Ecohydrology & Hydrobiology 17 (2017) 113–124

Contents lists available at ScienceDirect

Ecohydrology & Hydrobiology

journal homepage: www.elsevier.com/locate/ecohyd

Original Research Article

Spatio-temporal macroinvertebrate multi-index of biotic integrity (MMiBI) for a coastal river basin: a case study of River Tana, Kenya

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ARTICLE INFO

Article history:
Received 13 August 2015
Accepted 5 October 2016
Available online 21 October 2016

Keywords:
Spatio-temporal
Macroinvertebrate
MMiBI
Tana River Basin
Coastal ecosystem

ABSTRACT

In the wake of climate change events, spatio-temporal integrated ecosystem models and indices are useful decision support tools for ecosystem management. We developed a preliminary spatio-temporal macroinvertebrate multi-index of biotic integrity (MMiBI) in a major tropical river basin in Kenya. Separation power of Mann–Whitney *U* test ($p < 0.05$) qualified 11 metrics from triplicate macroinvertebrate samples collected seasonally in 33 microhabitats into the scoring system of 1, 3 and 5. Validation and strengthening procedure compared the final MMiBI with selected ($p < 0.05$) physico-chemical parameters and post-MMiBI fieldwork. Seasonal alternating pattern of MMiBI scores suggested that it was highly likely that temporal scores performance might rank lower as compared to spatial significance on ecosystem health delineation. Although a case study in a single river basin is presented, the indexing holistic approach can be of general use for any other coastal river basin as a low-cost biomonitoring tool as a prerequisite towards ecosystem sustainability of water resources.

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1. Introduction

Socio-economic growth is dependent on intensification of human activities such as forestry and urbanization which leads to water resources' degradation (Mereta et al., 2012; Kibena et al., 2014). Yet water resources such as rivers are vital to support biodiversity and provide socio-economic benefits to humans (e.g., Hajkowicz, 2006). Indexing and modelling of aquatic resources using biomonitoring and bioassessment are essential to protect and improve water quality for ecology and hydrology management while responding to anthropogenic stressors (Dickens and Graham, 2002). Bioassessment and biomonitoring is recognized as pertinent aspects of water resources management as a tool for achieving sustainable riverine ecosystems (Ndaruga et al., 2004). Therefore, multi-metric indices that apply a holistic bioassessment approach of water resources for sustainability of ecosystems have become a popular tool for global assessment of aquatic resources (Aura et al., 2010).

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http://dx.doi.org/10.1016/j.ecohyd.2016.10.001
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Prior water quality indices and models have been developed as biomarkers of ecosystem management for water resources sustainability (Masese and McClain, 2012). Globally (e.g., Karr and Chu, 1997, 2000) and at a regional level (e.g., Raburu et al., 2009; Aura et al., 2010; Gonzalo and Camargo, 2013), several models and indices integrating physico-chemical, hydraulic, and ecological water quality have been developed. Index of Biotic Integrity (IBI) stands out to be one of the best indices (Sabater et al., 2004) as it helps to gain a well-rounded perspective of the chemical, physical and biological conditions of a particular water resource. An Index of Biotic Integrity (IBI) is suitable because it satisfies the requirements that an index should be relevant, simple and easily understood by managers, scientifically justifiable, quantitative and acceptable in terms of cost (Aura et al., 2010).

Several global and regional studies on IBI have been developed to assess the ecosystem health of rivers (Kerans and Karr, 1994; Karr and Chu, 1997; Masese et al., 2009a,b; Raburu et al., 2009; Aura et al., 2010), but little information is available on coastal and seasonal aspects of such lotic systems. Due to limited field data, improvement of indices and models that are independent of high costs and climate change scenarios (e.g., studies on seasonal changes) are attracting renewed attention from ecologists, economists and managers to support water management for sustainability of such ecosystems (Smucker et al., 2015). Thus, there is an obvious need of biological indices for monitoring and scoring pollution and other types of degradation of African coastal lotic habitats to form the basis of global discussions on their temporal dynamics.

Little information is known on the holistic approach of water resources sustainability of River Tana Basin in terms of the physical, chemical and biological conditions. Yet, being the longest river in Kenya, it forms an important riverine basin ecosystem for various uses by human and for biodiversity management. Within Tana River Basin, various human activities (e.g., hydroelectric supply and flood control dams, agriculture, forestry, urbanization, industrialization) threaten the water and biological quality. Tana River Basin as a multifunctional ecosystem in Kenya, has several dams (Masinga, Kiambere, Kamburu, Gitaru, and Kindaruma) in the upper section that account for 60% of hydroelectric power supply in the country (Government of Kenya Report, 2007). There is more emphasis to reclaim the projects in Tana River Basin under the National Economic Food Security Project (NEFSP) to improve livelihoods through fish production, horticulture, livestock development among other enterprises. Notably, Tana River Basin is generally dry and prone to drought especially in June–August. Rainfall is erratic, with rainy seasons in March–May and October–December. Conflicts have occurred between farmers and nomadic peoples over access to water. Flooding is also a regular problem, caused by heavy rainfall in upstream areas of the Tana River (Government of Kenya Report, 2007). The Basin supports a fishery whose species diversity and abundance is influenced by the prevailing climatic conditions and long term impacts of human activities (Government of Kenya Report, 2012). Such unique features of Tana River Basin provide the necessity to assess the spatio-temporal status of the resource, especially in the lower sections of the river in order to ascertain the anthropogenic impact on the ecosystem.

Thus, based on the Tana River riparian influences due to the aforementioned anthropogenic activities, hydrological character and water retention of the basin is seasonally altered resulting in destructive flooding with increased precipitation (Government of Kenya Report, 2007). The human activities pose a challenge to the holistic physical, chemical and biological attributes of the basin as the water quality is degraded and quantity reduced during the dry seasons. Amid the inevitable climate change and increasing population, such problems will only exacerbate. In order to protect resources like those in Tana River Basin, there is need to regulate human influences using low-cost and integrated decision-support tools that monitor such changes. One major contribution is the development of new methodologies and tools such as IBI to assess and monitor the ecological integrity of such riverine ecosystems using species-environmental relationships (Karr and Chu, 1997, 2000) such as use of Macroinvertebrate Multi-index of Biotic Integrity (hereafter, MMiBI). This is because bioindicators such as benthic macroinvertebrates are superior to chemical analyses since they are widely employed in monitoring and assessing water quality of most freshwater bodies (e.g., Ndaruga et al., 2004; Mereta et al., 2012; Pace et al., 2012). They are increasingly studied and commonly used as indicators of ecological disturbance since they are long-lived and integrate varied levels and kinds of pollutants accumulated over a long period of time (Raburu et al., 2009), because of their sensitivity to environmental changes and ease of sampling (Morse et al., 2007).

