Aquat. Living Resour. 19, 31–46 (2006)
© EDP Sciences, IFREMER, IRD 2006
DOI: 10.1051/alr:2006003
www.edpsciences.org/alr

Aquatic
Living
Resources

A fish-based index of large river quality for French Guiana (South America): method and preliminary results*

Francisco Leonardo Tejerina-Garro[1], Bernard de Mérona[2,a], Thierry Oberdorff[3] and Bernard Hugueny[2]

[1] Centro de Biologia Aquática, Universidade Católica de Goiás, Campus II, Av. Bela Vista Jardim Olímpico, Cx P. 86, CEP 74506010, Goiânia, Goiás, Brazil
[2] Institut de Recherche pour le Développement (UR131), Antenne au Laboratoire d'Écologie des Hydrosystèmes fluviaux, Université Claude Bernard Lyon-1, 43 bd. 11 novembre 1918, 69622 Villeurbanne Cedex, France
[3] Institut de Recherche pour le Développement (UR131), Antenne au Muséum d'Histoire Naturelle, 43 rue Cuvier, 75231 Paris Cedex 05, France

Received 5 September 2005; Accepted 14 December 2005

Abstract – Growing pressures from development activities, on aquatic environments in South American countries, have created an urgent need for tools to measure the quality of aquatic systems. The index of biotic integrity, based on fish assemblages, elaborated in North America, constitutes a basis for the development of these tools. However, its direct application is problematic in regions having a rich and diverse ichthyofauna and where the knowledge of fish species ecology is incomplete. The response of taxonomic or trophic groups is often unknown and the selection of fish assemblage metrics cannot be based on previous knowledge, as in intensely studied regions. This work proposes a method, similar to the approach recently pursued in Europe, based on comparing the variability of fish assemblages in reference situations with their variability influenced by human-induced perturbations. The method was developed from 53 samples of 27 reaches in 7 hydrographic basins of French Guiana. For each of 28 fish assemblage descriptors, stepwise multiple linear regressions with 28 habitat variables were carried out. The residuals of the models obtained were used as candidate metrics independent of natural environmental factors. Nine metrics showing significant differences between reference and disturbed samples were selected to constitute the index. The index was validated by analysing a temporal data series obtained from a reach disturbed by dam construction.

Key words: Tropical fish / Freshwater fish / Fish assemblages / River quality / Biotic index

Résumé – Un indice de qualité des grands fleuves de Guyane française, basé sur les poissons : méthode et résultats préliminaires. Du fait de pressions croissantes des activités de développement sur l'environnement aquatique dans la plupart des pays d'Amérique du Sud, la nécessité de disposer d'outils de surveillance des milieux aquatiques est urgente. Il s'agit d'élaborer des indices capables de mesurer la qualité de ces milieux. L'indice d'intégrité biotique, basé sur les communautés de poissons, développé en Amérique du Nord, constitue une base pour le développement de ces outils. Cependant, son application directe pose problème dans des régions possédant une riche faune piscicole, et où les connaissances sur l'écologie des espèces ne sont que fragmentaires. En effet, la réponse de groupes taxonomiques ou trophiques à des perturbations est la plupart du temps inconnue ; la sélection de métriques des peuplements pour la constitution d'un indice ne peut donc se baser sur une connaissance préalable, comme c'est le cas dans les régions intensément étudiées. Aussi, nous proposons une méthode, similaire à celle développée en Europe, et basée sur la comparaison de la variabilité des peuplements de poissons en situation non perturbée avec celle observée sous l'effet de perturbations. La méthode a été développée à partir de 53 échantillons réalisés sur 27 sites de 7 bassins hydrographiques de Guyane française. Pour chacun de 28 descripteurs des peuplements de poissons, des relations ont été établies avec 28 variables d'habitat, à partir de régressions multilinéaires pas à pas. Les résidus de ces relations ont été utilisés comme métriques susceptibles d'entrer dans la constitution d'un indice. Neuf métriques présentant une différence significative entre prélèvements perturbés et non perturbés ont été retenues pour la constitution de l'indice. Une première validation de l'indice proposé a été réalisée par l'analyse d'une série temporelle dans une station perturbée par la construction d'un barrage hydroélectrique.

* Appendix is only available in electronic form at
http://www.edpsciences.org/alr

[a] Corresponding author: bernard.de-merona@univ-lyon1.fr

Article published by EDP Sciences and available at http://www.edpsciences.org/alr or http://dx.doi.org/10.1051/alr:2006003

1 Introduction

Most South America countries are confronted with the conflicting interests of development and environmental preservation. Deforestation, agriculture, urban and industrial development, installation of hydroelectric dams, mining and other anthropogenic activities cause various environmental impacts that degrade terrestrial and aquatic environments (Pringle et al. 2000). A first step in preserving the environment, and more specifically the aquatic one, is to develop tools to measure environmental quality in order to anticipate or predict environmental impacts (Hughes and Oberdorff 1999). The Index of Biotic Integrity (IBI) of Karr (1981) follows this framework. The IBI is based on the concept of biological integrity which is defined as "the ecological capacity of a system to support and maintain adaptable, balanced and integrated biological communities having a specific composition, diversity and functional organization comparable with those found in a natural habitat" (Karr and Dudley 1981).

The IBI, initially based on fish assemblages, was first conceived for the water courses of Central North America by Karr (1981) but, later it was adapted to the USA and Canada (Fausch et al. 1984; Miller et al. 1988; Dionne and Karr 1992; Simon and Lyons 1995) and subsequently for France (Oberdorff and Hughes 1992; Oberdorff and Porcher 1994), Belgium (Kestemont et al. 2000; Belpaire et al. 2000; Breine et al. 2004), Lithuania (Kesminas and Virbickas 2000), Africa (Hocutt et al. 1994; Hay et al. 1996; Hugueny et al. 1996; Kamdem-Toham and Teugels 1999), Asia (Ganasan and Hughes 1998), New Zealand (Joy and Death 2004), Australia (Harris 1995), Venezuela (Gutierrez 1994), Mexico (Lyons et al. 1995), Argentina (Hued and Bistoni 2005) and Brazil (Araujo 1998; Araujo et al. 2003; Bozzetti and Schultz 2004). According to Hughes and Oberdorff (1999), the adaptation of the IBI within and beyond North America preserved the same ecological framework; that is, it was based on the hypothesis of the predictive relationships between fish assemblages and the physical, chemical and biological conditions of the water courses. However, these applications made no explicit reference to local environmental factors potentially influencing fish assemblages, they used instead previous extensive knowledge of fish species' responses to disturbance in the studied region. Thus, applications of the IBI in regions where the ecology of fish is partially unknown must be based on supposed equivalents of some of the original IBI metrics (Hocutt et al. 1994; Gutierrez 1994; Hugueny et al. 1996; Ganasan and Hughes 1998; Araujo et al. 2003; Bozzetti and Schulz 2004). Based on the same premises established for fish IBI construction, a modified version using the reference condition approach (Wright 1995; Bailey et al. 2004) was proposed for all water courses of France and Europe (Oberdorff et al. 2002; Pont et al. in press). This approach consists of integrating environmental factors acting on fish assemblage structure under natural conditions to distinguish human-induced disturbances from natural variations. Numerous studies have indicated strong relationships between the structure and composition of local fish assemblages and aquatic habitat characteristics in the absence of human-induced perturbation (reviewed in Tejerina-Garro et al. 2005). The role of an index of biotic integrity is to identify and measure the deviation from this relationship in habitats perturbed by human activities (Karr et al. 1983).

French Guiana is an administrative department of France. This political status subjects it to the European Water Framework Directive (WFD), which requires restoration and maintenance of "healthy" aquatic ecosystems following the assessment of their hydro- morphological, chemical and biological characteristics. To conform to this requirement, the "Direction de l'Environnement" funded studies to develop water quality indices and initiate routine river surveys.

We propose a method of constructing a preliminary index of river quality in a poorly known tropical context in three steps: 1) search for fish-habitat relationships in undisturbed reaches, 2) select metrics sensitive to perturbations, and 3) construct a preliminary index by scoring the identified metrics. We used observations from 7 watersheds in French Guiana, South America.

2 Materials and methods

2.1 Study area

French Guiana has a surface area of 90 000 km^2 and is located between 51°40′ to 54°30′ E and 1°30′ to 5°50′ N (Fig. 1). It is surrounded by Brazil to the East and South and Suriname to the West. This area is part of the Precambrian Guiana Shield. Two contrasting zones exist in French Guiana: 1) the coastal plain lowlands (altitude <50 m) formed by marine sediments of the tertiary or quaternary period and covered by rainforest, marsh and/or savanna; 2) the inner uplands, formed mainly by volcanic and uplifted sedimentary rocks, covered by dense rainforest. The climate is equatorial, characterized by strong and frequent rains. The rainy season extends from November to July (average annual precipitation: 4500 mm) and the dry season from July to November (average annual precipitation oscillating from 100 to 800 mm) (CNRS/ORSTOM 1979). Air temperature is uniform all year long (average annual temperature in Cayenne, 25.8 °C).

The hydrographic network of French Guiana is characterized by extreme density due to the importance of the annual rains and the gentle terrain slopes. The river basins can be classified by size: large size basins such as the Maroni (65 830 km^2), medium size basins such as the Sinnamary (6565 km^2), and small size basins such as the Malmanoury (138 km^2). We sampled 27 reaches distributed within two large basins (Maroni and Oyapok), two medium sized ones (Sinnamary and Mahuri) and two small sized ones (Malmanoury and Karouabo) (Fig. 1 and Table 1). We also sampled a small tributary of the Kourou River for comparison with the coastal creeks. In each of these basins the reaches were chosen to include reference reaches and reaches with known human perturbations. The basins were chosen by considering access facilities, opportunities to use data from other research projects and existence of identified perturbations. With the exception of the two reaches on the Oyapok River, the reaches were sampled twice between 1998 and 2000. The two samples were planned to occur during high water and low water periods, however, due to natural variability of the rains and technical constraints this was not always possible. The 27 reaches were classified a priori as reference or disturbed.

