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MARINA GRANAI

Da diversidade à conectividade da ictiofauna em um reservatório amazônico

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“Dedico este trabalho ao meu муж. Together we can rule the galaxy!”

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“It is important to draw wisdom from different
places. If you take it from only one place,
becomes rigid and stale. Understanding others,
the other elements and the other nations will help
you become whole.”

Uncle Iroh

RESUMO

O Rio Madeira, um dos maiores afluentes do Amazonas, abriga mais de mil espécies de peixes. A construção do Complexo Hidrelétrico do Rio Madeira ocasionou mudanças ambientais no trecho do rio antes marcado pela presença de corredeiras. No Capítulo I, investigou-se a reestruturação da ictiofauna frente à formação do reservatório da UHE Santo Antônio (UHE SA), comparando os períodos antes (2009-2011) e depois do enchimento (2012-2019), baseado em dados gerados durante monitoramento conduzido pela UHE no âmbito do licenciamento ambiental do empreendimento. Foram calculados índices de diversidade taxonômica e funcional da ictiofauna. A assembleia foi avaliada através de análise de redundância canônica, análise de redundância baseada em distância e partição da diversidade beta com base na dissimilaridade de Bray-Curtis. Em todo o período, registrou-se 311 taxa (espécies e morfo-espécies) e 42.117 indivíduos. Embora os índices taxonômicos não tenham se relacionado com a formação do reservatório, foram observadas variações nos índices de diversidade funcional da assembleia, predominantemente relacionadas à variação balanceada na abundância das espécies, sem evidência de homogeneização da ictiofauna. Como o reservatório ainda se encontra na fase de colonização, são esperadas variações na ictiofauna. Nesse contexto, como medida de mitigação à manutenção da conectividade longitudinal após o barramento, sistemas de transposição de peixes (STPs) são implantados mundialmente em empreendimentos, com o objetivo de favorecer a continuidade do uso de habitats e rotas pelas assembleias de peixes. Contudo, a avaliação da efetividade desses sistemas permanece limitada, principalmente, por desafios metodológicos relacionados ao monitoramento, como a ausência de protocolos padronizados e de métricas comparáveis entre empreendimentos. Para suprir essa lacuna, desenvolveu-se um dispositivo experimental de captura de peixes (FishCD) instalado na saída a montante do STP da UHE SA. No Capítulo II, avaliou-se a efetividade do dispositivo para monitoramento padronizado de um STP. Após a ativação do dispositivo, o tempo de operação foi registrado e realizaram-se três inspeções diárias para remoção dos peixes capturados. Indivíduos foram identificados, medidos, pesados e soltos. A riqueza de espécies foi estimada por curvas de rarefação e pelo estimador Chao 1. As taxas de captura foram analisadas utilizando modelos lineares generalizados do tipo zero-inflated negative binomial (ZINB), considerando como preditores o período da captura (dia/noite), a vazão defluente e o esforço padronizado. Entre setembro/2024 e abril/2025, foram registrados 149 indivíduos: 18 Espécies, 7 Famílias e 2 Ordens. Todas as espécies registradas já haviam sido previamente observadas a montante e a jusante da barragem. Characiformes apresentaram maior riqueza (11 espécies) e

Siluriformes a maior abundância (106 indivíduos). A abundância foi maior durante a enchente e a cheia, em consonância com deslocamentos de migradores, e as capturas foram mais frequentes à noite. A vazão defluente exerceu efeito negativo nas capturas, enquanto o período do dia afetou apenas a probabilidade de eventos com captura zero. Operacionalmente, o FishCD proporcionou amostragem contínua sem necessidade de energia externa, protocolo de captura padronizado e operacional estável sob diferentes condições ambientais e operacionais, oferecendo evidências complementares àquelas geradas por tecnologias já estabelecidas. O FishCD configurou-se como ferramenta replicável para quantificar, em nível de espécie, o sucesso de passagem em STPs. Os dados gerados fornecem a base essencial para avaliações e para a gestão adaptativa de sistemas amazônicos regulados por barragens.

Palavras-chave: Atributos funcionais. Peixe Migrador Neotropical. Petrecho de pesca. Rio Madeira. Usina Hidrelétrica.

ABSTRACT

From diversity to connectivity of the ichthyofauna in an Amazonian reservoir

The Madeira River, one of the largest tributaries of the Amazon, harbours more than one thousand fish species. The construction of the Madeira River Hydropower Complex led to environmental changes in a river stretch previously characterised by the presence of rapids. In Chapter I, fish assemblage reorganisation associated with the formation of the Santo Antônio Hydropower Plant (HPP SA) reservoir was investigated by comparing the periods before (2009-2011) and after impoundment (2012-2019), based on data from an ichthyofaunal monitoring programme required under Brazilian environmental licensing. Taxonomic and functional diversity indices of the ichthyofauna were calculated, and assemblage patterns were evaluated using canonical redundancy analysis (RDA), distance-based redundancy analysis (dbRDA), and beta diversity partitioning based on Bray-Curtis dissimilarity. Across the entire period (2009-2019), 311 taxa (species and morphospecies) and 42,117 individuals were recorded. Although taxonomic diversity indices showed no association with reservoir formation, variation was observed in the functional diversity indices of the assemblage, predominantly related to balanced variation in species abundances, with no evidence of assemblage homogenisation. As the reservoir remains in a colonisation phase, further variation in the ichthyofauna is expected over time. In this context, fish passage systems (fishways) are widely implemented worldwide as a mitigation measure to maintain longitudinal connectivity following river impoundment, aiming to support the continued use of habitats and movement routes by fish assemblages. However, the assessment of the effectiveness of these systems remains limited, mainly due to methodological challenges related to monitoring, such as the lack of standardised protocols and comparable metrics across developments. To address this gap, an experimental fish capture device (FishCD) was developed and installed at the upstream exit of the fishway at HPP SA. In Chapter II, the effectiveness of the device for standardised monitoring of a fish passage system was evaluated. Following FishCD activation, operational time was recorded and three daily inspections were conducted to remove captured fish. Individuals were identified, measured, weighed, and released. Species richness was estimated using rarefaction curves and the Chao1 estimator. Capture rates were analysed using zero-inflated negative binomial (ZINB) generalised linear models, with diel period (day/night), outflow discharge, and standardised sampling effort as predictors. Between September 2024 and April 2025, 149 individuals were recorded, representing 18 species, 7 families, and 2 orders. All recorded species had been previously observed upstream and downstream of the dam. Characiformes exhibited the highest species richness (11 species), whereas Siluriformes accounted for the highest abundance (106 individuals). Abundance was higher during the rising-water and high-water periods, consistent with migratory movements, and captures occurred more frequently at night. Outflow discharge exerted a negative effect on capture rates, while diel period affected only the probability of zero-capture events. Operationally, the FishCD enabled continuous sampling without the need for external power supply, applied a standardised capture protocol, and maintained stable performance under a range of environmental and operational conditions, providing complementary evidence to that generated by established monitoring technologies. The FishCD proved to be a replicable tool for quantifying species-level passage success in fish passage systems. The data generated provide an essential basis for assessment and adaptive management of Amazonian river systems regulated by dams.

Keywords: Functional traits. Neotropical Migratory Fish. Fishing Gear. Madeira River. Hydroelectric Plant.

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INTRODUÇÃO GERAL

A bacia do Rio Madeira, situada na margem direita no Rio Amazonas, possui uma área de drenagem com aproximadamente 1,37 milhões de km^2, com elevações variando entre 20 e 6.450 metros, precipitação entre 300 e 6.000 mm/ano, e presença de corredeiras e cachoeiras. Por fazer parte da bacia Amazônica, seus rios são classificados em: (i) águas pretas (ácidas, escurecidas por compostos húmicos e com baixa concentração iônica); (ii) águas claras (alta transparência e química intermediária); e (iii) águas brancas (originárias da região andina, turvas e relativamente ricas em nutrientes e sólidos dissolvidos) (Bogotá-Gregory et al., 2020; Graça et al., 2025; Junk et al., 2011). A combinação entre heterogeneidade de relevo, variação climática e diferentes tipos de águas resulta em uma ampla diversidade de habitats, onde são registradas mais de 1.000 espécies de peixes (Carolsfeld et al., 2003; Graça et al., 2025; Queiroz et al., 2013).

O Rio Madeira é o segundo maior afluente do Amazonas, com extensão total de 3.240 km. O maior trecho do rio (1.815 km) encontra-se na Bolívia, onde é denominado Rio Beni. Formado a partir da confluência dos rios andinos Beni e Mamoré, o Rio Madeira percorre 1.425 km de extensão em território brasileiro, sendo classificado como um rio de águas brancas.

Da fronteira do Brasil com a Bolívia, até a cidade de Porto Velho, no estado de Rondônia, o Rio Madeira se caracterizava até setembro de 2011, pela presença de várias corredeiras (Queiroz et al., 2013; Torrente-Vilara, 2009; Torrente-Vilara et al., 2011). A corredeira mais a jusante localizava próxima ao centro urbano de Porto Velho/RO e a partir deste ponto, o Rio Madeira possui 1.345 km de trecho navegável até sua foz no Rio Amazonas. Antes de chegar em sua foz, vários afluentes despejam suas águas no Rio Madeira: em Rondônia, os rios Jamari e Machado e, no estado do Amazonas, os rios Marmelos, Manicoré e Aripuanã.

Devido ao enorme volume de água e à presença de corredeiras, o Rio Madeira foi alvo de estudos com o objetivo de expandir a matriz energética do país por meio da construção de empreendimentos hidrelétricos em seu leito e afluentes (Agência Nacional de Águas, 2012; Ministério de Minas e Energia, 1987, 2007, 2020). Em 1989, na cidade de Candeias do Jamari/RO, entrou em operação a Usina Hidrelétrica Samuel (UHE Samuel), construída no Rio Jamari (tributário localizado na margem direita do Rio Madeira), localizada a aproximadamente 90 km da confluência com entre os dois rios. Em 2008, iniciou-se a construção de duas usinas hidrelétricas (UHEs) no Rio Madeira, em Porto Velho/RO. Os eixos destas duas barragens foram posicionados de modo a aproveitar as formações naturais das Cachoeiras do Caldeirão do Inferno e Santo Antônio.

Entre essas cachoeiras, localizavam-se outras corredeiras que foram submersas com a formação dos reservatórios das UHEs. A Cachoeira do Teotônio, antigo marco geográfico na região, foi submersa pelo reservatório da UHE Santo Antônio (UHE SA), assim como as Cachoeiras do Macaco e Morrinhos. Da mesma forma, a Cachoeira de Jirau foi submersa com a construção da UHE Jirau, cujo eixo foi implantado na Cachoeira Caldeirão do Inferno. O trecho do Rio Madeira onde foram construídas as duas usinas caracteriza-se por um canal encaixado, com margens íngremes que alcançam até 15 metros de altura. A margem esquerda encontrava-se relativamente preservada, apesar da presença de ocupação humana, enquanto a margem direita era mais impactada, apresentando centros urbanos mais desenvolvidos (Queiroz et al., 2013).

A calha principal do Rio Madeira, ambiente lótico com alta concentração de sedimentos em suas águas, abriga uma ictiofauna apta a viver nessas condições. Espécies das ordens Characiformes e Siluriformes são as mais presentes na região (Graça et al., 2025). Observa-se a presença de espécies reofílicas, migradoras (e. g., *Brachyplatystoma rousseauxii* e *Prochilodus nigricans*), além de uma ampla diversidade de guildas tróficas: carnívoras/piscívoras (e.g., *Hydrolycus armatus*, *Calophysus macropterus*), detritívoras (e.g. *Potamorhina latior*), iliófagas (e.g., *Hypostomus* spp.), planctívoras (e.g., *Chaetobranchus flavescens*), invertívoras (e.g. *Triportheus* spp.) e onívoras (e.g. *Colossoma macropomum*). A composição e a abundância relativas dessas guildas variam com o pulso sazonal de cheias (P. B. Silva et al., 2020) e com a heterogeneidade ambiental do trecho devido às barreiras biogeográficas que atuavam como filtros seletivos em parte do ano (Cella-Ribeiro et al., 2017; Ohara, 2018; Torrente-Vilara, 2009; Torrente-Vilara et al., 2011).

As corredeiras do Rio Madeira, submersas pelos reservatórios, atuavam como barreiras biogeográficas naturais à fauna de peixes. Embora esse isolamento não fosse completo (Ohara, 2018; Torrente-Vilara et al., 2011), especialmente durante os períodos de cheia, a barreira à movimentação foi suficiente para moldar a estrutura da comunidade (Rahel, 2007; Torrente-Vilara et al., 2011), resultando em diferenças entre as assembleias a montante e a jusante das corredeiras (Torrente-Vilara et al., 2011). A formação do reservatório, ao conectar os esses trechos a montante e a jusante das corredeiras, pode promover a homogeneização das assembleias de peixes, ao permitir a dispersão de espécies antes restritas a determinados trechos do rio, levando a alterações na composição da ictiofauna (Rahel, 2007).

Além das alterações na conectividade e na estrutura das assembleias de peixes, a implantação de reservatórios está frequentemente associada ao desmatamento de áreas marginais dos corpos d'água, seja para a construção da barragem ou em decorrência de outras atividades antrópicas, como a agricultura. Essas alterações comprometem o equilíbrio do ecossistema aquático e podem resultar em mudanças na estrutura e composição da ictiofauna (A. A. Agostinho et al., 2008; Dala-Corte et al., 2020).

Agostinho et al. (1999) apontam que, após o represamento, os reservatórios passam por um período de instabilidade que pode durar entre 5 e 30 anos. Ao final desse período, a ictiofauna tende a alcançar uma relativa estabilidade, embora mudanças continuem ocorrendo de forma mais lenta. Em reservatórios de acumulação, como aqueles tipicamente construídos na bacia do Paraná, nas regiões Sul e Sudeste do Brasil, os primeiros 15 anos são considerados como “fase de colonização” (A. A. Agostinho et al., 1999), caracterizada por mudanças na riqueza e na abundância da ictiofauna, frequentemente associadas ao aumento da predominância de espécies com alta tolerância às variações ambientais (A. A. Agostinho et al., 2008, 2016; Olden, 2015; Pelicice et al., 2018). Padrão semelhante foi observados por Cella-Ribeiro et al. (2017) no Rio Madeira, onde se registrou aumento na riqueza e abundância de espécies após a formação do reservatório da UHE SA, acompanhado por alterações na estrutura trófica e mudanças na preferência alimentar de algumas espécies. Contudo, mesmo após a submersão da Cachoeira de Teotônio, sua atuação como barreira biogeográfica foi mantida, resultando em dissimilaridade entre as assembleias de peixes nos diferentes trechos do rio (Cella-Ribeiro et al., 2017).

Durante a fase de colonização, a assembleia de peixes tende a ser dominada por espécies de pequeno porte associadas a ambientes lênticos e por espécies com hábitos alimentares generalistas (A. A. Agostinho et al., 2008, 2016; Loures & Pompeu, 2018). Esse padrão resulta em elevada disponibilidade de presas de pequeno porte e de rápido crescimento, favorecendo indiretamente o aumento da abundância de espécies piscívoras na fase inicial de colonização do reservatório (A. A. Agostinho et al., 1999; Perônico et al., 2020). Nessa fase, a ictiofauna tende a se concentrar nas regiões mais preservadas do entorno do reservatório (A. A. Agostinho et al., 1999, 2008, 2016). Nesse contexto, a riqueza e a diversidade da assembleia aumentam progressivamente do barramento em direção à montante, ao longo do eixo longitudinal, bem como da região central para a zona litorânea, nas margens ao longo do eixo transversal, além de variarem entre os diferentes estratos da coluna d'água (A. A. Agostinho et al., 2008; Baumgartner et al., 2018; López-Rodríguez et al., 2024; E. F. Oliveira et al., 2004).

Após a fase de colonização e o término do período de instabilidade do reservatório (aproximadamente em 30 anos), espera-se uma reorganização da assembleia de peixes, caracterizada pelo aumento de espécies onívoras e planctívoras de pequeno porte, pela redução contínua de piscívoros e do tamanho corporal médio dos indivíduos, pela dominância de espécies pelágicas e pela diminuição de espécies bentônicas (A. A. Agostinho et al., 1999). Em reservatórios de acumulação, esta reorganização é desencadeada pelo aporte inicial de matéria orgânica e nutrientes após o enchimento, seguido por processos como assoreamento, aumento da turbidez, alterações térmicas e de oxigenação e eutrofização de longo prazo, os quais degradam a zona litorânea, simplificam os substratos e deslocam a produção primária para a coluna d'água.

Esse processo de envelhecimento do reservatórios (A. A. Agostinho et al., 1999), ou sucessão ecológica em reservatórios, também é influenciado por eventos climáticos que atuam sobre as bacias hidrográficas e aceleram as modificações das características físicas, químicas e biológicas dos corpos d'água (Hansen et al., 2020; Miranda et al., 2020; Miranda & Faucheux, 2022). Nesse contexto, as alterações na estrutura das assembleias de peixes são simultaneamente a matéria-prima e o produto do processo sucessional de reorganização da ictiofauna diante das transformações ambientais (Camara et al., 2024; Fernandez et al., 2024; C. Zhang et al., 2022).

Os efeitos gerados por empreendimentos hidrelétricos podem variar conforme as características do projeto, incluindo a extensão do reservatório, o tipo de barragem e o modo de operação, o que influencia nas alterações dos ecossistemas aquáticos e, por consequência, na ictiofauna (A. A. Agostinho et al., 2008; Baumgartner et al., 2018). Ambas as usinas do Rio Madeira operam com reservatórios reduzidos, aproveitando a vazão natural a montante do rio para gerar energia elétrica, o que as caracteriza como usinas a fio d'água (ANEEL, 2011; Tsuanyo et al., 2023; Venus et al., 2020). A operação de um reservatório a fio d'água é considerada menos prejudicial ao meio ambiente devido ao seu tamanho reduzido quando comparado a um reservatório de acumulação (Tsuanyo et al., 2023).

Diante dos diferentes contextos operacionais e as respostas ecológicas associadas, estudos sobre a ictiofauna em reservatórios, muitas vezes, empregam medidas clássicas de diversidade taxonômica (e.g. riqueza de espécies, diversidade de Shannon-Wiener e equitabilidade de Pielou) para descrever a assembleia de peixes (Schmera et al., 2023), sem considerar a filogenia ou características funcionais das espécies (Ricotta & Pavoine, 2024). Esses índices de diversidade consideram o número de espécies (e.g. riqueza) e a abundância relativa como base para seus cálculos (Magurran, 2013; Melo, 2008; Odum, 1983). No entanto, como diferentes espécies podem ocupar o mesmo nicho em função de semelhanças funcionais e filogenéticas, os índices de diversidade taxonômica fornecem apenas uma visão parcial da estrutura da assembleia (Cianciaruso et al., 2009; Melo, 2008).

Para complementar as informações fornecidas pelos índices de diversidade taxonômica, diversos estudos têm caracterizado o funcionamento das assembleias com base nos atributos funcionais das espécies (Carmona et al., 2021; Perônico et al., 2020; Queiroz-Sousa et al., 2019; C. Zhang et al., 2020, 2022; W. Zhang et al., 2023). Atributos funcionais são características morfológicas, fisiológicas e comportamentais que refletem a forma como indivíduos de uma determinada espécie utilizam o ambiente (Carmona et al., 2021; Díaz & Cabido, 2001; Gomes et al., 2023; Mouillot et al., 2013; Petchey & Gaston, 2006; Sobral, 2021; Violle et al., 2007). Nesse contexto, alterações ambientais tendem a resultar em mudanças nos atributos funcionais das assembleias.

Por estarem diretamente vinculados aos processos ecológicos, os atributos funcionais captam com maior sensibilidade os efeitos das perturbações (Chen et al., 2023; Mouillot et al., 2013; Sagouis et al., 2016), superando as métricas estritamente taxonômicas na detecção de reorganizações da assembleia. Assim, as análises de diversidade funcional da ictiofauna, que avaliam tanto a variedade de características funcionais dentro de cada assembleia local (diversidade alfa) quanto a variação dessas características entre diferentes assembleias (diversidade beta), permitem compreender de forma mais abrangente como a ictiofauna responde às mudanças ambientais (Comte et al., 2016; Hoeinghaus et al., 2006; Li et al., 2024; Mouillot et al., 2013; Queiroz-Sousa et al., 2019).

O Quadro 1 apresenta um exemplo de dissociação entre índices de taxonômica e funcional.

Quadro 1 | Mesma riqueza e mesma abundância, o que muda nas comunidades?

Tomando por exemplo a imagem de Teresa et al. (2021), as comunidades A e B apresentam a mesma riqueza (5 espécies) e a mesma abundância (5 indivíduos), porém divergem quanto à estrutura funcional.

Na comunidade A, os indivíduos exibem atributos semelhantes, o que implica em alta redundância funcional (características semelhantes), forte sobreposição de nichos e baixa ocupação do espaço funcional. Espera-se, portanto, menor diversidade de estratégias ecológicas e menor eficiência de particionamento de recursos.

Já na comunidade B, os atributos são mais diversos, indicando maior complementaridade entre espécies, maior expansão no espaço funcional e menor sobreposição de nichos, com exploração de diferentes micro-habitats e recursos.

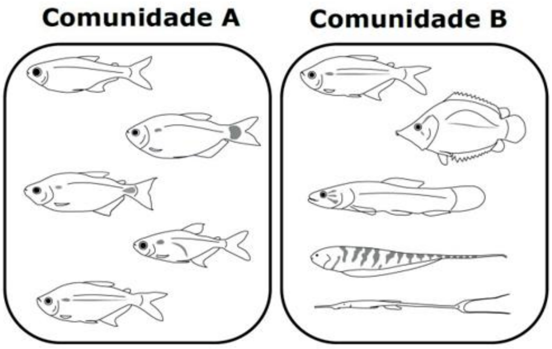

Figura 1. Comunidades A e B possuem a mesma riqueza (5 espécies) e mesma abundância (5 indivíduos), mas são funcionalmente distintas (espécies com atributos funcionais distintos). Imagem obtida em Teresa et al. 2021.

Apesar de equivalentes nos índices taxonômicos clássicos (riqueza e abundância iguais), as comunidades diferem substancialmente quando avaliadas por métricas sensíveis à variação dos atributos funcionais (e.g., FRic, FDis, FEve e outros).

A comparação ilustra a limitação de índices estritamente taxonômicos para captar mudanças na organização da assembleia e reforça a utilidade de abordagens baseadas em características para diagnósticos mais finos sobre estrutura, funcionamento e resposta a perturbações.

Além das alterações associadas à formação e ao envelhecimento do reservatório, a perda de conectividade resultante da presença da barragem também influencia a estrutura das assembleias de peixes. No Rio Madeira, embora as corredeiras funcionassem como filtros biogeográficos naturais, os trechos a montante e jusante não estavam completamente isolados (Ohara, 2018; Torrente-Vilara et al., 2011). Durante os períodos de enchente e cheia do Rio Madeira, o elevado volume de água criava conexões que facilitavam a migração rio acima de espécies migradoras de grande importância comercial e ecológica na região (Ayala et al., 2018; Doria et al., 2012, 2015). Dentre essas espécies, destaca-se *Brachyplatystoma rousseauxii*, reconhecida por realizar uma das mais longas migrações entre peixes de água doce na região Neotropical (Barthem et al., 2017), além de outros bagres migradores (Siluriformes) e de espécies migradoras de outras ordens, especialmente Characiformes, como *Brycon amazonicus* e *Prochilodus nigricans.*

Com o represamento do rio, esse cenário se modificou, uma vez que as espécies migradoras passaram a enfrentar uma barreira física que tem o potencial para interferir nos movimentos migratórios, principalmente no sentido montante (A. A. Agostinho et al., 2008, 2012; Hershey, 2021; Lira et al., 2017; Petrere Júnior, 1996). Para mitigar o potencial efeito de isolamento decorrente da formação do reservatório, uma das estratégias adotadas foi a implantação de um sistema de transposição de peixes (STP).

