ACTA
AMAZONICA

http://dx.doi.org/10.1590/1809-4392202201461

ORIGINAL ARTICLE

# Evaluation of fisheries management strategies using a biotic integrity index in floodplain lakes in the lower Solimões River, Amazonas, Brazil

Bruna de Souza ANDRADE[1]*, Carlos Edwar de Carvalho FREITAS[2], Maria Letizia PETESSE[3], Flávia Kelly SIQUEIRA-SOUZA[2]

[1] Instituto Nacional de Pesquisas da Amazônia (INPA), Av. André Araújo, 2936, Campus II, Aleixo, 69060-001 Manaus, Amazonas, Brazil

[2] Universidade Federal do Amazonas, Av. Gen. Rodrigo Otávio, 3000, Coroado II, 69077-000 Manaus, Amazonas, Brazil

[3] Instituto de Pesca de São Paulo, Av. Conselheiro Rodrigues Alves, 1252, Vila Mariana, 04014-900 São Paulo, Brazil

* Corresponding author: andrade.brunabio@gmail.com; https://orcid.org/0000-0002-3846-8499

## ABSTRACT

Fisheries is an important activity in the Amazon basin and potentially has a high impact on ichthyofauna. With the aim of achieving sustainability in fisheries of target species, protected areas were established within a fisheries management framework known as the Amazon Lakes Management System (ALMS). With the aim to evaluate the effect of ALMS on fish assemblages as a whole, we compared floodplain lakes with different levels of fish management in the lower Solimões River, in the Brazilian Amazon, developing an index of biotic integrity, named ALMS-IBI. We sampled fish in three lake categories: protected (PR), managed (MN), and commercially fished (CM). The ALMS-IBI was developed by selecting and testing fish assemblage metrics based on samplings carried out in 2017 and 2018, using gillnets with different mesh sizes. We captured 4565 fish of 113 species and seven trophic categories. The final index is composed of seven metrics related to species richness, trophic structure, and fish abundance, and characterized the fish assemblage in the PR and MN lakes as acceptable, and in the CM lakes as poor. Our results indicate that the ALMS-IBI can be an efficient method to monitor the whole fish assemblage in Amazonian floodplain lakes, and can be used as a complementary tool in the ALMS to assess environmental sustainability.

**KEYWORDS:** environmental quality, multimetric index, bioindicators, Amazon basin

# Avaliação da estratégia de manejo pesqueiro usando um índice de integridade biótica em lagos de várzea no baixo Rio Solimões, Amazonas, Brasil

## RESUMO

A pesca é uma atividade importante na bacia amazônica e tem alto impacto potencial sobre a ictiofauna. Com o objetivo de alcançar sustentabilidade na atividade pesqueira de espécies focais, foram estabelecidas áreas protegidas no marco de um plano de manejo pesqueiro conhecido como Sistema de Manejo dos Lagos da Amazônia (ALMS). Com o objetivo de avaliar o efeito do ALMS sobre toda a assembleia de peixes, nós comparamos lagos de várzea com diferentes níveis de manejo pesqueiro no baixo rio Solimões, na Amazônia brasileira, desenvolvendo um índice de integridade biótica, denominado ALMS-IBI. Amostramos peixes em três categorias de lagos: protegidos (PR), manejados (MN) e de pesca comercial (CM). O ALMS-IBI foi desenvolvido por meio da seleção e teste de métricas das assembleias de peixes, com base em amostragens realizadas em 2017 e 2018, usando redes de emalhar com diferentes tamanhos de malha. Capturamos 4565 peixes de 113 espécies e sete categorias tróficas. O índice final está composto de sete métricas relacionadas à riqueza de espécies, estrutura trófica e abundância de peixes, e caracterizou a assembleia de peixes como regular nos lagos PR e MN, e como pobre nos lagos CM. Nossos resultados indicam que o ALMS-IBI pode ser um método eficiente para monitorar a assembleia de peixes como um todo em lagos de várzea amazônicos, e pode ser usado como uma ferramenta complementar para avaliar a sustentabilidade ambiental no ALMS.

**PALAVRAS-CHAVE:** qualidade ambiental, índice multimétrico, bioindicadores, bacia Amazônica

**CITE AS:** Andrade, B.S.; Freitas, C.E.C.; Petesse, M.L.; Siqueira-Souza, F.K. 2022. Evaluation of fisheries management strategies using a biotic integrity index in floodplain lakes in the lower Solimões River, Amazonas, Brazil. *Acta Amazonica* 52: 289-298.

## INTRODUCTION

Aquatic environments suffer many anthropic impacts (*e.g.*, invasion of alien species, habitat destruction and fragmentation caused by hydromorphological modifications, eutrophication, climate change, fish overexploitation) that generate multiple pressures on the biota and on the environment as a whole (Poikane *et al.* 2017). The effects of environmental stressors can result in imbalance in ecosystem services of aquatic environments, such as habitat loss, species decline, drop in water level, and degradation of water quality (Bellwood *et al*. 2003; Hanna *et al*. 2018; Vári *et al*. 2021), challenging the stability and integrity of aquatic ecosystems (Fausch *et al*. 1990; Karr and Chu 1999).

One strategy used to protect aquatic ecosystems and their services (*e.g.,* water quality control, habitat provision, erosion prevention, provision of fertile soils, food, flood regulation, microclimate regulation) is the establishment of protected areas (Saunders *et al.* 2002). Protected areas have effectively prevented habitat degradation and biodiversity decline, and have also aided the recovery of fish stocks (Schram *et al.* 1995; Reid *et al.* 2001; Suski and Cook 2007).

In Amazonian aquatic environments, anthropic impacts are mainly related to overexploitation of commercial fish species (Barthem and Goulding 2007; Castello *et al.* 2011), posing a threat to both commercial and non-commercial species, and to the integrity of the aquatic environment. The effects on fish include decreased density of commercial species, reduced fish stocks, and changes in assemblage structure, composition and functionality (Barthem and Goulding 2007; Silvano *et al.* 2009; Castello *et al.* 2011; Silvano *et al.* 2017).

In the Brazilian Amazon, protected areas have been established as a strategy to achieve sustainable fishing management, in a framework known as the Amazon Lakes Management System (ALMS) (*Sistema de Manejo dos Lagos da Amazônia*, in Portuguese) (McGrath *et al.* 1993; Castro and McGrath 2001; Benatti *et al.* 2003). The ALMS is specifically focused on floodplain lakes (Aquino *et al.* 2007) and was implemented in the 1980s by local fishermen and researchers who were concerned about the overexploitation of fish species. In general, the ALMS is based on the assignment of fishing restriction categories to limit the irrestricted access of fishers to floodplain lakes, thus establishing a more controlled and sustainable use of fishery resources and the environment (Nolan *et al.* 2009). It is a management instrument regulated by the federal environmental protection agency, Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA) and the Amazonas state agency for the environment, *Secretaria do Estado do Meio Ambiente* (SEMA) through fisheries agreements to promote sustainable fisheries by riverine and commercial fishers, and consists of defining the rules for the access to and use of local fishing resources and/or fishing gear and techniques, applicable in the state of Amazonas, Brazil (SDS 2011).

The ALMS defines lake categories based on levels of fishing restrictions in the floodplains of the Solimões-Amazonas River and its main Andean and Pre-Andean tributaries, and respects the existing laws (McGrath 1998; Castro and McGrath 2001; SDS 2011; Campos-Silva and Peres 2016). Another characteristic of the ALMS is the rotation of fishing access to lakes, which is based on the population dynamics of the target species, as a management strategy to improve ecosystem integrity in the floodplain lakes.

The success of this management model has been attested by the recovery of stocks of the main target species, such as *Arapaima* spp., in several areas of the Brazilian Amazon (Almeida *et al.* 2009; Campos-Silva and Peres 2016; Campos-Silva *et al.* 2017). For example, protected lakes and lakes with some degree of protection showed larger populations of the target species than unprotected lakes (Campos-Silva and Peres, 2016; Campos-Silva *et al.* 2017).

In addition to the main effect on a single target species or group of target species, the co-management of lakes through ALMS can promote the conservation of fish diversity as a whole in aquatic environments (Medeiros-Leal *et al.* 2021). For example, in the lakes of Paciência Island, in the region of the lower Solimões River, the implementation of ALMS had a positive effect on the composition and structure of fish assemblages, increasing fish abundance, biomass, fish size and species richness (Medeiros-Leal *et al.* 2021).

In this context, the aim of this study was to develop an ecological index of biotic integrity (IBI) for the lakes included in the ALMS by using metrics of richness, abundance and trophic category, according Petesse e*t al.* (2016), to compare fish assemblages among lakes of different management categories. Since the development of IBI by Karr (1981), this tool has been used for the assessment of ecosystems exposed to distinct levels of environmental impacts (Carvalho *et al.* 2017; Carter *et al.* 2019), but few studies have used metrics of ecological integrity to assess the responses and effectiveness of management actions (Parrish *et al.* 2003; Tierney *et al.* 2009; Carter *et al.* 2019), and their application in sustainable fishing management programs is novel.

We evaluated fish richness metrics (*e.g.*, total number of species, number of species with moderate/high vulnerability), abundance (*e.g.*, total number of individuals, abundance of piranhas, equitability) and proportion of specialist trophic groups (*e.g.*, percentage of piscivorous individuals), and selected those metrics that best represented the variance among lake categories. We propose to use this index of biotic integrity (IBI) as a complementary aspect to ALMS, to measure environmental quality for fish assemblages as a whole, and name it ALMS-IBI. We evaluated whether ALMS-IBI can adequately measure the effect of fisheries management on Amazonian lakes, and hypothesized that ALMS-IBI scores are significantly higher in protected (no fishing) and managed lakes (restricted fishing) than in lakes where fishing is permitted without restriction.

