Fisheries Research 221 (2020) 105399

Contents lists available at ScienceDirect

Fisheries Research

journal homepage: www.elsevier.com/locate/fishres

Declining fisheries and increasing prices: The economic cost of tropical rivers impoundment

Maria Alice Leite Lima[a,b,*], Adriana Rosa Carvalho[c], Marcus Alexandre Nunes[d], Ronaldo Angelini[e], Carolina Rodrigues da Costa Doria[a]

[a] Laboratory of Ichthyology and Fisheries, Federal University of Rondônia, Porto Velho, 76801-059, Brazil
[b] Postgraduate Program in Regional Development and Environment (PGDRA), Federal University of Rondônia, Porto Velho, 76801-059, Brazil
[c] Department of Ecology, Federal University of Rio Grande do Norte, Natal, 59078-970, Brazil
[d] Department of Statistics, Federal University of Rio Grande do Norte, Natal, 59078-970, Brazil
[e] Department of Civil Engineering, Federal University of Rio Grande do Norte, Natal, 59078-970, Brazil

A R T I C L E I N F O

Handled by A.E. Punt

Keywords:
Amazon
Fisheries
Structural breaks
Impoundment effects
Price dynamics

A B S T R A C T

This work tests the null hypothesis that the coefficients of the total landings, landed values, mean catches and price per kg of migratory and resident species are constant over time following the installation of two large run-of-the-river hydroelectric dams in a large tropical river. To identify shifts in catches and economic returns due to river impoundment, we inspected daily landing data (25-year time series) and wholesale prices (19-year time series) for the Madeira River, the largest tributary of the Amazon River. Our results show that the period of decreasing catches and increasing prices observed for fisheries in the Madeira River matched the timings of the construction of the two dams. According to the results, both dams quickly changed catches and fish supply to market, which were immediately echoed in the price per kg of exploited fish species. Following the dam construction, prices rose for both fish that became scarce and fish that became abundant. Though catches declined 58% in 25 years, the price increased 49% over the same period, representing a high economic cost for the local population. Further, there was a clear decline in the catches of some species (e.g., the dourada and the curimatã), but increased catches of others (e.g., the sardine and the tucunaré). Moreover, some fluctuation patterns across years showed natural oscillations, or changes, in local habitats and even fishing efforts.

1. Introduction

Fisheries production in large tropical rivers is a key ecosystem service to human welfare and an important component of biodiversity supplying global economic value, markedly to developing countries (Balmford et al., 2002; Costanza et al., 1997). The exploitation of such biodiversity despite benefiting many people, has changed natural ecosystems leading to losses in biodiversity and ecosystem services, threatening the well-being of social groups more susceptible to poverty. As a result, over the last century, freshwater biodiversity has plummeted globally (Sala et al., 2000), mainly due to river impoundment that negatively impacts the ecosystem (Dudgeon et al., 2006; Lima et al., 2017; Santos et al., 2018), its biological diversity and local fish stocks (Agostinho et al., 2008; Bunn and Arthington, 2002; Pelicice et al., 2017).

Notwithstanding these effects, at the end of 2000's about 45,000 large dams (15 m in height) and 800,000 small dams had been built worldwide (WCD, 2000). In tropical areas, large impoundments keep growing and being considered as development actions (Hoeinghaus et al., 2009), continuously adding pressure on rich biological diversity (Agostinho et al., 2005). The Amazon basin has been affected by this trend and currently has 416 hydroelectric power plants in operation or under accelerated construction, and 334 are planned (Winemiller et al., 2016). The consequences of the rushed development and construction of these plants include underestimation of the disruption on the resilience of local ecosystems and of impacts on the livelihoods of local fishing communities (Santos et al., 2018).

The recent development of large-scale infrastructure projects, such as roads, waterways, mines, and hydroelectric dams, has threatened fisheries in Amazonian aquatic ecosystems (Lees et al., 2016),

* Corresponding author at: Postgraduate Program in Regional Development and Environment (PGDRA), Federal University of Rondônia, Porto Velho, 76801-059, Brazil.

E-mail addresses: alicelima.lima@gmail.com (M.A. Leite Lima), acarvalho.ufrn@gmail.com (A. Rosa Carvalho), marcus.nunes@gmail.com (M. Alexandre Nunes), ronangelini@gmail.com (R. Angelini), carolinarcdoria@gmail.com (C. Rodrigues da Costa Doria).

https://doi.org/10.1016/j.fishres.2019.105399
Received 26 April 2019; Received in revised form 29 September 2019; Accepted 30 September 2019
0165-7836/

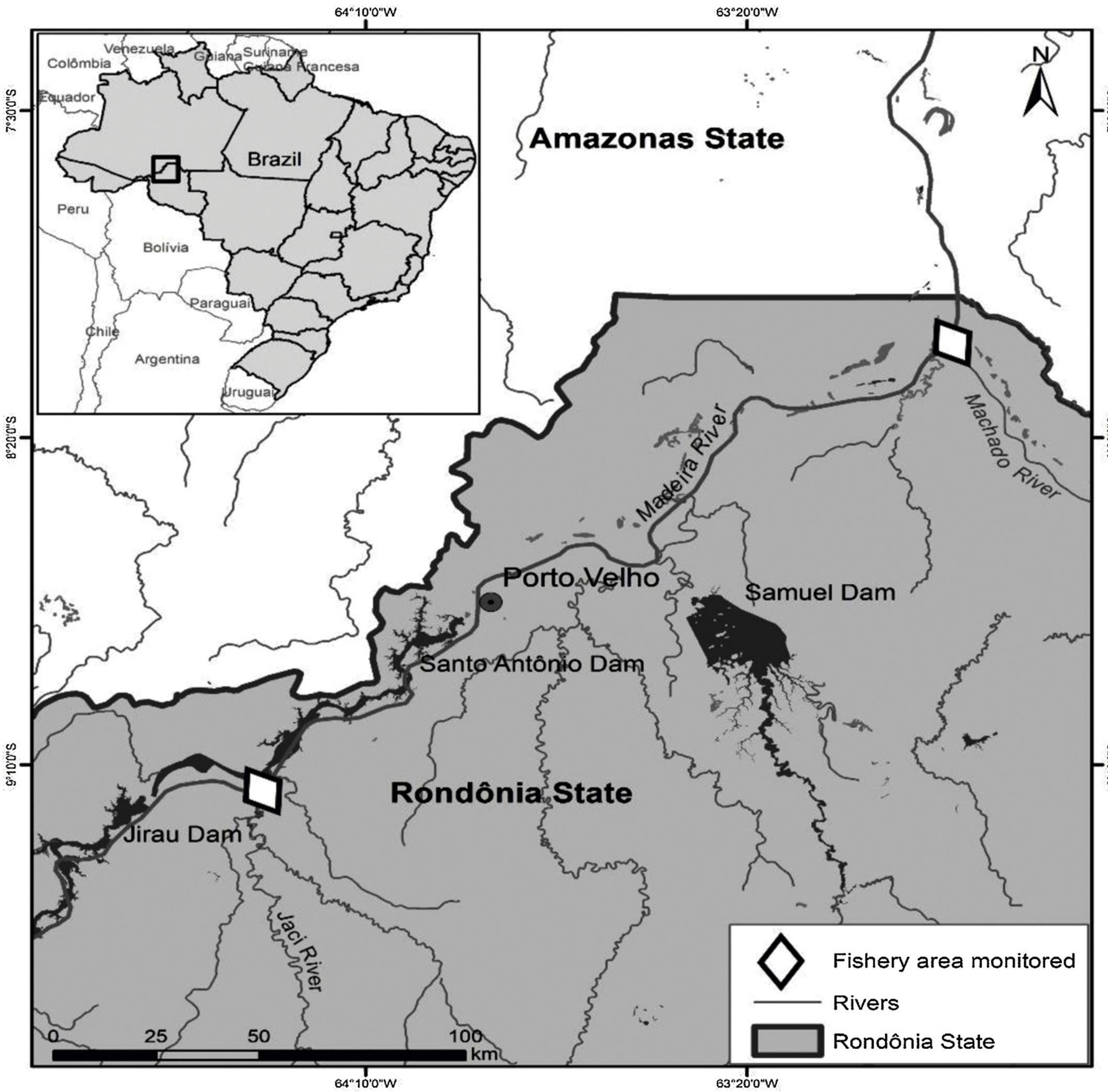

Fig. 1. Madeira River crossing Rondônia State (Brazil) with the Jirau and Santo Antônio Dams (close to Porto Velho City). The fishery area monitored by the Porto Velho fishermen's colony is also highlighted.

