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Comparison of temperate and tropical versions of Biological Monitoring Working Party (BMWP) index for assessing water quality of River Aturukuku in Eastern Uganda

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ABSTRACT

Despite their socio-economic and ecological importance, rivers are among the most threatened ecosystems. As a result, reliable and affordable monitoring system is fundamental for their effective management and conservation. The utility of Biological Monitoring Working Party, BMWP (England, E) index developed for the temperate region and BMWP-CR modified for Costa Rica, the tropics, were compared for assessing water quality of River Aturukuku in Eastern Uganda. Benthic macroinvertebrates were used in the biomonitoring study because of their wide spectrum of sensitivity to changes in water physico-chemical characteristics. The riverine water quality at upstream site (rural area), four sites in mid-stream (urban area) and one site downstream (rural area), were evaluated using the BMWP indices from February to October, 2018. The Shannon-Wiener diversity index (H′) and selected physico-chemical variables were used to validate performance of the BMWP indices. Although BMWP-CR included more local macroinvertebrate taxa for pollution sensitivity scores than BMWP (E), the performance of both indices was similar. The BMWP (E) and BMWP-CR classified river water quality as bad to moderate, whereas the associated Average Score Per Taxon (ASPT), from England, ASPT (E) and ASPT-CR from Costa Rica indicated moderate to very good category, across seasons. The H′ and physico-chemical variables classified river water quality as bad to moderate. The BMWPs and associated ASPTs allotted sites at urban effluent sources as moderate to very good, while those in rural settings as bad to good, contrary to allocations by H′ and physico-chemical variables. The two BMPs failed to reliably separate sites based on pollution gradient, attributable to biogeographical differences in environmental conditions and pollution tolerances among macroinvertebrates. There is need to adapt a biotic index such as BMWP-CR or develop an indigenous one for Uganda, through an intensive study on local macroinvertebrate assemblages.

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1. Introduction

Worldwide, freshwater ecosystems, particularly lotic ones, are facing high rates of decline in biodiversity due to loss of habitats and catchment degradation resulting from human activities, including land conversion for agriculture, settlement, urban development, industrial establishments, dam construction and pollution (Munir et al., 2016; Dudgeon, 2019). The deterioration in freshwater ecosystems′ functioning is exacerbated by the impact of invasive species, climate change and other emerging threats (Friberg, 2014; Reid et al., 2019). For example, the presence of a dam can reduce downstream flows, declines in precipitation may lead to reduction in the flows and water depths, and wildfires may trigger hydric erosion and associated siltation and compaction of the substrate; all inflicting impairment on survival of the species, including those which are legally protected, in lotic systems (Santos et al., 2015). The introduction of alien fish species has resulted in harming or even displacing native species (Kiruba-Sankar et al., 2018), whereas climate change, such as global warming, poses a threat to about 50% of global freshwater fish species (Reid et al., 2019) and most sensitive species of macroinvertebrates (Kakouei et al., 2018). Moreover, other threats such as the expansion of agriculture towards forested areas with disregard for the soil capability, termed environmental land use conflicts, especially in developing countries, have contributed to the decline of biodiversity (e.g., macroinvertebrates, Valle Junior et al., 2015).

Although lotic systems such as rivers are among the most diverse ecosystems, with immense ecological and socio-economic contributions, they are also the most threatened (Sabater and Elosegi, 2014). As a result, reliable monitoring and assessment programs are fundamental for underpinning effective management of water quality and conservation of freshwater ecosystems (Park and Hwang, 2016). For over a century, the quality of riverine ecosystems has been evaluated through biomonitoring, the systematic use of living organisms or their responses to assess the condition or changes in the environment (Li et al., 2010). Besides, biomonitoring is used to assess the quality and effectiveness of conservation and restoration measures of riverine ecosystems (Pander and Geist, 2013; Stubbington et al., 2018). It complements physical and chemical approaches in water quality assessment (Szczerbińska and Gałczyńska, 2015), which involve *in situ* measurements of variables (e.g., pH and dissolved oxygen) that vary within a short time and/or the collection of water samples for analyses of nutrients and other variables in the laboratory (Dirican, 2015). Based on a wide spectrum of sensitivity to changes in physical and chemical characteristics, benthic macroinvertebrates are the most widely used group in freshwater biomonitoring (Rossaro et al., 2011; Sziválk, and Csabai, 2012). The advantages in using macroinvertebrates for biomonitoring include, being heterogeneous, ubiquitous, abundant, relatively easy to collect, identify and enumerate (Bonada et al., 2006; Deborde et al., 2016). Further, the sedentary nature and long-life cycles of various species facilitate spatial analysis of pollution effects and traceability of such effects over several years (Bonada et al., 2006; Deborde et al., 2016). Whilst freshwater biomonitoring using macroinvertebrates may be less accurate than physical and chemical means, the latter reflects mainly momentary conditions, and is more expensive (Alba-Tercedor, 1996; Aazami et al., 2015).

Biotic indices, such as, the Biological Monitoring Working Party Score System (BMWP) and Average Score Per Taxon (ASPT) are used for biomonitoring riverine ecosystems. The BMWP was developed in 1976 (Biological Monitoring Working Party, 1978), and revised to a final version, 1980 (National Water Council, 1981) in England, as a simplified system to assess water quality, using benthic macroinvertebrates (Hawkes, 1997). Since its development, the BMWP has been widely used in many European countries (Li et al., 2010). Meanwhile, the BMWP was adapted for Spain (Alba-Tercedor, 1996), Thailand (Mustow, 2002), Colombia (Roldán, 2003), and modified for Costa Rica, as BMWP-CR (MINAE, 2007), among others. The adaptation of BMWP for other countries includes addition of new families and changes in some sensitivity scores (Mustow, 2002). However, developing countries, such as Uganda, have neither developed their own biotic indices nor adapted foreign ones for freshwater biomonitoring. Under such circumstances, freshwater ecologists in developing countries often use foreign indices developed from temperate and other biogeographical regions for biomonitoring. For example, Kwitonda (2013) used BMWP Index from England (Armitage et al., 1983) and South African Scoring System (SASS; Dickens and Graham, 2002), without modification, to assess streams within Kampala. Further, without modification, Sekiranda et al. (2004) and Nabirye et al. (2016) applied Hilsenhoff (Family) Biotic Index (Hilsenhoff, 1988) to evaluate the quality of water in northern Lake Victoria; Van Butsel et al. (2017) used SASS (Dickens and Graham, 2002) to assess water quality of Mpanga River, western Uganda.

The reliability and validity of macro-invertebrate indices from temperate and other bio-geographical regions for assessing pollution in tropical rivers, such as those in Uganda is contested (Elias et al., 2014a). Biotic indices, e.g., the BMWP are based on species assemblages and associated environmental sensitivities within their respective regions, which do not necessarily match with those in the tropics (Ochieng et al., 2019). Macroinvertebrate communities respond in different ways to varying environmental and climatic conditions or weather variables, such as temperature (Li et al., 2012). Although most of the macroinvertebrate families used in developing biotic indices (e.g., BMWP index) in the temperate regions are found in the tropics, they do not necessarily occupy the same niches (O'Callaghan and Kelly-Quinn, 2013), nor share similar sensitivities to water pollution (Buss and Salles, 2006).

The objective of this study was to compare on spatial and seasonal basis, the performance of two biotic indices, the BMWP (England, E, temperate region; National Water Council, 1981) and BMWP-modified for Costa Rica (BMWP-CR, tropical region; MINAE, 2007), for assessing water quality of River Aturukuku in Tororo, Eastern Uganda. We hypothesized that when BMWP (E) and BMWP-CR are used in biomonitoring a tropical riverine ecosystem in Eastern Uganda, the latter would be more reliable. The BMWP (E) and BMWP-CR were chosen because they are used globally (Li et al., 2010), and applied on qualitative and family-level data (Mustow, 2002). Further, their numerical form (index or score) render biological data comprehensible to non-biologists, who make decisions on the management of water bodies (Armitage et al., 1983). Besides, for developing countries, research capability and resources are often limiting, hence species-level identification of macroinvertebrates is seldom achieved.

