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Applying an index of biotic integrity based on fish assemblages in a West African river

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Abstract

A multivariate measure of river quality, the Index of Biotic Integrity (IBI), was adapted to a West African river, the Konkoure (Republic of Guinea). Fish assemblages were sampled using gill-nets during the dry season to provide data for the IBI. Ten metrics were subsequently defined. The capacity of the modified IBI to assess the impact of a bauxite treatment plant was tested. The IBI decreased as expected at the impacted station. Detailed examination of the data revealed that one family, the Mormyridae, was almost eliminated from the impacted station, suggesting that members of this familiy can be considered as intolerant species in future uses of the IBI in other African rivers.

Introduction

The rivers of the more industrialized countries are subject to an increasing number of diverse disturbances (Hughes & Noss, 1992). As a result, numerous methods have been developed that evaluate the impact of human activities on water quality and on the aquatic ecosystem. Biological indicators are commonly used and have proven to be efficient (Karr, 1991). New criteria for evaluating the biotic integrity of rivers have led to the development of more powerful indices. The biotic integrity of an ecosystem can be defined as its capacity to maintain a community with a species richness, composition and functional organisation comparable to that in similar ecosystems that are undisturbed by human activity (Karr & Dudley, 1981). In this context, an index of biotic integrity (IBI) based on fish assemblages was developed in the United States (Karr, 1981), and introduced into Europe (Oberdorff & Hughes, 1992). This index proved useful to highlight a large range of human disturbances, such as structural changes in habitat; water quality degradations and alterations in land use. African rivers are beginning to be touched by some of these disturbances and attempts are being made to apply the IBI in a few countries: Sierra Leone (Ganda, personal communication); Cameroon (Kandem Toham, personal communication); Namibia (Hocutt et al., in press) and Republic of Guinea (present study).

This study has two aims: (1) adapt the IBI to West African rivers, particularly those of Guinea and (2) present an example of its use.

Karr (1981) recommends the use of fish assemblages to evaluate the biotic integrity of rivers for the following reasons: (1) the biology and ecology of fish species is well known; (2) fish assemblages include several different trophic groups and are thus potentially good indicators of the surrounding conditions; (3) fish species are relatively easy to identify; (4) fish are present in all but the most polluted aquatic environments and (5) fish are popular and are therefore a means of alerting the public to disturbances in aquatic ecosystems.

In the African context, these statements hold true, but must be moderated. The biology, ecology and systematics of the species are generally not as well known as in temperate regions and hence, points 1–3 lose strength. Nonetheless, fish remain the best known group of purely aquatic African animals. For instance, benthic invertebrates, other candidates for the elaboration of an IBI (Karr, 1991), are poorly known. In West Africa, ecological and systematic knowledge is sufficient to meet the needs of an IBI based on fish assemblages. Good identification keys exist (Lévêque et al., 1990, 1992) and fish diets are generally known (Lauzanne, 1988; Paugy, 1994). Species sensitivity to general disturbances is the only area where information is missing.

In African rivers particularly, a 'fish index' can be supported because fish are often an important food resource for local human populations, and are thus guarded. Furthermore, undisturbed areas, which can be used to calibrate the IBI and its metrics are still numerous in Africa, making it ideal for the development of biotic indices.

Adaptation of the IBI to West African rivers

The IBI is based on a certain number of 'metrics', each of which is representative of an aspect of a fish assemblage. We will discuss those originally defined by Karr (1981) for a few rivers in North America, and where possible, adapt them to West Africa.

Metric 1: Number of species. This metric is based on the hypothesis that a disturbed environment will have fewer species than an undisturbed one, as species that are intolerant to the disturbance will be absent. This metric has not been changed.

Metric 2: Number of intolerant species. This metric is obviously justified, species that are known to not tolerate a wide range of disturbances will disappear first. We do not retain this metric because of limited knowledge of species responses to general disturbance.

Metrics 3–5 account for the functional diversity of the assemblage expressed as the number of species of different taxonomic groups. The underlying hypothesis is that a disturbed community will be less diversified than an undisturbed one.

