to Poff *et al.* (1997), changes in the natural flow regimes are the key driver of river ecosystem structure and influence sediment transport, capturing 25–30% of pre-disturbance discharge (Schmutz and Moog, 2018). Schmutz and Moog (2018) also identified that damming affects 48% of rivers worldwide, and the impacts might even trickle far downstream and compromise the health of the river ecosystem (Degefu *et al.*, 2015). In developing nations such as China, the impact has increased rapidly (Wang and Chen, 2010). Overall, dams are of concern as they modify the natural flow regimes and morphodynamical patterns of rivers, adversely affecting productivity, biodiversity, and ecosystem services in the downstream river and also in the associated reservoir (Nilsson *et al.*, 2005; Mbaka and Wanjiru, 2015; Ko *et al.*, 2020). Dams are also criticised because of the large area that the reservoirs cover and are hence responsible for the displacement of human communities from their original places. These problems are more severe in developing countries such as Ethiopia because of the lack of enough financial resources to equitably resettle people displaced by the projects (Reis *et al.*, 2011; Degefu *et al.*, 2015).

Using biological metrics derived from aquatic organisms, it is possible to assess the ecological health of rivers (McRae *et al.*, 2017; Deinet *et al.*, 2020), particularly to explore human-induced disturbances such as dams (Poff *et al.*, 1997). Macroinvertebrates represent a diverse group of relatively long-living taxa with limited mobility that react strongly and often predictably to human influences on aquatic systems (Cairns and Prall, 1993). As mentioned, many hydropower dams have already been built in Ethiopia, and many more are being planned (Degefu *et al.*, 2015). However, limited studies in the country have investigated the impact of hydropower dams on benthic macroinvertebrate assemblages. The goal of this study was, therefore, to assess the impacts of the KHD on macroinvertebrate assemblages along the Awash River in Ethiopia. To achieve this goal, i) we assessed the responses of macroinvertebrate assemblages to river damming; ii) we identified the key environmental factors that significantly affect macroinvertebrate assemblages along the Awash River. In view of the above objectives, we hypothesized that there are no macroinvertebrate assemblage differences between river reaches upstream, midstream, and downstream of the KHD.

METHODS

Study area and sampling sites

The Awash River basin, which covers an area of 112,696 square kilometers, is the most important and industrialized catchment in Ethiopia (Englmaier *et al.*, 2020). The Awash River in this basin originates near Entoto Mountain (Bussi *et al.*, 2021) and drains into Lake Abe on the Ethiopia-Djibouti border by crossing 1,250 kilometers northeast across the KHD (Englmaier *et al.*, 2020). The KHD, built in 1960s, is 42 meters high and has 42 megawatts of installed capacity (Degefu *et al.*, 2013). Three sampling sections were defined with 5 sites in each: upstream (US1-US5, near the source of the river), midstream (MS1-MS5, above the KHD), and downstream (DS1-DS5, below the KHD) river reaches (Fig. 1).

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Macroinvertebrate data collection, sampling, and identification

At each of the 15 sites, macroinvertebrate samples were collected in the dry season (from the middle of March to the middle of April), as this is a period when water levels are sufficiently low to enable sampling (Montana Department of Environmental Quality, 2012). According to Barbour *et al.* (1999), the most effective sampling season of the year is an important consideration for selecting an index period. In Ethiopia, there is a wet season (June to September) and a dry season (October to May), and almost all previous studies in Ethiopia sampled surface water during the dry season (Degefu *et al.*, 2013; Englmaier *et al.*, 2020). Samples of macroinvertebrates were collected using a rectangular frame pond net (20 x 30 cm) with a mesh size of 0.5mm (Barbour *et al.*, 1999). A 2-minute kick sample was collected along a 10-meter stretch of river at each site (Getachew *et al.*, 2022). The samples, preserved in 80% ethanol in plastic bags, were taken to the laboratory, where they were rinsed through 0.5 mm mesh sieves. Macroinvertebrates were sorted into broad taxonomic groups for later identification and stored in 80% ethanol. Each macroinvertebrate specimen was then identified at the lowest possible taxonomic level (usually genus or species). Based on similar climatic conditions relative to other non-tropical regions, we used identification keys developed for South Africa (Harrison, 2009; Lowe, 2009; Schael, 2010).

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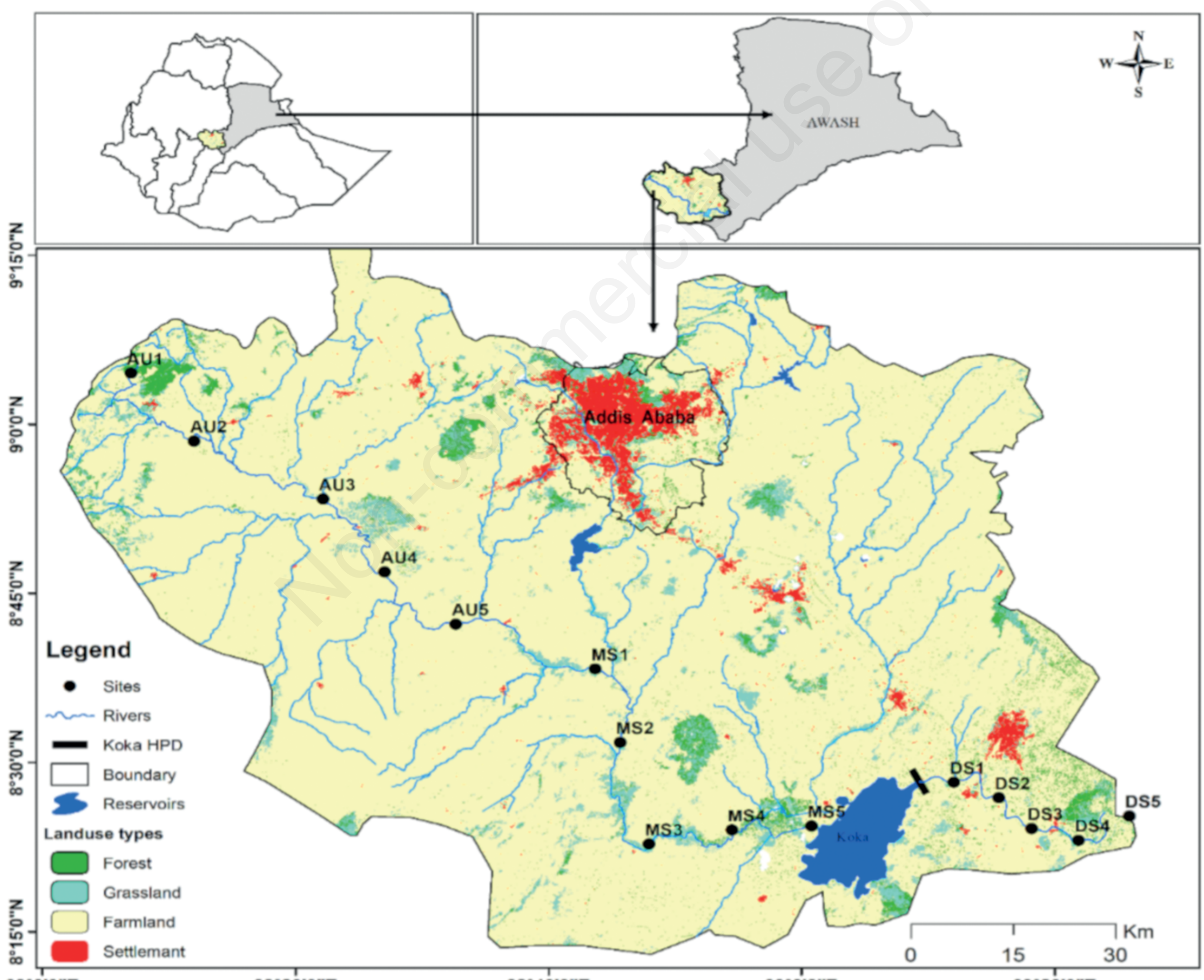

