Impacts of Koka Hydropower Dam on Benthic Macroinvertebrate Assemblages in the Awash River, Ethiopia

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Abstract

Macroinvertebrate distribution and community structure in lotic water are influenced by a range of natural environmental variables and other anthropogenic factors, particularly related to the construction of dams. Hydropower dams can alter natural flow regimes, sediment transportation, hydrochemical water quality, channel morphology, water temperature, and nutrient cycles and physically obstruct the dispersal and the migration of macroinvertebrates and fish communities. The objective of this study was to evaluate the impacts of the Koka Hydropower Dam on benthic macroinvertebrate assemblage structures.

Environmental, hydromorphological, and macroinvertebrate community data were collected along the course of the Awash River in the upper Awash basin. A total of 2305 macroinvertebrates assigned to 11 orders and 41 families were collected from 15 sites in the upstream, midstream, and downstream reaches. Ephemeroptera (32.2%), Diptera (24.6%), and Trichoptera (20.2%) represented 77% of the total abundances.

The canonical correspondence analyses, multivariate test, showed that river velocity and phosphate concentration explained the majority of the variation (60%) in macroinvertebrate community structure among the three reaches possibly due to the dam's impact in addition to other forms of impairment differences between reaches. The water quality categories of the river reach based on the Shannon diversity index at upstream, midstream, and downstream locations were high, moderate, and low, respectively. Based on the macroinvertebrate family-level information, this study found that the water quality degradation was most severe in the downstream reach. Overall, detection of the specific impacts of the dam is complicated by nutrient pollution in the river.

Introduction

A dam is a barrier constructed across the river to impound water for different purposes such as irrigation, flood control, hydropower, water supply, aquaculture, and navigation (Bhattarai et al., 2016). Hydropower dams provide substantial social and economic benefits to human society like electricity, flood risk alleviation, irrigation, and recreation (Orr et al., 2008; Reid et al., 2019), and they are a symbol of modernization and the ability to exploit natural resources (Teodoru & Wehrli, 2005). As per World Commission on Dams (WCD), on average, 160–320 new large dams have been constructed annually worldwide (Wcd, 2000). In the era of policy concern about climate change, hydropower generation from large dams continues to be an attractive investment choice in developing countries (Schulz & Adams, 2019). Schulz and Adams (2019), new hydropower dams are generally not planned using the WCD approach(Schulz & Adams, 2019). Nowadays, the increasing hydrological changes in river flow regime and resultant detrimental impacts on the riverine ecosystem health are recognized worldwide (Tharme, 2003; Kuriqi et al., 2020; Suwal et al., 2020).

From the basic ecological viewpoint, extreme flow fluctuations exert selective pressure on populations that dictate the relative success of different biological elements and regulate ecosystem process rates

(Resh et al., 1988; Castro et al., 2013). The modification of natural flow regimes alters river geomorphological and ecosystem processes, affecting diverse abiotic and biotic components (Poff & Zimmerman, 2010; Darwall et al., 2018; Krajenbrink et al., 2019). For example, hydropower dams alter natural flow regimes (Bunn & Arthington, 2002; Orr et al., 2008; Poff & Zimmerman, 2010; Sabater et al., 2018; Belmar et al., 2019; Ko et al., 2020), sediment transportation (Garcia De Jalon et al., 2015; Gabbud & Lane, 2016), hydrochemical water quality (Bhadja & Vaghela, 2013; Sabater et al., 2018; Reid et al., 2019; Winton et al., 2019), channel morphology (Sabater et al., 2018), water temperature (Dewson et al., 2007), nutrient cycles (Sabater et al., 2018), and physically obstruct the dispersal and the migration of macroinvertebrates and fish communities (Liermann et al., 2012; Hastings, 2014). These impacts may even propagate for hundreds of kilometers downstream of the dam and be ambiguous sources of environmental degradation (Winton et al., 2019) that can cause incessant disruptions in watercourses which finally result in changes in ecosystem integrity (Teodoru & Wehrli, 2005; Orr et al., 2008; Ko et al., 2020).

Macroinvertebrates, sensitive to alterations in the river environment and ecological conditions (Hastings, 2014; Su et al., 2019), are functionally and taxonomically diverse groups representing a large proportion of global freshwater biodiversity (Hauer & Resh, 2017). They represent local environmental changes better than other higher-level consumers such as fish (Ormerod & Tyler, 1993; Morse et al., 2007) and reflect several anthropogenic disturbances such as water flow variation, eutrophication, water pollution, climate change, and biological invasions (Fornaroli et al., 2018; Guareschi & Wood, 2019).

The life of a river is closely tied to its flow, which constantly fluctuates. Damming a river and altering its flow pattern generates several physical and biological impacts. The disruption of a river's flow obstructs its natural flow and affects the habitat (Liermann et al., 2012; Wiley, 2014). For example, Hastings (2014) showed that the reduction in the flow of rivers reduces the natural transportation of larger boulders, cobbles, and debris to downstream habitats (Hastings, 2014). However, the dam's impact on the river ecology depends on its size. According to the Wcd (2000) report, a dam size with a height of > 15m is defined as a large dam. On the other hand, Wiley (2014) classified dams into three types similarly based on the height measured from the base to its crest: a small dam (< 30m); a medium-height dam (30 and 100m), and a high dam (> 100m). Several authors reported that small dams have a relatively insignificant impact on the river biota (Sharma et al., 2005; Ambers, 2007). For example, Principe (2010) found minor changes in macroinvertebrate richness, diversity, and the density of filterer-collectors. On the other hand, large dams cause changes in the taxonomic composition of macroinvertebrate communities due to the alteration of fluvial habitats (Lessard & Hayes, 2003). Moreover, evidence showed that the hydraulic conditions, besides the physicochemical variables, were significant factors affecting benthic macroinvertebrate community structures (Tomsic et al., 2007).