# MATERIAL AND METHODS

## Study area

The study was carried out in six floodplain lakes located on Paciência Island in the lower stretch of the Solimões River, in Amazonas state, Brazil (Figure 1). The islands of the Amazonian floodplains support complex systems of relatively small and shallow lakes located close to each other, and that are often connected during the high water season (Freitas *et al.* 2010). Two of the six study lakes were assigned to each of three management categories, as defined by ALMS: 1) Preserved lakes (PR): Cacau Lake (3°18'32.9"S, 60°12'54.1"W) and Baixo Lake (3°18'09"S, 60°13'35.4"W). Fishing is prohibited in these lakes, to allow for the recovery of fish stocks; 2) Management lakes (MN): Sacambú Lake (3°18'46"S, 60°13'19"W) and Preto Lake (3°18'33.0"S, 60°13'09.5"W). In these lakes, the fishing of *Arapaima* spp. is regulated by annually pre-defined quotas of fish for each fisherman or fishing sector; and 3) Commercial use lakes (CM): Piranha Lake (3°16'57.3"S, 60°13'20.0"W) and Poção Lake (3°18'53.1"S, 60°11'10.5"W). These lakes are open to unrestricted fisheries, including commercial fisheries. The distance between lakes varies from 0.5 to 6 km and the PR and MN lakes are closer to each other than to the CM lakes (Figure 1).

The lakes have been assigned to these categories in 2011, when ALMS was implemented on Paciência Island. The ALMS established that lakes rotate periodically in category assignment, but the rotation has not yet occurred on Paciência Island. The ALMS co-management protocol determines that PR lakes are located further away from the Solimões River channel in order to make access difficult for commercial fishers, and that MN lakes are located closer to fisher communities, to facilitate the communitary monitoring of fishing. In the MN lakes, commercial fishing is prohibited, but subsistence fishing by the community and annual fishing (determined by quotas) of the target species of fisheries management (*Arapaima* spp.) are allowed.

## Fish sampling

Fish were collected in five sampling events during the hydrological periods of rising water (April), high water (June), receding water (August), and low water (December) in 2017, and rising water (April) in 2018, with a total of five sampling in each lake, totaling 30 sampling events overall. In each sampling event, we used ten gillnets, divided into two gillnets in each lake covering different points (aquatic herbaceous stands and open water). The gillnets had standardized dimensions of 15 m long and 2 m high, and mesh sizes varying from 30 to 120

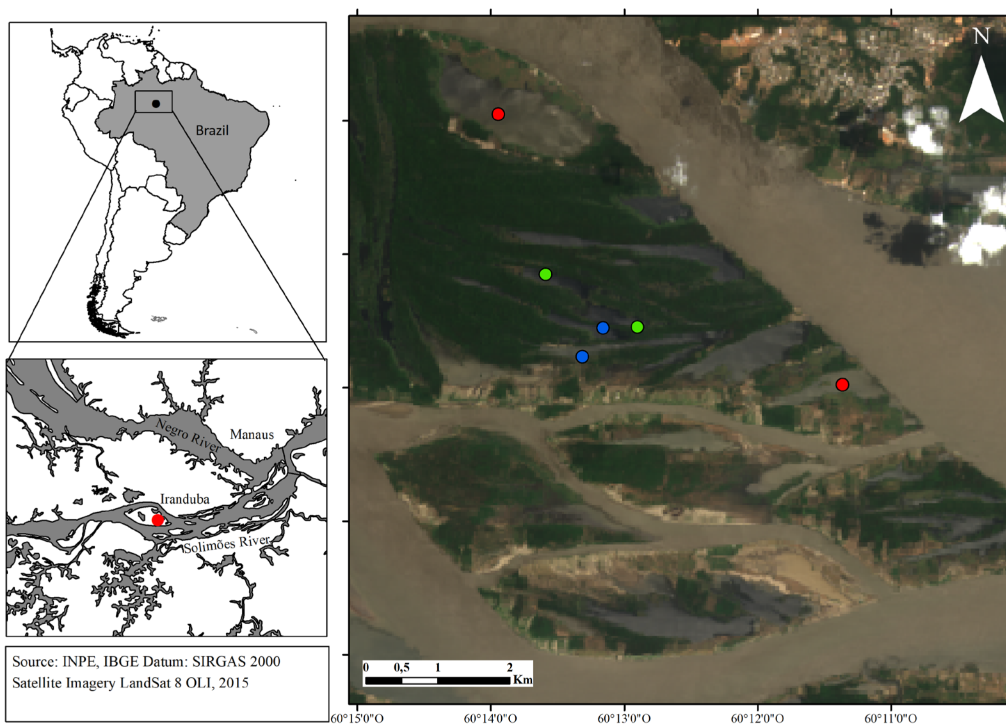


**Figure 1.** Location of the study area in northern Brazil, and of the studied lakes (red dot) on Paciência Island in the lower stretch of the Solimões River. The close up satelite image of Paciência Island shows the six sampled floodplain lakes. Green points = preserved lakes; blue points = managed lakes; red points = commercial fishing lakes. This figure is in color in the electronic version.

mm between opposite knots. The gillnets were placed in the water for eight hours, divided into two crepuscular phases: 5:00 until 9:00 a.m. and 5:00 until 9:00 p.m.. The sampled fish were identified using reference keys (Soares *et al.* 2008). The sampling was conducted under license # 62867-1 granted by IBAMA through the Brazilian Biodiversity Authorization and Information System (SISBIO) and was authorized by the ethics committee in the use of animals of Universidade Federal do Amazonas (UFAM) (protocol # 037/2017 CEUA/UFAM). The fishes were deposited in the ichthyological collection of Universidade Federal do Amazonas (UFAM), Manaus, Amazonas state, Brazil.

### Environmental variables

We measured the following environmental variables *in situ* during fish sampling using a portable digital multiparameter (Hanna HI9829) to record abiotic environmental information: temperature (ºC), pH, dissolved oxygen ($mg \cdot L^{-1}$), depth (m), conductivity (s/m) and transparency (m). The measurements were made on the five sampling events, in the early morning and late afternoon, near the surface and bottom of the lake.

### Reference lakes

In our study, the protected lakes (PR), Cacau and Baixo, were used as reference lakes for the biological integrity of the lake biota. Due to the absence of pristine sites in the region, these lakes were considered as reference condition (Whittier *et al.* 2007), i.e., the original state of biological integrity before any human disturbance. As these lakes are no-fishing lakes by the ALMS protocol, the fish assemblages are expected to exhibit natural cycles, unaffected by the fishing pressure elsewhere in the region. They are located in the interior of the island, far away from the main channel of the Solimões River.

### Candidate metrics and selection procedure

Candidate metrics for the ALMS-IBI were organized in three groups: species richness, trophic categories and species abundance. We used the set of metrics validated by Petesse *et al*. (2016), excluding "number of non-native species", as there is no record of exotic fish species in the Solimões River. We added four metrics, namely, "number of Cichlidae species"; "abundance of piranhas" (Serrasalmidae), referring to *Pygocentrus nattereri*, *Serrasalmus maculatus* and *S. altispinis*; "abundance of branquinhas" (Curimatidae), referring to *Psectogaster rutiloides*, *Potamorhina latior* and *P. altamazonica*; and "abundance of sardines" (Triportheidae), referring to *Triportheus albus*, *T. angulatus* and *T. auritus*. These metrics were added because these species are the most representative (in abundance) of their trophic groups in the floodplain environment and because we would like to assess how fisheries impact the abundance of these species. In total, we used 23 candidate metrics (Supplementary Material, Table S1).

The trophic categories used to develop the ALMS-IBI were based on a literature survey (Mérona and Rankin-de-Mérona 2004; Santos *et al.* 2009; Soares *et al.* 2008) and on information from the FishBase website (Froese and Pauly 2018). The following trophic groups were considered: omnivores, herbivores, detritivores, insectivores, invertivores, planktivores and piscivores. The ecological indices of richness (S), abundance (N), Shannon diversity index (H'), equitability of Pielou (J') and dominance of Berger-Parker (d), as described by Magurran (2004), were estimated to compose complementary biotic data on the fish assemblages. The indices were estimated for each sampling event in each lake, *i.e*., five sampling events in each lake.

We used the dataset of the references lakes (Baixo and Cacau lakes) to score metrics and we randomly sub-sampled the dataset to validate the ALMS-IBI, according to the criterion used by Petesse *et al.* (2016). The metrics were submitted to three tests in sequence: 1) metric range test; 2) metric sensitivity test; and 3) metric redundancy test (Petesse *et al.* 2016). The range test evaluated the distribution of values for each metric and the difference between the maximum and minimum values of the range. When the difference was < 4, the metric had no variability and was excluded in the initial phase (Petesse *et al.* 2016). For metrics with sequences of equal values (*e.g.*, sequence of zero values) the criterion of 75% of the values was used, which corresponds to the percentage calculation of equal values, according to the equation: n/N total x 100; where n = number of equal values in the metric; N total = total number of samples. Percentages over 75% meant that the metric had no variability and would probably not distinguish between reference sites and test sites, according to the criterion of Petesse *et al.* (2016). For the metrics "percentage of insectivorous individuals", "percentage of planktivorous individuals" and "abundance of branquinhas", the criterion of 75% of equal values was used. The "equitability" and "dominance" metrics were converted into percentages to apply the same criteria as to the other metrics.

The sensitivity test was carried out using box-plot graphical analysis to evaluated the ability of the metric to discriminate between reference lakes and test lakes by assessing the degree of overlap of quartiles and medians. The greater the overlap, the lower the sensitivity of the metric to distinguish between reference areas and impacted areas (Barbour *et al.* 1996; Hughes *et al.* 1998). The redundancy test evaluated the correlation between metrics, since highly correlated metrics do not respond to the assessment of environmental integrity (Seegert 2000). This analysis used Spearman's correlation coefficient (Petesse *et al.* 2016).

