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Ecosystem indicators to assess the sustainability of multispecific artisanal fisheries in face of the environmental changes in the Southwest Amazon Basin

Igor Rechetnicow Alves Sant'Anna[1], Vanessa Reis[2], Danielle Mendonça Pinto[1], Flávia Kelly Siqueira de Souza[3], Raniere Garcez Costa Sousa[1,*], Carolina Rodrigues da Costa Doria[1]

[1]Universidade Federal de Rondônia – Porto Velho (RO), Brazil.
[2]Griffith University – Rivers Institute – Brisbane, Australia.
[3]Universidade Federal do Amazonas – Departamento de Ciências Pesqueiras – Manaus (AM), Brazil.
*Corresponding author: ranieregarcez@unir.br

ABSTRACT
The construction of hydroelectric dams causes several impacts on the aquatic environment and affects its structure and functions with direct consequences for the provision of ecosystem services. Despite its importance, since 1970, fisheries in the Amazon region, especially small-scale artisanal fisheries practiced by traditional populations, have been facing serious problems arising from human and climatic actions. In this study, we evaluated the impacts of dam construction and the environmental responses on fisheries production and fishing sustainability indexes in the Madeira River, which is the largest tributary of the Amazon River, between 2000 and 2019. Fisheries landing data in the city of Porto Velho (RO), Brazil, and large-scale environmental variables were collected for each of the years in the period and were compared using general linear models. We found that dams, as well as changes in hydrological levels, affect fisheries production. The sustainability indexes showed a decline after the completion of the two dams. Moreover, catch-per-unit-of-effort showed a significant decline after the implementation of the dam. We therefore highlight the importance of understanding the factors that affect the sustainability of fisheries in large rivers in order to mitigate the impacts of large dams and other anthropogenic actions.

Keywords: Amazon basin; Madeira River; River dams; Data-limited fisheries.

Indicadores ecossistêmicos para avaliar a sustentabilidade da pesca artesanal multiespecífica diante das mudanças ambientais no sudoeste da Bacia Amazônica

RESUMO
A construção de hidrelétricas causa diversos impactos no meio aquático, afetando sua estrutura e funções, com consequências diretas na prestação de serviços ecossistêmicos. Apesar de sua importância, desde 1970 a pesca na região amazônica, principalmente a pesca artesanal de pequena escala praticada por populações tradicionais, vem enfrentando sérios problemas decorrentes das ações antrópicas e climáticas. Neste estudo, avaliamos os impactos da construção de barragens e as respostas ambientais na produção pesqueira e nos índices de sustentabilidade da pesca no Rio Madeira, o maior afluente do Rio Amazonas, entre 2000 e 2019. Dados de desembarque de peixes na cidade de Porto Velho (RO), Brasil, e variáveis ambientais de grande escala foram coletados para cada um dos anos referidos e comparados por meio de modelos lineares gerais. Descobrimos que as barragens, bem como as mudanças nos níveis hidrológicos, afetam a produção pesqueira. Os índices de sustentabilidade apresentaram queda após a conclusão das duas barragens. Além disso, as capturas por unidade de esforço mostraram declínio significativo após a barragem. Destacamos, portanto, a importância de entender os fatores que afetam a sustentabilidade da pesca em grandes rios, a fim de mitigar os impactos de grandes barragens e outras ações antrópicas.

Palavras-chave: Bacia Amazônica; Rio Madeira; Barragens; Pescarias com dados limitados.

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INTRODUCTION

The Amazon Basin encompasses a remarkably large network of tributaries and floodplains covering an area of 7,000,000 km^2 (Ferreira et al., 2013). It also has a high diversity of fish species, about 2,406, which is equivalent to 15% of the freshwater fish described on the planet (Jézéquel et al., 2020). This richness justifies the importance and success of fisheries in the region (Duponchelle et al., 2021; Fluet-Chouinard et al., 2018; Lowe-McConnell, 1987). It is estimated that Amazonian fisheries generate a revenue of around US$ 400 million every year (Duponchelle et al., 2021), and that around 48,200 commercial fishers and 111,800 subsistence fishers operate in the main channel of the Amazon River, generating 200,000 direct jobs and more than 1,000,000 indirect jobs in the Amazon Basin (Almeida et al., 2010; FAO, 2000).

Despite its importance, since 1970, fisheries in the Amazon region, especially small-scale artisanal fisheries practiced by traditional populations, have faced serious problems arising from human and climatic actions (Cruz et al., 2020; Winemiller et al., 2016). The fish stocks in the Amazon Basin are threatened by human activities that intensify habitat degradation and overexploitation of water resources (Castello & Macedo, 2016), and are especially affected by projects such as hydroelectric dams, ports and waterways, deforestation of floodplain areas, the introduction of invasive species, pollution, and the damming of streams for fish farming (Barthem & Goulding, 2007; Castello et al., 2013; Doria et al., 2020; Sousa et al., 2018; Winemiller et al., 2007).

Among these high impact activities, the construction of dams on large rivers stands out since it promotes numerous structural and functional changes in aquatic ecosystems (Arantes et al., 2021; Fearnside, 2015). Changes in the flood regime, obstacles for fish migration, reduction of oxygen levels above and below dams, and sediment retention are amongst the main changes identified and associated with the construction of dams (Freitas et al., 2012; Freitas et al., 2013). In addition, the construction of large and small hydroelectric dams threatens the regularity of the hydrological regime, inducing the loss of essential habitats for feeding, spawning, and growth, and which serve as a refuge of fish, thus affecting the conservation of aquatic biodiversity (Castello & Macedo, 2016; Fearnside, 2014).

The use of health index indicators for ecosystem studies allows one to understand patterns of structure and functioning of ecosystems over time and identify impacts, as well as allowing one to make predictions (Li et al., 2020). In the fisheries sector, indicators may include indices based on the weight of landings and species size (average weight of an individual and abundance), as well as indicators of trophic dynamics and energy transfer efficiency through food webs (Blanchard et al., 2010; Fabré et al., 2017; Libralato et al., 2008; Lima et al., 2020b). Among the trophic dynamic indicators, the trophic level allows the assessment of changes in the structure of the food web (Lira et al., 2021; Pauly et al., 1998) and can be used as a sustainability index (Collie et al., 2016; Cury et al., 2005; Philippsen et al., 2019). The fishing-in-balance indicator allows one to assess whether fishing activity is ecologically balanced (Kleisner & Pauly, 2011; Pauly et al., 2000). Likewise, the analysis of trophic interactions in the food web quantifies the loss in production and provides the basis for the definition of the L index, which considers ecosystem properties (such as primary and secondary production) and the characteristics of the fishing activity (Libralato et al., 2008; Pauly et al., 2000). Doria et al. (2018c) proposed the combination of using ecosystem methods to analyze artisanal fishing in the Amazon, focusing on exploited fish communities, using fishing in the Madeira Basin as a case study, in a situation/moment not modified by large projects (e.g., before the construction of the Madeira River hydroelectric plants). In the present study, we intended to refine this analysis considering the same area in an impact situation.