introducing major risks to a region that has also been impacted by overfishing (Petrere et al., 2004) and climate change (Freitas et al., 2012). There is clear evidence for the impact of man-made river regulation in the Amazon on bio-ecological processes within the river channel (Agostinho et al., 2008; Almeida et al., 2016; Bunn and Arthington, 2002; Sant'Anna et al., 2014; Santos et al., 2018; Ward and Stanford, 1995) and its effects on formation of shoals, breeding, feeding, nursery and refuge habitats (Bayley, 1991; Goulding, 1980; Nunes et al., 2019).

River connectivity regulates the quantity and species composition of the fish caught and the life cycles of fish species in the Amazon region (Halls and Welcomme, 2004; Jiménez-Segura et al., 2010; Junk et al., 1989; Lima et al., 2017). Fish species migrating thousands of miles from estuarine areas to the Andes for reproductive purposes, are particularly dependent on natural river processes. These fish, together with other products from agriculture and the rainforest, are the means of survival for approximately 30 million people (Anderson et al., 2018; Hauser et al., 2019).

Freshwater fisheries are an economic opportunity for millions of low-income families and provide the majority of dietary protein consumed by rural and urban communities in developing tropical countries (Allan et al., 2005). In Amazon, fish is an essential component in the diet of riverside and indigenous people, reaching up to 500 g of per capita daily fish consumption (Batista and Petrere, 2008; Batista et al., 1998; Isaac et al., 2015), highlighting the societal and economic importance of this ecosystem service in developing countries (Hoeinghaus et al., 2009). As a result, damming in the Amazon impacts both fishers and fish consumers in a region where riverine populations depend enormously and historically on natural resources for income generation, food security and subsistence (Santos et al., 2018).

Fishery production brings US$200 million/year to the Amazon region and employs 200,000 commercial fishers (Tundisi et al., 2014). Given the large array of changes caused by river damming, there are environmental and energetic costs to those who rely on fisheries, due to the changes in catch and species composition (Hoeinghaus et al., 2009; Santos et al., 2018; Wegener et al., 2017; Winemiller et al., 2016). Assessments on the impoundment effects in the Amazon River on fisheries' catches and on economical yields are rarely conducted due to our current lack of long-term time series data (Doria et al., 2016).

This work tests the null hypothesis that total landings (ton), landed values (R$), and coefficients estimated in the regression model of the mean catches and price/kg for migratory and resident species are constant over time following impoundment in a large tropical river. The basic premise is that the changes are caused by an external factor (e.g., anomalous flooding, overexploitation, or river impoundment), which could lead to fewer (or more) fish being landed. Alternatively, catches could be caused by changes in the market value of the fish traded. To identify shifts in catches and economic returns due to river impoundment, we inspected daily landings data (25-year time series) and wholesale prices (19-year time series) for the Madeira River, the largest tributary of the Amazon River, where two large run-of-the-river hydroelectric dams, Santo Antônio and Jirau, were installed between 2011 and 2012 (Hauser et al., 2019).

Specifically, we searched for structural breaks in: (a) total landings (from 1990 to 2014); (b) the average catch of species frequent in at least 60% of landings (or that had their prices registered in 60% of records); (c) the average catch by species category (migratory or resident); (d) the average price of fish; and (e) the landed values per total catches, per migratory species and per resident species. A finding of no significant changes over the course of the time series (particularly, after the river damming) may indicate that the impoundment did not impact fishing landings or the economics of the fishery. In contrast, significant changes indicate impacts. As most of the commercial fish species found in the Madeira River are migratory, and given the importance of catfish as a fishery resource throughout the Amazon basin (Barthem et al., 2017; Doria et al., 2012), any ecological and economic damages from Madeira River damming will have social and ecological impacts far beyond the limits of the Madeira basin.

2. Material and methods

2.1. Study area

The Madeira River (Fig. 1) is a white water river that flows through Brazil, Bolivia, and Peru, draining over 1.4 million km^2 (Latrubesse et al., 2005; Siqueira et al., 2015). The Upper Madeira River carries around 2.1 million tons of sediment per day (PCE et al., 2004). In total, the Madeira River contributes about half of the total sediment transported to the Atlantic Ocean through the Amazon River (Meade, 1994).

Currently, 1008 fish species have been catalogued in the Madeira River watershed: the highest freshwater fish species richness recorded in the world (Ohara et al., 2015). Further, fishery landings from the Madeira River are around 4000 tons per year (Doria et al., 2018), representing approximately 4% of the total Amazonian fish catch (Barthem and Goulding, 2007).

The vast majority of the Madeira River flows through the State of Rondônia ($\approx$ 1700 km^2) where approximately 60 species, including the culturally valuable and high-priced migratory catfishes, have been recorded in the main channel (Doria et al., 2012). Most of the fish catches are Characiforms, Siluriforms, and Perciforms (Lima et al., 2017). The Porto Velho fish market (called *Cai n'água*) in Rondônia is the main landing port and the major trading market in the region. The total fish landings there are around 566 $\pm$ 196 tons/year (Doria et al., 2018) and represents more than 90% of commercial catch in Madeira River, supporting approximately 1200 registered fishermen across the region (Doria et al., 2012). The fishing fleet consists of small wooden fishing vessels (more than 1000 units) with storage capacity between 250 and 600 kg (Doria et al., 2018). The Madeira River fishery occur around Porto Velho city ($\sim$200 km river stretch).

The main stretch of the Madeira River (within the Rondônia border) comprises roughly 18 rapids (Cella-Ribeiro et al., 2013), some of which represent important geographical barriers controlling fish distribution and migration (Goulding et al., 2003; Siqueira et al., 2015; Torrente-Vilara et al., 2011). In 2008, two other important barriers began to be constructed: the Santo Antônio and Jirau hydroelectric power plant dams (HPPs). By 2012, the reservoirs were fully flooded. The dams were installed in the middle portion of the Madeira River (Fig.1)—5 km (Santo Antônio HPP) and 136 km (Jirau HPP), respectively, from Porto Velho (the capital of Rondônia)—and removed two important waterfalls: Teotônio and Caldeirão do Inferno (Hauser et al., 2019). The Madeira River HPPs were the first to be implemented in a white water river in Brazil, as well as to use bulb type turbines (run-of-the-river), which flood relatively small areas (Fearnside, 2014).

2.2. Fisheries data

We used two data series: fishery landings data from 1990 to 2014 and records of species' prices per kg from 1994 to 2013. Data on fish price/kg from 1990 to 1994 were available, but were not used because Brazil's economic instability during this period resulted in large variations in currency values and daily prices, compromising the accuracy of the data.