Metric 3: Number of darter species (fish belonging to the Percidae family). In the present application this metric is replaced by the number of mormyrid species (Mormyridae family). In other studies this metric has been replaced by benthic specialist species (Hocutt et al., in press) and by benthic invertivores (Miller et al., 1988). As mormyrids feed on benthic invertebrates they make up a substantial part of benthic specialist species and benthic invertivores.

Metric 4: Number of sunfish species (fish belonging to the Centrarchidae family). Fausch et al. (1984) suggested that in tropical zones this metric should be replaced by the number of cichlids (Cichlidae family), which will be done here, as in Hocutt et al. (in press). Other authors (Oberdorff & Hughes, 1992) suggested water column species as a functional replacement. Due to their percomorph body plan favouring maneuvrability, cichlids are water column species well suited to live in structured edge or vegetated habitats (Winemiller et al., 1995).

Metric 5: Number of sucker species (fish belonging to the Catostomidae family). In this study this metric was replaced by the number of large, longlived, benthic siluriform species. Oberdorff & Hughes (1992) proposed large, long lived species as a replacement, and Steedman (1988) classified suckers and catfishes into the same metric. Our metric is thus a compromise between these two suggestions while retaining the association with river bottom displayed by suckers.

Metric 6: Percentage of individuals that are Green Sunfishes. This metric takes into account the dominance of this species in highly disturbed areas. In African waters the resistance of species to disturbance is not well enough known to enable an equivalent species, or group of species, to be suggested.

Metric 7: Percentage of individuals that are omnivorous species. The number of omnivores is meant to increase in disturbed environments, where specialized sources of food are rare or absent. This metric is unchanged.

Metric 8: Percentage of individuals that are invertivorous species. This metric is unchanged.

Metric 9: Percentage of individuals that are largely piscivorous species as adults. This metric is retained.

Metric 10: Number of individuals. In a disturbed environment the number of individuals is expected to be less than that observed in an undisturbed environment. This metric has not been modified.

Metric 11: Percentage of hybrid individuals. It is generally assumed that hybridization between species is favoured by changes in their habitats. This metric has not been modified.

Metric 12: Percentage of individuals with anomalies or that are disease ridden. When the environment is highly disturbed, disease and problems in development or growth may appear. This metric has not been modified.

Table 1 presents a summary of the original metrics and their adaptation to the fish assemblages of West Africa.