The present study evaluated whether ecosystem indicators responded to changes in fisheries landings in face of the environmental changes caused by human actions such as damming and deforestation. We also identified which indicators can best detect changes in fish production and make recommendations on how to use the indicators to better manage small-scale inland fisheries.

MATERIALS AND METHODS

Study area

Madeira River is the main river of the basin and is the largest tributary of the Amazon River (Sant'Anna et al., 2020a) and has the largest inventoried fish diversity in the world, with 1,057 fish species (Ohara et al., 2015). The most important fish market of the Madeira River is located in Porto Velho (approximately 460 thousand inhabitants; IBGE, 2022) (RO), Brazil, and was chosen as a case study, due to its representativeness in the Amazonian scenario, as a medium-sized market, with great importance for riverside communities in the region (Doria et al., 2012; Sant'Anna et al., 2020b); and for being affected by two large hydroelectric dams, Santo Antônio and Jirau (Doria et al., 2015; Hauser et al., 2019; Hauser et al., 2020; Lima et al., 2020a; Sant'anna et al., 2015).

The fishing activity developed in the region is classified as small-scale artisanal fishing, characterized by short, isolated trips, simple fishing gear of many types, low income from fishing, and its focus is on more than 10 predominant species (Doria et al., 2012; Doria et al., 2015). The activity is carried out along the Madeira River in communities and districts in the municipalities of Humaitá, in the state of Amazonas, and Porto Velho, which is also the capital of the state of Rondônia and where the main fish landing port of the region is located (Fig. 1).

This region has been influenced by the construction of two large dams, the Santo Antônio dam, 7 km upstream from Porto Velho and built between 2008 to 2011, and the Jirau dam, built at Caldeirão do Inferno Fall and completed in 2012, located 100 km upstream from Porto Velho (Doria et al., 2015; Santos et al., 2018).

In the period studied by Lima et al. (2020a), from 1990 to 2014, before the dams were closed, the average annual fishing production was 576.8 tons (± 294.8 tons) and 1,202 fishers worked in the region between Guajará-Mirim and Humaitá (Doria et al., 2015). In this region, the fishing fleet consists mainly of small non-motorized and motorized wooden canoes, with an average length of 6 m and a storage capacity of 250 to 600 kg. However, the number of large boats (motorized and with an average length of 9 m ± 2.3 m), which have a greater storage capacity (average of 2,500 kg), diminished over time and were not observed after the year 2005 (Doria et al., 2018c).

Data collection

Landings records were obtained concurrently from different sources to fill gaps in fisheries records, and due to the limited availability of data in the study region. The fisheries database consisted of the following information:

- Years 2000 to 2009, daily landing records kept by the Z-1 fishing association based on fisher's declarations;
- Years 2009 to 2019, fisheries landing records collected using a questionnaire by the state inspection authority;
- Bibliographic records and environmental data on human actions in Porto Velho and related to fisheries in the study area (Table 1).

The main landing port in the region is in Porto Velho, it is called Cai n'água port, and responsibility of the Z-1 fishing association. The daily records of fisheries landings (species categories and total weight landed in kg) made by the fishing association were obtained by a research group called South-Western Amazon Biodiversity

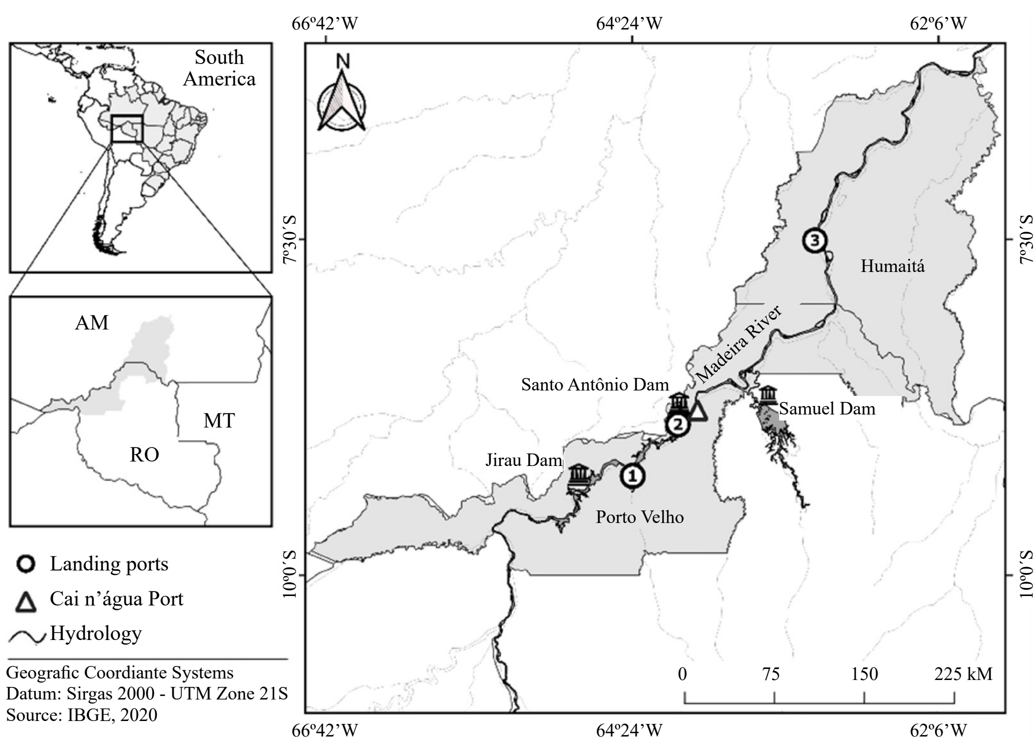

Figure 1. Geographic location of the fishing areas of municipalities in the Madeira River region, Brazil. The numbers indicate the fish landing ports: (1) Jaci-Paraná, (2) Vila Teotônio, in Rondônia, and (3) Humaitá, in Amazonas state, before arriving at Cai n'água port in the city of Porto Velho, Rondônia, Brazil.

Study Group, of the Laboratory of Ichthyology and Fisheries at the Universidade Federal de Rondônia. Additionally, data records from the questionnaires applied by the inspection authority were organized into spreadsheets to calculate the rate of capture-per-unit-of-effort (CPUE), and statistical analysis was performed.