2. Materials and methods

2.1. Study area and selection of sites

The study was undertaken along upstream-downstream reach (about 14.7 km) of River Aturukuku in Tororo, Eastern Uganda (Fig. 1.). Sampling was done during dry (February and August) and wet (July and October) seasons, 2018. The river reach is headwater of first and second stream order category (Fritz and Johnson, 2011), situated within a low-land (approximately 1173 m above sea level, a.s.l.) catchment area of Lake Kyoga. The area experiences humid equatorial type of climate, with rainfall and temperature range of 1215–1328 mm and 16.4–29.3 °C, respectively (Majaliwa et al., 2015). River Aturukuku is permanent in nature, and its mid channel water depth fluctuates from ≤0.5 to 1.0 m during dry and wet seasons, respectively. An upstream site, R (0°43′29"N; 34°14′15"E), within a rural area was established to benchmark minimally disturbed water quality conditions. It provided a basis for comparison with five other sites with diverse human activities and likely pollution within the downstream reach (UNEP-GEMS, 2008). Four sites were established at the mid-stream in Tororo municipality namely: T1 (0°42′90"N; 34°11′14"E), immediate downstream of abattoir effluent and far from R; T2 (0°42′15"N; 34°11′30"E), downstream T1; T3 (0°42′20"N; 34°10′52"E), immediate downstream of sewage effluent after T2; and T4 (0°42′25"N; 34°10′40"E), downstream T3. The last (sixth) site R2 (0°40′49"N; 34° 7′18"E) was established in a rural area at the extreme downstream T4 (Fig. 1.). Elevations (m, a.s.l.) at sites R, T1, T2, T3, T4 and R2 are 1,242, 1,171, 1,172, 1,172, 1166 and 1,126, respectively. The human population density within the upstream, Tororo municipality in the mid-stream and downstream areas of the river reach, based on the national population census (UBOS, 2014), was estimated at 406, 1516 and 529 persons/km^2, respectively.

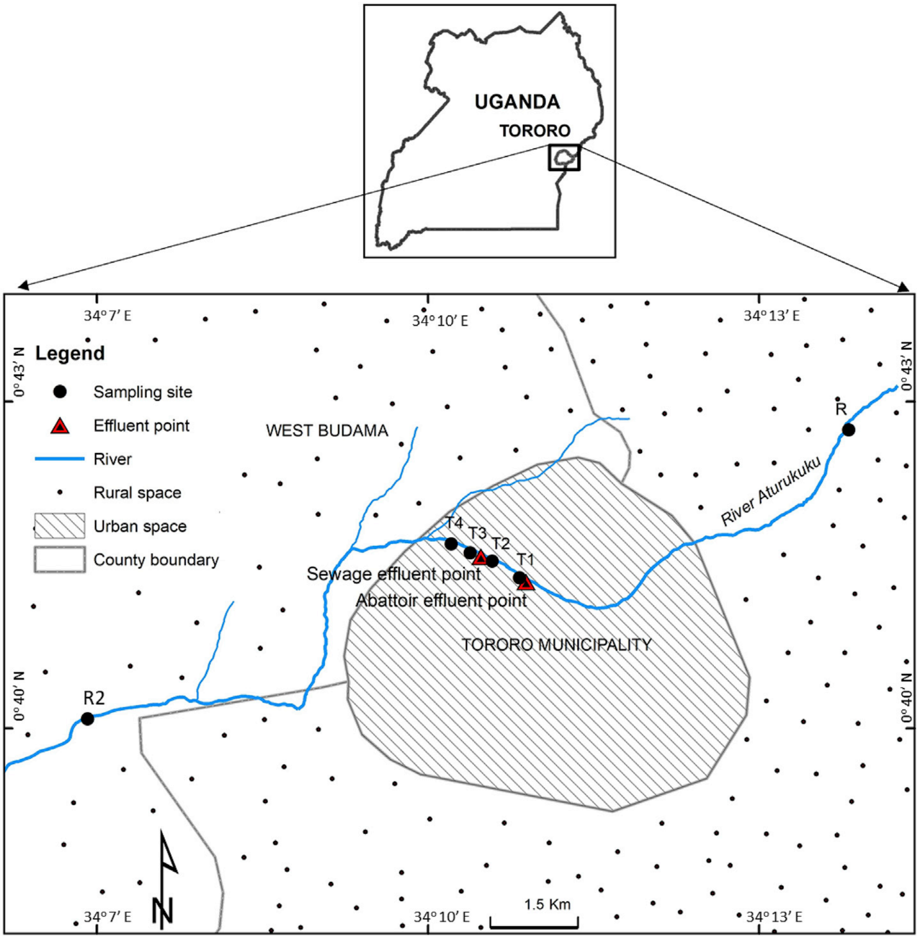

Fig. 1. Location of the study sites along River Aturukuku, Tororo, Eastern Uganda.

2.2. Habitat physical characteristics and human activities at the study sites

As macroinvertebrate assemblages in rivers relate to habitat physical characteristics, selected features such as epifaunal substrates, embeddedness, water velocity (speed of flow) and water discharge (volume of water flowing in a unit time), riffles, runs (Barbour et al., 1999; Silva et al., 2014; Jun et al., 2016), and human activities within the basin (Gichana et al., 2015), were measured during the dry and wet seasons. Epifaunal substrate refers to the relative quantity and variety of natural structures in the river (e.g., cobbles and boulders) and act as refugia, or sites for feeding, spawning and nursery grounds of macroinvertebrates and fish. Meanwhile embeddedness is the extent to which rocks (gravel, cobble and boulders) are covered or sunken into the silt, sand, or mud of the river bottom; affecting availability of the surface area to macro-organisms (Barbour et al., 1999). A riffle is a shallow part of the stream where water flows swiftly over completely or partially submerged obstructions to produce surface agitation, while a run is a relatively shallow part of a stream with moderate velocity and little or no surface turbulence (Fitzpatrick et al., 1998).

At every site, during each of the sampling seasons, water velocity was estimated three times using the floating object method (Blackburn-Lynch et al., 2016). In order to determine water discharge, river wetted width (bank to bank distance covered with water) was measured three times, and depending on its (wetted width) extent, the cross-sectional channel depths (distance between bottom and water surface) were measured four to six times, using a tape measure. Water discharge was calculated as a product of the mean values of river velocity, wetted width and cross-sectional channel depths (Blackburn-Lynch et al., 2016, Table 1).

2.3. Measurement of selected physico-chemical variables, water sample collection and analysis

All the measurements and collection of water samples for nutrient analysis were randomly done in triplicates, from either riffles found at sites T3 and R2 or runs at sites R, T1, T2 and T4 (Table 1). The sampling campaigns provided 72 samples in total, i.e., 2 dry and 2 wet seasons x 3 samples x 6 sites, for physico-chemical and nutrient variables. All sites were sampled on the same day during each campaign, around the middle of the dry season to avoid significant reduction of the water level, and shortly (about two weeks) after start of the rains to assess effects of contamination arising from run-offs, whilst avoiding untrue representation of the biota at the sites during flooding. The water pH, temperature (^{o}C), dissolved oxygen, DO (mg/L) and electrical conductivity, EC (μS/cm), were measured *in situ*, using calibrated multi-parameter meter (Multiprobe, Hach HQ40d). The water samples (1 L each), for analyses of total phosphorus, TP (mg/L); total nitrogen, TN (mg/L), Nitrate-nitrogen, NO_3^-N (mg/L) and total suspended solids, TSS (mg/L), were collected using Nalgene bottles (1000 mL). The water samples were placed in a cool ice box and transported to the National Fisheries Resources Research Institute (NaFIRRI) laboratory, where they were analyzed within 24 h, following standard methods (APHA, 1999).

2.4. Sampling and identification of benthic macroinvertebrates

Bottom substrates for benthic macroinvertebrates were randomly collected in triplicates, on the same day, and from the same habitats of the six sites (Table 1), where measurements of physico-chemical variables and collection of water samples were undertaken. The campaigns provided 72 samples in total, i.e., 2 dry and 2 wet seasons x 3 samples x 6 sites, for macroinvertebrates. The sampling was conducted from downstream towards upstream, to minimize collection of drifted organisms. It was quantitatively conducted using a rectangular-shaped dip net of 50 cm wide by 25 cm high aperture and attached to a bag-shaped Nitex net of 500 μm pore size (Barbour et al., 1999). The kick net was vertically fixed at suitable points, and based on its aperture; an area of 50×50 cm (2500 cm^2) was demarcated using two wooden pegs at its immediate upstream position. Using the toe of the boot, the bottom substrates within the 2500 cm^2 area were then shifted about for about 2 min to allow dislodging and capturing of both more mobile and sessile taxa. The macro-invertebrates dislodged were drifted by water current into the dip net, set downstream.