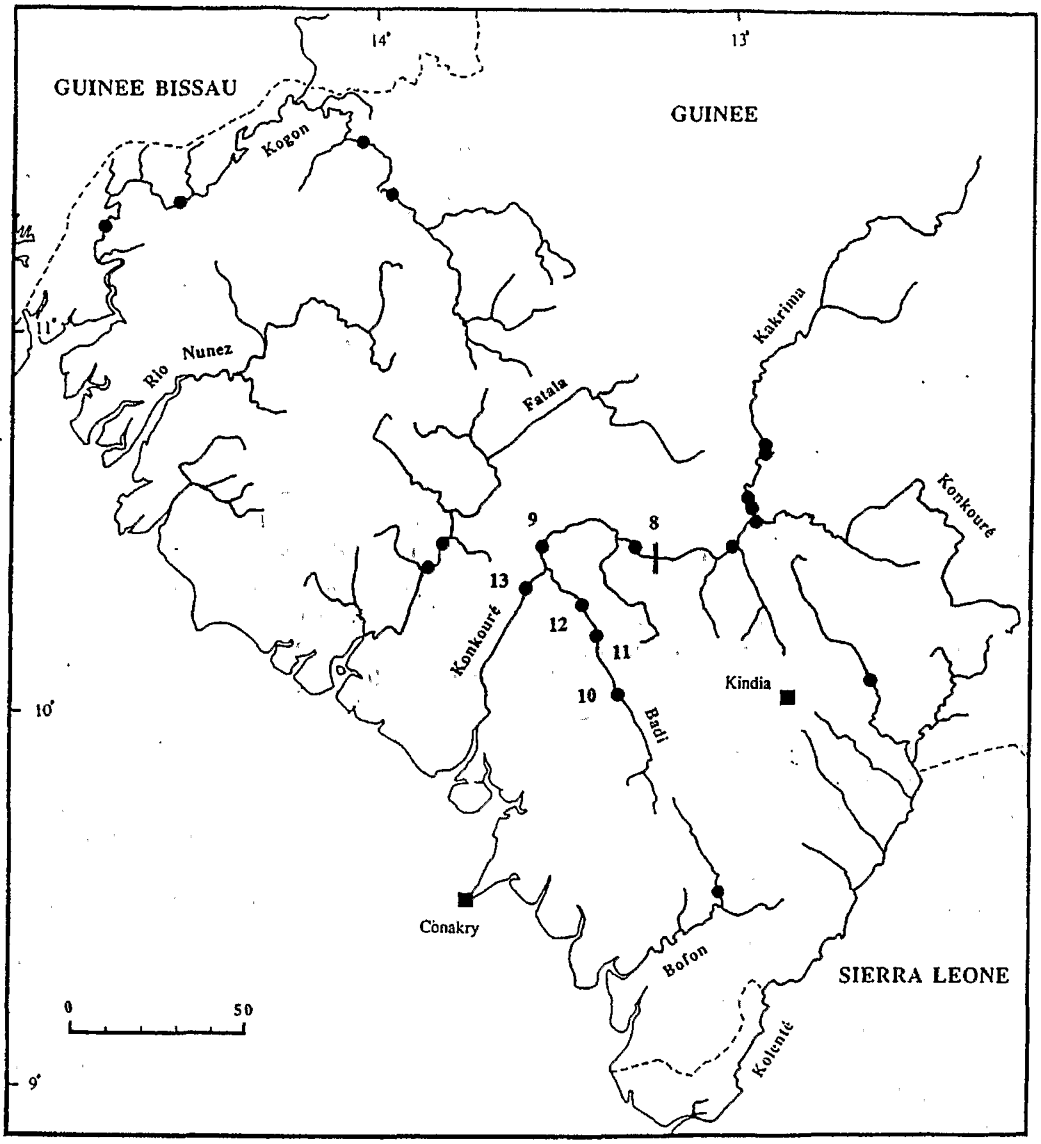

Figure 1. Map showing location of sampling sites on the Konkouré river in République de Guinée. Unnumbered points are sampling sites not considered in this study. At Fria (13) there are two sampling sites, I and R, respectively downstream and upstream from the reject. At locality 10 there are two sampling sites: one (B) is on a tributary of the Badi, the other (A) is on the Badi. Kaleta falls are represented by a bar crossing the river.

Application to a concrete case

The modified IBI is used on a length of the Konkouré basin in the Republic of Guinea. The river was chosen for the homogeneity of its ichthyofauna and because of the presence of a point source of pollution, which allows the IBI to be tested *in situ*. The Konkouré River, the largest Guinean river along the atlantic coast, drains an area described by White (1983) as a mosaic of lowland rain forest and secondary grassland. In the studied area the climatic regime is Guinean; mean annual rainfall is about 2800 mm, the rainy season ranges from June to November and rainfall reaches its maximum between July and September.

The study zone is made up of the Konkouré River downstream of the Kaleta waterfalls and its tributary, the Badi (Figure 1). Within this zone, the ichthyofauna is homogeneous (Hugueny, unpublished data), and no

Table 1. Fish community metrics from the original IBI and corresponding metrics adapted for this study. All metrics are assumed to display low values in disturbed localities except those in italics.

	Original metrics	Adapted metrics
1	Number of fish species	No change
2	Number of intolerant species	No equivalent
3	Number of darter species	Number of mormyrid species
4	Number of sunfish species	Number of cichlid species
5	Number of sucker species	Number of large benthic siluriform species
6	*Percent of individuals as Green sunfish*	No equivalent
7	*Percent of individuals as omnivores*	No change
8	Percent of individuals as invertivores	No change
9	Percent of individuals as piscivores	No change
10	Number of individuals	No change
11	*Percent of individuals as hybrids*	No change
12	*Percent of invividuals with anomalies*	No change

noticeable changes in the assemblages occur relative to the width of the river. Consequently, it is not necessary to standardize the species richness metrics according to the size of the river as is usually the case. At Fria, about 50 km from the river's mouth, there is a bauxite treatment plant. The most obvious disturbance this plant causes is an increase in pH due to the release of soda into the Konkouré. The pH is 10 at the reject, 8 where sampling is carried out in the impacted station, and between 6.2 and 7 elsewhere in the Konkouré. Unfortunatly, no access to the river further downstream from Fria was found. As a result, there is no way to assess downstream recovery.