Globally, IBIs that have been developed have little information on the temporal aspect of assessing their holistic performance on water resources and their sustainability. Due to limited resources for research, the lower sections of the basin (i.e. urbanization and forestry) were studied to assess the resultant longitudinal influence of anthropogenic activities on the holistic ecosystem status of the basin using temporal scales. We hypothesized that the MMiBI developed will account for various attributes and microhabitats that are evaluated based on their predictive accuracy and their ability to perform in a temporal scale in the lower Tana River Basin ecosystem scenarios. Therefore, in this study, we developed a spatio-temporal MMiBI for Tana River Basin, Kenya, as a decision-making support tool for river basin management for water resources sustainability.

2. Materials and methods

2.1. Study area

This study was conducted on Tana River Basin (Fig. 1). The 1014 km Tana River is the longest and major river in Kenya, and gives its name to the Tana River County. Its tributaries include the Thika, Sagana and Thuci. The river rises in the Aberdare Mountains and passes through the towns of Garissa, Hola and Garsen before entering the

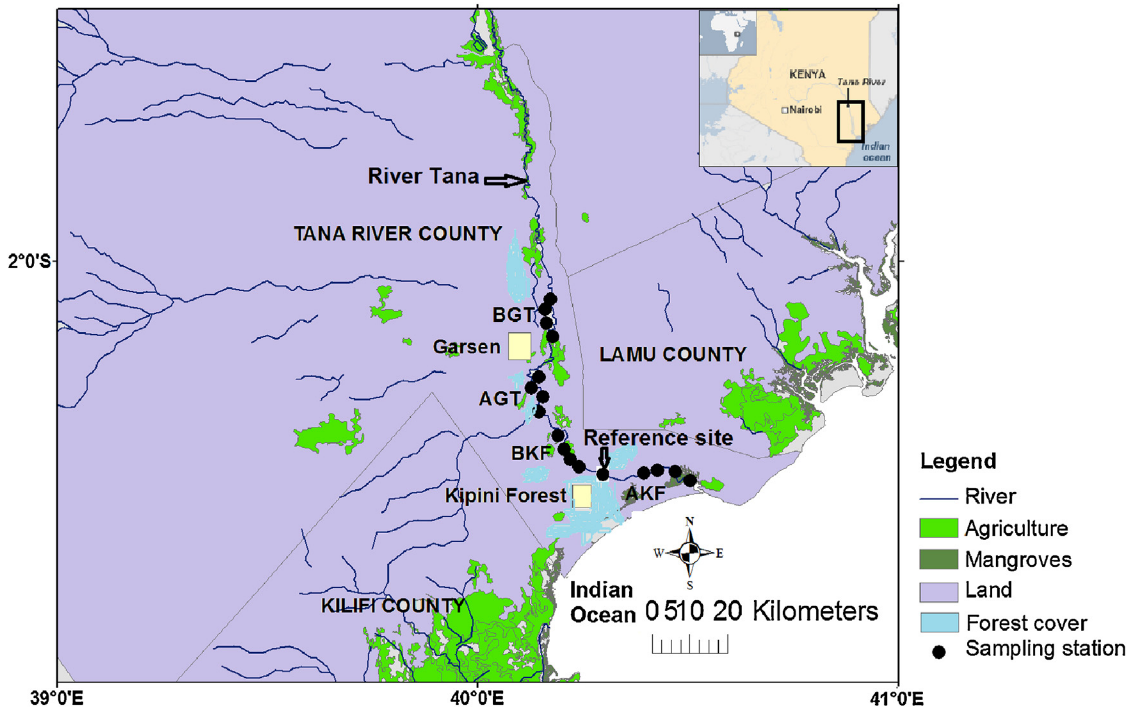

Fig. 1. Location of River Tana Basin, Kenya with Kipini forest and Garsen town that formed the basis of sampling stations. Sampling stations that included Before Garsen Town (BGT), After Garsen Town (AGT), Before Kipini Forest (BKF), Reference site (at Kipini forest), and After Kipini Forest (AKF) consisted of sampling sites (riffles, pools and runs). Land use cover in River Tana Basin consisted of agriculture, bare land, mangroves, forest cover and urbanization.

Indian Ocean at Ungwana Bay. Apart from the three major basins in Kenya, Tana River Basin is the major multifunctional ecosystem with an interesting case of the nexus between conflict and food security. A recent survey found the basin to be 79% food insecure and with an incidence of poverty at 62% (Government of Kenya Report, 2007). Tana River Basin comprises several areas of forest, woodland and grassland which are minor centres for species endemism (IUCN, 2012).

2.2. MMiBI development

Fig. 2 shows a summarized procedure towards the development of a spatio-temporal MMiBI for the lower Tana River Basin. The methodology was divided into four major sections i.e. characteristics of sampling stations, physico-chemical parameters, fauna collection, and data analysis and MMiBI calculations.

2.2.1. Characteristics of sampling stations

The sampling stations in this study were selected to represent urban impacts (near Garsen town) on riverine communities and forested area near the site where River Tana enters the Indian Ocean (Kipini forest). The sampling design was intended to provide a robust test of the MMiBI over a wide range of before and after the human impact intensities that involved three microhabitats (riffles, runs and pools). Microhabitats were sampled in triplicates and were randomly picked to avoid bias due to spatial variations in geological landscape and to give much variations at an approximate distance of 0.2 km. A total of 33 different microhabitats at each trip were sampled. Sampling sites included before Garsen town (BGT), after Garsen town (AGT), before Kipini forest (BKF), at Kipini forest (to assess its validity as a reference site) and after Kipini forest (AKF). Sampling surveys occurred in the rainy season (RS: May, November: 2013–2014) and dry season (DS: July, August: 2013–2014). Environmental data in each site were recorded quantitatively depending on the status of the river bank, environmental conditions of the buffer zone and adjacent land use according to methods suggested by Karaouzas et al. (2011). Variables were recorded aiming to give a detailed display of river and floodplain hydromorphology including local scale characteristics (average stream size, microhabitats type, mean depth, water velocity and discharge, mineral grain size, algal type, inorganic matter, tree litter vegetation type, bank type) and catchment-scale characteristics (land cover at catchment) (Table 1).