Both time series datasets (landings and price/kg) were recorded daily by Z-1 Fisher's Colony at the *Cai N'água* landing port in Porto Velho, where the catches were also traded. Landings data included the species and the total weight landed (in kg) while the economic time series included average price/kg and represented the ex-vessel price registered in the fish market. The long partnership between the researchers' group and the Laboratory of Ichthyology and Fisheries at the Federal University of Rondônia (UNIR) allowed the taxonomic identification of fish to the species level when possible (following Queiroz et al., 2013), as well as the supply of the datasets by the Fisher's Colony.

2.3. Data analysis

Unexpected changes in mean values from time series are structural breakpoints. The model for identifying such changes is the standard linear regression model (Zeileis et al., 2002). This approach defines the mean of a time series by coefficients that depend on time, which do not change if there is no structural break in the data.

Thus, in this work, we used the regression model Bai and Perron (2003): defined as $y_t = z_t' \delta_j + u_t,\ t = T_{j-1} + 1, \cdots, T_j,$ (1), for $j = 1, \cdots, m + 1$. In our application, we took y_t to be the total landings, landed values, and observed catches and price/kg of migratory and resident species at time t; z_t is the vector of covariates; δ_j is the correspondent vector of coefficients to be estimated; and u_t is the disturbance at time t. The unknown breakpoints are identified by the indices $T_1, \cdots, T_m$. Therefore, we are interested in estimating both the coefficients δ_j and the breakpoints T_j. We test the null hypothesis that the coefficients δ_j are constant over time following impoundment in a large tropical river. A coefficient change is caused by an external factor (e.g., an anomalous flooding, an overexploitation or a river impoundment), which could lead to fewer (or more) fish being landed and changes in the market value of the fish traded.

The structural break dates were identified using partitions of the dataset and were located at the lowest residual sums of squares (RSSs). The RSSs were calculated using the ordinary least squares cumulative sum (CUSUM) method. Since it is computationally expensive to test all possible partitions in a data set, the minimal RSS was sought through an optimal segmentation, as defined by Bai and Perron (2003). The algorithm found the minimal RSS for each number of breakpoints from 0 (no breakpoints) to 5, yielding six models, each with its Bayesian information criterion (BIC) value. The model with the lowest BIC was chosen as the one with the optimal number of breakpoints (Script available in Link research data).

Specifically, we searched for structural breaks in: (a) total landings (1990–2014); (b) the average catch of species frequent in at least 60% of landings (or that had their prices registered in 60% of records); (c) the average catch by species category (migratory or resident); (d) the average price of fish; and (e) the landed values per total catches and per migratory and resident species.

Weighted average price was the price per species. The price of species per kg was deflated for later conversion into US dollars. The inflation rate used followed the "National Index of Prices for Consumers" (IPCA). Landed values were estimated by multiplying the catch per species (in tons) by the weighted average price/kg/species (in

Table 1
Frequency of observation (%) in the catches (1990–2014) and migratory habits of species (≥ 1%) found in the landings by the small-scale fishery in Madeira River, Amazon. LD = Long-distance migration; MD = medium distance migration; R = resident.

Order/Family	Common name	Scientific name	N of observations in the landings (%)	Migratory habits
Osteoglossiforms				
Arapaimatidae	Pirarucu	*Arapaima gigas*	1.76	R
Osteoglossidae	Aruanã	*Osteoglossum bicirrhosum*	1.05	R
Clupeiforms				
Pristigasteridae	Apapá	*Pellona* spp.	2.19	MD
Characiforms				
Anostomidae	Piau	*Leporinus* spp.	2.88	MD
Characidae	Jatuarana	*Brycon amazonicus*	4.38	MD
	Matrinxã	*Brycon melanopterus*	1.12	MD
	Sardinha	*Triportheus* spp.	4.02	MD
Serrasalmidae	Pacu	*Mylossoma* spp.	4.49	MD
	Piranha	*Serrasalmus* spp.	1.91	R
	Tambaqui	*Colossoma macropomum*	2.36	MD
	Pirapitinga	*Piaractus brachypomus*	2.71	MD
Curimatidae	Branquinha	*Potamorhina* spp.	4.1	MD
Erythrinidae	Traíra	*Hoplias malabaricus*	2.28	R
Prochilodontidae	Curimatã	*Prochilodus nigricans*	4.7	MD
	Jaraqui	*Semaprochilodus* spp.	4.29	MD
Siluriforms				
Callichthyidae	Tamoatá	*Hoplosternum littorale*	1.29	R
Doradidae	Bacu	*	1.07	MD
Loricariidae	Bodó	*Hypostomus* spp.	1.83	R
Pimelodidae	Surubim	*Pseudoplatystoma* spp.	4.75	MD
	Surubim caparari	*Pseudoplatystoma tigrinum*	1.35	MD
	Jaú	*Zungaro zungaro*	3.18	MD
	Barbachata/barbado	*Pinirampus pirinampu*	2.08	MD
	Dourada	*Brachyplatystoma rousseauxii*	4.66	LD
	Babão	*Brachyplatystoma platynemum*	3.05	LD
	Filhote/piraíba	*Brachyplatystoma filamentosum*	2.99	MD
	Piramutaba	*Brachyplatystoma vaillantii*	1.25	MD
	Pirarara	*Phractocephalus hemioliopterus*	3.33	MD
	Jandiá	*Leiarius marmoratus*	1.22	MD
	Coroatá	*Platynematichthys notatus*	1.38	MD
	Mapará	*Hypophthalmus* spp.	1.85	MD
	Mandi	*Pimelodus* aff. *blochii*	2.47	MD
Perciforms				
Cichlidae	Tucunaré	*Cichla* sp.	3.37	R
	Acará	*	2.43	R
Sciaenidae	Pescada	*Plagioscion squamosissimus*	2.66	R
N.I	Mixed species	*	3.95	

USD).

3. Results

From 1990 to 2014, the small-scale fishery in the Madeira River exploited 53 species in a group called "salada" (mix of species), which comprised small-sized species, smaller individuals, and a few individuals from small catches (Table 1). The characiform (63%) and siluriform (25%) fish species comprised most of the landings. The mixture of species ("salada") represented 12% of total landings and included mainly characiforms, osteoglossiforms, and clupeiforms. Of all landings (n = 5505), 32 species represented less than 2% of the weight of the catches.

The mean annual catches for the entire period were around 576.8 t (± 294.8 t). The largest catch was registered in 2008 (1589.01 t), mainly due to catches of pacu (silver dollar fish, *Mylossoma* spp.) and barbachata (Flatwhiskered catfish, *Pinirampus pirinampu*), which represented 35% of total catches. The lowest catch was observed in 2014 (76.83 t).

3.1. Temporal changes in overall catches and prices

Overall catches and prices per year varied over the periods, especially after the start of HPP development (Fig. 2). Average landings had two structural breakpoints, indicating two change events in the mean of total catches for the period from 1990 to 2014 (Table 2; Fig. SM1). The overall weighted average price per kilo of all species harvested from 1994 to 2013 showed three structural breakpoints (Table 2; Fig. SM1).

The structural breakpoints in the catches indicated that, in 2005, mean catches increased by 65% (46.88 to 77.3). Later, they decreased by 58% (to 32.8) from 2008 to 2011, when a lower mean catch was established for total landings (Fig. SM1). With respect to price assessment, the structural breakpoint showed a decrease of 37.5% in 1999 and significant increases in 2006 (109.2%) and 2010 (49.3%; Fig. SM1; values in Table 2).

3.2. Structural breakpoints in average values of most frequent fish species

Of the nine species most frequently caught (≥ 60% of registers), only surubim (*Pseudoplatystoma* spp.) and sardines (*Triportheus* spp.) were not common in economic records (Table 3). Of the species common in both catch records and economic datasets, six had structural breakpoints in the mean catches (Fig. SM2; Table SM1, Supplementary material for details). The greatest extents of change with respect to the average catch were observed in the sardinha (sardine, *Triportheus* spp.) species, for which catches increased by 139.7% (1995), and the dourada (giant catfish, *Brachyplatystoma rousseauxii*), for which catches declined by 74.4% (in 2009).