In many countries, several studies have assessed the ecological impacts of dams using macroinvertebrates as indicator organisms (Mor et al., 2018; Krajenbrink et al., 2019; Wang et al., 2019). Despite the rapidly growing hydropower dam developments and constructions in Ethiopia, there are limited studies that evaluate their impacts on aquatic macroinvertebrates in the country. The objective of

this study was to assess the influence of the Koka hydropower dam (KHD) on macroinvertebrate community structure and composition (ecological implications) on the Awash River in the central part of Ethiopia.

Materials And Methods

Study area and sampling sites

Awash River Basin represents one of the most utilized landscape catchments in Ethiopia (Berhe et al., 2013; Degefu et al., 2013; Asfaw et al., 2014). It has an area of 112,696 km^2, is home to nearly 15 million people, and consequently is one of the most important basins in the country (Tesfaye & Wolff, 2014; Englmaier et al., 2020). This river emerges in the central highlands in Ethiopia at an altitude of 3000 m.a.s.l (Berhe et al., 2013; Likasa, 2013; Bussi et al., 2021) and is subjected to severe pollution primarily from its tributary rivers (Berhe et al., 2013; Asfaw et al., 2014; Getachew et al., 2020). The river flows 1250 km (Englmaier et al., 2020) through KHD to the northeast along the Great Rift Valley where it finally drains into Lake Abe (Likasa, 2013) at the border of Ethiopia and Djibouti (Berhe et al., 2013; Tesfaye & Wolff, 2014). The upper Awash River basin (UARB), located between 8° and 9^oN and $38^o$and 39^oE (Reys, 2016) (Fig. 1), covers an area of 11,600 km^2 including Addis Ababa city, the capital of Ethiopia (Bussi et al., 2021). The KHD, constructed in 1960 with an installed capacity of 42 MW (Degefu et al., 2015) and a dam height of 42 meters (Reis et al., 2011), is located in the UARB (Fig. 1).

Figure 1

There are more than 1000 meters of elevation drop from the source of Awash River at US1 (2386 m.a.s.l) to the downstream of KHD at DS5 (1345 m.a.s.l). Several tributary rivers and streams join the Awash River at different sections from the left and right sides (Fig. 1). Before sampling, 15 sites were selected with an average distance of 12 kilometers between two successive sampling points. Three study reaches were defined, upstream (US), midstream (MS), and downstream (DS) of KHD. Five sites in the US reach, selected near the source of the Awash River, are minimally impaired. However, five sites selected in the MS reach above the KHD, and five sites selected in the DS reach were highly degraded from anthropogenic activities (Degefu et al., 2013; Asfaw et al., 2014; Getachew et al., 2021a).

Aquatic habitat selection

Both macroinvertebrates and physicochemical samples were collected from riffle/run habitats. Habitats were defined using velocity to depth ratio validation and visual identification after Queensland_Government (2019). The habitat was categorized as riffle when the velocity to depth ratio is > 0.032/s, as a run for velocities between 0.0124/s and 0.032/s.

Macroinvertebrate sampling and identification

Two-minute kick samples (Getachew et al., 2021b) were collected with a D-frame dip net with a mesh size of 0.5mm (Barbour et al., 1999; Hauer & Resh, 2017) from riffle/run habitats. Immediately after the collection of samples, large leaves, sticks, and stones were individually rinsed and inspected for organisms and discarded. Samples were then preserved in 96% ethanol alcohol after Barbour et al. (1999) before being sorted and identified to the family level. By considering similar climatic conditions relative to other non-tropical regions, the South African benthic macroinvertebrate keys (Harrison, 2009; Schael, 2009) were used for taxa identification at the family level.

Physicochemical and benthic substrates data collection

The river width (m), depth (m), velocity (m/s), and temperature (^{o}C) were measured at each site. To determine the river depth at each site, an average of three measurements was taken. The surface river velocity was computed by dividing the 10m distance, measured at each site for sampling, by the average time the floating object took to travel that distance and multiplying the quotient by the correction factor (0.85) to estimate the average water velocity of the stream (Ngoma & Wang, 2018). The GPS reading such as altitude (m.a.s.l) and the location of sites were recorded in the field using a global positioning system (GERMIN-72H). Substrate types such as bedrock, boulder, cobble, gravel, fine gravel, and sand and fines were visually characterized by changes in habitat features at each sampling site. These substrate types have been converted to a single substrate index (SI) values after Jowett and Richardson (1990) using the formula: SI = 0.08* bedrock + 0.07* boulder + 0.06* cobble + 0.05 *gravel + 0.04*fine gravel + 0.03* sand and fines.

The DO, pH, temperature, and EC were measured in-situ using a digital handheld portable multi-parameter (HACH HQ40D probe). The Measurements were done starting from downstream to the upstream to minimize disturbance of the sites before sampling. Similarly, turbidity was measured using a handheld turbidity meter. Nitrate-nitrogen and phosphate were measured using DR3900, HACH spectrophotometer in the laboratory. The BOD_5 was also measured using the Winkler method (Carpenter, 2006).

Metric selection

Taxon richness (Mwedzi et al., 2020), Simpson index (Simpson, 1949), Shannon index, evenness, Margalef's index (Washington, 1984; Clarke et al., 2001), the Biological Monitoring Working Party (BMWP) score (Armitage et al., 1983), and ETHbios score developed for Ethiopia (Lakew & Moog, 2015), %EPT, and EPT richness calculated from orders of Ephemeroptera, Plecoptera, and Trichoptera were some of the selected metrics. Also, the functional feeding groups (FFG) such as %Scrapers, %Shredders, %Collectors, %Filterers, %Predators (Wang et al., 2013), were considered to evaluate differences in the macroinvertebrate communities between the three study reaches in the UARB.