2.2.2. Physico-chemical parameters

In every survey in both dry and rainy seasons, triplicate physico-chemical parameters from each microhabitat were measured at each site. In each season, a total 66 replicate samples for physico-chemical parameters were collected and categorized into microhabitat types per station.

Conductivity was measured in situ before 1000 h using a conductivity metre (OAKTONR, Model WD-35607-10, Singapore), while temperature and pH were also measured

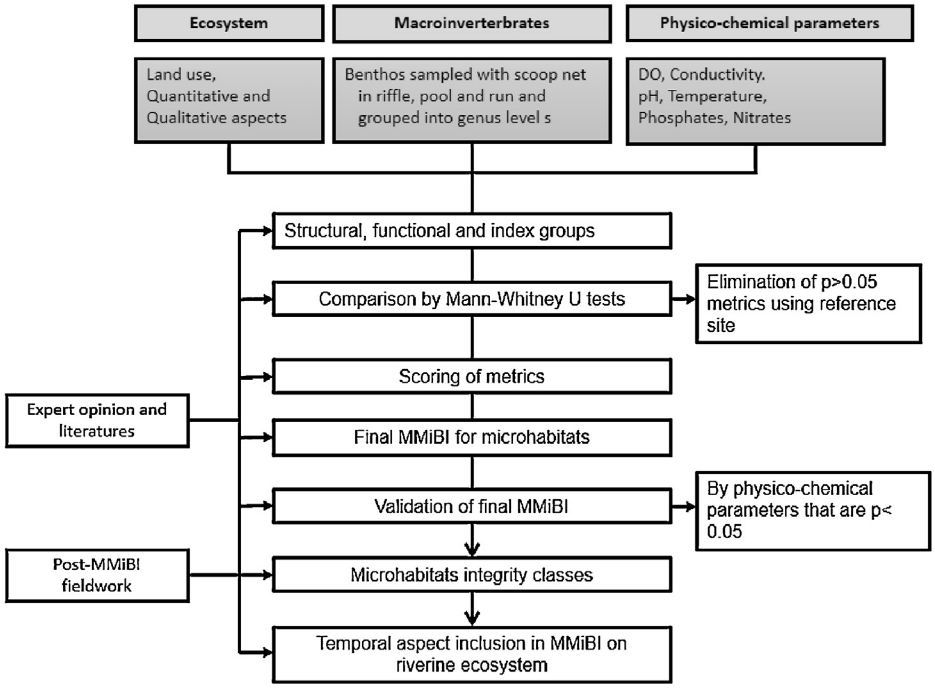

Fig. 2. Schematic representation towards the development of a Macroinvertebrate Multi-index of Biotic Integrity (MMiBI) for the lower Tana River Basin, Kenya.

in situ before 1000 h by a combined pH-and-temperature-metre, (OAKTONR, Model pH/Mv/°C METER, Singapore). The Winkler method was used to determine dissolved oxygen (DO) concentrations (APHA, 2000) using water samples from sites where macroinvertebrates were to be sampled. Water samples for total nitrogen (TN) and total phosphorous (TP) were collected from each site and analyzed according to Wetzel and Likens (2000). In order to get a representative water quality measurement, the three replicate samples of physico-parameters from each microhabitat per site were averaged per station (either before or after Garsen town or Kipini forest) and season (either dry or rainy season).

2.2.3. *Fauna collection*

A triplicate number of benthic macroinvertebrate samples from each microhabitat were sampled using a scoop net (1 m^2 covered bottom, with a 0.5 mm mesh size). In each season, a total 66 replicates for macroinvertebrate samples were collected and categorized into microhabitat types per station.

Macroinvertebrate samples from all the microhabitats per sampling site were hand-sorted and preserved in 70% alcohol, and kept separately according to microhabitat type and season. They were then identified using a microscope to genus level according to Merritt and Cummins (1997). The use of Merritt and Cummins (1997) from United States (US) keys was on the suppositions that US determination keys have previously been applied in the region (e.g., Raburu et al., 2009; Aura et al., 2010); and that the course taxonomic resolution still gives a good basis for the discrimination ability of the MMiBI. Mathooko (1998) and existing local literature (e.g., Ndaruga et al., 2004; Kibichii et al., 2007) for other taxa were used to further identify the macroinvertebrates and verify their African existence.

2.2.4. *Data analysis and MMiBI calculations*

Macroinvertebrates mean (±SE) relative abundance and dominance were analyzed based on methods suggested by Herrmann (1999) and Aura et al. (2010). The classifications into functional feeding groups (FFG), richness, composition and tolerance was based upon the metrics previously used in riverine ecosystems around the world (e.g., Kerans and Karr, 1994; Barbour et al., 1999), and those that have been recommended for African riverine ecosystems (Richards et al., 1997; Masese and McClain, 2012). Metrics were further adjusted following interpretations of community responses to different types of stressors in the region (Ndaruga et al., 2004; Kibichii et al., 2007; Kasangaki et al., 2008). The number of each macroinvertebrate taxon in each replicate sample was quantitatively converted into density, and relative percentage per microhabitat type.

Variations in the uncertainty of water quality as a function of the site categories and sampling dates were examined using a non-parametric Kruskal–Wallis ANOVA with the sites and dates as the main factors. This is because data were not normally distributed and attempts to normalize the data by transformations were unsuccessful. With no monthly or annual variations ($p > 0.05$) in physico-chemical parameters or macroinvertebrate abundance, both variables were grouped into seasons to account for the strong influence of dry and rainy seasons in the region (Government of Kenya Report, 2007). The physico-chemical parameters that showed spatio-temporal significant effects ($p < 0.05$) were related with MMiBI

Table 1
Spatio-temporal mean (±SE) values of physico-chemical variables, and environmental data in River Tana, Kenya.