### Procedure for scoring the metrics

For each metric selected for the reference lakes, the 65$^{th}$ percentile was used as the upper threshold, and the 32$^{nd}$

percentile was used as the lower threshold (Petesse *et al.* 2007). These thresholds were chosen based on a conservative approach and assumed that some metric values from the reference lakes could have relatively low integrity (*i.e.*, due to natural variability in the hydrologic cycle or to an anthropogenic effect that occurred in the past). A continuous score scale from zero (worst situation) to 10 (best situation) was used. Values above or equivalent to the 65$^{th}$ percentile received a score of 10, representing metrics of high integrity, while values equal to or below the 32$^{nd}$ percentile received a score of zero. For the values in between the percentiles, the following equation was used: (observed value – value of the 32$^{nd}$ percentile)/(value of the 65$^{th}$ percentile – value of the 32$^{nd}$ percentile) x 10. For low-integrity metrics, the score was inverted, and the values between the percentiles received scores according to the following equation: (value of the 65$^{th}$ percentile – observed value)/(value of the 65$^{th}$ percentile – value of the 32$^{nd}$ percentile) x 10 (Petesse *et al.* 2016).

To standardize the final index, the sum of the partial scores for each lake category was divided by the total number of metrics selected and multiplied by 10 in order to obtain the ALMS-IBI, which varied between 0 and 100. This interval was divided into four classes, where 0-25 (poor condition), *i.e.*, human pressure is dominant and the negative effects are evident; 26-50 (regular condition), *i.e.*, signs of human pressure are evident and negative effects begin to appear; 51-75 (good condition), *i.e.*, signs of apparent human interference, but the use of resources is environmentally sustainable; and 76-100 (excellent condition), *i.e.*, condition minimally impacted or without significant anthropogenic interference.

### Statistical analysis

The non-parametric Kruskal-Wallis test was used to compare the ALMS-IBI among the four hydrological periods. Data refer to two samplings per lake category for the high, receding and low water periods, and four samplings per lake category for the rising water period.

Each ALMS-IBI candidate metric was compared among lake categories using the non-parametric Kruskal-Wallis test, using 10 samples per lake category, *i.e.*, five samples for each lake. We also compared the final ALMS-IBI scores among lake categories, using the Kruskal-Wallis test, based on ten sampling values per category (five for each lake). The pairwise difference between lake categories was tested by Wilcoxon test with the adjustment method "hommel", a method considered robust and valid for independent samples. A significance level of 5% ($p \leq 0.05$) was used in all tests.

Non-metric multidimensional scaling (NMDS) was applied based on Manhattan distance, using lake categories as objects and the metrics of the biotic integrity index as attributes. The analyses were performed in R software using the Vegan package (R Core Team 2020). The metrics were standardized using the Z scores method.

## RESULTS

### Structure of the fish assemblages

A total of 4,565 fish specimens were collected, distributed in six orders, 25 families and 113 species (Supplementary Material, Table S2). Characiformes was the order with the highest proportion of species in MN (88.4%) and PR (75.4%) lakes, while Siluriformes had the highest proportion in CM lakes (36.9%). Most species belonged to Curimatidae and Serrasalmidae in the MN (52.8%) and PR (43.3%) lakes, and to Triportheidae and Pimelodidae in the CM lakes (50.2%). The Sacambú Lake (MN) had the highest species richness estimates and abundance of individuals, while Preto Lake (MN) had the highest species diversity. The PR lakes had the second highest species diversity and highest equitability. The Shannon diversity, Pielou equitability and Berger-Parker dominance indices were similar among the lakes (Table 1).

In the CM lakes, omnivores made up 51.9% of the fish sampled overall. In the MN lakes, omnivores (39.6%) and detritivores (36.3%) were similarly predominant. In the PR lakes, omnivores, carnivores and detritivores were equally abundant (approximately 29% each) (Figure 2), but the highest richness was of omnivores (S = 20).

**Table 1.** Structure and ecological indices of the fish assemblage in six floodplain lakes sampled on Paciência Island (lower Solimões River, Amazonas, Brazil) between April 2017 and April 2018. Values refer to the pooled data of five sampling events in each lake. Category (according to ALMS): PR = protected lake; MN = management lake; CM = commercial use lake. S = species richness; N = number of individuals; H′ = Shannon diversity index; J′ = Pielou equitability index; d = Berger-Parker dominance index.

| Lake | S | Number of orders | Number of families | N | H′ | J′ | d | Category |
|---|---|---|---|---|---|---|---|---|
| Baixo | 52 | 5 | 20 | 324 | 3.35 | 0.84 | 0.95 | PR |
| Cacau | 53 | 5 | 19 | 327 | 3.32 | 0.83 | 0.94 | PR |
| Sacambú | 83 | 6 | 21 | 2,099 | 3.21 | 0.77 | 0.93 | MN |
| Preto | 67 | 6 | 22 | 454 | 3.42 | 0.81 | 0.94 | MN |
| Piranha | 63 | 5 | 20 | 868 | 3.11 | 0.75 | 0.91 | CM |
| Poção | 63 | 5 | 19 | 493 | 3.12 | 0.75 | 0.91 | CM |

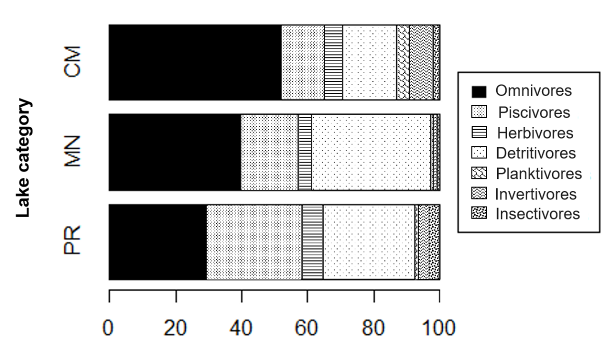


**Figure 2.** Relative abundance (%) of the trophic groups of fish sampled in six lakes on Paciência Island in the lower Solimões River (Brazil). Lakes are categorized according to the Amazon Lakes Management System (ALMS): PR = protected; MN = managed; CM = commercial use. Values refer to grouped data of five sampling events in each of two lakes per category.

### Environmental variables

Cacau Lake (PR) had a lower mean temperature (25.89 ºC) and higher mean transparency (0.91 m) (Table 2). Baixo Lake (PR) showed a mean temperature of 28.47 ºC and had the highest pH value (8.06). PR lakes showed highest transparency, highest pH and lowest temperatures. CM lakes had lowest pH, highest conductivity, lowest transparency and temperature above 29 °C. MN lakes, which are the deepest lakes (Table 2), had highest dissolved oxygen.

### Metric selection for the ALMS-IBI

All 23 metrics showed sufficient variability in the range test (Supplementary Material, Table S3) and thus proceeded to the next test. According to the sensitivity test (Supplementary Material, Figure S1 and S2), only seven metrics differentiated among lake categories (number of Siluriformes species, number of Cichlidae, percentage of omnivorous individuals, percentage of piscivorous individuals, equitability, abundance of sardines and abundance of piranhas) and were used in the redundancy test (Supplementary Material, Table S4) and included in the ALMS-IBI (Supplementary Material, Table S5).

The ALMS-IBI presented higher scores for the PR and MN lakes. The metric "percentage of piscivorous individuals" scored high in the PR lakes (Baixo = 8.87; Cacau = 8.09) and the metric "abundance of piranhas" scored high in the PR lakes (Baixo = 10; Cacau = 8.98) and MN lakes (Sacambú = 10; Preto = 9.79). The metric "number of Cichlid species" achieved mixed scores in the PR lakes (Cacau = 10; Baixo = 4) and scored high in MN lakes (Sacambú = 10; Preto = 10). The metrics "Percentage of omnivorous individuals" and "Abundance of sardines" were higher in CM lakes and consequently presented zero scores. The PR and MN lakes were classified as having regular integrity, while the CM lakes were classified as having poor integrity (Figure 3).

There was significant difference among lake categories for the metrics "percentage of piscivorous individuals" ($H = 9.8692$, $df = 2$, $p = 0.007$), "abundance of piranhas" ($H = 9.3626$, $df = 2$, $p = 0.009$) and "abundance of sardines" ($H = 10.472$, $df = 2$, $p = 0.005$). The ALMS-IBI varied significantly among lake categories ($H = 10.49$, $df = 2$, $p = 0.005$), with a significant difference both between PR and CM lakes (Wilcoxon test, $p = 0.041$) and between MN and CM lakes (Wilcoxon test, $p = 0.004$). The ALMS-IBI scores did not vary significantly among hydrological periods.

The NMDS (stress value = 0.0210) separated the three lake categories in the multivariate space (Figure 4). The metrics "percentage of piscivorous individuals" and "abundance of piranhas" tended to be associated with PR lakes, while the metrics "abundance of sardines" and "number of Cichlidae" were associated with CM and MN lakes. Equitability was associated with MN lakes, and the metric "number of Siluriformes" was associated with MN and CM lakes.

## DISCUSSION

Our results provide evidence that the ALMS co-management rules improve biotic integrity at the fish assemblage level. The ALMS has been implemented

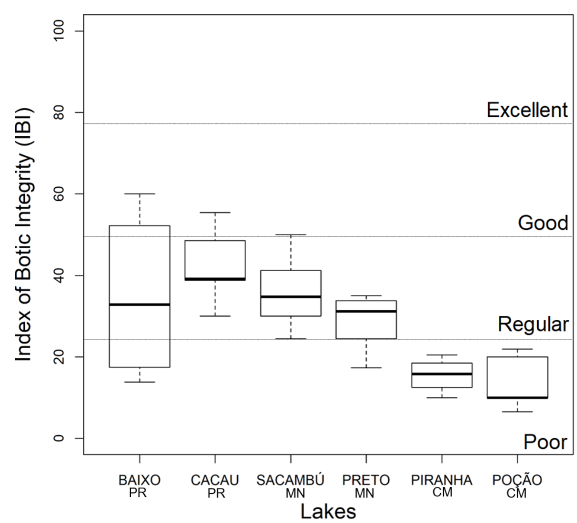


**Figure 3.** Score distribution of the index of biotic integrity of the Amazon Lakes Management System (ALMS-IBI) for six lakes on Paciência Island in the lower Solimões River (Brazil). The central line in bold represents the median, the box the 25-75% quartiles, and the bar represents the range. Values correspond to five samplings in each of two lakes per category.