Environmental data and information regarding human actions were extracted from official documents and national and state records throughout the study period (Table 1) and correlated with the variation of landings and sustainability indices.

Data analysis

Fisheries landing data were aggregated by month and year for the statistical analysis and estimation of average landings. The CPUE was obtained by fisheries landing records collected using a questionnaire by the state inspection authority (Eq. 1), from CPUE of the Amazon:

$$CPUE = C/Nf \cdot Nd \tag{1}$$

Where: C= catch in kg; Nf= number of fishers; Nd= number of days.

So that the monthly average from 2009 to 2019 could be estimated. After applying the Shapiro-Wilk's and Levene's tests, it was observed that the monthly production data (kg) and CPUE did not present normal distribution and homoscedasticity (Myers, 1990). Therefore, the non-parametric Kruskal-Wallis' test was performed to detect differences in the median, interquartile range, and in the distribution of records. Additionally, Dunn's post-hoc test was applied with Bonferroni's *p*-value adjustment method to visualize which records presented differences between them. The analyses were performed using the R statistical program (R Core Team, 2021).

The relative frequency was estimated according to species category, when necessary, since fishers assign common names to several species. Thus, the production levels of the categories and species estimated were those that represented at least 2% of the total weight captured in the year and were considered the main categories. To observe changes in fishing production in the studied period, in the analysis of total and specific fishing production, the records were grouped into the following five-year intervals: 2000–2004, 2005–2009, 2010–2014, and 2015-2019 (according to the Doria et al., 2018b). Carruthers et al. (2014) demonstrated the importance of testing catch-based methods with groups of three to ten years in order to observe trends over the years.

The feeding habits and trophic level of the main species were calculated based on studies in the region (Cella-Ribeiro et al., 2016; Doria et al., 2018b) and on information from the FishBase platform (Froese & Pauly, 2019). The first ecosystem indicator estimated was the annual trophic level (TL). This indicator is a weighted average of the TL of the species in the catches (as used in Doria et al., 2018c). The average TL varies over the years of study in such a way that, in the ecosystem assessment of fisheries, the functionality of the species is more relevant than the species itself. To analyze the dynamics of the fish in the trophic levels, the species were grouped into five categories: 2 to 2.49 (detritivores); 2.5 to 2.99 (herbivorous); 3 to 3.39 (omnivorous); 3.4 to 3.99 (carnivorous); and 4 to 4.5 (piscivorous).

Table 1. Environmental data and human actions with the potential to affect fisheries landings from 2000 to 2019, in the municipality of Porto Velho, Rondônia, Brazil.

Environmental changes and anthropization	Period	Source
Construction of hydroelectric dams	2008–2012	Lima et al. (2020a); Santos et al. (2018)
Average precipitation (mm)	2000–2019	Hydroweb (2021)
Average annual hydrological level (cm)	2000–2019	Hydroweb (2021)
Alerts on the maximum and minimum hydrological level of the Madeira River	2014, 2018, 2019	Rondônia (2017; 2018; 2019)
Average deforestation in the municipality of Porto Velho (km^2)	2008–2019	TerraBrasilis (2021)
Occurrence of El Niño and La Niña	2000, 2003, 2007, 2008, 2010, 2011, 2016 and 2018	Brazil (2021)
Average forest cover in the municipality of Porto Velho (km^2)	2000–2019	MapBiomas (2021)

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The fishing-in-balance (FiB) index (Pauly et al., 2000) was calculated to evaluate the efficiency of the fishing system using data of the catches during the study period, using Eq. 2:

$$FiBk = log(Yk(1/TE)^{TLk}) - log(Y0(1/TE)^{TL0}) \qquad (2)$$

Where: k = year (0 = base year); Y = catch (tons); TL = calculated average trophic level of catches; TE= transfer efficiency between trophic levels, set at 0.078 before the end of dam construction (before 2012) and 0.048, as of 2012, for this stretch of the Madeira River (Lima et al., 2020b).

The theoretical loss in secondary production due to exploitation was quantified by calculating the L index (Libralato et al., 2008). In this index, the ecosystem properties expressed as a function of the primary production required (PPR) to support the capture of each species, the TL of the species, the primary production of the basic chain (P1), and the energy transfer efficiency rate (TE) of trophic flows in the ecosystem are considered (Eq. 3).

$$L = -\frac{1}{P1}.Ln.TE.\sum_{i}^{m}.(PPRi.TE^{TLi-1}) \cong -PPR.TE^{TLc-1}/P1.Ln.TE \qquad (3)$$

The values of P1, PPRi, and TE used were from a food-chain model of the middle stretch of the Madeira River (Lima et al., 2020b). Due to the TE presenting different values, the L index was presented in two graphs, ten years before the construction of the first dam (from 2000 to 2009) and ten years later (2010 to 2019). In sustainable fisheries, reference values for the L index range from L = 0.021 (years before) to L = 0.007 (years after) (Libralato et al., 2008). The probability of sustainable fishing (Psust) was estimated using Eq. 4:

$$\text{Psust} = -L \cdot 1/(0.18+1) \qquad (4)$$

The resulting value was compared with the average value of the L index (Libralato et al., 2008).

Correlations between explanatory (environmental data and human actions) were assessed using correlation plots and Pearson's correlation tests, on the "corrplot" package in r. Variables with correlation greater than 0.4 were removed and not used in later models to avoid collinearity. We also evaluated variance inflation factors (VIF), which were generally < 3, indicating that there was no multicollinearity among the remaining variables (Myers, 1990). The environmental variables were log-transformed to ensure the fit to a normal distribution and to be used in subsequent modelling, since it is a requirement to use mixed linear models. Model diagnosis was performed using the package "predictmeans" in R. Thus, to understand the impacts on annual fisheries production (t) (2000 to 2019) and on the FiB indicator, the selected records were the following: hydrological level (cm), rainfall (mm), forest cover (%), deforestation in the geographical area of Porto Velho (km^2), fishing (days·n°. of fishers), years of El Niño and La Niña, and before and after installation of dams on the Madeira River in order to understand their effects on fisheries production and FiB. These analyses were performed in the R statistical program, package lme4 (R Core Team, 2021).

A mixed linear model is described by the distribution of two vector-valued random variables: Y, the response, and B, the random effects vector. The model that best suited the data with fisheries production was a mixed linear model. The model that most suited the data with the indicator FiB was a generalized linear model.