Table 1. Reaches sampled during the high water (H) and low water (L) seasons in French Guiana. The known or supposed environmental disturbances are indicated. IW: industrial waste, DAM: dam, GM: gold mining, UR: urban and/or agricultural diffuse pollution.

Code	Basin	Reach	Date	Disturbance	Season (water level)
1	Oyapok	Camopi River	May-1999		H
			Nov.-1998		L
2	Mahuri	Comté River upstream	Apr.-1999		H
			Oct.-1998		L
3	Mahuri	Comté River downstream	Apr.-1999		H
			Oct.-1998		L
4	Sinnamary	Saut Dalle	Jun.-1999		H
			Nov.-1999		L
5	Maroni	Grand Inini River	Jan.-1999		H
			Jun.-1999		H
6	Karouabo	Bridge	Apr.-1999	IW	H
			May-1998	IW	H
			May-2000	IW	H
7	Sinnamary	Kerenroch	Jul.-1999	DAM	L
			Dec.-1999	DAM	H
8	Mahuri	Kounana River upstream	Jul.-1998		L
			Dec.-1998		H
9	Mahuri	Kounana River downstream	Jul.-1998		L
			Dec.-1998		H
10	Sinnamary	Koursibo River	Jun.-1999		H
			Dec.-1998		H
11	Maroni	Langa Tabiki	Jun.-1999	GM+UR	H
			Oct.-1999	GM+UR	L
12	Sinnamary	Leblond River	Jun.-1999	GM	H
			Dec.-1998	GM	H
13	Malmanoury	Bridge	Apr.-1999		H
			May-2000		H
14	Maroni	Maripasoula upstream	Jan.-1999		H
			Jun.-1999		H
15	Maroni	Maripasoula downstream	Jan.-1999	GM+UR	H
			Jun.-1999	GM+UR	H
16	Mahuri	Orapu River upstream	Jul.-1998		L
			Dec.-1998		H
17	Mahuri	Orapu River downstream	Jul.-1998		L
			Dec.-1998		H
18	Oyapok	Camopi	Apr.-1999		H
19	Oyapok	Saint Georges	May-1999		H
20	Kourou	Des Pères River	Apr.-1999		H
			May-2000		H
21	Maroni	Petit Inini River	Jan.-1999	GM	H
			Jun.-1999	GM	H
22	Sinnamary	Deux Roros	Jun.-1998		H
			Nov.-1998		L
23	Sinnamary	Saulnier	Jun.-1998	DAM	H
			Dec.-1999	DAM	H
24	Maroni	Inini River	Jan.-1999	GM	H
			Jun.-1999	GM	H
25	Sinnamary	Takari Tanté	Jun.-1999		H
			Nov.-1998		L
26	Maroni	Tampock River	Jan.-1999		H
			Jun.-1999		H
27	Sinnamary	Venus	Jun.-1998	DAM	H
			Dec.-1998	DAM	H

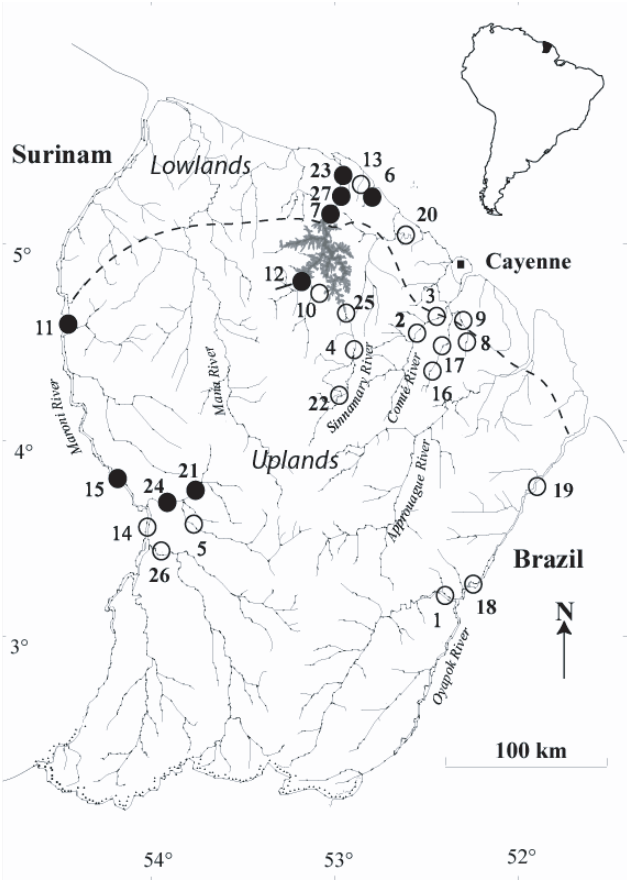

Fig. 1. Sampling reaches in French Guiana. The black area represents the Petit Saut reservoir. Disturbed reaches are indicated by filled circles. See Table 1 for identification of sampled reaches.

The reference reaches were those with no obvious signs of environmental disturbance or those with only minor signs of disturbance (18 reaches, 36 samples). The disturbed samples (17 samples, 9 reaches) were those with obvious environmental disturbance from gold mining, dam construction, urbanization or industrial activities (Table 1).

2.2 Environmental variables

To determine physical features for the rivers sampled, we followed the hierarchical linear spatial scale proposed by Imhof et al. (1996) for charactering watersheds. The reach hierarchy (10^1–10^4 m) was chosen for evaluation and measurement of qualitative and quantitative variables at a local scale.

In each of the 53 sampling occasions, 9 channel transects were established along an 800 m reach.

At each transect, local physical and biological habitat variables were noted. These were *riparian vegetation height* (low, medium, high), *riparian vegetation cover* (channel width covered by overhanging vegetation), presence/absence of *macrophytes* and *floating plant debris* and *channel width* (Rangefinder Yardage Pro Bushnell). *Substrate* type (rock, pebble, sand, mud, litter, or coarse plant debris), *channel depth* (Digital sounder Speedtech SM-5, Honda Electronics) and *flow velocity* (Flow meter General Oceanic, Mod. 2030) were determined at three or five points along each transect, depending on channel width. At the center of each reach, at one and two meter depths *dissolved oxygen*, *conductivity*, *water temperature* (YSI, mod. 85), *pH* (pH meter, WTW France, mod. pH330), *water transparency* (Secchi disk), and *turbidity* (turbimeter, Lamotte, mod. 2008) were measured. We also noted the tidal influence on the reach, the season based on water level at time of sampling and water depth at each gillnet location. The "sub-watershed" (10^4–10^8 m) and "watershed" (10^5–10^{10} m) hierarchy (Imhof et al. 1996) were chosen for measures of quantitative and qualitative variables at the basin scale. In this category 2 variables were included relative to the position of the reach in the basin: *distance from river-mouth* and *drainage area upstream of the reach*. The latter is considered as a rough indicator of mean annual discharge at the reach. *Distance from the river-mouth* was measured using Autocad Map (version 2.0) software and *drainage area upstream of the reach* was estimated on a map of French Guiana (IGN 1/500 000) using a planimeter. For more details about the sampling protocol for environmental variables, see Tejerina-Garro and Mérona (2001). These variables were subsequently transformed into a limited number of synthetic variables (Table 2).

2.3 Fish sampling and fish assemblage descriptors

Fish assemblages were sampled following a standardized sampling protocol described by Tejerina-Garro and Mérona (2000). In the same 800 m reach selected for sampling habitat variables, we deployed four batteries of 5 gill nets (25 m long by 2 m deep with mesh size from 15 to 35 mm) from 5 p.m. to 7 a.m.

From the fish assemblages data we computed three types of descriptors: global, taxonomic and trophic (Table 3).

The four global descriptors were:

- Species richness (*SR*), is a function of the size of the sample (the number of specimens collected) (Magurran 1988). This parameter was estimated by the residuals of the log-linear relationship between species number and number of specimens. This relationship was calculated for each individual basin from a number of collections by the Laboratory of Hydrobiology of the Institut de Recherche pour le Développement, Centre de Cayenne, between 1994 and 2000.
- The Simpson index of diversity $Is = 1/\Sigma p_i^2$ where p_i: relative abundance of species i in the sample.
- Equitability $Es = (Is-1)/(SR-1)$ where SR is the sample's species richness.
- The total number of specimens collected.

The taxonomic descriptors were the relative number of specimens and the relative number of species of the major fish families. Taxonomic classification is based on morphological characters and it is likely that form is linked to some aspects of the ecological role of species in the ecosystem (Matthews 1998). The families included in the analyses were the Auchenipteridae (water column Siluriformes), Characidae, Cichlidae, Curimatidae, Anostomidae, Pimelodidae, Ageneiosidae, Doradidae, Sternopygidae, Gymnotidae and Hypopomidae. Because some of these families were poorly represented in

Table 2. Quantitative and qualitative environmental variables considered for index development.