Fig. 1. The locations of the study area, sampling sites, Koka Reservoir, KHD, rivers, and the main land use categories along the upper Awash River basin in central Ethiopia. ArcGIS software version 10.5 was used to produce the map. The Water and Land Resources Information System (WALRIS) at www.wlrc-eth.org, and MapServer Ethiopia, both of which are located at www.mapserver-ethiopia.org were used to gather all raster and shape file data.

Environmental data collection

The river depth (m) and river velocity (m s^{-1}) were measured at each site. Out of the three depth measurements recorded at equal intervals across the wetted river bank width, these were used to determine the maximum river depth for use in the statistical analysis. Surface water velocity was measured using the techniques for estimating stream velocity as described in the United States Environmental Protection Agency (2017) protocol. The spatial locations and altitudes of the study sites were recorded using a global positioning system (GPS). At each site, the percentage compositions of the benthic substrates were estimated using the following particle size scales: sand (0.06-2 mm in diameter), fine gravel (2-10 mm), gravel (10-64 mm), cobble (64-256 mm), boulder (>256 mm), and bedrock (solid rock surfaces) (Jowett and Richardson, 1990). Substrate measurements were then transformed into a single variable by summing weighted substrate percentages to form a substrate index (SI). SI = 0.08% bedrock + 0.07% boulder + 0.06% cobble + 0.05% gravel + 0.04% fine gravel + 0.03% sand. The optimum substrate index for many invertebrates was just under cobble size (SI = 6) (Jowett and Richardson, 1990). Dissolved oxygen (DO), pH, temperature, and electrical conductivity (EC) were measured *in situ* on the same date using a digital handheld portable multi-parameter (HACH HQ40D probe). The turbidity of water was measured using a turbidity meter. In the field, 1.0 litre of water sample was collected from each site using acid-cleaned polyethylene containers at a depth of 20-30 cm and fixed with 3 drops of concentrated sulfuric acid. The samples were then transported to the laboratory for analysis. In the laboratory, nitrate-nitrogen (mg L^{-1}) and phosphate (mg L^{-1}) concentrations and biochemical oxygen demand (BOD_5) were analysed using the standard methods described in APHA (2005). To avoid disturbance of sites before sampling, measurements were undertaken from the downstream to the upstream direction.

Metric selection

Metrics that are relevant to the ecology of rivers within the region were selected to address several attributes of macroinvertebrate assemblages after Barbour *et al.* (1999). These included diversity measures such as evenness, Simpson, and Shannon indices (Nathan *et al.*, 2017), composition measures, including total taxon richness, EPT richness, and %EPT abundances (Lenat and Penrose, 1996), and functional approaches that reflect ecological integrity based on the information on both the structure and function of aquatic ecosystems (Cummins, 1974; Sumudumali and Jayawardana, 2021). We determined the functional traits for each taxon using the functional feeding groups in Dudgeon (1999), Merritt *et al.* (2002), and Merritt *et al.* (2006) (Tab. 1).

Data analysis

Biological data analysis

Univariate and multivariate tests were carried out using data on the composition, diversity, and functional measures of the macroinvertebrate assemblages. Under the assumption that the samples were selected at random and the observations were independent of one another, the Kruskal-Wallis test was used to determine whether there were significant differences among the three river reaches in terms of the univariate metrics (Tab. 1). The *post-hoc* analysis, using the Mann-Whitney U test in SPSS version 27, was performed on statistically significant ($p<0.05$) metrics from the Kruskal-Wallis test.

We used a one-way analysis of similarity (ANOSIM) to test the null hypothesis that there were no macroinverte-

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Tab. 1. Selected metrics used to assess the water quality of the Awash River and the impacts of the Koka hydropower dam and their description.

Metrics	Description	Metric type	References
% EPT	% EPT families in an ecosystem	Composition	Lenat and Penrose (1996)
EPT taxa	Total number of families belonging to the EPT orders	Composition	Lenat and Penrose (1996)
Total taxa	Number of different taxa represented in an ecological community	Composition	Nathan *et al.* (2017)
Evenness	The similarity of frequencies among units making up a population	Diversity	Nathan *et al.* (2017)
Simpson Index	A metric accounting for species richness and their relative abundance	Diversity	Nathan *et al.* (2017)
Shannon Index	A metric accounting for the total number of species and their distribution	Diversity	Nathan *et al.* (2017)
% Collectors	% of organisms that collect FPOM from the stream bottom	Function	(Merritt *et al.*, 2002)
% Predators	% of organisms that feed on other consumers or capture live prey	Function	(Merritt *et al.*, 2002)
% Shredders	% of animals feed on coarse terrestrial plant litter or aquatic macrophytes	Function	(Merritt *et al.*, 2002)
% Filterers	% of organisms feed on FPOM from the water column by variety of filters	Function	(Merritt *et al.*, 2002)
% Scrapers	% of organisms feed on live plants & particulates from substrate surfaces	Function	(Merritt *et al.*, 2002)

EPT, Ephemeroptera, Plecoptera, and Trichoptera.

brate assemblage differences at the locations of the upstream, midstream, and downstream river reaches. Macroinvertebrate data were transformed to log (x+1) to increase homoscedasticity, normality, and linearity (Clarke *et al.*, 2008). The degree of group separation for those samples represented by the similarity matrix can be assessed using the global R statistic (Clarke *et al.*, 2008). When R is closer to 1, there is good separation among the groups, and a value closer to zero shows weak separation of the groups (Clarke and Gorley, 2006). Similarity percentage (SIMPER) was used to identify the taxa that accounted for the most dissimilarity among the groups when significant differences in macroinvertebrate assemblages were found. Non-metric multidimensional scaling (NMDS) was used to visualise the separation of the sites in the three river reaches. According to Clarke and Gorley (2006), an NMDS ordination with a stress value of 0.1 is generally regarded as fair, and values below this indicate an excellent match. Furthermore, hierarchical cluster analysis of samples, known as linkage tree, was carried out based on the macroinvertebrate assemblage data (Clarke *et al.*, 2008). Cluster analysis explores groups that naturally occur within a data point and does not require the organization of data points into any predefined groups, which we also utilised to depict the groupings of the sites throughout the river reaches (Clarke *et al.*, 2008). PRIMER software was used to implement all of these multivariate analyses (Clarke and Gorley, 2006).