Stations/seasons	Garsen town				Kipini forest				Kruskal–Wallis test (χ^2; p values)	
	Before town (BGT)		After town (AGT)		Before forest (BKF)		After forest (AKF)			
Parameters	RS	DS	RS	DS	RS	DS	RS	DS	Stations	Seasons
Temperature (°C)	28.1 ± 0.0	29.1 ± 0.0	27.0 ± 0.0	29.0 ± 0.0	26.2 ± 0.0	27.2 ± 0.0	26.7 ± 0.0	28.3 ± 0.0	5.12; 0.06	4.13; 0.08
pH	6.8 ± 0.1	6.9 ± 0.7	6.8 ± 0.3	6.9 ± 0.5	6.7 ± 0.2	6.9 ± 0.1	6.8 ± 0.5	7.0 ± 0.3	7.33; 0.12	5.01; 0.10
Conductivity ($\mu S\,cm^{-1}$)	128.0 ± 1	134.0 ± 4	147.0 ± 5	164.0 ± 4	115.0 ± 2	110.0 ± 45	135.0 ± 4	130.8 ± 8	3.11; 0.03*	2.14; 0.04*
Oxygen ($mg\,L^{-1}$)	5.0 ± 0.02	4.6 ± 0.07	3.5 ± 0.0	3.1 ± 0.01	4.4 ± 0.03	4.8 ± 0.02	5.8 ± 0.03	4.6 ± 0.01	3.17; 0.03*	2.93; 0.02*
TP ($mg\,L^{-1}$)	0.16 ± 0.27	0.14 ± 0.31	0.49 ± 0.20	0.29 ± 0.10	0.35 ± 0.02	0.13 ± 0.02	0.30 ± 0.10	0.11 ± 0.23	6.15; 0.14	5.85; 0.09
TN ($mg\,L^{-1}$)	0.31 ± 0.14	0.47 ± 0.06	0.52 ± 0.1	0.61 ± 0.14	0.45 ± 0.0	0.70 ± 0.01	0.18 ± 0.04	0.20 ± 0.03	1.17; 0.02*	3.32; 0.04*
Average stream width (m)	4.31 ± 0.25		5.1 ± 0.40		3.3 ± 0.20		3.2 ± 0.10		5.52; 0.13	–
Mean depth (m)	0.41 ± 0.15		0.44 ± 0.18		0.52 ± 0.24		0.37 ± 0.24		3.41; 0.04*	–
Discharge ($L\,s^{-1}$)	23.4 ± 7.13	19.4 ± 4.09	29.6 ± 9.8	26.6 ± 6.9	31.8 ± 9.8	22.8 ± 4.7	20.6 ± 5.3	16.6 ± 6.2	8.22; 0.09	2.13; 0.02*
Mean water velocity ($m\,s^{-1}$)	0.22 ± 0.11	0.18 ± 0.10	0.17 ± 0.21	0.14 ± 0.19	0.34 ± 0.12	0.25 ± 0.18	0.10 ± 0.01	0.08 ± 0.03	3.26; 0.03*	2.19; 0.04*
Inorganic matter	Gravel		Sandy		Silt		Mud			
Organic matter	Very coarse		Coarse		Fine		Fine			
Mineral grain size (mm)	2–4		0.125–0.5		0.015–0.03		0.004–0.032			
Main tree litre vegetation type	Fallen leaves		Detritus		Detritus		Detritus			
Algal type	Absent		*Hydrodictyon* sp.		Sparse blue-green		Sparse *Euglena*			
Bank type	Slightly eroded		Highly eroded		Eroded		Slightly eroded			
Riparian area/Land use type	Scarce bushes/Agriculture/ Bare land		Agriculture Bare land/Grassland		Bushes/Grassland/ Agriculture		Bare land/Scarce mangroves cover			

* Refers to significant p level of Kruskal–Wallis ANOVA, $p < 0.05$.

Table 2
Results of Mann–Whitney *U* tests for the metrics discrimination using microhabitats of impaired sites [before Garsen town (BGT), after Garsen town (AGT), before Kipini forest (BKF) and after Kipini forest (AKF)] with microhabitats of reference site (at Kipini forest) in River Tana, Kenya during the rainy (RS: May, November) and dry (DS: July, August) seasons ($p < 0.05$ indicated by +for more than two cases of microhabitat pair-wise comparison).

Metrics	Rainy season	Dry season
Taxa richness		
Number Ephemeroptera taxa	<0.01+	0.01+
Number Hemiptera taxa	0.001+	0.01+
Number Diptera taxa	0.03+	0.02+
Number Decapoda taxa	0.11	0.12
Taxa composition		
Shannon diversity index	0.03+	0.03+
Whittaker index	0.02+	0.04+
% EPT: Diptera	0.21	0.11
% Gastropoda	0.10	0.08
% Hemiptera	0.02+	0.01+
% Odonata	0.42	0.25
Taxa tolerance		
% Tolerant taxa	0.03+	0.04+
% Dominant taxon	0.02+	0.01+
% 5 Dominant taxa	0.25	0.10
Trophic functions		
% Filterers	0.01+	0.02+
% Gatherers	0.02+	0.01+
% Predators	0.21	0.10
% Shredders	0.03+	0.04+

scores to validate and strengthen the final MMiBI which may indicate better performance of the index to organic pollution (Aura et al., 2010; Ofenboeck et al., 2010; Aschalew and Moog, 2015). Data analyses were carried out using STATISTICA version 8.0.

The macroinvertebrate community composition in each microhabitat per site was considered as metrics (Table 2). The metrics selected based on literature and expert opinions acted as indicator attributes in assessing the status of macroinvertebrate assemblages in response to perturbation along a gradient of human disturbance or environmental condition change (Mason, 2002). The development of MMiBI followed the methods suggested by Raburu et al. (2009) and Aura et al. (2010) but with modifications based on the local conditions and to accommodate the temporal pattern of the index. Modifications in this case were the interpretations of community responses to different types of stressors in the region that were mainly due to dry and rainy seasons, land use and pollution.

Temporal MMiBI was developed since the majority of the physico-chemical parameters varied significantly ($p < 0.05$) at a seasonal level. Additionally, there was need to develop a preliminary temporal MMiBI which could be improved in the future studies in order to accommodate climate change aspects. Shannon–Wiener diversity index (H′) to $\log_{10}$ (Dajoz, 2000) was included in the model as a biomarker of macroinvertebrate biodiversity and calculated per pooled station and microhabitat in each season. The beta diversity index was applied in order to evaluate the taxonomic similarity between stations' communities. Herein, we considered pairs of microhabitats and sites on which we applied Whittaker index (β_w) (Whittaker, 1972), calculated as: $\beta_w = (\mathrm{Sr}/\alpha \text{ mean}) - 1$, where Sr is the total richness in each pooled microhabitat or site and α mean the mean richness of both pooled microhabitats or sites in each season. Furthermore, we evaluated the ability of attributes to separate each microhabitat of impaired site (BGT, AGT, BKF, and AKF) from reference site (at Kipini forest) using Mann–Whitney *U* tests. This is because the non-parametric test is used to assess the uncertainty of intermediate (or impaired) classification difference with the reference classification. Reference site was chosen depending on the status of the river bank, environmental conditions of the buffer zone and adjacent land use in relation to the impaired sites (Raburu et al., 2009). Potential metrics for MMiBI scoring were identified when the tests showed significant differences ($p < 0.05$ in more than two cases of microhabitat pair-wise comparison) between site groups per season (Table 2).