Esses sistemas consistem em estruturas ou mecanismos projetados para permitir a passagem de peixes através da barragem, tanto em direção à montante quanto à jusante (A. A. Agostinho & Gomes, 2002; C. S. Agostinho et al., 2011; Clay, 1995). O tipo de STP implementado varia de acordo com as características do rio, da ictiofauna e do empreendimento. Entretanto, muitos dos modelos atualmente utilizados têm origem em estudos realizados na América do Norte, desenvolvidos principalmente para salmonídeos (Birnie-Gauvin et al., 2019; Clay, 1995; Kemp, 2016). Para os rios neotropicais, caracterizados por alta diversidade de peixes (Albert et al., 2020; Winemiller et al., 2016), esses modelos podem não ser eficazes (Baigún et al., 2007; Roscoe & Hinch, 2010).

O uso de STPs, originalmente baseados em modelos norte-americanos, não invalida o princípio da transposição de peixes como medida mitigadora, a qual permanece como a estratégia mais utilizada em empreendimentos hidrelétricos no mundo. No entanto, apesar de sua ampla implantação, a avaliação do desempenho e da efetividade dos STPs ainda constitui um grande desafio, sobretudo em função de limitações metodológicas e logísticas associadas ao monitoramento de longo prazo (Bunt et al., 2011, 2016; Kemp, 2016; Lira et al., 2017; Roscoe & Hinch, 2010). Entre os principais desafios destacam-se: (i) a ausência de protocolos padronizados de monitoramento (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017); (ii) as dificuldades em acompanhar o ciclo de vida completo das espécies migratórias (Hahn et al., 2020; Roscoe & Hinch, 2010; A. T. Silva et al., 2018); (iii) as limitações técnicas e tecnológicas para rastrear indivíduos em diferentes escalas espaciais e temporais (Cooke et al., 2012; Jepsen et al., 2002); e (iv) os altos custos operacionais envolvidos.

Como consequência desses desafios no monitoramento, há lacunas de dados sobre a real contribuição ecológica dos STPs para a restauração da conectividade fluvial, tornando incerta sua efetividade para a conservação das populações de peixes (A. A. Agostinho et al., 2012; C. S. Agostinho et al., 2011; Lira et al., 2017; Pelicice & Agostinho, 2012; Petrere Júnior, 1996; Pompeu et al., 2012). Nesse contexto, a superação dessas limitações de monitoramento é condição fundamental para o estabelecimento de evidências consistentes sobre o desempenho dos STPs na restauração da conectividade ecológica.

Por essa razão, estudos recentes têm enfatizado a necessidade de desenvolver ferramentas de monitoramento inovadoras e protocolos padronizados capazes de avaliar de forma mais confiável tanto a eficiência de atração quanto a de passagem

(Castro-Santos, 2009; Eggers et al., 2025; Kemp, 2016; Lira et al., 2017; A. T. Silva et al., 2018) em sistemas de transposição de peixes. Em resposta a essa demanda por metodologias inovadoras, foi desenvolvido, no âmbito do Programa de Pesquisa, Desenvolvimento e Inovação da Agência Nacional Energia Elétrica (ANEEL, PDI nº PD-06683-0320/2020), um dispositivo experimental para captura de peixes inspirado na *fishwheel* (ou "roda de peixes", em tradução livre), tradicionalmente utilizada em rios da América do Norte para a captura não letal de salmões (Begich & Wilburn, 2022; Reid & Ban, 2025; Saito et al., 2021; Zuray, 2005). Assim como os sistemas de transposição, as *fishwheels* foram originalmente desenvolvidas com foco em salmonídeos e empregadas em trechos livres dos rios (Link et al., 1996). O dispositivo utilizado neste estudo, por sua vez, foi projetado especificamente para operar na saída de peixes do STP e capturar a diversidade de espécies presentes no Rio Madeira.

O dispositivo funciona de maneira semelhante a uma roda d'água, sendo impulsionado pela correnteza e capturando peixes que nadam contra o fluxo do rio. Após a captura, os indivíduos são direcionados automaticamente para um compartimento com água, onde permanecem vivos e sem injúrias, permitindo sua identificação, marcação e posterior devolução ao ambiente com mínima interferência. O dispositivo foi desenvolvido para ser instalado na saída de peixes de STPs, com o objetivo de capturar e monitorar os indivíduos que completaram o percurso pelo sistema. Sua instalação é viável em qualquer empreendimento hidrelétrico que possua STP com saída de peixes, possibilitando a padronização de coleta entre os diferentes sistemas e a obtenção de dados consistentes e comparáveis sobre a ictiofauna que efetivamente concluiu o percurso.

Adicionalmente, o dispositivo foi projetado para minimizar o estresse, evitar injúrias e reduzir a mortalidade dos indivíduos durante a amostragem. Por ser aplicável a distintos empreendimentos, a comparação direta dos resultados obtidos permite avaliar a eficiência dos diferentes tipos de STPs, contribuindo para a identificação dos modelos mais adequados à conservação das espécies e fornecendo subsídios à tomada de decisão em projetos futuros de implantação e aprimoramento desses sistemas.

Diante das possíveis alterações na ictiofauna decorrentes da formação do reservatório (Agostinho et al., 1999, 2008, 2016), este estudo teve como objetivo investigar as variações na diversidade taxonômica e funcional da assembleia de peixes após a formação do reservatório da UHE Santo Antônio (CAPÍTULO I)[1]. Considerando que os atributos funcionais das espécies estão diretamente relacionados aos processos ecológicos, hipotetizou-se que, durante a fase de colonização do reservatório, a

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[1] Capítulo I foi elaborado de acordo com as recomendações do periódico *Freshwater Biology*.

diversidade funcional detectaria alterações não captadas pelos índices de diversidade taxonômica (Mouillot et al., 2013; Sagouis et al., 2016).

Adicionalmente, diante da ausência de protocolos padronizados de monitoramento que limitam a avaliação do desempenho de STPs, este estudo teve como objetivo avaliar o desempenho e a operação de um dispositivo para captura de peixes, bem como validá-lo como uma ferramenta replicável para quantificar a passagem de peixes em rios neotropicais (CAPÍTULO II)[2].

Os capítulos I e II foram redigidos em inglês, seguindo o formato e as diretrizes dos periódicos científicos aos quais serão submetidos. Ao final da dissertação, apresenta-se uma discussão e conclusão gerais, em português, integrando e contextualizando os principais resultados.

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[2] Capítulo II foi elaborado de acordo com as recomendações do periódico *Fish and Fisheries*.

CAPÍTULO I

TAXONOMIC AND FUNCTIONAL DIVERSITY OF FISH ASSEMBLAGES IN AN AMAZONIAN RUN-OF-THE-RIVER RESERVOIR

Abstract

1. Hydropower impoundments change hydrological regimes and habitat structure, leading to fish assemblage reorganization. Following reservoir formation, assemblages enter in a colonisation stage, in which species composition and ecological strategies are reshaped in response to the new environment. Trait-based functional diversity provides a clearer and more process-based assessment of how fish assemblages respond to environmental disturbances than taxonomic metrics alone. This study investigates the fish assemblage reorganisation associated with the formation of the Santo Antônio Hydropower Plant reservoir on the Madeira River, Brazil, using both taxonomic and functional diversity metrics to assess assemblage changes.
2. Data correspond to fish monitoring records from the Santo Antônio Hydropower Plant for the period 2009-2019. The dataset was divided into before and after reservoir phases. Fish were sampled using gill nets and longlines. Taxonomic and functional diversity indices were calculated. Relationships between diversity metrics and environmental predictors were evaluated using canonical redundancy analysis and distance-based redundancy analysis, considering river flow, reservoir phase and distance from the dam as explanatory variables. Beta diversity was partitioned into balanced variation and abundance gradients based on Bray-Curtis dissimilarity. Permutation-based ANOVA analyses to test predictor significance and dispersion differences.
3. A total of 311 taxa and 42,117 individuals were recorded. Flow and distance from the dam shaped taxonomic and functional metrics of the assemblage, while reservoir phase affected only functional diversity. Beta-diversity partitioning indicated that assemblage reorganisation was dominated by balanced variation in species abundances, with a minor contribution of abundance gradients. Traits shifted towards generalist feeders with subterminal/supra-terminal mouths, increased demersal/pelagic preferences, more elongated forked-finned mid-sized fishes, and higher proportions of migratory taxa, while piscivorous and carnivorous species and smaller size classes were less represented.
4. Fish assemblage reorganisation reflects responses to environmental modifications associated with damming. Following impoundment, the assemblage showed colonisation stage patterns: an increase in generalists taxa and fluctuating richness/abundance. However, an expected rise in piscivores/carnivores did not occur, and size structure shifted from small-bodied towards predominantly mid- and larger-bodied classes. Notably, before the impoundment fluctuations in richness and abundance were already present,

possibly due the river's strong seasonality. The predominance of balanced abundance variation indicates compositional reorganisation driven by reciprocal changes in species abundances, without evidence of assemblage homogenisation. These findings underscore the importance of integrating multiple dimensions of diversity to detect assemblage change, with functional diversity better expressing the changes in assemblage structure. As the reservoir remains in the colonisation stage coupled with climate change, further assemblage shifts are expected.

5. Incorporating functional analyses alongside taxonomic diversity indices enhances the ability of monitoring programs to detect early assemblage reorganisation in response to environmental disturbances. Adopting these approaches is essential for developing effective management and conservation actions in tropical river systems affected by dams and environmental change.

Keywords: neotropical fishes; trait-based analysis; hydropower plant; tropical freshwater ecosystems; Madeira River

I. Introduction

Ecological studies of fish assemblages in reservoirs often aim to understand how local changes in biodiversity, including shifts in species composition, functional traits, or trophic interactions, translate into broader ecological impacts following impoundment (Arantes et al., 2019; Cella-Ribeiro et al., 2017; A. G. Oliveira et al., 2018; Perônico et al., 2020; W. Zhang et al., 2023). Reservoirs are complex and dynamic systems that evolve based on the specific characteristics of each project (Agostinho et al., 1999, 2008; Perbiche-Neves & Camargo, 2018). Although differences in design, location, and ecological context lead to distinct effects on fish assemblages (Camara et al., 2024; Fernandez et al., 2024; C. Zhang et al., 2022), certain ecological changes tend to recur across reservoir systems.

After river damming, alterations in flow regimes, habitat structure, water quality, and connectivity are the main drives of changes in fish assemblage over time (Gogola et al., 2022; Loures & Pompeu, 2019; Miranda & Faucheux, 2022; Sagouis et al., 2016). The first 15 years following reservoir formation are often referred to as the "colonisation stage", a period marked by fluctuations in fish richness and abundance, with a general trend toward increased dominance of species tolerant to environmental variability (Agostinho et al., 1999). During this stage, assemblages are typically dominated by small-bodied, dietary generalist species with traits associated with tolerance to environmental variability, which confer ecological advantages under lentic conditions. The increased availability of abundant, fast-growing prey in turn supports a higher abundance of piscivorous species. Spatially, fish assemblages relatively preserved areas within the reservoir, with species richness and diversity increasing both longitudinally - towards upstream sectors where the system gradually transitions from lentic to lotic conditions -, and laterally, from the central basin towards littoral zones characterised by greater physical complexity and substrate availability (Agostinho et al., 2016; E. F. Oliveira et al., 2004).

Although taxonomic metrics based on species richness and relative abundance are valuable for describing the effects of damming on fish assemblages (Agostinho et al., 2008, 2016; E. F. Oliveira et al., 2004; Orsi & Britton, 2014; Petrere Júnior, 1996), they may overlook structural changes in assemblage composition and underrepresent shifts in functional diversity (Arantes et al., 2019; Carmona et al., 2021; Mouillot et al., 2013; Ricotta & Pavoine, 2024; Sagouis et al., 2016; W. Zhang et al., 2023), which are essential for understanding the long-term consequences of environmental change on biodiversity. To address these oversights, comparative measures of species gains and losses, combined with functional diversity indices (Carmona et al., 2021; Laliberté & Legendre, 2010; Ricotta et al., 2016; Ricotta & Pavoine, 2025; Teichert et al., 2018; Villéger et al., 2010, 2017), provide a more complete view of how fish assemblages reorganise and how these shifts influence ecological processes, including trophic dynamics and ecosystem functioning (Arantes et al., 2019; Cianciaruso et al., 2009; Melo, 2008; Mouillot et al., 2013; Sagouis et al., 2016).

Reservoir formation, particularly in large storage reservoirs, has frequently been associated with functional simplification of fish assemblages through the decline or local extinction of specialized species (e.g., large-bodied migrants, rheophilic taxa) (Agostinho et al., 1999, 2008, 2016; Fráguas & Pompeu, 2021; Loures & Pompeu, 2019; A. G. Oliveira et al., 2018; E. F. Oliveira et al., 2004). In contrast, early post-impoundment phases may be characterised by temporary increases in piscivorous species (Agostinho et al., 1999), driven by elevated prey availability, followed by long-term declines as productivity decreases and habitat complexity is reduced (Arantes et al., 2019; Perônico et al., 2020; C. Zhang et al., 2020). Over longer temporal scales in storage reservoirs, these compositional shifts reduce the diversity of ecological strategies within the assemblage, leading to a constriction of the functional space and a decrease in functional richness. Moreover, increased trait convergence, driven by the dominance of species with similar ecological roles, can further reduce functional divergence and regularity (Queiroz-Sousa et al., 2019; C. Zhang et al., 2022). In storage reservoirs, this structural homogenisation may compromise essential ecosystem processes and limit the resilience of fish assemblages to future disturbances, even when taxonomic richness remains unchanged (Li et al., 2023; Perônico et al., 2020). These patterns are context-dependent and have been primarily reported for storage reservoirs, with their magnitude varying according to hydrological operation and regional setting.

Therefore, understanding how fish assemblages respond to disturbance requires frameworks that prioritize ecological roles over taxonomic lists. In this regard, functional diversity has proven to be more effective than classical taxonomic indices in detecting changes in fish assemblages caused by environmental disturbances (Mouillot et al., 2013; Sagouis et al., 2016). Because functional traits are directly linked to ecological processes, they provide a more process-based means of assessing shifts in fish communities (Carmona et al., 2021; Chen et al., 2023; Sagouis et al., 2016). Consequently, analyses based on functional diversity offer a clearer understanding of how fish assemblages respond to environmental disturbances (Comte et al., 2016; Hoeinghaus et al., 2006; Mouillot et al., 2013; Queiroz-Sousa et al., 2019). In this context, assessing functional diversity is particularly relevant for large Neotropical rivers subject to impoundment, where environmental disturbances often result in changes in fish assemblage structure.

Among these systems, the Madeira River is one of the largest tributaries of the Amazon River, harbouring around 1.000 fish species (Graça et al., 2025; Queiroz et al., 2013). The regional assemblage is dominated by Characiformes and Siluriformes (Graça et al., 2025). Despite the high abundance of fish and the large number of species, endemism remains low due to the widespread distribution of species across the Amazon basin (Albert et al., 2020; Dagosta & Pinna, 2019). Madeira River's discharge follows a pronounced annual cycle, with a wet season (November-April) and dry season (May-October) (Espinoza et al., 2019; Graça et al., 2025). This pronounced seasonal cycle regulates essential ecological processes, especially fish reproduction and migration (Barthem et al., 2017; Godinho & Kynard, 2009; Torrente-Vilara et al., 2011), and shapes assemblage composition and abundance (Silva et al., 2020).

The Madeira Hydropower Complex, comprising the Santo Antônio and Jirau Hydropower Plants (HPPs), began construction in 2008 and operates under a run-of-the-river regime, which allows limited flow regulation on a daily or weekly basis, generating energy while preserving most of the river's natural flow (Tsuanyo et al., 2023; Venus et al., 2020). Due to its low water retention, run-of-the-river reservoirs are considered less environmentally impactful than conventional storage reservoirs (Tsuanyo et al., 2023).

This study investigates taxonomic and functional reorganisation of fish assemblages associated with reservoir formation by the Santo Antônio Hydropower Plant (SA-HPP) on the Madeira River, Brazil. Given that species traits are directly tied to ecological processes (Chen et al., 2023; Mouillot et al., 2013; Sagouis et al., 2016), a trait-based perspective offers a more precise framework than taxonomic indices alone for diagnosing assemblage reorganisation. It is therefore hypothesised that functional diversity may provide an earlier and clearer response to assemblage changes during the colonisation stage compared to taxonomic metrics.

II. Methods

Study area

The Madeira River main channel extends for 3.240 km, of which 1.425 km is located within the Brazilian states of Rondônia and Amazonas. The main channel is impounded by two dams situated in the municipality of Porto Velho, in Rondônia State, Brazil. This study was conducted in the reservoir of the Santo Antônio Hydropower Plant (SA-HPP).

Santo Antônio Hydropower Plant is a run-of-the-river hydropower plant that utilizes the natural flow of the Madeira River to generate electricity. The reservoir is relatively small (Tsuanyo et al., 2023), covering 546 km², with a volume of 2.888 × 10^6

m³, and depths ranging from 11.5 m to 27.5 m (Eletrobras, 2025). The operation water elevation ranges from 70.5 m and 71.3 m above sea level (a.s.l.). The second dam of the Madeira River Hydropower Complex, Jirau Hydropower Plant (Jirau-HPP), is located 100 km upstream of SA-HPP and directly discharges its waters into the SA-HPP reservoir. Thus, the SA-HPP reservoir area is constrained between its own dam downstream and the Jirau-HPP dam upstream.

Before impoundment, the area now occupied by the SA-HPP reservoir contained the rapids of Macaco, Teotônio and Morrinhos (Figure 1). Collectively, these rapids acted as effective biogeographical filters (Rahel, 2007; Torrente-Vilara et al., 2011), shaping species distribution and ecological connectivity. Among them, the Teotônio Rapids, the steepest and deepest was located 14 km upstream from the dam structure and functioned as a strong ecological filter, partially restricting fish movement between river sections (Cella-Ribeiro et al., 2017; Torrente-Vilara et al., 2011). In contrast, Macaco and Morrinhos, situated 5 km and 46 km from the dam, respectively, were less steep and allowed both navigability and fish movement.

Alongside the main channel, two tributaries flow into the SA-HPP reservoir: the Jaci-Paraná River on the right bank and the Jatuarana Stream on the left.

Figure 1 | Location of the main rapids and the Madeira Complex along the Madeira River near Porto Velho, Rondônia, Brazil. Panel A shows the river course in 2007, prior to reservoir formation, and Panel B shows the area after impoundment. Gray markers indicate the dam structures of SA-HPP and Jirau HPP. White markers indicate the former locations of the Macaco, Teotonio, and Morrinhos Rapids, now submerged within the Santo Antônio reservoir. Blue markers indicate the two tributaries that flow into the reservoir and were included in the monitoring program of SA-HPP. Image source: Google Earth © 2025 Airbus Retrieved Sep 15, 2025, from https://earth.google.com.

Dataset

Data were obtained from monitoring records associated with the Santo Antônio Hydropower Plant, including fish assemblage and environmental data. Fish sampling was carried out from April 2009 to September 2019 as part of the SA-HPP´s Ichthyofauna Monitoring Program, conducted by foundations, public-interest civil society organizations (OSCIPs), and private contractors, in accordance with Brazilian environmental licensing requirements and the corresponding permits (CGFAP Permit nº 51/2009; IBAMA/DILIC Permits nº 109/2011, 83/2012, 697/2016, and 697/2016 - RET 1/2018).

From 2009 to 2010, sampling was conducted monthly at three locations: the Jaci-Paraná confluence with the Madeira River (JAF), an upstream site in the Jaci-Paraná River (JAM), and the Jatuarana confluence with the Madeira River (JAT). In 2011, the frequency changed to bimonthly. After reservoir formation in 2012, three new sites were added in the upper (RSA), middle (RSM), and lower (RSB) areas of the reservoir (Figure 2). From 2017 onward, sampling shifted to bimonthly or quarterly. In 2018 and 2019, it became exclusively quarterly. All hydrological seasons were represented in the dataset each year, although the sampling interval varied over time, ranging from monthly to bimonthly or quarterly surveys (Table 1).

Table 1 | Number of fish sampling campaigns conducted at each sampling site across the different years of the study. Reservoir Phase indicates whether the year corresponds to the period before or after reservoir formation. JAF (Jaci-Paraná confluence with the Madeira River), JAM (upstream site in the Jaci-Paraná River), JAT (Jatuarana confluence with the Madeira River), RSA (upper reservoir area), RSM (middle reservoir area), and RSB (lower reservoir area near the dam). Sites RSB, RSM, and RSA were included in the monitoring program following reservoir formation in 2012, therefore, no data were collected at these locations prior to the reservoir filling. Sampling campaigns per year is the total number of sampling campaigns conducted at each site in the given monitoring year.

Reservoir Phase	Year	Sampling Sites						Sampling campaigns per year
		JAT	JAF	JAM	RSB	RSM	RSA	
Before	2009	9	9	9				27
	2010	8	8	8				24
	2011	6	6	6				18
After	2012	6	6	6	6	6	6	36
	2013	5	5	5	5	5	5	30
	2014	6	6	6	6	6	6	36
	2015	6	6	6	6	6	6	36
	2016	6	6	6	6	6	6	36
	2017	4	4	4	4	4	4	24
	2018	5	5	5	5	5	5	30
	2019	3	3	2	3	3	3	17

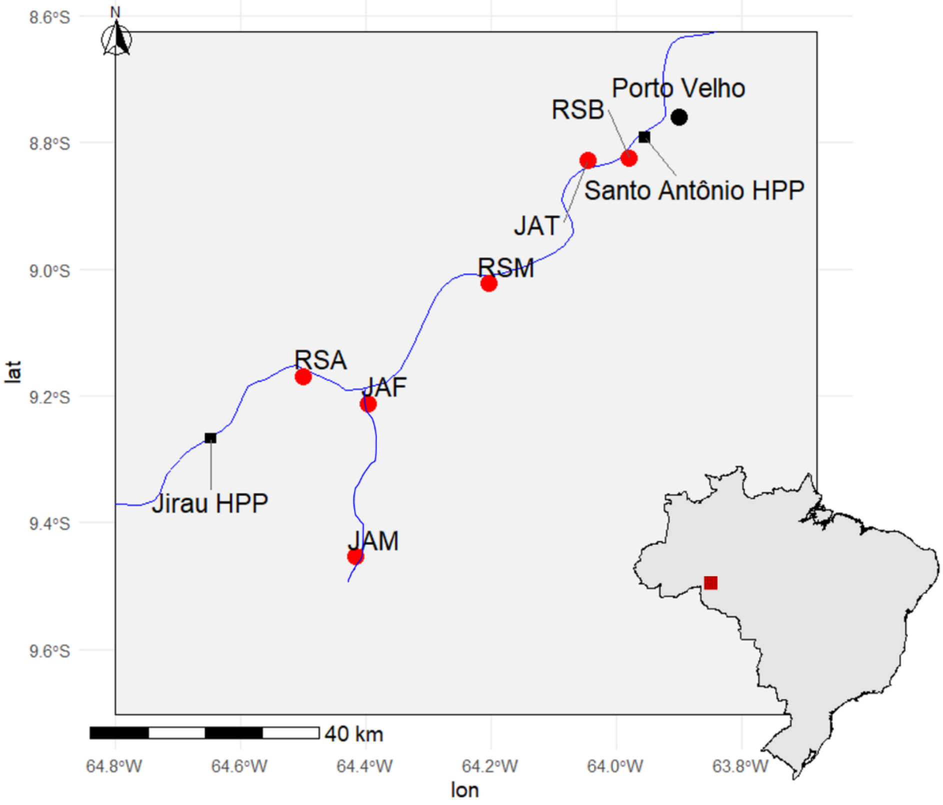

Figure 2 | The red dots represent the locations where fish samples were collected during the monitoring period: JAF (Jaci-Paraná confluence with the Madeira River), JAM (upstream site in the Jaci-Paraná River), JAT (Jatuarana confluence with the Madeira River), RSA (upper reservoir area), RSM (middle reservoir area), and RSB (lower reservoir area near the dam). The black square symbols indicate the locations of the Santo Antônio and Jirau HPPs. The black circle marks the downtown area of Porto Velho city. The Madeira River network is depicted in blue, providing context to the sampling locations. The inset map displays the highlighted region within Brazil for spatial reference.