The last economic change increased the average price per kilo for all six species in either 2010 or 2011. Catfish species had the highest price increase in 2010. Specifically, prices for the dourada, which represented 72% of all catches, increased 88% in relation to the average

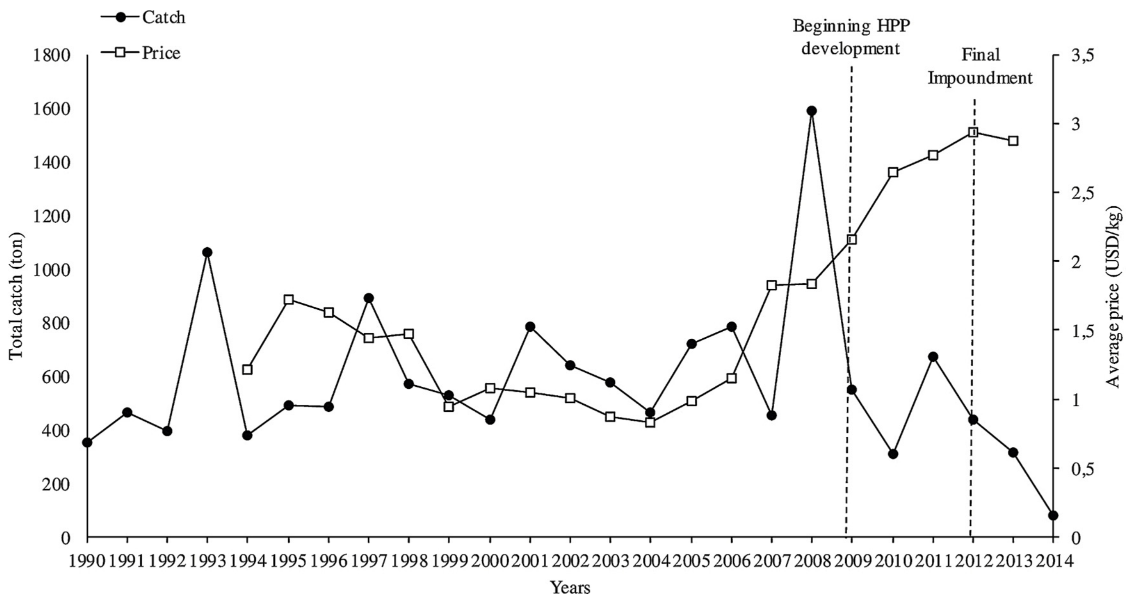

Fig. 2. Total annual catch (ton) of fish landed between 1990 and 2014 and the average annual price (USD) of the kilo of fish between 1994 and 2013 by the small-scale fishery in Madeira River, Amazon. HPP: Hydroelectric Power Plant.

value recorded since 1994. Half of this price increase (43.5%) occurred in 2010, one year after catches sharply dropped.

Of the most frequently caught species, only the pacu (silver dollar fish, *Mylossoma* spp.), jaraqui (kissing prochilodus, *Semaprochilodus* spp.), and surubim (barred sorubim, *Pseudoplatystoma* spp.) species did not experiences significant changes in catches during the period, but changed in average price/kg (Fig. SM3). These price changes occurred mainly in 2009, and one catfish species, the *Pseudoplatystoma* spp. (surubim), which was the third-most reported species in the landings, had the greatest price increase (175.2%; Fig. SM3; Table SM2, Supplementary material).

An additional three catfish species not common in landing records and with no change in catches also had significant price increases. In 2010, these were the babão (*B. platynemum*, by 62%) and filhote (*B. filamentosum*, by 57%) species. The price for jaú (*Zungaro zungaro*) increased in 2006 (Fig. SM4; Table SM3, Supplementary material).

3.3. Changes detected in catches and prices of migratory and resident species

Most of the migratory species caught were medium-distance migratory species (75%), followed by long-distance migratory species (e.g., the Amazon catfish, 14%). Resident species represented the lowest proportion of the catches (4.8%), and a group of fish that could not be identified constituted a slightly higher proportion (6.2%).

- The medium-distance migratory fish species did not exhibit structural breakpoints in average catches during the assessed period. However, their average price/kg showed three structural breakpoints that significantly increased their average values in June 1999, December 2006, and May 2010 (Fig. SM5; Table SM4, Supplementary material).
- The long-distance migratory species showed four significant changes in average catches (Fig. SM5; Table SM4, Supplementary material). Values fluctuated from an increase in average catches in 1995 (51.4% higher) to a drop of 63.8% that established the current lower average catch from 2009 onwards (Fig. SM5). Likewise, the average prices of these species changed four times (Fig. SM5; Table SM4, Supplementary material), and though they dropped twice (after 2005 and markedly after 2010), mean prices sharply increased (Fig. SM5). These patterns seemed to be mostly influenced by changes in the catches and price/kg of dourada (≈ 72% of long-distance

Table 2
Structural breakpoints (SB) in mean values of total catches identified from 1990 to 2014 and structural breakpoints in weighted average price per kilo from 1994 to 2013 to all species harvested by the small-scale fishery in Madeira River, Amazon. NSB = Number of structural breakpoints; delta-BIC = Bayesian information criteria; Date Interval: period of breakpoints identified; CI = Confidence interval relative to bolded date; Mean = structural breakpoint mean value (ton); [1] Lowest BIC, indicating the best fit. Bolded date in each interval indicates the specific month of identified structural breakpoints.

	NSB	BIC	Date Interval	CI (Lim_{inf}-Lim_{sup})	Mean
Total catch (ton)	0	3101			
	1	3098			
	2	3090[1]	1990/Jan – **2005/Mar**	2000/Jul - 2006/Feb	46.88
			2005/Apr – **2008/Dec**	2008/Sep - 2011/Feb	77.30
			2009/Jan – 2014/Dec		32.80
	3	3100			
Mean Price per kg	0	554			
	1	417			
	2	373			
	3	340[1]	1994/Jan – **1999/Jun**	1999/Feb - 1999/Dec	1.57
			1999/Jul – **2006/Nov**	2006/Oct - 2007/Jan	0.98
			2006/Dec – **2010/Oct**	2009/Dec - 2010/Nov	2.05
			2010/Nov – 2013/Dec		3.06
	4	448			

Table 3
Amplitude (%) and year of changes (in brackets and indicated by breakpoint analysis) observed in average values of species most frequently registered in landings and the economic dataset from Madeira River fishery. Negative signals show the percentage of reduction in catches while the absence of signals indicates the percentage of increase in catches. [1]Even though *Pseudoplatystoma* spp. (Surubim/Barred sorubim) was not frequent in economic dataset and did not change in catches it was shown because its influence on overall change of weighted average price from 1994 to 2013.

