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Review article

Biomonitoring as a prerequisite for sustainable water resources: a review of current status, opportunities and challenges to scaling up in East Africa

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ABSTRACT

Degradation of aquatic ecosystems in the Lake Victoria basin (LVB) and the rest of East Africa has elicited concern because of its bearing on social and economic development. Rapid population growth, industrialization and its associated urbanization, agricultural intensification and habitat loss have increased pressure on the integrity of water resources. Costs associated with traditional approaches to monitoring water quality have become prohibitive while not giving reliable early warning signals on resource condition to aquatic resource managers. The purpose of this paper is to explore approaches to developing macroinvertebrate- and fish-based biomonitoring tools in the LVB and East Africa and the challenges they face through a review of studies that have been carried out in the region. The hypothesis is that aquatic biota in the LVB provides cost-effective and integrative measures of the physical and chemical habitat conditions thus necessitating their use in assessment and monitoring of water resources. In the LVB macroinvertebrate and fish based indices of biotic integrity (IBIs) have demonstrated their utility in identifying sources of impairment, determining the extent of impacts and stand to give natural resource managers a scientifically defensible rationale for developing guidelines for conservation and management. Despite this significant step, however, adoption and use of indices as part of regular monitoring programs are yet to be realized. We recommend for the advancement and adoption of biological criteria as an integrated approach to monitoring human-induced stress in riverine ecosystems of the East Africa region.

Contents

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1. Introduction

Freshwater ecosystems are among the most threatened habitat types in the world and three of the world's greatest challenges over the coming decades will be biodiversity loss, climate change and water shortages (Dudgeon, 2010). In water stress countries, the problem of inadequate fresh water supply, complicated by ever-increasing demands, is already being experienced. In Africa, development initiatives have been identified that will increase demands for freshwater. Africa Water Vision 2025 outlines development targets that will include a doubling of the area of irrigated agriculture and a five-fold increase in overall water use for agriculture, industry and hydropower (UN-Water, 2003). Developing Africa's water resources without degrading ecosystems is a challenging but prudent goal (McClain et al., 2013), considering that a large proportion of rural populations depend directly on the ecological goods and services provided by rivers and river corridors – water, fish, other food sources (e.g. molluscs, crabs and vegetables) and fibre that contribute to meeting nutritional needs and livelihoods (Darwall et al., 2011).

In East Africa, unsustainable land-use and land-cover changes being witnessed (FAO, 2010) present another set of environmental challenges. At local levels, environmental impacts have manifested themselves in form of frequent droughts, intense flooding, receding of lake levels and reduced baseflow conditions in streams and rivers (Kadomura, 2005; Ligdi et al., 2010; Elisa et al., 2010; Obiero et al., 2012). In the LVB, the most pressing environmental concern is degradation of, not only the lake and consequent loss of water quality and biological diversity, but also degradation of streams and rivers draining into the lake (Ntiba et al., 2001; Njiru et al., 2008; Odada et al., 2009; Masese and McClain, 2012). Increased intensity of agriculture and deforestation have been linked to increasing magnitude and frequency of runoff events (Mutie, 2006; Mati et al., 2008), pesticide contamination (Osano et al., 2003), reduced baseflow (Elisa et al., 2010; LVBC, WWF-ESARPO, 2010), erosion and sedimentation of streams and rivers (Okungu and Opango, 2005) and nutrient loading into the lake (Scheren et al., 2000; Nyenje et al., 2010). Because of riparian habitat loss to deforestation and horticulture, river bank degradation by livestock and sand mining (Masese et al., 2009a,b; Raburu et al., 2009a,b; Raburu and Masese, 2012), and drainage of wetlands (Njuguna, 1996; Bavor and Waters, 2008), the hydrological character and water retention of many streams and rivers have been altered resulting in massive and destructive flooding during spates (Kadomura, 2003; UNEP, 2003). These activities pose a challenge to rivers that drain altered catchments as water quality is degraded and quantity reduced during the dry months. With the inevitable challenge of climate change amid a rapidly increasing human population, the problems will only be exacerbated. To protect these water resources, human influences should be regulated and water resource managers need to develop decision-support tools that monitor changes that occur.

A major contribution to the management and conservation of freshwater resources has been an improved understanding of species–environmental relationships, and development of new methodologies and frameworks to assess and monitor the ecological integrity of streams and rivers (Karr, 1981; Karr et al., 1986; Karr and Chu, 2000). It is now established that aquatic communities are good indicators used to assess the effects of different levels of human impact. This has been achieved through careful analysis of biological and ecological responses along gradients of human disturbance, the identification of indicator assemblages and species among assemblages with known responses to human alterations, identification of driving variables (both natural and human-induced) acting on aquatic ecosystems, and improved statistical techniques and approaches to detect effects across different types of aquatic ecosystems and regions (Karr, 1981; Karr and Chu, 2000; Whittier et al., 2007a,b; Stoddard et al., 2008). As a result, bioassessment and biomonitoring is now recognized as pertinent aspects of water resources management and conservation. Particular attributes of fish and macroinvertebrate communities exhibit clear responses to human disturbance and have been identified as useful bioindicators of ecological condition in streams and rivers (Karr et al., 1986;

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Rosenberg and Resh, 1993; Barbour et al., 1999). Multimetric indices are increasingly used to guide conservation actions because they yield policy relevant information for regulatory agencies and decision makers (Karr and Chu, 1999). Therefore, they have become a popular tool for regional assessment of aquatic resources in Europe (Hering et al., 2006) and the United States (USEPA, 2013; Stoddard et al., 2008). While the development and incorporation of biomonitoring tools into regular water resources management and conservation programs in other parts of the world is advancing, most African countries (except South Africa) are lagging behind.

Certainly, effective management of riverine ecosystems in the LVB and the rest of East Africa require up-to-date data on the various land-use and human activities, their impacts on aquatic resources and innovative and cost-effective means of monitoring changes that occur. The objective of this paper is to review current monitoring approaches being employed to monitor human perturbations on water resources in the LVB and to present the efforts that have been made to use bioindicators, mainly macroinvertebrates and fish, to monitor water resources in the basin. The potential for scaling up the use of bioindicators in the wider East Africa is also evaluated as an opportunity, the corresponding challenges highlighted and the way forward presented.

2. Biological monitoring framework and its rationale

A number of interrelated physical, chemical, and biological factors affect the ecological integrity of riverine ecosystems. These factors can be grouped into classes of chemical and physical water quality, flow regime, habitat structure, biotic interactions and energy sources (Karr, 1991; Karr and Chu, 2000). In addition to these, physical barriers to migratory corridors and alien species threaten ecological integrity of riverine ecosystems in the LVB. Monitoring actions traditionally focused on one aspect of ecological integrity, chemical and physical water quality, with regulatory efforts aimed at controlling individual parameters (Roux, 1997). However, this approach was only useful in controlling pollution coming from point sources. As a result, environmental conditions of aquatic ecosystems continued to decline worldwide (Karr and Chu, 2000). Altered water flow regimes in dammed streams and rivers; pollution from non-point sources such as cities, farms, and feedlots; destruction of habitats above and alongside rivers by development or logging; and invasions by alien species have all presented challenges. This continual degradation of surface waters, coupled with the failure of physical and chemical water variables to provide information on the overall condition of aquatic ecosystems (Roux, 1997; Dickens and Graham, 1998), prompted a shift of monitoring and management from a regulatory approach to a more integrated and holistic ecosystem approach (Cairns, 2003). This has resulted in monitoring which increasingly concentrates on the overall response of an ecosystem to all kinds of stressors. Biological assessment and monitoring has, thus, become an important tool in evaluating water quality and ecosystem integrity of water resources around the world (Karr and Chu, 2000; Moog and Chovanec, 2000) and the United States Environmental Protection Agency (USEPA) has been a leader in developing guidelines and the actual implementation (Barbour et al., 1999; USEPA, 2013).

Biological assessment is the evaluation of the biological condition of an ecosystem, based on surveys of the structural and functional organization of the community of resident biota (Karr et al., 1986; Barbour et al., 1999; USEPA, 2013). On the other hand biomonitoring is the periodic sampling of biota of a site or habitat such as a stream, river, lake or wetland (Barbour et al., 1999; USEPA, 2013). Bioindicators are superior to chemical analyses because long-lived aquatic organisms integrate varied levels and kinds of pollutants accumulated over long periods of time, and also respond to the combined action, whether additive, synergistic, or antagonistic. Within the same framework, macroinvertebrate- and fish-based indices of biotic integrity have been widely adopted and applied in riverine ecosystems where they have been useful in providing information on levels and sources of degradation and developing biological criteria for stream protection and restoration (Barbour et al., 1999, 2000; Weigel et al., 2002; Klemm et al., 2003).