Variable	Code	Description		Range	Transformation
Drainage basin upstream station	DRAIN	km^2		72–57 690	$\log(x)$
Distance to river mouth	DIST	km		5–359	$x^{0.2}$
Tidal influence	TIDE	0 = no; 1 = yes			binary coded
Season	SEAS	0 = low water; 1 = high water			binary coded
Channel width	WIDTH	mean of 9 measures, m		8–356	$\log(x)$
Vegetation cover	COV	mean width of the river covered by overhanging vegetation, m		0–59	$x^{-0.5}$
Substrate diversity	SUBDIV	Simpson diversity coefficient		2.4–6.0	$x^{0.5}$
Flow velocity	FLOW	mean of 9 measures(*), m s^{-1}		2–94	none
Channel depth	DEPTH	mean of 9 measures(*), m		1.5–17.0	$\log(x)$
Depth at gillnets	NETDEPTH	mean depth at gillnet locations, m		1.5–6.3	$\log(x)$
Conductivity	COND	$\mu S\ cm^{-1}$		15.8–441	none
Dissolved oxygen	OXY	$g\ L^{-1}$	mean of two	1.1–9.0	none
pH	PH		measures: at	5.0–6.8	none
Water transparency	TRANS	cm	8:00 and 18:00	20–198	$x^{0.4}$
Water turbidity	TURB	UTN		1.5–57.0	$x^{0.2}$
Water temperature	TEMP	°C		24.2–30.9	none
Macrophytes	MACR	0 = absent; 1 = present			binary coded
Debris	DEBR	% of points from 18 observations			none
Riparian vegetation height					
Low	VLO			0–50	none
Medium	VME	% of points from		0–94	none
High	VHI	18 observations		0–94	none
Channel substrate					none
rock	ROC			0–36	none
pebble	PEB			0–33	none
sand	SAN	% of points		0–61	none
litter	LIT	from 27 or 45		0–44	none
mud	MUD	observations		0–23	none
clay	CLA	depending on		0–12	none
trunks, branches	TRU	channel width		0–35	none

(*) in the middle of the channel

our samples we grouped together the three benthic Siluriform families (Pimelodidae, Ageneiosidae and Doradidae) and the three Gymnotiform families (Sternopygidae, Gymnotidae and Hypopomidae), which exhibit similar form and behavior.

Trophic descriptors included the number of specimens and the number of species in diet categories determined from previous stomach contents analysis (Tejerina-Garro 2001; Mérona et al. 2003, 2005). These categories were detritivores (organic layer and detritus), piscivores, terrestrial invertivores, aquatic invertivores, herbivores (remains of higher plants) and omnivores (similar proportions of plants and animals).

Biological descriptors and environmental variables were transformed to approach as much as possible a normal distribution (Tables 2 and 3).

2.4 Selection of fish assemblage metrics

The selection of fish assemblage metrics proceeded in two steps:

- We computed multilinear regressions between fish assemblage descriptors and environmental variables from the reference samples (36). To reduce the cases of co-linearity (Oberdorff et al. 1995) we used a forward stepwise selection of the environmental variables. We obtained a model of the form:

$$D = aP_1 + bP_2 + \ldots + xP_n + \text{error}$$

where D: descriptor of the fish assemblage
$P_1, P_2, \ldots, P_n$: environmental variables
a, b and x: regression coefficients
error: residual of the relation.

Environmental variables likely to be modified by human disturbances typically encountered in French Guiana were excluded before modeling the relationships between fish descriptors and environment variables of reference samples. In practice, that means excluding the physico-chemical parameters (water conductivity, pH, water transparency, turbidity and temperature). We checked a posteriori the independence of the two samples taken at the same station in computing correlations between the residuals calculated for these two samples. The results showed that for every descriptor but one (the number of Cichlids) this correlation was non significant.

- We compared the residuals of the preceding relations between reference samples and disturbed ones using t-tests. Residual values of descriptors that displayed statistical differences between the two groups were selected as metrics for inclusion in the final index.

Table 3. Fish assemblage descriptors tested for relationships with environmental variables.

Descriptors			
Type	Name	Code	Transformation
Global	Abundance	NUM	log(x)
	Species richness	SR	$x^{0.6}$
	Simpson diversity	DI	log(x)
	Equitability	EQ	$x^{0.6}$
Taxonomic relative abundance	Anostomidae	ANO	$x^{0.3}$
	Auchenipteridae	AUC	$x^{0.2}$
	Characidae	CHA	none
	Cichlidae	CIC	binary coded
	Curimatidae	CUR	scale
	Gymnotiformes	GYM	binary coded
	Benthic Siluriformes	SIL	$x^{0.3}$
Taxonomic relative richness	Anostomidae	ANOr	$x^{0.7}$
	Auchenipteridae	AUCr	$x^{0.5}$
	Characidae	CHAr	none
	Curimatidae	CURr	binary coded
	Benthic Siluriformes	SILr	$x^{0.8}$
Trophic relative abundance	Detritivores	DET	$x^{0.5}$
	Herbivores	HEB	log(x + 1)
	Piscivores	PIS	$x^{0.2}$
	Aquatic invertivores	AINV	scale
	Terrestrial invertivores	TINV	$x^{0.6}$
	Omnivores	OMN	$x^{0.4}$
Trophic relative richness	Detritivores	DETr	none
	Herbivores	HEBr	none
	Piscivores	PISr	log(x + 1)
	Aquatic invertivores	AINVr	binary coded
	Terrestrial invertivores	TINVr	log(x + 1)
	Omnivores	OMNr	$x^{1.2}$

2.5 Index development and metric notation

To homogenize the weight of metrics, each metric of the reference samples was standardized. The transformed distribution has a mean of 0 and a standard deviation of 1. The same procedure was used for samples from the disturbed reaches by using the values of the mean and standard deviation calculated from the reference samples. Then the standardized residuals were compared to the student distribution related to unilateral tests and a score was attributed accordingly (Table 4) (see Oberdorff et al. 2002 for details). This procedure means that 1) all metrics will vary between 0 and 5, 2) all metrics will have the same response to disturbance (i.e. a decrease) and 3) the scores are associated with the probabilities of type I error. For example, the probability of obtaining a score of 0 in a reference sample is 0.05. The final index score is the sum of the scores attributed to each metric.

Table 4. Metrics scoring based on the Student-t distribution.

Metric changes under disturbance	Class interval	Risk (%)	Score
Decreasing	>−0.67	>25	5
	]−0.67; −0.84]	20−25	4
	]−0.84; −1.04]	15−20	3
	]−1.04; −1.28]	10−15	2
	]−1.28; −1.65]	5−10	1
	<−1.65	<5	0
Increasing	<0.67	>25	5
	]0.67; 0.84]	20−25	4
	]0.84; 1.04]	15−20	3
	]1.04; 1.28]	10−15	2
	]1.28; 1.65]	5−10	1
	>1.65	<5	0

2.6 Interpretation of the index

A tentative interpretation of the index values was done by regressing the score of every metric against the environmental parameters supposed to be directly influenced by various disturbances and not included in the reference models (water conductivity, pH, water transparency and water temperature). A stepwise forward regression was used.

2.7 Index validation

To validate the index, we used time series data (December 1991 – December 2000) from a station located 30 km downstream from the Petit-Saut Dam, which included samples before and after dam closure (Mérona and Albert 1999; Mérona et al. 2005).

All statistical analyses were performed with the software SYSTAT®.

Table 5. Statistics of the linear regressions between descriptors and environmental variables. See Table 2 for variables codes and definitions.

Descriptor	Variable entered in model (sign of the effect)	*p*
Total number of individuals		0.062
Richness		0.086
Equitability	COV(+), VLO(+)	0.022
Simpson Index	COV(+), DEBR(-), VLO(+)	0.001
Taxonomic composition: Number of individuals		
Anostomidae	DRAIN(-),DEBR(-),DIST(+),SUBDIV(-),MACR(-),MUD(+),VLO(+)	0.000
Auchenipteridae	DEBR(+), DIST(-), SUBDIV(+), DEPTH(+)	0.000
Characidae	DEBR(+), SUBDIV(+), WIDTH(+), SAN(+), MUD(-), VLO(+)	0.000
Cichlidae	ROC(-), SEAS(-)	0.003
Curimatidae	LIT(-), SEAS(-)	0.000
Gymnotiformes	SUBDIV(-), WIDTH(+), TIDE(+), NETDEPTH(-), DEPTH(-), SAN(-)	0.000
Benthic Siluriformes	COV(+), MACR(+), NETDEPTH(-), PEB(+), SAN(-), VME(-)	0.001
Taxonomic composition: Number of species		
Anostomidae	DRAIN(-), FLOW(+), SUBDIV(-), NETDEPTH(-), ROC(+), PEB(+)	0.000
Auchenipteridae	DIST(-), TIDE(+), PEB(+), SEAS(+)	0.000
Characidae	FLOW(+), DEBR(-), DIST(+), DEPTH(+), PEB(-), SAN(+), SEAS(+)	0.000
Curimatidae	DEBR(-), DIST(+), SAN(-), MUD(-)	0.000
Benthic Siluriformes	NETDEPTH(-), MUD(-)	0.003
Trophic composition: Number of individuals		
Detritivores	DRAIN(+), SUBDIV(-), WIDTH(-), SAN(-), SEAS(-)	0.000
Herbivores	DEBR(-), DEPTH(-), VHI(-)	0.001
Piscivores	COV(+), SUBDIV(+), MACR(+), LIT(+)	0.000
Aquatic invertivores	COV(+), SUBDIV(+), SEAS(-), VLO(+)	0.001
Terrestrial invertivores	DRAIN(-), DEBR(+), SUBDIV(+), WIDTH(+), TIDE(+), SAN(+), MUD(+)	0.000
Omnivores	DEBR(-), SUBDIV(-), TIDE(-), SAN(-), MUD(+), TRU(+), SEAS(+)	0.000
Trophic composition: Number of species		
Detritivores	PEB(+), SEAS(-), VME(+)	0.000
Herbivores	MACR(+), TIDE(-)	0.009
Piscivores	DEBR(+), SUBDIV(+), TIDE(+), DEPTH(+), LIT(+), TRU(-)	0.000
Aquatic invertivores	COV(+), ROC(-), LIT(-), VLO(+)	0.000
Terrestrial invertivores	FLOW(+)	0.001
Omnivores	DRAIN(+), COV(-), SUBDIV(-), PEB(+), VME(-)	0.009

3 Results

A total of 124 fish species were collected. The number of species in the samples varied between 3 and 42 (Appendix).

