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PRIMEIRAS HIDRELÉTRICAS EM RIO DE ÁGUAS BRANCAS NA AMAZÔNIA: RESISTÊNCIA DA ICTIOFAUNA BENTÔNICA

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RESUMO

O barramento de rios para geração de eletricidade é uma das atividades mais impactantes em sistemas aquáticos. São inúmeras as consequências ambientais decorrentes de grandes barragens, incluindo impactos diretos nas propriedades biológicas, químicas e físicas dos rios. Recentemente a bacia Amazônica tem experimentado um *boom* de construção de hidrelétricas, colocando a sua grande biodiversidade em risco. Durante seis anos foram conduzidos estudos ictiofaunísticos no rio Madeira, três anos antes e três anos depois da construção em cascata de duas grandes barragens (UHEs Jirau e Santo Antônio). Foram amostradas assembleias de peixes dependentes da planície de inundação (pelágicos) e do fundo do canal principal do rio Madeira (bentônicos). Para entender os efeitos dos reservatórios em cascata minha tese foi organizada em três capítulos. O primeiro capítulo apresenta a relação peso-comprimento (RPC) de 112 espécies de peixes, representando 23 família e cinco ordens capturadas no rio Madeira. Oito novos registros de RPC foram apresentados para espécies amazônicas conforme informação da base Fish-Base. RPCs provem um registro histórico de dados para os coeficientes *a* e *b* antes do barramento do rio Madeira, permitindo comparações futuras dos impactos promovidos nesses parâmetros populacionais. O segundo capítulo traz um artigo sobre a distribuição espaço-temporal de jovens dos gigantes pimelodídios do gênero *Brachyplatystoma* spp. Larvas e juvenis estiveram presentes em baixas abundâncias em todas as áreas e durante todo o ciclo hidrológico. A presença de larvas e jovens da espécie *Brachyplatystoma rousseauxii* ao longo do canal do rio Madeira por todo o período hidrológico sugere eventos de desova assincrônica nas cabeceiras do rio Madeira. Finalmente, nós examinamos as primeiras respostas das assembleias de peixes, tanto pelágicos como bentônicos, três anos antes e três anos depois dos barramentos. As principais mudanças ocorreram nos atributos de riqueza, abundância e biomassa das espécies, sem diferenças notáveis na diversidade e equitabilidade das espécies antes e depois dos barramentos. Observamos alterações imediatas na assembleia associada a planície de inundação, e foram relacionadas as mudanças ambientais locais. Já a ictiofauna bentônica do rio Madeira não apresentou alterações após os barramentos. Nosso estudo é relevante no tocante as respostas da ictiofauna em relação ao tipo de reservatório fio d’água, com impactos menos intensos em relação a regiões com reservatórios tradicionais.

Palavras-chave: Amazônia; água branca; reservatórios em cascata; peixes de água doce.

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ABSTRACT

River damming for hydropower generation is one of the most impact activities in river systems. The environmental consequences of large dams are numerous, including direct impacts on biological, chemical and physical properties of rivers environments. Recently the Amazon basin are experiencing an unprecedented boom in construction of hydropower dams. Studies were conducted at six years in the Madeira River, three years before and three years after two reservoirs built in cascade in the largest white-water tributary of the Amazon River (Jirau and Santo Antônio dams'). Fish assemblages depending on floodplains areas (pelagic) and bottom main channel (benthic) were sampled with gillnets and bottom trawl net respectively. To understand the effect of reservoirs in cascade my thesis was organized into three chapters. The first chapter present the length–weight relationships (LWR) for 112 freshwater fish species representing 23 families and five orders captured in the Madeira River. Eight new LWR records are presented for Amazonian species as information for Fish-Base database. LWRs provided historical data about *a* and *b* coefficients prior to the damming of the Madeira River, allowing comparison of estimates to predicted future population parameters influenced by human impacts. The second chapter brings an article about the temporal and spatial distribution of young giants' catfish *Brachyplatystoma* spp. Larvae and juveniles were present in low abundances in all areas and during all hydrological periods. The presence of larvae and juveniles of *Brachyplatystoma rousseauxii* along the channel throughout the hydrological cycle suggests spawning regulated by asynchronous floods events in the headwaters of the Madeira River. Finally, we examined responses from fish assemblages depending on floodplains areas (pelagic) and bottom main channel (benthic) three years before and three years after the dams. Overall, there were major differences in species richness, biomass and density but no significant differences in fish diversity and evenness before and after dam construction. Differences in pelagic fish assemblage composition into both reservoirs were noticed immediately, and linked to the increasing in conductivity, deep, oxygen and temperature while channel bottom fish assemblage seems to not respond immediately to the dams' closure. Our study is especially relevant considering responses from fish assemblages to bulb turbines face to burgeoning new hydroelectricity development plans for rivers across Amazonia.

Keywords: Amazon; muddy-water; cascade reservoirs; freshwater fish.

SUMÁRIO

Capítulos seguem numeração de acordo com a origem da publicação.

INTRODUÇÃO

A vazão das águas representa a principal força reguladora dos ecossistemas de água doce, sendo responsável pela estrutura geológica e hidrológica, dinâmica produtiva, ciclagem de nutrientes, distribuição e evolução da biota de rios e riachos (Poff *et al.*, 1997). Apesar da relevância de se manter o fluxo das águas livre, algumas atividades humanas têm promovido fortes interferências pela construção de barragens em sistemas lóticos. O barramento de rios para geração de eletricidade é uma das principais atividades de impacto antrópico (Gido e Matthews, 2000), alterando drasticamente a vazão das águas em diferentes locais do mundo (Vörösmarty *et al.*, 2004; Syvitski *et al.*, 2005; Vörösmarty *et al.*, 2010). E os efeitos deletérios de impedir o fluxo natural das águas tem orientado países precursores no desenvolvimento desses sistemas a recuperar rios pela retirada de barragens (O'connor *et al.*, 2015).

Barrar um rio é um processo intenso e dramático que resulta na criação de um novo ecossistema (Baxter, 1977), com estrutura, biota e funcionamento diferenciados do sistema natural e regidos por processos semelhantes independente do sistema (rio) impactado (Agostinho *et al.*, 2008). Após o barramento do rio forma-se uma zona de reservatório, resultado da transição de um ambiente lótico representado pelo rio em suas condições naturais subitamente transformado em ambiente lêntico e, em alguns casos recentes em semi-lêntico (Furnas, 2005; Waters e Aggidis, 2015). A drástica modificação ambiental ocorre de forma rápida e intensa, impondo novos processos que resultam na extinção local de espécies cuja parte do ciclo de vida é interrompido, alterando a diversidade e a função ecológica dos rios (Bunn e Arthington, 2002; Agostinho *et al.*, 2008).

O Brasil tem nos rios uma importante fonte de geração de energia elétrica, com ~80% da energia atual proveniente de hidrelétricas (Matriz Energética Nacional 2030, 2007; Lees *et al.*, 2016; Prado *et al.*, 2016). A maior parte dos barramentos já instalados estão concentrados nas porções Sul, Sudeste e Centro-Oeste do Brasil (Agostinho *et al.*, 2008; Mello, 2011). Na Amazônia brasileira poucas usinas foram instaladas até a década de 90. Duas delas podem ser consideradas de pequeno porte, são elas a Curuá-Una, no Pará, e Paredão, no Amapá (Ferreira, 1993). Uma considerada de médio porte, a UHE Samuel no rio Jamari em Porto Velho-RO (Santos, 1995). E duas consideradas de grande porte, a UHE Tucuruí no Pará, e Balbina, que apesar de ser considerada de grande porte em função da grande área alagada, possui baixa eficiência de produção energética (9,44 km2/MW; Fearnside, 1990). No ano de 2002 foi finalizado o enchimento do reservatório da Usina de Lajeado (ou Usina Hidroelétrica Luís

Eduardo Magalhães), inundando um trecho do alto e médio rio Tocantins (Araújo *et al.*, 2009). Recentemente outras grandes hidrelétricas estão sendo construídas na Amazônia, Santo Antônio e Jirau no rio Madeira, e Belo Monte no rio Xingu. Todas com as obras em estágio avançado e seus reservatórios já foram estabelecidos (informações disponíveis em: http://www.santoantonioenergia.com.br/, http://www.energiasustentaveldobrasil.com.br/ e http://norteenergiasa.com.br/site/).

No caso do rio Madeira, as peculiaridades do seu trecho encachoeirado, como a grande vazão das águas e ausência de altas quedas d'água abruptas, exigiram especificidades para o aproveitamento da vazão com diferenças estruturais e técnicas hidráulicas distintas das hidrelétricas tradicionalmente instaladas no Brasil. As hidrelétricas tradicionais possuem modelos de turbinas que exigem alta queda d'água, e consequentemente reservatórios de acumulação com grandes extensões (Fearnside, 1990; Ferreira, 1993; Santos, 1995). A engenharia mecânica desenvolveu técnicas hidráulicas que foram aplicadas nos empreendimentos do rio Madeira e Xingu, onde turbinas do tipo *bulbo* permitem maior aproveitamento da vazão a partir de quedas menos acentuadas e com áreas de reservatórios reduzidos, o que tem sido denominado de usinas a fio d'água. Esses reservatórios não reservam grandes estoques de água e necessitam de pequena queda d'água < 20m (http://www.santoantonioenergia.com.br/pt/tecnologia). É importante ressaltar que independentemente do tipo de turbina utilizada, do tamanho do reservatório ou do tempo de permanência da água são esperados impactos promovidos em relação a alteração do fluxo natural do rio, além da barreira física da barragem que ocorrerá de qualquer forma (Fearnside, 2013; 2014).

Com grande disponibilidade de recursos hídricos, a Amazônia Legal vem sendo colocada como área central para solução energética dos países amazônicos. Pelo menos 191 hidrelétricas já em funcionamento (Lees *et al.*, 2016) e 330 planejadas ou em construção (Fearnside, 2016). Esses empreendimentos fazem parte de uma política internacional. O G20 estabeleceu o uso entre US$ 60 trilhões e US$ 70 trilhões para novas infraestruturas até 2030, financiados por grandes investidores como Banco Mundial, Asian Infrastructure Investment Bank (AIIB, Ásia) e Banco Nacional do Desenvolvimento (BNDS, Brasil). Em torno de 90 % desses projetos serão desenvolvidos nas nações em desenvolvimento, em regiões tropicais e subtropicais, com maiores riquezas biológicas e mais complexos ecossistemas (Laurance, 2016). Milhares de novas barragens fazem parte desse programa, e incluem o uso de rio amazônicos. Financeiramente, perdas por impactos e problemáticas oriundos desses novos barramentos na Amazônia podem ser maiores do que realmente irão valer esses empreendimentos (ver Fearnside, 2016).

Esses empreendimentos hidrelétricos em grandes rios da Amazônia tornaram-se prioridade para o governo brasileiro, com projeto de uso de pelo menos 20 cursos d'água até meados de 2020 (Eletrobrás, 1987; Matriz Energética Nacional 2030, 2007; Lees *et al.*, 2016; Prado *et al.*, 2016). A construção das duas UHE's no trecho de corredeiras do rio Madeira, Santo Antônio na cachoeira de Santo Antônio e Jirau na corredeira Caldeirão do Inferno (a jusante da cachoeira Jirau) fazem parte da consolidação desta política energética para região Amazônica.

A extensão territorial e hídrica da Amazônia é refletida na sua espetacular diversidade de espécies de peixes (Reis *et al.*, 2003; Buckup *et al.*, 2007; Winemiller e Willis, 2011). A ictiofauna da Amazônia ocupa uma diversidade de ambientes aquáticos, presentes tanto em águas claras, brancas como pretas (Sioli, 1968). São conhecidas pelo menos 2,3 mil espécies de peixes com ocorrência para região amazônica, e estima-se que muitas outras ainda poderão ser descritas a partir do aprofundamento dos estudos taxonômicos na região (Reis *et al.*, 2003; Buckup *et al.*, 2007; Winemiller e Willis, 2011; Ohara *et al.*, 2015). Mais recentemente, assembleias de peixes que vivem associados ao fundo do canal dos grandes rios amazônicos tem sido objeto de vários estudos que têm mostrado que muitas espécies de ocorrência rara em águas rasas são abundantes nesses ambientes mais profundos (Lopez-Rojas *et al.*, 1984; Mago-Leccia *et al.*, 1985; Cox Fernandes, 1995; Lundberg *et al.*, 1996; Cox Fernandes, 1999; Cox Fernandes *et al.*, 2004; Thomé-Souza e Chao, 2004; Cella-Ribeiro e Crampton, 2013; Lundberg *et al.*, 2013; Crampton e Cella-Ribeiro, 2013a; Crampton e Cella-Ribeiro, 2013b; 2013c; 2013d).

Essa crescente onda de novas UHEs em rios amazônicos disponibilizou um aporte financeiro para estudos ictiofaunísticos nas áreas a serem impactadas. Pesquisadores necessitam de tempo para desenvolver atividades de coleta, identificação e estudos específicos para conhecer a ictiofauna de um determinado local previamente a um impacto. Grandes esforços são empregados para compreender os padrões e processos relacionados a ictiofauna alvo do impacto. Contudo, o atual processo de licenciamento de empreendimentos no Brasil tem sido historicamente definido pelo estabelecimento de uma hidrelétrica antes de implementar estudos que reconheçam a dinâmica aquática do sistema e mensurem a real perda da biodiversidade. O descompasso entre a engenharia e os estudos ambientais prometem resultar em perdas irreparáveis na escala da bacia, frente ao número de empreendimento previstos para grandes rios Amazônicos (Lees *et al.*, 2016; Prado *et al.*, 2016; Winemiller *et al.*, 2016). Essa não é uma problemática exclusiva do Brasil, como observado em outros grandes rio do mundo considerados como áreas prioritárias para implementação de UHEs (além da bacia Amazônica, as bacias do Congo e Mekong; Winemiller *et al.*, 2016).

Uma limitação para entender as mudanças ambientais promovidas por atividades humanas é a ausência de informação prévia aos impactos (Doria *et al.*, 2014). Muitas vezes tem sido utilizadas compilações incompletas de dados pretéritos, obtidos em museus biológicos e outros registros históricos (Da Fonseca *et al.*, 2002) que não são suficientes para recuperar o estado pretérito da ictiofauna e seu funcionamento do sistema antes do impactado. Nesse sentido, rios com histórico de dados pretéritos a um barramento são considerados importantes modelos a serem estudados para o entendimento dos impactos causados por barramentos (Winemiller *et al.*, 2016). A possibilidade de conduzir estudos pretéritos aos impactos das UHEs Jirau e Santo Antônio, inserem o rio Madeira como um modelo importante a ser estudado.

Os estudos no rio Madeira foram possíveis pela parceria entre o Laboratório de Ictiologia e Pesca da Universidade Federal de Rondônia e Instituto Nacional de Pesquisas da Amazônia em parceira com Furnas Centrais Elétricas (2003 a 2005) e as empresas construtoras dos empreendimentos de Jirau e Santo Antônio, dentro do escopo do Programa Básico Ambiental: Programa de Monitoramento e Conservação da Ictiofauna (PBA, 2009 a 2013). PBAs tem como objetivo principal obter informações necessárias para subsidiar medidas de monitoramento para o manejo da ictiofauna local. A partir de minha experiência, pelo envolvimento nos estudos por mais de sete anos nas diferentes fases de implantação dos empreendimentos, desenvolvi meu projeto de doutoramento partindo do conhecimento de que "*O barramento de rios para formação de reservatórios podem alteram profundamente o ambiente e, consequentemente, a composição e a estrutura das assembleias de peixes*". Nesse sentido, a hipótese científica a ser testada no presente estudo é H_0*: o barramento do rio Madeira não altera a ictiofauna*. Nossa predição é de um modelo de reservatório semilêntico que mantém fluxo de água contínuo semelhante a algum período sazonal do estado natural do rio, especialmente na calha principal resultaria em um amortecimento dos impactos observados em reservatórios clássicos, com grandes áreas alagadas. Ainda, a ictiofauna responderia de maneira distinta ao impacto em função do seu habitat principal, onde peixes associados ao fundo do canal do rio teriam maior chance de manutenção de seu padrão natural quando comparado as espécies altamente dependentes da planície alagável.