Data analysis

We used the relative abundances of the macroinvertebrate taxa to carry out the multivariate tests to control for the sampling times and habitat differences during sampling. Several multivariate analyses

techniques were used to assess variations in macroinvertebrate community structure and composition between the three study reaches.

Analysis of Similarity (ANOSIM)

The ANOSIM (Clarke & Green, 1988), which allows a test of the null hypothesis that there are no macroinvertebrate community structure differences among groups of samples collected from three reaches (Clarke & Gorley, 2006), was carried out in PRIMER 6, using log(x + 1) transformed relative abundance data with 999 permutations. The differences among the reaches were measured by the global R statistic that was calculated as $R = \frac{4(\bar{r}_B - \bar{r}_W)}{n(n-1)}$, where $\bar{r}_B$ and $\bar{r}_W$ are the mean rank between-group similarity and within-group similarity, respectively, and *n* is the total number of samples (Cao et al., 2005). ANOSIM calculates a test statistic R that ranges from − 1 to 1 (Chapman & Underwood, 1999). When the R is closer to 1, there is a good separation among the groups and a value closer to 0 shows weak separation (Chapman & Underwood, 1999; Clarke & Gorley, 2006) and the negative values indicate that the samples within a group are less similar to one another than to the samples of other groups, probably due to inappropriate sampling designs (Chapman & Underwood, 1999). An associated P-values to R-statistics in the analysis of ANOSIM highlights the significance level of the test (Clarke & Gorley, 2006).

Non-metric multidimensional scaling (MDS)

The MDS plot in PRIMER 6 (Clarke & Gorley, 2006) was also used based on the relative abundance of the macroinvertebrate taxa to assess the (dis)similarity of macroinvertebrate communities in the three study reaches. Also, the data were log(x + 1) transformed to improve the normality of the data before ordination. The taxa resemblance matrix was then analyzed using Bray-Curtis similarity with 999 permutations as a function of sampling sites. As a rule of thumb, an MDS ordination with a stress value equal to or below 0.1 is considered fair while values equal to or below 0.05 indicate a good fit (Clarke, 1993). Also, we used the SIMPER routine in PRIMER (Clarke & Gorley, 2006) on the relative abundance data to determine the percentage (dis)similarity in macroinvertebrate communities between the three reaches.

Shannon diversity index

The Shannon diversity index was also analyzed using the natural logarithm in PRIMER to compare the diversity of macroinvertebrates in each reach. The water quality of the river in the upstream, midstream, and downstream reaches of KHD is determined from information provided in Fernando et al. (1998) and Baliton et al. (2020) as very high, high, moderate, poor, and very poor with a corresponding Shannon diversity index of ≥ 3.5, 3.00-3.49, 2.50–2.99, 2.00-2.49, and ≤ 1.99, respectively.

Kruskal-Wallis test

The Kruskal-Wallis test was carried out to assess differences in continuous dependent variables by a categorical independent variable with assumptions that the samples drawn from the population are

random and the observations are independent of each other. This test is used when the assumptions of one-way ANOVA are not met. The Kruskal-Wallis test was used to examine for significant differences in a range of metrics among the three study reaches. The metrics analyzed using the Kruskal-Wallis test include taxa richness, total abundance, the BMWP (Armitage et al., 1983), ETHbios (Aschalew & Moog, 2015), Shannon-Wiener index, evenness, Margalef's index, Simpson index, %Scrappers, %Shredders, %Collectors, %Filterer and %Predators, %EPT, and EPT richness. A post *hoc* test, for the statistically significant metrics results in a Kruskal-Wallis test, was then run to determine where the differences lay using a Bonferroni-corrected p-value to reduce the instance of false positives. The Bonferroni corrected P-values were analyzed in SPSS version 20 statistical software.

Canonical Correspondence Analysis (CCA)

Detrended Correspondence Analysis (DCA) was applied using CANOCO 4.5 (Ter Braak & Smilauer, 2002) to examine whether Redundancy Analysis (RDA) or Canonical Correspondence Analysis (CCA) would be appropriate (Ter Braak & Wiertz, 1994) to analyze the data. The DCA yielded gradient lengths that were higher than three standard deviations, therefore CCA was used for the data analysis. Macroinvertebrate abundance data were log-transformed, log(x + 1), before analysis to obtain homogeneity of variance. Environmental variables, except for pH, were square-root transformed in PRIMER statistical software (Clarke & Gorley, 2006) and standardized since the variables were measured in different units and transferred to CANOCO 4.5 model for analysis. In the CCA model, forward selection of environmental variables with Monte Carlo permutations of 499 was used to determine whether the variables exerted a significant P-value of < 0.05 impact on macroinvertebrate distributions. The statistical significance of eigenvalues and species-environment correlations generated by the CCA were tested using Monte Carlo permutations.

BIOENV routine

The BIOENV routine in PRIMER, the typical setup in the exploration of environmental variables that best correlate to sample similarities of the biological community (Clarke & Gorley, 2006), was also used in this study. In this case, the similarity matrix of the community is fixed, while subsets of the environmental variables are used in the calculation of the environmental similarity matrix. A Spearman correlation coefficient was then calculated between the two matrices and the best subset of environmental variables was identified with a 999 permutation test. The similarity matrix of environmental data was based on normalized "Euclidean" distances to remove the effect of different units between parameters and the biological variable matrix was based on a Bray-Curtis similarity measure on the log (x + 1) transformed data (Dixon, 2003; Clarke & Gorley, 2006).

Results

Multivariate Analyses

A total of 2305 macroinvertebrates belonging to 41 families and 11 orders were identified from the three reaches, each with five sites. Ephemeroptera (32.2%), Diptera (24.6%), and Trichoptera (20.2%) were the three dominant orders representing 77% of the total collected. The one-way ANOSIM test based on the relative abundance of the macroinvertebrate taxa showed significant differences among the three reaches (R = 0.915, P = 0.001, ANOSIM). Also, the pairwise ANOSIM test indicated a complete separation of the three reaches. The R-statistic for midstream/upstream, upstream/downstream, and downstream/upstream pairs were 0.784, 1.00, and 0.928, respectively, with P-values of 0.008 for all pairs.