The dataset included data from samples caught using gill nets (mesh sizes from 3, 4, 5, 6, 7, 8, 9, 10, 12, 14, 16, 18, and 20 cm opposite knots length) and long lines with 10 hooks (size 12/0). Gill nets were 20 m in length, with variable heights depending on the net panel, resulting in a total deployed net area of approximately 480 m² per sampling site. Both gears were deployed in littoral areas, inspected at four-hour intervals and retrieved after 24 hours of exposure at each site. Long lines were baited with locally occurring fish species (e.g., *Potamorhina* spp.), anchored at the shoreline and extended towards the main river channel, with a sinker placed at the distal end to maintain the gear in position.

Only fish genera/species of Actinopterygii with valid scientific names, following the taxonomic classification of Eschmeyer's Catalog of Fishes Database (Fricke et al., 2025), were considered in the analyses. Records that did not correspond to valid biological entities, or that reflected uncertainties and potential misidentifications, were excluded. Additionally, data on monthly river discharge over the years were sourced from the Reservoir Performance Monitoring System database - SAR (Agência Nacional de Águas, 2025).

Functional traits data

Eight functional traits related to feeding, locomotion, and reproduction were selected based on previous studies (Erős et al., 2012; Luck et al., 2012; Villéger et al., 2017; Terra et al., 2016; W. Zhang et al., 2023). The trait dataset comprised seven categorical traits (feeding behaviour, mouth position, habitat preference, body shape, caudal fin type, reproductive period, and migration behaviour) and one continuous trait (maximum body length) (Table 2).

The functional traits data were compiled from available literature (Alcalá & Betancourt, 2022; Carolsfeld et al., 2003; Cella-Ribeiro et al., 2016; Ohara et al., 2017; Queiroz et al., 2013; Reis et al., 2003; Röpke et al., 2024; Santos et al., 2006) and the FishBase database (Froese & Pauly, 2024). For species lacking trait information in the literature, data from congeneric species were used. When genus-level data were also unavailable, traits were inferred based on the patterns observed among other species of the same family recorded in this study. The functional traits assigned to each species are provided in Supporting Information (Appendix A).

Table 2 | Functional categories, functional traits, and the type or range of trait values used to describe the ecological strategies of fish species recorded in the study. Traits were selected based on their relevance to locomotion, feeding and reproductive behaviour.

Functional Category	Functional Traits	Type or range
Feeding	Feeding preference	Detritivorous; Hematophagous (blood-feeding); Herbivorous; Invertivorous (feeding exclusively on invertebrates); Carnivorous; Omnivorous; Piscivorous; and Planktivorous
	Mouth orientation	Terminal; Supra-terminal; Sub-terminal
	Water column position	Benthopelagic (lives and feeds near the bottom as well as in the water column); Demersal (lives close to the riverbed); Pelagic (lives and swims in the open water column of the river)
Locomotion	Body shape	Elongated; Anguilliform (eel-like); Rounded; Depressed (flattened vertically, from top to bottom); Fusiform (spindle-shaped, tapered at both ends)
	Caudal fin types	Rounded; Absent; Emarginate; Forked; Lunate; Pointed; Truncated
	Maximum body length	2.73 - 450 cm
Reproduction	Breeding season	Rainy season; Dry season
	Migration Behaviour	Migratory fish; Non-migratory fish

Data analysis

Sampling events were treated independently due to variation in sampling frequency across years and hydrological seasons, and due to the inclusion of three additional sampling sites only after reservoir filling in 2012.

The dataset was divided into two periods: (i) before reservoir formation (April 2009-November 2011); and (ii) after reservoir formation, during full hydropower operation (March 2012-September 2019). The reservoir filling occurred from September 2011 to January 2012.

Classical taxonomic diversity metrics, total fish abundance, species richness, Shannon-Wiener diversity, Simpson dominance, and Pielou's evenness were calculated for each sampling event.

Functional diversity was estimated using a multidimensional trait-based framework calculated using the *mFD* package (v1.0.7) (Magneville et al., 2022). Species traits were log-transformed when necessary (e.g., body size) to reduce the influence of extreme values. Species were grouped into Functional Entities (FEs) based on trait identity, and pairwise functional distances among FEs were calculated using the Gower metric with standardized trait scaling. A Principal Coordinates Analysis (PCoA) was then applied to construct a functional space, and the optimal number of dimensions was selected based on mean absolute deviation (MAD) plots. Eight functional diversity indices (Table 3) were calculated for each sampling event.

To reduce the influence of dominant taxa and meet assumptions of linearity in multivariate analyses, species abundance data used in ordination and composition analyses were Hellinger-transformed (Legendre & Gallagher, 2001) using the *vegan* package (Oksanen et al., 2001). River flow and sites distance from the dam structure were z-score standardized (mean-centred and scaled by the standard deviation) (Legendre & Legendre, 2012) to ensure comparability among predictors and prevent scale bias in constrained ordination models.

Redundancy analysis (RDA) was conducted separately for taxonomic and functional diversity indices, using the *vegan* package. Sampling events were excluded when functional indices could not be calculated due to low species richness or unmet mathematical requirements. In both analyses, river flow, reservoir phase (before and after filling) and distance from the dam structure were used as environmental predictors. In addition, diversity indices (taxonomic and functional) were included as response variables to assess their specific associations with the environmental predictors. Analyses of variance (ANOVA, with 999 permutations) were applied to each RDA model to assess the significance of the overall model, and predictors.

Assemblage beta diversity was assessed using Bray-Curtis dissimilarity (abundance-based). Abundance data was Hellinger-transformation to reduce the influence of dominant and rare species (Legendre & Gallagher, 2001). Beta diversity was partitioned into balanced variation (d_BC-bal), abundance gradient (d_BC-gra), and total dissimilarity (d_BC) components (Baselga, 2013) using the *betapart* package (Baselga et al., 2023). The abundance gradient component required Cailliez correction (Legendre & Legendre, 2012) for valid analysis. Permutational Multivariate Analysis of Variance (PERMANOVA) was applied to assess environmental predictors effects, followed by Permutational Analysis of Multivariate Dispersions (PERMDISP) to verify homogeneity of group dispersions (Anderson, 2006; Anderson et al., 2006). Distance-based redundancy analysis (dbRDA) with permutation-based ANOVA (999 permutations) was used to examine environmental predictors' influences on each beta diversity component.

Table 3 | Description of the functional diversity indices calculated in this study. Each index captures a distinct aspect of functional structure within fish assemblages. The indices are grouped according to the functional dimension they represent richness (extent of functional space), divergence (distribution and extremity of traits) and regularity (evenness of trait distribution).

Abbreviation	Names	Functional Dimension	Description	References
FE	Functional Entity richness	Richness	Groups of species sharing the exact same combination of trait values.	Magneville et al. (2022)
FRic	Functional Richness	Richness	Measures the volume of functional space occupied by the assemblage.	Villéger et al. (2008)
FDiv	Functional Divergence	Divergence	Quantifies whether species with extreme or specialized traits dominate functional space.	Villéger et al. (2008)
FDis	Functional Dispersion	Divergence	Weighted mean distance of each species to the community centroid in trait space.	Laliberté & Legendre (2010)
FEve	Functional Evenness	Regularity	Assesses how evenly trait values are distributed within the functional space.	Villéger et al. (2008)
FSpe	Functional Specialization	Divergence	Measures the degree of specialization based on the distance from the community centroid.	Mouillot et al. (2013)
FOri	Functional Originality	Divergence	Reflects the uniqueness of a species' trait combination within the assemblage.	Mouillot et al. (2013); Kondratyeva et al. (2019)
FMPD	Functional Mean Pairwise Distance	Divergence	Mean pairwise trait distance between all species in the assemblage.	Swenson (2011); Webb et al. (2002)
FNND	Functional Nearest Neighbor Distance	Divergence	Mean distance between each species and its nearest neighbour in trait space.	Webb et al. (2002); Pavoine et al. (2009)

Trait composition was described using relative frequencies of individuals per functional category, thereby standardising the data by total abundance and allowing comparisons between phases despite differences in sampling effort and total number of individuals. Graphical representations were used to explore associations between changes in the relative frequency of individuals across functional trait categories.

All analyses were performed using the R Studio (version 2025.05.0 Build 496) (R Core Team, 2024). The statistical significance criterion used for all tests was $p \leq 0.05$.

III. Results

A total of 42.117 individuals from 311 taxa (species and morpho-species), distributed across 7 Orders and 35 Families (Appendix B), were sampled between 2009 and 2019. Of these, 40 taxa were identified only to genus level, an inevitability in the Madeira River, where intense predation often yields partially consumed specimens that preclude species-level identification. Characiformes and Siluriformes dominated both phases, representing 93.7% of individuals before and 92.0% after reservoir formation. Thirty-three taxa were recorded exclusively before reservoir formation (8 identified only to genus level) and 100 exclusively after (25 identified only to genus level). Notably, all 33 genera documented before reservoir formation were also detected afterward.

Before reservoir formation, the confluence between Jaci-Paraná and Madeira rivers (JAF) exhibited the highest values of abundance, richness, Shannon-Wiener diversity and Simpson dominance, while the Jatuarana stream (JAT) and an upstream site in the Jaci-Paraná River (JAM) frequently showed the lowest values. These patterns reflect sampling events analysed independently, accounting for variation in sampling frequency across years and hydrological seasons. Despite the occurrence of high and low values, taxonomic diversity indices showed broad overlap among sampling events, indicating substantial intra-site variability prior to impoundment. After impoundment, highest values of abundance and richness were still recorded at JAF. The middle reservoir area (RSM), the lower reservoir area (RSB), and upstream site in the Jaci-Paraná River (JAM) had the maximum values of Shannon-Wiener diversity, Pielou's evenness and Simpson dominance, respectively. In December 2018, only a single individual was captured at JAT, resulting in the lowest value recorded after reservoir formation (Table 4). These extreme values occurred within a context of increased sampling effort after impoundment and should therefore be interpreted as event-specific rather than representative of general temporal trends.

Concerning functional diversity indices, the tributaries sites (JAM, JAF and JAT) each recorded maxima in at least one functional diversity index before impoundment. However, minima values were concentrated primarily at JAT and JAF. After impoundment, JAM, JAT and RSA each exhibited both maximum and minimum values across functional diversity indices (Table 5). The presence of extreme functional values within the same sites indicates high temporal variability in functional structure, rather than a unidirectional response to reservoir formation.

Temporal variation in taxonomic and functional diversity indices are presented in Appendix C and Appendix D, respectively.

Table 4 | Extreme values - maximum (Max.) and minimum (Min.) -, mean and standard deviation (SD) of taxonomic diversity indices across reservoir phases (Before and After impoundment), with the specific sites and dates where these values were recorded. The indices comprise Abundance, Pielou's Evenness, Richness, Shannon-Wiener Diversity, and Simpson Dominance, with separate columns indicating the Reservoir Phase (Before/After), Maximum Value (highest recorded value for each index), Site (sampling location where the extreme value occurred), Date (month/year of observation), and Minimum Value (lowest recorded value). The sampling sites include: JAF (Jaci-Paraná confluence with the Madeira River), JAM (upstream site in the Jaci-Paraná River), JAT (Jatuarana stream confluence with the Madeira River), RSM (middle reservoir area), and RSB (lower reservoir area near the dam). Sites RSB and RSM were included in the monitoring program following reservoir formation in 2012, therefore, no data were collected at these locations prior to the reservoir filling.

Taxonomic Diversity indices	Reservoir phase	Max.	Mean	SD	Min.	Site (max)	Date (max)	Site (min)	Date (min)
Abundance	Before	1006	139	175.341	6	JAF	Jun/2011	JAM	Feb/2010
	After	1533	61	189.316	1	JAT	Mar/2012	JAT	Dec/2018
Richness	Before	82	27	17.438	4	JAF	Jun/2011	JAM	Mar/2010
	After	65	22	12.862	1	JAF	Nov/2012	JAT	Dec/2018
Shannon-Wiener	Before	3.58	2.633	0.657	0.9	JAF	Jun/2010	JAM	Mar/2010
	After	3.42	2.51	0.568	0	RSM	Jun/2015	JAT	Dec/2018
Pielou's Evenness	Before	1	0.829	0.123	0.4	JAM	Feb/2010	JAT	Apr/2010
	After	1	0.848	0.113	0.3	RSB	Apr/2015	JAM	Jun/2019
Simpson Dominance	Before	0.96	0.887	0.126	0.4	JAF	Jun/2010	JAT	Apr/2010
	After	0.96	0.88	0.129	0	JAM	Jul/2016	JAT	Dec/2018

Table 5 | Extreme values - maximum (Max.) and minimum (Min.) -, mean and standard deviation (SD) of functional diversity indices across reservoir phases (Before and After impoundment), with the specific sites and dates where these values were recorded. The indices comprise Functional Entity richness (FE); Functional Richness (FRic), Functional Evenness (FEve), Dispersion (FDis), Functional Divergence (FDiv), Functional Specialization (FSpe), Mean Pairwise Distance (FMPD), Mean Nearest Neighbour Distance (FNND), and Functional Originality (FOri). The sampling sites include: JAF (Jaci-Paraná confluence with the Madeira River), JAM (upstream site in the Jaci-Paraná River), JAT (Jatuarana stream confluence with the Madeira River), RSA (upper reservoir area), and RSM (middle reservoir area). Sites RSM, and RSA were included in the monitoring program following reservoir formation in 2012, therefore, no data were collected at these locations prior to the reservoir filling.

Functional diversity indices	Reservoir phase	Max.	Mean	SD	Min.	Site (max)	Date (max)	Site (min)	Date (min)
FE	Before	81	27	17.06	8	JAF	Jun/2011	JAT	Jan/2010 Feb/2010
	After	65	23	12.265	7	JAF	Feb/2013	JAF	May/2014
					7			JAM	Apr/2013 Apr/2016 Jun/2017
					7			RSA	Mar/2015 Apr/2017
					7			RSM	Apr/2012
Fric	Before	0.394	0.064	0.1	0	JAF	Jun/2011	JAT	Jun/2009
	After	0.367	0.049	0.072	0	JAM	Dec/2016	RSM	Apr/2012
Feve	Before	0.833	0.622	0.076	0.506	JAT	Jan/2010	JAT	Oct/2010
	After	0.887	0.649	0.084	0.47	JAM	Feb/2017	JAT	Mar/2012
Fdis	Before	0.834	0.655	0.107	0.323	JAM	Apr/2011	JAT	Jun/2009
	After	0.862	0.677	0.086	0.377	RSA	Jun/2019	JAM	Jun/2017
Fdiv	Before	0.933	0.799	0.066	0.591	JAM	Sep/2009	JAT	Feb/2010
	After	0.947	0.832	0.053	0.642	RSA	Feb/2017	JAT	Oct/2014
Fmpd	Before	0.772	0.652	0.047	0.537	JAM	Apr/2011	JAF	Apr/2010
	After	0.793	0.669	0.044	0.528	JAT	Jun/2019	RSM	Apr/2012
Fnnd	Before	1.205	0.536	0.229	0.212	JAT	Dec/2009	JAT	Feb/2011
	After	1.31	0.594	0.226	0.118	JAM	Apr/2013	JAT	Apr/2012
Fori	Before	0.473	0.191	0.072	0.081	JAM	Apr/2011	JAT	Jun/2009
	After	0.411	0.179	0.073	0.048	JAT	Jun/2019	JAT	Mar/2018
Fspe	Before	0.7	0.552	0.053	0.471	JAM	Apr/2011	JAF	Feb/2011
	After	0.731	0.605	0.041	0.475	JAT	Jun/2019	JAM	Jun/2017

Madeira River´s flow in the reservoir showed a pronounced seasonal cycle (Figure 3), with wet-season peaks (March-May) and dry-season minima (September-November), alongside strong interannual variability (notably high peaks in 2014-2015 and 2019).

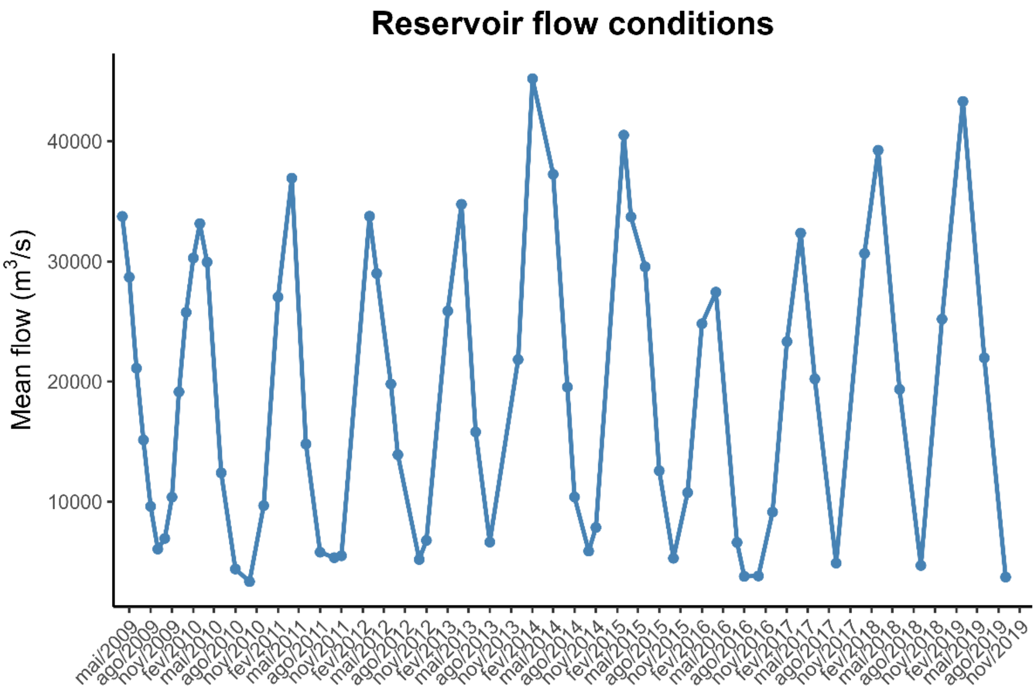

Figure 3 | Temporal variation of the monthly mean flow (m³/s) in the Madeira River within the SA-HPP reservoir area from 2009-2019. Values aggregated by month.

Environmental variables combined were significant for taxonomic diversity (F = 13.08, p = 0.001) and accounted for 10.16% of the variation. River flow (F = 30.37, p = 0.001) and distance from the dam structure (F = 6.63, p = 0.003) had significant effects, whereas reservoir phases did not (F = 2.24, p = 0.086) (Figure 4). were positively associated with distance from the dam structure and negatively associated with river flow, while Pielou's evenness showed a stronger association with reservoir phase.

Environmental predictors accounted for 7.79% of the variation in functional diversity (F = 13.16, p = 0.001). All variables contributed significantly: river flow (F = 19.77, p = 0.001), distance from the dam structure (F = 10.91, p = 0.001), and reservoir phase (F = 8.79, p = 0.001) (Figure 5). Functional entity richness (FE), originality (Fori) and richness (Fric) were positively associated with distance from the dam structure. Functional dispersion (FDis), mean pairwise distance (Fmpd), and specialization (Fspe) were positively associated with reservoir phase, while evenness (Feve), divergence (Fdiv) and nearest neighbour distance (Fnnd) were positively associated with river flow.

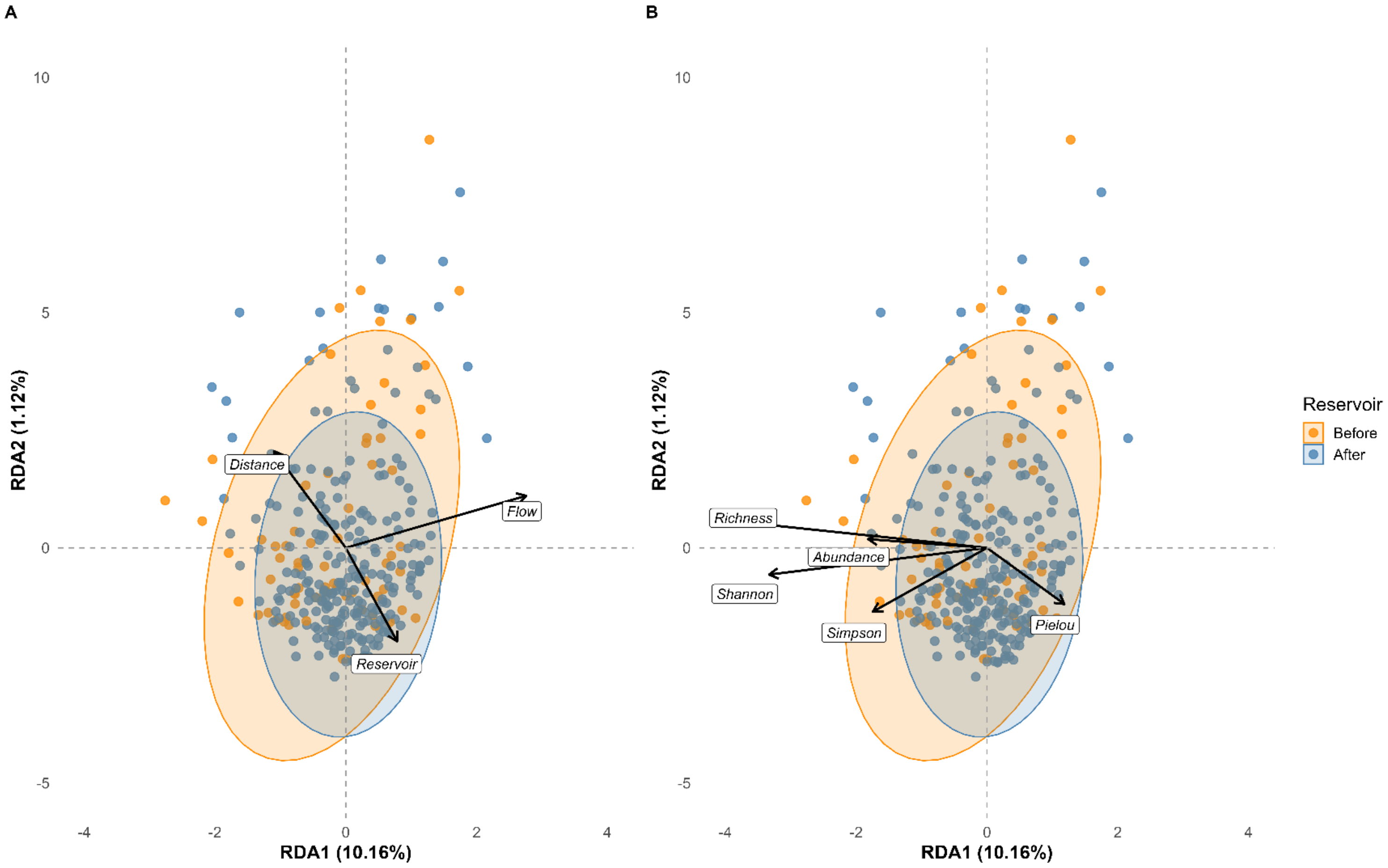

Figure 4 | Redundancy Analysis (RDA) of taxonomic diversity. Panel A: RDA constrained by river flow (Flow), distance from the dam (Distance), and reservoir phase (Before and After impoundment). Panel B: RDA constrained by taxonomic indices – abundance, richness, Shannon-Wiener diversity (Shannon), Simpson's dominance (Simpson) and Pileou's evenness (Pielou). Arrows represent environmental predictors (panel A) or taxonomic diversity indices (Panel B) scaled by their correlations with the ordination axes. Points represent sampling events, colored by reservoir phase: orange for Before and blue for After. Ellipses denote 95% confidence regions for each phase. Axis labels indicate the percentage of variance explained by each constrained axis.