A similar scoring system or criteria employed by Raburu et al. (2009) and Aura et al. (2010) of 1, 3 and 5 with the thresholds of median-ranges for each metric of 25th and 75th percentiles based on the reference site (at Kipini forest) was used, which has been commonly used in macroinvertebrate IBIs. Whereby, for each metric expected to decrease with degradation, values below the 25th percentile were scored as 1. Values between the 25th and 75th percentiles were scored as 3, and values above the 75th percentile were scored as 5 (Table 3).

In order to arrive at the final spatio-temporal MMiBI value for each sampling site, the scores for each metric were summed (Table 3). The highest expected value of 50 points served as a benchmark for the qualitative assessment using a suggested four-class scheme based on the distribution of MMiBI scores under the multi-metric river quality classes (Table 4). The maximum value of 50 points was used as the threshold based on the stressor–response relationships (Stevenson et al., 2004; Paulsen et al., 2008) of MMiBI scores. Whereas different approaches are used to group sites into condition category classes (e.g., excellent, good, fair, poor), the study used levels based on consensus from scientists using the riparian and riverine status observed to come out with a scenario to be desired by the public or management authority. Since there were no excellent integrity class status as observed during fieldwork, a slightly higher threshold range of >43 points was agreed that appeared to highly deviate from all the MMiBI final values. The lowest threshold range of <28 was awarded since a few microhabitats appeared to fall within such a category. The middle ranges were based on the higher (>43) and lower (<28) threshold integrity class ranges with an equal class size of six, but without losing their actual description as per the riverine ecosystem status. In order to validate the integrity classes awarded, a post-MMiBI fieldwork was conducted in order to verify the categories and descriptions awarded with the actual riverine status. The validation and strengthening of the final MMiBI scores was done using physico-chemical variables that only showed significant relationships ($p < 0.05$) in both sites and seasons since they were assumed to indicate river quality deterioration (Aschalew and Moog, 2015).

Table 3
Metrics used and scoring criteria (system) for the development of MMiBI in the forestry and urban ecosystems of Tana River Basin during the rainy (May, November) and dry (July, August) seasons (MMiBI, Macroinvertebrate Multi-index of Biotic Integrity; RS, Rainy Season; DS, Dry Season; Ri, Riffle; Ru, Run; P, Pool; maximum = 50 points). Riffles were absent at AKF. Sampling at (during) Kipini forest was used as reference site.

Sites/Seasons	Garsen town												Kipini forest										Scoring criteria		
	Before town (BGT)						After town (AGT)						Before forest (BKF)						After forest (AKF)						
	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS	RS	DS			
Metric for MMIBI	Ri	Ri	Ru	Ru	P	P	Ri	Ri	Ru	Ru	P	P	Ri	Ri	Ru	Ru	P	P	Ru	Ru	P	P	5	3	1
# Ephemeroptera taxa	5	5	5	5	3	3	5	3	1	1	1	1	1	1	1	1	3	1	5	3	3	3	16–10	10–5	<5
# Hemiptera taxa	3	3	5	3	3	1	1	1	3	1	3	1	3	3	3	5	1	1	5	5	3	1	24–13	13–7	<7
# Diptera taxa	3	3	1	5	5	5	1	3	3	1	1	3	3	5	3	3	5	5	1	3	5	3	11–9	9–4	4–1
Shannon diversity index (H′)	3	3	3	5	3	1	3	1	3	3	5	3	3	3	3	5	1	1	5	5	3	1	>3.05	3.05–2.7	<2.7
Whittaker index (β_w)	1	3	3	5	3	5	3	5	3	5	5	5	3	5	3	5	3	5	3	5	5	5	<0.40	0.40–0.60	>0.6
% Hemiptera	3	3	5	3	3	1	1	1	3	1	3	1	3	3	3	3	1	1	5	5	3	1	>10	5–10	<5
% Tolerant taxa	3	3	1	5	5	5	1	3	3	1	1	3	3	3	3	3	5	5	1	3	5	3	<30	30–40	>40
% Dominant taxon	3	5	3	3	1	1	5	3	5	3	5	5	5	3	3	3	5	5	3	5	1	5	>15	6–15	<6
% Filterers	3	1	3	3	5	5	1	1	3	3	1	1	3	3	1	3	1	1	5	3	5	5	>20	8–20	<8
% Gatherers	5	5	5	3	3	3	3	3	5	3	1	1	3	3	5	3	1	1	5	3	1	1	>25	13–25	<13
% Shredders	3	3	1	1	3	3	1	3	1	3	1	1	3	3	3	1	1	3	1	1	3	3	>14	8–14	<8
Total MMIBI Score	**35**	**37**	**35**	**41**	**37**	**33**	**25**	**27**	**33**	**25**	**27**	**25**	**33**	**35**	**31**	**35**	**27**	**29**	**39**	**41**	**37**	**31**			
Averaged MMIBI	**RS = 35.67; DS = 37**						**RS = 27.33; DS = 25.67**						**RS = 30.33; DS = 33**						**RS = 38; DS = 36**						

Bold values represent summation of IBI scores.

Table 4
Suggested threshold values of riverine ecosystem integrity classes for final Macroinvertebrate Multi-Index of Biotic Integrity (MMiBI) development showing the classification level and ranges for Tana River Basin, Kenya during the study period.