The MDS ordination plot using macroinvertebrate relative abundances also showed that the three river reaches were well separated from one another (Fig. 2). The low MDS tress level (0.04) showed no overlap in multidimensional space by the spread of the site within reaches except for the MS1 in the midstream reach (Fig. 2) that was closer to the upstream sites than the other sites in that reach.

Figure 2

The Shannon index values for the upstream, midstream and downstream reaches from KHD were 3.02, 2.72, and 2.32, respectively. The Kruskal-Wallis tests on this index and biotic and diversity indices as well as functional measures showed significant differences among the three site reaches except for the evenness index and %collectors of macroinvertebrate functional feeding groups (Table 1).

Table 1
The Mean ± SE, mean ranks, and the Kruskal-Wallis H tests for macroinvertebrate metrics in the upstream (US), midstream (MS), and downstream (DS) reaches from KHD in 2021. Metrics with significant differences in the Kruskal-Wallis H test are presented in bold

Metrics	Mean ± SE			Mean rank			K-W H test	
	US	MS	DS	US	MS	DS	K-W	P-value
Taxa (S)	34 ± 0.84	24 ± 1.41	14.8 ± 0.97	13.00	8.00	3.00	12.55	0.002
Abundance (N)	244.2 ± 35.17	124.8 ± 6.64	92 ± 11.14	13.00	7.30	3.70	11.02	0.004
Margalef's index	6.04 ± 0.06	4.77 ± 0.27	3.06 ± 0.14	13.00	8.00	3.00	12.5	0.002
Evenness	0.86 ± 0.01	0.86 ± 0.004	0.86 ± 0.02	7.40	7.90	8.70	0.22	0.895
Shannon index	3.02 ± 0.02	2.72 ± 0.05	2.32 ± 0.08	13.00	7.80	3.20	12.06	0.002
Simpson index	0.93 ± 0.004	0.90 ± 0.007	0.87 ± 0.01	3.00	8.60	12.4	10.27	0.006
%Predators	26.95 ± 1.31	23.29 ± 1.59	11.87 ± 1.77	11.80	9.20	3.00	10.22	0.006
%Collectors	34.21 ± 0.54	29.94 ± 3.23	37.70 ± 1.67	7.40	5.40	11.2	4.34	0.114
%Shredders	5.52 ± 0.63	5.99 ± 1.03	0.87 ± 0.36	10.20	10.80	3.00	9.44	0.009
%Scrapers	19.55 ± 0.80	11.96 ± 1.44	3.80 ± 0.61	13.00	8.00	3.00	12.5	0.002
%Filterers	13.77 ± 0.72	28.81 ± 3.64	45.76 ± 1.92	3.00	8.20	12.8	12.02	0.002
BMWP score	197 ± 4.64	114.8 ± 11.91	61.2 ± 6.38	13.00	8.00	3.00	12.5	0.002
ETHbios score	149 ± 3.11	98.2 ± 9.33	47.8 ± 5.12	13.00	8.00	3.00	12.5	0.002
%EPT	59.32 ± 1.56	52.28 ± 3.14	34.44 ± 1.41	12.00	9.00	3.00	10.52	0.005
EPT richness	9.6 ± 0.24	5.8 ± 0.66	4.2 ± 0.37	13.00	7.20	3.80	11.1	0.004

Table 1

The post *hoc* tests also highlighted significant differences ($P < 0.05$) between the upstream and downstream reaches (Table 2). However, the midstream and downstream reaches were not significantly

different except for % Shredders (P = 0.033) (Table 2).

Table 2
The pairwise post hoc test calculated for macroinvertebrate metrics having significant differences in the Kruskal Wallis test, 2021

Metrics	Reaches	TS	SE	P-Value	Metrics	Reaches	TS	SE	P-Value
Taxon richness	MS-DS	5.0	2.823	0.230	% Scrapers	MS-DS	5.0	2.828	0.231
	US-MS	5.0	2.823	0.230		US-MS	5.0	2.828	0.231
	DS-US	10.0	2.823	0.001		DS-US	10.0	2.828	0.001
Abundance	MS-DS	3.6	2.826	0.608	% Filterers	MS-DS	-5.2	2.828	0.198
	US-MS	5.7	2.826	0.131		US-MS	-4.6	2.828	0.312
	DS-US	9.3	2.826	0.003		DS-US	-9.8	2.828	0.002
Margalef's index	MS-DS	5.0	2.828	0.231	BMWP	MS-DS	5.0	2.828	0.231
	US-MS	5.0	2.828	0.231		US-MS	5.0	2.828	0.231
	DS-US	10.0	2.828	0.001		DS-US	10.0	2.828	0.001
Shannon index	MS-DS	4.6	2.823	0.310	ETHbios	MS-DS	5.0	2.828	0.231
	US-MS	5.2	2.823	0.197		US-MS	5.0	2.828	0.231
	DS-US	9.8	2.823	0.002		DS-US	10.0	2.828	0.001
Simpson index	MS - DS	3.4	2.803	0.675	%EPT	MS-DS	6.0	2.826	0.101
	US-MS	5.5	2.803	0.149		US-MS	3.0	2.826	0.865
	DS-US	8.9	2.803	0.004		DS-US	9.0	2.826	0.004
% Predators	MS-DS	6.2	2.828	0.085	EPT richness	MS-DS	3.4	2.793	0.670
	US-MS	2.6	2.828	1.000		US-MS	5.80	2.793	0.113
	DS-US	8.8	2.828	0.006		DS-US	9.20	2.793	0.003
% Shredders	MS-DS	7.2	2.826	0.033					
	US-MS	-0.6	2.826	1.000					
	DS-US	7.8	2.826	0.017					

US = upstream, MS = midstream, DS = downstream, SE = standard error, TS = test statistic. Significant differences are presented in bold. Tests are 2 – sided. The significance level is 0.05. P-values are adjusted by the Bonferroni correction for multiple tests

Table 2

Physicochemical water quality parameters

River depth and width, water temperature, EC, turbidity, pH, and BOD_5 showed a general increase from upstream to downstream reaches. On the other hand, water velocity, substrate index, and DO showed a general decrease from the upstream to the downstream reaches. The results for nitrate and phosphate concentrations were more variable. The highest concentrations were found in the midstream reach sites (MS1-5) before the KHD (Table 3).