**Table 2.** Limnological and physical-chemical parameters for six floodplain lakes sampled on Paciência Island (lower Solimões River, Amazonas, Brazil) between April 2017 and April 2018. Category (according to ALMS): PR = protected; MN = management; CM = commercial use. Values are the mean of five sampling events in each lake to bottom and surface.

| Environmental variable | Lake | | | | | |
|---|---|---|---|---|---|---|
| | Baixo (PR) | Cacau (PR) | Sacambú (MN) | Preto (MN) | Piranha (CM) | Poção (CM) |
| Dissolved oxygen (mg L$^{-1}$) | 1.7 ± 1.4 | 1.6 ± 0.8 | 2.6 ± 2.6 | 2.1 ± 1.4 | 2.4 ± 1.3 | 1.5 ± 1.0 |
| pH | 8.0 ± 0.7 | 7.6 ± 0.2 | 7.7 ± 0.2 | 7.6 ± 0.3 | 7.5 ± 0.4 | 7.6 ± 0.1 |
| Temperature (°C) | 28.4 ± 2.5 | 25.8 ± 7.4 | 29.4 ± 2.0 | 29.1 ± 1.5 | 29.1 ± 0.6 | 29.2 ± 1.2 |
| Conductivity (s m$^{-1}$) | 49.8 ± 10.8 | 53.0 ± 15.9 | 67.0 ± 27.8 | 45.1 ± 13.4 | 57.3 ± 10.6 | 60.0 ± 28.8 |
| Transparency (m) | 0.7 ± 0.2 | 0.91 ± 0.2 | 0.51 ± 0.1 | 0.8 ± 0.2 | 0.4 ± 0.1 | 0.6 ± 0.0 |
| Depth (m) | 4.6 ± 2.2 | 5.4 ± 1.8 | 4.8 ± 0.2 | 5.3 ± 2.2 | 4.4 ± 2.0 | 5.5 ± 2.5 |

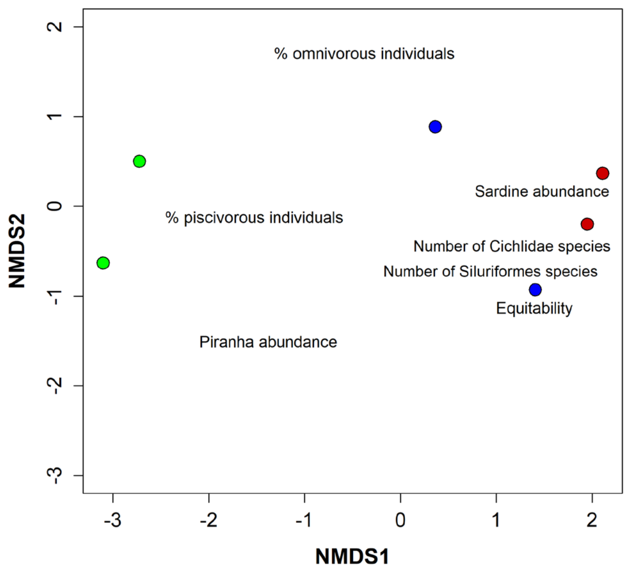


**Figure 4.** Non-metric multidimensional scaling (NMDS) showing the relation of lakes on Paciência Island in the lower Solimões River (Brazil) categorized according to the Amazon Lakes Management System (ALMS) with the metrics on fish assemblages selected for the index of biotic integrity (IBI). Green points = protected lakes; blue points = management lakes; red points = commercial fishing lakes. Stress-value = 0.0210. This figure is in color in the electronic version.

throughout Amazonas state, notably so in two sustainable development reserves in the middle Solimões River region that combine the preservation of the environment and the sustainable exploitation of natural resources (RDS Piagaçu-Purus on the lower Purus River, and RDS Mamurauá and Amanã). In both reserves, the management of *Arapaima* spp. is well established and shows positive results in the recovery of populations (Amaral *et al.* 2011). A study conducted in the lower Solimões River (Medeiros-Leal *et al*. 2021) showed that fisheries management has a positive effect on the structure and composition of the fish assemblage as a whole, increasing fish abundance and richness in the whole lake ecosystem. Likewise, the environmental quality of Amazonian Forest streams in areas of reduced impact logging was higher than in unmanaged logging exploitations (Prudente *et al.* 2018). In the Pantanal biome, a multi-metric index obtained high values of integrity (*e.g.*, good and excellent) in a protected area, confirming the effectiveness of the environmental protection (Polaz *et al.* 2017). Studies in other biomes also provided evidence for high biotic integrity in protected aquatic environments (Mancini *et al.* 2005; Reza *et al.* 2011; Sobczak *et al.* 2013; Fitzpatrick *et al.* 2015). In our study, the protected lakes (PR), where fishing is prohibited, may represent a reliable reference site for the assessment of fishing pressure on the biotic integrity of the ichthyofauna for this region, since they have better preserved fish assemblages in terms of species composition and trophic structure in comparison to what was found in the CM lakes.

The ALMS-IBI revealed that piscivorous fish, particularly piranhas, were predominant in the PR lakes, supporting other reports on a relatively higher proportion of fish of higher trophic levels in protected environments (Paine 1966; Araújo 1998; Ganasan and Hughes 1998). In the study by Petesse *et al.* (2016), the lakes with excellent biotic integrity presented the highest percentages of predatory (*i.e.,* carnivorous) fish. Carnivorous fish play a role in structuring the lower trophic levels (Jia *et al.* 2021), therefore changes in the proportion of fish belonging to high trophic levels can alter the fish assemblage to a higher dominance of lower trophic levels. Commercial fishing frequently focuses on large species of high trophic level (*e.g., Arapaima spp.*), which consequentially alters the fish assemblage (Jia *et al.* 2021) and explains the importance of this metric to assess biotic integrity in ALMS-IBI. In the case of piranhas, we attribute their higher abundance in the PR lakes relative to the CM lakes to the sedentary behavior of these species (Soares *et al.* 2008), which characterizes them as residents in the PR lakes, at a longer distance from the CM lakes and isolated in the low water period. In addition, the association of PR lakes with piscivorous fish was probably influenced by the tendency to higher transparency of the water column in these lakes, as had been observed in a previous study (Medeiros-Leal *et al*. 2021). Further studies should elucidate how much of the variation in limnological parameters in ALMS lakes is explained by the co-management as opposed to natural variability.

Species richness and abundance were lower in the PR lakes compared to the MN lakes, which may be associated with the greater abundance of predators in PR lakes, as these exert strong pressure on the assemblages (Both *et al.* 2009; Freitas *et al.* 2010), which could result in the lower richness in these lakes. In the presence of many predators, lateral displacement of prey may occur and result in decreased richness of fish, mainly in the high water period (Melo *et al.* 2007). The MN and PR lakes showed equitability in species abundance and higher Shannon diversity compared to the CM lakes, corroborating our results that PR and MN lakes have better biotic integrity. The proximity of the PR and MN lakes probably influences the structure of the fish assemblages in these lakes, as they were similar in diversity and evenness, which reflected in similar ALMS-IBI scores of regular biotic integrity. Similarity in fish assemblages of island lakes in the same region during the high water period was also observed by Freitas *et al.* (2010).

Regarding CM lakes, the lower biotic integrity scores for these lakes may be related to their environmental conditions (higher temperatures, less transparency and high conductivity), which are probably influenced by the influx of sediments and suspended material from the main channel of the Solimões River, to which these lakes are directly connected in the high water period. In addition, the current anthropic impacts in the CM lakes, such as pollution and habitat degradation by human presence and commercial fishing, probably also contributed to the lower ALMS-IBI scores for these lakes.

Changes in biotic integrity reflect the trophic structure of species (Bozzetti and Schulz 2004; Costa and Schulz 2010), and the dominance of one or a few species over the others (Casatti *et al.* 2009). A greater abundance and dominance of fish with omnivorous feeding habits and migratory behavior was recorded in the CM lakes, particularly sardines, which normally live in shoals (Ponte *et al.* 2016).

## CONCLUSIONS

Defining metrics that represent the effects of fisheries (particularly commercial fisheries) in the Amazon has been a challenge. Our results suggest that the ALMS-IBI was successful in characterizing fish assemblages in lakes in the lower Solimões River using a few selected biotic parameters, and may be a useful complementary tool for the evaluation of biotic integrity in different categories of ALMS lakes. Future studies should consider samples from a wider geographic range and the relationship between biotic and abiotic metrics in different lake categories to improve the ALMS-IBI. Despite the limitations of our study, our results show the benefits of the ALMS regarding the evaluation of management strategies for the sustainable use and conservation of biodiversity.

## ACKNOWLEDGMENTS

We are grateful to the fishers involved in the management system of Paciência Island for their support during the research.

## REFERENCES

Almeida, O.T.; Lorenzen, K.; McGrath, D.G. 2009. Fishing agreements in the lower Amazon: for gain and restraint. *Fisheries Management and Ecology*, 16: 61–67.

Amaral, E.; Sousa, I.S.; Gonçalves, A.C.T.; Carvalho, G.; Braga, R.; Ronan, P.; Vilena, J. 2011. *Manejo de Pirarucus (Arapaima gigas) em Lagos de Várzea de Uso Compartilhado entre Pescadores Urbanos e Ribeirinhos*. Instituto de Desenvolvimento Sustentável Mamirauá – IDSM/OS/MCTI, Tefé, 50p.