Environmental data analysis

We used the Kruskal-Wallis test to detect differences in the environmental variables between the three reaches in order to obtain comparable results with biological data. Finally, the relationships between environmental and biological variables were assessed using CANOCO 4.5. After preliminary Detrended Correspondence Analysis (DCA), we determined Canonical correspondence analysis (CCA) to be the most appropriate analysis based on the gradient length (Ter Braak and Wiertz, 1994). Except for pH, all environmental variables were square-root transformed (Clarke and Gorley, 2006). To determine whether the variables had a significant influence on the distribution of macroinvertebrates, forward selections of environmental factors were used in the CANOCO model with 499 Monte Carlo permutations. The statistical significance of the eigenvalues (λ) and the taxon-environment correlations generated by the CCA were tested using Monte Carlo permutations with a p-value of <0.05.

RESULTS

Environmental variables

The values of measured environmental variables and their statistical significance across the three river reaches are presented in Tab. 2. The midstream reach had the deepest water depths (minimum and maximum), whereas water velocity was, as expected, highest in the upper reaches, progressively decreasing in a downstream direction. The same applied to conductivity as well as water temperature, which varied from an average of 16.4°C in the upper reach to 23.7°C downstream of the dam. Oxygen concentrations were relatively low (mean value <8 mg L^{-1}) throughout the sites, with average values decreasing from the upstream to midstream and downstream reaches. BOD was low (2.78±0.55 mg O_2 L^{-1}) in the upper reaches, but values in

Tab. 2. The minimum, maximum, mean ± SE of measured environmental variables, and mean rank and the Kruskal-Wallis test among the upstream, midstream, and downstream river reaches on the Awash River in Ethiopia.

Variable	Upstream			Midstream			Downstream			Mean rank			Kruskal-Wallis
	Min	Max	Mean ±SE	Min	Max	Mean ±SE	Min	Max	Mean ±SE	US	MS	DS	test
River depth (m)	0.33	0.37	0.35±0.01	0.39	0.42	0.40±0.00	0.29	0.32	0.31±0.01	13.00	8.00	3.00	**12.50***
Velocity (m s^{-1})	0.40	0.46	0.42±0.01	0.29	0.38	0.34±0.02	0.15	0.22	0.18±0.01	13.00	8.00	3.00	**12.50***
Substrate index	5.79	7.33	6.62±0.26	5.42	5.68	5.60±0.05	3.54	4.02	3.83±0.11	13.00	8.00	3.00	**12.52***
Temperature (°C)	15.00	17.30	16.40±0.43	17.64	21.62	18.88±0.72	21.60	26.00	23.70±0.73	3.00	8.10	12.90	**12.28***
EC (µS cm^{-1})	32.00	154.00	101.20±20.58	193.00	564.00	405.20±83.27	505.00	563.00	532.20±11.74	3.00	10.10	10.90	**9.47***
Turbidity (NTU)	19.00	78.00	46.60±10.73	97.00	428.00	278.00±73.41	291.00	356.00	318.20±13.60	3.00	11.00	10.00	**9.50***
DO (mg L^{-1})	7.00	8.10	7.58±0.19	6.22	7.10	6.64±0.15	4.28	6.02	5.27±0.35	12.80	8.20	3.00	**12.02***
BOD_5 (mg L^{-1})	1.30	4.30	2.78±0.55	5.20	21.00	13.44±3.20	10.00	16.30	13.94±1.31	3.00	10.70	10.30	**9.41***
pH	6.80	7.50	7.12±0.12	7.30	8.38	7.88±0.22	7.70	8.08	7.83±0.07	3.40	10.50	10.10	**7.98***
NO_3-N (mg L^{-1})	0.86	4.01	2.21±0.59	4.21	7.88	5.58±0.71	0.70	3.25	1.77±0.47	6.10	13.00	4.90	**9.57***
Phosphate (mg L^{-1})	0.03	0.29	0.11±0.05	0.63	1.76	1.10±0.18	0.05	0.01	0.03±0.01	7.40	13.00	3.60	**11.18***

*US, upstream river reach; MS, midstream river reach; DS, downstream river reach; NTU, nephelometric turbidity units; DO, dissolved oxygen; BOD, biochemical oxygen demand; *metrics that have significant differences.*

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the midstream and downstream reaches were 13.44±3.20 and 13.94±1.31 mg O_2 L^{-1}, respectively. The mean concentration values of phosphate and nitrate at midstream were higher than downstream of KHD, which might be due to their being absorbed in the reservoir (Tab. 2).

Macroinvertebrate assemblages

A total of 2,305 individuals belonging to 73 species or genera, 43 families, and 12 orders were collected (Tab. 3, Tab. S1). The upstream sites showed the highest total taxon richness (61 at Site US1), between 19 and 26 EPT taxa richness, and 56.98 to 71.10% EPT abundance. Sites in the midstream section had values between upstream and downstream river reaches. Conversely, the downstream river reach had the lowest total taxa richness (21-28), EPT taxa richness (7-8), and % EPT abundance (21.25-31.65%) (Tab. 3). Regarding the macroinvertebrate functional feeding groups (FFGs), %Collectors generally increased from the upstream sites to the downstream direction. On the other hand, other FFGs showed variable values at all river reaches. Among the 12 orders identified from the three river reaches, Trichoptera dominated at all sites of the upstream river reach, whereas Diptera dominated at the midstream and downstream sections (Tab. 3).

The result of the Kruskal-Wallis test showed that there was no significant difference in %Predators, %Shredders, and %Filterers between the river reaches ($p>0.05$). However, there were significant differences in other metrics such as %Collectors, %Scrapers, the Shannon, Simpson, Evenness indices, total and EPT taxa richness, and %EPT abundances ($p<0.05$) (Tab. 4).

The *post-hoc* analysis of all metrics also highlighted significant differences ($p<0.05$) between the upstream-midstream, upstream-downstream, and midstream-downstream

Tab. 3. Composition, diversity, functional measures, and relative abundances of orders of macroinvertebrate taxa (genera or species) at each sampling site in the upstream (US), midstream (MS), and downstream (DS) river reaches of Koka hydropower dam in the Awash River basin in Ethiopia.