Table 3
The Mean (+/-SE), minimum and maximum values of measured physicochemical variables in the upstream, midstream, and downstream reaches from KHD on the gradients of Awash River from 15 April-17 May 2021. SE = standard error, Min = minimum, Max = maximum

Upstream (US1-5)				**Midstream (MS1-5)**			**Downstream (DS1-5)**		
Parameter	Mean± (SE)	Min	Max	Mean± (SE)	Min	Max	Mean± (SE)	Min	Max
River width (m)	17.9 ± 0.48	16.40	19.20	20.52 ± 0.60	19.40	22.50	21.66 ± 0.67	19.80	23.30
River depth (m)	0.43 ± 0.5	0.30	0.55	0.56 ± 0.03	0.45	0.63	0.73 ± 0.03	0.62	0.78
Water velocity (m/s)	0.35 ± 0.01	0.33	0.38	0.28 ± 0.02	0.24	0.32	0.18 ± 0.01	0.15	0.22
Substrate index	12.20 ± 0.73	10.00	14.00	9.92 ± 0.50	8.40	11.00	9.82 ± 0.07	9.60	10.00
Water temperature (^{O}C)	19.8 ± 0.5	18.3	21.4	23.6 ± 0.8	21.6	26.4	27.5 ± 0.3	26.5	28.0
EC (µS/cm)	101.2 ± 20.6	32.0	154.0	405.2 ± 83.3	193.0	564.0	532.2 ± 11.7	505.0	563.0
Turbidity (NTU)	46.6 ± 10.7	19.0	78.0	278 ± 73.4	97.0	428.0	318.2 ± 13.6	291.0	356.0
DO (mg/l O_2)	7.54 ± .22	6.80	8.10	5.56 ± 0.35	4.70	6.50	5.82 ± 0.22	5.20	6.20
BOD (mg/l O_2)	2.78 ± 0.55	1.30	4.30	13.44 ± 3.2	5.20	21.00	13.94 ± 1.31	10.00	16.30
pH (unit)	7.12 ± 0.12	6.80	7.50	7.56 ± 0.11	7.30	7.90	8.56 ± 0.13	8.10	8.90
NO_3-N (mg/l)	0.48 ± 0.22	0.10	1.30	8.68 ± 2.55	2.50	13.90	5.48 ± 0.83	3.90	8.41
Phosphate (mg/l P)	0.56 ± 0.16	0.04	0.92	4.32 ± 1.22	1.20	7.60	2.03 ± 0.23	1.30	2.60

Table 3

Relationships between environmental variables and macroinvertebrate assemblages

The ordination triplot diagram for the environmental variables, macroinvertebrates, and the site locations of the three reaches are presented in Fig. 3.

Figure 3

To identify environmental variables that are more closely associated with macroinvertebrate assemblage structures, the constrained ordination technique using CCA (Ter Braak & Verdonschot, 1995) was used. Each CCA axis is associated with an eigenvalue (λ) that indicates maximized dispersion of species scores along the axes (Ter Braak, 1987). The cumulative percentage variance of the species-environment relationship explained by Axis-1 and Axis-2 for this data set was $\lambda_{1+2} = 0.33$. As a rule of thumb, eigenvalue > 0.30 indicates strong gradients (Ter Braak, 1987). The CCA model showed that the variation in the distribution of macroinvertebrates was significantly correlated with river velocity (F = 9.83, P = 0.001) and phosphate concentration (F = 3.15, P = 0.001). These two variables explained 60% ($\lambda_{Velocity+Phosphate} = 0.3$, CCA) of the variation in macroinvertebrate community structure out of the total canonical variations (66%). The contribution to the variance by NO_3-N (6%), water temperature (6%), river depth (4%), BOD_5 (4%), pH (4%), EC (4%), DO (4%), river width (4%), substrate index (2%), and turbidity (2%) were not significant.

Further analysis of the BIOENV routine in PRIMER (Clarke & Gorley, 2006) showed that river velocity and depth, water temperature, EC, and phosphate had a combined influence on the structure and composition of macroinvertebrate communities with the Pearson correlation coefficient (ρ) of 0.842 (P-value = 0.001) at 999 permutations.

Discussion

Biological assessments aim to characterize the status of stream ecosystems by monitoring changes in the aquatic communities associated with anthropogenic disturbance (Jun et al., 2012). Highland areas, having sharp gradients, are characterized by variable climate, riparian vegetation, changes in water velocity (Hastings, 2014), and others that result in macroinvertebrate communities that are distinct from downstream reaches (Fureder et al., 2001). For example, Hammer and Linke (2003) and Tiemann et al. (2004) reported that the decrease in water flow velocity, typical of the reach downstream of the dam, causes changes in water quality shifting macroinvertebrate abundances by decreasing Ephemeroptera, Plecoptera, and Trichoptera; while increasing tolerant families such as Chironomidae. Similarly, in this study, the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT) were reduced in abundance in the downstream reaches compared to the upstream and midstream reaches. The statistical analysis test highlighted that only the upstream and the downstream reach pairs were significantly different. As a further example, the percent composition of Ephemeroptera was 65.3%, 24.7%, and 10.1% in the upstream, midstream, and downstream reaches, respectively.