Aquino, A.S.; Bocarde, F.; Lima, N.A.S.; Ruffino, M.L. 2007. Gestão participativa no manejo de recursos pesqueiros na Amazônia. In: Prastes, A.P.; Blanc, D. (Org.). *Áreas Aquáticas Protegidas como Instrumento de Gestão Pesqueira*. MMA/SBF, Brasília, 272p.

Araújo, F.G. 1998. Adaptação do índice de integridade biótica usando a comunidade de peixes para o rio Paraíba do Sul. *Revista Brasileira de Biologia,* 58: 547–558.

Barbour, M.T.; Gerritsen, J.; Griffith, G.E.; Frydenborg, R.; McCarron, E.; White, J.S.; Bastian, M.L. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society,* 15: 185–211.

Barthem, R.B.; Goulding, M. 2007. *An Unexpected Ecosystem: the Amazon as Revealed by Fisheries.* Amazon Conservation Association (ACA)/Sociedade Civil Mamirauá, Belém, 241p.

Benatti, J.H.; McGrath, D.G.; Oliveira, A.C.M.D. 2003. Políticas públicas e manejo comunitário de recursos naturais na Amazônia. *Ambiente & Sociedade*, 6: 137-154.

Bellwood, D.R.; Hoey, A.S.; Howard, C.J. 2003. Limited functional redundancy in high diversity systems: resilience and ecosystem function of coral reefs. *Ecology Letters,* 6: 281–285.

Both, C.; Solé, M.; Santos, T.G.; Cechin, S.Z. 2009. The role of spatial and temporal descriptors for neotropical tadpole communities in southern Brazil. *Hydrobiologia*, 624: 125–138.

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

Campos-Silva, J.V.; Peres, C.A. 2016. Community-based management induces rapid recovery of a high-value tropical freshwater fishery. *Scientific Reports,* 6: 34745.

Campos-Silva, J.V.; Peres, C.A.; Antunes, A.P; Valsecchi, J.; Pezzuti, J. 2017. Community-based population recovery of overexploited Amazonian wildlife. *Perspectives in Ecology and Conservation*, 15: 266–270.

Carter, S.K.; Fleisham, E.; Leinwand, I.I.F.; Flather, C.H.; Carr, N.B.; Fogarty, F.A.; *et al.* 2019. Quantifying ecological integrity of terrestrial systems to inform management of multiple-use public lands in the United States. *Environmental Management,* 64: 1–19. doi.org/10.1007/s00267-019-01163-w

Carvalho, D.R.; Leal, C.G.; Junqueira, N.T.; Castro, M.A.; Fagundes, D.C.; Alves, C.B.M.; Hughes, R.M.; Pompeu, P.S. 2017. A fish-based multimetric index for Brazilian savanna streams. *Ecological Indicators,* 77: 386–396.

Casatti, L.; Ferreira, C.P.; Langeani, F. 2009. A fish-based biotic integrity index for assessment of lowland streams in southeastern Brazil. *Hydrobiology,* 623: 173–189.

Castello, L.; McGrath, D.G.; Beck, P.S.A. 2011. Resource sustainability in small-scale fisheries in the Lower Amazon floodplains. *Fisheries Research,* 110: 356–364.

Castro, F.; McGrath, D. 2001. O manejo comunitário de lagos na Amazônia. *Parcerias Estratégicas,* 12: 112–126.

Costa, P.; Schulz, U. 2010. The fish community as an indicator of biotic integrity of the streams in the Sinos River basin, Brazil. *Brazilian Journal of Biology,* 70: 1195–1205.

Fausch, K.D.; Lyons, J.; Karr, J.R.; Angermeier, P.L. 1990. Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium,* 8: 123–144.

Fitzpatrick, B.M.; Harvey, E.S.; Langlois, T.J.; Babcock, R.; Twiggs, E. 2015. Effects of fishing on fish assemblages at the reefscape scale. *Marine Ecology Progress Series,* 524: 241–253.

Freitas, C.E.C.; Siqueira-Souza, F.K.; Guimarães, A.R.; Santos, F.A; Santos, I.L. 2010. Interconnectedness during high water maintains similarity in fish assemblages of island floodplain lakes in the Amazonian Basin. *Zoologia*, 27: 931–938.

Fricová, K.; Růžičková, J.; Hřebík, Š. 2007. Benthic macroinvertebrates as indicators of ecological integrity of lotic ecosystems in the Šumava National Park, Czech Republic. *Silva Gabreta,* 13: 39-55.

Froese, R.; Pauly, D. 2018. FishBase. (www.fishbase.org). Accessed on 14 Apr 2018.

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. *Freshwater Biology,* 40: 367–383.

Hanna, D.E.L.; Tomscha, S.A.; Ouellet-Dallaire, C.; Bennett, E.M. 2018. A review of riverine ecosystem service quantification: Research gaps and recommendations. *Journal of Applied Ecology*, 55: 1299–1311.

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. *Canadian Journal of Fisheries and Aquatic Sciences,* 55: 1618–1631.

Jia, Y.; Jiang, Y.; Lui, Y.; Sui, X.; Feng, X.; Zhu, R.; Chen, Y. 2021. Understanding trophic structure variation in fish assemblages of subtropical shallow lakes: Combined effects of ecosystem size, productivity, and disturbance. *Ecological Indicators*, 129: 107924.

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

Karr, J.R.; Chu, E.W. 1999. *Restoring Life in Running Waters: Better Biological Monitoring*. 1st ed. Island Press, Washington DC, 220p.

Magurran, A.E. 2004. *Measuring Biological Diversity*. Blackwell Science, Oxford, 261p.

Mancini, L.; Formichetti, P.; Anselmo, A.; Tancioni, L.; Marchini, S.; Sorace, A. 2005. Biological quality of running waters in protected areas: the influence of size and land use. *Biodiversity and Conservation*, 14: 351–364.

Medeiros-Leal, W.M.; Castello, L.; Freitas, C.E.C.; Siqueira-Souza, F.K. 2021. Single-species co-management improves fish assemblage structure and composition in a tropical river. *Frontiers in Ecology and Evolution,* 9: 604170.

Melo, T.L.; Tejerina-Garro, F.L.; Melo, C.E. 2007. Biological diversity of the fish assemblage in the Lower das Mortes River, Mato Grosso, Brazil. *Revista Brasileira de Zoologia*, 24: 657-665.

Mérona, B.; Rankin-de-Mérona, J. 2004. Food resource partitioning in a fish community of the central Amazon floodplain. *Neotropical Ichthyology,* 2: 75–84.

McGrath, D.; Câmara, E.; Futemma, C. 1998. Manejo comunitário de lagos de várzeas e o desenvolvimento sustentável da pesca na amazônia. *Novos Cadernos NAEA,* 1: 1-23. doi.org/10.5801/ncn.v1i2.4

McGrath, D.G.; De Castro, F.; Futemma, C.; De Amaral, B.D.; Calabria, J. 1993. Fisheries and the evolution of resource management on the lower Amazon floodplain. *Human Ecology,* 21 :167-195.

MMA. 2021. Ministério do Meio Ambiente. Biomas. (http://www.mma.gov.br/biomas). Accessed on 04 Oct 2022.

Nolan, K.S.; Fabré, N.N.; Batista, V.S. 2009. Landscape variables affecting fishery yield in lake systems of the Central Amazon region, Brazil. *Journal of Applied Ichthylogy,* 25: 294–298.

Paine, R.T. 1966. Food web complexity and species diversity. *The American Naturalist,* 100: 65-75.

Parrish, J.D.; Braun, D.P.; Unnasch, R.S. 2003. Are we conserving what we say we are? Measuring ecological integrity within protected areas. *BioScience,* 53: 851-860.

Pellizzaro, P.C.; Hardt, L.P.A.; Hardt, C.; Hardt, M.; Sehli, D.A. 2015. Gestão e Manejo de áreas naturais protegidas: contexto internacional. *Ambiente & Sociedade,* 18: 21–40.

Petesse, M.L.; Petrere-Jr, M.; Spigolon, R.J. 2007. Adaptation of the reservoir fish assemblage index (RFAI) for assessing the Barra Bonita reservoir (São Paulo, Brazil). *River Research and Applications,* 21: 188–193.

Petesse, M.L.; Siqueira-Souza, F.K.; Freitas, C.E.; Petrere, M. 2016. Selection of reference lakes and adaptation of a fish multimetric index of biotic integrity to six amazon floodplain lakes. *Ecological Engineers,* 97: 535–544.

Poikane, S.; Ritterbusch, D.; Argillier, C.; Białokoz, W.; Blabolil, P.; Breine, J.; *et al.* 2017. Response of fish communities to multiple pressures: Development of a total anthropogenic pressure intensity index. *Science of the Total Environmental,* 586: 502–511.

Polaz, C.N.M.; Ferreira, F.C.; Petrere-Jr., M. 2017. The protected areas system in Brazil as a baseline condition for wetlands management and fish conservancy: the example of the Pantanal National Park. *Neotropical Ichthyology,* 15: e170041.

Ponte, S.C.S.; Ferreira, L.C.; Bittencourt, S.C.S.; Queiroz, H.L.; Zacardi, D.M. 2016. Spatial and temporal variation of larvae Triportheus (Characiformes, Triportheidae) in the middle Solimões River, Central Amazon, Brazil. *Acta of Fisheries and Aquatic Resources*, 4: 71-81.

Prudente, B.S.; Pompeu, P.S.; Montag, L. 2018. Using multimetric indices to assess the effect of reduced impact logging on ecological integrity of Amazonian streams. *Ecological Indicators,* 91: 315–323.

R Development Core Team 2020. R: A language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.

Reid, D.M.; Anderson, D.M.; Henderson, B.A. 2001. Restoration of lake trout in Parry Sound, Lake Huron. *North American Jounal of Fisheries Management,* 21: 156–169.

Reza, M.I.H.; Abdullah, S.A. 2011. Regional index of ecological integrity: A need for sustainable management of natural resources. *Ecological Indicators,* 11: 220-229.