Metric type	MetricsUpstream reach					Midstream reach					Downstream reach				
	US1	US2	US3	US4	US5	MS1	MS2	MS3	MS4	MS5	DS1	DS2	DS3	DS4	DS5
Composition measures															
Total taxa richness	61.00	53.00	46.00	46.00	48.00	43.00	40.00	36.00	34.00	33.00	25.00	21.00	23.00	27.00	28.00
EPT richness	26.00	22.00	19.00	20.00	21.00	15.00	15.00	12.00	13.00	12.00	8.00	7.00	7.00	7.00	8.00
%EPT abundances	66.12	65.36	71.10	61.26	56.98	51.39	44.36	46.49	40.19	32.32	21.25	31.65	27.38	22.77	31.50
Diversity measures															
Evenness	0.92	0.92	0.89	0.92	0.94	0.93	0.94	0.95	0.95	0.94	0.95	0.96	0.96	0.95	0.95
Shannon index	1.64	1.58	1.48	1.54	1.59	1.52	1.51	1.48	1.45	1.43	1.33	1.27	1.31	1.37	1.38
Simpson index	0.97	0.97	0.95	0.96	0.97	0.96	0.96	0.96	0.96	0.95	0.95	0.94	0.95	0.95	0.95
Functional measures															
%Collectors	26.02	23.21	19.72	33.51	31.28	25.00	30.08	28.95	33.64	39.39	45.00	44.30	41.67	38.61	34.65
%Predators	19.78	19.64	20.18	10.99	11.17	20.14	14.29	20.18	15.89	9.09	7.50	5.06	10.71	20.79	17.32
%Shredders	1.90	1.43	0.46	0.00	2.79	3.47	2.26	0.00	3.74	2.02	2.50	1.27	0.00	0.99	0.79
%Filterers	33.33	36.79	42.66	40.31	35.75	39.58	42.11	44.74	39.25	37.37	41.25	49.37	38.10	30.69	41.73
%Scrapers	18.97	18.93	16.97	15.18	18.99	11.81	11.28	6.14	7.48	12.12	3.75	0.00	9.52	8.91	5.51
Relative abundance of orders															
Ephemeroptera	19.78	21.07	23.85	24.61	21.23	20.83	14.29	17.54	15.89	13.13	3.75	10.13	7.14	6.93	10.24
Odonata	5.15	5.71	3.67	5.24	1.68	6.94	0.75	4.39	3.74	0.00	0.00	0.00	2.38	0.99	1.57
Diptera	8.94	11.79	10.09	13.61	15.64	23.61	30.08	32.46	38.32	49.49	53.75	50.63	41.67	40.59	38.58
Hemiptera	5.15	3.57	5.96	2.62	4.47	6.25	7.52	4.39	4.67	2.02	1.25	0.00	0.00	12.87	12.60
Lepidoptera	0.00	0.00	0.00	0.00	2.79	2.78	2.26	0.00	3.74	2.02	0.00	0.00	0.00	0.00	0.00
Veneroida	1.08	1.07	0.00	3.14	2.79	1.39	3.01	2.63	0.93	2.02	12.50	12.66	8.33	9.90	2.36
Coleoptera	10.57	8.93	6.42	12.04	8.38	4.86	6.02	7.02	1.87	3.03	5.00	1.27	2.38	1.98	1.57
Trichoptera	40.38	38.57	40.83	36.65	35.75	30.56	30.08	28.95	24.30	19.19	17.50	21.52	20.24	15.84	21.26
Annelida	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.88	0.93	1.01	0.00	0.00	2.38	1.98	0.79
Plecoptera	5.96	5.71	6.42	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Basommatophora	2.98	3.57	1.83	1.05	5.59	1.39	4.51	0.00	1.87	6.06	3.75	0.00	9.52	8.91	5.51
Unionoida	0.00	0.00	0.92	1.05	1.68	1.39	1.50	1.75	3.74	2.02	2.50	3.80	5.95	0.00	5.51

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river reach pairs except for the Simpson Index and %Collectors in the upstream-midstream and %Scrapers in the midstream-downstream river reach pairs (Tab. 5).

Cluster analysis based on macroinvertebrate assemblage data was effective in grouping sites according to their level of anthropogenic impacts. At 65% Bray-Curtis similarity index (Fig. 2), three groups of sites were clustered: i) upstream river reach sites relatively less impacted by human activities; ii) midstream river reach sites upstream of the reservoir, which is impacted by urbanisation and agricultural activities; and iii) the downstream river reach sites that received all impacts from the upstream catchment. The ANOSIM test demonstrated significant differences in macroinvertebrate assemblages between the three river reaches (Global R = 0.968, p=0.001). SIMPER analysis also showed significant differences between all pairs of

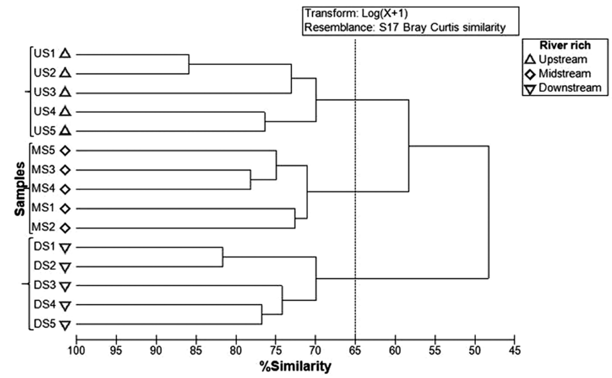

Fig. 2. A dendrogram (similarity cluster analysis) based on results of group average macroinvertebrate log(x+1) transformed data.

Tab. 4. The mean ± SE, mean ranks, and Kruskal-Wallis H tests for macroinvertebrate metrics in the upstream (US), midstream (MS), and downstream (DS) river reaches of Koka hydropower dam.

Metrics	Mean ± SE			Mean rank			Kruskal-Wallis H tests	
	US	MS	DS	US	MS	DS	K-W	p-value
	Composition measures							
% EPT	64.8±2.34	45.7±3.3	29.30±1.9	13.00	8.00	3.00	12.50	0.002*
EPT taxa richness	21.6±1.21	13.4±0.7	7.4±0.2	13.00	8.00	3.00	12.66	0.002*
Total taxa richness	50.8±2.85	37.2±1.9	24.8±1.3	13.00	8.00	3.00	12.52	0.002*
	Diversity measures							
Evenness index	0.92±0.01	0.9±0.004	0.96±0.002	3.40	7.60	13.00	11.58	0.003*
Simpson index	0.97±0.001	0.96±0.01	0.95±0.002	10.40	10.20	3.40	7.94	0.019*
Shannon index	1.7±0.03	1.5±0.02	1.3±0.02	12.60	8.40	3.00	11.58	0.003*
	Functional measures							
% Collectors	26.3±2.5	29.9±2.5	39.5±1.8	4.60	6.80	12.60	8.54	0.014*
% Predators	16.1±2.13	15.1±2.5	11.9±2.8	10.00	7.80	6.20	1.82	0.40
% Shredders	1.3±0.49	2.2±0.6	1.1±0.4	7.20	10.40	6.40	2.26	0.32
% Filterers	38.9±1.9	43±3.1	42.2±3.0	5.40	10.00	8.60	2.78	0.25
% Scrapers	17.49±0.8	9.3±1.2	5.4±1.7	13.00	7.20	3.80	10.82	0.004*

**Metrics that have significant differences.*

Tab. 5. The pairwise *post-hoc* test calculated for macroinvertebrate metrics having significant differences in the Kruskal-Wallis test.

Metrics	Upstream-midstream reach pairs				Upstream-downstream reach pairs				Midstream-downstream reach pairs			
	MR-US	MR-MS	M-W U	p-value	MR-US	MR-DS	M-W U	p-value	MR-MS	MR-DS	M-W U	p-value
%EPT	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*
EPT taxa richness	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.008*	8.00	3.00	0.00	0.008*
Total taxa richness	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*
Evenness index	3.40	7.60	2.00	0.028*	3.00	8.00	0.00	0.009*	3.00	8.00	0.00	0.009*
Simpson index	5.80	5.20	11.00	0.754	7.60	3.40	2.00	0.028*	8.00	3.00	0.00	0.009*
Shannon index	7.60	3.40	2.00	0.028*	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*
%Collectors	4.60	6.40	8.00	0.347	3.00	8.00	0.00	0.009*	3.40	7.60	2.00	0.028*
% Scrapers	8.00	3.00	0.00	0.009*	8.00	3.00	0.00	0.009*	7.20	3.80	4.00	0.076

*MR, mean rank, M-W U Mann-Whitney U test; *significant differences between river reach pairs with 2-sided test and a significance level of $p<0.05$.*

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river reaches, with the highest separation between the upstream-downstream river reach pairs (65.93%) with *Amphipsyche senegalensis* (7.11%) and *Cheumatopsyche sexfasciata* (5.43%) (Hydropsychidae), *Heptagenia* (4.85%) and *Leucrocuta* (4.48%) (Heptageniidae), and *Neoperla* (3.95%) (Perlidae) as the taxa with the highest contributions to the dissimilarity.