On the other hand, the percent composition of Diptera order was higher (37.8%) in the downstream reach compared to the upstream (32.4%) and midstream (29.8%) reaches. Specifically, the percent composition of the Chironomidae family was 20.7%, 21.9%, and 57.4% at the upstream, midstream, and downstream reaches, respectively. The review by Wang et al. (2020) highlighted that macroinvertebrate taxa with different tolerances to natural and anthropogenically modified environmental conditions, e.g. sensitive taxa, such as EPT, and tolerant taxa, such as Oligochaeta) also responded distinctly to dams because of changes in the downstream environment. Dams influence macroinvertebrate communities primarily by

altering habitat structure (Hastings, 2014). For example, the reduction in river velocity reduces the natural transportation of large boulders, cobbles, and debris to downstream habitats (Hastings, 2014). Other reports also showed that river damming imposes fundamental physical changes that in turn lead to chemical and biological changes (such as the production of harmful algal blooms) within the reservoir, posing a major threat to local and global biodiversity (Reid et al., 2019; Winton et al., 2019). The impacts may even propagate for hundreds of kilometers downstream of the dam and be a cryptic source of environmental degradation (Winton et al., 2019). As a result, the structure and composition of macroinvertebrate communities vary between reaches upstream and downstream of a dam, as demonstrated in the present study.

In agreement with other similar studies (Quevedo et al., 2018; Ko et al., 2020), both one-way ANOSIM and pairwise ANOSIM tests, in this study, showed significant differences among the three river reaches as also illustrated in the MDS ordination plot. Furthermore, the SIMPER analysis results showed 36.28%, 58.31%, and 41.08% dissimilarity between the upstream and midstream, upstream and downstream, and midstream and downstream reach pairs, respectively. The water quality categories of the reaches, based on the natural logarithm of the Shannon diversity index (Fernando et al., 1998; Baliton et al., 2020), varied. In the downstream reach, the water category was low, which highlights the impact was severe compared to the upstream and midstream reaches having high and moderate water quality, respectively, even in the presence of several anthropogenic influences in the selected sites (Mohammed; Taddese et al., 2009; Degefu et al., 2013; Asfaw et al., 2014; Englmaier et al., 2020; Getachew et al., 2020; Mersha et al., 2021). The mean phosphate concentration in the midstream reach was elevated, albeit it was lower than further downstream.

The impacts of the dam on several macroinvertebrate metrics were evaluated using the Kruskal-Wallis tests. These showed significant differences among the upstream, midstream, and downstream reaches except for the evenness index and % collectors. However, the post *hoc* test showed significant differences only between the upstream and downstream reaches. The midstream and downstream reaches did not show any difference in macroinvertebrate community structure, except for %shredders. The retention of fine particulate materials by the hydropower dam may explain their reduced abundance below the dam (Lamberti & Gregory, 2007).

In this study, the constrained ordination technique was used to identify environmental variables closely associated with macroinvertebrate community structures. River velocity explained high percentages of the variability (46%) ($F = 9.83$. $p = 0.002$) contrary to another finding (Wang et al., 2020). As per Popoola and Otalekor (2011), the river velocity is, directly and indirectly, important as it influences the riverbed and the amount of silt deposition which in turn affects the distribution of benthic organisms. With the assumption that each taxon prefers a particular combination of abiotic environmental parameters, Macura et al. (2016) also showed that the habitat suitability for organisms is strongly related to river velocity and depth. According to the information indicated in Xu et al. (2014), suitable flow velocities are in the range of 0.3 to 0.8 m/s, whereas unsuitable flow velocities are < 0.3 or > 0.8 m/s. The average velocity in this study gradually decreased from 0.35 m/s (upstream) to 0.284 m/s (midstream), and it

sharply fell to 0.176 m/s in the downstream reach from the dam. As per Xu et al. (2014), this river velocity is unsuitable for several intolerant macroinvertebrates such as EPT. The decrease in velocity in the downstream reach is considered to be largely due to the impacts of the dam. Bunn and Arthington (2002) and Reid et al. (2019) reported that riverine benthic macroinvertebrates are susceptible to dam-induced flow and thermal changes, and altered hydrochemistry (Bunn & Arthington, 2002; Reid et al., 2019). Similarly, Mor et al. (2018) reported that macroinvertebrate interspecific relationships and community structure might be affected by river flow alterations.

Next to the river velocity, phosphate concentrations explained about 14% of the total constrained variation in macroinvertebrate community structure. Dams particularly large ones, as the case in this study, may cause significant environmental changes related to nutrients (Mbaka & Mwaniki, 2015), which in turn affect macroinvertebrate community structure. The phosphate concentration gradually increased from the upstream reach sites to the midstream reach and then showed a gradual decrease after it passed through KHD into the downstream reach sites. Although reduced the phosphate concentrations below the dam were still elevated and this complicated efforts to identify the specific effects of the dam.

River velocity and depth, phosphate, as well as water temperature, and EC were highlighted by the BIOENV routine in PRIMER as having the strongest influence on macroinvertebrate community structure. All these parameters may be influenced by the dam. For example, a similar study by Wang et al. (2020) found that EC, velocity, and phosphate decreased at sites downstream of dams. Other environmental parameters, in this study, such as nitrate-nitrogen, BOD5, water temperature, pH, DO, river width, substrate index, and turbidity did not significantly affect the variation in the distribution of macroinvertebrates.