2.1. The index of biotic integrity (IBI)

The index of biotic integrity (IBI) is a holistic multimetric approach that involves the integration of a number of structural and functional attributes or metrics of a community into a composite index. The IBI, also called a multimetric index (MMI) is defined as any index that is based on the sum or ratings of several different attributes, termed metrics, with the rating of each metric based on quantitative expectations of what comprises high biotic integrity (Simon and Lyons, 1995). A metric is defined as a calculated term or enumeration representing some aspect of biological assemblage structure, function or other measurable characteristic that changes in a predictable way with human influence (Barbour et al., 1995).

The development and use of IBI dates back to 1981 following its use in the assessment of the biological integrity of warmwater streams in Midwestern United States using fish communities (Karr, 1981). It was designed to quantify characteristics of stream fish assemblages to assess biotic integrity, which is defined as the "capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region (Frey, 1977; Karr and Dudley, 1981). In its original form, the IBI became popular and was employed by many investigators (Karr and Chu, 1999). Since that time different versions of IBIs have been developed for different regions and ecosystems and have become standard tools for assessment of stream condition, particularly to address aquatic life uses and conservation (Davis and Simon, 1995; Guaro and Gubiani, 2013). Further modification saw its wide use outside the United States and in different habitats (for reviews see Miller et al., 1988; Simon and Lyons, 1995; Hughes and Oberdorff, 1999; Roset et al., 2007; Guaro and Gubiani, 2013). In Africa fish-based IBIs have been used in

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West Africa (Hugueny et al., 1996), Central Africa (Toham and Teugels, 1999), Southern Africa (Hocutt et al., 1994; Kleynhans et al., 1999) and in East Africa (Raburu and Masese, 2012). Because of its popularity, all continents and regions, except Antarctica, have used the index for bioassessment and biomonitoring purposes.

2.2. Use of different taxa in biomonitoring

While initially developed using fish assemblages (Karr, 1981; Karr et al., 1986), modifications and adoptions has seen development of IBIs based on different aquatic organisms, including bacteria, protozoans, diatoms, algae, macrophytes, macroinvertebrates and amphibians (Barbour et al., 1999; Rothrock et al., 2008; Lane and Brown, 2007; Kane et al., 2009). Depending on disturbance characteristics (intensity, frequency, duration) each group may respond at different rates and provide different information. Therefore, the choice of which particular assemblage to use will depend on the objective of the study and the type of ecosystem (river or stream, wetland, lake, reservoir, estuary) to be investigated. Of the many biological indicators available, the most widely used groups in lotic systems have been macroinvertebrates and fish. The reasons for the popularity of macroinvertebrates in current bioassessment practices (Rygg, 1988; Rosenberg and Resh, 1993; Metcalfe-Smith, 1994) include :

- their sedentary nature which allows determination of the spatial extent of an impact;
- they are comparatively long-lived than other aquatic organisms such as bacteria, protozoans, diatoms and algae and this allows temporal changes in abundance and age structure to be followed;
- they are abundant and diverse in most streams hence their large number of species produces a range of responses;
- they integrate conditions temporally, so like any biotic group, they provide evidence of conditions over long periods of time;
- their ease of identification to family level;
- sampling is easy, requires few people, and
- equipment is relatively inexpensive.

Fish is the second group of biological indicators that is quite popular in monitoring programs. A number of authors (Karr, 1981; Karr et al., 1986; Fausch et al., 1990; Simon and Lyons, 1995) have highlighted the advantages and disadvantages of using fishes as indicators of ecosystem health. Some of the advantages of using fishes as indicator organisms include:

- fishes are typically present in most aquatic systems, with the exception of highly polluted waters;
- fishes are relatively easy to identify;
- most samples can be processed in the field with the fishes being returned to the water;
- there is extensive life-history and environmental response information available for most fish taxa;
- fishes may exhibit obvious external anatomical pathology due to chemical pollutants;
- fishes often exhibit physiological, morphological, or behavioural responses to stresses;
- being mobile, sensitive fish species may avoid stressful environments, leading to measurable population patterns reflecting that stress;
- because fishes often range considerable distances, they have the ability to integrate diverse aspects of relatively large-scale habitats;
- as fishes are comparatively long lived, they provide a long-term record of environmental stress;
- fish communities include species from a variety of trophic levels (e.g. detritivores, herbivores, planktivores, benthic invertebrate feeders and piscivores) and thus reflect effects at all levels within the food web;
- fishes contain many functional guilds and are also able to integrate adverse effects on other components of the ecosystem, e.g. habitat disturbance in the river catchment; and
- because of recreational, subsistence and commercial fishing, as well as concerns regarding fish production and safety for human consumption, the public relates more directly to information about fish communities than to other aquatic biota.

3. Legal and policy framework basis for biomonitoring and assessment

The USA Clean Water Act and the European Union Water Framework Directive form the foundation for the large amount of biomonitoring and bioassessment in those nations. Similar legislation exists for South Africa (National Water Act), South Korea (National Stream Health Monitoring Act), Canada (Canada Water Act), New Zealand (Resource Management Act), and Australia (Sustainable Rivers Audit). In Kenya, the current legal framework addressing water resource management is the Environmental Management and Co-ordination Act (EMCA), 1999; the Water Act, 2002 and the Environmental Management and Co-ordination (Water Quality) Regulations, 2006. Part V article 42 of EMCA addresses protection of rivers, lakes and wetlands while Article 50 and 51 details the manner in which the National Environmental Management Authority (NEMA) shall ensure the conservation of biological diversity and prescribe adequate measures to ensure the conservation of biological resources in situ in Kenya. Even though the EMCA does not expressly mention biomonitoring, it provides an opportunity for its incorporation by emphasizing consultations with relevant lead agencies such as universities and research institutions. The Water Act on the other hand provides for the management, conservation, use and control of water resources and for the acquisition and regulation of rights to use water; to provide for the regulation and management of water supply and sewerage services. However, it does not have an express provision for aquatic habitat biomonitoring and bioassessment in particular. A third legislative document is the Environmental Management and Co-ordination (Water Quality) Regulations, 2006. These regulations apply to drinking water, water used for industrial purposes, water used for agricultural purposes, recreational uses, fisheries and wildlife, and water used for any other purposes. Part II clause 6 of the regulations provides for the protection of lakes, rivers, streams, springs, wells and other water sources. Even though this regulations recognize that resource quality in relation to a water resource, means the quality of all the aspects of a water resource including the physical, chemical and biological characteristics of the water, it does not go further to identify such cost-effective biomonitoring and assessment tools necessary to achieving these.

The rest of East African countries have recently reformed water laws and associated regulations and recognize water for basic human needs and environmental protection as the highest priorities (Uganda Law Reform Commission, 2000; GoT, 2009). While these water laws and regulations appreciate the need for the protection of water resources, there are no provisions for their bioassessment and biomonitoring. However, the laws and regulations provide for the protection of various components of water resources that constitute ecological integrity; maintenance of 'the reserve', which is the natural flow regime in streams and rivers, physico-chemical and biological attributes, in-stream and riparian conditions and the condition of aquatic biota (Water Act, 2002). In this review, we affirm that biomonitoring is the way forward and is a prerequisite for sustainable water resources management in East Africa.

4. Water quality monitoring in the Lake Victoria basin

Monitoring of water quality in the LVB has focused for a very long time on chemical and physical water quality variables with regulatory efforts aimed at controlling them below stipulated threshold levels (Table 1). Chemical analysis as an alternative approach to determining ecosystem health, however, faces a major drawback when addressing non-point sources of pollution. The method is also limited because it cannot provide information on the overall condition of aquatic ecosystems without associated biological data (Dickens, Graham, 1998). In rivers and their associated wetlands, chemical analysis is more limited considering the extreme variability of physico-chemical variables over small temporal scales that always occur. If chemical sampling is to be used to determine river condition, trends caused by human intervention would have to be separated from natural signals that show vast spatial and temporal fluctuation. Because of this limitation, the water quality situation in most rivers in the basin has been deteriorating over the years. This has been worsened by the rising human population and changes in land-use and land-cover (Odada et al., 2009).

In concert with the realization of the importance of utilizing biota in monitoring programs, a shift has been witnessed towards the use of aquatic communities as indicators of ecosystem integrity (Table 2). Most of these studies have been very useful in laying the basis for developing indices to guide monitoring practices in the basin. In LVB, Kenya, efforts to develop biomonitoring tools for aquatic resources have yielded promising results. Macroinvertebrate-based indices of biotic integrity have been developed for a number of rivers and streams in the upper reaches (Masese et al., 2009a; Raburu et al., 2009a; Aura et al., 2010) and in the lower reaches (Kobingi et al., 2009). Studies on entire riverine ecosystems include Raburu (2003) and Raburu et al. (2009b) based on macroinvertebrates and Raburu and Masese (2012) based on fish. Omukoto (2007) developed a fish-based IBI for the satellite lakes. The indices were meant to achieve at least one of the following objectives: (1) detect and characterize the ambient condition of water resources in the systems, (2) define spatial conditions in water and ecological conditions, and (3) identify thresholds for system stressors, that is, how much the systems have shifted from their natural pristine states or how much the systems can be disturbed without causing unacceptable changes to water quality or impairment of beneficial uses. In their various forms, the IBIs have demonstrated that they can delineate different forms of degradation, be it habitat loss on the riparian areas to animals and sand mining, industrial discharge of wastewater, domestic wastes, urban pollution or deforestation at the catchment level.