A proposta central dessa tese é caracterizar padrões e processos da ictiofauna em condições pré e pós a construção das hidrelétricas no rio Madeira. O primeiro capítulo traz a Relação Peso-Comprimento de espécies associadas as planícies de inundação do rio Madeira, muitas com importância para a pesca na região. O trabalho foi publicado como contribuição técnica na revista Journal of Applied Ichthyology, apresentando a relação peso-comprimento de 112 espécies do rio Madeira. Entre as espécies de grande importância na pesca da região, estão os grandes bagres migradores da família Pimelodidae. Esses grandes bagres possuem ciclo de vida longo e grande importância na pesca comercial em toda Amazônia (ex. dourada, *Brachyplatystoma rousseauxii*). Estas espécies utilizam toda a extensão da bacia amazônica para completar seu ciclo de vida, com reprodução no sopé Andino e deriva dos ovos e larvas sentido estuário (Barthem *et al.*, 1991; Barthem e Goulding, 1997; García Vásquez *et al.*, 2009; Duponchelle *et al.*, 2016). A construção em cascata de hidrelétricas no rio Madeira poderá comprometer seriamente a passagem desses ovos e larvas sentido estuário, prejudicando as futuras gerações. Especialmente por haver evidências de comportamento *homing* para uma das espécies do gênero, *B. rousseauxii* (Duponchelle *et al.*, 2016). Desta forma, avaliar e registrar o padrão de distribuição dessas espécies no canal do rio Madeira antes dos impactos das UHEs tornou-se primordial para comparações e avaliações futuras acerca dos impactos sofridos por essas populações. O segundo capítulo apresenta um estudo de caso referente a ocorrência de jovens dessas espécies de grandes bagres migradores, reconhecidas como as espécies mais afetadas pelos empreendimentos do Madeira de acordo com o Programa Básico Ambiental, Leme (2005). O artigo analisa os padrões espaço-temporal da distribuição desses jovens no antigo trecho de corredeiras do rio Madeira, pretéritos aos impactos ocorridos com a instalação das UHEs. Visando o entendimento dos impactos promovidos pelas UHEs do rio Madeira, o terceiro capítulo traz informações da ictiofauna nas duas condições estudadas, pré e pós barramento. Neste capítulo é testada a hipótese científica da tese, sendo ela H_0*: o barramento do rio Madeira não altera a ictiofauna*.

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CAPÍTULO 1

Length-weight relationships of fish from Madeira River, Brazilian Amazon, before the construction of hydropower plants.

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Technical contribution

Length–weight relationships of fish from Madeira River, Brazilian Amazon, before the construction of hydropower plants

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Summary

Length–weight relationships (LWRs) are presented for 112 freshwater fish species representing 23 families and five orders captured in the Madeira River, the largest white-water river tributary of the Amazon River. The allometry coefficient (b) of the LWR ($Wt = aSL^b$) ranged from 2.446 to 3.856 with a median value of 3.102. Eight new LWR records are presented for Amazonian species as information for FishBase. LWRs in the present study provide historical data on *a* and *b* coefficients prior to the damming of the Madeira River in November 2011, allowing comparison estimates of predicted future population parameters as influenced by human intervention.

Introduction

Length–weight relationships (LWRs) usually translate a common relationship between variables among fish populations as a useful tool to convert length to weight and vice versa (Merella et al., 1997). In fisheries biology, length–weight relationships are useful in determining weight and biomass when only length measurements are available, for indications of condition, and to allow comparisons of species growth between different regions (Koutrakis and Tsikliras, 2002). In some cases, it is easier to take measurements of weight rather than length; for example, in cephalopod species (Bello, 1991). However, in most fish and fisheries, weighing each organism individually under field conditions is very difficult; here, weight data can be converted to length by using the LWR (Dias et al., 2013).

The relationship between two variables, as observed in linear regressions, may be one of functional dependence of one on the other. That is, the magnitude of one of the variables (the dependent variable) is assumed to be determined by – i.e. is a function of – the magnitude of the second variable (the dependent variable), although the reverse situation is often not true. The slope of the regression line, the *b* value, expresses quantitatively the straight-line dependence of Y on X in the sample and the degree of dependence of Y on X (i.e.b#0) does not necessarily mean that there is dependence in the population (i.e.beta#0) (Zar, 1998).

An historical review shows that the intra-specific variance of the LWR may be quite large (from YOY to very senior specimens of a population) and that users should follow certain recommendations when using this relationship (Froese, 2006). Among the problems that may contribute to an increase in the LWR variability, Froese (2006) noted: a narrow range of body lengths in the sample; use of non-random samples; and use of one specific size gear for selection of specimens. Genera (sexes) are a category that can potentially result in LWR variability in the calculation: in commercial fisheries it is not possible in most species to distinguish females from males prior to their sale in the marketplace. Thus, despite these recommendations, an application of the LWR for adult fish populations (excluding YOY and immature specimens) should best fit the LWR to contribute to fisheries evaluation. Since LWRs may vary geographically (Sparre et al., 1989) it is often practical to make use of the local values and highly recommended to use the LWR relationships with data collected in the same area and close to the time of the study in order to minimize any bias in the weight estimation (Kimmerer et al., 2005).

This paper is a contribution that complements the current information on the biological parameters of commercial fish in the Madeira River Basin and represents a more complete list for this type of data on fish (commercial and non-commercial species), at least on the Brazilian side of the basin. Parameters of the length–weight relationships are reported for 112 freshwater fish species collected during four years of study and prior to the construction of two large power plants: Santo Antonio and Jirau.

Growth rates and length–weight relationships have been reported for six species in the Madeira River Basin on the Bolivian side: *Prochilodus nigricans* (Loubens and Panfili, 1995), *Colossoma macropomum* (Loubens and Panfili, 1997), *Pseudoplatystoma fasciatum* (*P. punctifer*, García-Dávila et al., 2013) and *P. tigrinum* (Loubens and Panfili, 2000), *Piaractus brachypomus* (Loubens and Panfili, 2001), *Plagioscion*

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squamosissimus (Loubens, 2003), *Pellona castelnaeana* (Le Guennec and Loubens, 2004); and on the Brazilian side: *Pellona castelnaeana* (Ikeziri et al., 2008), *Roestes molossus* (Torrente-Vilara et al., 2008), *Brachyplatystoma platynemum* and *Pinirampus pirinampu* (Sant'Anna et al., 2014), however a complete list of LWRs for most of the Madeira River freshwater fish has not been reported previously.

Material and methods

Study area

The Madeira River is a major waterway in South America (Albert et al., 2011) approximately 3250 km (2020 miles) in length. Environmental information from the Madeira River is available in Queiroz et al. (2013). Recently, the Santo Antônio and Caldeirão do Inferno Falls gave way to two large hydroelectric dams (Santo Antônio and Jirau power plants, respectively), and flooding permanently the Teotônio and Jirau waterfalls (Cella-Ribeiro et al., 2013).

Fish sampling

Fish fauna was sampled on 23 occasions, including the flood and dry seasons from 2008 to 2011 before the closing of the Santo Antônio and Jirau dams. Sampling sites were established at the mouth of the eight main tributaries and two varzea lakes of Madeira River in the Brazilian territory. A set of 13 gill nets (mesh sizes from 30 to 200 mm between opposite knots; total catch area = 480 m^2) was exposed over a 24-h period, with a specimen collection every 4 h. Fish specimens were maintained in insulated iceboxes and transported to the laboratory at the Universidade Federal de Rondônia in Porto Velho, Rondônia state, Brazil for measuring (SL, nearest 0.1 cm standard length and Wt, 0.01 gram precision). The fish specimens were later identified to species level, and an in-depth taxonomic revision of each species by family was performed by specialists (Queiroz et al., 2013). As gillnets are not efficient in catching some species of the genus *Brachyplatystoma* and *Pseudoplatystoma*, data from fisheries were used for those species obtained at the fish market and artisanal fisheries in the cities of Humaitá, Porto Velho, Nova Mamoré, Guajará-Mirim, Jaci-Paraná and Abunã, and the villages of Calama, São Carlos, Nazaré, São Sebastião and Cachoeira do Teotônio. In the specific case of *Brachyplatystoma rousseauxii*, specimens collected with gill nets between 2012 and 2013 were added in order to find a better coefficient of determination r^2 for these species. All data were collected under the Santo Antonio Energia Fish Conservation Program.

Data analysis

All species analyzed in this work were collected by gillnets in the Madeira River (Queiroz et al., 2013). From 112 species, SL and Wt pairs were plotted to identify and exclude possible outliers, represented by minor errors during laboratorial proceedings. The LWR was calculated using the power regression $W = aL^b$ (Haimovici and Velasco, 2000a,b), where a is the intercept and b the slope, W the weight and L the length. The degree of association between W and L was measured through the coefficient of determination (r^2). The regression coefficients 'a' and 'b' obtained for each species were compared to those published in FishBase that were obtained by Bayesian Hierarchical Approach (BHA) for LWRs estimated for practically all known 32000 species of fish (Froese et al., 2014). We tested for differences between our estimates for Madeira River fish and the values available in FishBase using a paired *t*-test. Our hypothesis is that if the differences are random we would find estimates of 'a' and 'b' above or below those in FishBase and the *t*-test would be not significant. If the differences were systematic, the paired *t*-test would otherwise give significant p values. Thus, consistent differences between the expected (in FishBase) and observed values would suggest important differences for the Madeira river fish.

To evaluate the species with new records of maximum lengths the references used were: Loubens and Panfili (1995, 2000, 2001, 2000), Loubens and Panfili (2001), Le Guennec and Loubens (2004), Ikeziri et al. (2008), Torrente-Vilara et al. (2008), García Vásquez et al. (2009), Giarrizzo et al. (2011, 2015), Silva et al. (2011), Vegh et al. (2014), FishBase database (Froese and Pauly, 2015).

Results

Overall, 28 559 specimens were analyzed, and the length–weight relationship of 112 fish species representing 23 families and five orders are presented in Table 1. The most diverse families were Serrasalminae (16 species), Pimelodidae (11 species) and Curimatidae (10 species), followed by Auchenipteridae (nine species), Characidae and Cichlidae (eight species each), Anostomidae (seven species), Doradidae, Hemiodontidae and Loricariidae (six species each), Cynodontidae (five species), Acestrorhynchidae (four species), Pristigasteridae and Prochilodontidae (three species each), Alestidae, Ctenoluciidae and Engraulidae (two species each). The families Achiridae, Cetopsidae, Erythrinidae and Sciaenidae were represented by only one species. All regressions were significant for all species ($P < 0.001$), with the coefficient of determination r^2 ranging from 0.95 to 0.99.

The allometry coefficient (b) of the LWR ($Wt = aSL^b$) ranged from 2.446 for *Squaliforma* cf. *emarginata* to 3.856 for *Anodus orinocensis*, with a mean value of 3.103. From the 112 species evaluated, eight are new to FishBase (unpublished LWR), and presented here: *Acestrorhynchus falcirostris* (Cuvier, 1819), *Brycon melanopterus* (Cope, 1872), *Boulengerella cuvieri* (Agassiz, 1829), *Hydrolycus scomberoides* (Cuvier, 1816), H. sp. 'rabo de fogo' (undescribed species; Queiroz et al., 2013), *Oxydoras niger* (Valenciennes, 1821), *Brachyplatystoma filamentosum* (Lichtenstein, 1819) and *Pseudoplatystoma punctifer* (Castelnau, 1855). These values are within the expected range of 2.0–4.0, as suggested by Le Cren (1951). However, of the 104 species remaining, some 41 revealed 'a' and/or 'b' values beyond the FishBase range prediction (Froese et al., 2014): 28 species for the linear coefficient (a) and 24 for the angular (b) coefficient, respectively (Table 1). The paired *t*-test confirmed linear coefficient 'a' for Madeira River species smaller than BHA means from FishBase for

Table 1
Descriptive statistics and length–weight relationship parameters for 112 fish species, Madeira River, Amazon

Order	Family	Species	N	SL (cm)		Wt (g)		*a*	*b*	r^2
				Min	Max	Min	Max			
Characiformes	Acestrorhynchidae	*Acestrorhynchus* cf. *pantaneiro* Menezes, 1992	113	12.30	26.50	24.83	288.09	0.00818 (0.00694–0.00962)	3.209 (3.152–3.265)	0.97
		Acestrorhynchus falcirostris (Cuvier, 1819)	735	10.00	45.00	19.40	1000.84	0.00547 (0.00502–0.00594)	3.172 (3.146–3.198)	0.95
		Acestrorhynchus heterolepis (Cope, 1878)	564	13.00	40.80	33.51	960.12	*0.00644* (*0.00586–0.00706*)	3.184 (3.154–3.213)	0.95
		Acestrorhynchus microlepis (Schomburgk, 1841)	928	7.90	22.00	4.90	141.80	0.00724 (0.00679–0.00772)	3.177 (3.153–3.200)	0.95
	Alestidae	*Chalceus epakros* Zanata & Toledo-Piza, 2004	33	9.50	15.10	16.86	72.37	0.01710 (0.01313–0.02226)	3.072 (2.961–3.182)	0.96
		Chalceus guaporensis Zanata & Toledo-Piza, 2004	187	9.00	**17.40**	12.82	122.00	0.01001 (0.00893–0.01120)	*3.302* (*3.258–3.346*)	0.97
	Anostomidae	*Laemolyta proxima* (Garman, 1890)	138	9.20	28.50	14.65	415.00	0.01353 (0.01186–0.01541)	3.126 (3.080–3.172)	0.97
		Laemolyta taeniata (Kner, 1859)	113	10.00	24.50	19.00	222.10	0.01174 (0.01006–0.01368)	3.119 (3.063–3.174	0.97
		Leporinus fasciatus (Bloch, 1794)	88	11.00	30.50	24.28	698.00	0.01215 (0.01033–0.01429)	3.172 (3.118–3.226)	0.98
		Leporinus friderici (Bloch, 1794)	170	8.60	33.00	13.01	1053.30	0.01722 (0.01609–0.01842)	3.119 (3.094–3.143)	0.99
		Rhytiodus argenteofuscus Kner, 1859	91	12.20	**32.00**	25.21	537.75	0.00510 (0.00402–0.00645)	3.296 (3.220–3.372)	0.95
		Rhytiodus microlepis Kner, 1859	36	12.20	33.20	21.73	460.00	*0.01286* (*0.00893–0.01850*)	2.998 (2.887–3.107)	0.95
		Schizodon fasciatus Spix & Agassiz, 1829	439	9.40	31.90	14.73	846.62	0.02231 (0.02084–0.02387)	2.925 (2.902–2.947)	0.97
	Characidae	*Brycon amazonicus* (Spix & Agassiz, 1829)	43	11.80	40.60	37.78	1890.99	0.02887 (0.02194–0.03797)	2.944 (2.859–3.029)	0.96
		Brycon melanopterus (Cope, 1872)	39	11.00	**38.00**	34.42	1332.80	0.01744 (0.01287–0.02362)	3.098 (3.000–3.195)	0.96
		Bryconops alburnoides Kner, 1858	97	7.00	**15.00**	4.00	48.35	0.00853 (0.00711–0.01021)	3.199 (3.124–3.272)	0.95
		Roeboides affinis (Günther,1868)	29	6.30	10.80	4.30	22.00	0.01647 (0.01246–0.02175)	3.035 (2.908–3.162)	0.95
		Roeboides myersi Gill, 1870	118	7.00	16.70	5.40	124.14	0.01150 (0.01004–0.01316)	3.245 (3.190–3.299)	0.97
		Triportheus albus Cope, 1872	394	9.30	21.60	11.98	154.00	0.01709 (0.01569–0.01861)	2.947 (2.914–2.980)	0.95
		Triportheus angulatus Cope, 1872	1282	6.30	**23.50**	6.32	211.00	0.02603 (0.02482–0.02728)	2.897 (2.878–2.915)	0.95
		Triportheus auritus (Valenciennes, in Cuvier & Valenciennes, 1850)	100	7.80	**25.50**	12.06	277.24	*0.03555* (*0.02956–0.04275*)	*2.666* (*2.604–2.727*)	0.95
	Ctenoluciidae	*Boulengerella cuvieri* (Agassiz, 1829)	50	20.20	44.40	51.45	906.53	0.00078 (0.00057–0.00107)	3.674 (3.575–3.771)	0.97
		Boulengerella maculata (Valenciennes, 1850)	23	18.80	29.90	42.00	228.50	0.00109 (0.00070–0.00170)	*3.578* (*3.439–3.715*)	0.97
	Curimatidae	*Curimata roseni* Vari, 1989	141	8.50	**19.50**	14.76	209.70	0.02934 (0.02652–0.03245)	2.995 (2.957–3.032)	0.98
		Curimata vittata (Kner, 1858)	82	8.10	**19.60**	12.18	231.64	0.02558 (0.02133–0.03067)	3.018 (2.951–3.085)	0.96
		Curimatella alburna (Müller & Troschel, 1844)	335	5.80	**18.80**	8.20	220.33	0.03230 (0.03035–0.03437)	2.997 (2.970–3.023)	0.97
		Curimatella meyeri (Steindachner, 1882)	45	8.20	**17.20**	16.00	152.50	0.01902 (0.01474–0.02452)	3.131 (3.034–3.227)	0.96
		Cyphocharax notatus (Steindachner, 1908)	75	6.90	14.70	9.80	72.09	0.03955 (0.03445–0.04540)	*2.813* (*2.754–2.870*)	0.97
		Potamorhina altamazonica (Cope, 1878)	1032	7.20	23.50	9.09	330.40	0.03409 (0.03241–0.03584)	2.869 (2.851–2.886)	0.96
		Potamorhina latior (Spix & Agassiz, 1829)	1807	6.50	**24.00**	7.77	271.00	0.02237 (0.02149–0.02328)	2.929 (2.915–2.943)	0.96
		Psectrogaster amazonica Eigenmann & Eigenmann, 1889	375	7.20	17.30	10.47	174.53	0.02615 (0.02413–0.02833)	3.086 (3.054–3.117)	0.96
		Psectrogaster rutiloides (Kner, 1858)	794	6.40	17.80	8.20	147.62	0.03724 (0.03534–0.03923)	2.874 (2.853–2.895)	0.96
		Steindachnerina bimaculata (Steindachner, 1876)	105	7.50	17.30	12.16	150.14	0.01910 (0.01616–0.02257)	3.130 (3.064–3.196)	0.96
	Cynodontidae	*Cynodon gibbus* Agassiz, in Spix & Agassiz, 1829	193	13.20	**32.20**	23.80	487.66	*0.00582* (*0.00492–0.00687*)	3.241 (3.188–3.293)	0.95
		Hydrolycus armatus (Schomburgk, 1841)	97	14.20	75.00	42.14	8500.00	*0.00998* (*0.00869–0.01146*)	3.109 (3.069–3.149)	0.98
		Hydrolycus scomberoides (Cuvier, 1816)	212	12.00	46.70	14.96	1670.00	0.00599 (0.00521–0.00687)	3.284 (3.240–3.328)	0.96
		Rhaphiodon vulpinus Agassiz, in Spix & Agassiz, 1829	600	20.50	55.80	51.22	1368.24	0.00387 (0.00350–0.00427)	3.146 (3.118–3.174)	0.96
		Roestes molossus (Kner, 1858)	155	10.60	**21.60**	25.48	209.61	0.01564 (0.01363–0.01792)	3.117 (3.067–3.166)	0.96