The mean dissimilarity between upstream and midstream groups was 51.33%. Here the taxa contributing to the dissimilarity were *Amphipsyche senegalensis* (6.44%) and *Cheumatopsyche sexfasciata* (6.23%) (Hydropsychidae), *Neoperla* (4.94%) (Perlidae), *Leucrocuta* (4.38%) and *Heptagenia* (3.73%) (Heptageniidae). Similarly, the mean dissimilarity between midstream and downstream groups was 48.35% with *Hydropsyche abyssinica* (4.92%) and *Amphipsyche senegalensis* (4.69%) (Hydropsychidae), and the genera *Orthocladius* (4.06%) (Chironomidae), *Heptagenia* (4.05%) (Heptageniidae), and *Hemerodromia* (3.87) (Empididae) as the taxa with the highest contributions to the dissimilarity. The NMDS ordination plot also demonstrated a clear separation of the three river reaches, each of which is represented by five sampling locations (Fig. 3).

Fig. 4 depicts the ordination triplot diagram for the environmental variables, macroinvertebrate assemblages, and

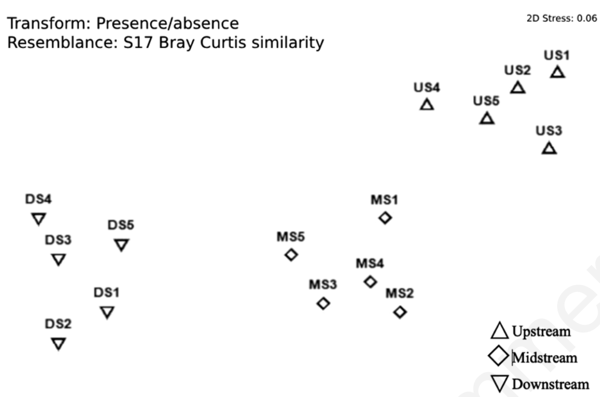

Fig. 3. The NMDS ordination plots of the macroinvertebrate assemblages at the three river reaches. The ordination plot was produced using PRIMER 6 software. Triangles, circles, and diamonds serve as symbols for the US, MS, and DS reaches, respectively.

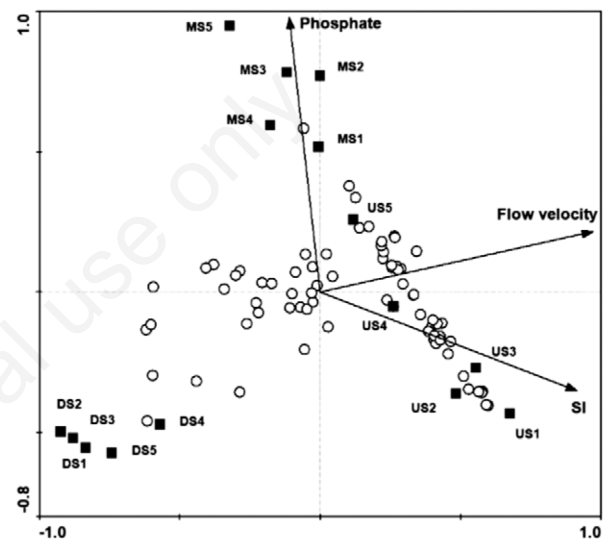

Fig. 4. The CCA ordination triplot illustrates the relationships between significant environmental variables at p<0.05 (arrows), macroinvertebrate taxa (circles), and sampling locations (squares) in the upstream, midstream, and downstream river reaches of the Koka hydropower dam. The figure was produced using the Canoco for Windows version 4.5. SI, substrate index.

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Tab. 6. The CCA results representing the independent and cumulative contribution of environmental factors in explaining the variation in the assemblages of macroinvertebrates in the three reaches of the Awash River.

Environmental factors	Eigenvalue (λ) for each factor)	% Explained by each factor	Cumulative Eigenvalue (λ)	Cumulative percentage	F-ratio	p-value
Velocity (m s^{-1})	0.28	40.58%	0.28	40.58	7.19	0.002*
Phosphate (mg L^{-1})	0.10	14.49%	0.38	55.07	3.05	0.002*
Substrate index	0.06	8.70%	0.44	63.77	1.8	0.016*
DO (mg O_2 L^{-1})	0.04	5.80%	0.48	69.57	1.47	0.106
River depth (m)	0.04	5.80%	0.52	75.37	1.39	0.136
NO_3-N (mg L^{-1})	0.04	5.80%	0.56	81.17	1.35	0.196
Electrical conductivity (µS cm^{-1})	0.03	4.35%	0.59	85.52	1.19	0.342
Temperature (°C)	0.03	4.35%	0.62	89.87	0.89	0.515
BOD_5 (mg L^{-1})	0.03	4.35%	0.65	94.22	0.89	0.514
pH	0.03	4.35%	0.68	98.57	1.17	0.318
Turbidity (NTU)	0.01	1.45%	0.69	100.00	0.45	0.778

*DO, dissolved oxygen; BOD, biochemical oxygen demand; NTU, nephelometric turbidity unit; *metrics that have significant differences.*

site locations within the three river reaches. Each CCA axis has an eigenvalue (λ) that represents the maximum dispersion of taxonomic scores along the axes. The cumulative percentage variance of the taxa-environment relationship explained by Axis-1 and Axis-2 was λ (1+ 2) = 0.38, with an eigenvalue of >0.30 indicating strong gradients. The CCA model showed that the variation in macroinvertebrate assemblages was significantly correlated with river velocity ($F = 7.19$, $p=0.002$), phosphate concentration ($F = 3.05$, $p=0.002$), and substrate index (SI) ($F = 1.8$, $p=0.016$) (Tab. 6). These three environmental variables explained 63.77% (λ= 0.44, CCA) of the total variation in macroinvertebrate assemblages. The contributions of the other variables ranged from 1.45% to 5.8% but were not significant.

DISCUSSION

The objective of our study was to assess the impacts of KHD on macroinvertebrate assemblage patterns based on their spatial locations. The ANOSIM test highlighted significant differences (Global $R = 0.968$, $p=0.001$) in macroinvertebrate assemblages between river reaches. This finding was consistent with previous studies conducted on dam impacts on the distribution of macroinvertebrates (Siziba, 2017; Ko *et al.*, 2020). We observed the lowest diversity of macroinvertebrate taxa in the downstream sites relative to those in the upstream and even midstream river reaches. More pollution-sensitive Trichoptera and Ephemeroptera taxa dominated upstream sites, whereas pollution-tolerant taxa in the Diptera order such as *Chironomus* (Chironomidae), *Aedes* (Culicidae), *Musca* (Muscidae), *Simulium and Prosimulium* (Simuliidae), *and Chrysogaster* (Syrphidae) were found in the downstream river reach. The results of earlier studies demonstrated similar changes in macroinvertebrate assemblages downstream of a dam in terms of changes in either the abundance or diversity of taxa and species richness (Sharma *et al.*, 2005; Ko *et al.*, 2020; Wang *et al.*, 2020).