The collected macroinvertebrates were concentrated using a washing bag (mesh size 500 μm) and each sample separately preserved in plastic sample bottles (100 mL) using 70% ethanol (Barbour et al., 1999; Kripa et al., 2013; Ojija et al., 2017). The samples were transported to NaFIRRI laboratory for immediate analysis. The macroinvertebrates were identified to family level, except for Oligochaeta (other taxon), using selected guides for freshwater macroinvertebrates (Pennak, 1953; Mandahl-Barth, 1954; Merrit and Cummins, 1978; Day and de Moor, 2002; de Moor et al., 2003a, 2003b; Stals and de Moor, 2007). The identified macroinvertebrate families were enumerated and their respective relative abundance (%) calculated, to indicate conditions in the river (Herman and Nejadhashemi, 2015).

2.5. Data analysis

The classifications of water quality on spatial and seasonal basis were compared using BMWP (E) and BMWP-CR, based on: (a) the number of invertebrate taxa that contributed to pollution sensitivity scores, and (b) separation of sites with relatively good water quality from polluted ones (e.g., O'Callaghan and Kelly-Quinn, 2013). Moreover, evaluation was conducted on the direction and strength to which the BMWP scores were correlated with macroinvertebrate diversity index (Shannon-Wiener index, H′), and selected physico-chemical variables, as supplemental indicators of water quality (Zeybek et al., 2014; Tan and

Table 1
Physical characteristics and human activities at the six study sites along River Aturukuku, Tororo, during the period February–October, 2018. Epifaunal substrate abundance (%): $^1 = \leq 5$, $^2 = 10–15$, $^3 = 45–50$, $^4 = \geq 80$. Estimates of mid channel depth, river wetted width and water velocity are mean values ± standard error, $n = 6$.

Site	Epifaunal substrate	Mid channel depth (m)	Wetted width (m)	Velocity (m/s)	Discharge (m^3/s)	Other characteristics and human activities
R	Fine sand^4, Silt^1	0.6 ± 0.1	2.1 ± 0.1	0.2 ± 0.0 (Slow)	0.2	• Rural setting, nearby (<1 km) homesteads • a run present, embeddedness absent • increased sand deposition in wet season • reduced (<50%) water level in dry season • small strips (5 m wide) of vegetation on banks • subsistence gardens, livestock grazing • rampant sand mining, fishing, washing.
T1	Silt^4, fine sand^3	0.5 ± 0.1	4.9 ± 0.1	0.2 ± 0.0 (Slow)	0.5	• Urban area, about 6 km downstream site R • receives untreated abattoir effluent • a run present, embeddedness absent • high deposition of fine sediment in wet season • reduced (<50%) water level in dry season • inflow of water from a protected spring • small strips (5 m wide) of vegetation on banks • subsistence gardens, livestock grazing • washing, fishing, solid waste disposal • crop irrigation by river channel diversion
T2	Soft mud^4, plant debris^1	0.3 ± 0.0	4.3 ± 0.1	0.6 ± 0.0 (Fast)	0.5	• Urban area, a run present, mbeddedness absent • reduced (75%) water level in dry season • small strips (<5 m wide) of vegetation on banks • gardens and planted trees dominate along banks • vehicle washing, bathing, crop irrigation • solid waste disposal, fishing.
T3	Boulders^4, stone^2,silt^1	0.2 ± 0.0	4.4 ± 0.3	0.9 ± 0.0 (Fast)	0.8	• Urban area, 200 m downstream sewage effluent • few riffles, embeddedness optimal (<25%) • about 25% substrate exposed in dry season • vegetation (>10 m wide) along upstream banks • gardens on banks, livestock grazing, fishing • washing, bathing and solid waste disposal.
T4	Clay^4, Coarse sand^2,	0.3 ± 0.0	3.5 ± 0.3	0.7 ± 0.0 (Fast)	0.5	• Urban area, near a slum • a run present, embeddedness absent, • reduced (75%) water level in dry season • inflow of water from a protected spring • planted elephant grass and trees on banks • gardens, livestock grazing, fishing, solid wastes
R2	Stones^3 coarse sand^3 silt^1	0.5 ± 0.0	7.9 ± 0.5	0.6 ± 0.0 (Fast)	1.7	• Remote rural, swampy area, 7 km downstream T4 • few riffles, embeddedness optimal (<25%) • about 25% substrate exposed in dry season • a stream inflow 250 m upstream, some vegetation • gardens, washing, fishing, livestock grazing.

Beh, 2015; Brraich and Kaur, 2017). Determination of the biotic metrics and application of the statistical methods, were conducted following specific criteria.

2.5.1. Biotic indices

The BMWP Score system is based on identification of macroinvertebrates to family level, except for Oligochaeta (National Water Council, 1981; MINAE, 2007). Macroinvertebrates identified at higher taxonomic level (except for Oligochaeta) are excluded from the calculation of the BMWP index. Each of the identified families is allocated a score, ranging from 1 to 10 (National Water Council, 1981) and 1 to 9 (MINAE, 2007). The most sensitive organisms (e.g., mayfly and stonefly nymphs), either score 10 (National Water Council, 1981) or 9 (MINAE, 2007), molluscs 3 and the least sensitive (Oligochaeta) 1 (National Water Council, 1981; MINAE, 2007). The values for each family are summed up independently from their abundance and generic diversity, thus sensitivity scores higher than 120 points indicate an undisturbed (excellent) aquatic system, while those lower than 15 points indicate highly polluted (poor) aquatic system (MINAE, 2007; Stein et al., 2008).

Based on the BMWP total scores, six levels of water quality may be established (Alba-Tercedor, 1996, Table 2). The total BMWP score can also be divided by the number of taxa to produce the Average Score Per Taxon, ASPT (Chapman, 1996), which

Table 2
Categories of water quality as defined by the numerical values and colors of the Biological Monitoring Working Party, BMWP, index (Alba-Tercedor, 1996; MINAE, 2007), and Average Score Per Taxon, ASPT, index (ASPT-CR, O'Callaghan and Kelly-Quinn, 2013; ASPT(E), Ganguly et al., 2018).

Water quality category	BMWP total score	BMWP color equivalents	ASPT (E) score	ASPT-CR score
Waters with excellent quality	>120	Light blue	>5.4	>6
Waters with good quality, no contaminations or obvious distortions	101–120	Dark blue	4.8–5.4	4–5
Waters with regular quality, Eutrophic, moderate contamination	61–100	Green	4.3–4.8	3–4
Waters with bad quality, Contaminated	36–60	Yellow	3.6–4.3	2–3
Waters with bad quality, very contaminated	16–35	Orange	3.0–3.6	1–2
Waters with very bad quality, extremely contaminated	<15	Red		

is less influenced by season and sample size (Armitage et al., 1983; Muralidharan et al., 2010). A high ASPT score (e.g., greater than 4) indicates clean water, containing a large number of high scoring taxa (Armitage et al., 1983; O'Callaghan and Kelly-Quinn, 2013). Specifically, ASPT (England, E) values (Ganguly et al., 2018) and the corresponding ones for Costa Rica, ASPT-CR (O'Callaghan and Kelly-Quinn, 2013), represent five levels (excellent, very good, good, moderate and bad) of water quality (Table 2). To ease comparison between the BMWP with six levels and ASPT with five levels, of water quality grading, we combined the last two scores (16–35 and < 15) of BMWP (Table 2) to represent bad category, and assigned numerical codes to only five levels of quality, i.e., 1, 2, 3, 4 and 5 (for bad, moderate, good, very good and excellent, respectively). The BMWP and ASPT scores were calculated for each sampling campaign and averages for four and two months used for spatial and seasonal water quality classifications, respectively.