Table 1
(Continued)

Order	Family	Species	N	SL (cm)		Wt (g)		*a*	*b*	r^2
				Min	Max	Min	Max			
Characiformes	Erythrinidae	*Hoplias malabaricus* (Bloch, 1794)	314	10.90	40.30	21.50	1351.41	0.01157 (0.01072–0.01247)	3.164 (3.140–3.186)	0.98
	Hemiodontidae	*Anodus elongatus* Agassiz, 1829	335	12.60	26.00	21.85	278.90	*0.00161 (0.00140–0.00185)*	*3.672 (3.625–3.717)*	0.95
		Anodus orinocensis (Steindachner, 1887)	104	14.10	26.70	31.00	319.31	*0.00097 (0.00077–0.00119)*	*3.856 (3.785–3.926)*	0.97
		Hemiodus immaculatus Kner, 1858	98	7.20	22.00	8.01	182.44	0.01873 (0.01575–0.02227)	2.957 (2.895–3.017)	0.96
		Hemiodus microlepis Kner, 1858	321	9.00	21.20	13.70	222.00	0.02318 (0.02127–0.02524)	2.958 (2.925–2.990)	0.96
		Hemiodus unimaculatus (Bloch, 1794)	508	8.50	22.00	13.34	259.00	0.01162 (0.01096–0.01230)	*3.232 (3.210–3.253)*	0.98
		Hemiodus sp. "rabo de fogo"	38	12.00	20.00	22.00	121.00	0.00282 (0.00196–0.00403)	3.543 (3.417–3.668)	0.96
	Prochilodontidae	*Prochilodus nigricans* Agassiz, 1829	639	7.20	38.20	9.84	1175.09	*0.02972 (0.02753–0.03207)*	2.933 (2.909–2.957)	0.96
		Semaprochilodus insignis (Jardine, 1841)	180	10.00	27.50	22.30	560.08	0.01573 (0.01353–0.01827)	3.193 (3.143–3.242)	0.96
		Semaprochilodus taeniurus (Valenciennes, 1817)	48	9.00	27.20	18.00	605.50	0.02953 (0.02275–0.03831)	2.886 (2.795–2.976)	0.95
	Serrasalmidae	*Metynnis guaporensis* Eigenmann, 1915	146	4.30	**16.00**	4.85	180.10	*0.06475 (0.06040–0.06939)*	2.881 (2.851–2.910)	0.98
		Metynnis lippincottianus (Cope, 1870)	45	8.20	**16.40**	22.25	205.37	*0.04470 (0.03462–0.05770)*	3.039 (2.937–3.140)	0.95
		Metynnis luna Cope, 1878	57	4.50	**12.90**	3.92	102.00	*0.04406 (0.03742–0.05187)*	3.091 (3.015–3.165)	0.97
		Myleus setiger Müller & Troschel, 1844	37	9.70	23.00	30.20	670.00	0.02049 (0.01581–0.02653)	*3.328 (3.231–3.424)*	0.97
		Myloplus asterias (Müller & Troschel, 1844)	39	6.60	19.10	13.78	360.40	*0.04465 (0.03687–0.05407)*	3.034 (2.960–3.108)	0.98
		Mylossoma aureum (Agassiz, 1829)	136	8.50	17.70	20.30	230.30	0.02266 (0.01985–0.02585)	3.183 (3.131–3.235)	0.97
		Mylossoma duriventre (Cuvier, 1818)	1298	5.40	24.90	6.60	736.00	0.02920 (0.02781–0.03065)	3.118 (3.099–3.137)	0.96
		Piaractus brachypomus (Cuvier, 1818)	51	9.00	31.50	22.47	1171.43	0.02891 (0.02345–0.03563)	3.116 (3.038–3.193)	0.97
		Pygocentrus nattereri Kner, 1858	822	7.00	21.40	13.77	543.41	0.03546 (0.03350–0.03753)	3.125 (3.103–3.146)	0.96
		Serrasalmus compressus Jégu, Leão & Santos, 1991	50	8.10	18.80	13.38	259.70	0.01639 (0.01325–0.02028)	3.245 (3.163–3.327)	0.97
		Serrasalmus eigenmanni Norman, 1929	192	5.20	19.60	3.27	394.00	0.01061 (0.00955–0.01179)	*3.458 (3.414–3.502)*	0.97
		Serrasalmus elongatus Kner, 1858	227	7.50	22.90	7.39	311.00	0.00890 (0.00831–0.00951)	*3.328 (3.302–3.353)*	0.99
		Serrasalmus hollandi Eigenmann, 1915	157	5.20	17.20	3.43	154.32	0.01829 (0.01681–0.01990)	3.247 (3.210–3.283)	0.98
		Serrasalmus maculatus Kner, 1858	28	6.50	18.80	8.90	288.17	0.02444 (0.02031–0.02939)	3.172 (3.093–3.249)	0.98
		Serrasalmus rhombeus (Linnaeus, 1766)	693	5.60	31.00	4.67	1277.00	0.01388 (0.01315–0.01465)	*3.314 (3.293–3.335)*	0.97
		Serrasalmus spilopleura Kner, 1858	129	5.20	19.30	5.01	277.41	0.02784 (0.02390–0.03242)	3.131 (3.068–3.194)	0.95
Clupeiformes	Engraulidae	*Jurengraulis juruensis* (Boulenger, 1898)	50	10.00	**17.90**	9.97	89.00	0.00327 (0.00256–0.00416)	*3.463 (3.368–3.557)*	0.97
		Lycengraulis batesii (Günther, 1868)	54	12.00	18.20	18.80	74.23	0.00630 (0.00495–0.00801)	3.222 (3.132–3.311)	0.96
	Pristigasteridae	*Ilisha amazonica* (Miranda Ribeiro, 1920)	32	13.80	19.70	31.90	98.39	0.00769 (0.00584–0.01011)	3.183 (3.084–3.282)	0.97
		Pellona castelnaeana (Valenciennes, 1847)	280	8.70	60.90	8.47	4000.30	0.01038 (0.00962–0.01119)	3.120 (3.098–3.142)	0.98
		Pellona flavipinnis (Valenciennes, 1836)	275	9.20	44.30	6.81	1347.00	0.01696 (0.01512–0.01901)	2.972 (2.934–3.008)	0.96
Perciformes	Cichlidae	*Astronotus crassipinnis* (Heckel, 1840)	39	12.00	21.00	75.00	484.54	*0.06911 (0.05542–0.08616)*	2.896 (2.817–2.974)	0.97
		Biotodoma cupido (Heckel, 1840)	62	5.50	**11.70**	5.42	55.32	0.01920 (0.01610–0.02288)	*3.293 (3.211–3.375)*	0.96
		Chaetobranchus flavescens Heckel, 1840	41	6.20	20.70	13.99	316.00	*0.18832 (0.16086–0.22047)*	*2.483 (2.421–2.544)*	0.98
		Cichla pleiozona Kullander & Ferreira, 2006	217	8.50	44.20	12.58	2400.00	0.01506 (0.01376–0.01647)	3.160 (3.131–3.189)	0.98
		Geophagus megasema Heckel, 1840	121	7.50	18.90	15.40	272.18	*0.04403 (0.03897–0.04974)*	2.908 (2.862–2.952)	0.97
		Heros spurius Heckel, 1840	77	6.20	15.00	13.32	192.30	*0.07634 (0.06393–0.09115)*	2.915 (2.843–2.987)	0.95
		Mesonauta festivus (Heckel, 1840)	60	3.50	12.10	2.00	97.41	0.03027 (0.02594–0.03532)	3.255 (3.184–3.325)	0.97
		Satanoperca jurupari (Heckel, 1840)	238	6.50	18.40	9.12	206.86	*0.03669 (0.03278–0.04106)*	2.989 (2.946–3.030)	0.95
Perciformes	Sciaenidae	*Plagioscion squamosissimus* (Heckel, 1840)	178	12.90	42.50	40.81	1576.34	*0.03217 (0.02766–0.03740)*	*2.855 (2.809–2.901)*	0.95
Pleuronectiformes	Achiridae	*Hypoclinemus mentalis* (Günther, 1862)	72	9.00	**21.60**	24.28	388.65	0.02807 (0.02448–0.03219)	3.099 (3.047–3.150)	0.98

Table 1
(Continued)

Order	Family	Species	N	SL (cm)		Wt (g)		*a*	*b*	r^2
				Min	Max	Min	Max			
Siluriformes	Auchenipteridae	*Ageneiosus atronasus* Eigenmann & Eigenmann, 1888	50	8.70	14.90	10.93	65.50	0.00861 (0.00682–0.01086)	*3.282* (*3.185–3.377*)	0.96
		Ageneiosus inermis (Linnaeus, 1766)	206	12.70	45.00	31.54	1532.42	*0.01779* (*0.01530–0.02067*)	*2.970* (*2.925–3.015*)	0.95
		Ageneiosus ucayalensis Castelnau, 1855	92	13.10	32.50	21.02	276.00	*0.01922* (*0.01657–0.02229*)	*2.755* (*2.707–2.803*)	0.97
		Auchenipterichthys thoracatus (Kner, 1857)	2706	5.10	**13.80**	2.10	62.39	0.02138 (0.02073–0.02205)	3.066 (3.052–3.079)	0.95
		Auchenipterus ambyiacus Fowler, 1915	246	10.50	21.30	17.64	130.40	0.01332 (0.01194–0.01485)	2.967 (2.928–3.006)	0.96
		Auchenipterus nuchalis (Spix & Agassiz, 1829)	110	10.20	**22.70**	16.03	146.30	0.00852 (0.00730–0.00994)	3.136 (3.081–3.191)	0.97
		Centromochlus heckelii (De Filippi, 1853)	185	5.30	**13.90**	2.76	44.87	*0.01822* (*0.01616–0.02054*)	3.024 (2.972–3.076)	0.95
		Parauchenipterus galeatus (Linnaeus, 1766)	106	6.80	**23.70**	12.88	464.70	0.02506 (0.02180–0.02879)	3.071 (3.018–3.123)	0.97
		Tatia aulopygia (Kner, 1857)	27	7.10	15.30	7.92	95.00	0.01379 (0.01027–0.01850)	*3.256* (*3.122–3.388*)	0.96
	Cetopsidae	*Cetopsis coecutiens* (Lichtenstein, 1819)	62	12.80	23.80	41.49	280.39	0.01874 (0.01493–0.02351)	3.024 (2.946–3.101)	0.96
	Doradidae	*Hemidoras stenopeltis* (Kner, 1855)	40	8.70	19.40	8.00	86.52	0.01095 (0.00860–0.01393)	3.088 (2.988–3.187)	0.96
		Nemadoras humeralis (Kner, 1855)	98	7.00	13.50	6.60	55.90	0.01908 (0.01608–0.02264)	3.061 (2.990–3.131)	0.95
		Opsodoras boulengeri (Steindachner, 1915)	89	6.20	19.30	4.90	129.10	0.02198 (0.01934–0.02497)	2.858 (2.810–2.906)	0.97
		Oxydoras niger (Valenciennes, 1821)	73	14.50	57.00	54.71	2927.87	0.02010 (0.01640–0.02463)	2.951 (2.891–3.010)	0.97
		Pterodoras granulosus (Valenciennes, 1821)	21	14.60	43.00	103.90	2147.00	*0.06583* (*0.04567–0.09486*)	*2.739* (*2.618–2.860*)	0.96
		Trachydoras paraguayensis (Eigenmann & Ward, 1907)	82	6.00	13.80	5.60	67.97	0.03779 (0.03246–0.04398)	2.869 (2.798–2.938)	0.95
	Hypoptopomatinae	*Hypoptopoma incognitum* Aquino & Schaefer, 2010	202	6.20	**10.80**	4.35	26.66	*0.01599* (*0.01430–0.01789*)	3.115 (3.065–3.165)	0.95
	Hypostominae	*Aphanotorulus unicolor* (Steindachner, 1908)	24	10.30	13.50	24.01	55.59	0.02403 (0.01898–0.03042)	2.973 (2.877–3.069)	0.97
		Hypostomus pyrineusi (Miranda Ribeiro, 1920)	28	11.10	21.20	37.07	314.93	*0.03713* (*0.02751–0.05010*)	2.928 (2.823–3.032)	0.97
		Pterygoplichthys lituratus (Kner, 1854)	144	12.00	36.50	19.03	1204.36	0.02064 (0.01730–0.02462)	3.047 (2.991–3.101)	0.95
		Squaliforma cf. *emarginata* (Valenciennes, 1840)	36	10.00	**36.50**	22.19	550.66	*0.09131* (*0.07532–0.11067*)	*2.446* (*2.384–2.508*)	0.98
	Loricariidae	*Loricaria cataphracta* Linnaeus, 1758	35	13.60	26.60	12.70	117.90	0.00578 (0.00427–0.00780)	2.998 (2.897–3.097)	0.96
	Pimelodidae	*Brachyplatystoma filamentosum* (Lichtenstein, 1819)	517	49.00	196.00	2000.00	100000.00	0.02169 (0.01902–0.02472)	2.914 (2.885–2.943)	0.95
		Brachyplatystoma rousseauxii (Castelnau, 1855)	1102	8.70	147.00	5.23	60000.00	0.01233 (0.01143–0.01329)	3.032 (3.014–3.049)	0.96
		Brachyplatystoma vaillantii (Valenciennes, 1840)	444	29.00	69.00	300.00	6000.00	*0.00121* (*0.00104–0.00139*)	*3.629* (*3.590–3.667*)	0.95
		Calophysus macropterus (Lichtenstein, 1819)	120	12.20	38.60	21.66	689.80	0.01340 (0.01184–0.01516)	3.000 (2.962–3.037)	0.98
		Hypophthalmus edentatus Spix & Agassiz, 1829	85	17.20	39.00	51.22	884.86	0.00611 (0.00489–0.00762)	3.167 (3.101–3.232)	0.96
		Hypophthalmus marginatus Valenciennes, 1840	135	25.20	47.00	99.73	1014.06	0.00261 (0.00209–0.00326)	*3.328* (*3.263–3.393*)	0.95
		Pimelodus blochii Valenciennes, 1840	552	5.80	21.20	3.81	203.93	0.01945 (0.01800–0.02101)	2.983 (2.954–3.012)	0.95
		Pseudoplatystoma punctifer (Castelnau, 1855)	283	19.60	97.00	81.58	9500.00	0.00997 (0.00858–0.01157)	3.029 (2.992–3.065)	0.96
		Sorubim elongatus Littmann, Burr, Schmidt & Isern, 2001	104	16.20	**32.20**	28.39	260.00	0.00409 (0.00328–0.00509)	3.135 (3.065–3.204)	0.95
		Sorubim lima (Bloch & Schneider, 1801)	90	18.50	38.50	36.98	591.51	*0.00095* (*0.00072–0.00125*)	*3.636* (*3.553–3.719*)	0.95
		Sorubim maniradii Littmann, Burr, Schmidt & Isern, 2001	119	13.90	**25.60**	21.07	131.97	*0.00499* (*0.00412–0.00604*)	3.123 (3.060–3.185)	0.95

N, number of individuals; SL, standard length; Wt, total weight. Bold = new length record, MaxSL. *Italics* = values of *a* and *b* outside range reported in FishBase. r^2, Pearson r-squared for log–logregression (all relationships significant at $P < 0.0001$).

the 28 species ($t_a = 3.77$, df = 27; $P < 0.001$); a similar result was obtained for the *a* set of species ($t_a = 5.49$, df = 103, $P > 0.05$). However, the angular coefficient '*b*' observed beyond the BHA range in FishBase could not be considered consistently different from the BHA mean for the 24 species ($t_b = 1.64$, df = 23, $P < 0.05$), or for the Madeira River *b* set of species ($t_b = 1.66$, df = 103, $P > 0.05$).