Table 1
Studies in riverine ecosystems in the Lake Victoria basin that have focused on various aspects of water quality, macroinvertebrates, fish and ecosystem function.

Group of organisms	Aspects studied	References
Fish	Biology and ecology	Okedi (1971), Balirwa (1979), Balirwa and Bugenyi (1980), Lowe-McConnell (1987), Welcomme (1988) and Manyala and Ochumba (1990)
	Effect of environmental factors on fishes	Balirwa and Bugenyi (1980) and Raburu (2003)
	Distribution and production	Ochumba and Manyala (1992)
Macroinvertebrates	Distribution and abundance, including influences of land use	Raburu (2003), Masese et al. (2009b) and Minaya et al. (2013)
Habitat quality	Instream and floodplain conditions	Raburu (2003) and Masese et al. (2009a)
Water quality	Physico-chemical parameters (TSS, BOD, DO, pH, conductivity, etc.)	Raburu (2003), Okungu and Opango (2005) and Masese and McClain (2012)
	Pesticides	Osano et al., 2003
	Heavy metals	Mwamburi (2003) and Oyoo-Okoth et al. (2010)
Functioning of streams and rivers	Energy sources and trophic relationships	Ojwang et al. (2007) and Masese and McClain (2012)

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Table 2
Mean (range) for selected water quality parameters at reference/unperturbed, moderately impaired and impaired condition categories in the Lake Victoria basin.

Physico-chemical parameters	Condition category		
	Reference/unperturbed	Moderately impaired	Impaired
Temperature (°C)	22.1 (20.2–23.7)	22.6 (20.2–24.1)	23.7 (22.8–24.4)
Dissolved oxygen (mg/l)	7.4 (6.3–8.6)	6.7 (3.6–6.8)	3.6 (2.4–4.4)
Conductivity (μS/cm)	75.2 (34.3–98.5)	134.8 (56–388.9)	258.8 (232.5–278.3)
pH	7.1 (6.4–7.6)	7 (6.3–7.8)	7 (5.9–7.9)
Turbidity (NTUs)	71.7 (42.1–98.2)	127.2 (49.7–241)	266 (152.6–372.4)
Total hardness (mg/l)	39.7 (26.3–59)	68.4 (26.3–159.2)	127.7 (101.5–157.3)
Total alkalinity (mg/l)	54.9 (26.1–90.1)	76.6 (24.9–190	144.1 (13.02–156.4)
Total dissolved solids (mg/l)	49.4 (31.8–76.2)	82.7 (27.5–151.9)	124.4 (110.1–136.7)
Total suspended solids (mg/l)	16.5 (12–23.6)	73.6 (62–80.1)	–
Total nitrogen (mg/l)	0.6 (0.4–1)	0.7 (0.6–0.9)	0.7 (0.6–1.2)
Total phosphorous (mg/l)	0.4 (0.3–0.8)	0.8 (0.6–1)	1.6 (1.4–2.1)

However, development of biological criteria for assessment and monitoring of aquatic ecosystems in the Lake Victoria basin face a major drawback caused by the scarcity of published biological community data from streams and rivers (Table 1). In particular, there are few data from relatively unimpaired sites and little historical information available to quantify how various attributes of biological communities in streams and rivers have been affected by environmental degradation. Development of an index of biotic integrity based on fish for aquatic ecosystems in the basin is also made difficult by several other factors:

- lack of ecological and life-history information about most fish species, such that their functional role in the biotic community is often uncertain;
- most low order streams in the basin are species poor, which limits the range and sensitivity of structural and compositional metrics that can be developed;
- an apparent tolerance of extreme environmental conditions displayed by many fish species, wide distribution in wetland areas that sometimes become anoxic during the dry periods, makes it hard to identify indicator taxa;
- most of the fish species are potamondromous, living in the lake and making periodic runs into the rivers to spawn, and yet some of the same species maintain permanent communities in the streams and rivers. This makes it difficult to clearly discern their breeding ecology;
- high degree of omnivory and shifts in food items consumed by a given species displayed by many fish species, often determined by the habitat where the fish occur, and variations within these habitats makes assignations to trophic groups difficult; and
- probable species extinctions in the rivers and streams caused by introduced species (as has happened for cichlids and other lacustrine species, makes it difficult to determine the diversity expected in natural habitats).

Despite these challenges, and as demonstrated elsewhere (e.g., Lyons et al., 1995; Ganasan and Hughes, 1998), available information meet the minimum threshold for developing IBIs for the region; knowledge of which species are native and exotic, their trophic and habitat guilds, and their relative tolerance to environmental degradation. Thus, applications of IBIs in different parts of the world has seen metrics being modified, deleted or new ones added to reflect regional differences in assemblage characteristics (Ganasan and Hughes, 1998; Hughes and Oberdorff, 1999; Roset et al., 2007).

4.1. Developing IBIs for the Lake Victoria basin

4.1.1. Assemblage characteristics

The diversity of benthic macroinvertebrate assemblages in riverine ecosystems in the Lake Victoria basin share similar attributes (Raburu, 2003; Masese et al., 2009b; Raburu et al., 2009a,b; Aura et al., 2010). Thirteen orders are common among the different rivers, and major differences that occur are attributed to human influences at the local level (Raburu et al., 2009a; Minaya et al., 2013). By composition, Ephemeroptera, Coleoptera, Diptera, Hemiptera and Trichoptera have high representation of families and genera. Taxa that display limited distribution and abundance include Lepidoptera, Megaloptera, and Collembola. The Ephemeroptera + Pleocoptera + Trichoptera (EPT) are considered to be the most intolerant to pollution in the region (Raburu, 2003; Ndaruga et al., 2004; Kibichii et al., 2007; Kasangaki et al., 2008). However, Plecoptera is only represented by one genera (*Neoperla spio*) while some families like Hydropsychidae (Trichoptera), Baetidae and Caenidae (Ephemeroptera) have been shown to be tolerant to organic pollution (Kibichii et al., 2007; Kasangaki et al., 2008; Masese et al., 2009a,b). Order Diptera is the most diverse and the many species respond differently to pollution, with the most tolerant (e.g.,) and Q6 the most sensitive (e.g., Athericidae and Rhagionidae) represented in the group (Masese et al., 2009a). Family Chironomidae (Diptera) has been widely used in bioassessment because of the many species that are represented in many trophic levels enables the group to respond to different sources and types of pollution (Wright and Burgin, 2009; Odume and Muller, 2011; Marchiori et al., 2012).

Fish diversity in the Lake Victoria was originally high. However, massive biodiversity loss has been reported in the lake (e.g., Ogutu-Ohwayo, 1990) but not much information is available about the condition in the influent rivers. Another problem is that few studies are available in the literature documenting fish species distributions in different rivers and their levels of endemism. However, given the high degree of potamodromy that has been reported for most non-cichlid species in the basin (Lowe-McConnell, 1987; Ochumba and Manyala, 1992), it is assumed that rivers in the basin share similar assemblage characteristics. In general, family Cyprinidae dominates riverine samples in terms of number of species and abundance. More species occur in the lower reaches of influent rivers, along lake margins and at the river-mouth wetlands (Balirwa and Bugenyi, 1980; Ochumba and Manyala, 1992; Gichuki et al., 2001). Six exotic species have been recorded in the rivers: *Oreochromis niloticus*, *O. leucostictus*, *Tilapia zillii*, *Lates niloticus*, *Rastreonobola argentea* and *Protopterus salmoides* (Ochumba and Manyala, 1992; Raburu, 2003; Raburu and Masese, 2012)

4.2. Classification of streams and rivers

Streams and rivers should be classified by virtue of their shared characteristics. This is to enable comparisons to be made such that during the evaluation of ecological conditions differences do not arise from factors outside anthropogenic causes. Classification minimizes natural influences on metric responses while at the same time maximizes variability to different human influences. Classification frameworks can be based on geography, e.g., ecoregions (Omernik, 1987), continuous variables or a combination. Continuous natural variables that have been found to be more useful for developing predictive IBI models include mean catchment area, elevation, annual runoff, mean summer and winter air temperature, channel slope, and geology (Pont et al., 2006, 2009; Moya et al., 2011). In the LVB, streams and rivers occur within the same ecoregion where they share similar climatic conditions, as evidenced by the distinct rainfall regime in East Africa as a whole (Rodhe and Virji, 1976). The amount of rainfall varies spatially as a result of changes in relief features. Mountainous areas generally receive more rainfall while low-lying areas in the floodplains of many rivers and at the lakeshore receive lower than average rainfall. However, because of longitudinal connections in terms of energy flow and assemblage characteristics, given that many of the non-cichlid species are migratory (Ochumba and Manyala, 1992) upstream–downstream comparisons of ecological conditions are feasible making it possible to develop monitoring indices for entire river basins.