Integrity class	Description	Ranges
1: Excellent	High quality and clear water (can see the bottom based on turbidity); low level of riparian degradation.	>43
2: Good	Good water quality; slight riparian degradation	36–42
3: Moderate	Moderate water quality; significant riparian degradation	29–35
4: Poor	Poor water quality; major/heavy riparian degradation	<28

3. Results

3.1. Physico-chemical and environmental data

The mean (±SE) values of the physico-chemical parameters as well as qualitative abiotic variables measured during the survey are shown in Table 1. Conductivity, DO and TN showed spatio-temporal variations (Kruskal–Wallis ANOVA; $p < 0.05$). Similar to TN, the highest mean (±SE) conductivity values were recorded at AGT (DS: 164.0 ± 4 $\mu S\,cm^{-1}$; RS: 147.0 ± 5 $\mu S\,cm^{-1}$) but with the lowest mean (±SE) DO levels, followed by AKF in the rainy season (135.0 ± 4 $\mu S\,cm^{-1}$). There were marked variations (Kruskal–Wallis ANOVA; $p < 0.05$) in mean (±SE) depth and mean (±SE) water velocities (Table 1). Organic and inorganic matter, mineral grain size, algal type and riparian vegetation showed gradual differences downstream towards the ocean.

3.2. Macroinvertebrate assemblages

A total of 11 orders, 20 families and 22 genera were sampled. The orders Ephemeroptera, Hemiptera and Diptera were the most diverse taxa, consisting of three families each (Appendix A). The highest taxonomic richness was recorded at AKF runs (27 taxa) while the lowest at AGT (17 taxa). Generally, the order Diptera was the most abundant at BKF (35%) and at BGT (29%). The intolerant group of Ephemeroptera, Plecoptera and Trichoptera (EPT) constituted 17% and 19% at BKF and BGT, respectively. The order Hemiptera dominated AKF (23%) and Oligochaeta was the most common at AGT (24%).

Furthermore, the EPT group (Ephemeroptera, Plecoptera and Trichoptera) dominated riffles and runs, whereas, tolerant taxa (e.g., Diptera, Oligochaeta) and less tolerant taxa (e.g., Hirudinea, Gastropoda) dominated the pools. For example, *Baetis* sp. (Ephemeroptera) was dominant in riffles at BGT and AGT with a relative mean (±SE) abundance of 24.10 ± 0.11 and 19.43 ± 0.09, respectively. Whereas, *Chironomus* sp. (Diptera) had the lowest relative mean abundance of 0.33 ± 0.01 and 0.20 ± 0.01 in runs at BGT and AGT, respectively. *Lumbricus* sp. (Oligochaeta) only dominated runs with the highest relative mean (±SE) values of 18.14 ± 0.09 and 23.73 ± 0.94 at BKF and AGT, respectively. *Polycentropus* sp., the only Trichoptera recorded in the study, dominated the riffles at BKF with a relative mean (±SE) abundance of 34.01 ± 0.41. *Velia* sp. (Hemiptera) was the most common in all the riffles and runs in all the sampled sites.

Lumbricus sp. (Oligochaeta) had the highest relative mean (±SE) abundance of 27.78 ± 0.95 in pools at BKF and 17.18 ± 0.85 at AGT. *Sphaerium* sp. (Gastropoda) had the highest relative mean (±SE) abundance value of 23.28 ± 0.06 in pools at AKF. Pools at BGT were dominated by *Chironomus* sp. with a mean (±SE) relative abundance of 26.1 ± 0.12, followed by *Tubifex* sp. (Oligochaeta) with a relative mean (±SE) abundance of 16.56 ± 0.07.

The low values of Whittaker index (β_w) were observed for the BKF-BGT pairs (RS: 0.40) and BKF-AGT (DS: 0.38) while highest β_w values were obtained with BKF-BGT (DS: 0.57), and BKF-AKF (RS: 0.58). AKF had the highest Shannon–Wiener diversity index (H′) (RS and DS: 3.09), followed by BKF (RS and DS: 3.06), whereas, in the dry season (DS) at BKF and AGT recorded the lowest Shannon–Wiener diversity of 2.68 and 2.54, respectively.

3.3. Macroinvertebrate multi-index of biotic integrity (MMiBI)

A total of 17 macroinvertebrate metrics that were longitudinally represented at least by more than one individual in more than two-thirds of the samples were used (Table 2). Of the 17 metrics that were selected, 11 of them differed significantly ($p < 0.05$) between sampling sites and seasons and thus they were assumed to have discrimination among them. These metrics were then used to create a MMiBI score for each microhabitat and an average MMiBI per site and season (Table 3). Those metrics that were statistically similar ($p > 0.05$) between sites and seasons (i.e. number of decapoda taxa, % EPT: diptera, % mollusca, % odonata, % 5 dominant taxa, and % predators-carnivores-scavangers, which engulf or pierce the prey) were not used in the scoring of the final MMiBI. Dissimilar functional feeding groups' metrics between sites included % filterers, % gatherers and % shredders.

Table 3 shows the calculated final MMiBI scores of Tana River Basin, Kenya. In the final MMiBI, AKF in the rainy season emerged with the highest average MMiBI (38.00 points) with good riverine ecosystem quality (Table 4). While AGT recorded the lowest MMiBI (25.67 points) with poor riverine quality and major riparian degradation. Thus, MMiBI was in the order of AKF, BGT, BKF and AGT, from the highest to the lowest, respectively. The highest MMiBI (AKF and BGT: 41) were also recorded in the runs. Additionally, alternating high and low performance of MMiBI scores occurred based on seasons. The validation and strengthening of the final MMiBI scores showed dependence ($R^2 > 0.50$; $p < 0.05$) with DO ($R^2 = 0.67$; $p < 0.05$; RS), and conductivity ($R^2 = 0.67$; $p < 0.05$; RS and DS) (Fig. 3). Significant but weak relationship and wide data variability of MMiBI scores with TN ($R^2 < 0.50$; $p < 0.05$; RS and DS) was noted.

4. Discussion

Globally, freshwater ecosystems are among the most threatened habitat types. The ecosystems greatest challenges over the coming decades will be biodiversity loss, climate change and water shortages (Dudgeon, 2010). In developing nations, the problem of inadequate fresh water supply, complicated by ever-increasing demands, is already being experienced. But developing water resources in less industrialized nations without degrading such ecosystems is a challenging but prudent goal (McClain, 2013), considering that a large proportion of rural populations depend directly or indirectly on the ecological goods and services provided by rivers and river corridors. In Africa in particular, unsustainable land use, land cover changes and environmental impacts have manifested themselves in form of drought, flooding and reduced baseflow in rivers (Elisa et al., 2010), deforestation and increased agriculture (Mati et al., 2008), erosion and sedimentation of rivers (Okungu and Opango, 2005). Such events and activities pose a challenge to rivers as riverine quality is degraded which affects the ecohydrology pattern, and quantity changes at seasonal levels due to changes in climate.