RESULTS

The highest fisheries production occurred in 2008 (1,425.7 tons). However, the lowest fisheries production was 76.8 t in 2014 (Fig. 2). The average annual fisheries production in the four five-year intervals was 465.8 t (± 318.9), with significant variation between the intervals (Kruskal-Wallis, X^2 = 26.44; degrees of freedom – df = 3; $p > 0.001$). Dunn's test showed that fisheries production in the period 2005–2009 (median = 45.90; interquartile = 99.39) was significantly different from that of 2000–2004 (median = 13.59; interquartile = 58.91) and of 2015–2019 (median = 7.84; interquartile = 20.84).

The CPUE reached its highest value in 2009, with 28.35 kg/fisher·day and the lowest value in 2013, with 11.8 kg/fisher· day, with an average value of 19.27 (±5.2). The annual CPUE values showed a statistically significant difference (Fig. 2) between the years (Kruskal-Wallis, X^2 = 59.793; df = 10; $p > 0.001$), and Dunn's test showed that the values estimated before the completion of the Santo Antônio and Jirau dams (2009, 2010 and 2011, and the median = 25.4; 23.9; 22.6, respectively) differ from the estimated values for the years 2016, 2017, 2018, and 2019 (median = 14.4; 12.9; 12.5; and 11.2, respectively), after completion of the dams.

During the study period, fish species of 53 taxa were identified in the landings. Between them, 20 categories (which use the common name for the species) represented 90% of the production, but 16 had at least 2% of representativeness in the production for the studied period (Table 2). In the first block of years, from 2000–2004, the categories with the highest percentage

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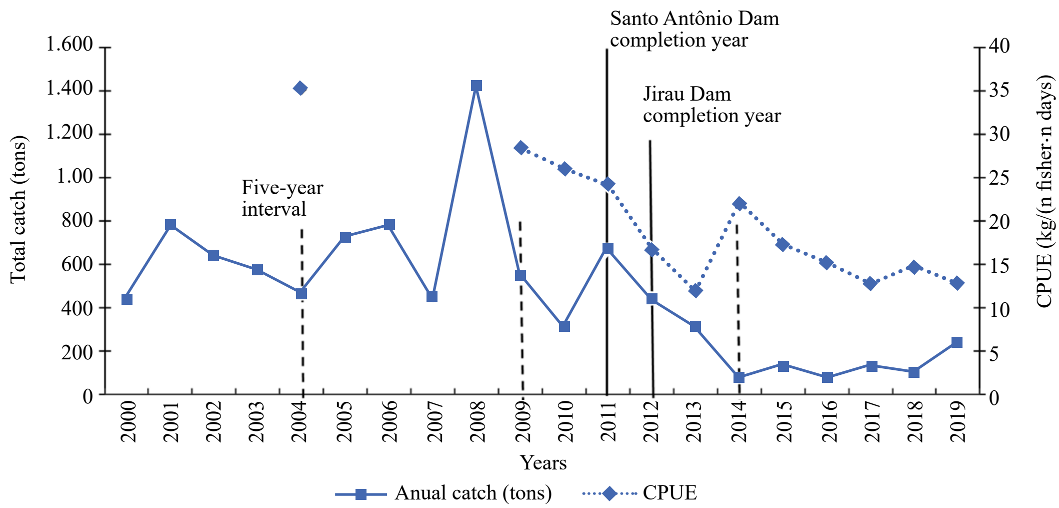

Figure 2. Annual fisheries production landed at the Cai n'água port in Porto Velho (Rondônia, Brazil; data source: Z-1 Fishing Association) capture-per-unit-of-effort (CPUE): 2004 (Doria et al., 2012); from 2009 to 2019; and the year of completion of the dam (Doria et al., 2015).

Table 2. Relative fishing production (main fish landed that presented > 2% of total landings, for at least one year) in the Cai n'água port in Porto Velho (RO), Brazil, at five-year intervals, main trophic category, migratory habit, and trophic level*.

Order Family	Taxon (fish category)	Fishery production (%)				Trophic category	Migratory habit	Trophic level
		2000–2004	2005–2009	2010–2014	2015–2019			
Characiformes Anostomidae	*Schizodon fasciatus* (Spix & Agassiz, 1829)	2.07	4.82	8.01	7.61	Herbivore	SD	2.0
Curimatidae	*Potamorhina altamazonica* (Cope, 1878); *P. latior* (Spix & Agassiz, 1829)	3.26	2.16	5.18	5.69	Detritivore	SD	2.0
Prochilodontidae	*Prochilodus nigricans* (Spix & Agassiz, 1829)	24.26	12.15	10.47	9.42	Detritivore	SD	2.3
	Semaprochilodus taeniurus (Valenciennes, 1821); *Semaprochilodus insignis* (Jardine, 1841)	10.84	8.66	7.74	4.81	Detritivore	SD	2.0

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Order Family	Taxon (fish category)	Fishery production (%)				Trophic category	Migratory habit	Trophic level
		2000–2004	2005–2009	2010–2014	2015–2019			
Characidae	*Brycon amazonicus* (Spix & Agassiz, 1829)	5.78	5.82	4.44	7.40	Omnivore	AD	2.0
	Triportheus spp.	7.20	3.72	4.07	3.34	Omnivore	AD	2.7
	Colossoma macropomum (Cuvier, 1816)	2.19	0.55	1.91	0.41	Omnivore	AD	2.0
Serrasalmidae	Myleus spp.; Mylossoma spp.	14.18	12.21	11.54	15.57	Herbivore	AD	2.0
Cichliformes Cichlidae	*Heros spurius* (Heckel, 1840); *Geophagus megasema* (Heckel, 1840); *Satanoperca jurupari* (Heckel, 1840);	0.47	1.40	4.53	3.14	Omnivore	R	2.3
	Cichla sp.	1.66	3.91	5.16	4.04	Carnivore	R	3.2
Siluriformes Pimelodidae	*Pinirampus pirinampu* (Spix & Agassiz, 1829)	0.10	7.05	4.93	2.78	Carnivore	AD	3.0
	Brachyplatystoma rousseauxii (Castelnau, 1855)	8.24	8.98	2.68	7.54	Carnivore	LD	3.2
	Brachyplatystoma filamentosum (Lichtenstein, 1819)	1.90	1.81	1.87	2.90	Carnivore	LD	3.5
	Zungaro zungaro (Humboldt, 1821)	0.33	1.15	1.90	3.09	Carnivore	AD	3.4
	Phractocephalus hemioliopterus (Bloch & Schneider, 1801)	1.46	1.34	1.79	1.83	Carnivore	AD	3.4
	Pseudoplatystoma spp.	3.39	4.12	3.79	6.83	Carnivore	AD	3.2

R= residents; LD= long distance; AD= average distance; SD= short distance; *the numbers in bold indicate the highest percentage fisheries production in the five-year intervals. Source: Doria et al. (2018c).

of production were *curimatã* (*Prochilodus nigricans*), *jaraqui* (*Semaprochilodus* spp.), and *pacu* (*Mylossoma* spp.). In the other intervals, the category *pacu* had the highest production, followed by *curimatã* and *dourado* (*Brachyplatystoma rousseauxii*) in 2005–2009. The *aracu* category (*Schizodon fasciatus*) had the third highest percentage of production in the years 2010–2014 and 2015–2019 (Table 2). The *jaú* (*Zungaro zungaro*) had a tenfold increase (0.33 to 3.09%) in its fisheries landing records, and the *surubim*/*caparari* category (*Pseudoplatystoma* spp.) doubled production from the first to the last evaluated period.