The only descriptors that did not show any significant relationship with environmental variables were the total number of individuals and species richness (Table 5). Complex relationships were detected between environmental variables and the other descriptors. The variables that most influenced fish assemblages were: substrate diversity, presence of debris, season, presence of sand, and riparian vegetation cover. Substrate diversity had a positive effect on the Auchenipteridae, Characidae, piscivores, and invertivores whereas the effect was negative on Anostomidae, Gymnotiformes, detritivores, and omnivores. Relative abundance of Auchenipteridae, Characidae and terrestrial invertivores along with relative richness of piscivores were favoured by the presence of debris, whereas the opposite effect was observed for assemblage diversity, relative number of Anostomidae, herbivores and omnivores, and relative richness of Characidae and Curimatidae. Season had a large influence on assemblage composition. The low water season favored the number of Cichlidae, Curimatidae, detritivores and aquatic invertivores while the high water season favored the number of omnivores and richness of Auchenipteridae and Characidae. Benthic fish were negatively affected by sandy substrate as shown by the negative effect of this variable on Gymnotiformes, benthic Siluriformes, Curimatidae and detritivores. Finally, greater riparian vegetation cover increased the equitability and diversity of fish assemblages and most of its effects on individual taxonomic or trophic categories were positive.

For 9 fish assemblage descriptors the mean residuals of the regression between descriptor and habitat were significantly different in the disturbed and reference samples (Table 6). Total number of individuals, diversity, relative number of piscivores, aquatic invertivores and terrestrial invertivores, and species richness of detritivores were lower in the disturbed samples, whereas, the number of individuals and the species richness of Anostomidae together with the species richness of omnivores were higher.

As expected, classification of samples by their index values produced a generally consistent match with our *a priori* separation into disturbed and reference reaches (Table 7 and Fig. 2). Supposed disturbed reaches exhibited values equal or lower than 35 (mean = 25.7), with the exception of one. Although some samples from reference reaches showed low index values, the mean value for reference reaches was significantly higher than that of disturbed reaches (mean = 36.0; $t = 4.78$; $p < 0.001$).

Table 6. Statistics of the comparisons between disturbed and reference samples. Significant differences are in bold.

	Reference		Disturbed		p
Metric	Mean	SD	Mean	SD	
Total Number	0.364	0.044	0.305	0.073	**0.003**
Equitability	0.002	0.126	0.021	0.148	0.679
Simpson Index	0.024	0.187	−0.131	0.276	**0.023**
Richness	3.969	1.086	3.683	1.439	0.237
Taxonomic composition: Number of individuals					
Anostomidae	0.008	0.326	0.417	0.615	**0.009**
Auchenipteridae	0.020	0.209	0.033	0.39	0.450
Characidae	0.012	11.903	−2.566	29.298	0.365
Cichlidae	0.002	0.409	0.009	0.595	0.484
Curimatidae	0.000	0.51	0.303	0.809	0.086
Gymnotiformes	0.010	0.297	0.108	0.590	0.263
Benthic Siluriformes	0.005	0.345	−0.187	0.840	0.188
Taxonomic composition: Number of species					
Anostomidae	0.080	1.269	1.924	2.338	**0.003**
Auchenipteridae	−0.005	0.593	0.327	0.902	0.090
Characidae	0.002	5.089	−3.183	14.774	0.200
Curimatidae	0.021	0.304	−0.159	0.596	0.127
Benthic Siluriformes	−0.001	2.693	0.287	5.617	1.422
Trophic composition: Number of individuals					
Detritivores	0.002	1.337	0.162	2.016	0.385
Herbivores	−0.003	0.336	0.029	0.344	0.381
Piscivores	0.004	0.212	−0.231	0.435	**0.024**
Aquatic invertivores	0.006	0.65	−0.392	0.698	**0.029**
Terrestrial invertivores	0.013	2.148	−1.880	4.212	**0.048**
Omnivores	−0.010	0.78	0.127	1.752	0.380
Trophic composition: Number of species					
Detritivores	−0.007	5.267	−5.550	11.136	**0.033**
Herbivores	0.001	3.911	2.604	6.894	0.081
Piscivores	0..010	0.093	−0.072	0.269	0.119
Aquatic invertivores	0.004	0.323	−0.051	0.549	0.352
Terrestrial invertivores	−0.006	0.131	−0.400	1.072	0.075
Omnivores	−0.003	9.042	20.38	20.175	**0.000**

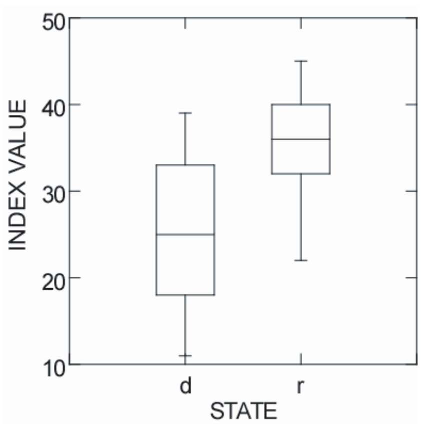

Fig. 2. Box plot of index values as a function of state of disturbance; d: disturbed; r: reference.

Among the environmental variables related to water quality, only conductivity indicated a significant change with perturbations (Table 8). However the changes were not uniform for the different types of perturbation. Downstream from the dam, conductivity and temperature were high. Reaches disturbed by gold mining were characterized by low water transparency and high conductivity, and the reach with industrial effluents had high transparency. Linear regressions of the sensitive metrics against the physico-chemical variables generated few relationships (Table 9). Dissolved oxygen had a positive effect on total abundance in the samples. Terrestrial invertivores were negatively affected by conductivity, temperature and transparency. The number of Anostomidae increased with temperature and the species richness of omnivores increased with conductivity and decreased with pH.

The index showed a clear sensitivity to Petit-Saut dam closure (Fig. 3, Table 10). The mean index value decreased significantly soon after closure, during the two years of reservoir filling ($p < 0.001$), and remained lower than in the pre-closure period in subsequent years ($p = 0.023$). Two metrics included in the index were particularly affected by dam closure: the number of captured individuals and the relative abundance of terrestrial invertivores.

4 Discussion

Elaboration of preservation programs using biotic indices requires a method integrating biotic responses through analysis

Table 7. Sample scores of individual metrics and total scores for the samples. Samples considered disturbed by human activities are in bold. See Table 3 for metrics codes and definitions. r: reference, d: disturbed sample.

Stations	State of disturbance	Metrics									Index value
		NUM	DI	ANO	PIS	AINV	TINV	ANOr	DETr	OMNr	
comA0499	r	5	5	5	5	5	5	5	5	5	45
roro0698	r	5	5	5	5	5	5	5	5	5	45
comB0499	r	5	5	5	5	5	5	5	5	5	45
tamp0699	r	5	5	5	5	5	5	5	5	5	45
taka1198	r	5	5	4	5	3	5	5	5	5	42
oraB1298	r	3	5	4	5	5	5	5	5	5	42
oraA1298	r	5	5	5	5	5	1	5	5	5	41
oraA0798	r	5	5	5	5	5	5	5	0	5	40
comB1098	r	5	5	5	5	5	5	5	0	5	40
pere0499	r	5	5	5	5	5	5	5	5	0	40
taka0699	r	5	5	5	5	5	5	5	5	0	40
gini0199	r	5	5	5	5	5	5	5	5	0	40
marB0199	**d**	**5**	**5**	**4**	**5**	**5**	**5**	**0**	**5**	**5**	**39**
oyaA0599	r	5	3	5	5	5	5	5	0	5	38
camo0599	r	5	3	5	5	5	0	5	5	5	38
marA0199	r	5	5	5	5	5	3	5	5	0	38
dalle1199	r	5	5	2	5	5	5	0	5	5	37
kouA1298	r	1	5	3	5	3	5	5	4	5	36
pere0500	r	5	5	5	5	5	0	1	5	5	36
saul0698	r	5	5	5	5	5	1	5	5	0	36
kouA0798	r	0	5	5	5	5	5	5	0	5	35
kouB1298	r	5	5	5	5	5	5	5	0	0	35
kour0699	r	5	5	5	5	5	5	5	0	0	35
roro1198	r	5	0	5	5	5	0	5	5	5	35
son0199	**d**	**5**	**5**	**5**	**5**	**5**	**5**	**0**	**5**	**0**	**35**
dalle0699	r	5	5	0	5	5	4	5	5	0	34
kour1298	r	5	5	5	5	5	5	3	0	0	33
lebl0699	**d**	**5**	**5**	**3**	**5**	**5**	**5**	**0**	**5**	**0**	**33**
camo1198	r	5	2	4	5	2	5	5	0	5	33
lang0699	**d**	**5**	**5**	**0**	**5**	**5**	**3**	**5**	**0**	**5**	**33**
lang1099	**d**	**5**	**5**	**0**	**5**	**3**	**0**	**5**	**5**	**5**	**33**
tamp0199	r	5	5	5	5	5	0	2	5	0	32
comA1098	r	5	5	5	5	5	2	5	0	0	32
karo0500	**d**	**5**	**5**	**5**	**5**	**5**	**5**	**1**	**0**	**0**	**31**
saul1299	r	5	5	5	5	5	1	5	0	0	31
kouB0798	r	0	5	5	5	5	1	0	5	5	31
malm0499	r	5	5	3	5	5	0	2	5	0	30
marA0699	r	4	0	5	5	5	5	1	0	5	30
oyaB0599	r	5	5	0	5	5	5	0	0	5	30
ven0698	**d**	**5**	**5**	**5**	**5**	**5**	**0**	**0**	**5**	**0**	**30**
malm0500	r	5	2	5	5	5	0	5	0	0	27
pini0199	**d**	**5**	**5**	**5**	**5**	**5**	**2**	**0**	**0**	**0**	**27**
gini0699	r	5	5	5	5	5	0	1	0	0	26
son0699	**d**	**5**	**5**	**5**	**5**	**5**	**0**	**0**	**0**	**0**	**25**
marB0699	**d**	**4**	**0**	**5**	**5**	**5**	**5**	**1**	**0**	**0**	**25**
pini0699	**d**	**5**	**3**	**5**	**5**	**5**	**2**	**0**	**0**	**0**	**25**
ven1298	**d**	**5**	**4**	**0**	**4**	**0**	**0**	**0**	**5**	**5**	**23**
oraB0798	r	5	2	1	5	3	5	1	0	0	22
lebl1298	**d**	**1**	**2**	**3**	**5**	**2**	**5**	**0**	**0**	**0**	**18**
kere0799	**d**	**1**	**3**	**1**	**2**	**5**	**0**	**5**	**0**	**0**	**17**
karo0499	**d**	**5**	**2**	**0**	**5**	**5**	**0**	**0**	**0**	**0**	**17**
kere1299	**d**	**4**	**0**	**0**	**5**	**2**	**0**	**5**	**0**	**0**	**16**
karo0598	**d**	**5**	**0**	**0**	**5**	**1**	**0**	**0**	**0**	**0**	**11**

Table 8. Comparison of selected water quality parameters in reference and disturbed samples.