Species most frequent in catches and economic dataset	Proportion of change (year) indicated by breakpoints	
	% of total catches in ton	% of market value in USD/kg
Brachyplatystoma rousseauxii (Dourada/Giant catfish)	88.48 (1995) − 64.79 (1999) 80.21 (2004) −74.43 (2009)	31.86 (2004) 43.28 (2007) 43.56 (2010)
Brycon amazonicus (Jatuarana/not termed in English)	− 66.63 (1997)	58.42 (2011)
Cichla sp. (Tucunaré/Peacock cichlid)	118.8 (2006)	− 48.20 (1998) 65.34 (2006) 66.46 (2011)
Potamorhina spp. (Branquinha/not termed in English)	− 56.42 (1995)	− 60.60 (1999) 182.05 (2006) 82.72 (2010)
Prochilodus nigricans (Curimatã/Black prochilodus)	− 61.37 (2008)	− 50.83 (1999) 149.15 (2006) 64.98 (2011)
Triportheus spp. (Sardinha/Sardine)	139.70 (1995) − 51.53 (2004)	Not frequent in economic dataset
Pseudoplatystoma spp.[1] (Surubim/Barred sorubim)	Not changed	75 (2006) 175 (2008) (Not frequent in economic dataset)
Mylossoma spp. (Pacu/Silver dolar fish)	Not changed	− 52.38 (2000) 105 (2006) 120.32 (2009)
Semaprochilodus spp. (Jaraqui/Kissing prochilodus)	Not changed	− 58.51 (1999) 123.21 (2006) 84 (2009)
Zungaru zungaru (Jaú/Gilded catfish)	Not frequent in total catches	126.85 (2006)
Brachyplatystoma platynemum (Babão/Slobbering catfish)	Not frequent in total catches	87.09 (2006) 62.06 (2010)
Brachyplatystoma filamentosum (Filhote/Piraíba/Kumakuma)	Not frequent in total catches	43.92 (2004) 36.68 (2007) 57.00 (2010)

migratory species landed; see Fig. SM2).

- Resident fish species had one breakpoint in average catches (Table SM4, Supplementary material), resulting in an increase in late 2006 (Fig. SM5). Temporal changes in resident species were likely influenced by landings of the tucunaré (Fig. SM2), which comprised 55% of the total catches in this group. The other resident species landed were those of acará/oscar (*Astronotus* spp., 27.2%), pirarucu (*Arapaima gigas*, 14%), aruanã/silver aruana (*Osteoglossum bicirrhosum,* 3.7%), and bodó and cascudo (*Hypostomus* spp.). The average prices of resident species experienced three change events, resulting in one price decrease (in late 1998) and two price increases (by 47.7% in 2006 and 64% in 2010; Figure SM5; Table SM4, Supplementary Material).

3.4. Structural breakpoints in average landed values

During the 20 years of the economic data records (from 1994 to 2013), the landed value (tons x price) of total catches decreased slightly (in 1997), then increased (in 2006), and, finally, decreased again (in 2009; Fig. SM6; Table SM6, Supplementary material). Among the most frequently caught species, the average landed value changed only for the dourada (*B. rousseauxii*), tucunaré (*Cichla* sp.) and pacu (*Mylossoma* spp.) species (Table SM6; Fig. SM7, Supplementary material).

The landed value of long-distance migratory species had three breakpoints during the observed time series. Again, the patterns matched the trends observed for the dourada (*B. rousseauxii*; Fig. SM2). Medium-distance migratory species experienced only one change in mean landed value (in 2007), while the average landed value of resident species underwent two change events, which increased average landed value in 2007 and 2009 (Fig. SM6; Table SM5, Supplementary material).

4. Discussion

Our results show that the period of decreasing catches and increasing prices observed for fisheries in the Madeira River matched the construction of the Santo Antônio and Jirau dams, which have been under construction since 2008 (Fearnside, 2014, 2015). According to the results, both dams quickly changed catches and fish supply to market, which were immediately echoed in the price per kg of exploited fish species. Additionally, after the dam construction, prices increased for both fish that became scarce and fish that have became abundant. Though catches declined 58% in 25 years, the price increase of 49% represents a high economic cost for the local population. Moreover, there was a clear decline in the catches of some species (e.g., dourada and curimatã), which contrasted with the increased catches of sardine and tucunaré. However, there was no pattern in this shift, since these last species showed fluctuations in different years (1995 and 2006) suggesting natural oscillations, changes in local habitats, or even changes in fishing efforts.

One of the most negative impacts of a dam is the blocking of migratory fish routes. It is widely known that many Amazonian species move hundreds of miles along the river channel, especially for reproductive purposes. The most vulnerable species is the migratory goliath catfish, which is highly economically profitable. These fish carry out the longest migration of any freshwater fish in the world, crossing several countries throughout their life cycle and making fishery management difficult (Goulding et al., 2019).

In economic terms, consistent excess supply can lead to a drastic decrease in revenue due to price drops, while a decrease in supply (fish) triggers increased prices. When a good (fish) is inelastic, changes in price do not affect people's consumption. In our case, in the Amazon basin, fish are an essential food item. The landing port considered here is the only one in the region and does not have competing markets (Doria et al., 2018). This means that fish landed are not being delivered to other consumers. Likewise, as Amazon has been recorded as the region with the greatest per capita consumption of fish in the world since 1980's (Batista and Petrere, 2008; Batista et al., 1998; Isaac and de Almeida, 2011), any effect of other competing food commodities in the market region would be negligible. In fact, the region has the highest fish intake in world ($\approx$ 169 kg person^{-1} year^{-1}; Doria et al., 2016; Isaac et al., 2015). This level of fish consumption results in an inelastic demand for fish, meaning that the buying behavior of consumers does not change in response to price increases or decreases. Thus, decreasing the fish supply in the Amazon will always result in an increase in fish prices, though fish consumption is unlikely to drop. A persistently low supply of Amazon fish may increase prices even more. The partial or total loss of inundated forests, flood pulses, and ecosystem services (e.g. fisheries) deeply threaten local livelihoods and equitable access to food and economic safety in the Amazon.

Therefore, a change in the river flow and flooding pattern is likely to shift trends in economic returns from fisheries. The destructive potential of damming Amazon Rivers includes habitat loss or alteration, changes in water quality and temperature, a disconnection among fish populations, higher fish mortality, reduced fish diversity, and disruptions to important ecological processes and ecosystem services, such as fisheries (Farias et al., 2010; Gubiani et al., 2010; Pelicice and Agostinho, 2008; Torrente-Vilara et al., 2011; Winemiller et al., 2016). With regard to Amazon fisheries, dams are potentially disastrous to many commercial fish stocks that are related to natural flood pulse dynamics and respond to flood alterations (Górski et al., 2012). The hydrologic changes caused by the construction of dams (Castello and Macedo, 2016) reflect the time, duration, and extension of the flood pulse (Junk et al., 1989; Richter et al., 1997; Van Looy et al., 2014), all of which are determining factors for habitat feasibility for fish feeding, spawning, growth, and refuge areas for many fish species (Barber et al., 2002; Górski et al., 2012; Lima et al., 2017). Further, food cycles have been related to patterns of fish landings (e.g. Arnade et al., 2005; Gates, 2000), and the temporal continuity of Amazon river level has been predicted to be the key variable controlling the continuity of fisheries' yield and, likely, the stability of production within the aquatic system (Vallejos et al., 2013).

Much of contemporary freshwater fisheries management research addresses the fisher rather than the fish (Winfield et al., 2016). However, in the Amazon, both have been disregarded by planners and entrepreneurs, who have failed to assess the true benefits and costs of large hydropower projects in Amazon rivers (Winemiller et al., 2016).

The growing displacement of prime uses of Amazonian rivers for fisheries should be evaluated against hydropower generation, taking into account both the economic costs of modifying downstream river flows and hydrology and the economic benefits of maintaining the integrity of upstream catchments (Emerton, 2005). If the current trend of decisions regarding hydropower exploitation in the Amazon persists, rural communities will not experience energy supply or job creation benefits that exceed the costs of lost fisheries, agriculture, and property (Winemiller et al., 2016). Indeed, in addition to the results presented here, there is also further evidence of recent catch declines (39% in 15 years) in a stretch of the Madeira River under the influence of the impoundments (Santos et al., 2018).

One limitation of our dataset was the lack of information on fishing efforts to confirm that the changes in catches were not due to lower fishing efforts. Theoretically, microeconomics, fishing effort, and climate could synergistically mediate change in catches affecting commercial landings (Stergiou et al., 1997).