All macroinvertebrate composition, diversity, and some functional measures (% collectors and %Scrapers) also showed significant differences between the three river reaches, consistent with an earlier study (Hauer and Resh, 2017). The results of the multivariate analysis also revealed considerable differences in the macroinvertebrate assemblages between the river reaches. The SIMPER analysis, for instance, indicated significant differences between all pairs of river reaches, with the upstream-downstream river reach pairs showing the greatest divergence and emphasising changes in macroinvertebrates. Several studies have highlighted the impacts of dams (Ko *et al.*, 2020; Bussi *et al.*, 2021; Mersha *et al.*, 2021). For example, Bussi *et al.* (2021) and Mersha *et al.* (2021) showed that variations in macroinvertebrate assemblages may be brought on by alterations in the physical characteristics of the river reaches as well as alterations in land use and water quality. The variations in these metrics might be related to several factors, such as the retention of sediments and nutrients in the reservoir, as demonstrated in the previous studies (Schmutz and Moog, 2018; Winton *et al.*, 2019; Cattaneo *et al.*, 2021).

Earlier studies showed that the Awash River basin is subjected to high climate variability and experiences intensive anthropogenic activities (Mersha *et al.*, 2021), such as water abstraction for various purposes and inputs of pollutants (Adeba *et al.*, 2015; Getachew *et al.*, 2020; Mersha *et al.*, 2021). For example, Mersha *et al.* (2021) estimated the abstraction of 1200 million cubic meters of water from the total annual surface water resource potential of 4600 million cubic meters in the Awash basin (Adeba *et al.*, 2015). Indeed, the cause of the variation might be associated with urbanization since the Awash basin is located in areas where several cities, including the capital Addis Ababa, are situated and many industries are concentrated (Englmaier *et al.*, 2020; Getachew *et al.*, 2020; Mersha *et al.*, 2021).

In our study, only three significant environmental factors -river flow velocity, phosphate concentration, and substrate index (SI)- significantly explained 63.8% of the variation in macroinvertebrate assemblages. Earlier studies also supported this finding (Winton *et al.*, 2019; Ko *et al.*, 2020). The river flow velocity accounted for the highest percentage (40.6%) of the variation in macroinvertebrate assemblages between the three river reaches ($F = 9.5$, $p=0.002$). In this regard, the development of the dam might decrease the downstream river flow as a result of impoundment, diversions for irrigation, or drought that limits the amount of suitable habitat for aquatic organisms. Petts and Maddock (1994) and Extence *et al.* (1999) highlighted that alterations in community composition may occur as a direct consequence of varying flow patterns or indirectly through associated habitat change. Ko *et al.* (2020) also showed that dams negatively alter the natural flow regime in terms of magnitude, frequency, and duration in the downstream river reach. Several other studies demonstrated that flow velocity may well contribute the most to macroinvertebrate assemblage change, as velocity affected nutrient transfer, the mobility of drifting species, and the morphology of benthic substrates (Matthaei *et al.*, 1997; Nelson and Lieberman, 2002; Brooks *et al.*, 2005; Wolmarans *et al.*, 2017). The sedentary behavior of the species may further limit their ability to disperse downstream of the dam (Cairns and Prall, 1993).

The average river velocities in the downstream, midstream, and upstream portions were 0.18, 0.33, and 0.45 m s^{-1}, respectively, highlighting that the downstream river velocity is unsuitable for most macroinvertebrate taxa. According to Xu *et al.* (2014), unsuitable flow velocities are either <0.3 or >0.8 m s^{-1}, whereas suitable flow rates fall between 0.3 and 0.8 m s^{-1}. On the other hand, Extence *et al.* (1999) allocated commonly identified freshwater species into one of the six groups based on recognized river flow associations as rapid (>1.0 m s^{-1}), moderate to fast (0.2-1.0 m s^{-1}), slow or sluggish flows (<0.2 m s^{-1}), and the rest three groups that are related to standing waters and drought-impacted sites. Jowett and Richardson (1990) also showed that the velocity preferences of macroinvertebrates may change with size or life stage. However, some taxa exist exclusively in very turbulent, high-velocity waters where they use sucker discs, hooks, or silk to remain attached to the substratum (Hauer and Lamberti, 2011; Hauer and Resh, 2017). For example, many Tipulidae, Trichoptera, and Plecoptera species that feed on coarse aquatic macrophytes (Shredders) (Merritt *et al.*, 2002) occur in riffles (Hauer and Lamberti, 2011). Li *et al.* (2009) determined that the optimum river velocity for various macroinvertebrate taxa, such as *Baetis,* is 0.4 m s^{-1}. Our study did not include flow sensitivity as LIFE index and we would like to recommend future studies include it in similar research on dam impacts.

The ranges of upstream and midstream river reach velocities were 0.4-0.46 m s^{-1} and 0.29-0.38 m s^{-1}, which are closer to the optimum value and have a better composition of macroinvertebrates, while the downstream river reach velocities ranging from 0.15 to 0.22 m s^{-1}. However, the more diverse macroinvertebrates, particularly in the upstream reach, maybe not only attributable to river velocity but also because of the presence of other conducive environmental characteristics such as lower temperatures, higher concentrations of DO, and more varied and coarse substrate compositions.

The other environmental parameter that explained 14.5% of the variation in macroinvertebrate distribution between the three river reach sections was differences in phosphate concentration. Earlier studies have reached similar conclusions (Ko *et al.*, 2020; Wang *et al.*, 2020). Particularly, large dams can significantly trap nutrients (Poff and Hart, 2002; Cattaneo *et al.*, 2021). For instance, Wang *et al.* (2020) reported that one sign of the effect of large dams on macroinvertebrate assemblage was a greater decline in nutrient levels at downstream sites, which has also been demonstrated in our study. The concentration of phosphate increased from the upstream to the midstream direction, probably because of differences in land uses such as agriculture and urbanisation, as also highlighted in various earlier studies (Englmaier *et al.*, 2020; Getachew *et al.*, 2020; Mersha *et al.*, 2021). However, the concentration of phosphate decreased in the downstream reach, which might partly be related to dam impacts, as supported by earlier studies. For example, Kunz *et al.* (2011) reported that more than 90% of the phosphorus was trapped in Kariba Dam on the Zambezi River.

The canonical correspondence analysis also showed that the changes in SI alone significantly explained 8.7% of the variation in the distribution of macroinvertebrates. According to Wang *et al.* (2020), damming disturbs the availability of suitable habitats for macroinvertebrates, and sediment trapping within reservoirs may have a significant impact. Wang *et al.* (2020) highlighted that the causes of macroinvertebrate richness reductions downstream of the dam were mainly attributed to changes in downstream substrate composition, *i.e.*, from coarse substrates (high SI) to fine substrates (low SI). Fine substrates can cause an accumulation on the fine and fleshy body parts of macroinvertebrates (such as gills and filter-feeding apparatus), making it difficult for aquatic residents to breathe and feed (Jones *et al.*, 2012).