2.5.2. Diversity index

Shannon-Wiener Index, H′ (Magurran, 2004) was calculated because in most cases species diversity increases with good water quality (Brraich and Kaur, 2017). Pooled data of macroinvertebrate species counts per sampling season (dry or wet) and for both seasons, were used to calculate H′ on seasonal and spatial basis, respectively. The H′ was calculated based on the formula:

$$H' = - \Sigma\ [(pi)(\ln pi)] \quad (1)$$

Thus, from formula (1), the proportion of species i relative to the total number of species (p_i) is calculated, multiplied by its natural logarithm ($\ln p_i$), then the resulting product across species summed up, and finally multiplied by negative one (-1).

2.5.3. Statistical methods

Statistical analyses on macroinvertebrate and physico-chemical data were undertaken using Minitab 18 Statistical Software; based on $p = 0.05$ as the alpha level of significance. Data were tested for normal distribution using Ryan-Joiner test for normality (Ukponmwan and Ajibade, 2017) before applying the parametric tests. To detect the effects of seasons (dry and wet) and sites (with different pollution levels) on physico-chemical variables, General Linear Model (GLM) under ANOVA was applied (Chapman et al., 2004). In this model, diagnostic plots of the residuals versus fit plot and normal probability were used to verify the assumption that the residuals were randomly distributed and had constant variance, and that they were normally distributed (Schützenmeister et al., 2012). To establish the significant variables in the outputs, a pairwise comparison based on Tukey method was used (Kwok et al., 2007). Relationship between BMWP and ASPT scores, H′, selected physico-chemical variables and habitat physical attributes, were determined using Pearson correlation coefficient (Abbaspour et al., 2017). The overall mean values of selected physico-chemical variables (e.g., pH, DO, TP and TN) were compared with the proposed benchmark (annual means) for riverine freshwater ecosystems by the International Water Quality Guidelines for Ecosystems (IWQGES, 2016), to confirm the water quality classifications generated using the biotic indices. To spell out key variables that explain a high variability in the data set, Principal Component Analysis (PCA) was used. This allowed selection of physico-chemical variables and habitat physical attributes associated with the major sources of variation in biological metrics and sites while minimizing redundant data. During this analysis, Principal Components (PCs) with eigenvalues greater than one were retained as those explaining the highest total variability in water quality. Biplots for the first two components were constructed out of the resultant scores and loadings to provide an overall view of the relations among the multi-variables within sites (Muriithi and Yu, 2015; Jabbar and Grote, 2019). Hierarchical cluster analysis involving Ward's linkage method, was performed on macroinvertebrate species compositions, to show major differences and similarities among the study sites based on pollution gradients (Mangadze et al., 2016).

3. Results

3.1. Status of physico-chemical variables on spatial and seasonal basis and water quality classification

The mean values of physico-chemical variables and their statistical differences on spatial and seasonal basis, are presented (Table 3). The Temperature values were highest at site T1 and lowest at R2 ($p < 0.001$), whereas NO_3^-N values were highest at sites T1, T2 and T3 and lowest at R2 ($p < 0.05$). Except for TP and TSS, the values of Temperature, pH, EC and NO_3^-N were higher

Table 3
Characteristics of selected water physico-chemical variables (mean ± standard error) and their differences on spatial ($n = 72$) and seasonal ($n = 36$) basis within River Aturukuku, February to October 2018. The *P*-value a = between sites, b = between the two seasons, bold and underlined = difference highly significant ($p < 0.01$), bold = difference significant ($p < 0.05$). Mean ± standard error based on $n = 12$ for each site and $n = 6$ for each season.

Variable	Space/season	Site R	T1	T2	T3	T4	R2	*P*-values **a**	**b**
								a	**b**
Temperature (°C)	Spatial	25.6 ± 0.4	27.2 ± 0.3	26.2 ± 0.6	24.7 ± 0.4	24.0 ± 0.5	23.1 ± 0.4	<**0.001**	
	Dry	26.5 ± 0.4	27.6 ± 0.4	27.3 ± 0.3	24.9 ± 0.1	24.1 ± 0.2	23.1 ± 0.1		
	Wet	24.6 ± 0.4	26.8 ± 0.5	25.1 ± 0.9	24.6 ± 0.9	23.8 ± 0.1	23.1 ± 0.7		**0.01**
pH	Spatial	6.8 ± 0.3	6.7 ± 0.3	6.9 ± 0.3	7.1 ± 0.2	7.3 ± 0.2	7.1 ± 0.4	0.50	
	Dry	7.5 ± 0.0	7.5 ± 0.0	7.6 ± 0.0	7.7 ± 0.1	7.9 ± 0.1	8.0 ± 0.1		
	Wet	6.0 ± 0.4	6.0 ± 0.5	6.2 ± 0.4	6.5 ± 0.3	6.6 ± 0.3	6.2 ± 0.6		<**0.001**
EC (μS/cm)	Spatial	244.4 ± 30.9	219.7 ± 8.9	305.3 ± 31.1	334.0 ± 36.4	317.3 ± 34.6	269.2 ± 33.5	0.12	
	Dry	328.8 ± 34.1	326.7 ± 15.8	374.3 ± 13.0	386.5 ± 11.7	392.0 ± 16.0	327.8 ± 13.8		
	Wet	159.9 ± 13.5	312.7 ± 9.1	336.3 ± 46.6	281.6 ± 67.9	242.6 ± 52.7	210.5 ± 58.0		<**0.001**
DO (mg/L)	Spatial	5.4 ± 0.3	4.8 ± 0.6	5.5 ± 0.2	5.6 ± 0.2	5.5 ± 0.3	6.0 ± 0.1	0.21	
	Dry	4.8 ± 0.5	3.9 ± 1.0	5.2 ± 0.3	5.2 ± 0.4	5.2 ± 0.5	6.2 ± 0.2		
	Wet	5.9 ± 0.3	5.7 ± 0.4	5.8 ± 0.1	5.9 ± 0.2	5.7 ± 0.3	5.8 ± 0.1		**0.01**
TP (mg/L)	Spatial	0.2 ± 0.0	0.3 ± 0.1	0.3 ± 0.1	0.4 ± 0.1	0.4 ± 0.1	0.2 ± 0.1	0.23	
	Dry	0.1 ± 0.0	0.2 ± 0.1	0.2 ± 0.1	0.4 ± 0.1	0.3 ± 0.0	0.2 ± 0.1		
	Wet	0.2 ± 0.1	0.4 ± 0.2	0.4 ± 0.2	0.4 ± 0.2	0.5 ± 0.2	0.3 ± 0.1		0.09
TN (mg/L)	Spatial	55.3 ± 25.0	34.8 ± 12.6	40.1 ± 17.4	37.2 ± 15.9	29.2 ± 11.0	47.1 ± 18.5	0.86	
	Dry	8.1 ± 2.4	15.5 ± 1.6	9.0 ± 1.4	9.4 ± 1.9	11.8 ± 1.6	17.9 ± 8.3		
	Wet	102.6 ± 42.9	54.2 ± 23.3	71.2 ± 30.8	64.9 ± 28.2	46.5 ± 20.2	76.4 ± 33.1		<**0.001**
NO_3^-N (mg/L)	Spatial	0.7 ± 0.3	1.1 ± 0.3	1.3 ± 0.3	1.3 ± 0.3	0.9 ± 0.1	0.3 ± 0.1		**0.01**
	Dry	1.1 ± 0.5	1.6 ± 0.5	1.9 ± 0.5	1.9 ± 0.4	1.2 ± 0.2	0.3 ± 0.1		
	Wet	0.3 ± 0.1	0.7 ± 0.2	0.6 ± 0.2	0.7 ± 0.2	0.5 ± 0.1	0.3 ± 0.1		<**0.001**
TSS (mg/L)	Spatial	129.4 ± 41.2	56.3 ± 5.6	71.1 ± 9.8	65.2 ± 9.5	82.2 ± 8.2	54.4 ± 9.1	0.06	
	Dry	77.4 ± 4.2	73.0 ± 3.7	74.9 ± 7.3	86.1 ± 10.5	104.2 ± 3.8	84.2 ± 2.9		
	Wet	181.2 ± 79.7	39.6 ± 3.7	67.4 ± 19.1	44.2 ± 10.4	60.2 ± 9.3	24.7 ± 2.2		0.37

during dry than wet season ($p < 0.001$), and those higher in wet than dry season were DO ($p < 0.05$) and TN ($p < 0.001$, Table 3). In comparison, the mean values of selected variables, such as, DO (4.8–6.0 mg/L), TP (0.2–0.4 mg/L) and TN (29.2–102.6 mg/L) at the study sites (Table 3) were close to benchmark means (DO, 3.0–6.0 mg/L; TP, >0.19 mg/L and TN, >2.5 mg/L; IWQGES, 2016), associated with bad to moderate water quality in a riverine freshwater ecosystem. According to IWQGES (2016), the DO values allotted all sites to moderate water quality; TP allotted R and R2 to moderate, and other sites to bad water quality; and TN allotted all sites to bad water quality. Overall, the physico-chemical variables classified the river water quality as bad to moderate categories.