The study zone was fished 25 times in 8 sites with gill nets during 1992 and 1993 (Table 2). We fished during the dry season (December to May) as most stations are inaccessible from the road during the rainy season. In West Africa the principal seasonal changes in fish assemblages occur between the dry and rainy seasons (Lévêque et al., 1988). Seasonal variability can thus be considered low in our data. Sampling is more efficient in pools which fit the following criteria: more than one meter deep and 500 m long, 70–200 m wide, at current velocities less than 0.2 m/s. One exception is locality 10B, a tributary of the Badi, which is about 30 m wide. However, as this locality is located at the confluence with the Badi, reduced river width has no obvious effect on the fish assemblage. Two gangs of gill nets were used, each with the following mesh sizes (in mm): 10, 12.5, 15, 17.5, 20, 22.5, 25 and 40. Nets were set for one night (from 16:00 to 8:00 the next morning).

Assignment of species to trophic categories is based on Paugy (1994), Lauzanne (1988) or Welcomme (1985). Species absent from lists provided by these authors, are assumed to belong to the same trophic category as congeneric species. Available stomach content analyses for Guinean species (N. Baude, unpublished data) lead to results concordant with the literature. The main discrepancy is for *Schilbe micropogon* which, from these data, is clearly a piscivore and not a omnivore as are most of its congenerics. On the basis of stomach content analyses, *Tylochromis* spp is classified as an omnivore. The trophic status of *Pelvicachromis* sp remains unknown. For fish identification keys given in Lévêque et al. (1990, 1992) have been used. In the Konkouré River the two species of *Petrocephalus* known to occur (*P. levequei* and *P. tenuicauda*) are morphologicaly close to each other preventing a confident diagnosis (R. Bigorne, personal communication). Consequently these two species have been grouped in our database. Two morphs of *Tylochromis* were found in the Konkouré river but we were unable to match them precisely with species described in West Africa. Thus, all *Tylochromis* were grouped in the same category in our database. One unindentified *Pelvicachromis* species was found. These taxonomic uncertainties affect a few metrics slightly.

We scored the 10 retained metrics for each station, except for the % hybrids (none were found) and % disease ridden individuals (difficult to assess with fish sampled in gill-nets). Each metric was then centered and reduced, and the IBI was calculated as the sum of metrics transformed in this way. This procedure is comparable to that generally used to calculate the IBI,

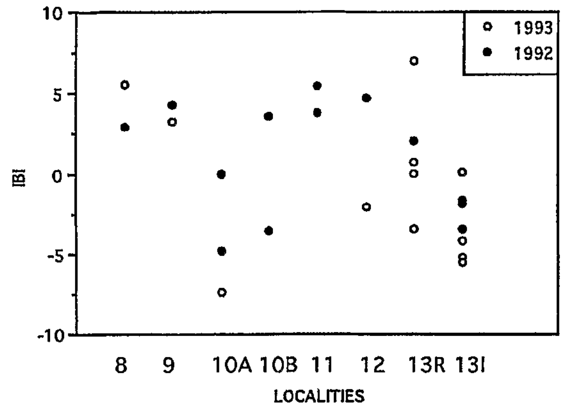

Figure 2. Plot of IBI scores by site and year. Sites are numbered as in Figure 1.