Sample variable	Reference (36)	Disturbed (17)	Type of perturbation: Dam (4)	Gold mining (6)	Diffuse (4)	Indus. Efflu. (3)
	Mean (SD)		Range			
Oxygen	6.3 (1.4)	5.9 (1.8)	4.2−6.8	5.8−7.9	5.8−8.2	1.1−5.5
Conductivity	26.6 (6.6)	31.6 (5.2)**	32.1−35.9	25.9−39.3	24.8−29.8	22.2−35.6
Temperature	25.8 (1.3)	26.5 (1.6)	27.0−27.9	24.6−25.5	26.0−30.9	25.6−27.8
Transparency	91 (38)	80 (55)	83−140	25−40	35−85	140−170

** significant difference at $p < 0.05$.

Table 9. Statistics of the linear regressions between the metrics composing the index and physicochemical variables. See Table 2 for codes and definitions. Significant regressions at $p < 0.05$ are in bold.

Descriptor	Relation (sign of the effect)	*p*
Total Number	OD(+)	**0.002**
Diversity		0.416
Piscivores		0.058
Aquatic invertivores		0.065
Terrestrial invertivores	COND(-), TEMP(-), TRANS(-)	**0.000**
Anostomidae	TEMP(+)	**0.025**
Richness Anostomidae		0.397
Richness detritivores		0.067
Richness omnivores	COND(+), pH(-)	**0.007**

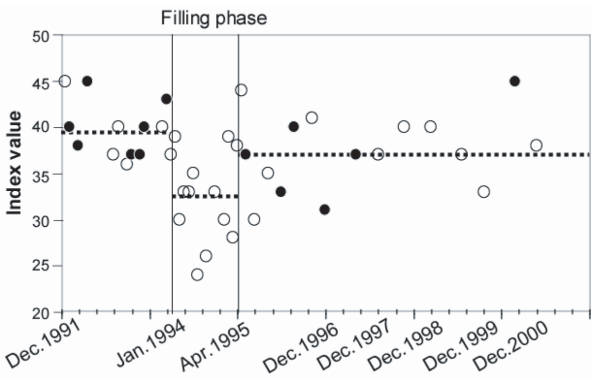

Fig. 3. Temporal changes in index values at Venus reach, downstream of the Petit Saut Dam in the Sinnamary River. The dotted lines represent the index mean in the three successive phases of the system evolution: pre-closure, filling and equilibration. Open circles indicate low water.

of patterns and process from local to ecosystem levels (Barbour et al. 1995). In this context, use of a multimetric approach has been relevant (Karr 1981). This approach takes into account the various aspects of the fish assemblages, which present different responses vis-à-vis the activities at the origin of disturbance (Hughes and Noss 1992; Simon and Lyons 1995) and gives an image at community, ecosystem and regional levels of ecosystems (Miller et al. 1988).

Fish-habitat relationships

A fundamental issue in developing biological indices of water course quality is to define reference conditions (Hughes 1995). There are two main difficulties in doing so. First, in many parts of the world, pristine conditions of rivers are rare and reference sites have to be defined as the least affected ones. Second, fish assemblages are subject to natural temporal and spatial variability, induced in part by aquatic habitat variability (Matthews 1998). In most of French Guiana, a very low human population density yields reaches lacking human interference (at least recently). Using data from these reaches, we were able to establish relationships between fish assemblages and their habitats. The relations we found were complex and it is outside the scope of this study to interpret in detail the influence of the aquatic habitat on the fish assemblage. However, our results were generally consistent with the available literature (see Matthews 1998, for a review of fish assemblage response to habitat variability). Variables most frequently included in the multilinear regressions between habitat and fish assemblage descriptors were those related to structure and diversity of substrate, structure of riparian vegetation, presence of plant debris or macrophytes and season. The nature of the substrate, associated with depth and flow velocity, has been considered a good predictor of the collective properties of fish assemblages (Meffe and Sheldon 1988; Sheldon and Meffe 1995). Riparian vegetation cover, along with the presence of debris and macrophytes have been repeatedly shown to affect fish distributions (Matthews and Hill 1979; Angermeier and Karr 1984; Sechnik et al. 1986; Penczak et al. 1994; Tait et al. 1994; Beechie and Sibley 1997; Mérigoux et al. 1998; Thévenet and Statzner 1999). Seasonal variability of fish assemblages in large rivers, although poorly documented, is generally recognized (Reash 1999). It can be the consequence of the dynamics of constituent populations (Welcomme 1979), fish movements (Winemiller and Jepsen 1998) or sampling efficiency related to modifications in water level. Due to this influence of season on the fish/habitat relationship, we recommend that water quality surveys sample fish in the same season. The more hydrologically stable, low water season is the most appropriate (Bozzetti and Schulz 2004; Pinto et al. in press; Plafkin et al. 1989).

Metric selection

Fish assemblages metrics included in river quality indices were generally chosen based on pre-existing detailed knowledge of fish species ecology in the considered region (Karr 1981; Barbour et al. 1995). The first formulation of the fish index of biotic integrity (Karr 1981) included various elements specific to the ichthyofauna of the region where it was first applied. Subsequently this index has been applied to other regions by adapting the initial metrics to differing taxa (Hughes and Oberdorff 1999). However modifying metrics for a species

Table 10. Index score of each metric at Venus station, Sinnamary River, between December 1991 and December 2000. Samples collected during the filling phase of the reservoir are in bold. See Table 3 for metric codes and definitions.

Date of sample	Descriptors									Index value
	NUM	DI	ANO	PIS	AINV	TINV	DETr	OMNr	ANOr	
Dec. 1991	5	5	5	5	5	5	5	5	5	45
Feb. 1992	5	5	5	5	5	5	5	3	2	40
Apr. 1992	5	5	5	4	5	5	4	0	5	38
Jun. 1992	5	5	5	5	5	5	5	5	5	45
Dec. 1992	5	1	5	2	5	5	4	5	5	37
Jan. 1993	2	5	5	5	5	5	3	5	5	40
Mar. 1993	5	5	0	5	5	1	5	5	5	36
Apr. 1993	5	3	3	2	5	5	4	5	5	37
Jun. 1993	5	2	5	5	5	2	3	5	5	37
Jul. 1993	5	5	5	5	5	5	5	5	0	40
Nov. 1993	0	5	5	5	5	5	5	5	5	40
Dec. 1993	5	5	5	5	5	5	5	5	3	43
Jan. 1994	**0**	**5**	**3**	**5**	**5**	**5**	**4**	**5**	**5**	**37**
Feb. 1994	**4**	**5**	**5**	**5**	**5**	**2**	**3**	**5**	**5**	**39**
Mar. 1994	**0**	**5**	**0**	**5**	**5**	**0**	**5**	**5**	**5**	**30**
Apr. 1994	**0**	**5**	**5**	**5**	**5**	**1**	**5**	**5**	**2**	**33**
May 1994	**3**	**0**	**5**	**5**	**5**	**0**	**5**	**5**	**5**	**33**
Jun. 1994	**0**	**4**	**5**	**5**	**5**	**1**	**5**	**5**	**5**	**35**
Jul. 1994	**2**	**0**	**5**	**0**	**0**	**5**	**5**	**5**	**2**	**24**
Sep. 1994	**0**	**3**	**5**	**1**	**5**	**2**	**5**	**5**	**0**	**26**
Nov. 1994	**1**	**5**	**5**	**0**	**5**	**2**	**5**	**5**	**5**	**33**
Jan. 1995	**1**	**1**	**5**	**1**	**5**	**2**	**5**	**5**	**5**	**30**
Feb. 1995	**1**	**5**	**5**	**5**	**5**	**3**	**5**	**5**	**5**	**39**
Mar. 1995	**2**	**1**	**5**	**0**	**5**	**0**	**5**	**5**	**5**	**28**
Apr. 1995	5	2	5	1	5	5	5	5	5	38
May 1995	4	5	5	5	5	5	5	5	5	44
Jun. 1995	4	5	5	1	5	5	5	5	2	37
Ago. 1995	0	5	1	3	5	1	5	5	5	30
Nov. 1995	5	0	5	0	5	5	5	5	5	35
Feb. 1996	5	5	1	0	5	5	5	5	2	33
May 1996	5	5	5	0	5	5	5	5	5	40
Sep. 1996	5	2	5	4	5	5	5	5	5	41
Dec. 1996	5	1	5	0	5	5	5	0	5	31
Jul. 1997	3	5	5	5	5	5	4	5	0	37
Dec. 1997	5	3	5	0	5	5	4	5	5	37
Jun. 1998	5	5	5	0	5	5	5	5	5	40
Dec. 1998	5	5	5	0	5	5	5	5	5	40
Jul. 1999	5	4	5	0	5	5	5	5	3	37
Dec. 1999	2	5	3	1	5	2	5	5	5	33
Jul. 2000	5	5	5	5	5	5	5	5	5	45
Dec. 2000	5	2	5	1	5	5	5	5	5	38

rich ichthyofauna for which little ecological information exists is difficult if not impossible. Like Hughes et al. (1998, 2004), McCormick et al. (2001) and Mebane et al. (2003), our method allowed selection of metrics without any a priori notion of which one could be affected by disturbances. In this study, 2 general metrics (total number of captured fish and diversity), 2 taxonomic metrics (number of individuals and number of species of Anostomidae) and 5 trophic metrics (number of individuals as piscivores, aquatic invertivores and terrestrial invertivores, number of species of detritivores and omnivores) were sensitive to human-caused disturbances.