3.2. Benthic macroinvertebrate families/other taxon and their contributions to BMWP scores

A total of 27 families/other taxon of benthic macroinvertebrates were found at the six study sites within River Aturukuku in Tororo, during the period February to October 2018. The number of families/other taxon were 9–19 at the six sites; the highest being at T3 (18), T4 (19) and R2 (17). Out of the 27 families/other taxon, 19 (70.4%) and 23 (85.2%) contributed towards scoring for the original BMWP (E) and BMWP-CR, respectively. All the families/other taxon scored for both biotic indices, except Protoneuridae, Elmidae, Muscidae, Tabanidae and Ceratopogonidae, which scored only for BMWP-CR and Ancylidae for only BMWP (E). Tricorythidae, Thiariidae, and Potamonautidae did not score for any of the biotic indices. The macroinvertebrate families with highest composition (>80%) in the river were Chironomidae, Simuliidae, Baetidae, Caenidae, Oligochaeta, Planorbidae, Libellulidae, Gomphidae and Coenagrionidae. Greater dominance by specific families was at site R, with Chironomidae forming 95% of the community composition, T1 (Oligochaeta, 66%), T3 (Simuliidae, 88%) and, T4 (Simuliidae, 87%), compared to sites T2 (Simuliidae, 45%) and R2 (Simuliidae, 49%, Table 4).

3.3. Biotic index scores, diversity index and water quality classification on spatial basis

The scores for BMWPs and ASPTs, the respective water quality classes and H′ on spatial basis, are shown (Table 5). The scores for BMWP (E) allotted sites R, T1 and R2 to bad water quality, while T2, T3 and T4 were moderate. Almost similar to this, the BMWP-CR scores indicated that sites R, T1, T2 and R2, were of bad quality, and T3 and T4, were of moderate water quality. The ASPT (E) scores allotted site T1 to moderate, R, T3, T4 and R2 to moderate, and T2 to very good. The ASPT-CR scores allotted sites R, T1, T2 and R2 to good, while T3 and T4 were very good. There was similarity in classification between BMWP (E) and BMWP-CR with a quality category of bad to moderate. Also, ASPT (E) and ASPT-CR were similar, being moderate to very good. However, the scores from both biotic indices did not clearly separate sites with relatively good water quality from

Table 4
Spatial occurrence (√ = present, relative abundance <50%; √* = present, relative abundance >60%; o = absent) of benthic macroinvertebrate families/other taxon at sites R, T1, T2, T3, T4 and R2, and their contributions towards scores (● = scored, X = did not score) for the BMWP (E) and BMWP-CR indices.

Family/other taxon	R	T1	T2	T3	T4	R2	BMWP (E)	BMWP-CR
Heptageniidae	o	✓	✓	✓	o	O	●	●
Baetidae							●	●
Leptophlebidae	o	o	o			O	●	●
Caenidae							●	●
Tricorythidae	o	o	o	o			X	X
Hydropsychidae	o	o					●	●
Lepidostomatidae	o	o	o	o		O	●	●
Libellulidae						O	●	●
Gomphidae							●	●
Protoneuridae	o					O	X	●
Coenagrionidae						O	●	●
Sphaeriidae	o			o	o		●	●
Thiaridae	o		o	o			X	X
Planorbidae						O	●	●
Ancylidae	o	o	o	o	o		●	X
Dyscitidae	o	o	o	o	o		●	●
Gyrinidae			o	o	o		●	●
Elmidae	o	o	o		o		X	●
Naucoridae	o	o	o				●	●
Potamonautidae	o	o	o	o	o		X	X
Simuliidae				√*	√*		●	●
Muscidae	o	o	o		o	O	X	●
Tabanidae	o	o	o			O	X	●
Chironomidae	√*						●	●
Ceratopogonidae	o	o	o	o			X	●
Glossiphoniidae	o		o			o	●	●
Oligochaeta	o	√*					●	●
Total present/scored	9	15	13	18	19	17	19	23
% present/scored	33.3	55.5	48.2	66.7	70.4	63.0	**70.4**	**85.2**

Table 5
BMWP scores, ASPT scores, diversity index and water quality classifications at study sites along River Aturukuku, Tororo. Water quality class 1 = bad, 2 = moderate, 3 = good, 4 = very good, and 5 = excellent.

Site	R	T1	T2	T3	T4	R2
BMWP (E)	23	27	37	44	38	31
Water quality class	1	1	2	2	2	1
ASPT (E)	4.4	3.9	5.1	4.8	4.8	4.6
Water quality class	3	2	4	3	3	3
BMWP-CR	19	26	34	46	38	28
Water quality class	1	1	1	2	2	1
ASPT-CR	3.7	3.6	3.9	4.3	4.4	4.0
Water quality class	3	3	3	4	4	3
H′	0.28	0.86	1.66	0.46	0.59	1.61

polluted ones. The H′ values were from 0.28 to 0.86 at sites R, T1, T3 and T4, and 1.61 to 1.66 at T2 and R2. The H′ (>1) separated sites T2 and R2 from other sites (Table 5).

3.4. Biotic index scores, diversity index and water quality classification on seasonal basis

During both dry and wet seasons (Table 6), water quality at sites R, T1 and R2 were allotted by BMWP (E) scores as bad, while T2, T3 and T4 were allotted moderate. These allocations were similar to those by the BMWP-CR scores during the dry season. However, in the wet season, the BMWP-CR scores allotted sites R, T1, T2, T4 and R2 to bad water quality, and T3 to moderate. In the dry season, the ASPT (E) scores allotted sites R and T1 as moderate, T2 and R2 good, and T3 and T4 as very good. The ASPT-CR scores allotted sites R, T1 and T2 to good quality, and T3, T4 and R2 to very good quality. In the wet season, the ASPT (E) scores indicated that water quality at sites T1 and T4 was moderate, good at T3 and R2, and very good at R and T2. The ASPT-CR scores allotted sites R, T1, T4 and R2 to good water quality, and T2 and T3 to very good. Both BMWP and ASPT scores indicated that urban/point source sites such as T3 and T4, compared to others, had relatively better water quality during each of the two seasons. The BMWPs and associated ASPTs, classified the river water quality as bad to very good, across the two seasons. In the dry season, the H′ values were lower (0.06–0.77) for sites R, T1 and T3, compared to sites T2, T4 and R2 (1.59–2.02). During the wet season, the H′ values were lower (0.47–0.52) for sites T3 and T4, compared to sites R, T1, T2 and

Table 6
BMWP scores, ASPT scores, diversity index and water quality classifications on seasonal basis at study sites along River Aturukuku, Tororo. D = dry season, W = wet season. Water quality class 1 = bad, 2 = moderate, 3 = good, 4 = very good, and 5 = excellent.

Site/season	R D	R W	T1 D	T1 W	T2 D	T2 W	T3 D	T3 W	T4 D	T4 W	R2 D	R2 W
BMWP (E)	19	28	24	31	39	36	42	46	41	36	30	33
Water quality class	1	1	1	1	2	2	2	2	2	2	1	1
ASPT (E)	3.9	4.9	3.7	4.2	4.8	5.3	4.9	4.6	5.3	4.2	4.8	4.4
Water quality class	2	4	2	2	3	4	4	3	4	2	3	3
BMWP-CR	15	23	24	28	39	30	47	44	44	33	24	33
Water quality class	1	1	1	1	2	1	2	2	2	1	1	1
ASPT-CR	3.3	4.0	3.6	3.7	3.4	4.4	4.4	4.1	5.1	3.7	4.2	3.9
Water quality class	3	3	3	3	3	4	4	4	4	3	4	3
H′	0.06	1.42	0.77	1.91	1.59	1.52	0.35	0.47	1.78	0.52	2.02	1.40

R2 (1.40–1.91). The sites T2 and R2 had relatively higher macroinvertebrate diversity (H′>1) during each of the two seasons, compared to other sites (H′<1; Table 6).