Discussion

This study provides new maximum length records for 27 species. *Colossoma macropomum* is a very important commercial fish in the Amazon. However, specimens landed at the Madeira River fish market were from Bolivia and mixed in with specimens from piscicultures, precluding our analysis for this species. The best information on the *C. macropomum* LWR is in Loubens and Panfili (1997), where $SL_{max} = 82.5$ cm ($a = -4.258$, $b = 2.903$, $r^2 = 0.99$, N = 864). *Pseudoplatystoma tigrinum* with $SL_{max} = 127$ cm ($a = -5.201$, $b = 3.15$, $r^2 = 0.99$, N = 551) in Loubens and Panfili (2000) was neither captured in our samples nor seen in the fish market. The LWRs calculated for *Pinirampus pirinampu* ($a = 0.0271$, $b = 2.81$, $SL_{max} = 74$ cm) and *Brachyplatystoma platynemum* ($a = 0.0462$, $b = 2.67$, $SL_{max} = 89$ cm) were published in Sant'Anna et al. (2014) using data from our samples. Most specimens captured in the Madeira River were adults and with a few juveniles. Madeira River is known for its schools of fish that perform yearly reproductive migrations. It is possible that these fish schools were growing and feeding in the same downstream river area before migrating to Madeira River each year, explaining the absence of juveniles in our gillnets samples.

The *a* and *b* parameters of the LWRs given in FishBase arise from a compilation of estimates from different genus or species belonging to the same family having the same body shape (Froese and Pauly, 2015). Indeed, 63 of 104 species have *a* and *b* parameters in agreement with coefficient values in FishBase; the differences observed in the coefficient values were reported by Allard et al. (2015) in the literature for Amazonian fish. Here, were report direct estimates of LWRs and are confident that our means and ranges encompass all size ranges for the adult species considered. However, our study notes that the *a* values are smaller than those estimated in FishBase, whereas the *b* are consistent. This suggests that Madeira River fish have the same proportional weight increments but are thinner when compared to BHA results (e.g. Table 1). Thin individuals, most often with empty stomachs and having low visceral fat deposits, may indicate a lower quality of habitat (Torrente-Vilara et al., 2011). In fact, a remarkable characteristic regarding Madeira River fish is that a typical lowland migratory fish in the stretch of rapids in our study area was represented by adult specimens in poor condition, possibly a result of a loss in energy due to migration.

Hydropower plants will almost certainly change ecological patterns; at the same time, they will favor some species more adaptable to new environmental conditions, yet will cause disturbances in others. As such, continued observations of the LWRs in these fish species are fundamental.

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CAPÍTULO 2

Temporal and spatial distribution of young Brachyplatystoma spp. (Siluriformes: Pimelodidae) along the rapids stretch of the Madeira River (Brazil) before the construction of two hydroelectric dams.

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Temporal and spatial distribution of young *Brachyplatystoma* spp. (Siluriformes: Pimelodidae) along the rapids stretch of the Madeira River (Brazil) before the construction of two hydroelectric dams

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Monthly (April 2009 to May 2010) bottom-trawl sampling for *Brachyplatystoma* species along the rapids stretch of the Madeira River in Brazil revealed that *Brachyplatystoma rousseauxii* larvae and juveniles were present in low abundances in all areas and during all hydrological periods. The presence of larvae and juveniles throughout the hydrological cycle suggests asynchronous spawning in the headwaters of the Madeira River.

Key words: dourada catfish; hydrological cycle; natural barriers; young-of-the-year.

The giant Pimelodid catfishes of the genus *Brachyplatystoma* inhabit the main channel of large Amazonian rivers (Lundberg & Akama, 2005). These heavily exploited migratory species are among the most important commercial fishes of the Amazon Basin (Barthem & Goulding, 1997, 2007; Petrere *et al.*, 2004). *Brachyplatystoma rousseauxii* (Castelnau 1855) has an exceptional life cycle. Adults reproduce in the headwaters of

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the main turbid water rivers in the pre-Andean portion of Ecuador, Peru, Colombia and Bolivia (García-Vásquez *et al.*, 2009; Vam Damme *et al.*, 2011; Agudelo *et al.*, 2013) and the larvae and juveniles drift thousands of km downstream to the Amazon Estuary, where they are resident for two years and then start migrating back towards the headwaters (Barthem & Goulding, 1997, 2007). During this migration, the fish grow and initiate their gonadal maturation, which is completed close to the headwaters (Barthem *et al.*, 1991; Barthem & Goulding, 1997; Alonso, 2002; García-Vásquez *et al.*, 2009).

Brachyplatystoma spp. are considered heavily or overexploited in Brazil, Colombia and Peru, where they are caught by driftnet and bottom longline fishing (Petrere *et al.*, 2004; García-Vásquez *et al.*, 2009; Agudelo *et al.*, 2013). The upper Madeira Basin represents the last area of the Amazon Basin where these species are considered to still be weakly exploited (Vam Damme *et al.*, 2011). This refuge, however, is now threatened by the construction of at least two hydroelectric power plants in the Brazilian part of the Madeira River, which might disrupt natural upstream breeding movements of these catfishes and the downstream drifting of their larvae and juveniles.

The aim of this study was to assess the spatial and temporal variation in the natural abundance of larvae and juveniles of *Brachyplatystoma* spp. along the rapids stretch of the Madeira River prior to the construction of the reservoirs of the Jirau and Santo Antonio power plants.

The Madeira River begins at the confluence of the Mamoré and Beni Rivers on the border between Brazil and Bolivia. The abrupt elevation change in the transition from the Brazilian central highlands to the Amazon lowlands results in a series of rapids, where the steepest waterfalls are situated in a 300 km river stretch between the cities of Guajará-Mirim and Porto Velho in the State of Rondônia, Brazil, where this study was conducted (Fig. 1). This section of river includes 19 major rapids, two of which, the Jirau and Teotônio Waterfalls, have abrupt falls (Cella-Ribeiro *et al.*, 2013). In this region, the river channel is between 350 and 1500 m wide, 3–33 m deep and water velocity is up to $2{\cdot}5\,\mathrm{m\,s^{-1}}$ in rapids and waterfalls. Three areas were sampled during this study: (A) upstream Jirau Waterfall, (B) between the Jirau and Teotônio Waterfalls and (C) downstream of the Santo Antonio Waterfall (Fig. 1). In each area, samples were collected from five stations, resulting in a total of 15 fixed sampling stations where fish populations were sampled monthly between April 2009 and May 2010 resulting in 210 samples (three areas × five stations × 14 months). Sampling was conducted using a 3 m long × 1 m wide bottom trawl that was hauled behind a boat, as described by López-Rojas *et al.* (1984). The net had a 5 mm mesh bag that was lined with 1 mm mesh netting. A 10 min trawl haul was performed at each sampling station at depths between 5 and 25 m according to the seasonal variation in the water level of the Madeira River.

Most of the sampled *Brachyplatystoma* spp. larvae and juveniles were already dead when pulled out of the water. The few individuals collected alive were euthanized with an overdose of anaesthetic (eugenol or clove oil) and subsequently fixed in a solution of 10% formalin. Fishes were transported to the Ichthyology and Fishing laboratory of the Universidade Federal de Rondônia (UNIR) for sorting, identification and measurement (standard length, L_S, mm). The developmental stage of each individual was determined based on morphological characteristics and pigmentation, according to Nakatani *et al.* (2001) and Leite *et al.* (2007). Voucher specimens were deposited in the UFRO-I fish collection (Universidade Federal de Rondônia, Porto Velho, Brazil).

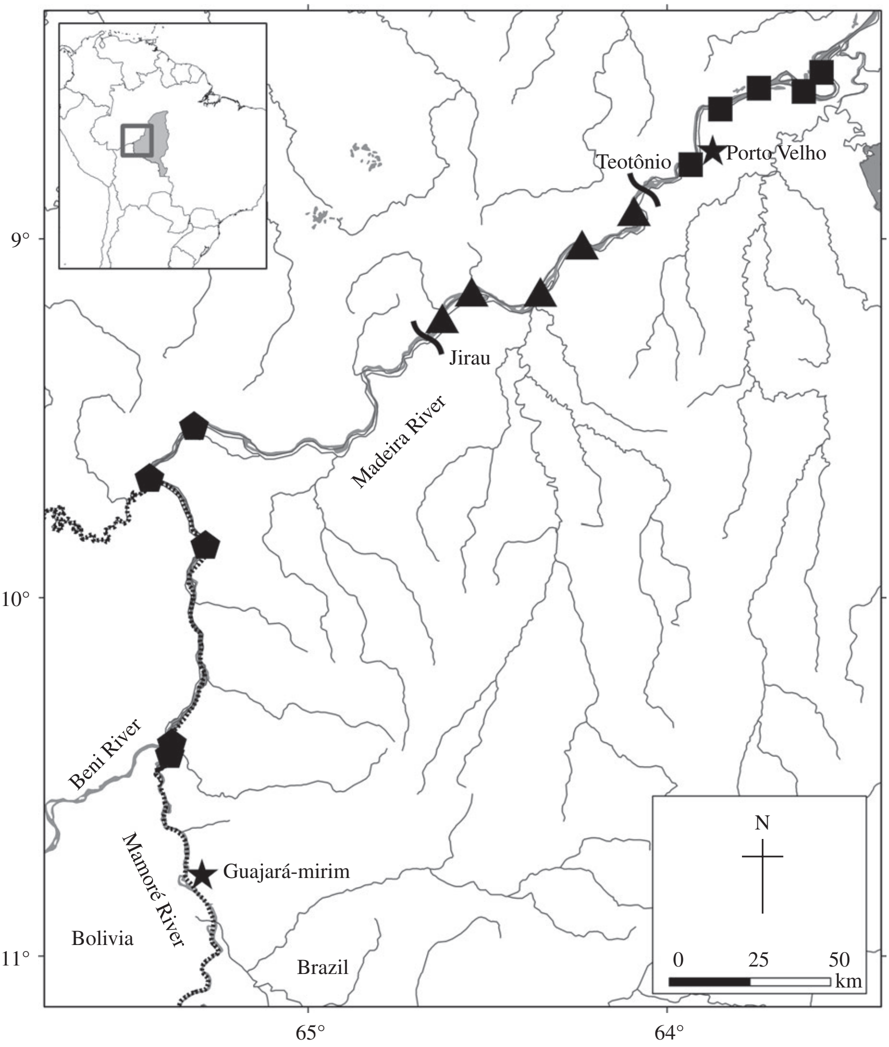

FIG. 1. Map of the study areas: A (⬟), B (▲) and C (■) (⋯, political boundaries; ⟍, waterfalls).

A total of 80 *B. rousseauxii* and 58 of other *Brachyplatystoma* species were collected. Forty four specimens (three juveniles and 41 larvae) were too small or damaged for species identification. The remaining 14 specimens were *Brachyplatystoma filamentosum* (Lichtenstein 1819) ($n = 1$), *Brachyplatystoma capapretum* Lundberg & Akama 2005 ($n = 8$) and *Brachyplatystoma platynemum* Boulenger 1898 ($n = 5$). Juveniles were more abundant than larvae for all species in all samples. No specimens (larvae or juveniles) of *B. capapretum* and *B. filamentosum* were collected upstream of the Jirau Waterfall (area A), and no larvae or juveniles of *B. platynemum* were observed in the samples taken between Jirau and Teotônio Waterfalls (area B). Below the rapids stretch (area C), almost all sampled individuals were juveniles.

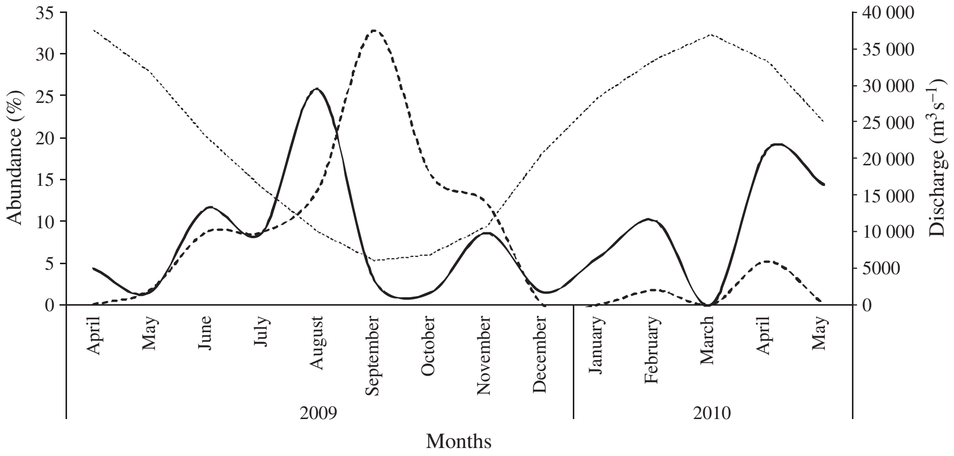

FIG. 2. Relative abundance of each *Brachyplatystoma* species per sampling month, all sampling stations pooled (- - - -, unidentified *Brachyplatystoma* spp.; ——, *Brachyplatystoma rousseauxii*; ··········, discharge $m^3 s^{-1}$).

Brachyplatystoma spp. larvae and juveniles were collected throughout the hydrological cycle, but mostly between June and November (dry season) (Fig. 2). There was, however, no difference in the abundance (ANOVA, d.f. = 2,437, $P > 0{\cdot}05$) and L_S (ANOVA, d.f. = 0,187, $P > 0{\cdot}05$) of larvae and juveniles of *B. rousseauxii* among the three sampling areas (Table I). Similarly, no significant temporal difference in the abundance of *B. rousseauxii* (grouping larvae + juveniles) was detected between the flood and dry periods (*t*-test, d.f. = 11, $P > 0{\cdot}05$; Table II).