However, Brazil has experienced steady economic stability since the start of 2000 until the end of 2014, minimizing the chance of microeconomic reasons causing changes in landings (and prices). The research team carried out fieldwork on Fisher's socioeconomics throughout the entire period (2009–2013). Their daily observations of and connections with the fishers ensured that the number of fishers and the time they spent fishing remained nearly unaltered, at least until 2014. However, at the end of 2014, Brazil fell into an economic crisis, and the Madeira River was simultaneously very affected by the largest flood in the past century. Many fishers had to temporarily leave their homes or lost their entire villages, drastically affecting local fisheries. Even those not affected by flooding in their neighborhoods faced obstructed access to fishing grounds. As a result, this year had fewer landings. The lowest catch observed in 2014 (76.83 t) was likely the result of lower catches being amplified by the fishers' withdrawal from fishing due to the extreme flooding. A much higher amplitude of differences and persistence changes in catches were registered in 2009, during the power plant's construction, while for prices, the significant differences were identified from 2010 ahead, therefore right after total impoundment. However, changes have been persistent and show a pattern that fits the typical regime shift induced by more or less abrupt transition (Gröger et al., 2011). The impoundment of the Madeira River seems to be the only extreme event driving the shifts observed in the landings and whole prices of fish through the period assessed here.

Finally, the database used and provided was only possible due to the participation of the fishers, who, in some places around the world, are reliable and can be used to inform management decisions (Bevilacqua et al., 2019; Carvalho et al., 2009; Philippsen et al., 2016). The use of this dataset to assess the impacts of the dam construction can be particularly valuable to inform the changes for every part of the socio-ecological system. Fisher's experience might be the best source of data in large tropical rivers in remote areas of the world, especially before impounding, when usually little information is available. But more important, fishery scientists have to inspire fishers and provide the means to allow the proper sampling of landing data and the price of the fish. We showed and understand the consequence of damming to the ecosystem. Perhaps the time has come to focus on the consequences for those who rely on fish supply as livelihood and economic insertion.

5. Conclusions

The 25-year time series of fish landings shows that catches were influenced not only by natural events, but also by the effects of the construction of two hydropower dams, which may have contributed to a reduction in the fish production and, consequently, a change in the price and landed value of species. Since fish are the main source of protein for Amazonian populations, the construction of hydroelectric plants in this region impacts not only the environment, but also people and their livelihoods. The Madeira River is a unique ecosystem that has been drastically impacted by two large hydroelectric projects. This river is vital not only to the life cycles of fish species, but also to the rural and urban populations dependent on fish.

The results of this study may help guide public policies to establish consistent and standardized studies on future hydroelectric projects in the Amazon basin. Extensive evidence has clearly demonstrated that the development of hydroelectric dams should be controlled in Amazonian rivers, mainly to maintain the rivers' natural hydrological regimes and the biological dynamics of their ecosystems. The changes in these dynamics impact the stocks and the fishing yield, with direct economic consequences. Studies started before the establishment of a large infrastructural project are needed to obtain continuous and standardized data that can guide post-dam conservation and mitigation strategies.

Acknowledgements

We thank the Fishermen's Colony for the long-term monitoring data, and CNPq (Conselho Nacional de Desenvolvimento Científico e Tecnológico) for the support and fellowship to MALL.

Appendix A. Supplementary data

Supplementary material related to this article can be found, in the online version, at doi:https://doi.org/10.1016/j.fishres.2019.105399.

References

Agostinho, A., Pelicice, F., Gomes, L., 2008. Dams and the fish fauna of the neotropical region: impacts and management related to diversity and fisheries. Braz. J. Biol. 68, 1119–1132. https://doi.org/10.1590/s1519-69842008000500019.

Agostinho, A.A., Thomaz, S.M., Gomes, L.C., 2005. Conservation of the biodiversity of Brazil's inland waters. Conserv. Biol. 19, 646–652. https://doi.org/10.1111/j.1523-1739.2005.00701.x.

Allan, J.D., Abell, R., Hogan, Z., Revenga, C., Taylor, B.W., Welcomme, R.L., Winemiller, K., 2005. Overfishing of inland waters. Bioscience 55, 1041–1051.

Almeida, R.M., Lovejoy, T.E., Roland, F., 2016. Brazil's Amazon conservation in peril. Science 80 (353), 228. https://doi.org/10.1126/science.aag2510.

Anderson, E.P., Jenkins, C.N., Heilpern, S., Maldonado-Ocampo, J.A., Carvajal-Vallejos, F.M., Encalada, A.C., Rivadeneira, J.F., Hidalgo, M., Cañas, C.M., Ortega, H., Salcedo, N., Maldonado, M., Tedesco, P.A., 2018. Fragmentation of Andes-to-Amazon connectivity by hydropower dams. Sci. Adv. 4, 1–7. https://doi.org/10.1126/sciadv.aao1642.

Arnade, C., Pick, D., Gehlhar, M., 2005. Testing and incorporating seasonal structures into demand models for fruit. Agric. Econ. 33, 527–532.