A major contribution to the streams and rivers management and conservation has been an improved understanding of biota–environmental relationships, and development of new frameworks and indices to assess and monitor their ecological integrity for sustainable ecosystem health. This is because aquatic communities such as macroinvertebrates have emerged as good indicators in assessing effects of different levels of anthropogenic influence (Aura et al., 2010). This has been achieved through multi-dimensional assessment of physical, biological, chemical and ecological responses along a gradient of human disturbances, identification of indicator biota, driving variables acting on river ecosystems, and improved statistical methodologies to detect effects across different riverine ecosystems (Karr and Chu, 1997; Barbour et al., 1999). Such factors have made bioassessment and biomonitoring to be recognized globally as pertinent aspects of water resources and management. Thus, in order to protect and conserve water resources for stream management, there is need to develop low-cost and climate change independent decision support tools that monitor changes that occur for water resource managers in the long term. At the moment, MMiBI herein forms a

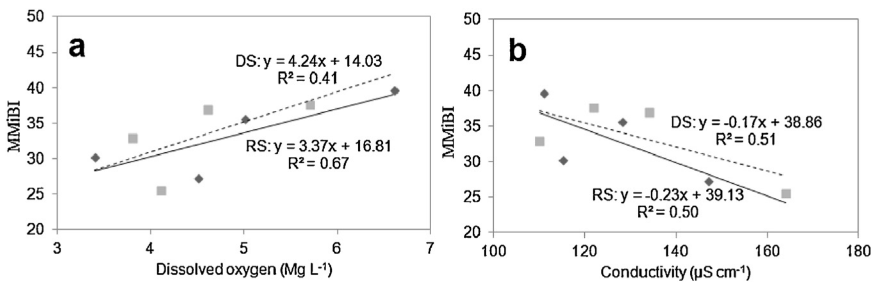

Fig. 3. Relationship plots (at $p < 0.05$) for validation and strengthening of the MMiBI scores with the final MMiBI values against (a) Dissolved Oxygen and, (b) conductivity for Tana River Basin during the rainy (RS: continuous line) and dry (DS: dotted line) seasons.

support tool that could be adopted and adjusted in the future for stream management in coastal areas due to seasonal changes for sustainable holistic approach of ecosystem conservation.

The study was based on the lower Tana River Basin that may capture the resultant effects of the river usage as well as for urban and forested sections. More adjustments for the MMiBI may be required in the future to compare the upper sections of the river with the coastal lower sections. In the MMiBI developed, only those taxa that were considered as tolerant or intolerant by consensus of most researchers and experts were designated as tolerant and intolerant in this study (Karr and Chu, 1997; Kasangaki et al., 2008). Although, there is still a huge debate on the taxa tolerance of macroinvertebrate to pollution and anthropogenic influence (Aura et al., 2010). Taxa that were considered to be intolerant to perturbation included the EPT group (i.e. Ephemeroptera, Plecoptera and Trichoptera) (Herrmann, 1999). Plecoptera and Trichoptera were excluded in the metric discrimination (Table 2) since both taxa did not show significant relationships ($p < 0.05$) after pair-wise comparison with reference site using Mann–Whitney U tests. Although, Plecoptera and Trichoptera have been found to be important in many riverine sites (Karr and Chu, 1997).

Almost similar proportional percentages of predator individuals were observed at reference sites in relation to the impaired sites which could have disqualified the % predators application in the final MMiBI (Table 2). Processing of organic material and turnover of detritivores in the forested regions must have been substantial to support the predator populations. Gatherers are generalists that thrive in depositional zones with abundant fine particulate organic matter. Changes in riparian land use and decrease in sedimentation at degraded sites away from the Indian Ocean might have influenced the domination of the shredders more than gatherers, in turn increasing the shredders population (Cheshire et al., 2005). However, the accuracy of functional feeding groups and their response to riparian degradation could have been biased since we used information from literature as there is limited published information on the macroinvertebrate functional feeding groups in the region.

Minimal riparian degradation at AKF was evident due to the highest average MMiBI (38.00 points: good riverine ecosystem quality) recorded. A high MMiBI is usually coupled with good water quality parameters, either high abundance or diversity of intolerant macroinvertebrates, and slight riparian degradation (Masese et al., 2009a; Aura et al., 2010). But for poor riparian ecosystems, intolerant species disappear in the early stages of degradation due increased nutrient levels, highly eroded banks, and decreased dissolved oxygen (Griffith et al., 2005) that was common at AGT. On the other hand, relatively high conductivity levels at AKF were associated with the presence of scarce mangrove cover and brackish soils in the area which acted as a buffer and trapping zone of ions from the nearby zones (Alongi and Christoffersen, 1992). Alongi and Christoffersen (1992) further mentioned that mangroves presence also favours high macroinvertebrate diversity hence the highest MMiBI in AKF recorded. Nevertheless, sections of the estuary in the tidal zone (in this case, near station AKF) are undoubtedly exposed to periodic backflows of saline water (chloride concentration or general salinity). This must have a bearing on the qualitative structure of the macroinvertebrate fauna and so on the results concerning water quality in station AKF in the present study. We recommend further investigation on salinity variation and other estuarine characteristics on the MMiBI developed that were not assessed and thus not included in the results due to minimal resources. The highest MMiBI (AKF and BGT: 41) were also recorded in the runs. This might be because riffles and runs have been considered to possess higher taxa richness of better water quality, and diverse habitat variation than the pools (Herrmann, 1999).

Those considered tolerant included most genera belonging to Diptera, Oligochaeta, Gastropoda and Hirudinea, % tolerant taxa, and % 5 dominant taxa in each site. Such taxa form composition attributes of relative abundance and dominance to provide information on the make-up of the assemblage by assessing the relative contribution of the macroinvertebrates to the total fauna. In this respect, the percentage 5 dominant genera in every order are commonly used as a measure of dominance and evenness (Masese et al., 2009a,b). Tolerant group at BKF and AGT was abundant, as well as low MMiBI score of 25.67 points (poor riverine ecosystem quality and major riparian degradation) recorded. Dipterans seemed to have high relative abundances at BKF and AGT that were associated with agricultural activities and eroded river banks in both stations, as well as urbanization in the latter. Urbanization (Ndaruga et al., 2004) and agriculture (Kasangaki et al., 2008) of a watershed have been found to significantly alter stream water quality even in the absence of direct industrial or municipal discharges (Kari and Rauno, 1993). Similarly, BGT and BKF had eroded river banks and the existence of agricultural activities which might have accounted for lower EPT levels recorded.