The mean TL in fisheries landings increased from 2005 onwards, with variations over the period (2.55 ± 0.2; Fig. 3a)

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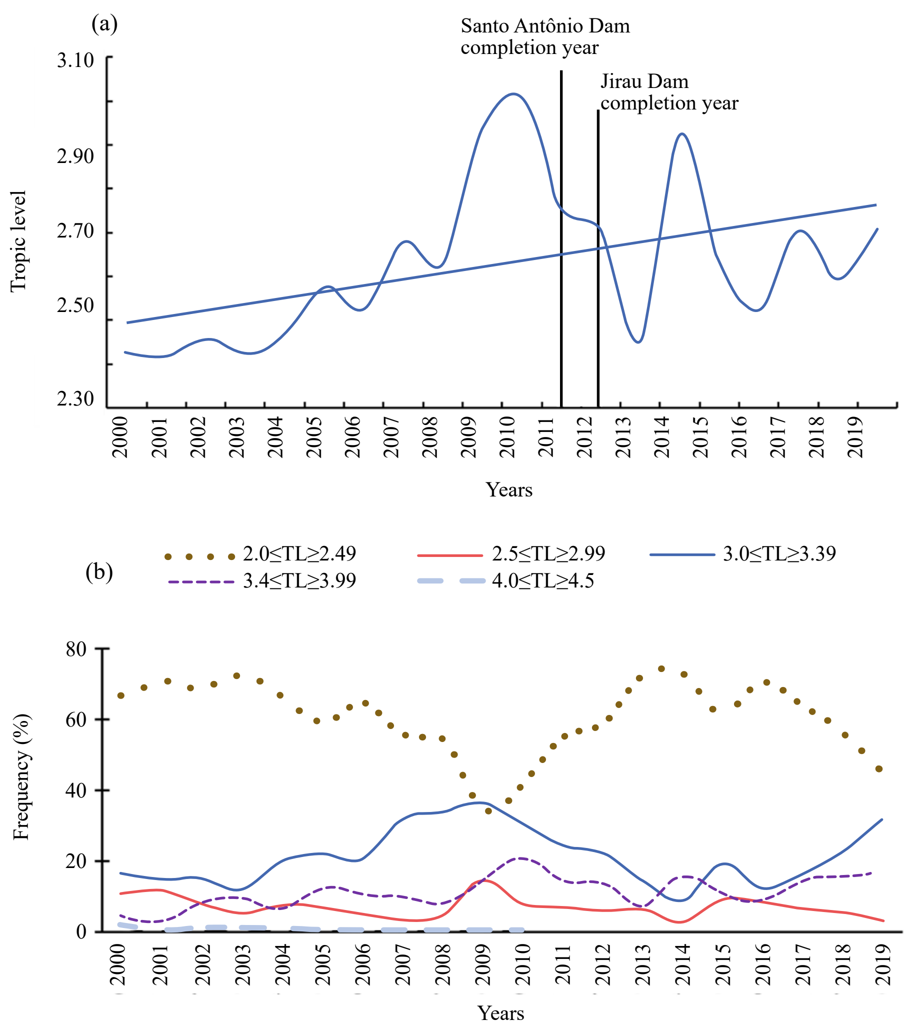
and a tendency to stabilize from 2016 onwards. The highest value was reached in 2010, and it was due to the increase in landings of carnivorous and piscivorous fish. In 2013, a large production of detritivores was responsible for raising the TL to 2.45 (Fig. 3a), and, after the great flood of 2014, the percentage of piscivorous fish was higher than in the years that preceded it. As of 2017, there has been an even greater increase in carnivores and piscivores. However, 2009 was the only year in the period in which the landing of carnivores and piscivores were higher than the landings of other trophic categories (Fig. 3b).

The FiB showed an upward trend between 2000 and 2009, when it began to decline, reaching a negative value (-0.044) in 2013, which remained until 2018. In 2019, FiB returned to a positive value (Fig. 4), but with a value nine times lower than the mean of the years that presented the highest values, from 2008 to 2012 (0.57 ± 0.1).

The drop in landed production was similar to that shown by the L index, which, after damming and the decrease in TE, showed lower values than in other years (Fig. 5). In the 10 years prior to the construction of the dams (until 2009), the L index had an average value of 0.03 ± 0.013 (Fig. 5a), and the Psust was 80%. From 2010 to 2019, five years (from 2014 to 2018) had values lower than 0.007, and an average value of 0.01 ± 0.010 (Fig. 5b), and the Psust was 90%.



Source: Z1 Fishing Association.

Figure 3. Annual average trophic level (TL) of captures. (a) Frequency (%) of kg of all trophic levels of the categories landed and (b) for each category's trophic level (TL) ranging from detritivores to piscivores: 2 to 2.49; 2.5 to 2.99; 3 to 3.39; 3.4 to 3.99; and 4 to 4.5, landed at the Cai n'água port in the city of Porto Velho (RO), Brazil.