Santos, G.M.; Ferreira, E.J.G.; Zuanon, J.A.S. 2009. *Peixes Comerciais de Manaus*. 2nd ed. Editora INPA, Manaus, 146p.

Saunders, D.L.; Meeuwig, J.J.; Vincent, C.J. 2002. Freshwater protected areas: Strategies for conservation. *Conservation Biology,* 16: 30–41.

Schram, S.T.; Selgeby, J.H.; Bronte, C.R.; Swanson, B.L. 1995. Population recovery and natural recruitment of lake trout at Gull Island shoal, Lake Superior. *Journal of Great Lakes Research,* 21: 225–232.

SDS. 2011. Secretaria de Estado do Meio Ambiente e Desenvolvimento Sustentável do Estado do Amazonas. Instrução Normativa # 002/2011. (https://diario.imprensaoficial.am.gov.br/portal/visualizacoes/pdf/13983/#/p:21/e:13983). Accessed on 05 Oct 2022.

Seegert, G. 2000. The development, use and misuse of biocriteria with an emphasis on the index of biotic integrity. *Environmental Science e Policy,* 3: 51-58.

Silvano, R.A.; Ramires, M.; Zuanon, J. 2009. Effects of fisheries management on fish communities in the floodplain lakes of a Brazilian Amazonian Reserve. *Ecology of Freshwater Fish,* 18: 156–166.

Silvano, R.A.; Hallwass, G.; Juras, A.A.; Lopes, P.F. 2017. Assessment of efficiency and impacts of gillnets on fish conservation in a tropical freshwater fishery. *Aquatic Conservation,* 27: 521-533.

Soares, M.G.M.; Costa, E.L.; Siqueira-Souza, F.K.; Anjos, H.D.B.; Yamamoto, K.C.; Freitas, C.E.C. (Org.). 2008. *Peixes de Lagos do Médio Rio Solimões*. 1st ed. EDUA, Manaus, 176p.

Sobczak, J.R.S.; Valduga, A.T.; Restello, R.M.; Cardoso, R.I. 2013. Conservation unit and water quality: the influence of environmental integrity on benthic macroinvertebrate assemblages. *Acta Limnologica Brasiliensia,* 25: 442-450.

Suski, C.D.; Cooke, S.J. 2007. Conservation of aquatic resources through the use of freshwater protected areas: Opportunities and challenges. *Biodiversity and Conservation,* 16: 2015–2029.

Tierney, G.L.; Faber-Langendoen, D.; Mitchell, B.R.; Shriver, G.W..; Gibbs, J.P. 2009. Monitoring and evaluating the ecological integrity of forest ecosystems. *Frontiers in Ecology and the Environmental,* 7: 308–316.

Vári, A.; Podschun, S.A.; Erős, T.; Thomas, H.; Beáta, P.; Ioan-Cristian, I. *et al.* 2021. Freshwater systems and ecosystem services: Challenges and chances for cross-fertilization of disciplines. *Ambio*, 51: 135-151.

Whittier, T.R.; Stoddard, J.L.; Larsen, D.P.; Herlihy, A.T. 2007. Selecting reference sites for stream biological assessments: best professional judgment or objective criteria. *Journal of the North American Benthological Society,* 26: 349–360.

**RECEIVED:** 06/05/2022
**ACCEPTED:** 08/09/2022
**ASSOCIATE EDITOR:** Cristhiana Röpke

## **SUPPLEMENTARY MATERIAL** (only available in the electronic version)

Andrade *et al*. Evaluation of fisheries management strategies using a biotic integrity index in floodplain lakes in the lower Solimões River, Amazonas, Brazil

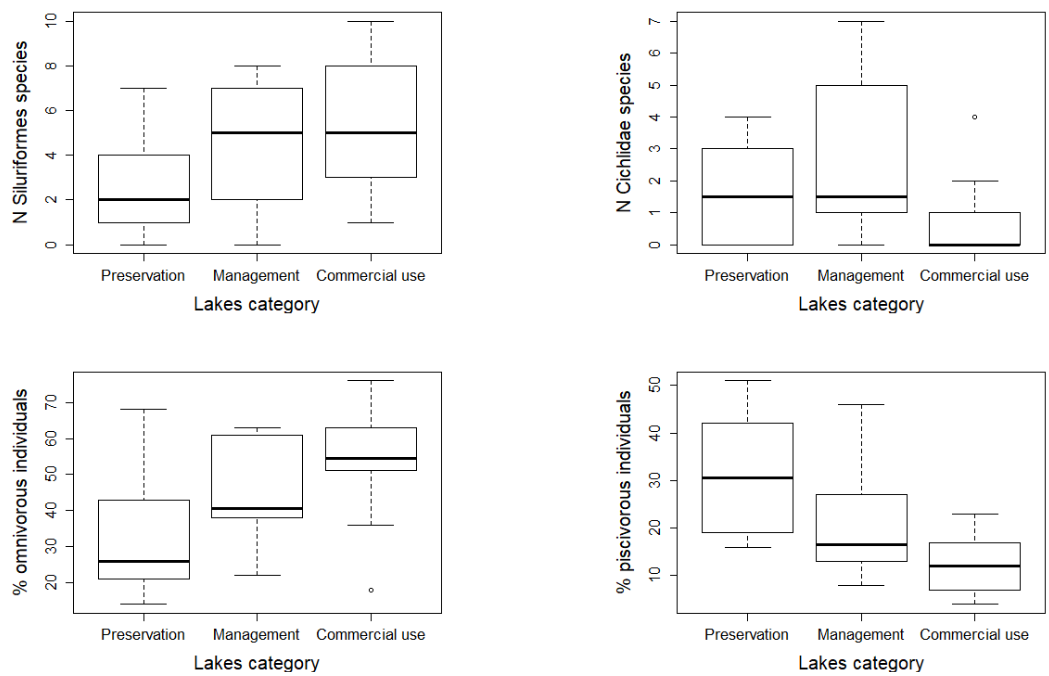


**Figure S1.** Sensitivity test for metrics "N Siluriformes species", "N Cichlidae species", "% omnivorous individuals" and "% piscivorous individuals" among ALMS lake categories for the development of an index of biotic integrity for ALMS (ALMS-IBI). Values for each lake category are from five sampling events events in each of two floodplain lakes on Paciência Island (lower Solimões River, Amazonas, Brazil) between April 2017 and April 2018.

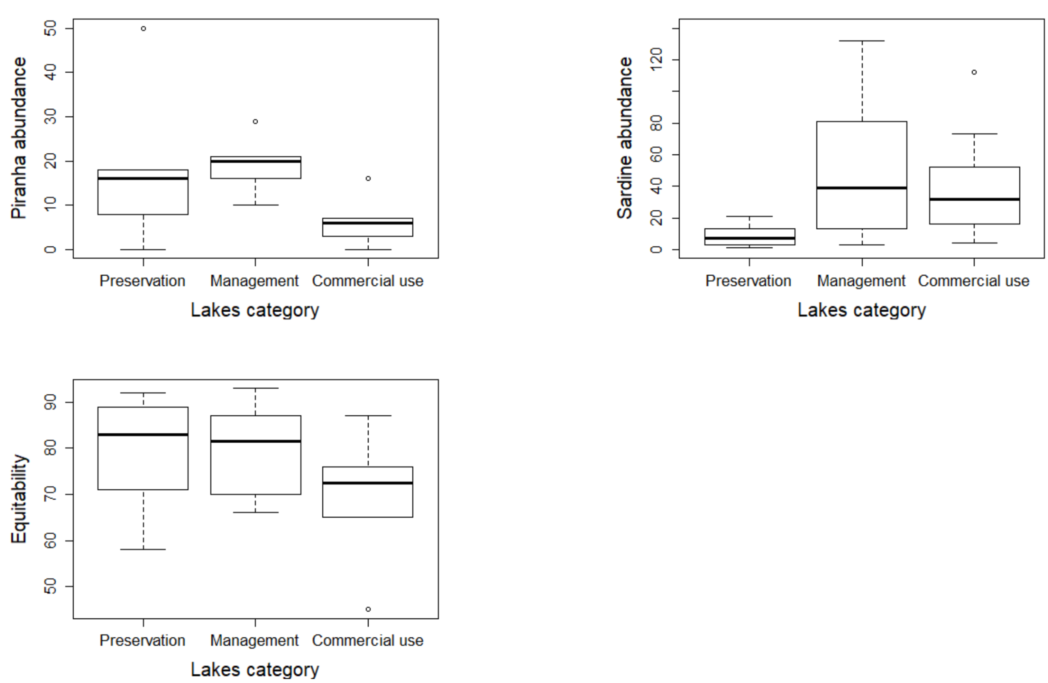


**Figure S2.** Sensitivity test for metrics "Piranha, abundance", "Sardine abundance" and "Equitability" among ALMS lake categories for the development of an index of biotic integrity for ALMS (ALMS-IBI). Values for each lake category are from five sampling events events in each of two floodplain lakes on Paciência Island (lower Solimões River, Amazonas, Brazil) between April 2017 and April 2018.