Variations in Shannon–Wiener (H′) and Whittaker indices (β_w) values obtained could be due to variations in physico-chemical parameters related to sampling stations that could have been influenced by longitudinal land use activities. For example, high conductivity levels in AGT may have influenced the osmoregulation of the aquatic invertebrates leading to sensitive freshwater organisms either to adopt or are phased out (Spiels and Mitsch, 2000). Lower Shannon–Wierner diversity and Whittaker indices levels at BGT and AGT than other stations were associated with agricultural activities, eroded banks, and the open access for livestock invasion that was evidenced in the sites. Other than agricultural activities and eroded banks (e.g., Kasangaki et al., 2008), the herbivory of aquatic vegetation and nutrient input via urine, faecal deposition and trampling of sediments which was a common phenomenon in these areas, could have had direct impact on the river as observed by Griffith et al. (2005) in the study that was done on southern Rocky Mountain streams. Thus, MMiBI was in the order of AKF, BGT, BKF and AGT, from the highest to the lowest, respectively which closely corresponded with the coupling of physico-chemical parameters, macroinvertebrate attributes, and riparian characteristics of the riverine ecosystem (Tables 1 and 3).

Seasonal alternating performance of MMiBI scores suggested that it was highly likely that either rainy or dry seasons may be ranked lower as compared to spatial influence on delineation of ecosystem health. Thus MMiBI developed might firstly be influenced by longitudinal occurrence of human activities i.e. forestry and urbanization before the temporal influences are considered. But with the index suitability description and scores, such as those recorded in this study, any slight differences in MMiBI scores in either dry or rainy seasons herein, suggested a large variation in ecosystem status on a temporal scale. For example, rains and floods have been known to increase the flow velocity as well as water volumes in rivers as opposed to periods of drought (IFM, 2006) which might have changed the microhabitat status that were sampled in this study. As similarly noted by Smucker et al. (2015), flow velocity in this study was mainly influenced by rainy season, water discharge, land cover type and mean depth of the stations that consisted of microhabitats (Table 1). The current study advocates for further studies to be conducted on the temporal aspect of indices, especially in areas with pronounced seasonality aspects such as rainy and dry periods recorded in this study.

The results from functional analysis showed that a significant relationship ($R^2 > 0.50$; $p < 0.05$) was observed between MMiBI scores and most physico-chemical parameters responsible for structuring benthic macroinvertebrate communities. MMiBI showed dependency ($R^2 > 0.50$; $p < 0.05$) with DO ($R^2 = 0.67$; $p < 0.05$; RS), and conductivity ($R^2 = 0.67$; $p < 0.05$; RS and DS) which may indicate better performance of the index to organic pollution under varying seasons (Fig. 3). Significant but weak relationship of MMiBI scores with TN ($R^2 < 0.50$; $p < 0.05$; RS and DS) indicated that total nitrogen may be a limiting factor from the organic pollution in Tana River Basin ecosystem.

5. Conclusions

The results of this study revealed the spatio-temporal structural occurrence of macroinvertebrates in relation to physico-chemical parameters that are affected by longitudinal anthropogenic influence in a tropical and coastal river basin. The MMiBI developed provided a degree of quantification of the longitudinal anthropogenic impacts which can be applied on a holistic approach of water resource management based on ecohydrology concept by monitoring ecological conditions of coastal riverine basins in developing nations. The spatio-temporal index developed could also be used as baseline information for future projections of climate change impacts on riverine ecosystems in areas with pronounced seasonality changes. Studies on the index and model predictions based on the existence of hydroelectric dams in the upper riverine sections as well as inclusion of estuarine characteristics such as salinity concentrations could improve the index further for coastal water resource management.

Conflict of interest

No conflict of interest.

Ethical statement

We are purely trained researchers with the information herein only meant for research purposes and therefore we account our output based on research and information dissemination. Notably, the work herein has not been previously published or submitted to another journal, but only to this journal.

Acknowledgements

Sincere thanks to all who gave their time to participate in the study, to our technicians Joseph Kilonzi and Mary Mukono. The authors thank the Kenya Marine and Fisheries Research Institute Librarian, Elijah Mokaya for additional literature search.

Funding

We thank International Foundation of Science (IFS) for funding the project. Kenya Marine and Fisheries Research Institute provided logistics and facilitated the research activities.

Appendix A

Summarized taxanomic list of benthic macroinvertebrates (based on current literature) found at the sampling stations in River Tana, Kenya. x, means present; BGT, before Garsen town; AGT, after Garsen town; BKF, before Kipini forest; and AKF, after Kipini forest.

Order	Family	Genus	BGT	AGT	BKF	AKF
Ephemeroptera	Baetidae	*Baetis* sp.	x	x	x	x
	Caenidae	*Caenis* sp.	x		x	x
	Heptagenidae	*Heptagenia* sp.	x	x	x	x
Hemiptera	Gerridae	*Gerris* sp.	x	x		x
	Veliidae	*Velia* sp.	x	x	x	x
	Nepididae	*Nepus* sp.	x		x	x
Coleoptera	Elmidae	*Elmis* sp.	x	x	x	
	Hydraenidae	*Hydraena* sp.	x		x	

Appendix A (*Continued*)

Order	Family	Genus	BGT	AGT	BKF	AKF
Diptera	Chironomidae	*Chironomus* sp.	x	x	x	x
	Culicidae	*Culicida* sp.	x	x	x	x
	Tipulades	*Limonia sp.*			x	x
Odonata	Agriidae	*Agrion* sp.	x		x	
Oligochaeta	Lumbriculidae	*Lumbricus* sp.	x	x	x	x
	Tubificidae	*Tubifex* sp.	x	x	x	x
Plecoptera	Nemouridae	*Nemoura* sp.	x	x	x	
	Leuctridae	*Leuctra* sp.	x		x	
Gastropoda	Unionidae	*Pisidium* sp.	x	x		x
		Sphaerium sp.	x	x	x	x
Bivalvia	Lymnaidae	*Lymnaea* sp.	x	x	x	
Trichoptera	Polycentropodidae	*Polycentropus* sp.	x	x	x	x
Hirudinea	Erpobdellidae	*Erpobdella* sp.	x	x		x
		Glossiphonia sp.	x	x	x	x

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