Overall, this study found significant differences among the three study reaches in terms of several metrics addressing the structures, composition, and functional feeding attributes of macroinvertebrates that relate, directly or indirectly, to flow regime modifications. Even, in the presence of several anthropogenic disturbances particularly in the urban areas (Asfaw et al., 2014; Getachew et al., 2020; Mersha et al., 2021), a large number of industries in Addis Ababa and the surrounding cities (Degefu et al., 2013; Getachew et al., 2020), and trace metal pollution (Aschale et al., 2016), the assemblage structures of the upstream and midstream reaches were far better than the downstream reach. However, the assessment of the impact of the dam is partly complicated by these water quality problems that originate upstream of the dam.

Conclusion

A combination of physicochemical and hydromorphological variables in this study explained why macroinvertebrate community structure and composition differed across the three study reaches. Apart from phosphate, the water flow change (river velocity) was highlighted as a key variable structuring the macroinvertebrate communities and a variable that can be altered by damming. Environmental flow assessments (EFAs) are fundamental considerations in the integration of the water-energy-ecosystem nexus (Kuriqi et al., 2020) but have received negligible attention worldwide (King et al., 1999) including in

Ethiopia. Although other possible mechanisms for observed differences may exist in the study reaches, such as water pollution, the results can help stakeholders consider and minimise the impact of existing and proposed hydropower dams on aquatic ecology by implementing EFAs and maintaining environmental flow requirements (EFRs) in rivers downstream of the hydropower dam in Ethiopia and elsewhere.

Declarations

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Availability of data and materials

Data are available from the corresponding authors upon reasonable request.

Conflict of interest

The authors declare no conflict of interest.

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Figures

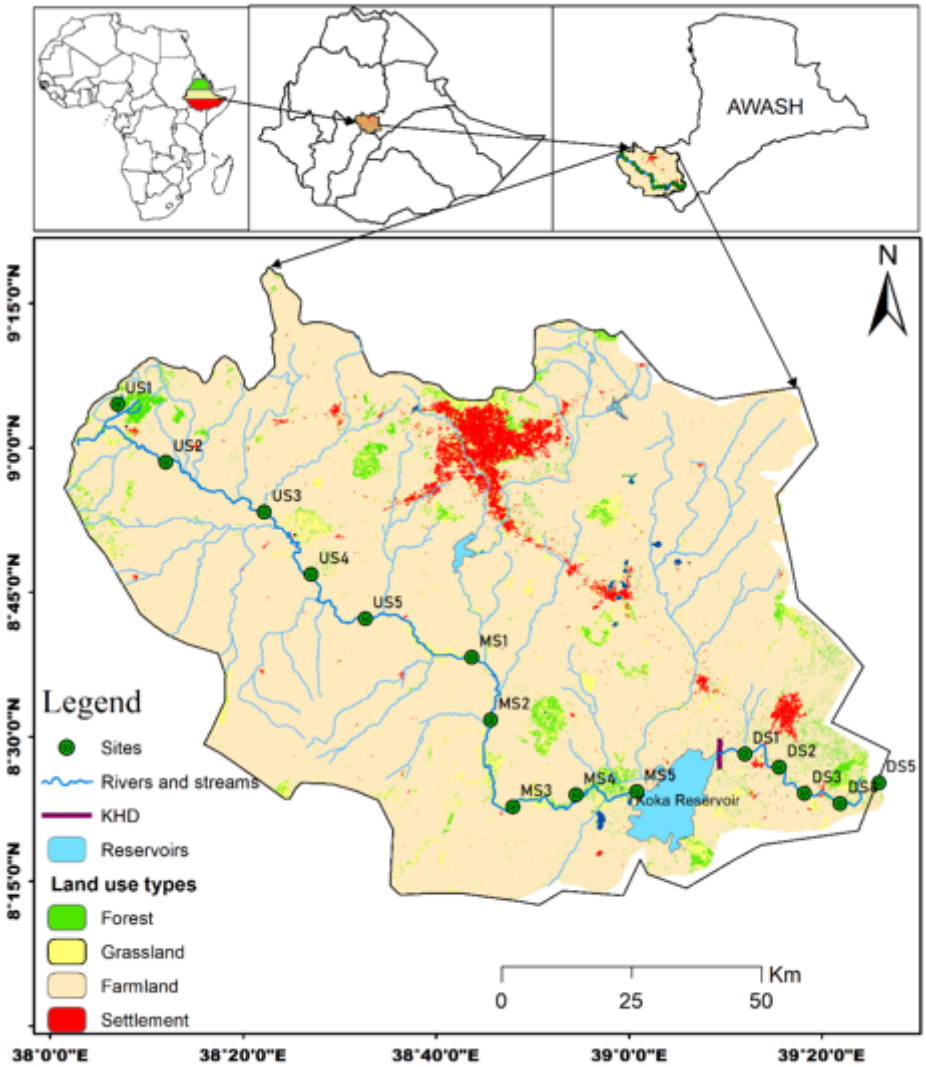

Figure 1

The locations of sampling sites, land-use types, and Koka hydropower dam in the Upper Awash River Basin in the central part of Ethiopia, 2021 using GIS 10.5