**Table S1.** Candidate metrics for the development of an index of biological integrity for the Amazon Lakes Management System (ALMS-IBI). ER = metric response to disturbance: decrease (↓) or increase (↑) in presence of the environmental disturbance, according to Petesse *et al.* (2016) and study present.

| Category | Metrics | ER | Description |
|---|---|---|---|
| Species richness | Total number of species | ↓ | Measure of diversity and generally decreases with environmental degradation. |
| | Number of species with moderate/high vulnerability | ↓ | Assesses the importance of species with moderate and high vulnerability in relation to fishing, according to Fishbase (2018). |
| | Number of orders | ↓ | Higher richness of orders is expected in an environment with good quality. |
| | Number of families | ↓ | Higher richness of families is expected in an environment with good quality. |
| | Number of Characiformes species | ↑ | Water column Characiforms are ecological equivalents of the "sunfish" group originally proposed by Karr (1981). Used by Petesse *et al.* (2016) to indicate poor quality conditions. Characiformes are widely distributed in tropical environments and are representative of the floodplain fish assemblage. |
| | Number of Siluriformes species | ↓ | Siluriformes can be used as an ecological equivalent of the "darter" group used by Karr (1981). Assesses the conditions of the bentho-pelagic habitat and migratory fish. |
| | Number of Perciformes species | ↑ | Used as the ecological equivalent of the "sunfish" group in tropical and subtropical environments (Hughes *et al*. 1999). Higher values represents environments with poor conditions. |
| | Number of Cichlidae species | ↓ | Suggested by Hocutt *et al*. (1994) and Hugueny *et al*. (1996) as indicators of margin conditions, since these fish use this environment for spawning and nest building (Hughes *et al*. 1999). High numbers indicate good quality of the habitat. |
| | Number of omnivorous species | ↑ | Measures changes in the food chain. Omnivores alter their diet according to food availability. Higher numbers indicate low quality environments. |
| | Number of piscivorous species | ↓ | Species with specialist feeding habit Lower numbers reflect low habitat quality. |
| Trophic category | Percentage of omnivorous individuals | ↑ | Omnivorous species can indicate poor quality environments as they can alter their diet with changes in the environment. |
| | Percentage of detritivorous individuals | ↓ | Species with specialist feeding habits. Higher proportions reflect good environmental quality. |
| | Percentage of herbivorous individuals | ↓ | Species with specialist feeding habits. Higher proportions reflect good environmental quality. |
| | Percentage of insectivorous individuals | ↓ | Species with specialist feeding habits. Higher proportions reflect good environmental quality. |
| | Percentage of piscivorous individuals | ↓ | Species with specialist feeding habits. Higher proportions reflect good environmental quality. |
| | Percentage of planktivorous individuals | ↓ | Species with a specialist feeding habits. Higher proportions indicate good environmental quality. |
| Abundance | Abundance of sardines | ↑ | Sardines represent the omnivorous feeding habit in floodplain environments and are highly abundant in lakes impacted by fisheries. Higher abundance indicates low environmental quality. |
| | Abundance of piranhas | ↓ | Piranhas represent the piscivorous feeding habit and are highly abundant in flood-plain environments. Higher abundance indicates good environmental quality. |
| | Abundance of branquinhas | ↓ | Branquinhas are the most abundant group of detritivores in floodplain environments. Higher abundance indicates good environmental quality. |
| | Total number of individuals | ↓ | High quality environments support larger numbers of individuals. |
| | Total number of individuals with moderate/high vulnerability | ↓ | Number of individuals belonging to species that are sensitive or vulnerable to environmental degradation. Larger numbers indicate good environmental quality. |
| | Dominance | ↑ | Assesses the relative abundance of species tolerant to environmental degradation. Low values indicate good environmental quality. |
| | Equitability | ↓ | Assesses the distribution of abundances of species. Higher values indicate good environmental quality. |

**Table S2.** Number of individuals of each fish species sampled in three lakes of different categories according to the Amazon Lakes Management System (ALMS) on Paciência Island, lower Solimões River (Brazil). Data from five sampling events between April 2017 and April 2018. ALMS category: PR = preservation lakes; MN = management lakes; CM = commercial use lakes.

| Order/family/species | Lake category | | |
|---|---|---|---|
| | PR | MN | CM |
| **Gymnotiformes** | - | 3 | 1 |
| Rhamphichthyidae | - | 1 | - |
| *Rhamphichthys rostratus* (Linnaeus, 1766) | - | 1 | - |
| Gymnotidae | - | 1 | - |
| *Electrophorus electricus* (Linnaeus, 1766) | - | 1 | - |
| Sternopygidae | - | 1 | 1 |
| *Eigenmannia limbata* (Schreiner & Miranda Ribeiro, 1903) | - | 1 | - |
| *Eigenmannia macrops* (Boulenger, 1897) | - | - | 1 |
| **Characiformes** | 491 | 2258 | 772 |
| Acestrorhynchidae | 29 | 81 | 14 |
| *Acestrorhynchus falcatus* (Blochi, 1794) | - | 5 | 3 |
| *Acestrorhynchus falcirostris* Cuvier, 1819) | 29 | 76 | 11 |
| Anostomidae | 41 | 102 | 59 |
| *Leporinus fasciatus* (Bloch, 1794) | - | 3 | - |
| *Leporinus friderici* (Bloch, 1794) | 1 | 14 | 1 |
| *Leporinus trifasciatus* Steindachner, 1876 | 3 | 3 | - |
| *Rhytiodus argenteofuscus* Kner, 1858 | - | - | 1 |
| *Rhythiodus microlepis* Kner, 1858 | 19 | 31 | 40 |
| *Schizodon fasciatum* Spix & Agassiz, 1829 | 18 | 28 | 15 |
| *Schyzodon vittatus* (Valenciennes, 1850) | - | 23 | 2 |
| Cynodontidae | 3 | 25 | 24 |
| *Raphiodon vulpinus* Spix & Agassiz, 1829 | 3 | 30 | 23 |
| *Cynodon gibbus* (Agassiz, 1829) | - | 1 | - |
| *Hydrolycus scomberoides* (Cuvier, 1819) | - | 4 | 1 |
| Bryconidae | - | 11 | - |
| *Brycon amazonicus* (Spix & Agassiz, 1829) | - | 7 | - |
| *Brycon cephalus* (Günther, 1864) | - | 3 | - |
| *Brycon hilarii* (Valenciennes, 1850) | - | 1 | - |
| Characidae | 9 | 76 | 18 |
| *Chalceus macrolepdotus* Cuvier, 1818 | 6 | 53 | 1 |
| *Charax gibbosus* (Linnaeus, 1758) | - | 3 | 1 |
| *Roeboides myersii* Gill, 1870 | 2 | 19 | 15 |
| *Tetragonopterus argenteus* Cuvier, 1816 | 1 | 1 | 1 |
| Curimatidae | 114 | 800 | 118 |
| *Curimata inormata* (Vari, 1989) | - | 7 | - |
| *Curimata ocellata* (Eigenmann & Eigenmann, 1889) | 3 | 2 | 4 |
| *Curimata vittata* (Kner, 1858) | 14 | 13 | 4 |
| *Curimatela meyeri* (Steindachner, 1882) | 1 | 22 | 7 |
| *Potamorhina altamazonica* Eigenmann & Eigenmann, 1889 | 14 | 234 | 27 |
| *Potamorhina latior* (Spix & Agassiz, 1829) | 80 | 120 | 45 |
| *Psectogaster amazônica* Eigenmann & Eigenmann, 1889 | 2 | 77 | 11 |
| *Psectogaster rutiloides* (Kner, 1858) | - | 325 | 4 |
| *Steindachnerina bimaculata* (Steindachner, 1876) | - | - | 6 |

**Table S2.** Continued.

| **Order/family/species** | **Lake category** | | |
|---|---|---|---|
| | **PR** | **MN** | **CM** |
| Erythrinidae | 6 | 13 | 16 |
| *Hoplerythrinus unitaeniatus* (Spix & Agassiz, 1829) | - | 3 | 10 |
| *Hoplias malabaricus* (Bloch, 1794) | 6 | 10 | 6 |
| Hemiodontidae | 8 | 45 | 33 |
| *Anodus elongatus* (Agassiz, 1829) | - | 7 | 10 |
| *Hemiodus immaculatus* Kner, 1858 | 2 | 12 | 10 |
| Hemiodus sp. | 1 | 25 | 11 |
| *Hemiodus unimaculatus* (Bloch, 1794) | 5 | 1 | 2 |
| Iguanodectidae | 2 | 1 | - |
| *Bryconops caudomaculatus* (Günther, 1864) | 1 | 1 | - |
| *Bryconops melanurus* (Bloch, 1794) | 1 | - | - |
| Prochilodontidae | 42 | 76 | 13 |
| *Prochilodus nigricans* Agassiz, 1829 | 19 | 23 | 10 |
| *Semaprochilodus insignis* (Jardine & Schomburgk, 1841) | 22 | 19 | 2 |
| *Semaprochilodus taeniurus* (Valenciennes, 1817) | 1 | 34 | 1 |
| Serrasalmidae | 148 | 426 | 91 |
| *Colossoma macropomum* Cuvier, 1818) | 23 | 20 | 8 |
| *Metynnis altidorsalis* (Ahl, 1923) | - | 1 | - |
| *Metynnis argenteus* (Ahl, 1923) | - | 8 | - |
| *Metynnis hypsauchen* (Müller & Troschel, 1844) | - | 24 | - |
| *Metynnis luna* Cope, 1878 | - | - | 1 |
| *Myloplus asterias* (Müller & Troschel, 1844) | - | 3 | - |
| *Myloplus rubripinnis* (Müller & Troschel, 1844) | - | 2 | - |
| *Mylossoma aureum* (Spix & Agassiz, 1829) | - | 15 | 14 |
| *Mylossoma duriventre* (Cuvier, 1818) | 13 | 215 | 29 |
| *Piaractus brachypomus* Cuvier, 1818) | 5 | 7 | 1 |
| *Prystobrycon calmoni* (Steindachner, 1908) | - | 2 | 1 |
| *Pygocentrus nattereri* Kner, 1858 | 21 | 69 | 13 |
| *Serrasalmus altispinis* Merckx, Jégu & Santos, 2000 | 11 | 49 | 12 |
| *Serrasalmus compresus* Jégu, Leão & Santos, 1991 | - | 4 | - |
| *Serrasalmus elongatus* Kner, 1858 | 6 | 18 | 4 |
| *Serrasalmus maculatus* Kner, 1858 | 61 | 71 | 6 |
| *Serrasalmus rombeus* (Linnaeus, 1766) | 8 | 18 | 1 |
| Triportheidae | 89 | 497 | 396 |
| *Triportheus albus* Cope, 1872 | 67 | 298 | 268 |
| *Triportheus angulatus* (Spix & Agassiz, 1829) | 21 | 187 | 106 |
| *Triportheus auritrus* (Valenciennes, 1850) | 1 | 12 | 22 |
| **Clupeiformes** | 11 | 59 | 73 |
| Pristigasteridae | 11 | 59 | 42 |
| *Pristigaster cayana* (Cuvier, 1829) | 6 | - | - |
| *Pellona castelnaeana* (Valenciennes, 1847) | 2 | 1 | 12 |
| *Pellona flavipinis* (Valenciennes, 1847) | 3 | 58 | 30 |