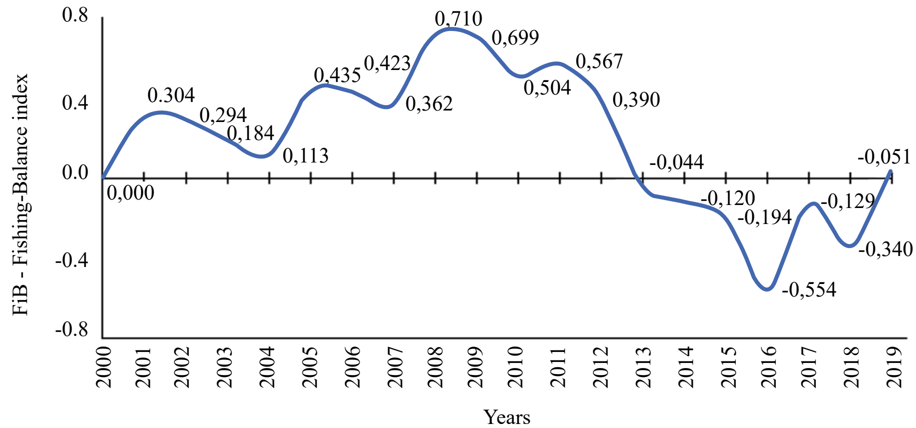
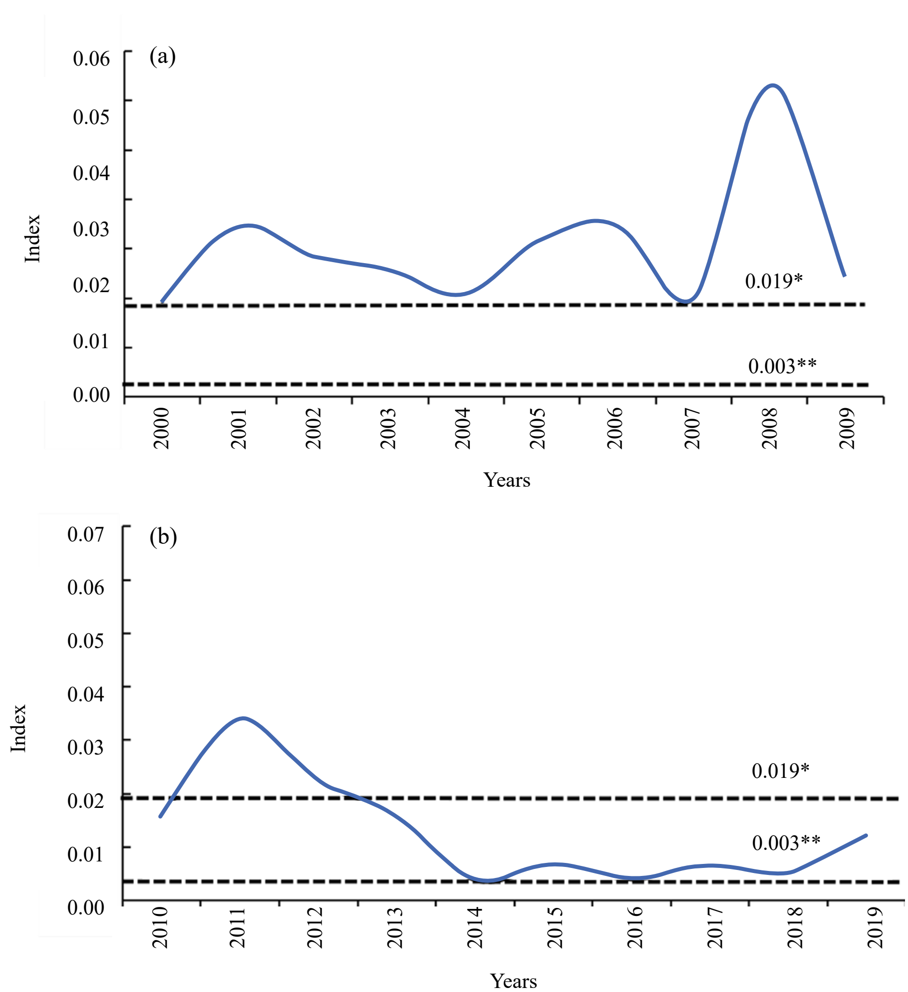


Figure 4. Fishing-in-balance (FiB) estimated from the categories and species landed at the port of Cai n'agua in Porto Velho (RO), Brazil.



- - -: minimum limits showing the differences in L index *before and **after dam closures.

Figure 5. L index (primary production required) (a) ten years before the construction of dams (from 2000 to 2009) and (b) ten years after (2010 to 2019).

Using a correlation matrix with the records, it was observed that the significant parameters, in relation to fisheries production (*catch*) and the FiB indicator (fib), were the hydrological level (wl), precipitation (prec), forest cover in the geographical area of Porto Velho (*forest*), fishing effort (effort), and the series of years in which El Niño or La Niña occurred.

The model that best suited the data on fishery production was a mixed linear model, in which the random effect of dams influenced 99.9% of the catch, and the hydrological level explains 78% of the catch variation (Table 3). The model that best suited the data with the FiB indicator was a generalized linear model, which explains 61% of the variation of the index over the years, and the absence of dams was significant. In the series of data under study from 2000, when separating 10 years before the construction of the first dam (2000–2009) and 10 years after (2010 to 2019), several transformations in the environment were observed (Fig. 4).

The results of the model indicate that fisheries production was significantly influenced by the hydrological level, and FiB was significantly affected by the installation of the dams (Table 4).

DISCUSSION

In the 20 years of fisheries data evaluated in this study, changes were observed in the annual averages of fisheries production in the main landing port of the region of Porto Velho. Landings over the past 10 years (2010–2019) fell by 74%, which was accompanied by a 66% drop in mean CPUE. This index suffered reduction between 2015 and 2016 because many fishers abandoned the activity (Dutka-Gianelli et al., 2022). The number of fishers registered by Sant'Anna et al. (2020b) in 2012 for this fishing port was 376, which is almost a third of those registered in 2009 and 2010.

The fishers who did not abandon the activity during the first years after the implementation of the dams increased their fishing efforts to maintain catch levels, which justifies the increase in CPUE. However, as they increased the effort to maintain the catch, consequently, they increased expenses with their fishing expeditions. Due to the drop in fisheries production, more fishers decided to look for another activity. According to the market intermediaries, fishers are leaving the activity to work in mining, as it is a more profitable activity, which may explain the almost 40% decrease of fishers in the activity, which is why the traders are choosing to buy fish from the neighboring municipalities of Labrea and Humaitá.

The change in catch composition also affected the profitability of the activity. This happened because there was a marked shift from higher value fish to lower value fish up until 2014 (as observed by Lima et al., 2020a), and this was accentuated in the following years. There is an increase in the percentage of production (200%) of medium-distance migratory catfish (average price US$ 3.34) and a 50–70% reduction in the landing of long-distance catfish (average price US$ 4.45) (Lima et al., 2020a).