The observed decrease in richness and abundance scores with disturbance was expected.

Fish abundance is a common surrogate for system productivity, and disturbed sites usually support fewer fish than high quality sites (Hughes and Oberdorff 1999). A decrease in assemblage diversity following ecosystem stress has been reported in many occasions (Tramer and Rogers 1973; Kushlan 1976; Kaestler and Herricks 1977; Sheldon 1988; Gray 1989; Travnichek and Maceina 1994; Lae 1995; Penczak and Gomes 2000). Because diversity indices integrate the number of species in the assemblage and the proportion of individuals per species, the metric based on diversity was excluded from the index elaborated by Karr (1981). However, for the same reason, this metric could be more sensible to disturbance. This seems to be the case in our data set where richness and evenness did not show any significant decrease with perturbation whereas the diversity index did.

The Anostomidae was the only family reacting to disturbances with an increase in number of individuals and number of species. Thirteen species of Anostomidae are present in French Guiana (Planquette et al. 1996). They are, in general, water column species, omnivores with trends to herbivory, eating leaves and fruits (Santos et al. 1984; Boujard et al. 1990; Planquette et al. 1996). This feeding habit favors this family in disturbed environments. Indeed, the ability to use marginal vegetation when available and to take advantage of other food resources when the exogenous supply becomes scarce, frees these species from prey scarcity caused by water quality disturbances. Moreover, according to Karr (1981), a disturbed aquatic environment can support a large set of omnivores. This is possible because, when the specific components of basic food become rare because of the disturbance, the opportunist characteristics of omnivores make them more adaptable than specialist fish (Karr et al. 1986). We also found that the number of omnivore species increased in disturbed environments.

Conversely, most specialized consumers showed a significant decrease in relative abundance (piscivores, terrestrial invertivores and aquatic invertivores) or relative richness (detritivores) in disturbed reaches. Piscivores are frequent in habitats where the water is transparent and conductive. In French Guiana rivers, this trophic group is formed mainly by visual predators, which makes them particularly sensitive to decreased water transparency according to the model of Rodriguez and Lewis (1997).

The diet of aquatic invertivores is based on benthic invertebrates, which are affected by many human-induced disturbances like substrate modification, flow regulation, or sewage (William and Winget 1979; Ward and Stanford 1979, 1982; Statzner et al. 2001). Terrestrial invertivores depend on terrestrial arthropods falling from the riparian vegetation (Horeau et al. 1996). Thus a reduced canopy cover induced either by a reduction in peak flows or by degradation of riparian vegetation is detrimental to these species. Detritivores, by our definition, are those species feeding upon the substrate organic layer. This includes periphytic algae, whose development is limited by low transparency and substrate siltation (Power 1984).

Sensitivity of the proposed index

Due to the method we used to select metrics, separation of disturbed and reference samples should be good if not perfect. However some samples appeared to be incorrectly positioned in the index scale. There are various reasons for this result. First, the index integrates 9 metrics, which should react differently to different disturbances. Second, our reach classification into disturbed and reference could have been erroneous in some cases. For example, the sample taken in the Maroni River downstream from Maripasoula in January 1999, was considered disturbed by the presence of the city, but received a high index score. At the time of sampling, the annual flooding had begun in the region and it is likely that urban pollution was diluted by the high discharge (Pinto et al. in press). Last, certain reference reaches could have suffered natural perturbations, leading to negative impacts on their fish assemblages. This was probably the case with the Malmanoury River, a small coastal creek, which suffered a severe drought in the year preceding the May 2000 sample.

Tentative interpretation of index variability

Although the small number of samples precludes a statistical analysis of the changes in physico-chemical parameters, in the disturbed reaches relative to the origin of the perturbation, some trends were revealed by the data (Table 8). Downstream of the Petit-Saut dam, as in similar situations in tropical areas (Imevbore 1970; Mérona et al. 1987), water temperature, conductivity and transparency increased. Changes in water quality of impounded rivers are extremely diverse, depending on overlapping factors like local climate, morphometry and stratification dynamics of the reservoir (Hannan et al. 1979; Petts 1984). In the case of the Petit-Saut Dam, changes in parameters resulted from internal reservoir processes that were passed downstream by the release of water: heating of superficial layers of water, mineralization of organic material and sedimentation of transported solids (Richard 1996; Richard et al. 1997).

Reaches disturbed by gold mining had high conductivity, and low temperature and transparency. In French Guiana, a common technique to extract gold consists of removing large amounts of soil for the separation of the precious metal (Marteau 2001). The result is the incorporation of large quantities of suspended material in the rivers which increases conductivity, diminishes transparency and the development of periphyton, phytoplankton and aquatic macrophytes. This, in turn, lowers endogenous oxygen production.

The only reach that was possibly affected by industrial effluents is located next to the Guianese Space Center of Kourou. In the absence of any specific chemical analysis of the water, it was impossible to identify toxic compounds, or even to assert the presence of effluents. However, direct observation in the field led us to suspect the existence of episodic pollution incidents. The high transparency, observed generally at that site, was probably related to its particular hydrodynamics. The Karouabo River estuary is partially closed by a sandy bar and consequently, the flow is greatly reduced in the mid-course, a situation which favors sedimentation.

Fish abundance was related to dissolved oxygen. However, we did not detect any difference in this parameter between disturbed and reference reaches, even when separating reaches by the origin of disturbance. The samples below Petit-Saut dam were taken 4 years after dam completion. Oxygen depletion was observed during the first year of reservoir filling, but this problem was rapidly solved by construction of a sill below the turbines to re-oxygenate the water and eliminate most reduced chemical compounds (Gosse and Grégoire 1997). As for the reaches disturbed by gold mining activities, they are located in small rivers with many rapids that could re-oxygenate the water, despite the large increase in turbidity.

The relative number of terrestrial invertivores was very sensitive to disturbances: 8 samples out of a total of 17 have scored 0 for this metric. It was negatively related to conductivity, temperature and transparency and affected by a large range of disturbances. Conductivity was higher in reaches affected by gold mining and those downstream from Petit-Saut Dam, whereas temperature was higher only downstream from Petit-Saut Dam and transparency higher only in the stations affected by industrial effluents.

The last relationship observed was that of omnivorous species richness with conductivity and pH. With one exception, all the samples considered disturbed had high relative numbers of omnivorous species. Logically this metric, which increases under disturbances, is the opposite of terrestrial invertivores.

Preliminary test of the index

The computation of the proposed index for the samples collected downstream of Petit-Saut Dam, before and after dam closure provided a clear image of the effect of this perturbation. Because, this was a rough preliminary approach, we did not consider a possible temporal autocorrelation. However, considering the large sample-to-sample variation, we doubt that the elimination of temporal autocorrelation would have greatly modified the results. Despite the variations in the index from one sample to another, there was a drastic decrease in score soon after dam closure and low values were observed during the whole filling phase. Afterwards, habitat conditions improved somewhat, resulting in an increase in index scores, which, nevertheless, were significantly lower than in the pre-closure phase (t-test; $p = 0.023$). Examination of individual scores of the different metrics revealed that the relative abundance of terrestrial invertivores was most affected by dam closure, followed by a decrease in the total numbers of fish captured and a decrease in relative abundance of piscivorous species. This result corroborates previous studies of the effects of the Petit-Saut Dam on fish assemblages downstream (Mérona and Albert 1999; Mérona et al. 2005): a rapid and drastic decrease in fish capture and altered fish feeding strategies.

5 Conclusion

Given the difficulties in adapting the fish IBI developed by Karr (1981) to regions where the knowledge of fish ecology is poor, this work presents a complementary method that does not require any a priori knowledge of the reaction of fish to disturbance. The method is based on relations between habitat characteristics and fish assemblages in natural (reference) situations. Evidently, this premise limits its application to regions with many pristine environments. Being based on statistical relations, the construction of an index should include many observational data and we are aware that our data set was too small for establishing a definitive and stable index of water quality. In particular, we could not check overfitting in our models because no independent data set was available to conduct a validation procedure. However, at least four considerations reinforce our confidence in this approach. First, relationships between fish assemblage metrics and environmental variables were coherent with the results of a parallel study on fish/habitat relationships using multivariate analysis (Tejerina-Garro 2001). Second, effects detected by the metrics selected were coherent with the general knowledge of changes in fish assemblages with various disturbances. Third, use of the same metrics in a dam induced perturbation detected this perturbation. Lastly, the way this index was constructed allows improvements with additional new data from research on species-environment relationships. This preliminary approach was the result of the Environment Commission of French Guiana Authority, whose objective was to set up a permanent survey of rivers that would include rivers and reaches others than those we surveyed.