Substrate preferences of macroinvertebrates vary from taxa to taxon. For example, stoneflies, cased caddisflies, and Diptera showed a preference for a substrate index of more than 6 (boulder/cobble) and the substrate index preference of beetles is about 5.6 (gravel/cobble) (Jowett and Richardson, 1990). The optimum substrate index for the majority of macroinvertebrates was just under cobble size (SI = 6) (Jowett and Richardson, 1990). The calculated mean SI for the upstream, midstream, and downstream sites were 6.63, 5.6, and 3.83, respectively, indicating that the downstream sites had substrates between sand (SI = 3) and gravel (SI = 4) which is far below the optimum SI. The absence of pollution-sensitive taxa such as *Heptagenia* and *Leucrocuta* (Heptageniidae), *Neoperla* (Perlidae*),* *Ecnomus similis* (Ecnomidae), and the dominance of pollution-tolerant taxa such as *Chironomus* (Chironomidae) in the downstream river reach might partly be associated with a reduction of SI.

The overall view is that the impacts of dams could be seen as eco-deficits (a condition related to insufficient water to meet the needs of the ecosystem) or eco-surplus (a condition related to water exceeding what is needed by the ecosystem) (Serfas, 2012). Several authors showed that dams alter the quantity, quality, and availability of water in relation to the natural environment (Gracey and Verones, 2016; Winton *et al.*, 2019). Our research has identified important environmental factors such as flow regulation, which might be related to changes in river flow velocity, changes in SI, and variations in nutrient loads to the downstream river reach that have caused changes in macroinvertebrate assemblage patterns between the reaches upstream and downstream of the KHD. However, the lower reaches of this river also carry the impact of pollution from further upstream, thus confounding efforts to attribute impacts directly to the presence of the dam.

CONCLUSIONS

Although dams often deliver economic services such as hydropower, flood risk alleviation, water supply, recreation, and more, there is evidence that dams have impaired the key functions of rivers in providing diverse habitats and maintaining ecosystem integrity. A combination of environmental and hydromorphological variables in this study explained why macroinvertebrate assemblages differed across the three river reaches above and below the KHD. Our study demonstrated that there is a significant difference between the three river reaches in terms of macroinvertebrate compositions, diversity, and some functional metrics. Changes in river flow regime due to changes in water velocity, variations in nutrient loads (phosphate concentration), and SI were highlighted as key variables determining the assemblage patterns of macroinvertebrates. Although other possible mechanisms for the observed differences may exist, the findings of this study are useful for the assessment of hydropower dam impacts on rivers in this region.

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REFERENCES

Adeba D, Kansal M, Sen S, 2015. Assessment of water scarcity and its impacts on sustainable development in Awash basin, Ethiopia. Sustain Water Resour Manage 1:71-87.

APHA, 2005. Standard Methods for the Examination of Water and Wastewater, 19th ed. American Public Health Association, Washington, DC.

Barbour MT, Gerritsen J, Snyder B, Stribling J, 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish. US Environmental Protection Agency, Office of Water, Washington, DC.

Brooks AJ, Haeusler T, Reinfelds I, Williams S, 2005. Hydraulic microhabitats and the distribution of macroinvertebrate assemblages in riffles. Freshwater Biol 50:331-344.

Bussi G, Whitehead PG, Jin L, Taye MT, Dyer E, Hirpa FA, et al., 2021. Impacts of climate change and population growth on river nutrient loads in a data scarce region: the Upper Awash River (Ethiopia). Sustainability 13:1254.

Cairns JR, Prall JR, 1993. A history of biological monitoring using benthic macroinvertebrates, p. 159-194.In: DM Rosenberg and VH Resh (eds.), Freshwater biomonitoring and benthic macroinvertebrates. Springer, New York.

Cattaneo F, Guillard J, Diouf S, O'Rourke J, Grimardias D, 2021. Mitigation of ecological impacts on fish of large reservoir sediment management through controlled flushing - The case of the Verbois dam (Rhone River, Switzerland). Sci Total Environ 756:144053.

Clarke KR, Gorley RN. 2006. PRIMER v6: User Manual/ Tutorial. PRIMER-E Ltd.

Clarke KR, Somerfield PJ, Gorley RN, 2008. Testing of null hypotheses in exploratory community analyses: similarity profiles and biota-environment linkage. J Expe Mar Biol Ecol 366:56-69.

Cummins K, 1974. Structure and function of stream ecosystems. Bioscience 24:631-641.

Degefu DM, He W, Zhao JH, 2015. Hydropower for sustainable water and energy development in Ethiopia. Sustain Water Resour Manage 1:305-314.

Degefu F, Lakew A, Tigabu Y, Teshome K, 2013. The water quality degradation of upper Awash River, Ethiopia. Ethiop J Environ Stud Manag 6:58-66.

Deinet S, Scott-Gatty K, Rotton H, Marconi V, McRae L, Baumgartner LJ, et al., 2020. The Living Planet Index (LPI) for migratory freshwater fish – Technical Report. World Fish Migration Foundation. Available from: https://worldfishmigrationfoundation.com/wp-content/uploads/2020/07/LPI_report_2020.pdf

Dudgeon D, 1999. Tropical Asian streams: zoobenthos, ecology and conservation. Hong Kong University Press, Hong Kong: 844 pp.

Englmaier GK, Hayes DS, Meulenbroek P, Terefe Y, Lakew A, Tesfaye G, et al., 2020. Longitudinal river zonation in the tropics: Examples of fish and caddisflies from the endorheic Awash River, Ethiopia. Hydrobiologia 847:4063-4090.

Extence CA, Balbi DM, Chadd RP, 1999. River flow indexing using British benthic macroinvertebrates: a framework for setting hydroecological objectives. Regul River 15:543-574.

Gernaat DEHJ, Bogaart PW, Vuuren DPv, Biemans H, Niessink R, 2017. High-resolution assessment of global technical and economic hydropower potential. Nat Energy 2:821-828.

Getachew M, Mulat WL, Mereta ST, Gebrie GS, Kelly-Quinn M, 2020. Challenges for water quality protection in the greater metropolitan area of Addis Ababa and the upper Awash basin, Ethiopia – time to take stock. Environ Rev 29:87-99.

Getachew M, Mulat WL, Mereta ST, Gebrie GS, Kelly-Quinn M, 2022. Refining benthic macroinvertebrate kick sampling protocol for wadeable rivers and streams in Ethiopia. Environ Monit Assess 194:196.

Gracey EO, Verones F, 2016. Impacts from hydropower production on biodiversity in an LCA framework - review and recommendations. Int J Life Cycle Assess 21:412-428.

Hagos YG, Andualem TG, Mengie MA, Ayele WT, Malede DA, 2022. Suitable dam site identification using GIS-based MCDA: a case study of Chemoga watershed, Ethiopia. Appl Water Sci 12:69.

Harrison J, 2009. Guides to the Freshwater Invertebrates of Southern Africa. Volume 10: Coleoptera Guides to the Freshwater Invertebrates of Southern Africa. Volume 10: Coleoptera. Water Research Commission Report: 263 pp.

Hauer FR, Lamberti GA, 2011. Methods in stream ecology. Academic Press, San Diego: 674 pp.

Hauer FR, Resh VH, 2017. Macroinvertebrates, p. 297-319. In: F.R. Hauer and G.A. Lamberti (eds.), Methods in stream ecology. Academic Press, San Diego.