Concomitantly, the composition of trophic levels underwent several changes, as observed via the trophic level indicator. The highest TL value was in 2010 (3.01), and it is attributed to the increase in landings of carnivorous species of the Pimelodidae

Table 3. The results of the correlation matrix of records regarding fisheries production and fishing-in-balance. Fisheries production (***catch***); fishing-in-balance indicator (fib); hydrological level (wl); precipitation (prec); forest cover in the geographical area of Porto Velho (***forest***), fishing effort (effort); and the series of years in which El Niño or La Niña occurred[#].

	year	catch	wl	prec	deforest	el_nino	la_nina	cpue	forest	fib	effort
year		-0.608**	0.418	0.520*	0.833***	-0.022	-0.043	-0.635**	-0.989***	-0.497*	-0.756**
catch	-0.608**		-0.260	-0.110	-0.357	0.303	-0.174	0.292	0.543*	0.775***	0.516
wl	0.418	-0.260		0.473*	0.026	-0.068	-0.326	-0.158	-0.396	-0.209	-0.197
prec	0.520*	-0.110	0.473*		0.045	0.110	-0.475*	-0.294	-0.493*	0.013	-0.553*
deforest	0.833***	-0.357	0.026	0.045		0.220	-0.238	-0.742**	-0.843***	-0.643*	-0.215
el_nino	-0.022	0.303	-0.068	0.110	0.220		-0.250	0.050	0.019	0.077	-0.223
la_nina	-0.043	-0.174	-0.326	-0.475*	-0.238	-0.250		0.065	0.014	-0.087	-0.209
cpue	-0.635**	0.292	-0.158	-0.294	-0.742**	0.050	0.065		0.663**	0.404	0.765**
forest	-0.989***	0.543*	-0.396	-0.493*	-0.843***	0.019	0.014	0.663**		0.422	0.740**
fib	-0.497*	0.775***	-0.209	0.013	-0.643*	0.077	-0.087	0.404	0.422		0.186
effort	-0.756**	0.516	-0.197	-0.553*	-0.215	-0.223	-0.209	0.765**	0.740**	0.186	

[#]Computed correlation used Pearson-Method with pairwise-deletion; *significant level of 0.05; **significant level of 0.01; ***significant level of 0.001.

Table 4. Results of the mixed linear model (MLM) and generalized linear model (GLM) for the catch and the fishing-in-balance indicator models, respectively. The parameters analyzed were effort, hydrological level, precipitation, forest cover, and presence of La Niña and El Niño phenomena.

Catch model			
Predictors	Estimates	CI	P-value
(Intercept)	-57.92	-186.29–70.45	0.279
Effort	0.65	-0.50–1.79	0.192
Hydrological level	-4.02	-7.99–-0.06	0.048*
Precipitation	4.06	-0.36–8.48	0.063
Forest cover	4.32	-5.29–13.92	0.280
La Niña / El Niño	-0.25	-1.72–1.22	0.665
Random effects			
σ^2	0.17		
$\tau_{00\ dam}$	0.17		
$\tau_{11\ dam, wl}$	0.00		
$\rho_{01\ dam}$	-1.00		
no $_{dam}$	2		
Comments	14		
Marginal R^2	0.779		
FiB model			
Predictors	Estimates	CI	P-value
(intercept)	106.45	-124.73–337.63	0.367
Effort	-0.49	-2.95–1.97	0.697
Hydrological level	-19.03	-39.24–1.18	0.065
Precipitation	7.12	-3.98–18.22	0.209
Forest cover	-0.59	-14.48–13.29	0.933
El Niño	-2.14	-4.73–0.44	0.105
La Niña	0.77	-2.46–4.00	0.639
Dam (Before)	2.02	0.02–4.02	0.048*
N observations	9		

*Significance of < 0.05; CI= confidence interval.

family, such as *Brachyplatystoma filamentosum* and *Pinirampus pirinampu*, which are species that represent the migratory group and were important for fisheries before the construction of the dams. In 2010, most of the landings were of carnivorous and piscivorous fish, and the greatest decline was in detritivorous fish, such as *branquinha* (*Potamorhina altamazonica* and *P. latior*) and *pacu*, in addition to *curimatã* (*Prochilodus nigricans*). Therefore, there was an inverse relationship between these landed fish that has never previously been observed since data began to be recorded, in 1990 (Doria et al., 2018c). Similarly, it does not correspond to the trend of fisheries landings in the central Amazon (Doria et al., 2018a).

From 2010, during the construction of the dams, the average TL decreased due to the return of detritivorous fish, and, in 2013, it decreased due to the increased production of the jaraqui category (detritivore); while, in 2014, the increase was in the landing of carnivores such as *Cichla* and *Hoplias*, possibly due to the more favorable environmental condition for detritivores and sedentary carnivores in the year 2013. In a post-damming environment (more enriched environments due to plant/animal mortality) and the following year, carnivores emerge due to the high occurrence of prey (detritivorous fish). In addition, in this year, there was a flood of proportions greater than those that had occurred previously, i.e., 2,2 m higher than the greatest

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previously recorded (Justina et al., 2014). In this period, the waters reached more lakes, which have sedentary species that some fishers could access. In addition, the records also showed an increase in Pimelodidae, with growth in the landing of the category *Pseudoplatystoma* and the *barba-chata* catfish (*Pinirampus pirinampu*). It is necessary to monitor this index in landings because, in the last years of the presented series, it seemed to stabilize around 2.5 to 2.7.

The FiB indicator is highly dependent on catches and their TL in the reference year (2000 in the present study). Until 2012, the FiB observed in the fisheries in the port of Porto Velho was FiB > 0, demonstrating that the fishing pressure on the Madeira River is relatively low, similar to that observed by Doria et al. (2018c), and sustainable (Pauly et al., 2000). On the other hand, the FiB < 0 observed for the values in the years 2013 to 2018 reflects a system that is working less efficiently than it should due to intense fishing pressure (Pauly et al., 2000). However, it was observed in the study area that this pressure did not occur, as the CPUE decreased, as well as the number of fishers. Both the statistical analysis of fisheries production and the CPUE show that the values before and after the construction of the dams differ. In other words, regardless of the effort (greater or lesser considering the number of fishers), production was affected by other issues, probably not related to fishing pressure.

The FiB index is not only sensitive to the historical development of fisheries, but also to differences in fisheries status over time, more precisely than any other single index derived from catch statistics (Cury et al., 2005). Thus, the index only decreases when catches do not expand as expected, which allows one to assess whether a fishery is ecologically balanced or not (Pauly et al., 2000). Therefore, due to its integrative nature, it is believed that the FiB provides a better indicator of changes in ecosystems than the composition of the catch (Garcia & Staples, 2000).