During the 14 months of the study, the bottom-trawl sampling of the Madeira River resulted in very low overall abundances of larvae and juvenile fishes compared with other studies in the Amazon and Orinoco Basins (López-Rojas *et al.*, 1984; Cox-Fernandes *et al.*, 2004; Thomé-Souza & Chao, 2004). As the same sampling protocol and gear yielded larger abundances in these studies, the low abundances observed in the Madeira can be considered as a robust result. This is further emphasized by the fact that sampling effort was increased by doubling the duration of trawl transects and reducing mesh size in this study. Such results are consistent with the low overall fish abundance observed along the rapids stretch of the Madeira River (Torrente-Vilara *et al.*, 2011).

The increased abundance of *Brachyplatystoma* spp. (including *B. rousseauxii*) late larvae and juveniles during the autumn and dry season may suggest that these species use the periods of lower water flow of the Madeira River for the downstream drifting and migration of young individuals towards the Amazon Estuary. Downstream drifting during the low water season may constitute a strategy to minimize the chances of juveniles being diverted to the floodplains in the intermediate portions of the basin and not reaching the Amazon Estuary, as hypothesized by García-Vásquez *et al.* (2009).

Brachyplatystoma rousseauxii is a typical periodic spawner (Winemiller & Rose, 1992), characterized by its large size, long generation time, long lifespan, the production of large numbers of small eggs and a breeding cycle synchronized with the hydrological periodicity (García-Vásquez *et al.*, 2009). In tropical rivers with

TABLE I. Standard length (L_S) range (mean ± S.D.) of the specimens of *Brachyplatystoma* species captured in the three sampling areas (A, upstream from the Jirau Waterfall; B, between the Jirau and Teotônio Waterfalls; C, downstream from the Teotônio Waterfall) along the Madeira River on Brazilian territory. *n*, number of individuals in each area (values in parentheses show the abundance of larvae and of juveniles in the samples; *, one damaged; **, three damaged; ***, five damaged; ****, six damaged)

	Area A		Area B		Area C	
Species	*n*	L_S (mm)	*n*	L_S (mm)	*n*	L_S (mm)
Brachyplatystoma capapretum	2* (0:1)	17·2	5 (2:3)	11·4–20·8 (15·5 ± 3·8)	1 (1:0)	16·3
Brachyplatystoma filamentosum	0		1 (1:0)	8·7	0	
Brachyplatystoma platynemum	3 (0:3)	14·2–53·6 (40·3 ± 22·6)	1 (0:1)	19·8	1* (0:0)	
Brachyplatystoma rousseauxii	13 (3:10)	11·9–22·6 (16·7 ± 4·3)	40** (9:28)	8·7–25·1 (17·5 ± 3·6)	27 (0:27)	9·1–25·3 (17·0 ± 3·8)
Unidentified *Brachyplatystoma* spp.	6 (3:3)	7·2–13·3 (10·8 ± 2·3)	19**** (13:0)	7·8–11·1 (9·5 ± 0·8)	19*** (14:0)	8·2–12·8 (10·5 ± 1·2)
All species pooled	24 (6:17)	7·2–53·6 (18·3 ± 11·8)	66 (34:32)	7·8–25·1 (15·3 ± 4·7)	48 (21:27)	8·2–25·3 (15·0 ± 4·4)

Table II. Relative abundance of each *Brachyplatystoma* species per sampling month, all sampling stations pooled (values in parenthesis indicate the abundance of larvae and juveniles, respectively)

	2009									2010				
Species	April	May	June	July	August	September	October	November	December	January	February	March	April	May
B. capapretum		1 (1:0)	2 (2:0)			2 (1:1)	1 (0:1)						2 (0:2)	
B. filamentosum							1 (1:0)							
B. platynemum			1 (0:1)		2 (1:1)	2 (0:2)								
B. rousseauxii	3 (1:2)	1 (0:1)	8 (1:7)	6 (2:4)	18 (6:12)	2 (1:1)	1 (0:1)	6 (0:6)	1 (0:1)	4 (0:4)	7 (2:5)		13 (1:12)	10 (1:9)
Unidentified *Brachyplatystoma* spp.			2 (1:1)	5 (5:0)	6 (6:0)	15 (15:0)	7 (7:0)	7 (7:0)			1 (0:1)		1 (0:1)	

predictable seasonal flow fluctuations, however, spawning of periodic species is usually restricted to a short period (a few months) during the rising waters (Tedesco & Hugueny, 2006). According to Agudelo *et al.* (2000, 2013) and García-Vásquez *et al.* (2009), *B. rousseauxii* and its congeners have relatively long breeding seasons (>5 months) in the headwaters of the Amazon River (Colombian and Peruvian Amazon). The occurrence of larvae and juveniles of *B. rousseauxii* in the rapids stretch throughout the year suggests that *B. rousseauxii* spawn at slightly different times in the rivers that compose the headwaters of the Madeira River in Bolivia and Peru, resulting in more than one peak of late larvae and juveniles crossing the rapids stretch in the Brazilian portion of the basin. This is consistent with observations of previous studies in the upper Madeira. Maximum larval densities occur during the period of highest flow between October and December in the upper Madre de Dios (Cañas & Pine, 2011; Cañas & Waylen, 2012). In the upper Mamoré River, *B. rousseauxii* spawns between February and May (Vam Damme *et al.*, 2011), whereas in the upper Beni River females with ripe gonads were observed from October to May (F. Carvajal & F. Duponchelle, unpubl. data). Therefore, the existence of different breeding periods corroborates molecular data on the presence of distinct populations of *B. rousseauxii* in different parts of the Madeira River basin (Carvajal-Vallejos *et al.*, 2014).

Another striking result of this study is the low abundance of larvae and juveniles collected, despite the considerable sampling effort, for such highly fecund species (García-Vásquez *et al.*, 2009). It is possible that a large proportion of the larvae that hatched in the Bolivian and Peruvian portions of the basin might be retained in the extensive flooded area of the Bolivian Amazon, thus resulting in the very low abundances in the Brazilian portion of the Madeira River. This contradicts, however, the standing hypothesis that the Amazon Estuary is essential for the completion of the *B. rousseauxii* life cycle. Small specimens (<1 kg) of these species are, however, very rarely observed in this portion of the basin (Carvajal-Vallejos *et al.*, 2014) and it is therefore unlikely that a large proportion of the larvae hatched in the Upper Madeira River were finding growth areas above the rapids. An alternative explanation is that only a small fraction of the *B. rousseauxii* larvae hatched in the upper Madeira survive long enough to reach the rapids stretch of the river, and ultimately the Amazon Estuary. If this holds true, it highlights the fragility of the life cycle of this species, and how it may depend on a small number of juveniles reaching the estuary, at least from the Madeira River basin. It also further stresses the question of how the downstream movements of such a low number of juveniles will be affected by the dams in the Madeira River. The presence of the dams may also impair the upstream migration of pre-adults, further reducing the production of larvae in the headwaters. While a fish pass was built at the Santo Antonio hydroelectric power plant, its efficiency to allow the upstream movements of *Brachyplatystoma* spp. has not been tested. Monitoring the relative abundances of a long-term data of *Brachyplatystoma* juveniles is therefore essential to evaluate the long-term effects of the dams on *Brachyplatystoma* species in the Madeira River basin.

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CAPÍTULO 3

Fish community responses after built two reservoir in cascade in the Amazon.

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Temporal fish community responses of two cascade reservoirs in the Amazon

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Keywords:	Amazon, muddy-water, cascade reservoirs, damming, freshwater fish, richness, species composition

Temporal fish community responses of two cascade reservoirs in the Amazon

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ABSTRACT

The environmental consequences of large dams are numerous, including direct impacts to the biological, chemical and physical properties of rivers environments. Dams fragment the riverine ecosystem, isolating populations of species living up and downstream of the dam and cutting off migrations and other species movements. We examined responses from fish assemblages depending on floodplains areas (pelagic) and bottom main channel (benthic) three years before and three years after two reservoirs built in cascade in the largest white-water tributary of the Amazon River. Jirau and Santo Antônio were the first dams constructed in an Amazonian tributary with white water, great flow variability, and fish diversity, such as the Madeira River. Overall, there were major differences in species richness, biomass and density but no significant differences in fish diversity and evenness before and after dam construction. Differences in pelagic fish assemblage composition into both reservoirs were noticed immediately, and linked to the increasing in conductivity, deep, oxygen and temperature while channel bottom fish assemblage seems to not respond immediately to the dams' closure. Our study is especially relevant considering responses from fish assemblages to bulb turbines in the face of burgeoning new hydroelectricity development plans for rivers across Amazonia.

KEY WORDS Amazon; muddy-water; cascade reservoirs; damming; freshwater fish; richness; species composition.

Introduction

Damming has one of the most severe form of changes riverine ecosystems. Dressed in clean energy, dam construction has been increasing faster causing a tsunami of social and environmental problems in tropical rivers (Finer and Jenkins, 2012; Castello *et al.*, 2013; Lees *et al.*, 2016; Prado *et al.*, 2016). Dams fragmented nearly two-thirds of the world's largest rivers (Dynesius and Nilsson, 1994; Nilsson *et al.*, 2005) and tropical rivers are now the primary frontier of dam construction due to its free-flowing natural conditions (Vörösmarty *et al.*, 2004; Finer and Jenkins, 2012; Fearnside, 2013; 2014; Lima Junior *et al.*, 2015; Fearnside, 2016; Winemiller *et al.*, 2016).

Dams are not planned randomly, they are mostly built at waterfalls and rapids to optimize energy production by blocking turbulent sections of rivers. Once blocked, the lotic section of the river is replaced by a lentic water body, hence modifying river's flow, the transport of sediments and nutrients load from upstream to downstream areas, most of which with unknown effects at regional scale (Liermann *et al.*, 2012; Fearnside, 2013; Lees *et al.*, 2016). At basin scale, lentic sections of river are permanent barrier to dispersal of aquatic organisms in spite of the frequent construction of transposition systems (Pelicice and Agostinho, 2008; Pelicice *et al.*, 2015). Yet, dams stabilize the river flow, thus drastically changing the natural flood pulse of tropical rivers, which is the main trigger to longitudinal and lateral fish movements (Junk *et al.*, 1989; Winemiller, 1989; Cox Fernandes, 1997; Dugan *et al.*, 2010), causing long term ripple effects (Winemiller *et al.*, 2016).

The effect of dams as a permanent barrier is not limited to the basin scale. The new lentic environment creates new habitats where an increase of temperature is followed by a decrease in dissolved oxygen (Prchalová *et al.*, 2009). These new environments facilitate the establishment of opportunistic and non-native species, thus modifying alfa and beta diversity at the local scale (Agostinho *et al.*, 2008; Pelicice and Agostinho, 2009; Vitule *et al.*, 2009; Agostinho *et al.*, 2015). An heterotrophic period is locally expected during the first stage of the reservoir filling which promotes increases in fish abundance and richness (see Petrere, 1996; Agostinho *et al.*, 2008; Gao *et al.*, 2010; Agostinho *et al.*, 2015; Lima *et al.*, 2015). The increase in fish richness might be a result of permanent flood on a host of natural habitats with their specific fish fauna assemblage (beaches, riparian zone) sheltering species temporarily. Furthermore, the permanent flood inputs terrestrial organic matter resulting in an increase in productivity of all trophic levels (Agostinho *et al.*, 2008; Agostinho *et al.*, 2015).

Spatially, fish assemblages are distributed along traversal (Cox Fernandes, 1997; Stewart *et al.*, 2002) and depth gradients of river (Sheldon, 1968). The dependence of fish on floodplain habitats in large river systems is recognized (Welcomme, 1979; Junk *et al.*, 1989; Agostinho and Zalewski, 1995), and for many pelagic fish species, the floodplain forest is a key source of food (Goulding, 1980; Saint-Paul *et al.*, 2000). Fish assemblages from the deep, main channel habitats of large tropical rivers are rarely addressed in the literature. In the last two decades, new taxa of highly specialized benthic fish species have been described including blind fishes adapted to live in the bottom (Lopez-Rojas *et al.*, 1984; Mago-Leccia *et al.*, 1985; Lundberg and Py-Daniel, 1994; Lundberg *et al.*, 1996; Stewart *et al.*, 2002; Cox Fernandes *et al.*, 2004; Thomé-Souza and Chao, 2004). However, independently of addressed pelagic or benthic fish assemblage, little is known about the process regulating the transition of fish assemblage after the establishment of a dam due to the lack of ecological information prior the dam construction. As a result, such information has to be reconstructed retrospectively using often-incomplete data from museum's collections and historical records (da Fonseca *et al.*, 2002; Vitule *et al.*, 2012). In this sense, rivers with historical data pre-impoundment are valuable models for understanding how running waters are impacted by dams, especially in tropical rivers which concentrate much of freshwater biodiversity.

In this study we describe the responses of fish assemblages during the heterotrophic period of the first two reservoirs built in cascade in the largest white-water tributary of the Amazon River. We sampled sites inside the two reservoirs and downstream areas before and after the dams' closure to test the hypothesis that changes in river habitat associated with dam closure would (i) increase species richness, abundance, and diversity, (ii) increase fish assemblages dissimilarity faster in pelagic fish assemblage than benthic fish assemblage due its level of dependence of the flood pulse.

Materials and methods

Study area and sampling design

Madeira River is the largest muddy-water tributary of the Amazon River and was blocked to create two reservoirs in cascade: Santo Antônio (blocked in 2011) and Jirau (blocked in 2012). Both reservoirs were the first case of a dam constructed with bulb turbine in the Amazonia. Bulb turbines are known for having a high efficiency in energy generation (e. g., more than 90%, Andre, 1976). Unlike other turbines, this high efficiency can be reached over a wide range of flow and head water characteristics through the use of adjustable guide vanes and runner blades (Waters and Aggidis, 2015). As a consequence, smaller reservoirs can be built and the river maintains free flowing sections with use of this type of turbine compared to traditional ones (i.e., accumulation reservoir). Our study was conducted both in reservoirs and downstream river sections, and all sites have been sampled before and after the dams' closure.

We examined two data sets collected in the Madeira River. The first one referred to fish assemblages depending on floodplains areas, hereafter pelagic fish assemblage, sampled by gillnets in the sites: (1) Mutum-Paraná River, (2) São Lourenço Creek; (3) Karipunas Creek; (4) Jaciparaná River; (5) Jatuarana Creek; (6) Belmont Creek; and (7) Machado River (Fig. 1). Sites 1-2, and 3-5 were flooded to create Jirau and Santo Antonio Dams, respectively. Sites 6 and 7 are positioned downstream both reservoirs (Fig. 1). The second data set was sampled in three transects (A Jirau reservoir area; B Santo Antonio reservoir area; and C downstream reservoirs) by using bottom trawl nets, hereafter benthic fish assemblages, and we hypothesized they would be less impacted by dams because its low level of dependence of a local floodplain. Each transect was composed of five collections of ten minutes each, combined as a single sample. A detailed description of Madeira River can be found elsewhere (Torrente-Vilara *et al.*, 2011; Queiroz *et al.*, 2013). All data were collected under the Santo Antonio Energia (SAE) and Energia Sustentável do Brasil (ESBR) Fish Conservation Programs.

Biological data

Fish sampling was conducted bimonthly during six years: 2004 and 2009, 2010 to 2011 before the dams closure (September 2011), and from 2012, 2013 and 2014 after damming. Pelagic fish assemblage have been sampled with a set of 13 gillnets (sites 1 to 7) with mesh size varying from 30 to 200 mm between opposite knots, totaling 431-m^2 per sample left in the water for 24 hours. Benthic fish assemblages were sampled in transects composed of five collections of ten minutes each combined as a single sample. Sampling was conducted using a 3 m × 1 m bottom trawl net that was hauled behind a boat (Lopez-Rojas *et al.*, 1984; Cella-Ribeiro *et al.*, 2015). The net had a 5 mm mesh bag that was lined with 1 mm mesh netting.

Pelagic fish assemblages were sampled 36 times (18 samples before and 18 samples after dam closure) whereas benthic fish assemblages were sampled 24 times (12 samples before and 12 after dam closure). Machado River (site 7) was not sampled in the first year of study (2004); Mutum-Paraná River (site 1), São Lourenço River (site 2), and Karipunas River (site 3) were not sampled in April, August and October 2014 due to the risk of sampling during an exceptional flood in Madeira River Basin (Fearnside 2014a). The individuals were euthanized using a lethal dose of clove oil (Eugenol) and immediately placed on ice (fishes collected by gillnets) or preserved in a 10% formalin solution (fishes collected by trawl net). The fish specimens were identified following (Queiroz *et al.*, 2013).