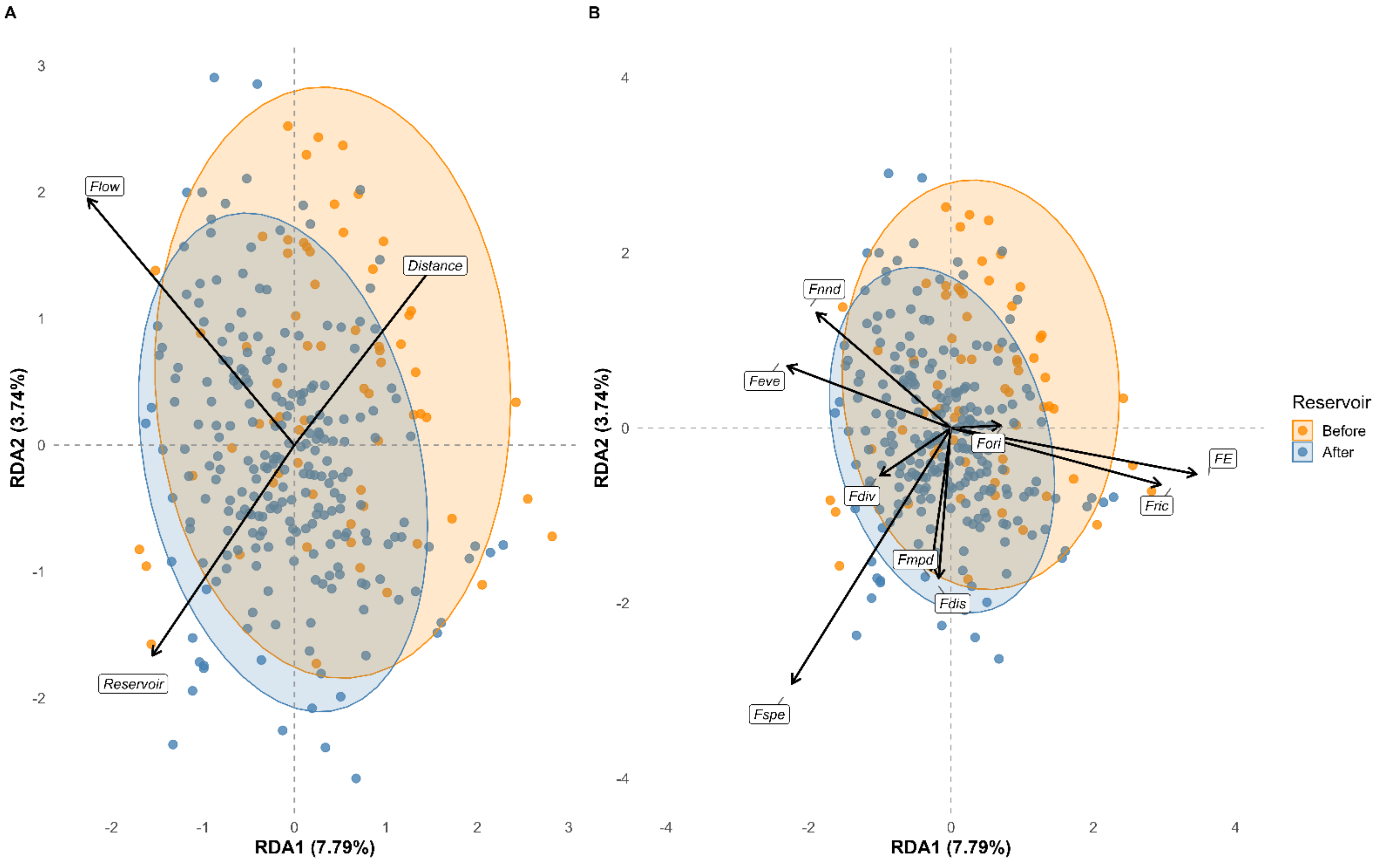

Figure 5 | Redundancy Analysis (RDA) of functional diversity. Panel A: RDA constrained by river flow (Flow), distance from the dam (Distance), and reservoir phase (Before and After impoundment). Panel B: RDA constrained by functional indices – Functional Entity richness (FE), Phylogenetic Dispersion (FDis), Mean Pairwise Distance (Fmpd), Mean Nearest Neighbor Distance (Fnnd), Phylogenetic Evenness (Feve), Phylogenetic Richness (Fric), Phylogenetic Divergence (Fdiv), Phylogenetic Originality (Fori), and Phylogenetic Specialization (Fspe) (Panel B). Arrows represent environmental predictors (on the left) or functional diversity indices (Panel A) scaled by their correlations with the ordination axes. Points represent sampling events, colored by reservoir phase: orange for Before and blue for After. Ellipses denote 95% confidence regions for each phase. Axis labels indicate the percentage of variance explained by each constrained axis.

Partitioning of beta diversity based on Bray-Curtis dissimilarity indicated that 95.9% of the variation between reservoir phases was explained by the balanced variation in abundance component (d_BC-bal), while only 4.1% corresponded to abundance gradient (d_BC-gra). Differences between phases were primarily driven by reciprocal shifts in species abundances, rather than by consistent unidirectional losses or gains of individuals across the assemblage. This analysis considered the full species assemblage recorded in each phase. The dBC-bal and dBC were significantly associated with reservoir phase, distance from the dam structure, and reservoir phase (Figure 6, Table 6), with no evidence of differences in multivariate dispersion. In contrast, d_BC-gra showed no significant group-level structure and high dispersion heterogeneity, indicating a minor and spatially inconsistent contribution to beta diversity.

Table 6 | Results of Permutational Multivariate Analysis of Variance (PERMANOVA), permutation-based Analysis of Variance (ANOVA) for individual predictors using distance-based Redundancy Analysis (dbRDA), and Permutational Analysis of Multivariate Dispersions (PERMDISP) for beta diversity components based on Bray-Curtis dissimilarities. Beta diversity was partitioned into balanced variation (d_BC-bal), abundance gradient (d_BC-gra), and total dissimilarity (d_BC) components. PERMANOVA results include F-values, R^2, and p-values for the full model. dbRDA results represent F-values and p-values from permutation-based ANOVA for individual predictors. For Abundace gradient, PERMANOVA results shown are after Cailliez correction. PERMDISP results (F/p-values) indicate differences in multivariate dispersion between reservoir phases. Results were considered significant at $p < 0.05$.

Component	Analysis	Predictor	F-value	R^2	p-value	PERMDISP (F/p-value)
Balance variation (dBC-bal)	PERMANOVA	Full model	18.67	0.153	0.001	0.16 / 0.664
	dbRDA	Reservoir Phase	4.10		0.001	
		Flow	1.7		0.001	
		Distance	3.89		0.001	
Abundance gradient (dBC-gra)	PERMANOVA	Full model	0.89	0.008	1	94.64 / 0.001
	dbRDA	Reservoir Phase	0.73		1	
		Flow	1.17		0.001	
		Distance	0.76		1	
Total (dBC)	PERMANOVA	Full model	16.51	0.137	0.001	0.69 / 0.397
	dbRDA	Reservoir Phase	9.93		0.001	
		Flow	4.19		0.001	
		Distance	9.35		0.001	

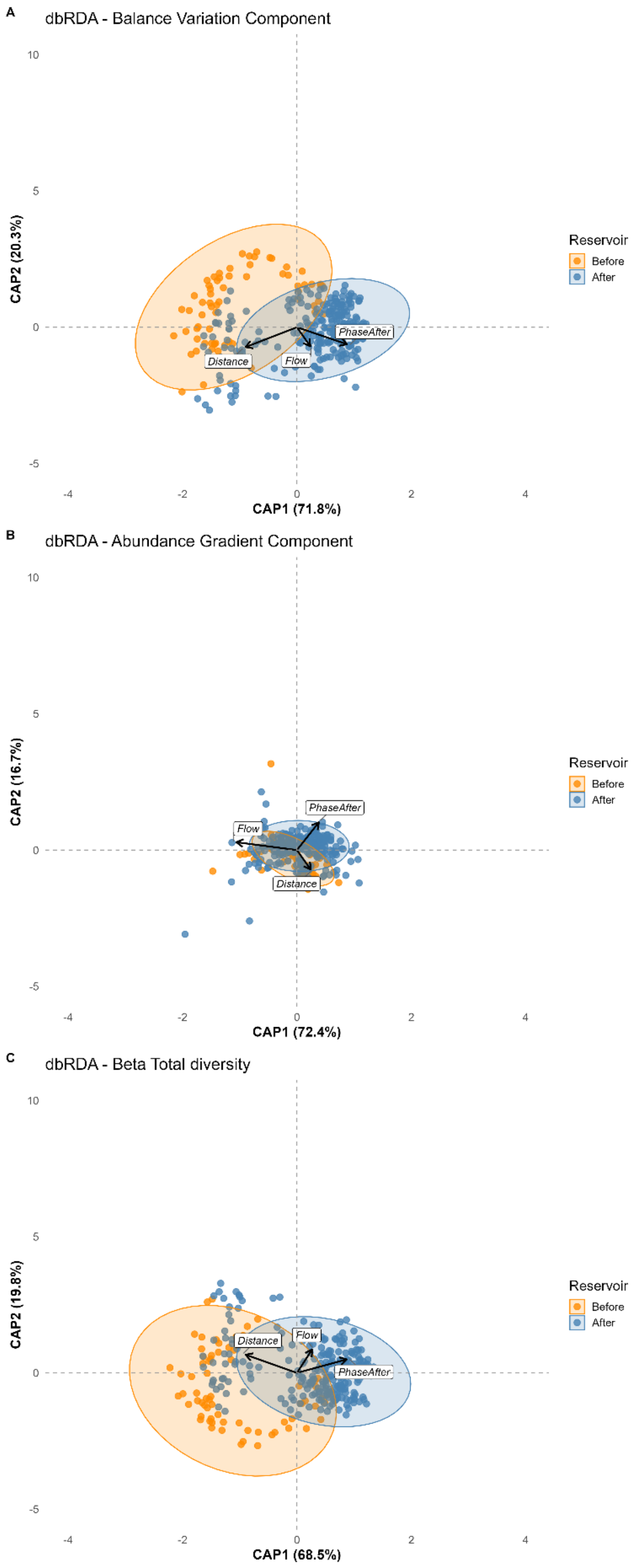

Figure 6 | Distance-based Redundancy Analysis (dbRDA) based on Bray-Curtis dissimilarities for the balance variation (Panel A), abundance gradient (Panel B) and Beta Total Diversity (Panel C). Analyses were constrained by river flow (Flow), distance from the dam (Distance), and reservoir phase (Before and After the impoundment). Arrows represent environmental predictors scaled by their correlations with the canonical axes. Points correspond to individual sampling events, colored by reservoir phase: blue for After and orange for Before. Ellipses denote 95% confidence regions for each phase. Axis labels indicate the percentage of variance explained by each constrained axis.

Based on the trait data, after reservoir formation the assemblage shows a higher frequency of generalists (omnivores and detritivores) and a reduction of piscivores and carnivores. These feeding preference shifts coincide with a greater share of subterminal and supra-terminal mouths, consistent with broader foraging habitats. Demersal and pelagic preferences also increase. Based on long-term fish monitoring data, elongated body forms and forked caudal fins predominate in both periods but are more frequent after impoundment, consistent with sustained swimming performance. Migratory taxa became more prevalent, accompanied by an increase in the relative proportion of species with non-seasonally restricted reproduction (i.e. capable of reproducing in both hydrological seasons) from approximately 26.2% to 35.0%, while species whose reproduction is restricted to the high-water season declined from about 71.6% to 62.5%, although they remained the dominant group. Regarding body size, the assemblage shifts from small to mid-sized fish after impoundment: the 12-21 cm class declines (37.6% to 22.8%), while 22-31 cm rises (19.3% to 35.3%). Several larger classes (≥52 cm up to 451 cm) show modest gains. Frequency of individuals in each functional trait category before and after impoundment are present at Appendix E.

IV. Discussion

Prior to impoundment, the section of the Madeira River was characterised by a deeply incised channel with steep banks reaching up to 15 meters (Queiroz et al., 2013), and dominated by rapids, which acted as biogeographic filters (Torrente-Vilara et al., 2011). These natural barriers may have restricted the presence of species, maintaining a functionally distinct assemblage upstream and downstream. Following the impoundment, the environment became accessible to a broader set of species, initiating the colonisation stage described by Agostinho et al. (1999), during which assemblages restructure in response to new environmental conditions as predicted by Agostinho et al. (2016).

A reorganisation of the fish assemblage was observed following reservoir formation, primarily driven by balanced variation in species abundances, rather than by unidirectional abundance gradients. This pattern indicates a reconfiguration of relative species abundances without evidence of consistent directional losses across the assemblage, and therefore without a tendency towards assemblage homogenisation (Baselga 2010; 2012; Baselga et al. 2023). The dominance of balanced abundance variation was consistent with the observed shifts in functional diversity, despite the absence of significant associations between taxonomic diversity indices and reservoir formation. This decoupling between taxonomic and functional responses has been observed in others reservoirs (Queiroz-Sousa et al., 2019; Sagouis et al., 2016; C. Zhang et al., 2020, 2022; Li et al., 2023; Jia et al., 2020) as functional metrics tend to capture subtle ecological changes more effectively than taxonomic metrics alone (Mouillot et al., 2013; Sagouis et al., 2016).

Both river flow and spatial gradients significantly shaped taxonomic and functional diversity patterns. As part of the Amazon Basin, the Madeira River has well-defined hydrological seasonality (Graça et al., 2025). The run-of-the-river reservoir retains pronounced river-like conditions, characterised by a very low water residence time and sustained flow within the impounded reach, features not always observed in other run-of-the-river systems (Tsuanyo et al., 2023; Venus et al., 2020). This hydrodynamic context likely underlies the observed influence of river flow on fish assemblage structure. In the SA-HPP reservoir, all indices (taxonomic or functional) exhibited temporal variability throughout the year, both before and after reservoir formation, yet no consistent patterns were detected. These fluctuations may reflect the influence of seasonal river flow in shaping assemblage structure, rather than directional changes in functional traits within the assemblage. Importantly, for most sites, variability among individual sampling events exceeded differences attributable to reservoir phase, indicating that short-term hydrological conditions played a major role in structuring assemblage metrics. This contrasts with storage reservoirs, where stronger flow regulation typically imposes a tighter ecological filter and promotes taxonomic and functional homogenisation (Agostinho et al., 2008, 2016; Arantes et al., 2019; A. G. Oliveira et al., 2018; C. Zhang et al., 2020, 2022).

The spatial gradient along the reservoir (longitudinal gradient) also contributed significantly to assemblage differentiation, as reported in other Neotropical reservoirs (Agostinho et al., 2016). Historically, the sector of the river closest to the dam was located between the Santo Antônio Rapid downstream and the Macaco and Teotônio rapids upstream, a configuration that already imposed natural constraints on species distributions prior to impoundment. Within this context, although no clear longitudinal gradient was detected across the SA-HPP reservoir, a marked spatial contrast was observed between the area closest to the dam and upstream sites. Consistent with its historically constrained position, the site nearest the dam (RSB) supported assemblages that were comparatively less diverse and more even under both taxonomic and functional approaches. This local pattern is likely related to hydrodynamic conditions near the dam, particularly reduced flow velocity and increased water depth. Even under a run-of-the-river regime, characterised by weak limnological stratification and short water residence times, environmental conditions tend to differ most strongly in the vicinity of the dam, which may restrict the range of taxa able to occupy this sector and result in a more constrained functional space.

In contrast, the upstream and tributary sites exhibited higher diversity and lower evenness. Despite the reservoir relatively limited area, approximately 100 km in length, differences in vegetation structure and human activity that exist along the reservoir may have increased habitat heterogeneity and supported assemblage resilience. This spatial heterogeneity, combined with littoral-pelagic gradients and vertical depth-related stratification in habitat conditions (Baumgartner et al., 2018), likely increased diversity of available habitats and ecological niches. Furthermore, littoral zones typically provide shelter and abundant food resources, facilitating the colonisation of species with diverse functional strategies (Agostinho et al., 1999; Camara et al., 2024). Consequently, this supports greater assemblage resilience and explains the higher diversity compared to the site closer to the dam.

Contrary to findings in other Neotropical reservoirs, where piscivorous and carnivorous species often increase after impoundment (Agostinho et al., 2016; Perônico et al., 2020), the assemblage exhibited a reorganisation of trophic structure, with lower representation of higher trophic levels and greater contribution of generalist and lower-trophic feeders. Omnivores and detritivores became more prominent, with modest gains among herbivores, invertivores and plankton feeders. This trophic structure is characteristic of the early colonisation phase described by Agostinho et al. (1999), where the new environmental conditions favour generalist species over specialised predators. Moreover, body size structure further reflects this transitional state, shifting from a prevalence of small-bodied fish to mid-sized individuals, alongside increases in larger classes (≥ 52 cm). This pattern contrasts with the typical expectation of dominance by small-bodied species after the reservoir formation (Agostinho et al., 1999), reflecting a transitional assemblage structure characterised by concurrent shifts in trophic composition, body size and functional traits.

Elongated and fusiform taxa already predominated before impoundment and became slightly more frequent afterwards, while rounded bodies declined. The predominant morphotypes are suited to irregular flows and heterogeneous habitats, as in the SA-HPP reservoir. Elongated and fusiform shapes, narrow peduncles, and larger caudal fins are associated with sustained, continuous swimming via body-caudal-fin propulsion, enhancing hydrodynamic efficiency for longer distances (Blake, 2004) and an affinity for rapid flow environment (Leavy & Bonner, 2009). These traits are fundamental for exploiting dispersed resources and maintaining position in the persistent flows (Blake, 2004), reinforcing river flow as a major influence structuring the assemblage in the reservoir. Finally, the inclusion of main-channel sampling sites after impoundment likely improved the detection of species associated with high-flow environments, which may have contributed to the increased representation of elongated and fusiform morphotypes observed after reservoir formation.

Changes in fish assemblages are expected after impoundments (Agostinho et al., 2016; Loures & Pompeu, 2019; Perônico et al., 2020). However, following reservoir formation, three additional sampling sites were incorporated into the monitoring program. These sites could only be included after the reservoir was established, as access had previously been limited by the rapids. Additionally, monitoring efforts were carried out over a longer period after the reservoir was formed than before impoundment. These factors may also have increased species detectability and contributed to observed differences in assemblage structure. Similar patterns were reported by Oliveira et al. (2018), who suggested that longer sampling periods can increase the probability of detecting rare species, potentially influencing richness metrics and concealing functional losses. Although increased sampling effort after impoundment likely enhanced species detectability, several patterns observed in median values of taxonomic and functional indices suggest that detected differences cannot be attributed solely to sampling intensity.

The observed reorganisation in functional structure underscores the importance of integrating multiple dimensions of diversity. This approach complements classical taxonomic diversity, as functional diversity was more sensitive to anthropogenic disturbances (Mouillot et al., 2013; Sagouis et al., 2016; Teichert et al., 2018). Furthermore, as the reservoir is still undergoing a colonisation stage (Agostinho et al., 1999), changes in assemblage structure may continue to occur over time.

Additionally, reservoir aging combined with climate change may further affect the environmental conditions and reshape species interactions and patterns of coexistence (Miranda & Faucheux, 2022). Maintaining consistent ichthyofaunal surveys over time is essential to detect ecological changes and enhance the accuracy of impacts in Neotropical rivers. In addition to classical taxonomic diversity indices, incorporating functional analyses into monitoring programs is strategic, as functional traits often respond earlier to environmental disturbances. The data generated by these monitoring programs are essential for guiding future decision-making and supporting adaptive management and conservation strategies in the face of changing environmental conditions.

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Supporting Information

Appendix A | Taxonomic classification and functional traits of fish species recorded in the study. Functional traits were classified according to feeding preference (FD; Carnivorous - Car, Detritivorous - Det, Hematophagous - Hem, Herbivorous - Heb, Iliophage - Ili, Invertivorous - Inv, Omnivorous - Omn, Piscivorous - Pis, Planktivorous - Plk), mouth position (MP; Terminal - Ter, Sub-terminal - Sub, Supra-terminal - Sup), water column preference (WCP; Benthopelagic - BPe, Demersal - Dem, Pelagic - Pel), body shape (BS; Anguilliform - Ang, Depressed - Dep, Elongated - Elo, Fusiform - Fus, Rounded - Rou), caudal fin type (CFT; Absent - Abs, Emarginated - Ema, Forked - For, Lunate - Lun, Pointed - Poi, Rounded - Rnd, Truncate - Tru), reproductive period (RP; Dry season - Dry, Wet season - Wet, Both seasons - Both), migratory behaviour (MB; Migratory - M, Non-migratory - N-M), and total length (TL; in centimeters).