The values of the L index until 2009 ranged from 0.018 to 0.063, and, from 2010 to 2019, the values ranged from 0.004 to 0.023. In other words, the index showed the unsustainability of fisheries from 2014 to 2018 (see Libralato et al., 2008). However, the Psust from 2000 to 2009 was 80%, and, from 2010 to 2019, it was 90%. This indicator has greater value with the increase of species between the higher TL than in marine fisheries, since the high sustainability of fisheries in open oceans is consistent with the relatively low yields and high TL of catches made in this environment (Caddy et al., 1998). From 2000 to 2009, the percentage of production of species categories with a TL of ≥ 3 was less than half of the total; from 2010 to 2019, this percentage was equal to 60%. Due to the flood in 2014 and the construction of the dams, this resulted in an increase in species with high TL (TL ≥ 3) (Agostinho et al., 2016). Thus, Psust was not suitable for analyzing an ecosystem affected by the construction of dams.

However, the sum of the impacts of the construction of the two large dams with other events (natural or anthropogenic) in the same basin can cause environmental changes and affect fishery landings (Castelo et al., 2013; Santos et al., 2018). In the Madeira River, it was observed that the capture of fish has a direct relationship with the variation of the hydrological level (Lima et al., 2017). In other words, any change in dynamics, whether caused by dams, waterways or extreme floods and droughts, will lead to changes in fisheries.

The impacts of anthropogenic (dams) and natural (e.g., El Niño and La Niña) origin that affect the flood pulse of rivers are mainly responsible for changes in fish production in freshwater rivers, since the flood regime is the controlling factor of fish stocks in Neotropical rivers (Lowe-McConnell, 1987). In the Amazon Basin, fish migrations are closely linked to seasonal fluctuations in the hydrological regime (Goulding, 1980; Junk et al., 1989). Reproductive migrations of diverse species of Characiformes are synchronized with the rainy season and the rising-water period, presumably to optimize environmental conditions for egg hatching, larval and juvenile growth, and survival (Duponchelle et al., 2021). The longitudinal reproductive migrations of most large Siluriformes are carried out upstream, towards the Andean piedmont during the high-water period (Barthem & Goulding, 1997; Barthem et al., 2017; Duponchelle et al., 2016; Hauser et al., 2019; Hauser et al., 2020; De Lima and Araujo-Lima, 2004). Large catfish also have a relevant ecological function as top predators of the food chain; therefore, changes in their abundance can have profound consequences for the ecosystem (Angelini et al., 2006), which can produce a cascade effect on other species in the food chain (Primack & Rodrigues, 2001; Primack & Ros, 2002).

When the fluctuations no longer govern the available environments and the previously available ones have changed, they demand a transformation from the existing ichthyofauna that results in a decrease in the abundance and slow adaptation of the more resilient species and, consequently, the change in fisheries to different trophic guilds (Pinaya et al., 2016; Sousa et al., 2021). The construction of two large dams in the main channel of the middle Madeira River, coupled with changes in the hydrological level, precipitation, and the ecological behavior of fish species, may contribute to the reduction of fisheries

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production values (Barros et al., 2020; Sant'Anna et al., 2020a; Sousa et al., 2020; Sousa et al., 2021).

In addition, another major threat to the maintenance of fish stocks that use the flooded forest is the loss of habitats (Arantes et al., 2019; Crampton et al., 2004) that comes from unusual flooding (e.g., El Niño) or deforestation. In the Tapajós River, deforestation was the main factor in the decrease in fish biomass, as the reduction of allochthonous food resources affects fish communities (Capitani et al., 2021; Carvalho et al., 2018). The same may be happening in the study region, where deforestation has already exceeded the value of 400 km² per year in 2019, and, since 2012, this scenario has increased, and forest cover is gradually decreasing. Forest loss in floodplains works together with climate change and promotes a considerable decrease in the probability of fish species persisting in the long term (Barros et al., 2020; Herrera-R et al., 2020).

Global climate change affects the hydrological cycle in the Amazon Basin, altering rainfall and evaporation levels, with lower rainfall, especially during the dry season (Castello & Macedo, 2016; Davidson et al., 2012). Extreme droughts can be catastrophic for fish for a short period due to the large reduction in aquatic environments (Duponchelle et al., 2021). Although several fish species can move to the river channel during the dry season, some resident species of lakes remain in the biotope and are unable to survive if the drought is severe, as happened in 2016 (Rondônia, 2017). Studies have observed evidence that extreme droughts occasionally alter the biomass of fish, affect reproduction of some species and at certain times cause local extinction (Humphries & Baldwin, 2003; Röpke et al., 2022). In large floods, the effect is different, for example, the consequences of the great flood of 2014 apparently led to an increase in production, given the numbers of the landings in 2015; however, the impact on the well-being of traditional populations was destructive for 3,758 families (CGU, 2014). Nonetheless, more detailed investigations are necessary to assess the real impacts of this anomalous event.

The cumulative effects of these environmental impacts in a basin can harm important economic activities, such as fishing, and jeopardize the main ecological processes performed by fish in tropical rivers. This has negative effects on ecosystem functioning and on income, food security, and livelihoods of millions of people in tropical regions (Arantes et al., 2019; Brismar, 2004; Tallis et al., 2015; Villarroya et al., 2014). Thus, it is important to develop studies that allow the integrated analysis of factors that have the potential to affect fisheries and that can contribute to the conservation and sustainability of the fisheries resources used by traditional communities. Although it has reduced in recent decades, artisanal fishing remains an important activity for hundreds of communities located on the banks of the main Amazonian rivers. In the Madeira Basin by itself (Bolivia, Peru, Brazil), it is estimated that there are more than 5,000 riverine families involved in fisheries (Doria et al., 2018b). In addition to specific jobs, such as that of the fisher, fishing generates a series of complementary activities (e.g., restaurants, markets/processing plants and fishmongers).

CONCLUSION

The indicators of the TL, FiB, and L indexes are useful for diagnosing the situation of the fisheries after major environmental changes, whether from anthropic actions or not. Furthermore, models can help to understand the relationship between environmental changes and their consequences for fisheries. Better understanding of the fisheries in large rivers can be obtained using such tools and has the potential to provide more precise information for decision making regarding the sustainability of fisheries.

CONFLICT OF INTEREST

Nothing to declare.

DATA AVAILABILITY STATEMENT

All data relevant to the study are included in the article and as supporting information.

AUTHORS' CONTRIBUTIONS

Conceptualization: Sant'Anna, I.R.A., Reis, V.; Writing – original draft: Sant'Anna, I.R.A.; **Data curation:** Sant'Anna, I.R.A., Pinto, D.M.; **Writing – review & edition:** Sant'Anna, I.R.A., Souza, F.K.S., Sousa, R.G.C., Doria, C.R.C.; **Resources:** Reis, V.; **Investigation:** Reis, V.; **Sofware:** Pinto, D.M.; **Formal Analysis:** Souza, F.K.S.; **Validation:** Sousa, R.G.C.; **Supervision:** Doria, C.R.C.; **Final approval:** Doria, C.R.C.

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