Natural variation in the distribution of macroinvertebrates and fish assemblages is given consideration in the classification of streams and rivers for evaluating metric responses. In this regard, differences in stream size, defined by catchment area or stream order (Strahler, 1957) are commonly used. However, stream order has been criticized because it is a poor predictor of stream size (Hughes and Omernik 1983; Hughes et al., 2011). Even though stream orders have been used previously as predictors of stream size in the LVB, we recommend that future studies also consider using catchment size and other continuous natural variables such as altitude, distance from source, reach slope, wetted width, presence/absence of a natural lake upstream, geological type and flow regime (Pont et al., 2006). In the Nyando River basin, ≥4th order streams recorded more number of macroinvertebrate taxa than those below 4th order (Raburu, 2003; Raburu et al., 2009b). For fish assemblages, most first order streams in the forested upper reaches have no fish. In some cases one or two species of *Barbus* or/and *Clarias* have been recorded (Raburu, 2003; Raburu and Masese, 2012; Masese and McClain, 2012). In other cases natural barriers, like waterfalls, have been found to prevent upstream movement of fish with sites upstream recording a limited numbers of species. For example, upstream of Odino Falls in the Nyando River and Tenwek Falls in the Nyangores River–Mara River occur one species of genus *Clarias*; *Clarias theodorae* and *C. liocephalus*, respectively.

4.3. Selection of reference conditions

Establishment of reference conditions is the most critical issue during development of the index of biotic integrity (Davis and Simon, 1995). Reference sites act as benchmarks against which other sites are compared to determine the degree of their impairment (Stoddard et al., 2006; Herlihy et al., 2008). Reference conditions also act as a measure of the success of interventions (elimination of stressors, reintroductions or restoration). However, the definition of a reference condition means has different interpretations to differed people. It can be used to mean historical condition, least-disturbed condition, minimally disturbed condition or best attainable condition (Stoddard et al., 2006). For the sake of this review, we adopt the definition by Reynoldson et al. (1997) that a reference condition is a condition that is representative of a group of minimally impaired or 'least-disturbed' sites organized by selected physical, chemical and biological characteristics. This definition recognizes that completely undisturbed sites are virtually nonexistent and even remote waters are impacted by factors such as atmospheric pollution (Roux, 1997). In some cases, streams can be identified that have experienced a minimal degree of human influence and are said to be in a minimally disturbed reference condition (Stoddard et al., 2006). However, these locations are rare because of widespread human influence, the ones available are sometimes located in inaccessible areas, and often are not representative of entire river networks or ecoregions. Sometimes it is necessary to reconstruct reference conditions where none exixts. In this regard, two approaches have been:

- use of literature and expert opinion or local knowledge to reconstruct conditions in terms of habitat and water quality conditions expected in least-disturbed sites; however, this is difficult because most parts of Africa lack historical data and expert opinion is always subjective and sometimes lacking;
- data is usually collected on water quality and habitat characteristics across a gradient of human influence to detect biological responses to changes in environmental conditions; the *posteriori* approach (Barbour et al., 1999; Whittier et al., 2007a; Herlihy et al., 2008). The reference conditions are then selected based on the best values observed.

The two approaches have been used, sometimes in combination, during IBI development in the LVB. Sampling across a gradient of human influence to select best observed conditions or metric values for use as reference condition requires that sites are selected to ensure that streams and rivers representing the full gradient of human disturbance, different communities and hydrogeomorphic classes are adequately represented in the data set (Karr and Chu, 1999). This is followed by sampling for water and habitat quality, macroinvertebrate and/or fish. In addition, previous published works and those in the grey literature have been used to establish trends in water quality (Table 2), assemblage distributions and their responses to changes in environmental conditions. For the fish IBI, local knowledge has also been utilized to give information on trends observed in fish abundance, species occurrence and distributions (Raburu and Masese, 2012).

When developing IBIs for large areas, e.g., countries or continents, selection of reference conditions presents a number of challenges because of extreme heterogeneity in environmental conditions and biological communities. A number of studies have presented detailed approaches on how reference conditions can be defined in such cases (Herlihy et al., 2008; Hering et al., 2004; Pont et al., 2007). However, it is often difficult to have a set of reference conditions that can be applicable across large scales. This is because human disturbances are not uniform at large scales and, even if they were to be, ecological communities respond differently to similar stressors because of inherent differences imposed by geomorphic and climatic conditions. In such cases, scaling approaches are employed to develop ecoregion-specific reference conditions (Whittier et al., 2007a; Herlihy et al., 2008). However, a consequence of setting ecoregion-specific thresholds is that it is no longer possible to compare directly the overall biological condition of sites in different ecoregions because each ecoregion is graded against its own least-disturbed condition, which might differ greatly from those in other ecoregions (Herlihy et al., 2008). Despite these shortcomings, the choice of reference conditions should be standardized, in order to make environmental assessments comparable.

4.4. Collection of biological data

Collection of biological data for use in index development should be done quantitatively and this should be standardized across all sites to minimize sampling error. If an existing data set is to be used, details should be available on when, where, how, why and by whom it was collected. Sampling for macroinvertebrates and fish assemblages often follow standard rapid methods as defined by Barbour et al. (1999). For macroinvertebrate, triplicate riffle samples collected during baseflow conditions are often the most appropriate. However, some studies in the basin have also used samples collected from macrohabitats sampled in proportion to their abundance or from riffles, pools and runs, with samples from these habitats pooled into a single site-composite. For fish, the index that has been developed in the basin used samples collected by an electrofisher, which is the most common and reliable method. The method obtains quantitative samples by standardizing the time spent sampling and the area sampled.

For macroinvertebrates, the methods described above are only useful if the number of samples is small. In cases where large areas are involved and many sites should be sampled, there is a need for compromise while making sure that information needed to answer questions pertinent to water-quality management is not compromised. While such surveys are constrained by a number of factors, including management objectives, time lines, and institutional constraints related to capabilities, funding plays an overwhelming role (Hughes and Peck, 2011). In order to reduce sampling time and the costs involved, a systematic site-scale design is used rather than a single targeted-habitat approach for riffles because of time constraints, limited expertise of field crews in classifying habitat types, and training time (Hughes and Peck, 2011). A standardized study reach is then selected for each sampling point and a fixed number and area of collection points is distributed systematically throughout the reach (Hughes and Peck, 2011). Standard Kick subsamples are then collected using a dip-net from the collection points and composited to reduce shipping, processing, and analysis costs (Barbour et al., 1999). Such reach-wide sampling designs are also easy to apply consistently at most sites thereby increasing the comparability of samples (Gerth and Herlihy, 2006). This method is especially useful in streams where riffles are not representative of the site, such as sand-bottom streams common in savanna areas (Hughes and Peck, 2011).

4.5. Metric selection and testing

Successful application of the multimetric index depends on a rigorous process to identify and test or evaluate metrics (Karr et al., 1986; Karr and Chu, 2000). Metrics should reflect specific and predictable responses of a biological community to human impacts, including single and cumulative effects. Metrics can be selected a priori to objectively measure a given type of disturbance based on expected response of the assemblage to that particular type of stressor (Weigel et al., 2002) or *posteriori* based on empirical relationships based on statistical relationships with measures of disturbance, like water chemistry and habitat quality (Klemm et al., 2003). The a priori approach corresponds with the use of reference conditions and knowledge of responses expected among metrics that are selected based on prior knowledge of their variability among different environmental condition categories. On the other hand *posteriori* approach corresponds to sampling a cross a gradient of human disturbance where prior knowledge of metric expectations and responses to human influences is lacking.

The *posteriori* approach is more common because it offers testing of a large number of metrics, which provide a wider scope and a more rigorous assessment of perturbations, because, if wrongly selected using the a priori approach, the metric may fail to capture differences in environmental quality. In this way a metric is included in the final index based on its demonstrated ability to communicate resource condition in question rather than on its historical performance. In the Lake Victoria basin, the two approaches have been used by objectively selecting some metrics from literature, which are known to respond well to some particular stressors similar to the ones in the basin and other metrics were selected that corresponded with the different levels of degradation in the river basins.