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Characiformes	Acestrorhynchidae	*Acestrorhynchus falcatus*	Pis	Ter	Bpe	Elo	For	Wet	N-M	30.0
Characiformes	Acestrorhynchidae	*Acestrorhynchus falcirostris*	Pis	Ter	Bpe	Elo	For	Both	M	45.0
Characiformes	Acestrorhynchidae	*Acestrorhynchus heterolepis*	Pis	Ter	Bpe	Elo	For	Wet	M	40.8
Characiformes	Acestrorhynchidae	*Acestrorhynchus microlepis*	Pis	Ter	Bpe	Elo	For	Wet	M	37.0
Characiformes	Acestrorhynchidae	*Acestrorhynchus pantaneiro*	Pis	Ter	Bpe	Elo	For	Wet	N-M	35.2
Characiformes	Acestrorhynchidae	*Acestrorhynchus* sp.	Pis	Ter	Bpe	Elo	For	Wet	N-M	37.6
Characiformes	Acestrorhynchidae	*Roestes molossus*	Pis	Sup	Bpe	Elo	For	Wet	N-M	21.6
Characiformes	Anostomidae	*Abramites hypselonotus*	Her	Ter	Bpe	Fus	For	Wet	M	14.0
Characiformes	Anostomidae	*Laemolyta proxima*	Her	Ter	Bpe	Elo	For	Wet	M	28.5
Characiformes	Anostomidae	*Laemolyta taeniata*	Omn	Ter	Bpe	Elo	For	Wet	M	28.8
Characiformes	Anostomidae	*Leporellus vittatus*	Omn	Sub	Bpe	Fus	For	Wet	M	30.0
Characiformes	Anostomidae	*Leporinus desmotes*	Omn	Ter	Bpe	Elo	For	Wet	M	22.0
Characiformes	Anostomidae	*Leporinus fasciatus*	Omn	Ter	Bpe	Elo	For	Wet	M	37.0
Characiformes	Anostomidae	*Leporinus friderici*	Her	Ter	Bpe	Elo	For	Both	M	40.0
Characiformes	Anostomidae	*Leporinus jamesi*	Omn	Sub	Pel	Elo	For	Wet	M	15.5
Characiformes	Anostomidae	*Leporinus trimaculatus*	Omn	Ter	Bpe	Elo	For	Wet	M	23.2
Characiformes	Anostomidae	*Megaleporinus trifasciatus*	Omn	Ter	Bpe	Elo	For	Wet	M	29.9
Characiformes	Anostomidae	*Pseudanos gracilis*	Omn	Sup	Bpe	Fus	For	Wet	M	18.1
Characiformes	Anostomidae	*Pseudanos trimaculatus*	Omn	Sup	Bpe	Fus	For	Wet	M	18.8
Characiformes	Anostomidae	*Rhytiodus argenteofuscus*	Her	Sub	Bpe	Fus	For	Wet	M	32.0

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Characiformes	Anostomidae	*Rhytiodus microlepis*	Her	Sub	Bpe	Fus	For	Wet	M	33.2
Characiformes	Anostomidae	*Rhytiodus* sp.	Her	Sub	Bpe	Fus	For	Wet	M	34.0
Characiformes	Anostomidae	*Schizodon fasciatus*	Her	Sup	Bpe	Fus	For	Wet	M	40.0
Characiformes	Bryconidae	*Brycon amazonicus*	Omn	Ter	Bpe	Fus	For	Wet	M	46.2
Characiformes	Bryconidae	*Brycon melanopterus*	Omn	Ter	Bpe	Fus	For	Dry	M	38.0
Characiformes	Bryconidae	*Brycon pesu*	Omn	Ter	Bpe	Fus	For	Wet	M	18.9
Characiformes	Bryconidae	*Salminus brasiliensis*	Car	Ter	Bpe	Fus	Ema	Wet	M	100.0
Characiformes	Bryconidae	*Salminus* sp.	Car	Ter	Bpe	Fus	Ema	Wet	M	16.4
Characiformes	Chalceidae	*Chalceus epakros*	Omn	Ter	Bpe	Fus	For	Wet	M	17.4
Characiformes	Chalceidae	*Chalceus guaporensis*	Inv	Ter	Bpe	Fus	For	Wet	M	17.4
Characiformes	Characidae	*Acestrocephalus pallidus*	Pis	Ter	Bpe	Fus	For	Wet	N-M	9.7
Characiformes	Characidae	*Agoniates anchovia*	Pis	Sup	Bpe	Elo	For	Dry	N-M	14.8
Characiformes	Characidae	*Agoniates halecinus*	Pis	Sup	Bpe	Elo	For	Wet	N-M	26.8
Characiformes	Characidae	*Astyanax anterior*	Omn	Ter	Bpe	Fus	For	Wet	N-M	11.1
Characiformes	Characidae	*Astyanax bimaculatus*	Omn	Ter	Bpe	Rou	For	Wet	M	17.5
Characiformes	Characidae	*Astyanax* sp.	Omn	Ter	Bpe	Fus	For	Wet	N-M	12.6
Characiformes	Characidae	*Catoprion mento*	Pis	Ter	Bpe	Rou	For	Wet	N-M	15.0
Characiformes	Characidae	*Charax* sp.	Car	Sup	Bpe	Fus	For	Wet	N-M	16.6
Characiformes	Characidae	*Ctenobrycon spilurus*	Omn	Ter	Pel	Rou	For	Wet	N-M	8.0
Characiformes	Characidae	*Cynopotamus gouldingi*	Car	Ter	Bpe	Fus	For	Wet	N-M	16.5
Characiformes	Characidae	*Galeocharax goeldii*	Car	Ter	Bpe	Fus	For	Wet	N-M	16.5
Characiformes	Characidae	*Jupiaba zonata*	Omn	Ter	Bpe	Fus	For	Wet	N-M	4.7
Characiformes	Characidae	*Knodus heteresthes*	Inv	Ter	Bpe	Fus	For	Wet	N-M	5.1
Characiformes	Characidae	*Moenkhausia bonita*	Inv	Ter	Pel	Elo	For	Wet	N-M	8.5
Characiformes	Characidae	*Moenkhausia cotinho*	Omn	Ter	Bpe	Elo	For	Wet	N-M	6.1
Characiformes	Characidae	*Moenkhausia dichroura*	Omn	Ter	Bpe	Elo	For	Wet	N-M	10.0
Characiformes	Characidae	*Moenkhausia grandisquamis*	Omn	Ter	Bpe	Rou	For	Wet	N-M	10.0

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Characiformes	Characidae	*Moenkhausia jamesi*	Omn	Ter	Bpe	Elo	For	Wet	N-M	6.8
Characiformes	Characidae	*Moenkhausia lepidura*	Car	Ter	Bpe	Elo	For	Wet	N-M	10.5
Characiformes	Characidae	*Moenkhausia* sp.	Omn	Ter	Bpe	Elo	For	Wet	N-M	7.6
Characiformes	Characidae	*Parecbasis cyclolepis*	Car	Ter	Pel	Elo	For	Wet	N-M	8.0
Characiformes	Characidae	*Poptella compressa*	Omn	Ter	Bpe	Rou	For	Wet	N-M	10.2
Characiformes	Characidae	*Roeboides affinis*	Car	Sup	Bpe	Fus	For	Wet	N-M	11.0
Characiformes	Characidae	*Roeboides myersii*	Car	Sup	Bpe	Fus	For	Both	N-M	22.0
Characiformes	Characidae	*Tetragonopterus argenteus*	Car	Ter	Pel	Rou	For	Wet	N-M	11.2
Characiformes	Characidae	*Tetragonopterus chalceus*	Omn	Ter	Pel	Rou	For	Wet	N-M	12.7
Characiformes	Chilodontidae	*Caenotropus labyrinthicus*	Omn	Sub	Dem	Elo	For	Wet	N-M	25.3
Characiformes	Chilodontidae	*Caenotropus schizodon*	Car	Sub	Pel	Elo	For	Wet	N-M	9.1
Characiformes	Curimatidae	*Curimata inornata*	Det	Sub	Bpe	Elo	For	Wet	M	21.0
Characiformes	Curimatidae	*Curimata knerii*	Det	Sub	Bpe	Rou	For	Wet	M	17.7
Characiformes	Curimatidae	*Curimata roseni*	Det	Sub	Bpe	Elo	For	Wet	M	19.5
Characiformes	Curimatidae	*Curimata vittata*	Det	Sub	Bpe	Elo	For	Wet	M	19.6
Characiformes	Curimatidae	*Curimatella alburnus*	Det	Sub	Bpe	Elo	For	Wet	M	18.8
Characiformes	Curimatidae	*Curimatella dorsalis*	Det	Sub	Bpe	Elo	For	Wet	M	14.9
Characiformes	Curimatidae	*Curimatella immaculata*	Det	Sub	Bpe	Elo	For	Wet	M	18.8
Characiformes	Curimatidae	*Curimatella meyeri*	Det	Sub	Bpe	Elo	For	Wet	M	18.4
Characiformes	Curimatidae	*Curimatella* sp.	Det	Sub	Bpe	Elo	For	Wet	M	12.7
Characiformes	Curimatidae	*Cyphocharax notatus*	Det	Sub	Bpe	Elo	For	Wet	M	14.7
Characiformes	Curimatidae	*Cyphocharax plumbeus*	Det	Sub	Bpe	Elo	For	Wet	M	17.8
Characiformes	Curimatidae	*Cyphocharax* sp.	Det	Sub	Bpe	Elo	For	Wet	M	17.3
Characiformes	Curimatidae	*Potamorhina altamazonica*	Det	Sub	Bpe	Elo	For	Both	M	27.0
Characiformes	Curimatidae	*Potamorhina latior*	Det	Sub	Bpe	Elo	For	Both	M	24.0
Characiformes	Curimatidae	*Potamorhina pristigaster*	Det	Sub	Bpe	Elo	For	Wet	N-M	21.9
Characiformes	Curimatidae	*Psectrogaster amazonica*	Det	Sub	Bpe	Elo	For	Both	M	19.0

Order	**Family**	**Species**	**FD**	**MP**	**WCP**	**BS**	**CFT**	**RP**	**MB**	**TL (cm)**
Characiformes	Curimatidae	*Psectrogaster essequibensis*	Det	Sub	Bpe	Rou	For	Both	M	16.9
Characiformes	Curimatidae	*Psectrogaster rutiloides*	Det	Sub	Bpe	Elo	For	Both	M	17.8
Characiformes	Curimatidae	*Steindachnerina bimaculata*	Det	Sub	Bpe	Elo	For	Both	M	17.3
Characiformes	Curimatidae	*Steindachnerina dobula*	Det	Sub	Bpe	Elo	For	Both	M	16.3
Characiformes	Curimatidae	*Steindachnerina hypostoma*	Det	Sub	Bpe	Elo	For	Both	M	9.8
Characiformes	Curimatidae	*Steindachnerina leucisca*	Det	Sub	Bpe	Elo	For	Wet	M	15.1
Characiformes	Curimatidae	*Steindachnerina planiventris*	Det	Sub	Bpe	Elo	For	Wet	M	8.7
Characiformes	Cynodontidae	*Cynodon gibbus*	Pis	Sup	Pel	Elo	For	Wet	M	32.2
Characiformes	Cynodontidae	*Hydrolycus armatus*	Pis	Sup	Pel	Elo	Ema	Wet	M	89.0
Characiformes	Cynodontidae	*Hydrolycus scomberoides*	Pis	Sup	Bpe	Elo	Ema	Both	M	117.0
Characiformes	Cynodontidae	*Rhaphiodon vulpinus*	Pis	Sup	Pel	Elo	Poi	Both	M	80.0
Characiformes	Erythrinidae	*Hoplias malabaricus*	Pis	Sup	Bpe	Elo	Tru	Both	N-M	65.0
Characiformes	Gasteropelecidae	*Thoracocharax stellatus*	Inv	Sup	Pel	Rou	For	Wet	N-M	6.7
Characiformes	Hemiodontidae	*Anodus elongatus*	Omn	Ter	Pel	Fus	For	Both	M	30.3
Characiformes	Hemiodontidae	*Anodus orinocensis*	Omn	Ter	Pel	Fus	For	Both	M	27.5
Characiformes	Hemiodontidae	*Argonectes longiceps*	Omn	Ter	Bpe	Fus	For	Dry	M	26.0
Characiformes	Hemiodontidae	*Hemiodus immaculatus*	Her	Ter	Bpe	Fus	For	Wet	M	24.7
Characiformes	Hemiodontidae	*Hemiodus microlepis*	Det	Ter	Bpe	Fus	For	Wet	M	23.9
Characiformes	Hemiodontidae	*Hemiodus semitaeniatus*	Her	Ter	Bpe	Fus	For	Wet	M	20.0
Characiformes	Hemiodontidae	*Hemiodus* sp.	Her	Ter	Bpe	Fus	For	Dry	M	23.5
Characiformes	Hemiodontidae	*Hemiodus unimaculatus*	Det	Ter	Bpe	Fus	For	Wet	M	32.9
Characiformes	Iguanodectidae	*Bryconops alburnoides*	Inv	Ter	Bpe	Fus	For	Wet	N-M	15.0
Characiformes	Iguanodectidae	*Bryconops caudomaculatus*	Omn	Ter	Bpe	Fus	For	Wet	N-M	12.4
Characiformes	Iguanodectidae	*Bryconops giacopinii*	Omn	Ter	Bpe	Fus	For	Dry	N-M	18.0
Characiformes	Prochilodontidae	*Prochilodus nigricans*	Det	Ter	Bpe	Elo	For	Wet	M	52.5
Characiformes	Prochilodontidae	*Semaprochilodus insignis*	Det	Ter	Bpe	Elo	For	Wet	M	29.5
Characiformes	Prochilodontidae	*Semaprochilodus taeniurus*	Det	Ter	Bpe	Elo	For	Wet	M	27.2

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Characiformes	Serrasalmidae	*Colossoma macropomum*	Omn	Ter	Bpe	Rou	For	Wet	M	108.0
Characiformes	Serrasalmidae	*Metynnis guaporensis*	Her	Ter	Pel	Rou	Ema	Wet	M	16.0
Characiformes	Serrasalmidae	*Metynnis hypsauchen*	Her	Ter	Pel	Rou	Ema	Wet	M	18.0
Characiformes	Serrasalmidae	*Metynnis lippincottianus*	Her	Ter	Pel	Rou	For	Wet	M	16.4
Characiformes	Serrasalmidae	*Metynnis luna*	Her	Ter	Pel	Rou	Ema	Wet	M	12.9
Characiformes	Serrasalmidae	*Metynnis maculatus*	Her	Ter	Pel	Rou	Ema	Wet	M	18.0
Characiformes	Serrasalmidae	*Myleus setiger*	Her	Ter	Bpe	Rou	For	Wet	M	28.7
Characiformes	Serrasalmidae	*Myloplus asterias*	Her	Ter	Bpe	Rou	Ema	Wet	M	25.0
Characiformes	Serrasalmidae	*Myloplus lobatus*	Her	Ter	Bpe	Rou	For	Dry	M	24.5
Characiformes	Serrasalmidae	*Myloplus rubripinnis*	Her	Ter	Bpe	Rou	For	Dry	M	41.5
Characiformes	Serrasalmidae	*Myloplus* sp.	Her	Ter	Bpe	Rou	For	Wet	M	21.0
Characiformes	Serrasalmidae	*Mylossoma aureum*	Omn	Ter	Bpe	Rou	Ema	Wet	M	20.0
Characiformes	Serrasalmidae	*Mylossoma duriventre*	Her	Ter	Bpe	Rou	Ema	Wet	M	25.0
Characiformes	Serrasalmidae	*Piaractus brachypomus*	Omn	Ter	Pel	Rou	For	Wet	M	88.0
Characiformes	Serrasalmidae	*Pristobrycon* sp.	Pis	Ter	Bpe	Rou	For	Dry	N-M	19.8
Characiformes	Serrasalmidae	*Pygocentrus nattereri*	Pis	Ter	Pel	Rou	For	Wet	N-M	50.0
Characiformes	Serrasalmidae	*Serrasalmus compressus*	Car	Ter	Bpe	Rou	For	Both	N-M	19.0
Characiformes	Serrasalmidae	*Serrasalmus eigenmanni*	Car	Ter	Bpe	Rou	For	Wet	N-M	35.0
Characiformes	Serrasalmidae	*Serrasalmus elongatus*	Car	Ter	Bpe	Elo	For	Wet	N-M	30.0
Characiformes	Serrasalmidae	*Serrasalmus hollandi*	Car	Ter	Bpe	Rou	For	Wet	N-M	18.5
Characiformes	Serrasalmidae	*Serrasalmus maculatus*	Car	Ter	Pel	Rou	For	Wet	N-M	34.5
Characiformes	Serrasalmidae	*Serrasalmus rhombeus*	Pis	Ter	Bpe	Rou	Ema	Both	N-M	41.5
Characiformes	Serrasalmidae	*Serrasalmus* sp.1	Car	Ter	Bpe	Rou	For	Wet	N-M	11.2
Characiformes	Serrasalmidae	*Serrasalmus* sp.2	Car	Ter	Bpe	Rou	For	Dry	N-M	18.6
Characiformes	Serrasalmidae	*Serrasalmus* sp.3	Car	Ter	Bpe	Rou	For	Wet	N-M	19.3
Characiformes	Serrasalmidae	*Serrasalmus* sp.4	Car	Ter	Bpe	Rou	For	Wet	N-M	14.6
Characiformes	Serrasalmidae	*Serrasalmus* sp.5	Car	Ter	Bpe	Rou	For	Wet	N-M	23.6

Order	**Family**	**Species**	**FD**	**MP**	**WCP**	**BS**	**CFT**	**RP**	**MB**	**TL (cm)**
Characiformes	Serrasalmidae	*Serrasalmus spilopleura*	Car	Ter	Bpe	Rou	Ema	Wet	N-M	31.0
Characiformes	Triportheidae	*Triportheus albus*	Inv	Sup	Bpe	Elo	For	Wet	M	25.7
Characiformes	Triportheidae	*Triportheus angulatus*	Omn	Sup	Bpe	Elo	Poi	Wet	M	23.8
Characiformes	Triportheidae	*Triportheus auritus*	Omn	Sup	Bpe	Elo	Poi	Wet	M	26.5
Characiformes	Triportheidae	*Triportheus culter*	Omn	Sup	Bpe	Elo	Poi	Dry	M	24.5
Characiformes	Triportheidae	*Triportheus curtus*	Omn	Sup	Bpe	Elo	Poi	Wet	M	10.7
Characiformes	Triportheidae	*Triportheus rotundatus*	Omn	Sup	Bpe	Elo	Poi	Wet	M	26.8
Clupeiformes	Engraulidae	*Anchoviella carrikeri*	Plk	Sub	Pel	Elo	For	Wet	M	6.5
Clupeiformes	Engraulidae	*Jurengraulis juruensis*	Plk	Sub	Pel	Elo	For	Wet	M	17.9
Clupeiformes	Engraulidae	*Lycengraulis batesii*	Car	Sub	Pel	Elo	For	Both	M	30.0
Clupeiformes	Pristigasteridae	*Ilisha amazonica*	Car	Sup	Pel	Elo	For	Dry	N-M	19.7
Clupeiformes	Pristigasteridae	*Pellona castelnaeana*	Pis	Sup	Pel	Elo	For	Wet	M	80.0
Clupeiformes	Pristigasteridae	*Pellona flavipinnis*	Car	Sup	Pel	Elo	For	Both	M	73.0
Gymnotiformes	Apteronotidae	*Adontosternarchus balaenops*	Inv	Ter	Bpe	Ang	Rnd	Wet	N-M	25.0
Gymnotiformes	Apteronotidae	*Adontosternarchus clarkae*	Car	Ter	Bpe	Ang	Rnd	Wet	N-M	18.6
Gymnotiformes	Apteronotidae	*Apteronotus albifrons*	Inv	Sub	Bpe	Ang	Rnd	Dry	N-M	50.0
Gymnotiformes	Apteronotidae	*Apteronotus bonapartii*	Inv	Sub	Bpe	Ang	Rnd	Dry	N-M	38.0
Gymnotiformes	Apteronotidae	*Apteronotus* sp.	Car	Sub	Bpe	Ang	Rnd	Dry	N-M	26.3
Gymnotiformes	Apteronotidae	*Compsaraia compsa*	Inv	Sub	Bpe	Ang	Rnd	Wet	N-M	33.9
Gymnotiformes	Apteronotidae	*Parapteronotus hasemani*	Car	Ter	Bpe	Ang	Rnd	Wet	N-M	38.1
Gymnotiformes	Apteronotidae	*Porotergus* sp.	Car	Sub	Bpe	Ang	Abs	Wet	N-M	28.4
Gymnotiformes	Apteronotidae	*Sternarchella schotti*	Inv	Ter	Bpe	Ang	Rnd	Wet	N-M	40.0
Gymnotiformes	Apteronotidae	*Sternarchella sima*	Car	Sub	Bpe	Ang	Rnd	Wet	N-M	40.0
Gymnotiformes	Apteronotidae	*Sternarchella* sp.	Inv	Sub	Bpe	Ang	Rnd	Dry	N-M	27.1
Gymnotiformes	Apteronotidae	*Sternarchorhynchus chaoi*	Inv	Sub	Bpe	Ang	Rnd	Wet	N-M	16.4
Gymnotiformes	Apteronotidae	*Sternarchorhynchus sp*	Inv	Sub	Bpe	Ang	Rnd	Wet	N-M	47.0
Gymnotiformes	Gymnotidae	*Electrophorus electricus*	Car	Sup	Bpe	Ang	Abs	Wet	N-M	250.0

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Gymnotiformes	Gymnotidae	*Gymnotus carapo*	Car	Sup	Bpe	Ang	Abs	Wet	N-M	76.0
Gymnotiformes	Gymnotidae	*Gymnotus curupira*	Car	Sup	Bpe	Ang	Abs	Wet	N-M	23.5
Gymnotiformes	Gymnotidae	*Gymnotus* sp.	Car	Sup	Bpe	Ang	Abs	Wet	N-M	5.4
Gymnotiformes	Hypopomidae	*Brachyhypopomus pinnicaudatus*	Car	Sub	Bpe	Ang	Abs	Wet	N-M	18.6
Gymnotiformes	Hypopomidae	*Steatogenys elegans*	Inv	Sub	Bpe	Ang	Abs	Wet	N-M	29.4
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys marmoratus*	Car	Ter	Bpe	Ang	Abs	Wet	N-M	65.0
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys pantherinus*	Car	Ter	Bpe	Ang	Abs	Dry	N-M	55.0
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys rostratus*	Car	Ter	Bpe	Ang	Abs	Wet	N-M	100.0
Gymnotiformes	Sternopygidae	*Distocyclus conirostris*	Car	Ter	Bpe	Ang	Abs	Wet	N-M	50.0
Gymnotiformes	Sternopygidae	*Eigenmannia limbata*	Inv	Ter	Bpe	Ang	Abs	Wet	N-M	48.5
Gymnotiformes	Sternopygidae	*Eigenmannia macrops*	Car	Ter	Bpe	Ang	Abs	Wet	N-M	25.2
Gymnotiformes	Sternopygidae	*Rhabdolichops troscheli*	Omn	Sup	Bpe	Ang	Abs	Wet	N-M	49.0
Gymnotiformes	Sternopygidae	*Sternopygus macrurus*	Car	Ter	Bpe	Ang	Abs	Both	N-M	141.0
Osteoglossiformes	Arapaimidae	*Arapaima gigas*	Car	Sup	Dem	Fus	Tru	Dry	N-M	450.0
Perciformes	Cichlidae	*Acaronia nassa*	Car	Ter	Bpe	Elo	Tru	Wet	N-M	16.0
Perciformes	Cichlidae	*Aequidens* sp.	Omn	Sub	Bpe	Elo	Tru	Wet	N-M	9.0
Perciformes	Cichlidae	*Aequidens tetramerus*	Omn	Sub	Bpe	Elo	Tru	Dry	N-M	19.2
Perciformes	Cichlidae	*Astronotus crassipinnis*	Omn	Ter	Bpe	Rou	Tru	Both	N-M	25.0
Perciformes	Cichlidae	*Biotodoma cupido*	Omn	Sub	Bpe	Rou	Ema	Both	N-M	11.7
Perciformes	Cichlidae	*Chaetobranchus flavescens*	Plk	Ter	Bpe	Rou	Tru	Wet	N-M	26.0
Perciformes	Cichlidae	*Cichla pleiozona*	Pis	Sub	Bpe	Elo	Tru	Both	N-M	44.2
Perciformes	Cichlidae	*Crenicichla cyanonotus*	Car	Sup	Bpe	Fus	Rnd	Both	N-M	14.8
Perciformes	Cichlidae	*Crenicichla reticulata*	Car	Sup	Bpe	Fus	Rnd	Both	N-M	29.5
Perciformes	Cichlidae	*Crenicichla* sp.	Car	Sup	Bpe	Fus	Rnd	Both	N-M	16.2
Perciformes	Cichlidae	*Crenicichla strigata*	Car	Sup	Bpe	Fus	Rnd	Both	N-M	30.0
Perciformes	Cichlidae	*Geophagus megasema*	Omn	Sup	Bpe	Rou	Rnd	Both	N-M	18.9
Perciformes	Cichlidae	*Heros spurius*	Pis	Sub	Bpe	Rou	Rnd	Both	N-M	15.0