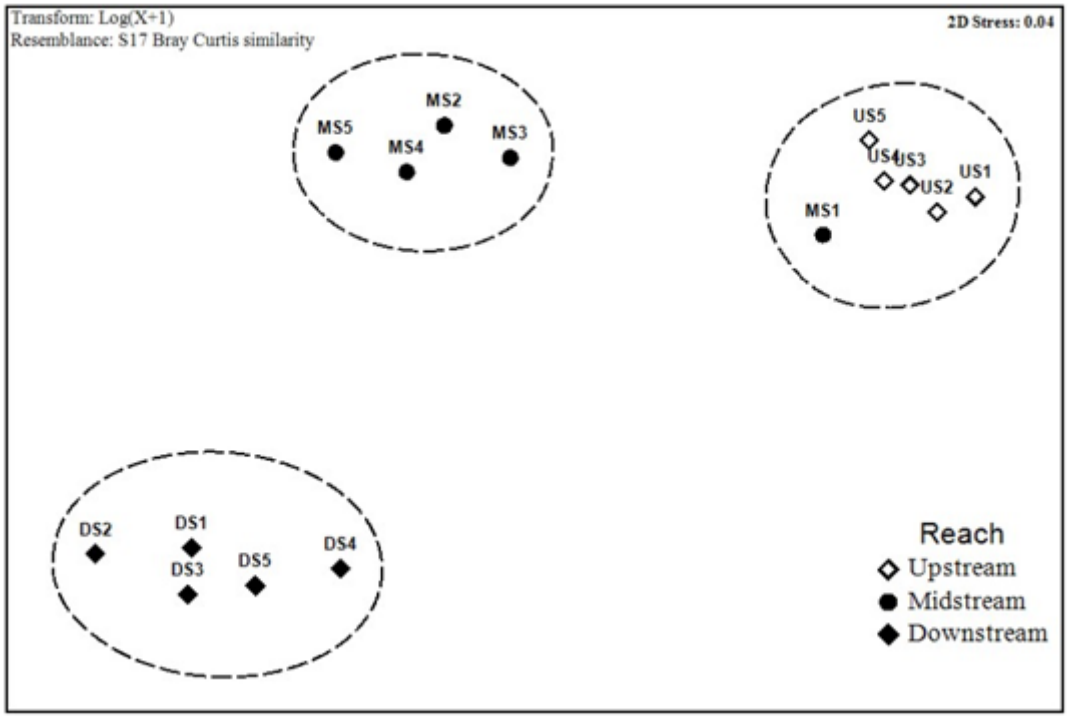

Figure 2

The MDS ordination plot with an overlay of clusters with 70% similarity of macroinvertebrate community structure based on relative abundances along the Awash River in 2021 using PRIMER version 6

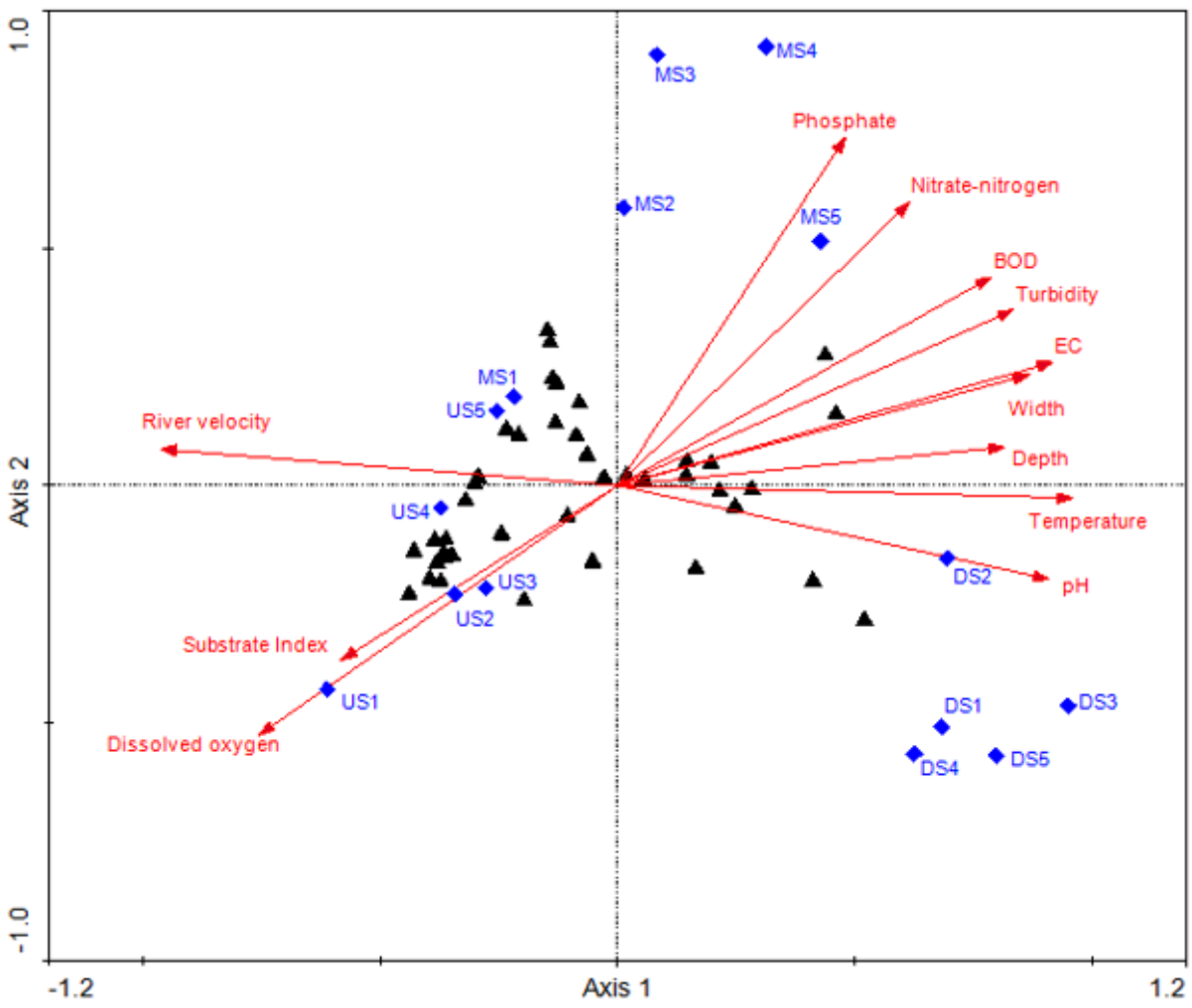

Figure 3

The CCA triplots showing relationships between sites (Diamond), taxa (triangles), and environmental variables (arrows) in the upstream, midstream, and downstream reaches of the Koka Hydropower dam in 2021 using CANOCO 4.5