Jones J, Murphy J, Collins A, Sear D, Naden PS, Armitage P, 2012. The impact of fine sediment on macro-invertebrates. River Res Appl 28:1055-1071.

Non-commercial use only

Jowett IG, Richardson J, 1990. Microhabitat preferences of benthic invertebrates in a New Zealand river and the development of in-stream flow-habitat models for Deleatidium spp. New Zeal J Mar Fresh 24:19-30.

Ko NT, Suter P, Conallin J, Rutten M, Bogaard T, 2020. Aquatic macroinvertebrate community changes downstream of the hydropower generating dams in Myanmar-potential negative impacts from increased power generation. Front Water 2:573543.

Kunz MJ, Wüest A, Wehrli B, Landert J, Senn DB, 2011. Impact of a large tropical reservoir on riverine transport of sediment, carbon, and nutrients to downstream wetlands. Water Resour Res 47:W12531.

Lenat D, Penrose, 1996. History of the EPT taxa richness metric. Bull N Am Benthol Soc 13:305-306.

Li F, Cai Q, Fu X, Liu J, 2009. Construction of habitat suitability models (HSMs) for benthic macroinvertebrate and their applications to instream environmental flows: A case study in Xiangxi river of Three Gorges Reservoir Region, China. Prog Nat Sci 19:359-367.

Lowe S, 2009. Guides to the Freshwater Invertebrates of Southern Africa. Volume 8: Insecta II. Hemiptera, Megaloptera, Neuroptera, Trichoptera and Lepidoptera. Afr J Aquat Sci 34:203-204.

Matthaei CD, Werthmüller D, Frutiger A, 1997. Invertebrate recovery from a bed-moving spate: the role of drift versus movements inside or over the substratum. Arch Hydrobiol 140:221-235.

Mbaka JG, Wanjiru MM, 2015. A global review of the downstream effects of small impoundments on stream habitat conditions and macroinvertebrates. Environ Rev 23:257-262.

McRae L, Deinet S, Freeman R, 2017. The diversity-weighted Living Planet Index: Controlling for taxonomic bias in a global biodiversity indicator. PLoS One 12: 0169156.

Merritt R, Cummins K, Berg M, Novak J, Higgins M, Wessell K, Lessard J, 2002. Development and application of a macroinvertebrate functional-group approach in the bioassessment of remnant river oxbows in southwest Florida. J N Am Benthol Soc 21:290-310.

Merritt R, Cummins K, Hauer F. 2006. Trophic relationships of macroinvertebrates, p. 585-609. Methods in stream ecology. Academic Press, San Diego.

Mersha AN, de Fraiture C, Masih I, Alamirew T, 2021. Dilemmas of integrated water resources management implementation in the Awash River Basin, Ethiopia: irrigation development versus environmental flows. Water Environ J 35:402-416.

Montana Department of Environmental Quality, 2012. Sample collection, sorting, taxonomic identification, and analysis of benthic macroinvertebrate communities standard operating procedure. Accessed: 6 June 2021. Available from: https://deq.mt.gov/files/Water/WQPB/QAProgram/Documents/WQPBWQM-009_rev3_Final.pdf

Nathan B, Hannah C, Dubniczki H, Natasha G, Howard A, Castilleja O, et al., 2017. Ecology for all. American Institute of Biological Sciences, Oxford University Press.

Nelson SM, Lieberman DM, 2002. The influence of flow and other environmental factors on benthic invertebrates in the Sacramento River, USA. Hydrobiologia 489:117-129.

Nilsson C, Reidy CA, Dynesius M, Revenga C, 2005. Fragmentation and flow regulation of the world's large river systems. Science 308:405-408.

Petts GE, Maddock I, 1994. Flow allocation for in-river needs, p. 289-307. In: P Calow and GE Petts, (eds.), The River's handbook: hydrological and ecological principles. Blackwell Scientific Publications, London.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, et al., 1997. The natural flow regime. BioScience 47:769-784.

Poff NL, Hart DD, 2002. How dams vary and why it matters for the emerging science of dam removal: an ecological classification of dams is needed to characterize how the tremendous variation in the size, operational mode, age, and number of dams in a river basin influences the potential for restoring regulated rivers via dam removal. BioScience 52:659-668.

Reis J, Culver TB, McCartney M, Lautze J, Kibret S, 2011. Water resources implications of integrating malaria control into the operation of an Ethiopian dam. Water Resour Res 47: W09530.

Schael D, 2010. Guides to the Freshwater Invertebrates of Southern Africa. Volume 9: Diptera. Afr J Aquat Sci 34: 303-304.

Schmutz S, Moog O, 2018. Dams: ecological impacts and management, p. 111-127. In: S Schmutz and J Sendzimir (eds.), Riverine ecosystem management: science for governing towards a sustainable future. Springer, Cham.

Serfas DH, 2012. Assessing the impacts of dams on nutrient and sediment loading in the Kalamazoo river using the soil and water assessment tool (SWAT). Master Thesis, Western Michigan University.

Sharma CM, Sharma S, Borgstrom R, Bryceson I, 2005. Impacts of a small dam on macroinvertebrates: a case study in the Tinau river, Nepal. Aquat Ecosyst Health Manage 8:267-275.

Siziba N, 2017. Effects of damming on the ecological condition of urban wastewater polluted rivers. Ecological Engineering 102:234-239.

Sumudumali RGI, Jayawardana JMCK, 2021. A review of biological monitoring of aquatic ecosystems approaches: with special reference to macroinvertebrates and pesticide pollution. Environ Manage 67:263-276.

Ter Braak CJ, Wiertz J, 1994. On the statistical analysis of vegetation change: a wetland affected by water extraction and soil acidification. J Veg Sci 5:361-372.

United States Environmental Protection Agency, 2017. Volunteer Stream Monitoring: A Methods Manual. Accessed: 17 May 2023. Available from: http://www.water-research.net/Waterlibrary/Lake/STREAM.PDF

Wang J, Ding C, Heino J, Jiang X, Tao J, Ding L, et al., 2020. What explains the variation in dam impacts on riverine macroinvertebrates? A global quantitative synthesis. Environ Res Lett 15:124028.

Wang Q, Chen Y, 2010. Status and outlook of China's free-carbon electricity. Renew Sustain Energ Rev 14:1014-1025.

World Commission on Dams, 2000. Dams and development: A new framework for decision-making: The report of the World Commission on dams. Earthscan, Oxford: 404 pp.

Winton RS, Calamita E, Wehrli B, 2019. Reviews and syntheses: Dams, water quality and tropical reservoir stratification. Biogeosciences 16:1657-1671.

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Wolmarans CT, Kemp M, de Kock KN, Wepener V, 2017. The possible association between selected sediment characteristics and the occurrence of benthic macroinvertebrates in a minimally affected river in South Africa. Chem Ecol33:18-33.

Xu M, Wang Z, Pan B, Yu G, 2014. The assemblage characteristics of benthic macroinvertebrates in the Yalutsangpo River, the highest major river in the world. Front Earth Sci 8:351-361.

Zhang AT, Gu VX, 2023. Global Dam Tracker: A database of more than 35,000 dams with location, catchment, and attribute information. Sci Data 10:111.

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Online supplementary material:

Tab. S1. The genus and species of macroinvertebrate presence/absence data collected from the Awash River above and below the Koka hydropower dam and categorised by their order and family.