Acknowledgements. This work was founded by the "Direction régionale de l'environnement" (DIREN) Guyane through the Grant 2366 CPER9498. The first author received a grant from IRD (Grant 32801A). Many thanks to Jean-Claude Bron, Roland Ruffine and Michel Tarcy for their field assistance, and to Marc Barral, Sébastien Lebossé and Frédéric Soulard for the first computation of statistical relationships. We also thank Robert M. Hughes and the two anonymous referees for their in-depth analysis of the manuscript and their useful comments.

Appendix (only available in electronic form). Fish species, taxonomic classification, trophic position, and the number of fish captured in the 53 samples – of 27 reaches in 7 hydrographic basins – carried out for the construction of an index of biological integrity for rivers in French Guiana. Taxonomic classification based on Planquette et al. (1996), Keith et al. (2000) and Le Bail et al. (2000).

References

Angermeier P.L., Karr J.R., 1984, Relationships between woody debris and fish habitat in a small warmwater stream. Trans. Am. Fish. Soc. 113, 716-726.

Araujo F.G., 1998, Adaptação do indice de integridade biótica usando a comunidade de peixes para o rio Paraiba do Sul. Rev. Bras. Biol. 58, 547-558.

Araujo F.G., Fichberg I.,Teixeira Pinto B.C., Peixoto M.G., 2003, A preliminary index of biotic integrity for monitoring the condition of the Rio Paraiba do Sul, Southeast Brazil. Environ. Manage. 32, 516-526.

Bailey R.C., Norris R.H., Reynoldson T.B., 2004, Bioassessment of freshwater ecosystems using the reference condition approach. New York, Springer.

Barbour M.T., Stribling J.B., Karr J.R., 1995, Multimetric approach for establishing biocriteria and measuring biological condition. In: Davis W.S., Simon T.P. (Eds.) Biological assessment and criteria. Tools for water resource planning and decision making. London, Lewis Publ., pp. 63-77.

Beechie T.J., Sibley T.H., 1997, Relationships between channel characteristics, woody debris and fish habitat in Northwestern Washington streams. Trans. Am. Fish. Soc. 126, 217-229.

Belpaire C., Smolders R., Vanden Auweele I., Ercken D., Breine J., Van Thyne G., Ollevier F., 2000, An index of biotic integrity characterizing fish populations and the ecological quality of Flandrian water bodies. Hydrobiologia 434, 17-33.

Boujard T., Sabatier D., Rojas-Beltran R., Prévost M.F., Renno J.F., 1990, The food habits of three allochthonous feeding Characoids in French Guyana. Rev. Ecol.-Terre Vie 45, 247-258.

Bozzetti M., Schulz U.H., 2004, An index of biotic integrity based on fish assemblages for subtropical streams in southern Brazil. Hydrobiologia 529, 133-144.

Breine J., Simoens I., Goethals P., Quataert P., Ercken D., Van Liefferinghe C., Belpaire C., 2004, A fish-based index of biotic integrity for upstream brooks in Flanders. Hydrobiologia 522, 133-148.

CNRS/ORSTOM, 1979, Atlas des DOM: La Guyane. Edn. CNRS, Paris, 36 planches.

Dionne M., Karr J.R., 1992, Ecological monitoring of fish assemblages in Tennessee River reservoirs. In: McKenzie D.H., Hyatt D.E., McDonald V.J. (Eds.) Ecological indicators. Chapman & Hall, pp. 259-281.

Fausch K.D., Karr J.R., Yant P.R., 1984, Regional application of an index of biotic integrity based on stream fish communities. Trans. Am. Fish. Soc. 113, 39-55.

Ganasan V., Hughes R.M., 1998, Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh). India Freshw. Biol. 40, 367-383.

Gosse P., Grégoire A., 1997, Dispositif de réoxygénation artificielle du Sinnamary à l'aval du barrage de Petit-Saut (Guyane). Hydroécol. Appl. 9, 23-56.

Gray J.S., 1989, Effects of environmental stress on species rich assemblages. Biol. J. Linn. Soc. 37, 19-32.

Gutierrez M.A.R., 1994, Utilizacion de la ictiofauna como indicadora de la integridad biotica de los rios Guache y Guanare, Estado Portuguesa, Venezuela. Master thesis, Universidad Nacional Experimental de los Llanos Occidentales "Ezequiel Zamora".

Hannan H.H., Fuchs I.R., Whitenberg D.C., 1979, Spatial and temporal patterns of temperature, alkalinity, dissolved O_2, and conductivity in an oligo-mesotrophic deep-storage reservoir in central Texas. Hydrobiologia 66, 209-221.

Hay C.J., van Zyl B.J., Steyn G.J., 1996, A quantitative assessment of the biotic integrity of the Okavango River, Namibia, based on fish. Water South Afr. 22, 263-284.

Harris J.H., 1995, The use of fish in ecological assessments. Aust. J. Ecol. 20, 65-80.

Hocutt C.H., Johnson P.N., Hay C., van Zyl B.J., 1994, Biological basis of water quality assessment: the Kavango River, Namibia. Rev. Hydrobiol. Trop. 27, 361-384.

Horeau V., Cerdan P., Champeau A., Richard S., 1996, Importance des apports exogènes dans le régime alimentaire de quelques poissons de "criques" du bassin versant du fleuve Sinnamary (Guyane française). Rev. Ecol.-Terre Vie 51, 29-41.

Hued A.C. Bistoni M.A., 2005, Development and validation of a biotic index for evaluation of environmental quality in the central region of Argentina. Hydrobiologia 543, 279-298.

Hughes R.M., 1995, Defining acceptable biological status by comparing with reference conditions. In Davis W.S., Simon T.P. (Eds.) Biological assessment and criteria. Tools for water resource planning and decision making. Boca Raton, Lewis Publ., pp. 15-30.

Hughes R.M., Howlin S., Kaufmann P.R., 2004, A biointegrity index for coldwater streams of western Oregon and Washington. Trans. Am. Fish. Soc. 133, 1497-1515.

Hughes R.M., Kaufmann P.R., Herlihy A.T., Kincaid T.M., Reynolds L., Larsen D.P., 1998, A process for developing and evaluating indices of fish assemblage integrity. Can. J. Fish. Aquat. Sci. 55, 1618-1631.

Hughes R.M., Noss R.F., 1992, Biological diversity and biological integrity: current concerns for lakes and streams. Fisheries 17, 11-19.

Hughes R.M., Oberdorff T., 1999, Applications of IBI concepts and metrics to waters outside the United States and Canada. In: Simon T.P. (Ed.) Assessing the sustainability and biological integrity of water resources using fish communities. London, CRC Press, pp. 79-96.

Hugueny B., Camara S., Samoura B., Magassouba M., 1996, Applying an index of biotic integrity based on fish communities in a west African river. Hydrobiologia 331, 71-78.

Imevbore A.M.A., 1970, The chemistry of the River Niger in the Kainji Reservoir area. Arch. Hydrobiol. 67, 412-431.

Imhof J.G., Fitzgibbon J., Annable W.K., 1996, A hierarchical evaluation system for characterizing watershed ecosystem for fish habitat. Can. J. Fish. Aquat. Sci. 53, 312-326.

Joy M.K., Death R.G., 2004, Application of the index of biotic integrity methodology to New Zealand freshwater fish communities. Environ. Manage. 34, 415-428.

Kaestler R.L., Herricks E.E., 1977, Analysis of data from biological surveys of streams: diversity and sample size. Water Resour. Bull. 3, 125-136.

Kamdem-Toham A., Teugels G.G., 1999, First data on an index of biotic integrity (IBI) based on fish assemblages for the assessment of the impact of deforestation in a tropical West African river system. Hydrobiologia 397, 29-38.

Karr J.R., 1981, Assessment of biotic integrity using fish communities. Fisheries 6, 21-27.

Karr J.R., Angermeier P.L., Schlosser I.J., 1983, Habitat structure and fish communities of warmwater streams. EPA Environ. Res. brief, EPA-600/D-83-094, 1-6.

Karr J.R., Dudley D.R., 1981, Ecological perspective on water quality goals. Environ. Manage. 5, 55-68.

Karr J.R., Fausch K.D., Angermeier P.L., Yant P.R., Schlosser I.J., 1986, Assessing biological integrity in running waters. A method and its rationale. Illinois Natural History Survey Spec. Publ. 5, 1-28.

Kesminas V., Virbickas T., 2000, Application of an adapted index of biotic integrity to rivers of Lithuania. Hydrobiologia 422/423, 257-270.

Kestemont P., Didier J., Depiereux E., Micha J.C., 2000, Selecting ichthyological metrics to assess river basin ecological quality. Arch. Hydrobiol. 121, 321-348.

Kushlan J.A., 1976, Environmental stability and fish community diversity. Ecology 57, 821-825.

Lae R., 1995, Climatic and anthropogenic effects on fish diversity and fish yields in the Central Delta of the Niger River. Aquat. Living Resour. 8, 43-58.

Lyons J., Navarro-Pérez S., Cochran P.A., Santana E., Guzman-Arroyo M., 1995, Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in west-central Mexico. Conserv. Biol. 9, 569-584.

Magurran A.E., 1988, Ecological diversity and its measurement. London, Chapman & Hall.

Marteau P., 2001, Mines et Carrières. In: Barret J. (Ed.) Atlas illustré de la Guyane. CNES, IESG, IRD, Région Guyane Publ., pp. 90-93.

Matthews W.J., 1998, Patterns in freshwater fish ecology. New York, Chapman & Hall.

Matthews W.J., Hill L.G., 1979, Influence of physico-chemical factors on habitat selection by red shiners, *Notropis lutrensis* (Pisces: Cyprinidae). Copeia 1979, 70-81.