Order	**Family**	**Species**	**FD**	**MP**	**WCP**	**BS**	**CFT**	**RP**	**MB**	**TL (cm)**
Perciformes	Cichlidae	*Lugubria adspersa*	Car	Sup	Bpe	Fus	Rnd	Dry	N-M	29.0
Perciformes	Cichlidae	*Lugubria johanna*	Car	Sup	Bpe	Fus	Rnd	Dry	N-M	28.3
Perciformes	Cichlidae	*Lugubria marmorata*	Car	Sup	Bpe	Fus	Rnd	Dry	N-M	28.0
Perciformes	Cichlidae	*Mesonauta festivus*	Her	Sub	Bpe	Rou	Rnd	Dry	N-M	12.1
Perciformes	Cichlidae	*Satanoperca jurupari*	Omn	Sub	Bpe	Rou	Tru	Both	N-M	24.0
Perciformes	Cichlidae	*Satanoperca* sp.	Omn	Sub	Bpe	Rou	Tru	Both	N-M	17.6
Perciformes	Cichlidae	*Saxatilia inpa*	Car	Sup	Bpe	Fus	Rnd	Both	N-M	18.4
Perciformes	Cichlidae	*Saxatilia semicincta*	Car	Sup	Bpe	Fus	Rnd	Both	N-M	17.1
Perciformes	Sciaenidae	*Pachypops fourcroi*	Pis	Sub	Bpe	Elo	Tru	Both	N-M	25.0
Perciformes	Sciaenidae	*Pachyurus paucirastrus*	Car	Sub	Bpe	Elo	Tru	Both	N-M	14.8
Perciformes	Sciaenidae	*Plagioscion montei*	Car	Sub	Pel	Elo	Poi	Both	N-M	34.0
Perciformes	Sciaenidae	*Plagioscion squamosissimus*	Car	Sub	Bpe	Elo	Poi	Both	N-M	60.0
Pleuronectiformes	Achiridae	*Hypoclinemus mentalis*	Car	Sup	Dem	Rou	Rnd	Dry	N-M	21.6
Siluriformes	Aspredinidae	*Bunocephalus coracoideus*	Omn	Sub	Dem	Dep	Rnd	Wet	N-M	11.0
Siluriformes	Auchenipteridae	*Ageneiosus inermis*	Pis	Sub	Pel	Elo	Tru	Wet	M	64.8
Siluriformes	Auchenipteridae	*Ageneiosus* sp.1	Car	Sub	Pel	Elo	For	Wet	M	10.3
Siluriformes	Auchenipteridae	*Ageneiosus* sp.2	Car	Sub	Pel	Elo	For	Wet	M	17.3
Siluriformes	Auchenipteridae	*Ageneiosus* sp.3	Car	Sub	Pel	Elo	For	Wet	M	29.7
Siluriformes	Auchenipteridae	*Ageneiosus ucayalensis*	Car	Sub	Pel	Elo	For	Wet	M	35.2
Siluriformes	Auchenipteridae	*Ageneiosus uranophthalmus*	Car	Sub	Dem	Elo	For	Wet	M	19.8
Siluriformes	Auchenipteridae	*Ageneiosus vittatus*	Car	Sub	Pel	Elo	Ema	Wet	N-M	22.0
Siluriformes	Auchenipteridae	*Auchenipterichthys coracoideus*	Omn	Ter	Dem	Elo	Ema	Wet	N-M	19.2
Siluriformes	Auchenipteridae	*Auchenipterichthys longimanus*	Inv	Ter	Bpe	Elo	Ema	Wet	N-M	22.8
Siluriformes	Auchenipteridae	*Auchenipterichthys thoracatus*	Omn	Ter	Bpe	Elo	Ema	Wet	N-M	13.8
Siluriformes	Auchenipteridae	*Auchenipterus ambyiacus*	Inv	Ter	Bpe	Elo	For	Wet	N-M	24.4
Siluriformes	Auchenipteridae	*Auchenipterus brachyurus*	Car	Sup	Bpe	Elo	For	Wet	N-M	16.5
Siluriformes	Auchenipteridae	*Auchenipterus britskii*	Car	Ter	Bpe	Elo	For	Wet	N-M	15.7

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Siluriformes	Auchenipteridae	*Auchenipterus nuchalis*	Car	Ter	Pel	Elo	For	Both	N-M	27.0
Siluriformes	Auchenipteridae	*Auchenipterus* sp.	Omn	Ter	Pel	Elo	For	Wet	N-M	20.9
Siluriformes	Auchenipteridae	*Centromochlus heckelii*	Inv	Ter	Pel	Elo	For	Wet	N-M	13.9
Siluriformes	Auchenipteridae	*Epapterus dispilurus*	Pis	Sup	Dem	Elo	Ema	Wet	N-M	14.5
Siluriformes	Auchenipteridae	*Tatia aulopygia*	Inv	Ter	Bpe	Elo	Ema	Wet	N-M	15.9
Siluriformes	Auchenipteridae	*Tatia intermedia*	Inv	Ter	Bpe	Elo	Ema	Wet	N-M	12.0
Siluriformes	Auchenipteridae	*Trachelyopterus galeatus*	Omn	Ter	Dem	Elo	Tru	Wet	N-M	30.0
Siluriformes	Auchenipteridae	*Trachelyopterus porosus*	Omn	Ter	Dem	Elo	Tru	Wet	N-M	22.2
Siluriformes	Auchenipteridae	*Trachelyopterus* sp.	Omn	Ter	Dem	Elo	Tru	Wet	N-M	22.9
Siluriformes	Auchenipteridae	*Trachycorystes trachycorystes*	Pis	Sup	Dem	Elo	Tru	Wet	N-M	35.0
Siluriformes	Auchenipteridae	*Tympanopleura atronasus*	Car	Sub	Pel	Elo	Ema	Wet	N-M	14.9
Siluriformes	Auchenipteridae	*Tympanopleura brevis*	Car	Sub	Pel	Elo	For	Wet	N-M	16.0
Siluriformes	Auchenipteridae	*Tympanopleura longipinna*	Car	Sub	Bpe	Elo	For	Wet	N-M	8.1
Siluriformes	Callichthyidae	*Hoplosternum littorale*	Omn	Sub	Dem	Elo	Ema	Wet	N-M	26.3
Siluriformes	Cetopsidae	*Cetopsis candiru*	Pis	Ter	Bpe	Elo	For	Dry	N-M	26.3
Siluriformes	Cetopsidae	*Cetopsis coecutiens*	Pis	Sub	Bpe	Elo	For	Dry	N-M	27.3
Siluriformes	Doradidae	*Acanthodoras spinosissimus*	Car	Sub	Dem	Elo	Tru	Wet	N-M	13.7
Siluriformes	Doradidae	*Amblydoras affinis*	Car	Ter	Dem	Elo	Tru	Wet	N-M	13.2
Siluriformes	Doradidae	*Anadoras weddellii*	Omn	Sub	Dem	Elo	Tru	Wet	N-M	15.0
Siluriformes	Doradidae	*Astrodoras fulcro*	Car	Sub	Dem	Elo	For	Wet	N-M	9.2
Siluriformes	Doradidae	*Centrodoras brachiatus*	Car	Sub	Dem	Elo	For	Wet	N-M	41.0
Siluriformes	Doradidae	*Hemidoras boulengeri*	Omn	Sub	Dem	Elo	For	Wet	N-M	19.3
Siluriformes	Doradidae	*Hemidoras morrisi*	Inv	Sub	Dem	Elo	For	Wet	N-M	14.4
Siluriformes	Doradidae	*Hemidoras stenopeltis*	Omn	Sub	Dem	Elo	For	Wet	N-M	19.4
Siluriformes	Doradidae	*Hemidoras stuebelii*	Inv	Sub	Dem	Elo	For	Wet	N-M	15.2
Siluriformes	Doradidae	*Megalodoras uranoscopus*	Omn	Sub	Dem	Elo	For	Wet	N-M	60.0
Siluriformes	Doradidae	*Nemadoras elongatus*	Omn	Sub	Dem	Elo	For	Wet	N-M	16.5

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Siluriformes	Doradidae	*Nemadoras humeralis*	Omn	Sub	Dem	Elo	For	Wet	N-M	16.3
Siluriformes	Doradidae	*Ossancora asterophysa*	Omn	Sub	Dem	Elo	For	Wet	N-M	16.8
Siluriformes	Doradidae	*Ossancora fimbriata*	Omn	Sub	Dem	Elo	For	Wet	N-M	20.0
Siluriformes	Doradidae	*Ossancora punctata*	Omn	Sub	Dem	Elo	For	Wet	N-M	11.3
Siluriformes	Doradidae	*Oxydoras niger*	Det	Sub	Dem	Elo	For	Wet	N-M	100.0
Siluriformes	Doradidae	*Platydoras armatulus*	Car	Sub	Dem	Elo	For	Wet	N-M	43.0
Siluriformes	Doradidae	*Pterodoras granulosus*	Her	Sub	Dem	Elo	For	Wet	M	90.1
Siluriformes	Doradidae	*Scorpiodoras liophysus*	Car	Sub	Bpe	Elo	For	Wet	N-M	15.7
Siluriformes	Doradidae	*Trachydoras brevis*	Inv	Sub	Dem	Elo	For	Wet	N-M	13.9
Siluriformes	Doradidae	*Trachydoras microstomus*	Inv	Sub	Dem	Elo	For	Wet	N-M	5.9
Siluriformes	Doradidae	*Trachydoras paraguayensis*	Inv	Sub	Dem	Elo	For	Wet	N-M	19.2
Siluriformes	Doradidae	*Trachydoras* sp.	Inv	Sub	Dem	Elo	For	Wet	N-M	15.5
Siluriformes	Doradidae	*Trachydoras steindachneri*	Inv	Sub	Dem	Elo	For	Wet	N-M	16.0
Siluriformes	Heptapteridae	*Pimelodella boliviana*	Car	Sub	Dem	Elo	For	Wet	N-M	9.0
Siluriformes	Heptapteridae	*Pimelodella howesi*	Car	Sub	Dem	Elo	For	Dry	N-M	7.9
Siluriformes	Heptapteridae	*Pimelodella* sp.	Car	Sub	Dem	Elo	For	Dry	N-M	23.7
Siluriformes	Loricariidae	*Acanthicus hystrix*	Ili	Sub	Dem	Elo	Tru	Wet	N-M	53.0
Siluriformes	Loricariidae	*Ancistrus dubius*	Det	Sub	Dem	Elo	Tru	Dry	N-M	12.6
Siluriformes	Loricariidae	*Ancistrus* sp.1	Det	Sub	Dem	Elo	Tru	Dry	N-M	16.0
Siluriformes	Loricariidae	*Ancistrus* sp.2	Det	Sub	Dem	Elo	Tru	Dry	N-M	14.3
Siluriformes	Loricariidae	*Ancistrus* sp.3	Det	Sub	Dem	Elo	Tru	Dry	N-M	12.5
Siluriformes	Loricariidae	*Aphanotorulus emarginatus*	Det	Sub	Dem	Dep	Ema	Dry	N-M	33.6
Siluriformes	Loricariidae	*Aphanotorulus unicolor*	Det	Sub	Bpe	Dep	Lun	Dry	N-M	13.9
Siluriformes	Loricariidae	*Farlowella smithi*	Her	Sub	Dem	Elo	Ema	Wet	N-M	10.0
Siluriformes	Loricariidae	*Hemiodontichthys acipenserinus*	Car	Sub	Dem	Elo	Tru	Wet	N-M	13.4
Siluriformes	Loricariidae	*Hypoptopoma incognitum*	Det	Sub	Bpe	Elo	Ema	Wet	N-M	10.8
Siluriformes	Loricariidae	*Hypostomus hoplonites*	Ili	Sub	Dem	Elo	Ema	Wet	N-M	32.5

Order	**Family**	**Species**	**FD**	**MP**	**WCP**	**BS**	**CFT**	**RP**	**MB**	**TL (cm)**
Siluriformes	Loricariidae	*Hypostomus plecostomus*	Det	Sub	Dem	Elo	For	Wet	N-M	28.0
Siluriformes	Loricariidae	*Hypostomus pyrineusi*	Her	Sub	Bpe	Elo	Tru	Wet	N-M	26.0
Siluriformes	Loricariidae	*Hypostomus* sp.	Her	Sub	Bpe	Elo	Tru	Wet	N-M	9.5
Siluriformes	Loricariidae	*Loricaria cataphracta*	Det	Sub	Dem	Dep	Ema	Wet	N-M	29.5
Siluriformes	Loricariidae	*Loricariichthys acutus*	Ili	Sub	Dem	Dep	Tru	Wet	N-M	28.0
Siluriformes	Loricariidae	*Loricariichthys maculatus*	Ili	Sub	Dem	Dep	Tru	Wet	N-M	26.3
Siluriformes	Loricariidae	*Loricariichthys nudirostris*	Ili	Sub	Dem	Dep	Tru	Wet	N-M	23.5
Siluriformes	Loricariidae	*Loricariichthys* sp.1	Ili	Sub	Dem	Dep	Tru	Wet	N-M	34.0
Siluriformes	Loricariidae	*Loricariichthys* sp.2	Ili	Sub	Dem	Dep	Tru	Wet	N-M	25.8
Siluriformes	Loricariidae	*Peckoltichthys bachi*	Det	Sub	Dem	Elo	Ema	Wet	N-M	14.0
Siluriformes	Loricariidae	*Pterygoplichthys lituratus*	Det	Sub	Dem	Elo	Ema	Both	N-M	37.0
Siluriformes	Loricariidae	*Pterygoplichthys pardalis*	Det	Sub	Dem	Elo	Ema	Dry	N-M	57.8
Siluriformes	Loricariidae	*Rineloricaria castroi*	Det	Sub	Dem	Elo	Ema	Wet	N-M	12.5
Siluriformes	Loricariidae	*Rineloricaria formosa*	Det	Sub	Dem	Dep	Ema	Wet	N-M	15.2
Siluriformes	Loricariidae	*Sturisoma lyra*	Det	Sub	Dem	Dep	Ema	Wet	N-M	25.0
Siluriformes	Pimelodidae	*Brachyplatystoma capapretum*	Pis	Ter	Bpe	Elo	For	Wet	M	58.1
Siluriformes	Pimelodidae	*Brachyplatystoma filamentosum*	Pis	Ter	Dem	Elo	For	Wet	M	360.0
Siluriformes	Pimelodidae	*Brachyplatystoma platynemum*	Pis	Ter	Dem	Elo	For	Wet	M	100.0
Siluriformes	Pimelodidae	*Brachyplatystoma rousseauxii*	Pis	Ter	Dem	Elo	For	Wet	M	192.0
Siluriformes	Pimelodidae	*Brachyplatystoma tigrinum*	Pis	Ter	Dem	Elo	For	Wet	M	60.0
Siluriformes	Pimelodidae	*Brachyplatystoma vaillantii*	Pis	Ter	Dem	Elo	For	Both	M	150.0
Siluriformes	Pimelodidae	*Calophysus macropterus*	Pis	Ter	Dem	Elo	For	Wet	M	40.0
Siluriformes	Pimelodidae	*Duopalatinus peruanus*	Car	Ter	Dem	Elo	For	Wet	M	15.0
Siluriformes	Pimelodidae	*Hemisorubim platyrhynchos*	Car	Sup	Dem	Elo	For	Wet	M	61.0
Siluriformes	Pimelodidae	*Hypophthalmus edentatus*	Plk	Ter	Pel	Elo	Rnd	Wet	M	57.5
Siluriformes	Pimelodidae	*Hypophthalmus fimbriatus*	Plk	Ter	Dem	Elo	For	Dry	M	46.5
Siluriformes	Pimelodidae	*Hypophthalmus marginatus*	Plk	Ter	Dem	Elo	For	Dry	M	56.0

Order	Family	Species	FD	MP	WCP	BS	CFT	RP	MB	TL (cm)
Siluriformes	Pimelodidae	*Hypophthalmus* sp.	Plk	Ter	Dem	Elo	For	Dry	M	53.3
Siluriformes	Pimelodidae	*Leiarius marmoratus*	Car	Ter	Dem	Elo	For	Wet	M	100.0
Siluriformes	Pimelodidae	*Phractocephalus hemioliopterus*	Omn	Ter	Dem	Elo	Tru	Wet	M	135.0
Siluriformes	Pimelodidae	*Pimelodina flavipinnis*	Car	Sub	Dem	Elo	For	Wet	M	39.0
Siluriformes	Pimelodidae	*Pimelodus blochii*	Omn	Sub	Bpe	Elo	For	Wet	M	35.0
Siluriformes	Pimelodidae	*Pimelodus ornatus*	Pis	Sub	Bpe	Elo	For	Wet	M	38.5
Siluriformes	Pimelodidae	*Pinirampus pirinampu*	Omn	Sub	Dem	Elo	For	Wet	M	120.0
Siluriformes	Pimelodidae	*Platynematichthys notatus*	Pis	Ter	Dem	Elo	For	Wet	M	80.0
Siluriformes	Pimelodidae	*Platysilurus mucosus*	Omn	Sub	Dem	Elo	For	Wet	M	20.0
Siluriformes	Pimelodidae	*Platystomatichthys sturio*	Car	Sub	Dem	Elo	For	Wet	M	40.0
Siluriformes	Pimelodidae	*Propimelodus caesius*	Car	Sub	Dem	Elo	For	Wet	M	17.0
Siluriformes	Pimelodidae	*Pseudoplatystoma punctifer*	Pis	Sub	Dem	Elo	For	Wet	M	102.0
Siluriformes	Pimelodidae	*Pseudoplatystoma tigrinum*	Pis	Ter	Dem	Elo	For	Wet	M	78.5
Siluriformes	Pimelodidae	*Sorubim elongatus*	Car	Sub	Dem	Elo	For	Wet	M	32.2
Siluriformes	Pimelodidae	*Sorubim lima*	Car	Sub	Dem	Elo	For	Wet	M	54.2
Siluriformes	Pimelodidae	*Sorubim maniradii*	Car	Sub	Dem	Elo	For	Wet	M	25.6
Siluriformes	Pimelodidae	*Sorubimichthys planiceps*	Car	Sub	Dem	Elo	For	Wet	M	150.0
Siluriformes	Pimelodidae	*Zungaro zungaro*	Pis	Ter	Dem	Elo	Tru	Wet	M	140.0
Siluriformes	Trichomycteridae	*Megalocentor echthrus*	Car	Sub	Bpe	Elo	Tru	Wet	N-M	2.73
Siluriformes	Trichomycteridae	*Plectrochilus machadoi*	Hem	Sub	Bpe	Elo	For	Wet	N-M	9.3
Siluriformes	Trichomycteridae	*Vandellia sanguinea*	Hem	Sub	Dem	Elo	For	Wet	N-M	8.4

Appendix B | Taxonomic classification (Order, Family, and Species) and occurrence of fish species recorded in the study. Specie occurrence (indicated by "●") before and after reservoir filling is shown in the columns: Before and After.

Order	Family	Species	Before	After
Characiformes	Acestrorhynchidae	*Acestrorhynchus falcatus*	●	
Characiformes	Acestrorhynchidae	*Acestrorhynchus falcirostris*	●	●
Characiformes	Acestrorhynchidae	*Acestrorhynchus heterolepis*	●	●
Characiformes	Acestrorhynchidae	*Acestrorhynchus microlepis*	●	●
Characiformes	Acestrorhynchidae	*Acestrorhynchus pantaneiro*	●	
Characiformes	Acestrorhynchidae	*Acestrorhynchus* sp.		●
Characiformes	Acestrorhynchidae	*Roestes molossus*	●	●
Characiformes	Anostomidae	*Abramites hypselonotus*	●	●
Characiformes	Anostomidae	*Laemolyta proxima*	●	●
Characiformes	Anostomidae	*Laemolyta taeniata*	●	●
Characiformes	Anostomidae	*Leporellus vittatus*		●
Characiformes	Anostomidae	*Leporinus desmotes*		●
Characiformes	Anostomidae	*Leporinus fasciatus*	●	●
Characiformes	Anostomidae	*Leporinus friderici*	●	●
Characiformes	Anostomidae	*Leporinus jamesi*		●
Characiformes	Anostomidae	*Leporinus trimaculatus*		●
Characiformes	Anostomidae	*Megaleporinus trifasciatus*	●	●
Characiformes	Anostomidae	*Pseudanos gracilis*		●
Characiformes	Anostomidae	*Pseudanos trimaculatus*	●	●
Characiformes	Anostomidae	*Rhytiodus argenteofuscus*	●	●
Characiformes	Anostomidae	*Rhytiodus microlepis*	●	●
Characiformes	Anostomidae	*Rhytiodus* sp.		●
Characiformes	Anostomidae	*Schizodon fasciatus*	●	●
Characiformes	Bryconidae	*Brycon amazonicus*	●	●
Characiformes	Bryconidae	*Brycon melanopterus*	●	●
Characiformes	Bryconidae	*Brycon pesu*		●
Characiformes	Bryconidae	*Salminus brasiliensis*		●
Characiformes	Bryconidae	*Salminus* sp.		●
Characiformes	Chalceidae	*Chalceus epakros*	●	
Characiformes	Chalceidae	*Chalceus guaporensis*	●	●
Characiformes	Characidae	*Acestrocephalus pallidus*	●	
Characiformes	Characidae	*Agoniates anchovia*	●	●
Characiformes	Characidae	*Agoniates halecinus*	●	●
Characiformes	Characidae	*Astyanax anterior*	●	●
Characiformes	Characidae	*Astyanax bimaculatus*		●
Characiformes	Characidae	*Astyanax* sp.		●
Characiformes	Characidae	*Catoprion mento*	●	
Characiformes	Characidae	*Charax* sp.	●	●
Characiformes	Characidae	*Ctenobrycon spilurus*	●	●
Characiformes	Characidae	*Cynopotamus gouldingi*	●	●
Characiformes	Characidae	*Galeocharax goeldii*	●	●
Characiformes	Characidae	*Jupiaba zonata*		●
Characiformes	Characidae	*Knodus heteresthes*	●	