(a) *Metrics based on macroinvertebrate assemblages*

Potential macroinvertebrate attributes were categorized into community structure, taxonomic composition, individual condition and functional processes, e.g., functional feeding groups (Table 3) using metrics previously used in riverine ecosystems around the world (e.g., Kerans and Karr, 1994; Barbour et al., 1999; Weigel et al., 2002; Klemm et al., 2003), and those that have been recommended for African riverine ecosystems (Richards et al., 1997). New metrics were also included 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). Testable hypotheses for these classes of attributes were then proposed regarding the direction (increase, decrease, no change or variable) to increasing levels of human disturbance (Table 3). In total twenty-two metrics were selected for evaluation. The large number of metrics evaluated provides an opportunity to capture different forms and levels of degradation (e.g., Whittier et al., 2007b; Feld and Hering, 2007).

(b) *Metrics based on fish assemblages*

Metrics were selected to reflect major fish community attributes classified under species richness and composition, indicator species, trophic groups, reproductive function, abundance and condition (Table 4). From among these classes, common metrics that have been widely applied in developing fish-based indices were selected (e.g., Karr, 1981; Hocutt et al., 1994; Hugueny et al., 1996; Toham and Teugels, 1999; Kleynhans, 1999) and modified to suit local fish assemblages (Raburu, 2003; Omukoto, 2007; Raburu and Masese, 2012). In addition, fish-based studies in satellite lakes, river floodplains, wetlands and associated ecosystems (lake margins and rivermouth wetlands) were also useful in providing information on the biology, ecology, production, taxon richness,

Table 3
Metrics for macroinvertebrates that have been considered and evaluated for development of an index of biotic integrity for riverine ecosystems in the Lake Victoria basin, Kenya, and their predicted responses to increased levels of perturbation.

Metric	Metric definition	Predicted response to increased perturbation
1. Simpson richness index	Value of Simpson richness index	Decrease
2. Number Ephemeroptera taxa[b,c,d,e]	Total number of mayfly taxa	Decrease
3. Number Plecoptera taxa[b,c,d,e]	Total number of stonefly taxa	Decrease
4. Number Trichoptera taxa[b,c,d,e]	Total number of caddisfly taxa	Decrease
5. Number Ephemeropter-Plecoptera-Trichoptera genera	Total number of taxa from mayfly, stonefly and caddisfly orders	Decrease
6. Total number of taxa[a,d]	All different taxa at a site	Decrease
7. Per cent EPT individuals[a,b,c,d,]	Per cent individuals from mayfly, stonefly and caddisfly orders	Decrease
8. Per cent non-insect individuals[d,e]	Per cent of individuals no belonging to the insect orders	Increase
9. Per cent individuals in 3 or 5 dominant taxa	Relative abundance of 3 most dominant taxa	Increase
10. Per cent individuals in dominant taxa[b,c,e]	Relative abundance of most dominant taxa	Increase
11. BMWP-ASPT[a]	BMWP-ASPT index value	Decrease
12. Per cent Diptera individuals[e]	Per cent midge individuals	Increase
13. EPT: Diptera individuals[e]	Ratio of mayfly + stonefly + caddisfly to midges (individuals)	Decrease
14. Per cent coleopteran individuals[d]	Per cent of beetle individuals	Decrease
15. Shannon diversity index	Value of Shannon diversity index	Decrease
16. Number intolerant taxa[b,c]	Total number of taxa belonging to pollution intolerant taxa	Decrease
17. Per cent intolerant individuals	Per cent of individuals in pollution sensitive taxa	Decrease
18. Per cent tolerant individuals[a,b,c,d,e]	Per cent of individuals in pollution tolerant taxa	Increase
19. Per cent filterer individuals	Filter fine organic material	Increase
21. Per cent scraper individuals	Feed on epiphytes	Decrease
22. Ratio scrapers:filterers[b]	Ration of scrapers to filter feeders	Variable
24. Per cent predator individuals[b,c,d,e]	Carnivores-scavangers, engulf or pierce prey	Decrease
24. Per cent shredder individuals	Feed on leaf litter	Decrease
25. Per cent gatherer individuals[c,d]	Collect fine deposited organic material	Variable
26. Per cent gatherer genera[a,b,e]	Collect fine deposited organic material	Variable
27. Number of individuals (per 1 m^2)[b]	Total abundance of individuals per 1 m^2	Variable

The hysterics indicate metrics that have been selected for inclusion in final IBIs in the Lake Victoria basin.
[a] Kobingi et al. (2009).
[b] Masese et al. (2009a).
[c] Raburu et al. (2009a).
[d] Raburu et al. (2009b).
[e] Aura et al. (2010).

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Table 4
Potential metrics that were considered for development of a fish-based index for monitoring riverine rivers in the Lake Victoria drainage basin their source, predicted responses to pollution.

Metrics	Predicted response
Species richness and composition	
Number of native species[a,b,c,d,e]	Decrease
Number *Barbus* species[b]	Decrease
Number catfish species[b]	Variable
Number cichlid species[d]	Decrease
Number cyprinid species[b]	Decrease
Number of rheophilic species[b,#]	Decrease
Per cent *Barbus* species[c]	Decrease
Per cent catfish species[c]	Variable
Per cent cichlid species[d]	Increase
Per cent clariid species[c]	Increase
Indicator species	
Number benthic species (excluding clarriids)[d]	Decrease
Per cent benthic species (excluding clariids)[b,e]	Decrease
Per cent *Barbus* individuals[d]	Decrease
Per cent benthic individuals[d]	Decrease
Per cent catfish individuals[d]	Increase
Per cent clariid individuals[d]	Increase
Per cent cichlid individuals[d]	Increase
Per cent cyprinid individuals[d]	Decrease
Per cent cyprinid species[d,e]	Decrease
Number of exotic species[c,e]	Increase
Per cent exotic species[d]	Increase
Number intolerant species[a,b,e]	Decrease
Per cent intolerant species[c]	Decrease
Per cent tolerant individuals[d,e]	Increase
Per cent tolerant species[b]	Increase
Trophic metrics	
Proportion as detritivore individuals[c,e]	Variable
Proportion as carnivores[a,b]	Decrease
Proportion as insectivores[a,b,c,e]	Decrease
Proportion as omnivores[a,b,c,e]	Increase
Reproductive function	
Proportion as mature individuals[c]	Decrease
Abundance and condition	
Total number of individuals[a,b,c]	Decrease
Number of individuals per 50 m of sampling[b,c,e]	Decrease
Modified index of well-being[c]	Decrease

The hysterics gives the source of the metrics.
[a] Karr (1981).
[b] Toham and Teugels (1999).
[c] Raburu (2003).
[d] New metric.
[e] Designate metrics included in the final index (Raburu and Masese, 2012).

assemblage characteristics and associated trends (Mavuti, 1989; Opiyo, 1991; Ochumba and Manyala, 1992; Gichuki et al., 2001; Goudswaard et al., 2002; Aloo, 2003; Raburu, 2003; Omukoto, 2007), in addition to highlighting the different types and intensities of human activities that threaten the integrity of fish assemblages in the basin. In total 33 metrics were evaluated and 12 were selected for inclusion in the final index.

4.6. Metric evaluation

Final metrics for development of an index of biotic integrity are selected, from among many that are initially considered, based various criteria. The criteria used depend on whether reference conditions are to be used or whether metrics are to be evaluated following their responses to measures of environmental conditions. This has resulted in many metrics in the original index suggested by Karr (1981) being replaced or adapted to regional conditions and have become difficult to compare globally or at large scales (Pont et al., 2009; Moya et al., 2011). This lack of standardization hinders a broad use of the original IBI and derived IBIs in the management of water resources (Fausch et al., 1990). In order to extend the IBIs to multiple scales, some studies have suggested the use of functional instead of taxonomic metrics, including the main factors known to affect the structure of fish assemblages, together with a regional biological variable (Pont et al., 2006, 2007, 2009; Stoddard et al., 2008). However, as indicated earlier, human influences are not uniform across different landscapes, the factors that influence biological communities can be confounded by natural variability of environmental conditions, and the metrics are not always sensitive at large scales (Stoddard et al., 2008). Stoddard et al. (2008) noted that a multimetric index (MMI) developed for US wadeable streams was less sensitive than the regional multimetric index in discriminating between impacted sites, but also indicated that it can still be used effectively for a national assessment.