Environmental variables

Physicochemical characteristics of the water were measured at each sampling in all sites. The variables dissolved oxygen (mg1^{-1}), water temperature (^{o}C), pH, conductivity (μSm^{-1}), turbidity (UNT), water transparency (cm), depth (m), width (m) and water speed (m/s) were used to evaluate the effects of local environmental changes on fish assemblage changes after dams closure in Madeira River.

Data analyses

All data analyses have been performed in R environment (R Core Team 2014) using the *vegan* packages (Oksanen *et al.*, 2016). We conducted mostly parametric analyses, but we used the corresponding nonparametric tests when variable distributions did not meet the normality assumption (checked using Shapiro-Wilk test) or number of samples were not equal between both periods. To test the differences in fish assemblage attributes (richness, abundance, biomass, diversity indices and composition dissimilarities) and environmental variables at each site between before and after dams construction, we compare all samples from both periods (before *vs* after). The number of samples per site were: sites 1 and 3 (18 samples before *vs* 15 samples after), site 2 (17 samples before *vs* 14 samples after), sites 4, 5 and 6 (18 samples before *vs* 18 samples after), site 7 (12 samples before *vs* 16 samples after), sites A, B and C (12 before *vs* 12 after).

Richness, abundance and biomass

Richness (S) was measured as the absolute number of species present in each sample. Species abundance and biomass were defined as catch per unit effort, expressed as individuals or weight/431m^2/24h for pelagic fish assemblages. Benthic fish assemblage was evaluated by individuals/10 minutes of trawling. We tested differences in fish species richness, abundances and biomass before and after dam construction with *t*-student and Mann-Whitney non-parametric tests.

Diversity indices

We tested for differences in fish diversity before and after dams construction with the Shannon-Wiener (H') and the Simpson's diversity index (D, Simpson 1949), which differ in their theoretical foundation and interpretation (Magurran, 2004). The Shannon index is affected both by the number of species and the evenness of species abundances, whereas the

Simpson index gives more weight to common or dominant species. Contrary to the Shannon index, rare species with only a few individuals will not affect Simpson index. Pielou's (J) index quantifies how even is abundance of species among samples, is derived from the Shannon index and is constrained between 0 and 1; the lower the variation among species abundance, the higher the J value. Shannon (H'), Simpson's (D), and Pielou's J represent thus variation in species diversity and evenness and were calculated for both pelagic and benthic fish assemblages. Differences in these indices before and after dam construction were tested with *t*-student and Mann-Whitney tests.

Composition dissimilarities before and after dams closure in each site

Relative abundance of pelagic and benthic fish species were first square-rooted transformed to reduce the influence of abundance/common species on the analyses, then pairwise dissimilarity matrix among sites were calculated using the Bray-Curtis distance metric. We tested for overall differences in fish species assemblages before and after dam closure using Permutational Analysis of Variance (PERMANOVA) with 999 permutations. Then, we summarized species assemblages with Non-Metric Multidimensional Scaling (NMDS) in two dimensions to visualize major differences before and after dams closure in Madeira River in only two dimensions (i.e. two NMDS axis). NMDS is a powerful unconstrained ordination method frequently used for indirect gradient analysis in community ecology (Minchin, 1987). We also tested for differences in the NMDS scores (2 axes) before and after dams closure in Madeira River with univariate ANOVAs. The NMDS, PERMANOVA and ANOVA were performed using the functions metaMDS, adonis and anova respectively.

Environmental variables

First, the environmental variables were tested in four data set according to the habitat (pelagic or benthic) and the river section (reservoir or downstream). We calculated the Euclidean distance matrix for each data set. Then, to test differences in environmental characteristics before and after dam closure we performed the test PERMANOVA (999 permutations) for each data set. After that, to test the differences in each environmental variable at each site among before and after dams construction we used *t*-student and Mann-Whitney non-parametric tests.

Influence of environmental variables on fish assemblage impacted by dams closure

We used the NMDS axes to test changes in species composition related to changes in local habitat characteristics. For this analyze, fish assemblages were grouped in four data set according the fish assemblage (pelagic or benthic) and river section (reservoir or downstream). For each data set, the model included the scores of the first two NMDS dimensions (species composition) as dependent variables and the $\log_{10}$-transformed local habitat variables as independent (dissolved oxygen mgl^{-1}, water temperature ^{o}C, pH, conductivity μSm^{-1}, turbidity UNT, water transparency cm, depth m, width m and water speed (m/s). Four models were performed; MOD-1 with pelagic fish from reservoir area, MOD-2 with benthic fish from reservoir area, MOD-3 with pelagic fish from downstream and MOD-4 with benthic fish from downstream.

Results

Richness, abundance, biomass, and diversity

A total of 45,387 specimens representing 438 fish species, 41 families, and 11 orders were recorded. Pelagic and benthic fish assemblages' were composed of 365 and 138 fish species, respectively. Only 66 (15 %) fish species were common to both sampling methods. Pelagic fish assemblages increased in richness in most sites positioned in both reservoirs areas (Fig. 2; Table 1; sites 1 to 5). Fish abundance and biomass increased in all but Mutum-Paraná River (site 1) sites positioned upstream and downstream the reservoirs. No differences were detected for richness, abundance, and biomass in sites from downstream reservoirs (Fig. 2; Table 1; sites 6 and 7), and benthic fish assemblage from all sites (Fig. 4; Table 1; sites A, B and C). Diversity index described by H' and D revealed differences in pelagic fish assemblage composition after dams closure only for Karipunas Creek, and Jaciparaná River (Fig. 3; Table 2; sites 3 and 4 locate between Santo Antonio and Jirau dams). No differences were detected for diversity indices in most sites in spite of the considered fish assemblage (Fig. 3 and 4; Table 2; sites 6, 7, A, B and C).

Changes in species composition

Pelagic fish assemblage before and after dam closure differed for most sampling sites (PERMANOVA: Fig. 5; Table 3; sites 1 to 5), whereas sites 6 and 7 were similar in terms of species composition. Before and after changes in benthic fish assemblage were detected only downstream the dams (PERMANOVA: Fig. 5; Table 3; site C), whereas upstream sites showed no effect at all. The ANOVA detected NMDS scores variation before and after dams closure only for pelagic fish assemblage in sites 1, 2, 3 (all of them in Jirau UHE), and 7 (Machado River, a downstream area) (Table 3).

Influence of environmental variables on the fish assemblages

Reservoir area showed significant differences for local environmental variables between before and after dams closure (PERMANOVA: Table 4). Those variables were width, transparence, turbidity, conductivity, temperature, dissolved oxygen and pH for tributaries (sites 1 to 5) and, water speed and dissolved oxygen for Madeira River main channel (sites A and B). No differences for environmental variables were detected for downstream reservoirs (tributaries sites 6, 7 and main channel C, PERMANOVA: Table 4).

The multiple regression models showed local environmental conditions (conductivity, depth, oxygen, and temperature) as the best predictor to explain changes in pelagic fish assemblages in the reservoir area (MOD 1; sites 1 to 5; Fig. 6; Table 5) while depth explain composition variation in a downstream river section (MOD-3; sites 6 and 7; Fig. 6; Table 5). Changes in benthic fish assemblages in the reservoir (sites A and B) were not related to local factors measured in this study (MOD-2; sites A and B; Fig. 6; Table 5) whereas dissolved oxygen explains downstream changes in benthic fish assemblages (MOD-4; site C; Fig. 6; Table 5).

Discussion

Our study represents the first assessment of dams closure in cascade in the largest white-waters Amazon River. Differences in pelagic fish assemblage composition into both reservoirs were noticed immediately, and linked to the increasing in conductivity, water temperature, and reduction in the dissolved oxygen and deep. However, benthic fish assemblage from reservoir area seems to not respond immediately to the dams closure. A secondary set of results were the maintenance of pelagic fish assemblage downstream the dams closure in cascades. These assemblages, immediately downstream (site 6, Belmont creek, 30 km from Santo Antonio Dam), and Machado River (site 7, 300 km from Santo Antonio Dam) did not change in terms of fish species assemblages probably due to the maintenance of local habitats.

The increase in richness, abundance and biomass of pelagic fish assemblages from tributaries flooded by the reservoirs are associated to immediate changes that occurred in the environment during the upsurge phase of the reservoir when a large amount of organic matter is available (see Agostinho *et al.*, 2015). The large flood caused by the reservoir resulted in an increase in space available for fish and other aquatic organisms (Fernandes *et al.*, 2009). Local areas previously spread over a variety of habitats and segregated by waterfalls (Torrente-Vilara *et al.*, 2011) changed due to the homogenizations and modification of resources' availability (Agostinho *et al.*, 2008; Fernandes *et al.*, 2009; Wang *et al.*, 2012; Lima *et al.*, 2015). However, there was an exception to the increase in abundance and biomass in the Mutum-Paraná River (site 1). Mutum-Paraná River is positioned upstream from the dam and with open access to upstream Madeira River where a small stretch of waterfall resists the drowning of Jirau Dam.

On the other hand, the structure of the benthic fish assemblage in the main channel of the Madeira River remained the same during the three years after the dams. Two hypotheses could explain this lack of response from this benthic community: (i) few environmental changes took place in the main channel of the Madeira River due a lower impact caused by bulb turbine when compared to traditional reservoirs (see Table 4) and (ii) species from common habitats around the area has weak potential to occupy niches in the bottom. In fact, the bottom of the Madeira River has a rare fish fauna composed mostly by Gymnotiformes and small Siluriformes species well adapted to the conditions of deep habitat (Lundberg and Py-Daniel, 1994; Cox Fernandes *et al.*, 2004; Covain and Fisch-Muller, 2007; Lundberg *et al.*, 2013; Queiroz *et al.*, 2013) supporting this hypothesis.

The literature has demonstrated that large dams resulted in a decrease in fish abundance and an alteration in fish composition downstream river (Petrere, 1996; Hallwass *et al.*, 2013; Lima *et al.*, 2015). Fish density seems to be generally high in the first few kilometers downstream of dams and due high concentration of migratory and non-migratory rheophilic fish (Pompeu and Martinez, 2006; Agostinho *et al.*, 2012; Lima *et al.*, 2015). However, the results of this study revealed that during the three years immediate after dams construction in the Madeira River, the pelagic fish assemblage attributes from downstream remained the same. These findings seem to be related to the maintenance of the local environmental factors downstream dams during the first three years after the dams closure also. There was an increase of marginal pelagic species in the benthic assemblage downstream dams (site C). The high erosion of the Madeira River banks downstream dams (Fearnside, 2014) forced the shoreline species towards to the main channel. Channeling the water from the spillway of the UHE Santo Antonio altered the downstream flow of the dam to a single drop compared to the old Santo Antônio waterfall (blasted), suddenly increasing the erosion in the downstream area of the UHE (Fearnside, 2014).

Though *Oreochromis niloticus* has been observed during these first three years of monitoring (da Graça *et al.*, 2013), a massive pool of invasive species was not observed colonizing the Madeira River reservoirs. Differences in pelagic fish assemblages' composition from reservoirs areas were more associated to changes in the abundances of species that already inhabited the area. Migratory Characiformes (N_{pre}= 1,179 and N_{post}= 9,624; e.g. the omnivorous *Triportheus* spp., the frugivorous *Mylossoma* spp., the detritivorous *Potamorhina* spp., *Psectrogaster* spp., the carnivorous *Pellona* spp.) and filter-feeding species (planktivorous; N_{pre}= 158 and N_{post}= 2,184; e.g. *Jurengraulis* juruensis, *Hypophthalmus* spp., *Anodus* spp., *Chaetobranchus flavescens*) were caught more frequently in the reservoirs. There was an increase in catches of *Plagioscion squamosissimus* and *Pterodoras granulosus* juveniles (less than 5 cm) in the Madeira River channel. These species have the potential to dominate the reservoirs of the Madeira River, as observed in other Brazilian reservoirs, including the Tocantins River, Amazon (Agostinho *et al.*, 2008; Araújo *et al.*, 2013). A set of species that depend on flood pulse to reproduce had high abundance in the reservoirs. If those set of species were able to adapt their life cycle to remaining rivers section between the reservoirs (as the tributary Jaciparaná River with 12163.20 km^2 sub-basin area), maybe they will persist in Jirau and Santo Antonio dams. Otherwise, blocking their life cycle could lead to local extinctions or a depletion of migrating species (Porcher and Travade, 1992; Kuussaari *et al.*, 2009; Duponchelle *et al.*, 2016). Still, some species that were previously abundant in the rapids stretch of tributaries, especially piscivores such as the genus *Acestrorhynchus* spp., have become scarce in the reservoirs suggesting low adaptability of this species to the new reservoir conditions and possibly local changes in trophic dynamics (but see Pereira *et al.*, 2016).

Jirau and Santo Antônio dams' were the first dams to be built with bulb turbines in a system of white-water, with great flow, and high fish species richness, such as the Madeira River (Queiroz *et al.*, 2013). Although we confirmed an increase in some of the fish assemblage attributes from Madeira River reservoirs' expected for the heterotrophic phase, it is still unknown how a river as the Madeira can withstand high species richness after the high magnitude impact caused by these dams. Reductions in species richness and fish diversity are also expected after this heterotrophic phase in the reservoir (Agostinho *et al.*, 2008) but bulb turbines seems to damp severe effects downstream dams. Standard, long-term monitoring is recommended after the stabilization of these reservoirs, to further evaluate the capacity to support this rich ichthyofauna, especially for benthic fish assemblage.

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Table 1. Richness, abundance and biomass comparison before and after dams closure in Madeira River. Main channel the absolute abundance and biomass per 10 minutes trawling. * Significant (*t-test*, $P < 0.05$) differences. * $P < 0.05$; ** $P < 0.01$ and *** $P < 0.001$.

Sites		Richness (S)			Abundance (ind/480m^2/24h, except to sites A, B, and C)			Biomass (weight/480m2/24h) , except to sites A, B, and C)			N
		PRE total (range) mean ± SD	POST total (range) mean ± SD		PRE total (range) mean ± SD	POST total (range) mean ± SD		PRE Total (range) mean ± SD	POST total (range) mean ± SD		
Reservoir area	1	133 (8−58) 20±14	162 (20−81) 38±15	**	2.1 (0.02−0.82) 0.14±0.19	7.24 (0.09-3.00) 0.48±0.71		435.98 (6.39-139.47) 29.06±32.61	1057.69 (15.01-379.49) 70.51±88.24		32
	2	118 (2−59) 20±14	164 (24−84) 45±15	***	2.61 (0.01-0.95) 0.18±0.25	10.69 (0.12-2.02) 0.76±0.52	**	419.99 (0.59-149.78) 29.99±37.45	1404.16 (24.71-313.38) 100.30±78.92	**	30
	3	140 (4−68) 22±16	143 (19−83) 42±17	**	3.22 (0.01-1.50) 0.21±0.36	12.83 (0.10-2.69) 0.85±0.78	**	565.73 (3.04-314.98) 37.71±77.78	2044.79(10.36-277.04) 172.19±76.10	**	32
	4	109 (12−30) 23±5	164 (7−65) 39±15	***	5.45 (0.04-0.75) 0.30±0.21	13.49 (0.02-2.41) 0.74±0.56	**	463.14 (4.84-51.00) 25.73±10.87	1662.20 (1.22-224.17) 92.345±65.49	***	36
	5	104 (8−29) 18±6.3	137 (9−63) 31±17	***	2.88 (0.05-0.34) 0.16±0.09	13.66 (0.04-3.19) 0.75±0.96	**	415.39 (7.92-76.04) 23.07±18.02	1220.63 (9.32-146.71) 67.81±41.51	***	36
	A	65 (2−30) 13±9	48 (2−29) 11±9		205 (2−44) 18±13	194 (2−68) 18±20					24
	B	44 (4−15) 10±3.7	62 (2−35) 12±10.6		333 (7−73) 28±18.6	906 (5−412) 75±138.1					24
Downstream reservoirs	6	94 (8−38) 19±7.4	105 (7−44) 19±9.9		5.21 (0.06-0.69) 0.28±0.18	4.39 (0.02-0.84) 0.24±0.22		603.81 (3.58-123.20) 33.54±32.12	451.48 (1.10-92.29) 25.08±26.44		36
	7	97 (16−39) 26±6.5	133 (7−52) 23±13.2		2.51 (0.06-0.41) 0.20±0.10	2.64 (0.02-0.92) 0.22±0.25		357.89 (5.81-55.73) 29.82±15.73	325.37 (1.36-128.18) 27.11±36.71		28
	C	49 (3−20) 9±5.2	45 (2−23) 7.5±5.8		301 (3−85) 25±23.2	347 (5−119) 29±39.1					24

Table 2. Diversity index, Simpson's (D), Shannon's (H) and Pielou's (J) comparison before and after dams closure in Madeira River. * Significant (*t-test*, $P < 0.05$) differences. * $P < 0.05$; ** $P < 0.01$ and *** $P < 0.001$.