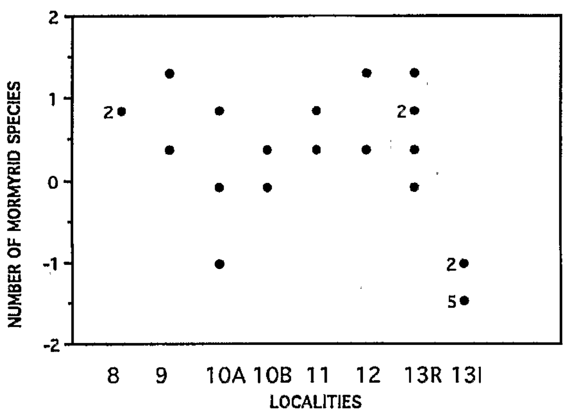

Figure 3. Scores of the metric 'number of mormyrid species'. Sites are numbered as in Figure 1. The number of coincident points is given by a numeral.

which consists of creating a constant number of classes for each metric, while losing the least possible information. Metrics expected to increase with disturbance are multiplied by -1 before summing. The average IBI observed at the disturbed area, downstream of Fria, is compared to the IBI's of the undisturbed stations. The same was done for each metrics to determine those that lead to variation of the IBI. We used one-tailed, non-parametric Mann-Whitney rank tests, because of the small sample size and the expected lower values in the impacted site.

Results and discussion

Both fishing results and the biological characteristics needed to calculate the IBI are shown in Table 2. Spatio-temporal variability of the IBI score is displayed in Figure 2. The IBI, as expected, is lower at the station downstream of the bauxite treatment plant but also in station 10A where there is no known disturbance. Thus the spatio-temporal variation of the IBI scores in reference sites is quite high. Karr et al. (1987) observed that sites displaying low changes in IBI score through time were those with the higher mean IBI score. This pattern is not observed in our data. Despite this spatio-temporal variation of IBI scores, the difference between reference sites and treatment site is statistically significant (Table 3). Among the metrics included in the index, three show the opposite trend to that expected (Table 3). These are the number of cichlid species, the number of large benthic siluriform species, and the percentage of individuals that tend to be piscivorous. Of the metrics that evolve as expected, four have a significantly lower value in the disturbed zone than in the reference zones (Table 3). These are species richness; the number of mormyrid species; the percentage of individuals as omnivores and the percentage of individuals as invertivore species. Considering the number of mormyrid species the difference between values observed in reference sites and treatment site is particularly striking as, except for one value, there is no overlap (Figure 3).

In order to use the IBI correctly, a certain number of precautions must be taken. First, the number of species in the community must be sufficiently high so that one or a few species do not cause important and unexpected variations in the IBI (Bramblett & Fausch, 1991). This is not a problem here, as even in the disturbed station there is always at least 8 species. Another condition is that the 'natural' spatio-temporal variability of the community that is likely to come out in the IBI rnust be known (Karr et al., 1987). This is one of the most limiting conditions, as it requires much field data, an uncommon situation in many countries. By sampling only large pools, the fishing method limits variation between environments sampled and consequently the spatial variation in the assemblages. Nevertheless, more data about spatio-temporal variability are needed, particularly those dealing with assemblage changes along the channel which must be integrated in the IBI. It is also recommended that sampling be representative of the assemblage. Fishing with gill nets is problematic as this is a selective method. It is possible that certain species that may significantly influence a metric are inadequatly sampled and lead to a decrease in IBI sensitivity. For example, cichlids are generally not efficiently sampled with gill nets, and for this family only very large variations in number or in species

Table 2. Number of individuals per species collected using gill-nets from 25 samples within the Konkouré river. Sites are numbered as in Figure 1. 131 = samples from the impacted locality. Biological features: M = mormyridae, C = cichlidae, S = large benthic siluriform species, I = invertivore, O = omnivore, P = piscivore.