In the LVB both reference conditions and testing a large number of metrics to select sensitive ones for inclusion in IBIs have been used. First, for each metric, scatter plots are examined for linearity, skewness, and kurtosis (Clarke and Ainsworth, 1993). Metrics are then transformed appropriately for normality and then examined for response as a function of stream size/order. This is to determine whether changes in river size have influences on metric values. Where reference sites are used, sites are grouped into three condition categories (reference or undisturbed, moderate or impaired) according to an independent assessment of environmental conditions based on water quality or habitat conditions. The three condition categories represent a gradient of human influence for the region that ranges from "unperturbed/reference" (near pristine or natural) through "moderately impaired" to "impaired". Metrics are then evaluated for their variability as a result of changes in stream order/size (Fig. 1). Because the two sources of variability in metric responses (river size and environmental conditions) are mutually exclusive i.e., independent of one another, a two-way analysis of variance (ANOVA) is often used, with stream order/size and condition category as main effects (Zar, 2001). For metrics that show no significant effect of stream order, one-way ANOVA is re-run with site condition category as the main effect. A Bonferroni multiple range test is then used to indicate differences among condition categories for each metric. Metrics that do not separate "impaired" condition category from either "moderately impaired" or "reference" conditions are eliminated from further consideration. The separation power of a metric that delineates between "impaired" conditions from "reference" conditions is evaluated using box plots (Fig. 2). Separation power is defined as the degree of overlap between boxes (i.e., 25th and 75th quartiles) in box plots of

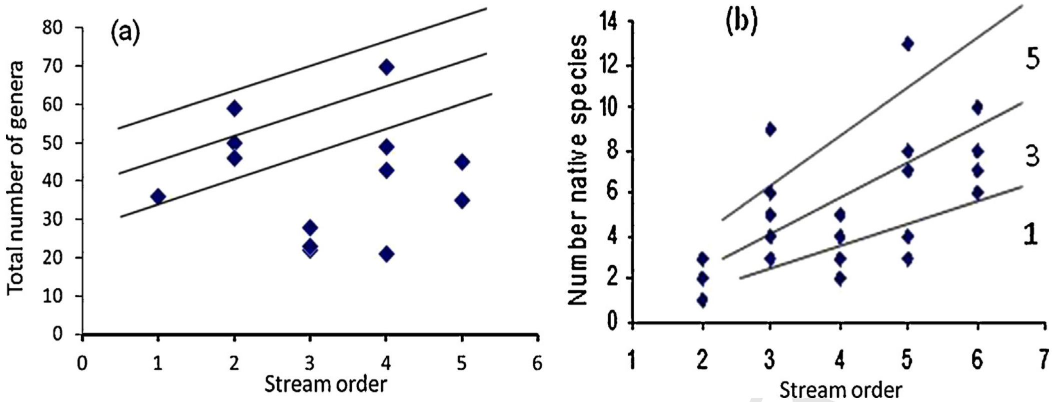
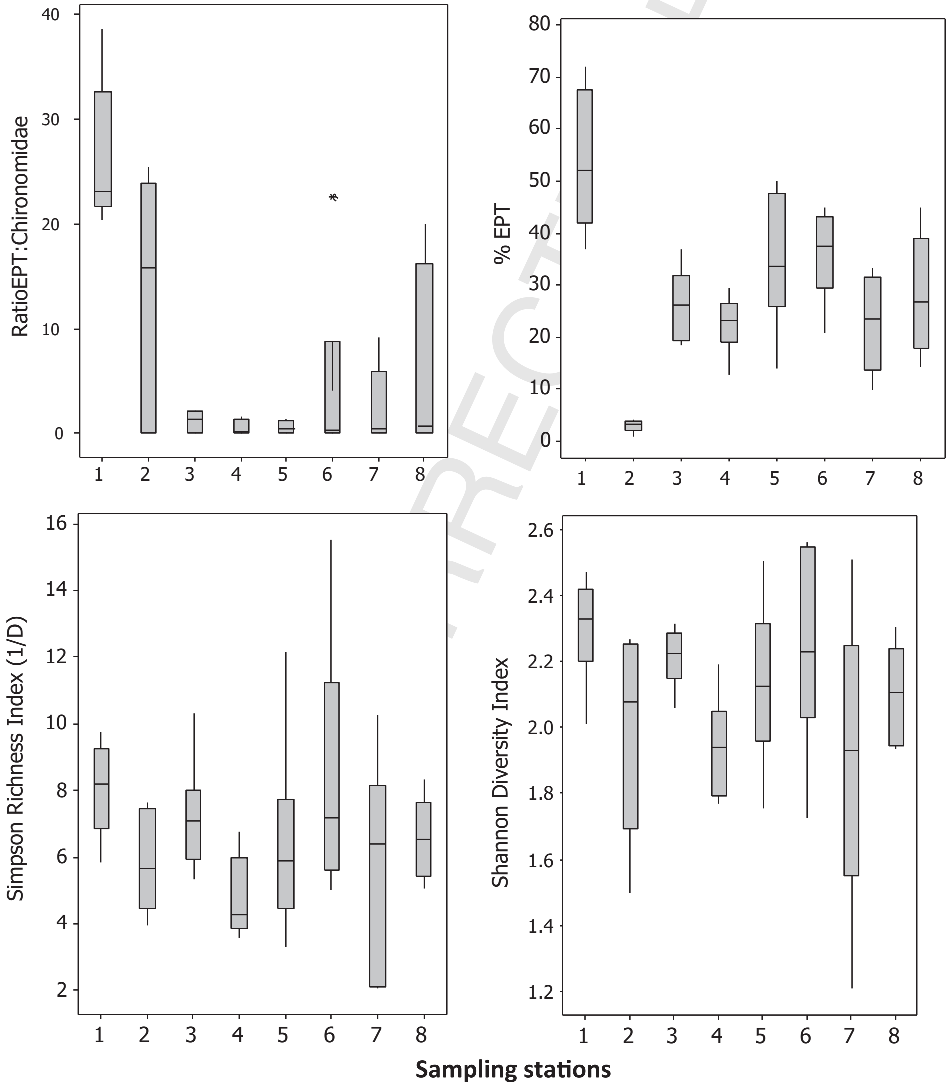


Fig. 1. Examples of metrics that are affected by changes in stream order/size in the Lake Victoria basin: (a) total number of genera of macroinvertebrates and (b) total number of native species of fish. (a) is based on macroinvertebrate samples from River Nyando (Raburu et al., 2009b) while (b) is based on fish samples collected from the Rivers Sondu-Miriu, Nyando and Nzoia (Raburu and Masese, 2012).



Q8 Fig. 2. Examples of box-and-whisker plots used to test the separation power of metrics. Metric (a) ratio EPT: Chironomidae and (b) percentage EPT perform well as they show a clear variability along gradient of human influence among some sites and there is little overlap between the distributions as the sites are discriminated according to their level of degradation. In contrast, (c) Simpson richness index (1/*D*) and (d) Shannon diversity index perform poorly as they show a weak relationship with the gradient and there is overlap between the distributions.

the values of the metric for reference and impaired sites (Barbour et al., 1999). Although calibrating metrics by stream size has a long history, other natural variables (e.g., slope, wetted width, air temperature) may alter metric performance. Thus, a number of these variables should be evaluated and influential ones selected for use during metric modelling and selection (Pont et al., 2006, 2007).

Sometimes the use of reference conditions is not possible. This normally happens due to: (1) widespread degradation that makes it difficult to identify sites that have not been impaired by human activity, (2) lack of historical records on environmental conditions, that predate the advent of human impairment, that can be used as reference conditions, and (3) sampling a small number of sites such that the number of reference sites is not statistically adequate for the calculation of metric thresholds for developing the scoring criteria. In such a situation, responsiveness of metrics to disturbance gradients are evaluated against physico-chemical water quality and habitat quality parameters by correlation or regression analysis. In the Lake Victoria basin disturbance gradients that have been considered include general disturbance (habitat quality index score, channel morphology habitat score, riparian land use score, riparian zone and bank erosion score), channel alteration (channel modification score) sedimentation (embeddedness score, turbidity, substrate quality score), acidity (pH, alkalinity, salinity, conductivity and hardness), nutrients (nitrogen and phosphorus) and decomposition (DO and BOD) among others (Raburu, 2003; Masese et al., 2009a,b; Raburu et al., 2009a,b; Raburu and Masese, 2012). Metrics that do not show any relationship with any of the disturbance parameters are eliminated from further consideration.

By using one or both of the approaches described above, it is often possible to remain with an appreciable number of metrics, some of them autocorrelated in their response to environmental conditions. This problem is solved by testing for redundancy among the remaining metrics, usually by correlation analysis. Metrics with a correlation coefficient $(r) \geq 0.85$ are considered redundant with one another (Clarke and Ainsworth, 1993), in which case only one should be included in the final index. Criteria for selection of a metric from among a group of redundant metrics include use of separation power, whereby a metric with the highest separation power is selected, or selecting a metric that displays a wide response to disturbance parameters. Other factors to consider include ecological significance, interpretability, regional significance and whether the metric is prominent among developed IBIs in other regions or studies.