3.5. Relationship between physico-chemical variables, biotic index scores and habitat physical attributes

There was no significant correlation between H′ and BMWP (E), ASPT (E), BMWP-CR and ASPT-CR ($p > 0.05$). Among the physico-chemical variables, Temp negatively correlated with ASPT-CR, while DO positively correlated with ASPT (E) ($p = 0.01$). TP had a positive correlation with BMWP (E) and BMWP-CR ($p < 0.01$; Table 7). Among the habitat physical attributes, positive correlations were between MP and ASPT (E) ($p = 0.05$); BST and BMWP (E), and BMWP-CR ($p = 0.01$); Fast F and BMWP (E), and BMWP-CR ($p < 0.001$), ASPT (E) ($p < 0.01$), and ASPT-CR ($p = 0.02$). Negative correlations were between FS and BMWP (E), and BMWP-CR ($p < 0.01$); Slow F and BMWP (E), and BMWP-CR ($p < 0.001$), ASPT (E) ($p < 0.01$), and ASPT-CR ($p = 0.02$; Table 7). There was a high positive correlation between BMWP (E) and BMWP-CR ($p < 0.001$), and ASPT (E) and ASPT-CR ($p < 0.001$). Other positive correlations were between BMWP (E) and ASPT (E) ($p < 0.01$), BMWP (E) and ASPT-CR ($p < 0.01$), BMWP-CR and ASPT (E) ($p = 0.02$), and BMWP-CR and ASPT-CR ($p < 0.01$; Table 7).

3.6. Major parameters contributing to high variability in data set among the study sites

The PCA correlation biplot shows that 68.6% of the total variance in the data is attributable to first (44.9%) and second (23.7%) Principal Components (PCs) (Fig. 2). The first axis of the PCA, significantly correlated with Cond, TP, pH, Fast F, BMWP (E), ASPT (E), BMWP-CR and ASPT-CR; separating the urban sites T2, T3 and T4 from others. On the opposite left side, the first axis negatively correlated with TN, FS and Slow F, associated with the rural upstream site R. The second axis of the PCA had positive correlation with Temp and NO_3^-N, and a negative one with DO on the opposite bottom side; separating the urban/point source site T1 from the rural downstream site R2, respectively. The PCA biplot showed a high dissimilarity in water quality among sites R, R2, T1 and cluster of T2, T3 and T4. Despite being in a cluster, site T2 was more positioned around the origin, relatively far from T3 and T4 (Fig. 2).

3.7. Similarity groupings of the study sites based on macroinvertebrate species compositions

The cluster analysis on differences and similarities in macroinvertebrate species compositions among study sites, provided three clusters. It separated sites R and R2 (rural areas) and T2 (urban but not close to effluent source), from T1 (urban and close to abattoir effluent), and T3 and T4 (urban and close to sewage effluent) (Fig. 3).

4. Discussion

4.1. Overall performance of the BMWP-CR and BMWP (E) indices

The use of benthic macroinvertebrates in biomonitoring the quality of riverine ecosystems (Guareschi et al., 2017), have for over a century (Buss et al., 2015) been mainly in the United Kingdom, many European countries, United States of America, Australia, New Zealand and Canada (Bo et al., 2016). Due to its reliability as a tool in freshwater ecology management, it continues to be adopted for several developing countries in South America, East Asia and Africa (Buss et al., 2015). We compared the utility of the original BMWP from England, BMWP (E) and that modified for Costa Rica (BMWP-CR) to assess the water quality of River Aturukuku in eastern Uganda.

The inclusion of more taxa in BMWP-CR than BMWP (E), in the present calculation of the index, showed that the former could accommodate a wider range of local benthic macroinvertebrates with varying pollution sensitivities for use in Ugandan rivers. The macroinvertebrates, and their respective pollution sensitivity levels used to develop BMWP-CR are more similar to those in this study (tropical region) than for England (temperate region). This is attributable to differences in macroinvertebrate occurrences (Boulton et al., 2008; Ochieng et al., 2019) and their respective pollution sensitivities (Kwok et al.,

Table 7
Pearson correlation coefficient (r) for biotic and diversity indices versus physico-chemical variables ($n = 18$) and habitat physical attributes (bottom substrate FS = fine sand and silt, MP = soft mud and plant debris, BST = boulders and stones, CCS = Clay and coarse sand, SS = stones and coarse sand; water velocity Slow F = slow flowing water, Fast F = fast flowing water). Correlations in bold and underlined = highly significant ($p < 0.01$), bold = significant ($p < 0.05$).

Attributes	BMWP (E)	ASPT (E)	BMWP-CR	ASPT-CR
H′	0.20	0.43	0.13	0.25
Temperature (°C)	−0.35	−0.45	−0.26	**−0.55**
pH	0.03	0.08	0.11	0.16
EC (μS/cm)	0.23	−0.02	0.34	0.12
DO (mg/L)	0.41	**0.58**	0.21	0.35
TP (mg/L)	**0.69**	0.16	**0.67**	0.38
TN (mg/L)	0.07	0.10	−0.05	0.04
NO_3^-N (mg/L)	0.20	−0.04	0.35	−0.06
TSS (mg/L)	−0.21	0.29	−0.24	0.19
FS (%)	**−0.67**	−0.19	**−0.71**	−0.34
MP (%)	0.25	**0.46**	0.16	−0.07
BST (%)	**0.56**	0.18	**0.59**	0.31
CCS (%)	0.30	0.15	0.32	0.39
SS (%)	−0.09	0.05	−0.12	0.08
Slow F (%)	**−0.81**	**−0.67**	**−0.75**	**−0.56**
Fast F (%)	**0.81**	**0.67**	**0.75**	**0.56**
BMWP (E)	1.00	**0.66**	**0.97**	**0.61**
ASPT (E)	1.00	**0.54**	**0.75**	
BMWP-CR	1.00	**0.58**		
ASPT-CR	1.00			

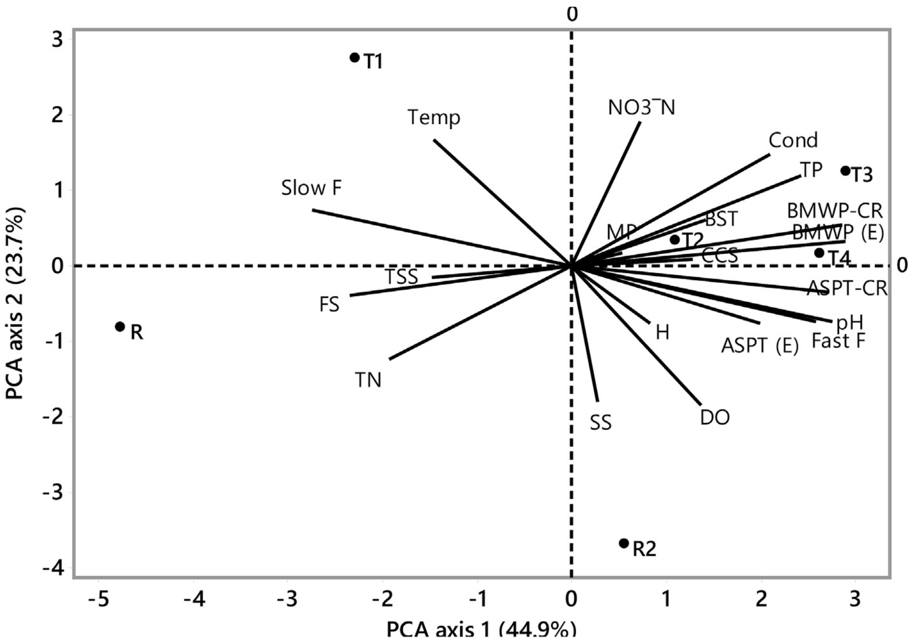

Fig. 2. PCA correlation biplot of the 20 variables (eight physico-chemical, four biotic indices, one diversity index and seven habitat physical attributes) at sites. Physico-chemical: Temp = temperature, pH, Cond = electrical conductivity, DO = dissolved oxygen, TP = total phosphorus, TN = total nitrogen, NO_3^-N = nitrate-nitrogen, TSS = total suspended solids. Biotic and diversity indices: BMWP (E) = Biological Monitoring Working Party (England), BMWP-CR = Biological Monitoring Working Party - Costa Rica, ASPT (E) = Average Score Per Taxon (England), ASPT-CR = Average Score Per Taxon - Costa Rica, H = Shannon-Wiener index (H′). Bottom substrates: FS = fine sand and silt, MP = soft mud and plant debris, BST = boulders and stones, CCS = Clay and coarse sand, SS = stones and coarse sand. Water velocity: Slow F = slow flowing water, Fast F = fast flowing water.