McCormick F.H., Hughes R.M., Kaufmann P.R., Herlihy A.T., Peck D.V., 2001, Development of an index of biotic integrity for the Mid-Atlantic Highlands Region. Trans. Am. Fish. Soc. 130, 857-877.

Mebane C.A., Maret T.R., Hughes R.M., 2003, An index of biological integrity (IBI) for Pacific Northwest rivers. Trans. Am. Fish. Soc. 132, 239-261.

Meffe G.K., Sheldon A.L., 1988, The influence of habitat structure on fish assemblage composition in Southeastern blackwater streams. Am. Midl. Nat. 120, 225-240.

Mérigoux S., Ponton D., Mérona B. de, 1998, Fish richness and species-habitat relationships in two coastal streams of French Guiana, South America. Environ. Biol. Fishes 51, 25-39.

Mérona B. de, Albert P., 1999, Ecological monitoring of fish assemblages downstream of a hydroeletric dam in French Guiana (South America). Regul. Rivers Res. Manage. 15, 339-351.

Mérona B. de, Carvalho J.L., Bittencourt M.M., 1987, Les effets immédiats de la fermeture du barrage de Tucurui (Brésil) sur l'ichtyofaune en aval. Rev. Hydrobiol. Trop. 20, 73-84.

Mérona B. de, Vigouroux R., Horeau V., 2003, Changes in food resources and their utilization by fish assemblages in a large tropical reservoir in South America (Petit-Saut Dam, French Guiana). Acta Oecol. 24, 147-156.

Mérona B. de, Vigouroux R., Tejerina-Garro F.L., 2005, Alteration of fish diversity downstream from Petit-Saut Dam in French Guiana. Implication of ecological strategies of fish species. Hydrobiologia 551, 33-47.

Miller D.L., Leonard P.M., Hughes R.M., Karr J.R., Moyle P.B., Schrader L.H., Thompson B.A., Daniels R.A., Fausch K.D., Fitzhugh G.A., Gammon J.R., Halliwell D.B., Angermeier P.L., Orth D.J., 1988, Regional applications of an index of biotic integrity for use in water resource management. Fisheries 13, 12-20.

Oberdorff T., Hughes R.M., 1992, Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. Hydrobiologia 228, 117-130.

Oberdorff T., Guégan J.-F., Hugueny B., 1995, Global scale patterns of fish species richness in rivers. Ecography 18, 345-352.

Oberdorff T., Pont D., Hugueny B., Porcher J.-P., 2002, Development and validation of a fish-based index (FBI) for the assessment of "river health" in France. Freshw. Biol. 47, 1720-1734.

Oberdorff T., Porcher J.-P., 1994, An index of biotic integrity to assess biological impacts of salmonids farm effluents on receiving waters. Aquaculture 119, 219-235.

Penczak T., Agostinho A.A., Okada E.K., 1994, Fish diversity and community structure in two small tributaries of the Paraná River, Paraná State, Brazil. Hydrobiologia 294, 243-251.

Penczak T., Gomes L.C., 2000, Impact of engineering on fish diversity and community structure in the Gwda River Basin, north Poland. Polk. Archiw. Hydrobiol. 47, 131-141.

Petts G.E., 1984, Impounded rivers. Perspectives for ecological management. Chichester NY, John Wiley & Sons.

Pinto B.C.T., Araujo F.G., Hughes R.M., in press, Effects of landscape and riparian condition on a fish index of biotic integrity in a large southeastern Brazil river. Hydrobiologia.

Plafkin J.L., Barbour M.T., Porter K.D., Gross S.K., Hughes R.M., 1989, Rapid bioassessment protocols for use in streams ans rivers: benthic macroinvertebrates and fish. EPA/444/4-89/001. Washington DC, US Environmental Protection Agency.

Planquette P., Keith P., Le Bail P.-Y., 1996, Atlas de poissons d'eau douce de Guyane (tome I). Collection du Patrimoine naturel, Vol. 22. IEGB – MNHN, INRA, CSP, Paris, Minist. Environ.

Pont D., Hugueny B., Beier U., Goffaux D., Melcher A., Noble R., Rogers C., Roset N., Schmutz S., in press, Assessing river biotic condition at the continental scale: an European approach using functional metrics and fish assemblages. J. Appl. Ecol.

Power M.E., 1984, Habitat quality and the distribution of algae-grazing catfish in a Panamanian stream. J. Anim. Ecol. 53, 357-374.

Pringle C.M., Scatena F.N., Paaby-Hansen P., Núñez-Ferrera M., 2000, River conservation in Latin America and the Caribbean. In: Boon P.J., Davies B.R., Petts G.E. (Eds.) Global perspectives on river conservation: science, policy and practice. Chichester, John Wiley & Sons, pp. 41-77.

Reash R.J., 1999, Considerations for characterizing Midwestern large river habitats. In: Simon T.P. (Ed.) Assessing the sustainabillity and biological integrity of water resources using fish communities. Boca Raton, CRC Press, pp. 463-474.

Richard S., 1996, La mise en eau du barrage de Petit-Saut (Guyane française). Hydrochimie 1- du fleuve Sinnamary avant la mise en eau, 2- de la retenue pendant la mise en eau 3- du fleuve en aval. Thèse de doctorat, Univ. Aix-Marseille I.

Richard S., Arnoux A., Cerdan P., 1997, Evolution de la qualité physico-chimique des eaux de la retenue et du tronçon aval depuis le début de la mise en eau du barrage de Petit-Saut. Hydroécol. Appl. 9, 57-83.

Rodriguez M.A., Lewis Jr. W.M., 1997, Structure of fish assemblages along environmental gradients in floodplain lakes of the Orinoco River. Ecol. Monogr. 67, 109-128.

Santos G.M. dos, Jegu M., Mérona B. de, 1984, Catálogo de peixes comerciais do baixo Rio Tocantins, projeto Tucurui. Manaus, ELETRONORTE/CNPq/INPA.

Sechnik C.W., Carline R.F., Stein R.A., Rankin E.T., 1986, Habitat selection by smallmouth bass in response to physical characteristics of a simulated stream. Trans. Am. Fish. Soc. 115, 314-321.

Sheldon A.L., 1988, Conservation of stream fishes: patterns of diversity, rarity, and risk. Conserv. Biol. 2, 149-156.

Sheldon A.L., Meffe G.K., 1995, Path analysis of collective properties and habitat relationships of fish assemblages in coastal plain streams. Can. J. Fish. Aquat. Sci. 52, 23-33.

Simons T.P., Lyons J., 1995, Application of the index of biotic integrity to evaluate water resource integrity in freshwater ecosystems. In: Davis W.S., Simon T.P. (Eds.) Biological assessment and criteria: tools for water planning and decision making. Boca Raton, Lewis Publ., pp. 245-262.

Statzner B., Bis B., Doledec S., Usseglio-Polatera P., 2001, Perspectives for biomonitoring at large spatial scales: a unified measure for the functional composition on invertebrate communities in European running waters. Basic Appl. Ecol. 2, 73-85.

Tait C.K., Li J.L., Lambert G.A., Pearsons T.N., Li H.W., 1994, Relationship between riparian cover and the community structure of high desert streams. J. N. Am. Benthol. Soc. 13, 45-56.

Tejerina-Garro F.L., 2001, Étude des relations habitat-poisson dans les eaux courantes de Guyane française pour l'évaluation de la qualité du milieu aquatique. Thèse de doctorat. Univ. Montpellier II, Sciences et Techniques du Languedoc.

Tejerina-Garro F.L., Maldonado M., Ibañez C., Pont D., Roset N., Oberdorff T., 2005, Effects of natural and anthropogenic environmental changes on riverine fish assemblages: a framework for ecological assessment of rivers. Braz. Arch. Biol. Techn. 48, 91-108.

Tejerina-Garro F.L., Mérona B. de, 2000, Gill net sampling standardisation in large rivers of French Guiana (South America). Bull. Fr. Pêche Piscic. 357/360, 227-240.

Tejerina-Garro F.L., Mérona B. de, 2001, Spatial variability of biotic and abiotic factors of the aquatic habitat in French Guiana. Reg. Rivers Res. Manage. 17, 157-169.

Thévenet A., Statzner B., 1999, Linking fluvial fish community to physical habitat in large woody debris: sampling effort, accuracy and precision. Arch. Hydrobiol. 145, 57-77.

Tramer E.J., Rogers P.M., 1973, Diversity and longitudinal zonation in fish populations of two streams entering a metropolitan area. Am. Midl. Nat. 90, 366-374.

Travnichek V.H., Maceina M.J., 1994, Comparison of flow regulation effects on fish assemblages in shallow and deep water habitats in the Tallapoosa River, Alabama. J. Freshw. Ecol. 9, 207-216.

Ward J.V., Stanford J.A., 1979, Ecological factors controlling stream zoobenthos with emphasis on thermal modification of regulated streams. In: Ward J.V., Stanford J.A. (Eds.) The ecology of regulated streams. New York, Plenum Press, pp. 35-55.

Ward J.V., Stanford J.A., 1982, Effects of reduced and perturbed flow below dams on fish food organisms in Rocky Mountain trout streams. In: Grover J.H. (Ed.) Allocation of fishery resources. Rome, FAO, 493-501.

Welcomme R.L., 1979, Fisheries ecology of floodplain rivers. London, Longman.

Williams R.D., Winget R.N., 1979, Macroinvertebrate response to flow manipulation in the Strawberry River, Utah (USA). In: Ward J.V., Stanford J.A. (Eds.) The ecology of regulated streams. New York, Plenum Press, pp. 365-376.

Winemiller K.O., Jepsen D.B., 1998, Effects of seasonality and fish movement on tropical food webs. J. Fish Biol. 53 (Suppl. A), 267-296.

Wright J.F., 1995, Development and use of a system for predicting macroinvertebrates in flowing waters. Aust. J. Ecol. 20, 181-197.