	Site	8	8	9	9	10A	10A	10A	10B	10B	11	11	12	12	13R	13R	13R	13R	13R	131	131	131	131	131	131	131
	Month	05	05	02	03	02	05	05	02	05	02	05	05	05	01	03	05	05	12	02	05	03	03	05	05	12
	Year	1992	1993	1992	1993	1992	1992	1993	1992	1992	1992	1992	1992	1993	1992	1993	1993	1993	1993	1992	1992	1993	1993	1993	1993	1993
Polypterus palmas	P	0	0	0	1	0	0	0	2	1	1	0	0	0	0	1	0	0	0	3	1	0	1	0	0	0
Papyrocranus afer	O	0	0	2	2	0	0	0	2	1	0	1	2	2	1	4	0	1	0	1	1	0	0	1	1	1
Brienomyrus brachyistius	M, I	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hippopotamyrus paugyi	M, I	1	9	1	0	1	1	0	1	0	4	1	2	0	2	7	3	0	3	0	0	0	0	0	0	0
Marcusenius mento	M, I	2	0	2	3	3	1	0	0	2	3	3	8	11	2	0	5	4	1	0	0	0	0	0	1	0
Marcusenius thomasi	M, I	5	9	7	2	3	0	0	0	5	7	13	12	2	17	10	7	3	2	0	0	0	0	0	0	0
Mormyrop anguilloides	M, I	0	3	2	1	1	0	0	2	0	0	0	2	1	0	0	0	0	2	0	0	0	0	0	0	0
Mormyrus tapirus	M, I	2	2	1	0	0	0	0	0 0	0	1	1	0	1	1	1	0	1	0	0	0	0	0	0	0	
Petrocephalus sp.	M, I	16	185	5	12	3	9	6	6	4	18	41	4	28	85	16	12	5	6	0	0	0	0	0	0	
Hepsetus odoe	P	0	0	1	0	0	0	1	1	0	1	0	0	5	2	2	1	0	0	0	0	0	2	0	0	0
Brycinus longipinnis	O	0	0	0	0	3	6	8	78	19	2	15	0	19	1	0	2	0	0	0	0	0	0	0	3	0
Brycinus macrolepidotus	O	12	44	13	13	10	6	15	4	11	5	19	10	40	51	32	4	11	26	8	20	22	84	18	17	60
Brycinus nurse	O	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hydrocynus forskalii	P	2	2	1	1	2	7	2	8	1	3	5	7	1	0	1	0	0	2	1	0	1	1	1	0	0
Barbus macrops	O	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	7	0
Auchenoglanis occidentalis	S.I	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	1	1	0
Chrysichthys johnelsi	S,I	0	0	1	2	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	3	2	2	4	0
Chrysichthys maurus	S,I	12	17	12	19	0	1	0	6	1	4	5	8	0	22	10	15	3	3	1	7	9	2	4	12	10
Schilbe micropogon	P	0	0	10	8	3	1	8	22	5	49	11	7	19	29	0	2	1	35	15	1	2	5	6	10	6
Malapterurus electricus	S,P	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synodontis ansorgii	S, I	0	0	0	0	0	0	2	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synodontis levequei	S, I	1	1	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Synodontis thysi	S, I	0	0	0	0	0	0	0	1	0	0	1 0	0	0	0	0	0	0	0	0	0	0	0	0	0	
Synodontis waterloti	S. I	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0
Lates niloticus	P	4	3	2	2	1	1	0	0	0	2	1	3	0	2	2	4	1	0	1	5	3	3	6	4	3
Hemichromis fasciatus	C,P	7	1	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	1	0	0	2	2	1
Pelvicachromis sp.	C	1	0	0	0	0	0	0	0	0	0	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0
Sarotherodon caudomarginatus	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
Tilapia brevimanus	C	0	0	1	1	0	1	0	1	3	0	1	0	0	0	0	0	0	0	2	1	2	4	0	3	1
Tilapia rheophila	C	0	0	0	0	0	0	0	1	1	0	0	4	0	0	0	3	1	0	0	0	0	0	0	0	0
Tylochromis sp.	C, O	0	0	1	0	0	1	0	4	0	0	1	0	0	0	1	1	3	1	3	2	4	1	3	19	1
Ctenopoma kingsleyae	I	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0	0	0	0	0	0	0

Table 3. Results of the Mann-Whitney test of metrics and IBI ranks between samples from the reference sites and the treatment site. One-tailed probabilities are given. a: mean rank in treatment samples are higher than or equal to mean rank in reference samples; b: not computed. Between brackets are given probabilities obtained when metrics are computed without the Mormyridae.