2007) between tropical and temperate regions. However, the failure of BMWP-CR and BMWP (E) to separate sites with relatively good water quality from polluted waters limit their utility and validity for River Aturukuku and other riverine ecosystems in Uganda. For example, the allocation by BMWP-CR and BMWP (E), and their associated ASPT scores, of sites such as T3 and T4 (downstream municipal sewage effluent), and R and R2 (in rural settings), mostly to water quality of relatively high (moderate to very good), and low (bad to good) categories, respectively, is unusual in a river, compared to allocations by the Shannon-Wiener diversity index (H′) and physico-chemical variable values.

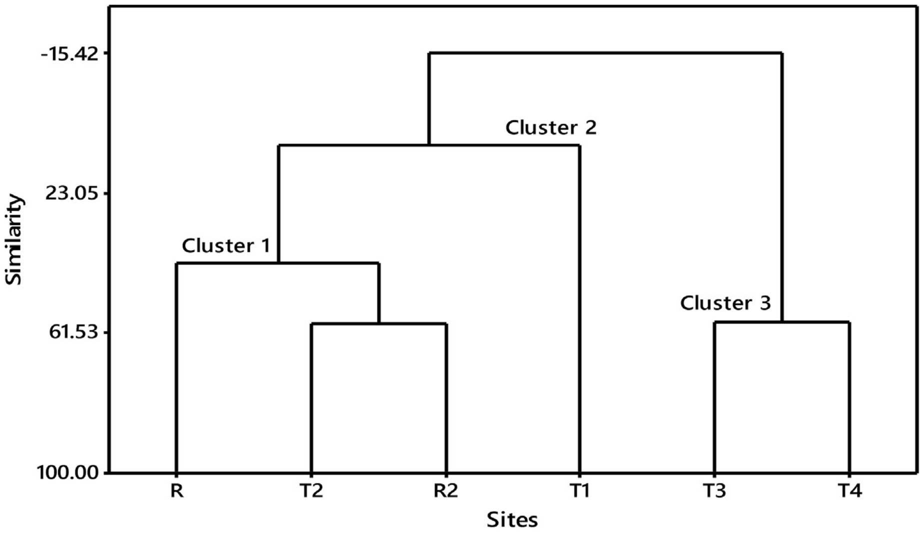

Fig. 3. Classification of study sites with macroinvertebrate species compositions using a hierarchical cluster analysis.

4.2. *Macroinvertebrate community indications*

Higher occurrence of families such as Chironomidae, Oligochaeta and Caenidae, with some species tolerant to highly polluted habitats (Matlou et al., 2017; Aera et al., 2019), and Simuliidae, Baetidae, Libellulidae, Gomphidae and Coenagrionidae, with some species tolerant to moderate pollution (Matlou et al., 2017; Shafie et al., 2017), indicated that the river is under environmental stress. Dominance by Oligochaeta at site T1 (municipal abattoir effluent), is attributable to some species being tolerant of reduced DO in an organically polluted environment (Rosa et al., 2014) and feeding habit on sediment rich in bacteria (Timm, 2012). Predominance by Simuliidae at sites T3 and T4 (downstream sewage effluent) was probably associated with presence of particulate organic matter and diatoms (Ruggiero et al., 2006; Tornés et al., 2018) and fast flowing water that favored their filter-feeding habit (Monaghan et al., 2001; Ocón et al., 2013). Subsistence agriculture and sand mining within the river basin at site R, may have contributed to enrichment of the river bed with fine sand and silt (Koehnken and Rintoul, 2018; Cornejo et al., 2019) and subsequent smothering of interstitial spaces for most sensitive taxa (Canning and Death, 2019), making the tolerant ones such as the Chironomidae dominant (Warkentine and Rachlin, 2015). Absence of taxa dominance in sites T2 and R2 indicated less impact by pollutants (Herman and Nejadhashemi, 2015). The low H' (<1) mostly found at sites R, T1, T3 and T4 on both spatial and seasonal basis, compared to T2 (quite far downstream abattoir point source) and far downstream rural site, R2 ($1<H'<3$), represented high and moderately polluted water, respectively (Wilhm and Dorris, 1968; Welch, 1992). It also suggests differences in negative effects on macroinvertebrates due to point and non-point source pollution (Gonzalo and Camargo, 2013; Gichana et al., 2015; Sharifinia et al., 2016).

4.3. *Physico-chemical variable indications*

Present values of physico-chemical variables such as DO, TP and TN (Table 3), were either close or above thresholds indicated for impairment of freshwater ecosystems (Canning and Death, 2019; IWQGES, 2016), and therefore, likely to pose adverse effects on biota. When DO concentration drops (<5 mg/l) organisms such as macroinvertebrates and fish can become stressed or asphyxiated. Nitrogen and phosphorus are the most common of concern in freshwater ecosystems. In case the TP and TN concentrations in the system increase beyond 0.05 and 0.5 mg/L, respectively, they can trigger environmental impairment such as increased algal and macrophyte growth and associated adverse effects to macroinvertebrates (Canning and Death, 2019). Based on the benchmarks (e.g., Canning and Death, 2019; IWQGES, 2016), the DO values allotted all sites to moderate water quality; TP allotted R and R2 to moderate, and other sites to bad water quality; and TN allotted all sites to bad water quality. Based on strong influence that DO and TP concentrations have on macroinvertebrate species occurrence, compared to very limited one by TN (Friberg et al., 2010), the water quality allocations by the former variables were probably more reliable indication than the latter. Overall, the physico-chemical variables classified the river water quality as bad to moderate, similar to H' for macroinvertebrate assemblages.

Moderate water quality at R2, as indicated by both physico-chemical variables and H', compared to other sites, is attributable to its location at the extreme (7 km) downstream remote area, from influence of pollution in municipality (Fig. 1). It is probable that an upstream-downstream self-purification of pollutants such as NO_3^-N and TSS in water (Ifabiyi, 2008; Kushwaha and Agarhari, 2017), occurred and contributed to the moderate water quality at R2. Inflow of a stream at about 250m upstream of site R2, may have contributed to recruitment of water, organic matter and sediment, thus improving the habitat quality in the river and forming a moderately high diversity spot (Rice et al., 2008), compared to other sites. In addition to remoteness, presence of large strips of natural vegetation along the banks at R2, probably minimized sedimentation and nutrient enrichment from non-point sources, leading to improved water quality and macroinvertebrate diversity (Luke et al., 2018). Relatively higher DO at site R2 than other sites in the upstream (Table 3; Fig. 2) is attributable to cooler water temperature (Rajwa-Kuligiewicz et al., 2015) during the early morning sampling, compared to other sites sampled afterwards in the day. Larger wetted width size at R2, compared to other sites upstream (Table 1), was also a likely factor that increased the complexity, size and stability of habitats for the moderate macroinvertebrate diversity (Vander Vorste et al., 2017). Moderate water quality ($1<H'<3$) at site T2 (urban area), similar to that at site R2 (remote rural downstream area), is attributable to similarities in physical attributes such as fast flowing water and natural vegetation along the banks, and influence from human activities, between the two sites (Table 1). The bad to moderate water quality indicated by H′ and TP at the reference site R in the upstream, compared to the downstream site R2 (moderate), is contrary to good and excellent quality expected at the upstream (Mbaka et al., 2014; Sharifinia et al., 2016; Wan Abdul Ghani et al., 2018). This portrays the increasing anthropogenic impacts on freshwater ecosystems, even in presumed minimally disturbed habitats (Stoddard et al., 2006), such as site R.