Bai, J., Perron, P., 2003. Computation and analysis of multiple structural change models. J. Appl. Econom. 18, 1–22. https://doi.org/10.1002/jae.659.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R.E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 80 (297), 950–953.
Barber, P.H., Palumbi, S.R., Erdmann, M.V., Moosa, M.K., 2002. Sharp genetic breaks among populations of Haptosquilla pulchella (Stomatopoda) indicate limits to larval transport: patterns, causes, and consequences. Mol. Ecol. 11, 659–674. https://doi.org/10.1046/j.1365-294X.2002.01468.x.
Barthem, R., Goulding, M., 2007. Um Ecossistema Inesperado: a Amazônia revelada pela pesca. Sociedade Civil Mamirauá.
Barthem, R.B., Goulding, M., Leite, R.G., Cañas, C., Forsberg, B., Venticinque, E., Petry, P., Ribeiro, M.L.D.B., Chuctaya, J., 2017. Goliath catfish spawning in the far western Amazon confirmed by the distribution of mature adults, drifting larvae and migrating juveniles. Nat. Publ. Gr. 7, 1–13. https://doi.org/10.1038/srep41784.
Batista, V., Petrere Jr, M., 2008. Spatial and temporal distribution of fishing resources exploited by the Manaus fishing fleet, Amazonas, Brazil. Braz. J. Biol. 67, 651–656. https://doi.org/10.1590/s1519-69842007000400009.
Batista, V.S., Inhamuns, A.J., Freitas, C.E.C., Freire-Brasil, D., 1998. Characterization of the fishery in river communities in the low-Solimoes/high-Amazon region. Fish. Manage. Ecol. 5, 419–435. https://doi.org/10.1046/j.1365-2400.1998.550419.x.
Bayley, P.B., 1991. The flood pulse advantage and the restoration of river-floodplain systems. Regul. Rivers Res. Manage. 6, 75–86. https://doi.org/10.1002/rrr.3450060203.
Bevilacqua, A.H.V., Angelini, R., Steenbeek, J., Christensen, V., Carvalho, A.R., 2019. Following the fish: the role of subsistence in a fish-based value chain. Ecol. Econ. 159, 326–334. https://doi.org/10.1016/j.ecolecon.2019.02.004.
Bunn, S.E., Arthington, A.H., 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ. Manage. 30, 492–507. https://doi.org/10.1007/s00267-002-2737-0.
Carvalho, A.R., Williams, S., January, M., Sowman, M., 2009. Reliability of community-based data monitoring in the Olifants River estuary (South Africa). Fish. Res. 119–128. https://doi.org/10.1016/j.fishres.2008.08.017.
Castello, L., Macedo, M.N., 2016. Large-scale degradation of Amazonian freshwater ecosystems. Glob. Change Biol. 22, 990–1007. https://doi.org/10.1111/gcb.13173.
Cella-Ribeiro, A., Torrente-Vilara, G., Hungria, D.B., Oliveira, M., 2013. As corredeiras do rio Madeira. In: Queiroz, L.J., Torrente-Vilara, G., Ohara, W.M., Pires, T., Zuanon, J., Doria, C.R.C. (Eds.), Peixes Do Rio Madeira. Dialeto Latin American Documentary, pp. 56–63 São Paulo, SP.
Costanza, R., D'Arge, R., Groot, R.D., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., Van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260. https://doi.org/10.1016/S0921-8009(98)00020-2.
Doria, C.R., da, C., Machado, L.F., Brasil de Souza, S.T., Lima, M.A.L., 2016. A pesca em comunidades ribeirinhas na região do médio rio Madeira, Rondônia. Novos Cad. NAEA 19, 163–188. https://doi.org/10.5801/ncn.v19i3.2499.
Doria, C.R.C., Lima, M.A.L., Angelini, R., 2018. Ecosystem indicators of a small-scale fisheries with limited data in Madeira River (Brazil). Bol. Inst. Pesca 44, e317. https://doi.org/10.20950/1678-2305.2018.317.
Doria, C.R.C., Ruffino, M.L., Hijazi, N.C., Cruz, R.L., 2012. A pesca comercial na bacia do rio Madeira no estado de Rondônia, Amazônia brasileira. Acta Amaz. 42, 29–40.
Dudgeon, D., Arthington, A.H., Gessner, M.O., Kawabata, Z.-I., Knowler, D.J., Lévêque, C., Naiman, R.J., Prieur-Richard, A.-H., Soto, D., Stiassny, M.L.J., Sullivan, C.A., 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biol. Rev. 81, 163–182. https://doi.org/10.1017/S1464793105006950.
Emerton, L., 2005. Values and Rewards: Counting and Capturing Ecosystem Water Services for Sustainable Development. IUCN - The World Conservation Union.
Farias, I.P., Torrico, J.P., García-dávila, C., Santos M da, C.F., Hrbek, T., Renno, J.-F., 2010. Are rapids a barrier for floodplain fishes of the Amazon basin? A demographic study of the keystone floodplain species Colossoma macropomum (Teleostei: Characiformes). Mol. Phylogenet. Evol. 56, 1129–1135. https://doi.org/10.1016/j.ympev.2010.03.028.
Fearnside, P.M., 2015. Tropical hydropower in the clean development mechanism: Brazil's Santo Antônio Dam as an example of the need for change. Clim. Change 131, 575–589. https://doi.org/10.1007/s10584-015-1393-3.
Fearnside, P.M., 2014. Impacts of Brazil's Madeira River dams: unlearned lessons for hydroelectric development in Amazonia. Environ. Sci. Policy 38, 164–172. https://doi.org/10.1016/j.envsci.2013.11.004.
Freitas, C.E.C., Siqueira-Souza, F.K., Humston, R., Hurd, L.E., 2012. An initial assessment of drought sensitivity in Amazonian fish communities. Hydrobiologia 159–171. https://doi.org/10.1007/s10750-012-1394-4.
Gates, J.M., 2000. Input substitution in a trap fishery. ICES J. Mar. Sci. 57, 89–108. https://doi.org/10.1006/jmsc.1999.0561.
Górski, K., Van den Bosch, L.V., Van de Wolfshaar, K.E., Middelkoop, H., Nagelkerke, L.A.J., Filippov, O.V., Zolotarev, D.V., Yakovlev, S.V., Minin, A.E., Winter, H.V., De Leeuw, J.J., Buijse, A.D., Verreth, J.A.J., 2012. Post-damming flow regime development in a Large Lowland River (Volga, Russian Federation): implications for floodplain inundation and fisheries. River Res. Appl. 28, 1121–1134. https://doi.org/10.1002/rra.1499.
Goulding, M., 1980. The Fishes and the Forest: Explorations in Amazonian Natural History. University of California Press.
Goulding, M., Barthem, R., Ferreira, E.J.G., 2003. The Smithsonian Atlas of the Amazon. Smithsonian Institution Press, Washingyon, D.C.
Goulding, M., Venticinque, E., Ribeiro, M.L.D.B., Barthem, R.B., Leite, R.G., Forsberg, B., Petry, P., Lopes da Silva-Júnior, U., Polliana, S.F., Cañas, C., 2019. Ecosystem-based management of Amazon fisheries and wetlands. Fish Fish 20, 138–158. https://doi.org/10.1111/faf.12328.
Gröger, J.P., Missong, M., Rountree, R.A., 2011. Analyses of interventions and structural breaks in marine and fisheries time series: detection of shifts using iterative methods. Ecol. Indic. 11, 1084–1092. https://doi.org/10.1016/j.ecolind.2010.12.008.
Gubiani, E.A., Gomes, L.C., Agostinho, A.A., Baumgartner, G., 2010. Variations in fish assemblages in a tributary of the upper Paraná River, Brazil: a comparison between pre and post-closure phases of dams. River Res. Appl. 26, 848–865. https://doi.org/10.1002/rra.
Halls, A.S., Welcomme, R.L., 2004. Dynamics of River fish populations in response to hydrological conditions: a simulation study. River Res. Appl. 20, 985–1000. https://doi.org/10.1002/rra.804.
Hauser, M., Doria, C.R.C., Santos, R.V., García-Vasquez, A., Pouilly, M., Pécheyran, C., Ponzevera, E., Torrente-Vilara, G., Bérail, S., Panfili, J., Darnaude, A., Renno, J.-F., García-Dávila, C., Nuñez, J., Ferraton, F., Vargas, G., Duponchelle, F., 2019. Shedding light on the migratory patterns of the Amazonian goliath catfish, Brachyplatystoma platynemum, using otolith 87Sr/86Sr analyses. Aquat. Conserv. Mar. Freshw. Ecosyst 1–12. https://doi.org/10.1002/aqc.3046.
Hoeinghaus, D.J., Agostinho, A.A., Gomes, L.C., Pelicice, F.M., Okada, E.K., Latini, J.D., Kashiwaqui, E.A.L., Winemiller, K.O., 2009. Effects of river impoundment on ecosystem services of large tropical rivers: embodied energy and market value of artisanal fisheries. Conserv. Biol. 23, 1222–1231. https://doi.org/10.1111/j.1523-1739.2009.01248.x.
Isaac, V., Almeida, M., Cruz, R., Nunes, L., 2015. Artisanal fisheries of the Xingu River basin in Brazilian Amazon. Braz. J. Biol. 75, 125–137. https://doi.org/10.1590/1519-6984.00314bm.
Isaac, V.J., de Almeida, M.C., 2011. El Consumo de pescado en la Amazonía brasileña. Roma. .
Jiménez-Segura, L.F., Palacio, J., Leite, R., 2010. River flooding and reproduction of migratory fish species in the Magdalena River basin, Colombia. Ecol. Freshw. Fish 19, 178–186. https://doi.org/10.1111/j.1600-0633.2009.00402.x.
Junk, W.J., Bayley, P.B., Sparks, R.E., 1989. The flood pulse concept in river-floodplain systems. In: Proceedings of the International Large River Symposium. Can. Spec. Publ. Fish. Aquat. Sci. pp. 106.
Latrubesse, E.M., Stevaux, J.C., Sinha, R., 2005. Tropical rivers. Geomorphology 70, 187–206. https://doi.org/10.1016/j.geomorph.2005.02.005.
Lees, A.C., Peres, C.A., Fearnside, P.M., Schneider, M., Zuanon, J.A.S., 2016. Hydropower and the future of Amazonian biodiversity. Biodivers. Conserv. 25, 451–466. https://doi.org/10.1007/s10531-016-1072-3.
Lima, M.A.L., Kaplan, D.A., Rodrigues da Costa Doria, C., 2017. Hydrological controls of fisheries production in a major Amazonian tributary. Ecohydrology 1–19. https://doi.org/10.1002/eco.1899.
Meade, R.H., 1994. Suspended sediments of the modern Amazon and Orinoco rivers. Quat. Int. 21, 29–39.
Nunes, M.U.S., Hallwass, G., Silvano, R.A.M., 2019. Fishers' local ecological knowledge indicate migration patterns of tropical freshwater fish in an Amazonian river. Hydrobiologia 3. https://doi.org/10.1007/s10750-019-3901-3.
Ohara, W.M., de Queiroz, L.J., Zuanon, J., Torrente-Vilara, G., Vieira, F.G., Doria, C.D.C., 2015. Fish collection of the Universidade Federal de Rondônia: its importance to the knowledge of Amazonian fish diversity. Acta Sci. Biol. Sci. 37, 251–258. https://doi.org/10.4025/actascibiolsci.v37i2.26920.
PCE, Furnas, CNO, 2004. Complexo Hidrelétrico do Rio Madeira. Estudos de viabilidade do AHE Jirau. Processo N° 48500.000104/03-53. Rio de Janeiro - RJ. .
Pelicice, F.M., Agostinho, A.A., 2008. Fish-passage facilities as ecological traps in large neotropical rivers. Conserv. Biol. 22, 180–188. https://doi.org/10.1111/j.1523-1739.2007.00849.x.
Pelicice, F.M., Azevedo-Santos, V.M., Vitule, J.R.S., Orsi, M.L., Lima Junior, D.P., Magalhães, A.L.B., Pompeu, P.S., Petrere Jr, M., Agostinho, A.A., 2017. Neotropical freshwater fishes imperilled by unsustainable policies. Fish Fish 1–15. https://doi.org/10.1111/faf.12228.
Petrere, M., Barthem, R.B., Córdoba, E.A., Gómez, B.C., 2004. Review of the large catfish fisheries in the upper Amazon and the stock depletion of piraíba (Brachyplatystoma filamentosum Lichtenstein). Rev. Fish Biol. Fish. 14, 403–414. https://doi.org/10.1007/s11160-004-8362-7.
Philippsen, J.S., Minte-Vera, C.V., Okada, K.E., Carvalho, A.R., Angelini, R., 2016. Fishers' and scientific histories: an example of consensus from an inland fishery. Mar. Freshw. Res.
Queiroz, L.J., Torrente-Vilara, G., Ohara, W.M., Pires, T.H.S., Zuanon, J., Doria, C.R.C., 2013. Peixes do rio Madeira. Dialeto Latin American Documentary, 1. ed. Porto Velho -RO.
Richter, B.D., Baumgartner, J.V., Wigington, R., Braun, D.P., 1997. How much water does a river need? Freshw. Biol. 37, 231–249.
Sala III, O.E., F.S.C, Armesto, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M., Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science 80 (287), 1770–1774.
Sant'Anna, I.R.A., Doria, C.R.C., Freitas, C.E.C., 2014. Pre-impoundment stock assessment of two Pimelodidae species caught by small-scale fisheries in the Madeira River (Amazon Basin – Brazil). Fish. Manage. Ecol. 1–8. https://doi.org/10.1111/fme.12082.
Santos, R.E., Pinto-Coelho, R.M., Fonseca, R., Simões, N.R., Zanchi, F.B., 2018. The decline of fisheries on the Madeira River, Brazil: the high cost of the hydroelectric dams in the Amazon Basin. Fish. Manage. Ecol. 1–12. https://doi.org/10.1111/fme.12305.
Siqueira Jr, J.L., Tomasella, J., Rodriguez, D.A., 2015. Impacts of future climatic and land cover changes on the hydrological regime of the Madeira River basin. Clim. Change 129, 117–129. https://doi.org/10.1007/s10584-015-1338-x.
Stergiou, K.I., Christou, E.D., Petrakis, G., 1997. Modelling and forecasting annual fisheries catches: comparison of regression, univariate and multivariate time series