Metrics	p
Number of species	0.005 (a)
Number of mormyrid species	<0.001 (b)
Number of cichlid species	a (a)
Number of large benthic siluriform species	a (a)
% of individuals as omnivores	0.002 (0.215)
% of individuals as invertivores	<0.001 (0.487)
% of individuals as piscivores	a (a)
Number of individuals	0.284 (a)
IBI	0.009 (b)

richness are detected. Nonetheless, even if gill netting influences the sensitivity of the IBI, the results are not biased if a standard protocol is followed. Gill nets are also often the only sampling method that can be used in large rivers as is the case here. Moreover, low conductivity of most West African rivers precludes electrofishing.

The IBI showed the effect of !he bauxite treatment plant at Fria on the fish assemblages. The metrics that change significantly (and in the expected direction) between the treatment and reference zones are species richness; the number of mormyrid species; the % invertivores, and the % omnivores. The increase in omnivores and the decrease in invertivores at the disturbed site suggests a change in food supply. However, the family Mormyridae makes up 75% of the invertivore individuals. Thus it is possible that the decrease in invertivores is due to the decline of this family in the treatment site for a reason other than food availability. In the same way, species richness may be affected by the absence of mormyrids species in the treatment site. If the analysis is redone without the mormyrids, none of the metrics affected by this change differ significantly between the reference and treatment zones (Table 3). This result suggests that the mormyrids are entirely responsible for the variation in the IBI observed between the reference and treatment zones. This conclusion is all the more plausible given that only two mormyrids were found in the treatment area. Despite the lack of data in the literature, it is common knowledge that mormyrids are very sensitive to many stressors. The absence of this family is probably explained by their high sensitivity to the physico-chemical conditions, which is an increase in pH here, rather than the disappearance or decline in their main food source, benthic invertebrates. Another point worth discussing is the increase in the number of cichlids species in the disturbed zone when a decrease would be expected. This metric may actually behave unexpectedly. Certain cichlid species, for instance some *Sarotherodon* and *Tilapia* species, are very tolerant, as has been shown by their successfull introduction into diverse environments (Welcomme, 1988). The usefulness of this metric therefore depends on the proportion of tolerant species among the cichlids. Apparently tolerant cichlids species dominate the fauna of the Konkouré. This hypothesis needs further confirmation. However representatives of this family are never present in high numbers in the samples. which means interpretations based on this metric are difficult. Although the usefulness of the IBI on the Konkouré is conclusive, the cases of the mormyrids and cichlids underline the importance of detailed knowledge on the tolerances of species to major types of disturbance. The results suggest that the mormyrids are sensitive species and 'number of mormyrid species' could be an equivalent for the original metric 2 as well as for metric 3 (Table 1). In the same way, metric 6 could include the percentage of individuals belonging to certain cichlid species. Representatives from the genus *Clarias* or *Barbus*, which are often the last species to survive in extreme conditions (Lévêque, 1990) also fill this role.

Hocutt et al. (In press) proposed different metrics than ours in a modified IBI for the Kavango river in Southern Africa. However, as there is no test in an impacted situation, there is no way of deciding if their metrics work better than ours. Species tolerance to anthropogenic disturbance was not integrated in their metrics which is needed in Southern Africa as well as in West Africa. They emphasize the high seasonal and spatial (longitudinal) variability observed for some metrics in pristine situations and this is also a concern in West Africa. For instance, studies on longitudinal zonation of fish assemblages provide mixed results in West African rivers and a clear pattern hardly emerges (Welcomme & de Mérona, 1988). Furthermore a substantial interannual variability in IBI values is expected in West Africa because fish assemblages display high interannual variability there (Hugueny et al., 1995)

One of the advantages of the IBI is that it is potentially sensitive to a variety of types of disturbance: organic or physico-chemical pollution, decline in habitat structure, etc. The next logical step is to test the modified IBI with other disturbance situations to establish its breadth of use. New metrics will also eventually be suggested, or certain metrics adapted to render its use as general as possible. We hope that ongoing and future projects will apply this method in African rivers and provide data leading to improvements in our ability to assess biotic integrity.

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