Sites		Simpson's index (D)			Shannon's index (H')			Pielou's J		N
		PRE	POST		PRE	POST		PRE	POST	
		total (range) mean±SD	total (range) mean±SD		total (range) mean±SD	total (range) mean±SD		total (range) mean±SD	total (range) mean±SD	
Reservoir area	1	0.95 (0.69−0.95) 0.86 ± 0.06	0.95 (0.66−0.96) 0.88 ± 0.08		3.87 (2.18−3.23) 2.69±0.29	3.83 (1.50−3.24) 2.61±0.50		0.76 (0.72−1) 0.86±0.67	0.78 (0.56−0.93) 0.80±0.12	32
	2	0.84 (0.32−0.92) 0.77 ± 0.15	0.81 (0.30−0.93) 0.78 ± 0.19		3.15 (0.50−3.08) 2.17±0.57	2.98 (0.89−3.29) 2.57±0.72		0.62(0.52−0.97) 0.78±0.14	0.62 (0.28−0.88) 0.68±0.17	30
	3	0.95 (0.60−0.95) 0.85 ± 0.10	0.96 (0.84−0.95) 0.92 ± 0.03	**	3.59 (1.11−3.55) 2.48±0.66	3.94 (2.37−3.53) 2.98±0.32	**	0.72 (0.62−0.95) 0.84±0.06	0.80 (0.64−0.89) 0.82±0.07	32
	4	0.94 (0.41−0.92) 0.77 ± 0.15	0.83 (0.71−0.92) 0.86 ± 0.05	*	3.58 (0.99−2.88) 2.21±0.57	2.88 (1.82−3.28) 2.64±0.36	**	0.70 (0.34−0.93) 0.71±0.17	0.61 (0.58−0.93) 0.74±0.08	36
	5	0.89 (0.35−0.93) 0.80 ± 0.14	0.94 (0.73−0.94) 0.85 ± 0.058		3.03 (0.94−2.91) 2.22±0.57	3.52 (1.72−3.35) 2.49±0.42		0.61 (0.36−0.95) 0.78±0.13	0.76 (0.51−0.94) 0.76±0.11	36
	A	0.91 (0.62−0.91) 0.80±0.10	0.83 (0.48−0.90) 0.71±0.12		3.17 (1.04−2.66) 2.01±0.62	2.42 (0.67−2.46) 1.57±0.54		0.74 (0.67−1) 0.88±0.09	0.62 (0.55−1) 0.84±0.15	24
	B	0.92 (0.61−0.90) 0.77±0.10	0.90 (0.48−0.87) 0.77±0.11		3.04 (1.15−2.47) 1.84±0.45	2.87 (0.67−2.58) 1.81±0.48		0.78 (0.69−0.98) 0.85±0.09	0.69 (0.62−1) 0.86±0.12	24
Downstream reservoirs	6	0.93 (0.58−0.94) 0.79 ± 0.10	0.92 (0.60−0.90) 0.80 ± 0.09		3.33 (1.25−3.20) 2.14±0.52	3.17 (1.36−2.90) 2.16±0.46		0.71 (0.49−0.88) 0.74±0.11	0.70 (0.55−0.96) 0.77±0.12	36
	7	0.96 (0.75−0.94) 0.88±0.05	0.95 (0.67−0.94) 0.88±0.07		3.98 (2.18−3.23) 2.69±0.29	3.63 (1.50−3.24) 2.61±0.50		0.81 (0.67−0.91) 0.83±0.07	0.79 (0.76−0.97) 0.85±0.06	28
	C	0.91 (0−0.85) 0.62±0.23	0.90 (0.50−0.86) 0.70±0.13		2.89 (0−2.32) 1.36±0.66	2.87 (0.69−2.37) 1.50±0.50		0.77 (0.65−0.94) 0.83±0.10	0.75 (0.57−1) 0.86±0.15	24

Table 3. Non-metric multidimensional scaling (nMDS) results (% of capture – R^2 and stress) of fish assemblages composition basead on the Bray-Curtis metric of dissimilarity, PERMANOVA test and ANOVA test before and after dams closure in Madeira River. Significance in each local are marked with a line (p-value).

Sites		R^2	stress	PERMANOVA		ANOVA	
				Pseudo-F	*P*-value	*F*	*P*-value
Reservoirs area	1	0.51	0.261	1.729	0.001	11.206	0.005
	2	0.55	0.258	2.385	0.001	14.276	0.000
	3	0.54	0.216	2.763	0.001	16.476	0.000
	4	0.67	0.177	2.981	0.001	1.021	0.319
	5	0.51	0.260	2.912	0.001	0.353	0.556
	A	0.49	0.188	1.053	0.351	0.105	0.748
	B	0.42	0.246	1.276	0.061	0.424	0.521
Downstream reservoirs	6	0.46	0.259	1.252	0.086	1.001	0.324
	7	0.53	0.247	1.327	0.051	12.941	0.001
	C	0.53	0.215	1.526	0.032	0.111	0.741

Table 4. Mean (range) of environmental characteristics in the sampling periods before and after dams closure in Madeira River. *Variables that showed significant differences (*t*-student test or Mann-Whitney test) among prior to, and after dams closure. * $P < 0.05$; ** $P < 0.01$ and *** $P < 0.001$.

Local	Reservoirs						Downstream reservoirs			
	Tributaries			Main channel			Tributaries		Main channel	
PERMANOVA	*Pseudo*-F = 14.025, *P* = 0.001			*Pseudo*-F = 6.808, *P* = 0.001			*Pseudo*-F = 1.124, *P* = 0.356		*Pseudo*-F = 1.695, *P* = 0.112	
Variables	mean (range) n=166			mean (range) n=48			mean (range) n=64		mean (range) n=24	
	PRE	POST		PRE	POST		PRE	POST	PRE	POST
Water speed (m/s)	< 0.1	< 0.1		1.05 (0.32–1.75)	0.68 (0.1–1.27)	**	< 0.1	< 0.1	1 (0.72–1.43)	0.87 (0.49–1.09)
Water transparency (m)	1.20 (0.3–2.08)	1.12 (0.1–6.30)	**	0.16 (0.05–0.50)	0.15 (0.05–0.56)		0.5 (0.06–1.5)	0.45 (0.1–1.08)	15 (8–32)	11.8 (6.4–17.6)
Width (m)	32 (8–80)	399 (6–1680)	***	877 (324–1,386)	885 (332–1,594)		85.4 (1.5–291)	86.3 (10–302)	1,161 (846–1,625)	1,090 (590–1,364)
Deep (m)	5.7 (1–13.3)	6 (1.5–15)		13.5 (5.9–23.1)	13.9 (3.7–19.2)		5.7 (1.4–19)	4.6 (1–10.2)	10.8 (5.8–17.3)	10.3 (6.2–16.4)
Turbidity (UNT)	12.2 (1.88–126)	36.3 (1.1–372)	***	180 (39.2–428.6)	255 (48.3–879)		127 (1.7–651)	72 (7.8–447)	201 (71–534)	267 (61–710)
Conductivity (µS/cm)	9.7 (1–32.1)	25.1 (6–111.5)	***	72.6 (32–135.4)	61.5 (23.4–115.5)		37.1 (3–110)	36 (4–104)	67.3 (26.2–128)	68.2 (29.6–116.8)
pH	5.9 (4.4–7.9)	6.2 (4–7.9)	*	6.9 (5.7–7.9)	7.04 (6.2–7.7)		6.2 (5.2–7.5)	6.4 (5.5–7.5)	6.9 (6.3–7.9)	7.1 (6.3–7.7)
Dissolved oxygen (mg.l^{-1})	4.3 (1.5–9.7)	5.3 (1–9.8)	**	3.86 (0.43–7.9)	5.27 (0.25–9.1)	*	3.9 (1.3–6.9)	4.8 (1.13–7.7)	5.85 (2.40–7.9)	7.22 (4.9–8.9)
Temperature	26.7 (23.9–31.2)	28 (23.9–36.2)	***	28.3 (25.4–31)	28.2 (24.7–31.6)		28.2 (25.5–32)	28 (25.5–32.2)	28.3 (25.7–30.5)	28.0 (25.6–29.8)

Table 5. Regression of fish compositional (NMDS scores) against explanatory variables (Log_{10}) related to the floodplain fish assemblages from reservoir area prior to, and after damming Madeira River. Variables that showed significance in each model are marked with a line (*p*-value).

Variables	MOD-1 Reservoir Pelagic		MOD-2 Reservoir Benthic		MOD-3 Downstream Pelagic		MOD-4 Downstream Benthic	
	$R^2 = 0.246$; $P < 0.001$ AIC = 30.440		$R^2 = 0.27$; $P < 0.05$ AIC = 0.88		$R^2 = 0.39$; $P < 0.001$ AIC = 16.573		$R^2 = 0.39$; $P < 0.05$ AIC = 8.415	
	Regression coefficients (β)	*P*	Regression coefficients (β)	*P*	Regression coefficients (β)	*P*	Regression coefficients (β)	*P*
Water speed (m/s)	not included		-0.080	0.830	not included		-0.724	0.300
Water transparency (m)	0.063	0.317	-0.420	0.311	-0.005	0.975	-0.023	0.960
Width (m)	0.050	0.099	-0.141	0.684	-0.149	0.105	0.263	0.807
Depth (m)	0.128	0.035	-0.551	0.207	0.492	0.000	-0.089	0.924
Turbidity (UNT)	0.066	0.114	0.116	0.794	0.004	0.955	0.075	0.810
Conductivity (µS/cm)	0.152	0.008	0.142	0.693	0.132	0.280	0.192	0.563
pH	-0.124	0.681	5.032	0.165	-0.229	0.832	4.700	0.112
Dissolved oxygen ($mg.l^{-1}$)	0.160	0.028	0.112	0.610	0.101	0.487	0.852	0.025
Temperature	1.921	0.004	-1.431	0.669	-2.106	0.148	3.646	0.187

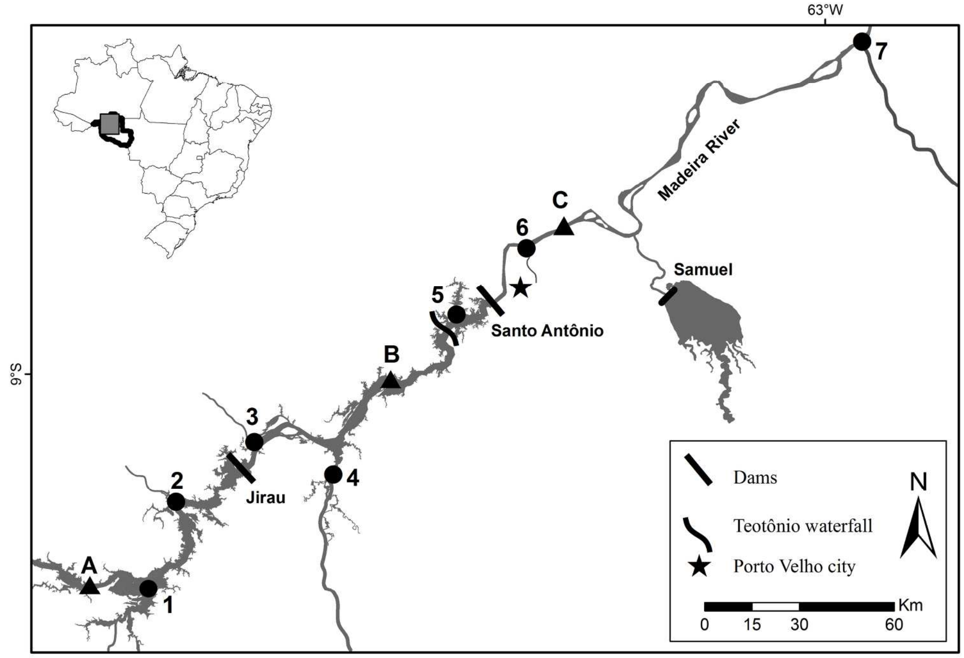

Fig. 1. Study area and the sampling sites in Jirau reservoir area (1 - Mutum-Paraná River, 2 - São Lourenço Creek; in Santo Antonio reservoir area (3 - Karipunas Creek, 4 - Jaciparaná River, 5 - Jatuarana Creek), and downstream river sections (6 - Belmont Creek and 7 - Machado River). Main channel transects A - Jirau reservoir area, B - Santo Antônio reservoir area and C - downstream dams.
210x148mm (300 x 300 DPI)

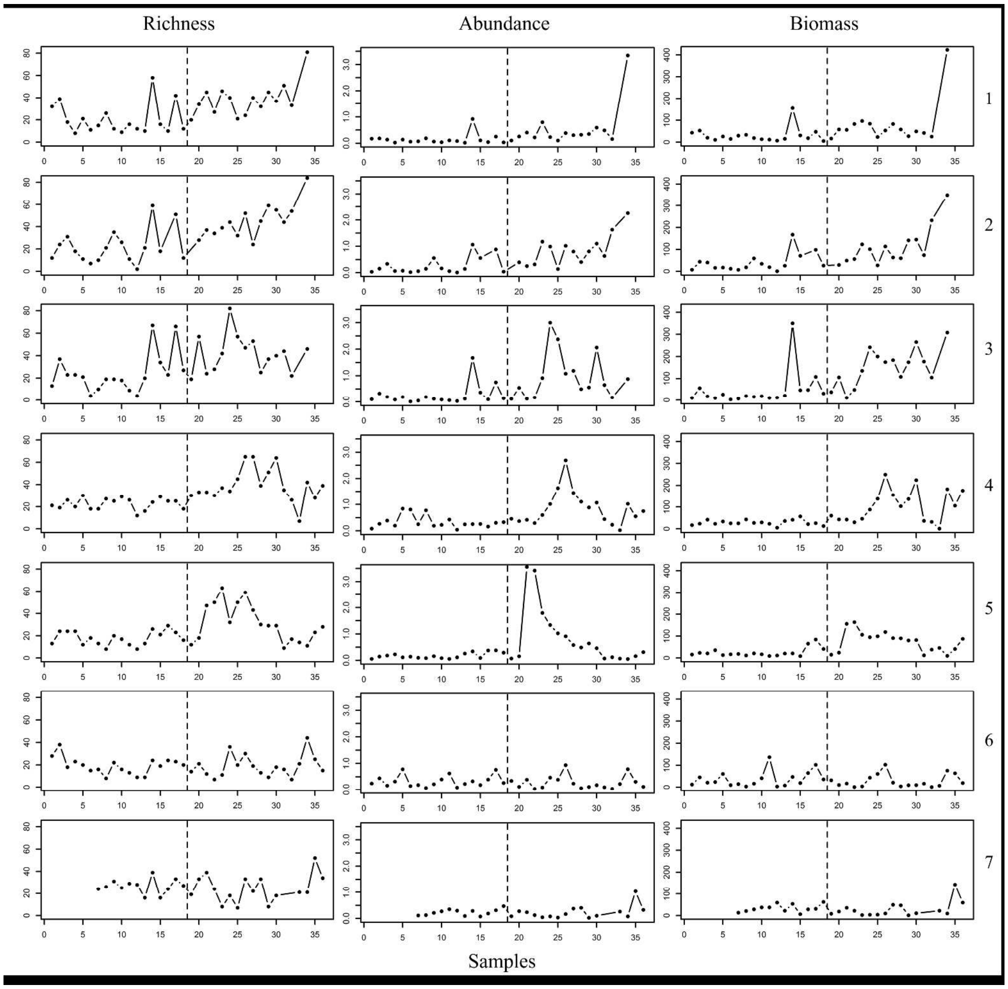

Fig. 2. Time series for richness, abundance (ind/431m2/24h) and biomass (weight/431m2/24h) of pelagic fish assemblage sampled by gillnets (1 to 7, indicated on the right side) before and after dams closure in Madeira River. Dashed line indicates the Santo Antônio dam closure.
635x621mm (150 x 150 DPI)