**Table S2.** Continued.

| Order/family/species | Lake category | | |
|---|---|---|---|
| | PR | MN | CM |
| **Osteoglossiformes** | 33 | 4 | 1 |
| Arapaimidae | 21 | - | - |
| *Arapaima gigas* (Schinz, 1822) | 21 | - | - |
| Osteoglossidae | 12 | 4 | 1 |
| *Osteoglossum bicirrhossum* Cuvier, 1829) | 12 | 4 | 1 |
| **Cichliformes** | 42 | 83 | 11 |
| Cichlidae | 42 | 80 | 4 |
| *Acarichthys heckelii* (Müller & Troschel, 1849) | 17 | 45 | 3 |
| *Acaronia nassa* (Heckel, 1840) | 1 | 1 | 1 |
| *Astronotus crassipinnis* (Heckel, 1840) | 1 | - | - |
| *Aequidens tetramerus* (Heckel, 1840) | - | 1 | - |
| *Chaetobranchus flavescens* Heckel, 1840 | 2 | 1 | - |
| *Cichassoma amazonarum* Kullander, 1983 | - | 1 | - |
| *Cichla monoculus* Spix & Agassiz, 1831 | 2 | 7 | - |
| *Geophagus proximus* Castelnau, 1855 | - | 7 | - |
| *Heros notatus* (Jardine, 1843) | - | 1 | - |
| *Heros severus* Heckel, 1840 | 4 | 3 | - |
| *Mesonauta festivus* (Heckel, 1840) | 2 | 8 | - |
| *Satanoperca acuticeps* (Heckel, 1840) | 2 | 2 | - |
| *Satanoperca jurupari* (Heckel, 1840) | 11 | 3 | - |
| **Perciformes** | - | 3 | 7 |
| Sciaenidae | - | 3 | 7 |
| *Plagioscion squamosissimus* (Heckel, 1840) | | 3 | 7 |
| **Siluriformes** | 74 | 144 | 503 |
| Auchenipteridae | 33 | 37 | 117 |
| *Ageneiosus inermis* (Linnaeus, 1766) | 2 | 2 | 1 |
| *Ageneiosus ucayalensis* Castelnau, 1855 | 13 | 13 | 74 |
| *Auchenipterichthys punctatus* (Valenciennes, 1840) | - | 1 | - |
| *Auchenipterus nuchalis* (Spix & Agassiz, 1829) | 2 | 4 | 13 |
| *Centromochlus heckelii* (De Filippi, 1853) | - | 1 | - |
| *Trachelyopterus galeatus* (Linnaeus, 1766) | 16 | 16 | 27 |
| *Tympanopleura rondoni* | - | - | 2 |
| Callichthyidae | 14 | 14 | 15 |
| *Hoplosternum litoralle* (Hancock, 1828) | 14 | 14 | 15 |
| Doradidae | 4 | 9 | 25 |
| *Nemadoras humeralis* (Kner,1855) | - | - | 3 |
| *Oxydoras niger* (Valenciennes, 1821) | - | 2 | 6 |
| *Platlydoras hancockii* (Valenciennes, 1840) | - | 1 | - |
| *Pterodoras granulosus* (Valenciennes, 1821) | 4 | 6 | 16 |
| Loricariidae | 16 | 30 | 58 |
| *Anchistrus dolichopterus* Kner, 1854 | 1 | - | - |
| *Hypoptopoma gulare* Cope, 1878 | - | - | 1 |
| *Hypoptopoma incognitum* Aquino & Schaefer, 2010 | - | - | 1 |
| *Hypostomus cochliodon* Kner,1854 | - | - | 1 |

**Table S2.** Continued.

| Order/family/species | Lake category | | |
|---|---|---|---|
| | PR | MN | CM |
| *Loricariichthys acutus* (Valenciennes, 1840) | 3 | 3 | - |
| *Loricaria cataphracta* Linnaeus, 1758 | 7 | 11 | 21 |
| *Loricariichthys nudirostris* Kner,1853 | 4 | - | 4 |
| *Pterygoplichthys pardalis* (Castelnau, 1855) | 1 | 16 | 30 |
| Pimelodidae | 7 | 58 | 288 |
| *Calophysus macropterus* (Lichtenstein, 1819) | - | 5 | 6 |
| *Hypophthalmus edentatus* Spix & Agassiz, 1829 | - | 4 | 39 |
| *Hypophthalmus fimbriatus* Kner, 1854 | - | - | 1 |
| *Hypophthlalmus marginatus* Valenciennes, 1840 | 1 | - | 3 |
| *Phractocephalus hemiliopterus* (Bloch & Schneider, 1801) | - | - | 1 |
| *Pimelodus blochii* Valenciennes, 1840 | 6 | 37 | 184 |
| *Pinirampus pirinampu* (Spix & Agassiz, 1829) | - | 1 | 6 |
| *Pseudoplatystoma tigrinum* (Valenciennes, 1840) | - | 1 | 2 |
| *Pseudoplatystoma punctifer* Castelnau, 1855) | - | 1 | - |
| *Sorubim elongatus* Littmann, Burr, Schmidt & Isern, 2001 | - | 4 | 38 |
| *Sorubim lima* (Bloch & Schneider, 1801) | - | 5 | 8 |
| **Total abundance (N)** | **651** | **2553** | **1361** |

**Table S3.** Metrics range test for the development of an index of biological integrity for the Amazon Lakes Management System (ALMS-IBI), according to Petesse *et al.* (2016) and the present study.

| Category | Candidate metrics | Range |
|---|---|---|
| Species richness | Total number of species | 31 |
| | Number of species with moderate/high vulnerability | 17 |
| | Number of orders | 5 |
| | Number of families | 13 |
| | Number of Characiformes species | 7 |
| | Number of Siluriformes species | 38 |
| | Number of Perciformes species | 10 |
| | Number of Cichlidae species | 7 |
| | Number of piscivorous species | 19 |
| | Number of omnivorous species | 15 |
| Trophic category | Percentage of omnivorous individuals | 62 |
| | Percentage of detritivorous individuals | 62 |
| | Percentage of herbivorous individuals | 27 |
| | Percentage of insectivorous individuals | 8 |
| | Percentage of piscivorous individuals | 47 |
| | Percentage of planctivorous individuals | 19 |
| Abundance | Sardine abundance | 48 |
| | Piranha abundance | 602 |
| | Branquinha abundance | 121 |
| | Total number of individuals | 48 |
| | Total number of individuals with moderate/high vulnerability | 131 |
| | Equitability | 81 |
| | Dominance | 144 |

**Table S4.** Correlation test among metrics for the development of an index of biological integrity for the Amazon Lakes Management System (ALMS-IBI). sp silur= number of siluriform species; sp cichlid = Cichlidae richness; % ind oniv = percentage of omnivorous individuals; % ind pisciv = percentage of piscivorous individuals; abund sard = sardine abundance; abund piranh = piranha abundance; equitab = equitability.

| | sp silur | sp cichlid | % ind oniv | % ind pisciv | abund sard | abund piranh | equitab |
|---|---|---|---|---|---|---|---|
| **sp silur** | 1.00 | | | | | | |
| **sp cichlid** | -0.20 | 1.00 | | | | | |
| **% ind oniv** | -0.16 | 0.06 | 1.00 | | | | |
| **% ind pisciv** | -0.40 | 0.01 | -0.40 | 1.00 | | | |
| **abund sard** | 0.12 | 0.10 | 0.62 | -0.53 | 1.00 | | |
| **abund piranh** | 0.12 | 0.39 | -0.24 | 0.12 | -0.006 | 1.00 | |
| **equitab** | -0.22 | 0.04 | -0.32 | 0.69 | -0.38 | 0.18 | 1.00 |

**Table S5.** Selection tests for candidate metrics of fish assemblages to compose an index of biotic integrity (IBI) for use within the context of the Amazon Lakes Management System (ALMS). (√) indicates the metric passed the test; (-) indicates that the metric did not pass the test.

| Metric category | Metric | Range test | Sensitivity test | Redundancy test |
|---|---|---|---|---|
| Species richness | Total number of species | √ | - | - |
| | Number of species with moderate to high vulnerability | √ | - | - |
| | Number of orders | √ | - | - |
| | Number of families | √ | - | - |
| | Number of Characiformes species | √ | - | - |
| | Number of Siluriformes species | √ | √ | √ |
| | Number of Perciformes species | √ | - | - |
| | Number of Cichlidae species | √ | √ | √ |
| | Number of omnivorous species | √ | - | - |
| | Number of piscivorous species | √ | - | - |
| Trophic category | Percentage of omnivorous individuals | √ | √ | √ |
| | Percentage of detritivorous individuals | √ | - | - |
| | Percentage of herbivorous individuals | √ | - | - |
| | Percentage of insectivorous individuals | √ | - | - |
| | Percentage of piscivorous individuals | √ | √ | √ |
| | Percentage of planktivorous individuals | √ | | |
| Abundance | Abundance of sardines | √ | √ | √ |
| | Abundance of piranhas | √ | √ | √ |
| | Abundance of branquinhas | √ | - | - |
| | Total number of individuals | √ | - | - |
| | Total number of individuals with moderate to high vulnerability | √ | - | - |
| | Dominance | √ | - | - |
| | Equitability | √ | √ | √ |