Order	Family	Species	Before	After
Characiformes	Characidae	*Moenkhausia bonita*		●
Characiformes	Characidae	*Moenkhausia cotinho*		●
Characiformes	Characidae	*Moenkhausia dichroura*	●	●
Characiformes	Characidae	*Moenkhausia grandisquamis*	●	●
Characiformes	Characidae	*Moenkhausia jamesi*	●	●
Characiformes	Characidae	*Moenkhausia lepidura*	●	●
Characiformes	Characidae	*Moenkhausia* sp.		●
Characiformes	Characidae	*Parecbasis cyclolepis*	●	●
Characiformes	Characidae	*Poptella compressa*	●	●
Characiformes	Characidae	*Roeboides affinis*	●	●
Characiformes	Characidae	*Roeboides myersii*	●	●
Characiformes	Characidae	*Tetragonopterus argenteus*	●	●
Characiformes	Characidae	*Tetragonopterus chalceus*	●	
Characiformes	Chilodontidae	*Caenotropus labyrinthicus*	●	●
Characiformes	Chilodontidae	*Caenotropus schizodon*	●	●
Characiformes	Curimatidae	*Curimata inornata*	●	●
Characiformes	Curimatidae	*Curimata knerii*	●	●
Characiformes	Curimatidae	*Curimata roseni*	●	●
Characiformes	Curimatidae	*Curimata vittata*	●	●
Characiformes	Curimatidae	*Curimatella alburnus*	●	●
Characiformes	Curimatidae	*Curimatella dorsalis*		●
Characiformes	Curimatidae	*Curimatella immaculata*		●
Characiformes	Curimatidae	*Curimatella meyeri*	●	●
Characiformes	Curimatidae	*Curimatella* sp.		●
Characiformes	Curimatidae	*Cyphocharax notatus*	●	●
Characiformes	Curimatidae	*Cyphocharax plumbeus*	●	●
Characiformes	Curimatidae	*Cyphocharax* sp.		●
Characiformes	Curimatidae	*Potamorhina altamazonica*	●	●
Characiformes	Curimatidae	*Potamorhina latior*	●	●
Characiformes	Curimatidae	*Potamorhina pristigaster*		●
Characiformes	Curimatidae	*Psectrogaster amazonica*	●	●
Characiformes	Curimatidae	*Psectrogaster essequibensis*	●	●
Characiformes	Curimatidae	*Psectrogaster rutiloides*	●	●
Characiformes	Curimatidae	*Steindachnerina bimaculata*	●	●
Characiformes	Curimatidae	*Steindachnerina dobula*		●
Characiformes	Curimatidae	*Steindachnerina hypostoma*	●	
Characiformes	Curimatidae	*Steindachnerina leucisca*	●	
Characiformes	Curimatidae	*Steindachnerina planiventris*	●	
Characiformes	Cynodontidae	*Cynodon gibbus*	●	●
Characiformes	Cynodontidae	*Hydrolycus armatus*	●	●
Characiformes	Cynodontidae	*Hydrolycus scomberoides*	●	●
Characiformes	Cynodontidae	*Rhaphiodon vulpinus*	●	●
Characiformes	Erythrinidae	*Hoplias malabaricus*	●	●
Characiformes	Gasteropelecidae	*Thoracocharax stellatus*		●
Characiformes	Hemiodontidae	*Anodus elongatus*	●	●
Characiformes	Hemiodontidae	*Anodus orinocensis*	●	●

Order	Family	Species	Before	After
Characiformes	Hemiodontidae	*Argonectes longiceps*		●
Characiformes	Hemiodontidae	*Hemiodus immaculatus*	●	●
Characiformes	Hemiodontidae	*Hemiodus microlepis*	●	●
Characiformes	Hemiodontidae	*Hemiodus semitaeniatus*		●
Characiformes	Hemiodontidae	*Hemiodus* sp.	●	●
Characiformes	Hemiodontidae	*Hemiodus unimaculatus*	●	●
Characiformes	Iguanodectidae	*Bryconops alburnoides*	●	●
Characiformes	Iguanodectidae	*Bryconops caudomaculatus*	●	●
Characiformes	Iguanodectidae	*Bryconops giacopinii*		●
Characiformes	Prochilodontidae	*Prochilodus nigricans*	●	●
Characiformes	Prochilodontidae	*Semaprochilodus insignis*	●	●
Characiformes	Prochilodontidae	*Semaprochilodus taeniurus*	●	●
Characiformes	Serrasalmidae	*Colossoma macropomum*	●	●
Characiformes	Serrasalmidae	*Metynnis guaporensis*	●	●
Characiformes	Serrasalmidae	*Metynnis hypsauchen*	●	●
Characiformes	Serrasalmidae	*Metynnis lippincottianus*	●	
Characiformes	Serrasalmidae	*Metynnis luna*	●	●
Characiformes	Serrasalmidae	*Metynnis maculatus*	●	●
Characiformes	Serrasalmidae	*Myleus setiger*	●	●
Characiformes	Serrasalmidae	*Myloplus asterias*	●	●
Characiformes	Serrasalmidae	*Myloplus lobatus*	●	●
Characiformes	Serrasalmidae	*Myloplus rubripinnis*	●	●
Characiformes	Serrasalmidae	*Myloplus* sp.		●
Characiformes	Serrasalmidae	*Mylossoma aureum*	●	●
Characiformes	Serrasalmidae	*Mylossoma duriventre*	●	●
Characiformes	Serrasalmidae	*Piaractus brachypomus*	●	●
Characiformes	Serrasalmidae	*Pristobrycon* sp.	●	●
Characiformes	Serrasalmidae	*Pygocentrus nattereri*	●	●
Characiformes	Serrasalmidae	*Serrasalmus compressus*	●	●
Characiformes	Serrasalmidae	*Serrasalmus eigenmanni*	●	●
Characiformes	Serrasalmidae	*Serrasalmus elongatus*	●	●
Characiformes	Serrasalmidae	*Serrasalmus hollandi*	●	●
Characiformes	Serrasalmidae	*Serrasalmus maculatus*	●	●
Characiformes	Serrasalmidae	*Serrasalmus rhombeus*	●	●
Characiformes	Serrasalmidae	*Serrasalmus* sp.1	●	
Characiformes	Serrasalmidae	*Serrasalmus* sp.2	●	●
Characiformes	Serrasalmidae	*Serrasalmus* sp.3		●
Characiformes	Serrasalmidae	*Serrasalmus* sp.4		●
Characiformes	Serrasalmidae	*Serrasalmus* sp.5		●
Characiformes	Serrasalmidae	*Serrasalmus spilopleura*	●	●
Characiformes	Triportheidae	*Triportheus albus*	●	●
Characiformes	Triportheidae	*Triportheus angulatus*	●	●
Characiformes	Triportheidae	*Triportheus auritus*	●	●
Characiformes	Triportheidae	*Triportheus culter*	●	●
Characiformes	Triportheidae	*Triportheus curtus*		●
Characiformes	Triportheidae	*Triportheus rotundatus*		●

Order	Family	Species	Before	After
Clupeiformes	Engraulidae	*Anchoviella carrikeri*		●
Clupeiformes	Engraulidae	*Jurengraulis juruensis*	●	●
Clupeiformes	Engraulidae	*Lycengraulis batesii*		●
Clupeiformes	Pristigasteridae	*Ilisha amazonica*		●
Clupeiformes	Pristigasteridae	*Pellona castelnaeana*	●	●
Clupeiformes	Pristigasteridae	*Pellona flavipinnis*	●	●
Gymnotiformes	Apteronotidae	*Adontosternarchus balaenops*		●
Gymnotiformes	Apteronotidae	*Adontosternarchus clarkae*		●
Gymnotiformes	Apteronotidae	*Apteronotus albifrons*		●
Gymnotiformes	Apteronotidae	*Apteronotus bonapartii*		●
Gymnotiformes	Apteronotidae	*Apteronotus* sp.		●
Gymnotiformes	Apteronotidae	*Compsaraia compsa*		●
Gymnotiformes	Apteronotidae	*Parapteronotus hasemani*		●
Gymnotiformes	Apteronotidae	*Porotergus* sp.		●
Gymnotiformes	Apteronotidae	*Sternarchella schotti*		●
Gymnotiformes	Apteronotidae	*Sternarchella sima*		●
Gymnotiformes	Apteronotidae	*Sternarchella* sp.		●
Gymnotiformes	Apteronotidae	*Sternarchorhynchus chaoi*		●
Gymnotiformes	Apteronotidae	*Sternarchorhynchus sp*		●
Gymnotiformes	Gymnotidae	*Electrophorus electricus*	●	●
Gymnotiformes	Gymnotidae	*Gymnotus carapo*		●
Gymnotiformes	Gymnotidae	*Gymnotus curupira*		●
Gymnotiformes	Gymnotidae	*Gymnotus* sp.		●
Gymnotiformes	Hypopomidae	*Brachyhypopomus pinnicaudatus*		●
Gymnotiformes	Hypopomidae	*Steatogenys elegans*	●	●
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys marmoratus*	●	●
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys pantherinus*		●
Gymnotiformes	Rhamphichthyidae	*Rhamphichthys rostratus*	●	●
Gymnotiformes	Sternopygidae	*Distocyclus conirostris*		●
Gymnotiformes	Sternopygidae	*Eigenmannia limbata*	●	●
Gymnotiformes	Sternopygidae	*Eigenmannia macrops*		●
Gymnotiformes	Sternopygidae	*Rhabdolichops troscheli*		●
Gymnotiformes	Sternopygidae	*Sternopygus macrurus*	●	●
Osteoglossiformes	Arapaimidae	*Arapaima gigas*	●	●
Perciformes	Cichlidae	*Acaronia nassa*	●	
Perciformes	Cichlidae	*Aequidens* sp.		●
Perciformes	Cichlidae	*Aequidens tetramerus*	●	●
Perciformes	Cichlidae	*Astronotus crassipinnis*	●	●
Perciformes	Cichlidae	*Biotodoma cupido*	●	●
Perciformes	Cichlidae	*Chaetobranchus flavescens*	●	●
Perciformes	Cichlidae	*Cichla pleiozona*	●	●
Perciformes	Cichlidae	*Crenicichla cyanonotus*		●
Perciformes	Cichlidae	*Crenicichla reticulata*		●
Perciformes	Cichlidae	*Crenicichla* sp.		●
Perciformes	Cichlidae	*Crenicichla strigata*	●	
Perciformes	Cichlidae	*Geophagus megasema*	●	●

Order	Family	Species	Before	After
Perciformes	Cichlidae	*Heros spurius*	●	●
Perciformes	Cichlidae	*Lugubria adspersa*	●	
Perciformes	Cichlidae	*Lugubria johanna*		●
Perciformes	Cichlidae	*Lugubria marmorata*	●	●
Perciformes	Cichlidae	*Mesonauta festivus*	●	●
Perciformes	Cichlidae	*Satanoperca jurupari*	●	●
Perciformes	Cichlidae	*Satanoperca* sp.	●	
Perciformes	Cichlidae	*Saxatilia inpa*	●	
Perciformes	Cichlidae	*Saxatilia semicincta*		●
Perciformes	Sciaenidae	*Pachypops fourcroi*		●
Perciformes	Sciaenidae	*Pachyurus paucirastrus*		●
Perciformes	Sciaenidae	*Plagioscion montei*	●	
Perciformes	Sciaenidae	*Plagioscion squamosissimus*	●	●
Pleuronectiformes	Achiridae	*Hypoclinemus mentalis*	●	●
Siluriformes	Aspredinidae	*Bunocephalus coracoideus*		●
Siluriformes	Auchenipteridae	*Ageneiosus inermis*	●	●
Siluriformes	Auchenipteridae	*Ageneiosus* sp.1		●
Siluriformes	Auchenipteridae	*Ageneiosus* sp.2	●	●
Siluriformes	Auchenipteridae	*Ageneiosus* sp.3		●
Siluriformes	Auchenipteridae	*Ageneiosus ucayalensis*	●	●
Siluriformes	Auchenipteridae	*Ageneiosus uranophthalmus*	●	●
Siluriformes	Auchenipteridae	*Ageneiosus vittatus*	●	
Siluriformes	Auchenipteridae	*Auchenipterichthys coracoideus*	●	●
Siluriformes	Auchenipteridae	*Auchenipterichthys longimanus*	●	
Siluriformes	Auchenipteridae	*Auchenipterichthys thoracatus*	●	●
Siluriformes	Auchenipteridae	*Auchenipterus ambyiacus*	●	●
Siluriformes	Auchenipteridae	*Auchenipterus brachyurus*	●	●
Siluriformes	Auchenipteridae	*Auchenipterus britskii*	●	
Siluriformes	Auchenipteridae	*Auchenipterus nuchalis*	●	●
Siluriformes	Auchenipteridae	*Auchenipterus* sp.		●
Siluriformes	Auchenipteridae	*Centromochlus heckelii*	●	●
Siluriformes	Auchenipteridae	*Epapterus dispilurus*	●	●
Siluriformes	Auchenipteridae	*Tatia aulopygia*	●	●
Siluriformes	Auchenipteridae	*Tatia intermedia*		●
Siluriformes	Auchenipteridae	*Trachelyopterus galeatus*	●	●
Siluriformes	Auchenipteridae	*Trachelyopterus porosus*	●	●
Siluriformes	Auchenipteridae	*Trachelyopterus* sp.	●	
Siluriformes	Auchenipteridae	*Trachycorystes trachycorystes*	●	
Siluriformes	Auchenipteridae	*Tympanopleura atronasus*	●	●
Siluriformes	Auchenipteridae	*Tympanopleura brevis*	●	●
Siluriformes	Auchenipteridae	*Tympanopleura longipinna*		●
Siluriformes	Callichthyidae	*Hoplosternum littorale*	●	●
Siluriformes	Cetopsidae	*Cetopsis candiru*		●
Siluriformes	Cetopsidae	*Cetopsis coecutiens*	●	●
Siluriformes	Doradidae	*Acanthodoras spinosissimus*	●	
Siluriformes	Doradidae	*Amblydoras affinis*	●	

Order	Family	Species	Before	After
Siluriformes	Doradidae	*Anadoras weddellii*		●
Siluriformes	Doradidae	*Astrodoras fulcro*	●	
Siluriformes	Doradidae	*Centrodoras brachiatus*	●	●
Siluriformes	Doradidae	*Hemidoras boulengeri*	●	●
Siluriformes	Doradidae	*Hemidoras morrisi*	●	●
Siluriformes	Doradidae	*Hemidoras stenopeltis*	●	●
Siluriformes	Doradidae	*Hemidoras stuebelii*	●	●
Siluriformes	Doradidae	*Megalodoras uranoscopus*		●
Siluriformes	Doradidae	*Nemadoras elongatus*		●
Siluriformes	Doradidae	*Nemadoras humeralis*	●	●
Siluriformes	Doradidae	*Ossancora asterophysa*	●	●
Siluriformes	Doradidae	*Ossancora fimbriata*		●
Siluriformes	Doradidae	*Ossancora punctata*	●	●
Siluriformes	Doradidae	*Oxydoras niger*	●	●
Siluriformes	Doradidae	*Platydoras armatulus*	●	●
Siluriformes	Doradidae	*Pterodoras granulosus*	●	●
Siluriformes	Doradidae	*Scorpiodoras liophysus*	●	●
Siluriformes	Doradidae	*Trachydoras brevis*	●	●
Siluriformes	Doradidae	*Trachydoras microstomus*	●	●
Siluriformes	Doradidae	*Trachydoras paraguayensis*	●	●
Siluriformes	Doradidae	*Trachydoras* sp.		●
Siluriformes	Doradidae	*Trachydoras steindachneri*		●
Siluriformes	Heptapteridae	*Pimelodella boliviana*		●
Siluriformes	Heptapteridae	*Pimelodella howesi*	●	●
Siluriformes	Heptapteridae	*Pimelodella* sp.	●	
Siluriformes	Loricariidae	*Acanthicus hystrix*		●
Siluriformes	Loricariidae	*Ancistrus dubius*	●	●
Siluriformes	Loricariidae	*Ancistrus* sp.1	●	
Siluriformes	Loricariidae	*Ancistrus* sp.2	●	●
Siluriformes	Loricariidae	*Ancistrus* sp.3		●
Siluriformes	Loricariidae	*Aphanotorulus emarginatus*	●	●
Siluriformes	Loricariidae	*Aphanotorulus unicolor*		●
Siluriformes	Loricariidae	*Farlowella smithi*		●
Siluriformes	Loricariidae	*Hemiodontichthys acipenserinus*	●	
Siluriformes	Loricariidae	*Hypoptopoma incognitum*	●	●
Siluriformes	Loricariidae	*Hypostomus hoplonites*	●	●
Siluriformes	Loricariidae	*Hypostomus plecostomus*	●	●
Siluriformes	Loricariidae	*Hypostomus pyrineusi*	●	●
Siluriformes	Loricariidae	*Hypostomus* sp.	●	
Siluriformes	Loricariidae	*Loricaria cataphracta*	●	●
Siluriformes	Loricariidae	*Loricariichthys acutus*	●	●
Siluriformes	Loricariidae	*Loricariichthys maculatus*	●	●
Siluriformes	Loricariidae	*Loricariichthys nudirostris*		●
Siluriformes	Loricariidae	*Loricariichthys* sp.1		●
Siluriformes	Loricariidae	*Loricariichthys* sp.2	●	
Siluriformes	Loricariidae	*Peckoltichthys bachi*	●	●

Order	Family	Species	Before	After
Siluriformes	Loricariidae	*Pterygoplichthys lituratus*	●	●
Siluriformes	Loricariidae	*Pterygoplichthys pardalis*	●	●
Siluriformes	Loricariidae	*Rineloricaria castroi*	●	
Siluriformes	Loricariidae	*Rineloricaria formosa*		●
Siluriformes	Loricariidae	*Sturisoma lyra*	●	
Siluriformes	Pimelodidae	*Brachyplatystoma capapretum*		●
Siluriformes	Pimelodidae	*Brachyplatystoma filamentosum*		●
Siluriformes	Pimelodidae	*Brachyplatystoma platynemum*		●
Siluriformes	Pimelodidae	*Brachyplatystoma rousseauxii*	●	●
Siluriformes	Pimelodidae	*Brachyplatystoma tigrinum*		●
Siluriformes	Pimelodidae	*Brachyplatystoma vaillantii*	●	●
Siluriformes	Pimelodidae	*Calophysus macropterus*	●	●
Siluriformes	Pimelodidae	*Duopalatinus peruanus*		●
Siluriformes	Pimelodidae	*Hemisorubim platyrhynchos*	●	●
Siluriformes	Pimelodidae	*Hypophthalmus edentatus*	●	●
Siluriformes	Pimelodidae	*Hypophthalmus fimbriatus*		●
Siluriformes	Pimelodidae	*Hypophthalmus marginatus*	●	●
Siluriformes	Pimelodidae	*Hypophthalmus* sp.		●
Siluriformes	Pimelodidae	*Leiarius marmoratus*	●	●
Siluriformes	Pimelodidae	*Phractocephalus hemioliopterus*		●
Siluriformes	Pimelodidae	*Pimelodina flavipinnis*		●
Siluriformes	Pimelodidae	*Pimelodus blochii*	●	●
Siluriformes	Pimelodidae	*Pimelodus ornatus*	●	●
Siluriformes	Pimelodidae	*Pinirampus pirinampu*	●	●
Siluriformes	Pimelodidae	*Platynematichthys notatus*	●	●
Siluriformes	Pimelodidae	*Platysilurus mucosus*	●	●
Siluriformes	Pimelodidae	*Platystomatichthys sturio*		●
Siluriformes	Pimelodidae	*Propimelodus caesius*		●
Siluriformes	Pimelodidae	*Pseudoplatystoma punctifer*	●	●
Siluriformes	Pimelodidae	*Pseudoplatystoma tigrinum*	●	●
Siluriformes	Pimelodidae	*Sorubim elongatus*	●	●
Siluriformes	Pimelodidae	*Sorubim lima*	●	●
Siluriformes	Pimelodidae	*Sorubim maniradii*	●	●
Siluriformes	Pimelodidae	*Sorubimichthys planiceps*	●	●
Siluriformes	Pimelodidae	*Zungaro zungaro*		●
Siluriformes	Trichomycteridae	*Megalocentor echthrus*		●
Siluriformes	Trichomycteridae	*Plectrochilus machadoi*		●
Siluriformes	Trichomycteridae	*Vandellia sanguinea*		●

Appendix C | Distribution of abundance (number of individuals), species richness (Richness), Shannon-Wiener diversity (Shannon) and Simpson dominance (Simpson), and Pielou's evenness (Pielou) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

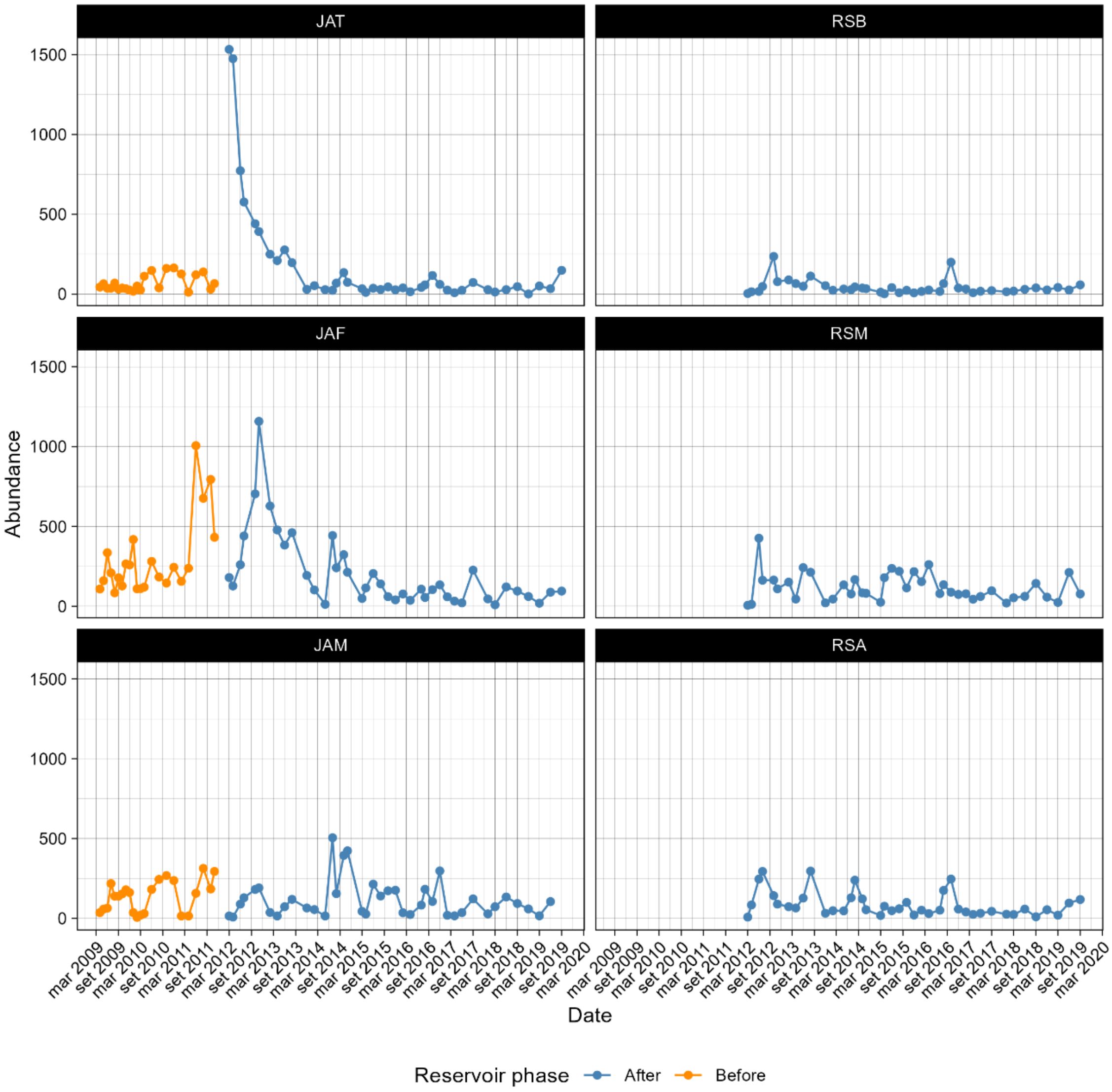

Figure 7 | Distribution of abundance (number of individuals) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