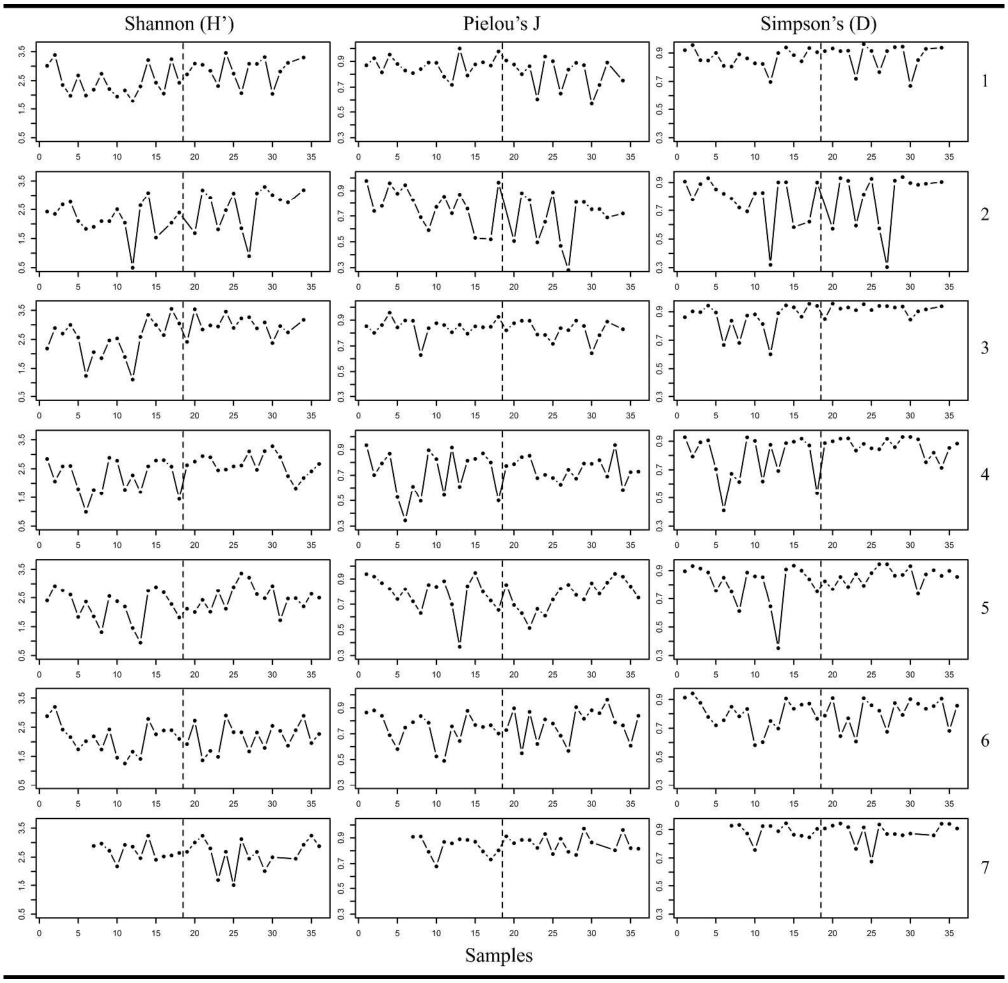

Fig. 3. Time series of Shannon (H'), Pielou's J and Simpson's (D) diversity indices for pelagic fish assemblage sampled by gillnets sites (1 to 7, indicated on right side) before and after dams closure in Madeira River. Dashed line indicates the Santo Antônio dam closure.
632x615mm (150 x 150 DPI)

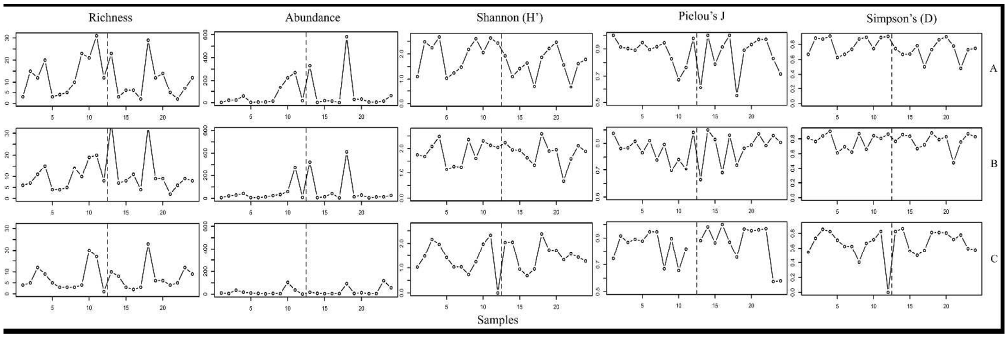

Fig. 4. Time series richness, abundance (ind/10min/trawlling), diversity index Shannon (H'), Pielou's J and Simpson's (D) of benthic fish assemblages sampled by trawlnet (A, B and C transects) before and after dams closure in Madeira River. Dashed line indicates dam closure.
1046x346mm (150 x 150 DPI)

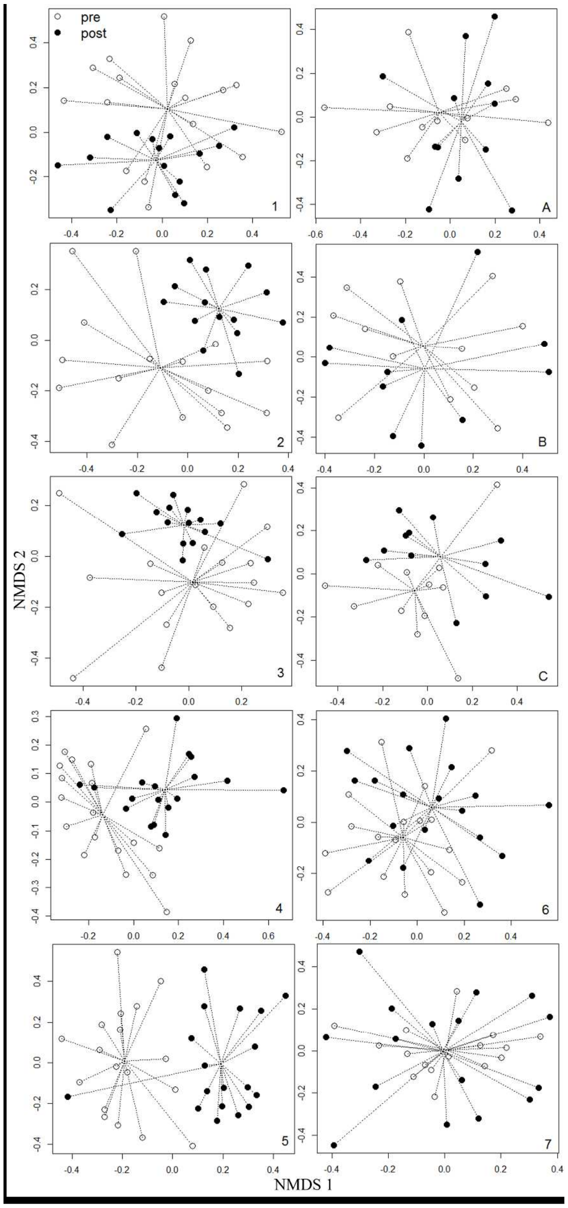

Fig. 5. Non-metric multidimensional scaling (nMDS) plot of pelagic and benthic fish assemblages' composition based on the Bray-Curtis dissimilarity index before (white points) and after (black points) dams closure in Madeira River. Dashed line indicate the distance of each point to the centroid group. Numbers and letters indicate the sampling site.
266x570mm (150 x 150 DPI)

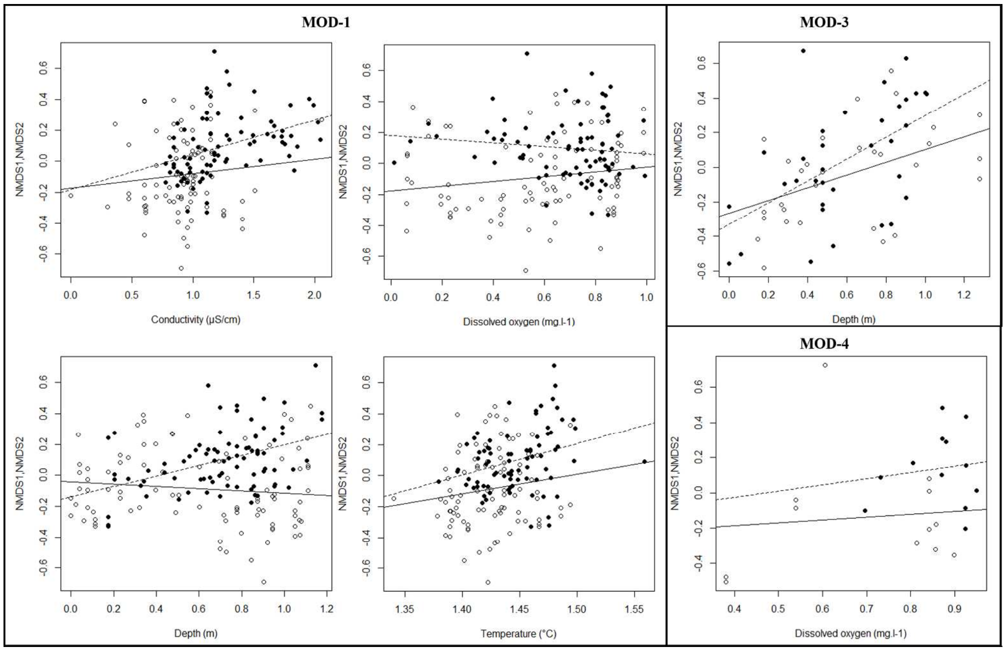

Fig. 6. Comparison among scores of non-metric multidimensional scaling analysis (first two NMDS dimensions) and the environmental variables (Log10) that showed significant differences (regression models 1, 3 and 4; see Table 5) before and after dams closure in Madeira River. White points (before) and black points (after).
423x272mm (150 x 150 DPI)

CONSIDERAÇÕES FINAIS

As primeiras respostas da ictiofauna em relação aos empreendimentos instalados no rio Madeira ocorreram com diferentes intensidades de acordo com o hábitat da assembleia de peixes. A ictiofauna associada ao canal principal do rio Madeira não apresentou alterações importantes na área dos reservatórios. Duas hipóteses foram apresentadas para explicar a resistência dessa ictiofauna perante os empreendimentos, (i) poucas mudanças ambientais nesse habitat e (ii) espécies dos habitats presentes na área do reservatório supostamente possuem baixa ou nenhuma capacidade de ocupar nichos tão específicos relacionados às espécies do fundo do canal do rio.

Ao contrário do que foi observado para a assembleia de peixes do canal, alterações imediatas foram detectadas na ictiofauna associada as planícies de inundação. O aumento de riqueza, abundância e diversidade nessa assembleia são esperados para essa fase heterotrófica e inicial da formação de reservatórios clássicos, quando ocorre um *input* de matéria orgânica no sistema (ver Petrere, 1996; Agostinho *et al.*, 2008; Gao *et al.*, 2010; Agostinho *et al.*, 2015; Lima *et al.*, 2015). A única exceção foi observada para a assembleia de peixes de planície inundável no rio Mutumparaná (diferenças observadas apenas para riqueza de espécies, e.g. Tabelas 01 e 02, capítulo 1), posicionado no extremo montante do reservatório de Jirau e diretamente conectado com o trecho encachoeirado de 130 km remanescente da construção das usinas em cascata no rio Madeira. É de grande importância manter esse trecho de corredeiras intacto, possivelmente isolando os efeitos deletérios dos reservatórios. Alterações na ictiofauna desse hábitat foram relacionadas as alterações físico-químicas nas águas dos afluentes do antigo trecho de corredeiras.

Imediatamente a jusante de barragens é esperado um impacto na ictiofauna, podendo ser observado já nos primeiros anos após o barramento do rio (Lima *et al.*, 2015). Nos três primeiros anos após o barramento do rio Madeira não foram observadas alterações significativas na ictiofauna associada a planície de inundação imediatamente a jusante dos empreendimentos. A jusante dos barramentos foi detectada a inclusão de espécies reconhecidamente marginais (espécies presentes em praias e na vegetação ripária) na assembleia de peixes associada ao canal principal do rio Madeira, justificando a modificação pontual da estrutura da assembleia de peixes do fundo do canal do rio Madeira no transecto C (e.g. Fig. 05, quadro C, capítulo 1). A inclusão dessas espécies foi atribuída as mudanças no fluxo principal do rio Madeira, com consequente desbarrancamento das margens na região imediatamente jusante da UHE de Santo Antônio (Fearnside, 2014).

A tese demonstra que generalizações sobre impactos causados por barramentos ocultam respostas associadas a enorme heterogeneidade dos rios e da paisagem Amazônica. Avaliar novos modelos hidráulicos desenvolvidos e aplicados em rios de declividade não abrupta pode minimizar ou retardar alguns impactos, abrindo uma discussão sobre modelos alternativos para a construção de hidrelétricas na Amazônia. Além disso, a implementação do método de arrasto bentônico em estudos de monitoramento deve ser considerada como essencial, uma vez que esta ictiofauna exclusiva responde de forma distinta ao que tem sido observado até então para os demais hábitats e não é capturada por metodologias tradicionalmente implementadas nesses estudos como redes de espera, redes de cerco e puçás. Os reservatórios de Jirau e Santo Antônio constituem ambientes novos para a ictiofauna local. O panorama aqui apresentado é o resultado de alterações ocorridas em um espaço de 160 km (distância fluvial entre Santo Antônio UHE e Foz do rio Mutumparaná) em um tempo considerado curto para a vida útil de um reservatório, e um período de estudo que não contempla a expectativa de vida da maioria das espécies envolvidas no estudo (como exemplo, espécies de ciclo de vida longo como os grandes bagres *Brachyplatystoma* spp.). Neste sentido, respostas conclusivas sobre a nova organização da ictiofauna somente poderá ser obtidas no futuro, após monitoramento da fase em que seja possível o entendimento da “estabilização ambiental” de ambos reservatórios no rio Madeira.

MEMORIAL DO DOUTORADO

Durante o desenvolvimento do corpo principal dessa tese de doutorado elaborei cinco trabalhos sobre padrões naturais da ictiofauna do rio Madeira e um trabalho de registro das corredeiras e cachoeiras da região estudada. Estes apesar de importantes para a compreensão das possíveis modificações que virão ocorrer no grupo alvo desse estudo, não estão vinculados diretamente a questão central da tese, e por isso não foram incluídos como capítulos no presente documento. Dados gerados acerca da condição pristina as modificações do sistema do rio Madeira representam um marco histórico para a ictiofauna da região e servirão para futuras comparações e avaliações dos impactos promovidos pelas UHEs tanto na ecologia quanto na pesca.

Um estudo inédito de identificação e revisão taxonômica das espécies associadas ao canal principal do rio Madeira foi realizado. Como resultado publiquei, juntamente com o especialista/sistemata do grupo taxonômico avaliado (Gymnotiformes), cinco capítulos no livro *Peixes do rio Madeira* (Queiroz *et al.*, 2013, capítulos 40, 41, 42, 43 e 44, volume 3, disponíveis online em www.santoantonioenergia.com.br/peixesdoriomadeira).

Ainda no livro *Peixes do Rio Madeira*, desenvolvi um trabalho de registro histórico das corredeiras e cachoeiras do antigo trecho de corredeiras do rio Madeira, na porção brasileira da bacia (Queiroz *et al.*, 2013, capítulo 3, disponível online em www.santoantonioenergia.com.br/peixesdoriomadeira). Esse trecho de aproximadamente 290 Km do rio Madeira tinha grande importância na distribuição e movimentação da ictiofauna da bacia, atuando como um filtro ecológico (ver Torrente-Vilara et al., 2011). No ano de 2011 a maior parte desse trecho de corredeiras foi afogado pelos reservatórios das UHEs Jirau e Santo Antônio.

Sem dúvidas a maior riqueza da minha região é o Rio Madeira (Foto Diogo Barbalho Hungria).