Despite the significant progress that has been made to screen a number of metrics for inclusion in IBIs in the LVB, there is overreliance on professional judgement to select metrics and the statistical approaches used are not rigorous enough (Stoddard et al., 2008). Recent metric screening processes have indicated that classes of metrics are artificial and that more rigorous statistical evaluations can yield more discriminatory IBIs (Van Sickle, 2010). To develop MMIs or IBIs, especially for large scales, a shift has been witnessed from the use of subjective approaches such as professional judgement towards the use of statistical principles and processes (Fore and Grafe, 2002; Bramblett et al., 2005; Stoddard et al., 2008). This is also due to an increase in the number of candidate metrics that are being evaluated (e.g., Whittier et al., 2007b; Feld and Hering, 2007), often accompanied by a decrease in ecological knowledge about each metric (Van Sickle, 2010). The statistical approach involves a series of steps and tests to select a set of desirable metrics that incorporate a minimum number of inherent assumptions (Whittier et al., 2007b; Stoddard et al., 2008). Tests are applied sequentially such that metrics that fail a test are not considered for further and only those metrics that display all of the desired characteristics are used to build a final MMI or IBI (Whittier et al., 2007b; Stoddard et al., 2008). In summary, after metrics have been selected from different classes that capture the different attributes of the biological assemblage's biotic integrity (Karr, 1981; Karr et al., 1986) the statistical steps involved include (1) range test, (2) reproducibility (signal-to-noise test), (3) correlation with natural gradients, (4) testing for responsiveness, (5) final metric selection and check for metric redundancy, and (7) range test for metric scores (see details in Whittier et al., 2007b; Stoddard et al., 2008). Desirable metrics for inclusion in final MMIs or IBIs should have sufficient variability in data values among sites (data range), should be reproducible (temporal stability), should be responsive to stressor gradients, and should be independent from other metrics (Kurtz et al., 2001; Klemm et al., 2003; Hering et al., 2004). Despite the promise offered by statistical approaches towards improving the performance of MMIs, a number of clarifications are still needed with more research to identify the attributes of high-performing MMIs (Van Sickle, 2010). For instance, Van Sickle (2010) reported that during redundancy testing, the choices should be based on the mean metric correlation of the IBI and that several alternative IBIs should be examined. Though some of these steps and statistical approaches have been applied in the LVB as shown above, they are often not sequentially followed and some have not been considered. We recommend that more rigorous statistical tests and approaches be used in future developments of IBI in the region.

4.6.1. Scoring criteria

The interval 1, 3, 5 scoring system used in the Lake Victoria basin has been commonly used in developing fish and macroinvertebrate IBIs (Karr, 1981; Kerans and Karr, 1994; Barbour et al., 1999; Raburu et al., 2009a,b). The discrete scores (1, 3, 5) are attributed according to the measured conditions and their deviation from the least-disturbed reference conditions; higher scores are attributed to the best conditions (Karr, 1991). The condition at these least-disturbed sites represents the best-available chemical, physical, and biological habitat conditions given the current state of the catchmnets. However, the discrete scores attributed are in part subjective, because they are largely based on professional judgement (Howe et al., 2007). Discrete scoring can also have the effect of increasing the variability of the final IBI and limit its ability to differentiate among ecological condition categories (Blocksom, 2003; Stoddard et al., 2008). Professional judgement may vary, hindering a standardization of the index and increasing the probability of type I error or other errors associated with metric selection (Norris and Hawkins, 2000). To reduce subjectivity and noise, continuous scoring is recommend (0–1 or 0–10) in which the lowest value is zero and the highest one or ten (McCormick et al., 2001; Hering et al., 2006; Whittier et al., 2007b).

However, the continuous scoring system has not been evaluated to test its suitability in the LVB. To come up with the final total score or IBI value for each study site, each individual metric receives a score depending on its departure from the baseline value set for that metric, which corresponds to least-disturbed conditions for the region. Two criteria are commonly used to come up with the baseline value. The first criterion is used when reference conditions are used and the 25th and 75th percentiles of reference values are used as the upper bound and lower bound, respectively (Table 5). For metrics that decrease with impairment, sites are receive a score of: 5 if the value of the metric is >25th percentile of reference site values, 3 if the value lies between the 25th percentile of reference and the 50th percentile of impaired site values, and 1 if the value is >50th percentile of impaired site values. For metrics that increase with impairment, sites are scored a value of: 5 if the value of the metric is <75th percentile of reference site values, 3 if the value lies between the 75th percentile of reference and the 50th percentile of impaired site values, and 1 if the value is >50th percentile of impaired site values. The second criterion applies when reference conditions are not used and the best value obtained, after sampling across all sites in the study, is used as the baseline. This criterion mostly applies when most sites are degraded and it is not possible to establish reference conditions (e.g., Ganasan and Hughes, 1998; Masese et al., 2009a; Raburu and Masese, 2012). For positive metrics (i.e., those that increase with improving conditions), the highest value of a metric across all sites is trisected (Barbour et al., 1999). Values above the upper one-third received a score of 5, those in the middle received a score of 3 while those in the lower one-third received a score of 1, corresponding to unimpaired, intermediate and impaired biota, respectively (Barbour et al., 1999). For negative metrics, which decreased with improving condition, the metric is trisected but scoring is done in reverse, i.e. values above the upper one third received a score of 1, those in the middle range a score of 3 while those in the lower one-third, a score of 5. To obtain the final index score for each site values of the scores for each metric are summed.

Table 5
Method for calculating scores using 1, 3, 5 scoring system based on reference conditions.

Score	Calculation
Value of metric decrease with impairment	
5	>25th percentile of reference sites
3	<25th percentile of reference sites and >50th percentile of impaired sites
1	>50th percentile of impaired sites
Value of metric increase with impairment	
5	<75th percentile of reference sites
3	>75th percentile of reference sites and <50th percentile of impaired sites
1	>50th percentile of impaired sites

4.6.2. Condition categories and narrative descriptions

One of the oldest criticisms of indices, including the index of biological integrity, is that by converting biological data into numerical values, it loses its ecological significance. However, the use of narrative description of what the values stand for in terms of the integrity of the ecosystem in question is the best response to the criticism. The idea is to divide index scores into condition categories that convey different information in terms of how much they have been influenced by human activities. A narrative description of different condition category classes represents an interpretation of the index scores and makes it easier for nonscientists and other end-users to understand what is being presented.

Different approaches are used to group sites into condition category classes (e.g., good, fair, poor). These include using levels desired by the public or management authority, percentiles of frequency distributions of conditions at all sites or at reference sites, or thresholds in stressor–response relationships (Blocksom, 2003; Stevenson et al., 2004; Paulsen et al., 2008). In the Lake Victoria basin, methods that have been used to establish 3 or 5 condition categories include using percentiles of frequency distributions of IBI scores at reference and impaired sites to distinguish different environmental conditions at the study sites. The 50th percentile of IBI score at reference sites is used to separate "excellent" from "good" conditions while the 25th percentile is used to separate "good" from "fair" sites. The 75th percentile of IBI scores at impaired sites is used to separate "fair" from "poor" sites while the 50th percentile separates "poor" from "very poor" conditions (Raburu et al., 2009a,b). Where reference conditions are not used, the highest IBI score is used to separate sites into condition categories. To come up with 3 classes, the 75th percentile is used to separate "good" from "fair" conditions, while the 50th percentile is used to separate "fair" from "poor" conditions (e.g., Kobingi et al., 2009). Sometimes the maximum IBI score expected at a given site is used to separate sites into classes using percentiles (e.g., Masese et al., 2009a) or established ranges (Karr et al., 1986; Raburu and Masese, 2012).

When continuous scoring is used, condition categories are established by deciding on how to set ceiling and floor values for each metric, i.e., what values of a metric indicate good biological condition (score = 10) and what values indicate poor condition (score = 0). Usually, the 95th percentile of the reference-site distribution of values for each metric are used as the scoring ceiling and the 5th percentile of the distribution of values at all sites as the scoring floor (Blocksom, 2003; Stoddard et al., 2008). Alternatively, only reference-site distribution of values is used to establish condition categories. The 5th and 25th percentiles of the reference-site distributions are used as thresholds for assigning any individual site to a condition class (Paulsen et al., 2008). Sites with indicator scores <5th percentile of reference distribution are considered to be outside of the least-disturbed reference distribution and

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Table 6
Total FIBI scores, integrity classes and the narrative description of their attributes.

Condition category	Narrative description
Good	Least-disturbed streams and rivers in the basin including those in forested catchments, no human activity within 30 m of the riparian zone and no human settlement or industrial activity within 100 m, no observable impact of human activities.
Fair	Minimal human activity with natural vegetation maintained along the river banks. No human activity within 15 m of the riparian zone. No point sources of pollution.
Poor	Human activity in the river and riparian zone include agriculture, grazing and settlement, sand mining, vehicle and laundry washing, disposal of municipal and industrial wastes and wastewater.

are classified in poor condition while those sites with indicator scores >25th percentile of the reference distribution are considered to be within the range of least-disturbed sites and subsequently classified in good condition. Sites with indicator scores between the 5th and 25th percentiles of reference distribution are classified in fair condition (Paulsen et al., 2008).