4.4. Variable relationships and indications

The macroinvertebrate community and physico-chemical variable indications, to a great extent, are in agreement with the outputs of PCA biplot (Fig. 2), where sites T3 and T4 (downstream sewage effluent) associated with increasing Cond, TP and pH, were separated from T1 (downstream abattoir effluent) with increasing Temp and NO_3^-N, R (rural upstream) with increasing fine sand and silt at river bottom, slowness in water flow and TN, and R2 (rural far downstream) with increasing DO. Sewage and abattoir effluent discharges contribute to increased levels of Cond, TP and NO_3^-N in the receiving waters (Raheem and Morenikeji, 2008; Figueroa-Nieves et al., 2014), such as those point-source areas within urban, compared to rural settings. The similarity clusters of study sites using species composition (Fig. 3) provides almost similar separation of sites based on pollution gradient. The PCA biplot (Fig. 2) and cluster analysis (Fig. 3) demonstrated effects of the studied variables (physico-chemical, nutrients and habitat physical attributes) on macroinvertebrate species composition and/or biotic metrics (Aazami et al., 2015). The significant correlation of PCA axis 1 with BMWP and ASPT indices, Cond, TP, pH and Fast F (Fig. 2), indicate that these were the four primary environmental variables affecting water quality and structuring macroinvertebrate species composition in River Aturukuku. These show that effluents at point sources (e.g., T3) contribute to increase in the nutrient (e.g., TP) concentrations and electrical conductivity of the river water (Rezende et al., 2014), which in addition to physical attributes (e.g., fast flowing water), structured the macroinvertebrate community (Aazami et al., 2015).

4.5. Comparisons with related studies and future prospects

The overall classifications of river water quality by BMWP (E) and BMWP-CR (bad and moderate), and ASPT (E) and ASPT-CR (moderate and very good), corroborate the high correlations between them (Table 7) and indicate similarity in their performance during this study. However, neither the BMWP (E), nor BMWP-CR indices, was effective for assessing and reliably separating less polluted from more polluted sites along the river. Insignificant correlations of H′ with BMWP and ASPT scores (Table 7), is contrary to studies that found positive correlation (Kalyoncu and Zeybek, 2011; Tan and Beh, 2015). Positive correlation between BMWP (E) and BMWP-CR, and TP, is also contrary to findings of Czerniawska-Kusza (2005), confirming limitations of the two biotic indices in assessing water quality of River Aturukuku. Limitations to reliable categorization of water quality along a river gradient using non-indigenous biotic indices have been reported (Kalyoncu et al., 2011; Zeybek et al., 2014; Abbaspour et al., 2017). As suggested (Zeybek et al., 2014; Abbaspour et al., 2017), for BMWP-CR, which included more local macroinvertebrate taxa for scoring water pollution sensitivity to be used in Uganda, there is need to adapt its biotic values to local conditions. In assessing assumptions underlying scoring systems worldwide, Chang et al. (2013) found significant dissimilarities in tolerance values for macroinvertebrates among countries within the same biogeographical regions. Indeed, Spain (Alba-Tercedor, 1996), Brazil (Junqueira and Campos, 1998; Monteiro et al., 2008), Argentina (Capitulo et al., 2001), Thailand (Mustow, 2002), Colombia (Roldán, 2003) and Costa Rica (MINAE, 2007), have adapted and modified the original BMWP (E) index, based on local environmental characteristics.

As performed in a similar study (Mustow, 2002), adaptation of BMWP-CR for application in assessing the Ugandan rivers, requires removal of families that are not present in Uganda and replacing them with the local ones such as Tricorythidae, Thiaridae and Potamonautidae that did not score during the calculations (Table 4). Furthermore, it will require assigning each of these families with pollution tolerance indices, following intensive studies. In Africa, some efforts have been made to develop macroinvertebrate indices. Among these are: SASS (South African Scoring System, Chutter, 1994) based on Biological Monitoring Working Party (BMWP) method developed in UK (Hawkes, 1997), and has been modified to the level of SASS Version 5 (Dickens and Graham, 2002); and The Namibian Scoring System (NASS) version 2 (Palmer and Tailor, 2004) and Tanzania River Scoring System (TARISS, Kaaya et al., 2015), developed based on SASS and SASS Version 5, respectively. Accordingly, these indices were developed through modification of other indices, purposely to allow reliable bio-assessment of water quality in specific ecoregions, by differentiating polluted from minimally polluted (reference) sites.

During their adaptation for application in new environments, adjustments on previous taxa scoring list to accommodate any new local taxa and their pollution tolerance values, was a key aspect. Similarly, Tumusiime et al. (2019) tested the reliability of the TARISS for assessing water quality in Uganda but found it not fully applicable to rivers in Uganda and East Africa at large, without revising and modifying its scoring taxa list. This is attributed to differences in environmental conditions and pollution tolerances among macroinvertebrate taxa at regional (Rodrigues et al., 2016) and even local levels (Buss and Salles, 2006), although universal application of macroinvertebrate indices have been reported Bere and Nyamupingidza, 2014). Relatedly, Borisko et al. (2007) found factors such as stream type or season of sampling, among the most important source of variation in summary macroinvertebrate index values, making interchange of the indices applicable within a localized but not large area.

Among the challenges affecting efforts to develop regional macroinvertebrate indices for Africa, and Uganda in particular, is lack of accessible taxonomical and ecological information describing macroinvertebrate communities at minimally (reference), moderately and highly impacted sites in the region (Elias et al., 2014b). Most rivers have also been severely impacted by human activities, reducing availability of reference sites for developing indices (Kaaya et al., 2015). Furthermore, the macroinvertebrates are identified using mainly guides developed from temperate and other regions despite varying taxa among biogeographical regions, besides limited taxonomic expertise, field, laboratory and museum facilities (Ochieng et al., 2019). To adapt a foreign or develop entirely Uganda-based biotic index, for effective bio-assessment, there is need to develop an accessible database of macroinvertebrate taxonomical, pollution sensitivity scores and ecological information. Benchmarking the methods used in developing the foreign indices currently in use in the region, and conducting training of scientists (e.g., taxonomic skills) in collaboration with foreign experts in invertebrate studies, are crucial. Restoration and conservation of riverine ecosystems will contribute to provision of reference sites for index development. Accordingly, it would be appropriate to either adapt a foreign index such as BMWP-CR to local environmental characteristics or develop entirely Uganda-based biotic index, for effective biomonitoring. We reject the hypothesis that when BMWP (E) and BMWP-CR are used in biomonitoring a tropical riverine ecosystem in eastern Uganda, the latter would be more reliable. Due to lack of indigenous biotic indices for freshwater biomonitoring in Uganda, it is advisable to apply the foreign ones cautiously, whilst including diversity indices (e.g., H′) and selected physico-chemical variables for validation.

5. Conclusions

The results indicate that neither BMWP-CR, adapted for tropical environment, nor BMWP (E) from temperate region, separated sites with relatively good water quality from polluted waters along the river. Although BMWP-CR included more local macroinvertebrate taxa for pollution sensitivity scores than BMWP (E), the performance of both indices was similar. Overall, the BMWP (E) and BMWP-CR classified the river water quality as bad to moderate, while ASPT (E) and ASPT-CR indicated moderate to very good quality. The diversity index (H′) and physico-chemical variables classified river water quality as bad to moderate. The BMWP-CR and BMWP (E), and their associated ASPT scores, allotted water quality at sites such as T3 and T4 (downstream sewage effluent) to relatively high quality (moderate and very good), while R and R2 (in rural settings) to low quality (bad to good), contrary to allocations by H′ and physico-chemical variable values.

Failure of BMWP-CR and BMWP (E) in separating sites based on pollution gradient is attributable to differences in environmental conditions and pollution tolerances among macroinvertebrate taxa at regional, country and local area levels. Correlation between the scores of the two biotic indices, diversity index and selected physico-chemical variables, contrary to those reported elsewhere, could be attributed to these differences. This limits the utility and validity of the two indices for River Aturukuku and other riverine ecosystems in Uganda. In order to apply a biotic index such as BMWP-CR, for biomonitoring riverine ecosystems in Uganda, there is need to adapt its biotic values to the local environmental conditions, through an intensive study. Development of indigenous biotic index would be ideal for the purpose of biomonitoring and management of the country's riverine ecosystems.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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