methods. Fish. Res. 29, 55–95. https://doi.org/10.1016/0165-7836(95)00389-4.
Torrente-Vilara, G., Zuanon, J., Leprieur, F., Oberdorff, T., Tedesco, P.A., 2011. Effects of natural rapids and waterfalls on fish assemblage structure in the Madeira River (Amazon Basin). Ecol. Freshw. Fish 20, 588–597. https://doi.org/10.1111/j.1600-0633.2011.00508.x.
Tundisi, J.G., Goldemberg, J., Matsumura-Tundisi, T., Saraiva, A.C.F., 2014. How many more dams in the Amazon? Energy Policy 74, 703–708. https://doi.org/10.1016/j.enpol.2014.07.013.
Vallejos, R.O., Fabré, N.N., da Silva Batista, V., Acosta, J., 2013. The application of a general time series model to floodplain fisheries in the Amazon. Environ. Model. Softw. 48, 202–212. https://doi.org/10.1016/j.envsoft.2013.07.004.
Van Looy, K., Tormos, T., Souchon, Y., 2014. Disentangling dam impacts in river networks. Ecol. Indic. 37, 10–20. https://doi.org/10.1016/j.ecolind.2013.10.006.
Ward, J.V., Stanford, J.A., 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regul. Rivers Res. Manage. 11, 105–119. https://doi.org/10.1002/rrr.3450110109.
WCD, 2000. Dams and development: A New Framework for Decision-Making. London.
Wegener, M.G., Schramm, H.L., Neal, J.W., Gerard, P.D., 2017. Effect of fishing effort on catch rate and catchability of largemouth bass in small impoundments. Fish. Manage. Ecol. 1–11. https://doi.org/10.1111/fme.12268.
Winemiller, K.O., McIntyre, P.B., Castello, L., Fluet-Chouinard, E., Giarrizzo, T., Nam, S., Baird, I.G., Darwall, W., Lujan, N.K., Harrison, I., Stiassny, M.L.J., Silvano, R.A.M., Fitzgerald, D.B., Pelicice, F.M., Agostinho, A.A., Gomes, L.C., Albert, J.S., Bara, E., Petrere Jr, M., Zarfl, C., Mulligan, M., Sullivan, J.P., Arantes, C.C., Sousa, L.M., Koning, A.A., Hoeinghaus, D.J., Sabaj, M., Lundberg, jg, Armbruster, J., Thieme, M.L., Petry, P., Zuanon, J., Torrente-Vilara, G., Snoeks, J., Ou, C., Rainboth, W., Pavanelli, C.S., Akama, A., Van Soesbergen, A., Sáenz, L., 2016. Balancing hydropower and biodiversity in the Amazon, Congo, and Mekong. Science 80 (351), 128–129. https://doi.org/10.1126/science.aac7082.
Winfield, I.J., Baigún, C., Balykin, P.A., Becker, B., Chen, Y., Filipe, A.F., Gerasimov, Y.V., Godinho, A.L., Hughes, R.M., Koehn, J.D., Kutsyn, D.N., Mendoza-Portillo, V., Obedorff, T., Orlov, A.M., Pedchenko, A.P., Pletterbauer, F., Prado, I.G., Rosch, R., Vatland, S.J., 2016. International perspectives on the effects of climate change on inland fisheries. Fisheries 41, 399–405. https://doi.org/10.1080/03632415.2016.1182513.
Zeileis, A., Kleiber, C., Walter, K., Hornik, K., 2002. Testing and Dating of Structural Changes in Practice (No. 2002), Reduction of Complexity in Multivariate Data Structures SBF. 475. https://doi.org/10.1016/S0167-9473(03)00030-6.