Different numbers of condition category classes have been used in the literature but 5 (excellent, good, fair, poor, and very poor) and 3 (good, fair, poor) are common. The number of classes used depends on the management objectives, the level of human influence and the procedure used to develop the scoring criteria, i.e., the use of reference conditions versus the use of the best value observed across all sites. In the Lake Victoria basin, we propose that given the high level of degradation being witnessed and the difficulty of using reference conditions for establishing metric expectations/scoring criteria, 3 condition category classes are the most appropriate (Table 6). Because most streams and river reaches are in the "fair" and "poor" conditions, it is feasible to achieve the "good" condition as opposed to "excellent" conditions that are too ambitious to achieve given the resources and goodwill available to do so. However, this can be revised in future as the status of water resources improve.

4.7. Index performance and validation

The suitability of any index developed is measured by its ability to give a true picture of what it is meant to measure. For the IBI, this is achieved by the combined ability of the different metrics in the final index to respond to different levels of human influence, however subtle they may be, while at the same time being independent from natural environmental variability (Pont et al., 2007; Stoddard et al., 2008). In the Lake Victoria basin, a number of impact types on water resources that have been identified include habitat degradation, flow variations, introduction of exotic species, water quality degradation as a result of land use change, municipal and industrial wastewater discharges and changes in energy sources and organic matter processing in streams and rivers. Indices developed in the basin should therefore be able to identify sources of pollutants and point out any source of impairment from both point and non-point sources. The indices so far developed have thus far been able to achieve this goal with some degree of success. This means that water resource managers have a scientifically defensible rationale for strengthening management efforts to improve the present status. Once restoration programs have been put in place, the indices should be able to assess their success.

Validation of IBI is done by applying it to assess a new set of targeted sites of known quality and evaluating how well it predicts their level of human influence. Validation of most of the indices developed has not been done, especially the macroinvertebrate-based indices. However, the fish-based index developed was able to be validated using an independent data set collected from different sites that ranged from severely degraded to least-disturbed. The success of the validation is a function of the number of sites the index is able to place in their rightful condition category or class. The independent data set is used to avoid circularity and post hoc justification that often arises when data used to develop an index is the same data for its validation.

5. Biomonitoring opportunities and challenges

5.1. Scaling up

The use of IBIs for monitoring of surface waters, not only in the LVB but also in the entire East Africa region, holds much promise given the level of success that previous developments have achieved. There is an increasing trend to use aquatic communities as indicators of environmental quality in the region, even though most of the work has been done outside the framework of multimetric indices (e.g., Shivoga, 2001; Ndaruga et al., 2004; Kibichii et al., 2007; Masese et al., 2009b; Minaya et al., 2013). However, in order to move from the traditional approach of concentrating much effort on measuring physico-chemical parameters, which is often inconsistent because of the high expenses involved leading to gaps in data collection, there is a need for a shift to the use of biological indicators. Use of MMIs and IBIs is a good alternative in this regard.

However, inadequate reference information, which is useful as a means of establishing community expectations following restoration initiatives, is a major hindrance to development of river health indices in the region. This limits our ability to set expectations in terms of what is to be achieved. Development of IBIs is faced with another challenge of lack of historical information depicting unperturbed structural and functional organization of aquatic biota in most aquatic ecosystems. However, as demonstrated already, other approaches exist that can be used to come up with metric expectations to serve the similar purpose of measuring human influence in terms of departures from what we perceive to be baseline integrity. For now, this baseline integrity are the least-disturbed conditions that represent the best-available chemical, physical, and biological habitat conditions given the current state of human influence.

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5.2. Adoption

Any assessment and monitoring method developed should not only be scientifically sound but also be cost-effective, transferable to different river basins and regional conditions and easy to use. It should also be within legal and institutional frameworks established to monitor, manage and protect target resources. For instance, the USA Clean Water Act and the European Union Water Framework Directive form the legal foundations upon which bioassessment and biomonitoring programs are based (Hering et al., 2004, 2006, 2010; USEPA, 2013). Fortunately, plans to further Africa's water resources development are being formulated within the framework of integrated water resources management (IWRM), which seeks to develop and manage water in a manner that maximizes economic and social benefits for multiple water users without degrading ecosystems (GWP, 2000). The commitment of African nations to this framework is reflected in national water laws that have adopted IWRM as the guiding framework. The principles are also captured in Africa Water Vision 2025, which among other objectives, seeks to achieve adequate quantity and quality of water for sustainable ecosystems and biodiversity (UN-Water, 2003). The East African countries have water laws and associated regulations that recognize water for basic human needs and environmental protection as the highest priorities (Uganda Law Reform Commission, 2000; GoK, 2002; GoT, 2009). While the existing water laws and regulations have provisions for maintaining ecological integrity of streams and rivers and how they should be protected, no provision has been provided on how their ecological integrity should be assessed. This is, in part, a failure on the part of water resources and environment managers to established decision-support tools to assist in their work. Additionally, lack of linkages between research outputs and their applications to solve existing problems is evident in the continued degradation of water resources. Throughout the development of these monitoring tools and methodologies, specific collaboration has not been sought among scientists and agencies that are responsible for water resources protection and management in order to collect and share data, seek feedback on the applicability of the methods and dissemination of outputs.

6. Conclusions and way forward

For the management of the riverine ecosystems in the LVB, the different indices that have been developed are indicative of a changing environment under the influence of human activities. A similar situation is common among other river basins and catchments in the East Africa. With increasing human populations in catchment areas, the situation is likely to be exacerbated. The challenge is to mitigate deleterious trends and practices to improve the current condition of water resources. Simple measures such as maintaining forest buffers or riparian zone restoration, which have been found to be useful in improving river health (Kasangaki et al., 2006), would be a good place to start.

There is a need to adopt and strengthen existing IBIs because they reliable for identifying sources of impairment, and can also be used as a monitoring and evaluation tool to identify streams and rivers where restoration activities are needed and to monitor trends in biotic integrity and biodiversity over time (Lyons et al., 1995). While the most common use of the IBI has been as a tool of environmental protection rather than that of conservation biology, its use in the LVB can be very beneficial in terms of monitoring the diversity of assemblages that have been affected by exotic introductions. In this regard, long-term monitoring and evaluation of conservation activities become an important part of conservation biology, and the IBI is a powerful, quantitative tool for assessments at the community/ecosystem level. However, as we advocate for the use of existing IBIs in biomonitoring and bioassessment programs, considerations should be given to emerging scientific advances in metric calibration and scoring criteria, including identification of reference conditions and setting management objectives.

While the IBI can reveal important and unique insights into ecosystem health and biodiversity, it is not a quick fix to environmental problems that surface waters in the East Africa face, nor is it meant to replace other, proven types of environmental monitoring and biodiversity assessment. Data on the physical and chemical attributes of an ecosystem remain indispensable to the preservation and restoration of aquatic ecosystem integrity and biodiversity. Other communities such as diatoms, amphibians and birds should also be incorporated in biomonitoring programs. However, when time and resources are limited, relying solely on the IBI will often prove cost-effective for characterizing biotic integrity and biodiversity across broad landscapes, for identifying the biotic communities and ecosystems most in need of conservation, and for monitoring trends in ecosystem integrity and biodiversity over time. Nevertheless, it will always be advantageous to base biodiversity conservation activities on a combination of measures (see Angermeier et al., 1993) and to use the IBI as but one part of a broad, holistic effort to quantify ecosystem integrity and biodiversity (Noss, 1990).

7. Recommendations

We believe that IBIs are easier to use and efficient tools to monitor aquatic environments, especially in East African countries where resources for intensive monitoring programs and long-term studies are limited. As a matter of necessity, there is a need to bridge gaps in knowledge that exist and contribute towards the development of new monitoring tools for water resources and strengthening existing ones:

- Due to lack of historical information on most riverine ecosystems that pre-date the human perturbations, classification of human pressures will largely rely on expert judgement, especially for the characterization of physical disturbances (morphological and hydrological). Thus, the accuracy of these impact evaluations needs to be improved by training of scientists and standardizing the assessment criteria.

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- Management efforts should focus on the specific natures of impairment, along with enforcement of existing wastewater discharge standards, restoration of degraded habitats, and mitigation of further degradation, based on accurate assessment and interpretation of component metrics and an understanding of amounts and types of human disturbance.
- Consideration should be given to development of appropriate identification keys for East Africa fauna that can enable easy identification of aquatic macroinvertebrates.
- Tolerance limits need to be verified for local fauna by qualitative toxicological tests or by direct gradient analysis to identify the various environmental optima for various taxa.
- There is a need for training of more water quality professionals utilizing biological indicators as one of the methods of assessing the integrity of surface waters.

Conflict of interest

None declared.

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