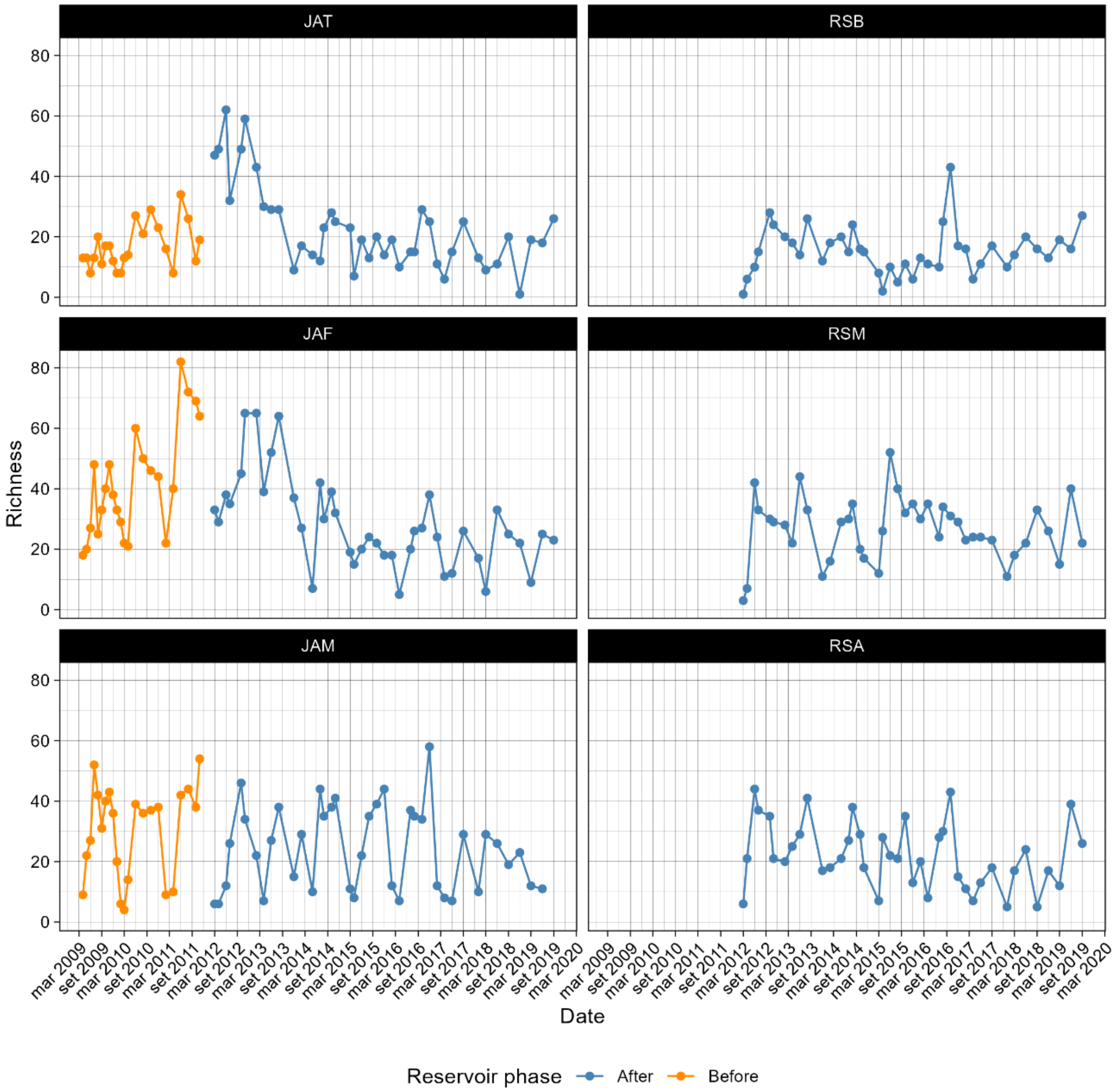

Figure 8 | Distribution of species richness (Richness) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

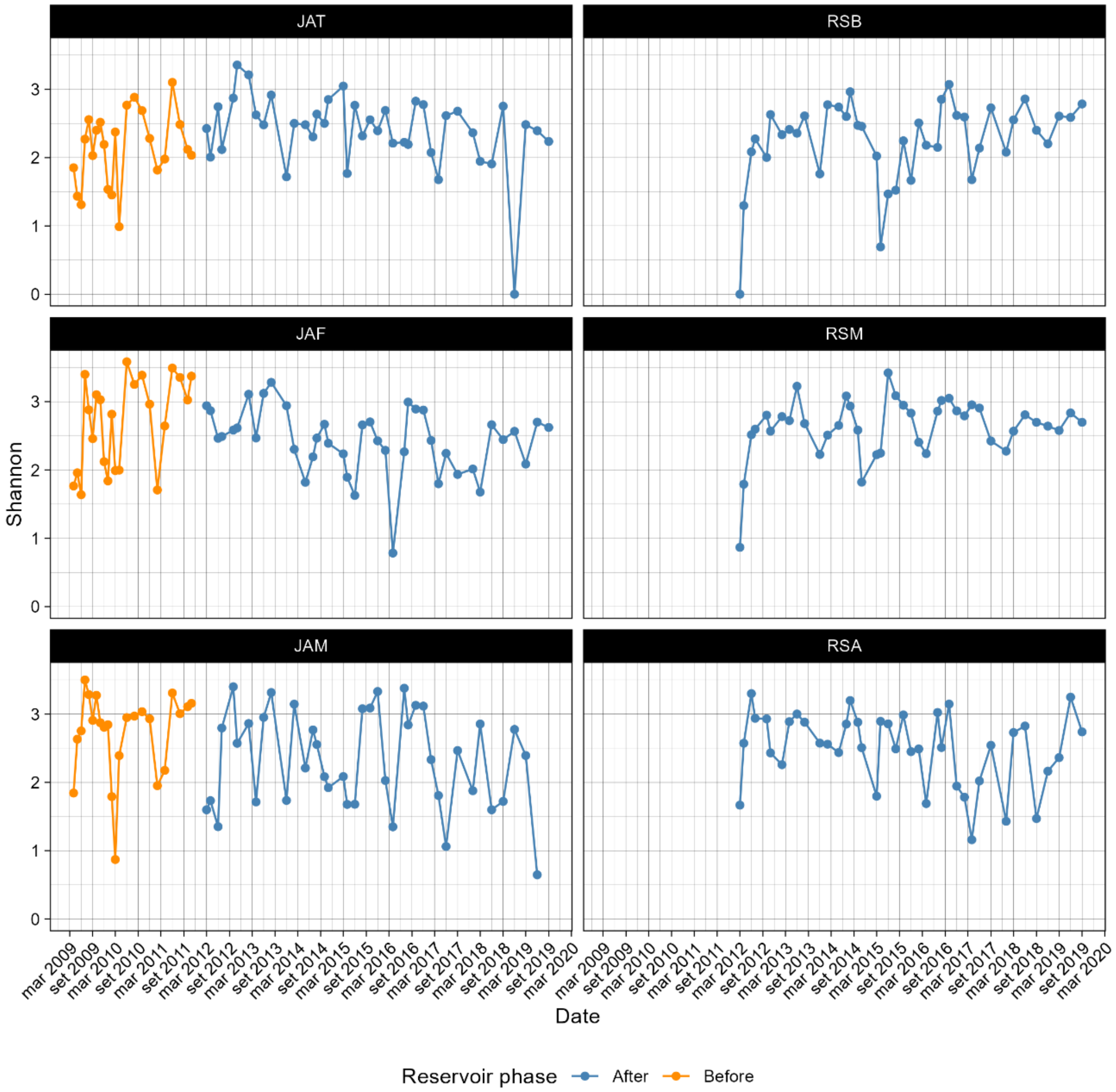

Figure 9 | Distribution of Shannon-Wiener diversity (Shannon) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

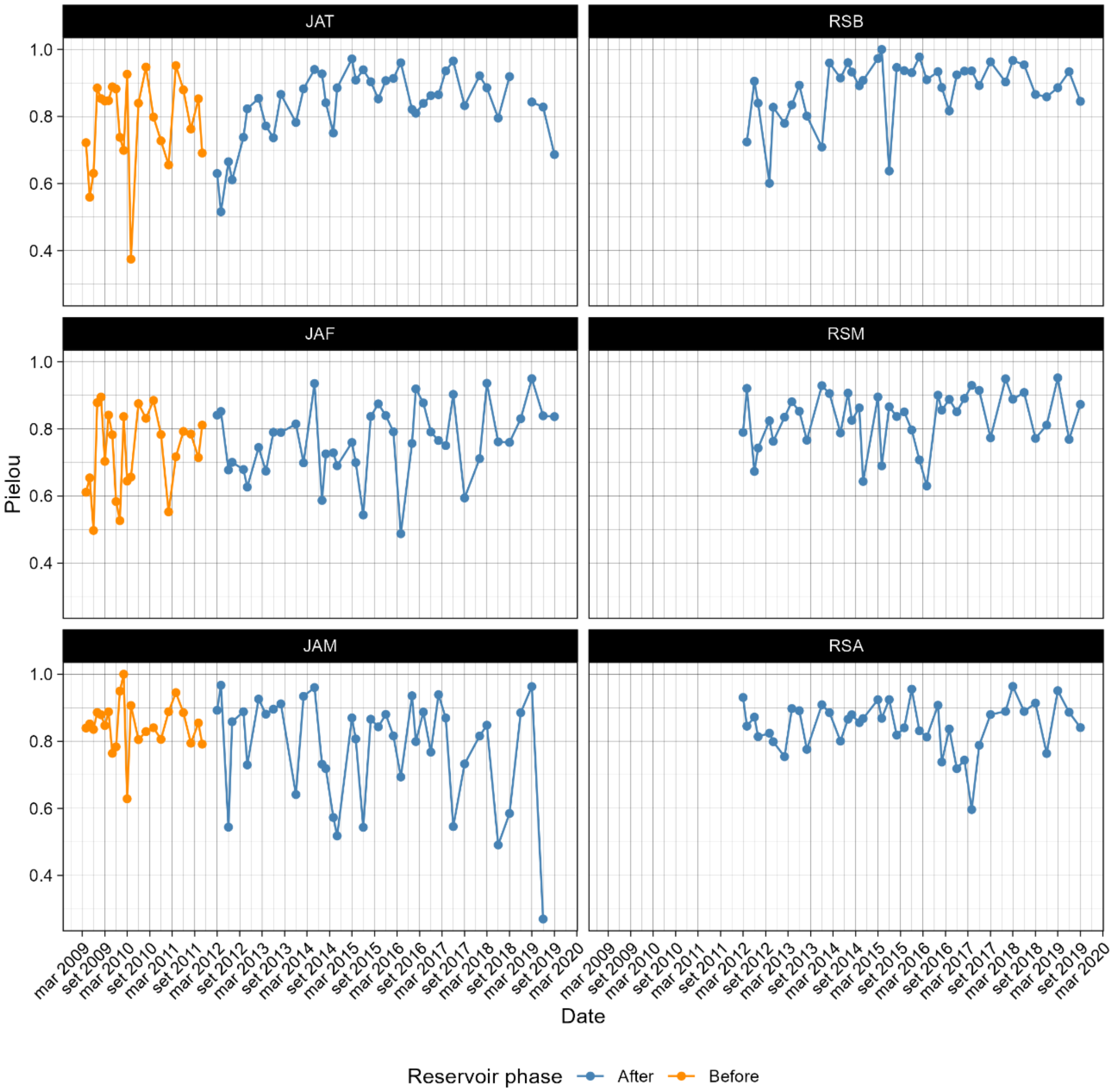

Figure 10 | Distribution of Pielou's evenness (Pielou) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

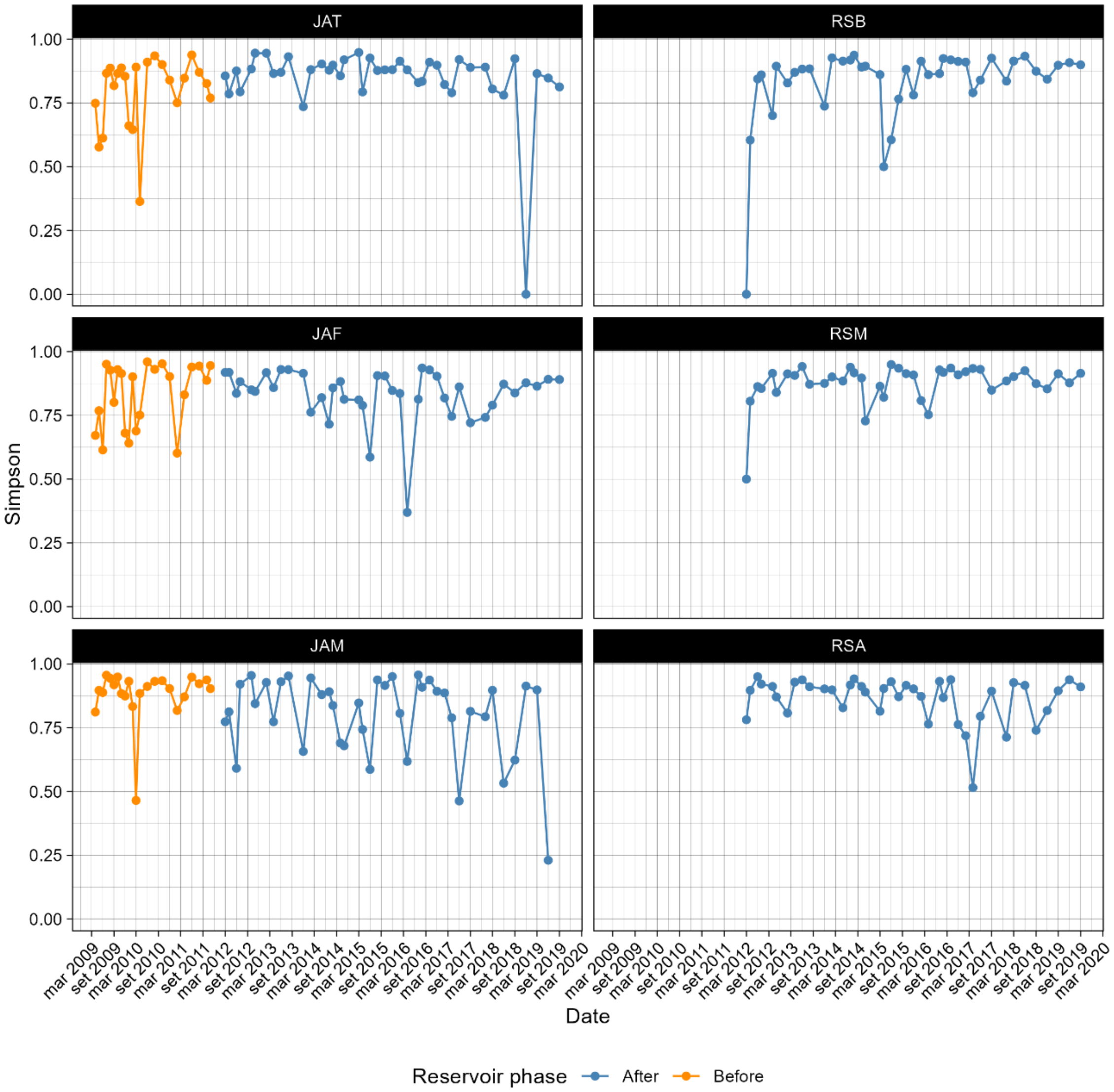

Figure 11 | Distribution of Simpson dominance (Simpson) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

Appendix D | Distribution of Functional diversity indices - Functional Entity richness (FE), Functional Richness (FRic), Functional Evenness (FEve), Dispersion (FDis), Functional Divergence (FDiv), Functional Specialization (FSpe), Mean Pairwise Distance (FMPD), Mean Nearest Neighbour Distance (FNND), and Functional Originality (FOri) - recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

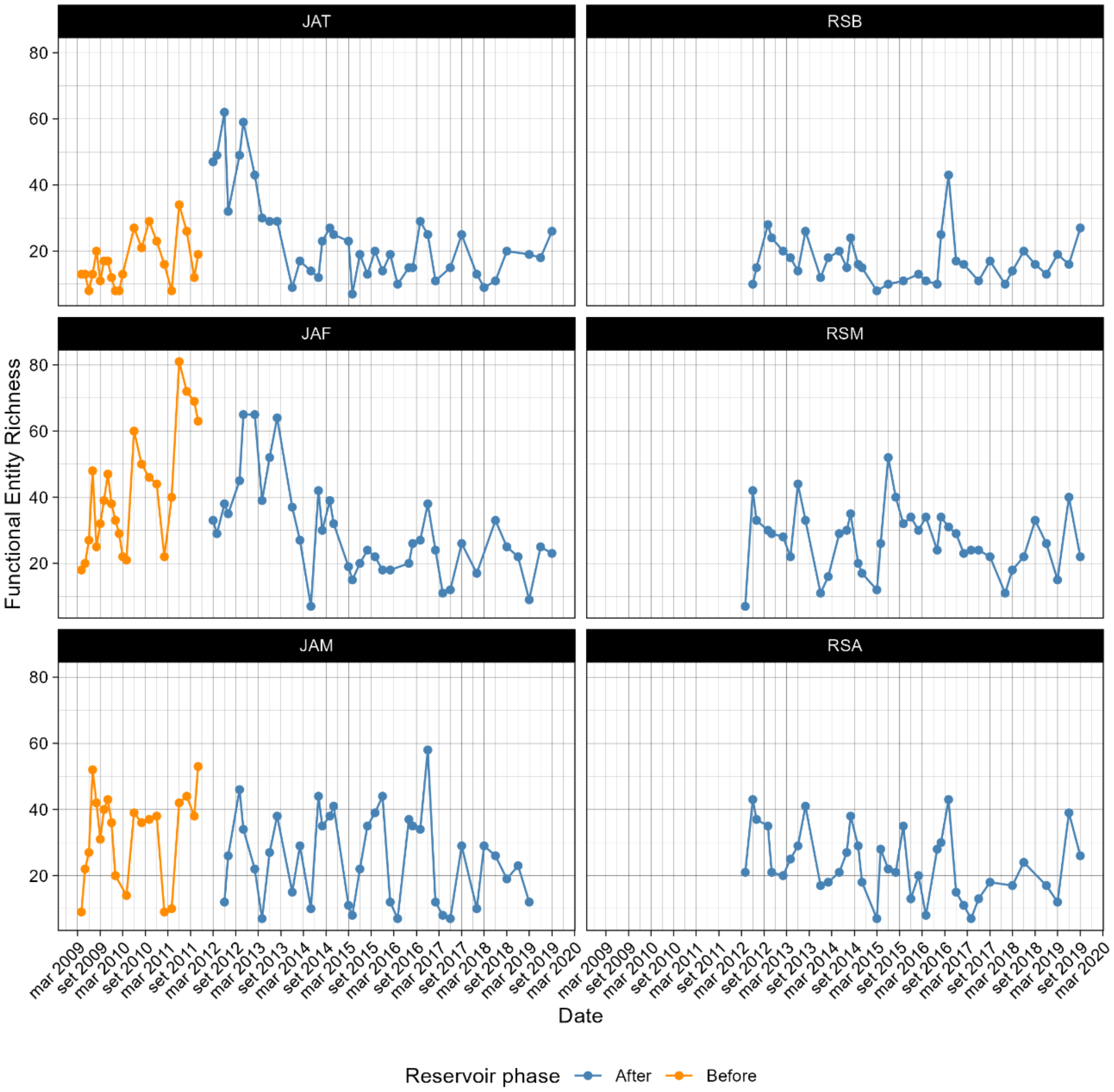

Figure 12 | Distribution of Functional Entity richness (FE) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

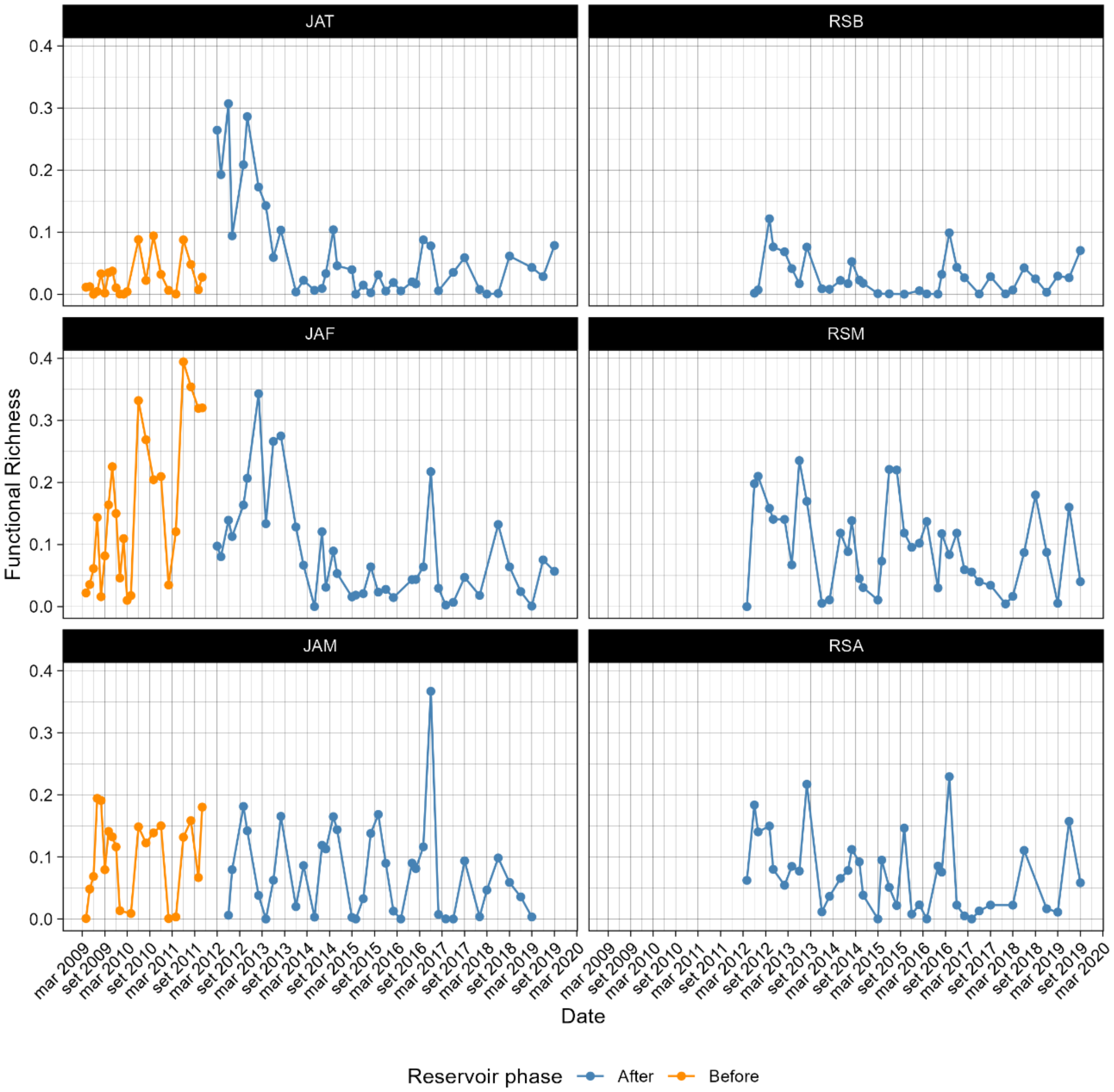

Figure 13 | Distribution of Functional Richness (Fric) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

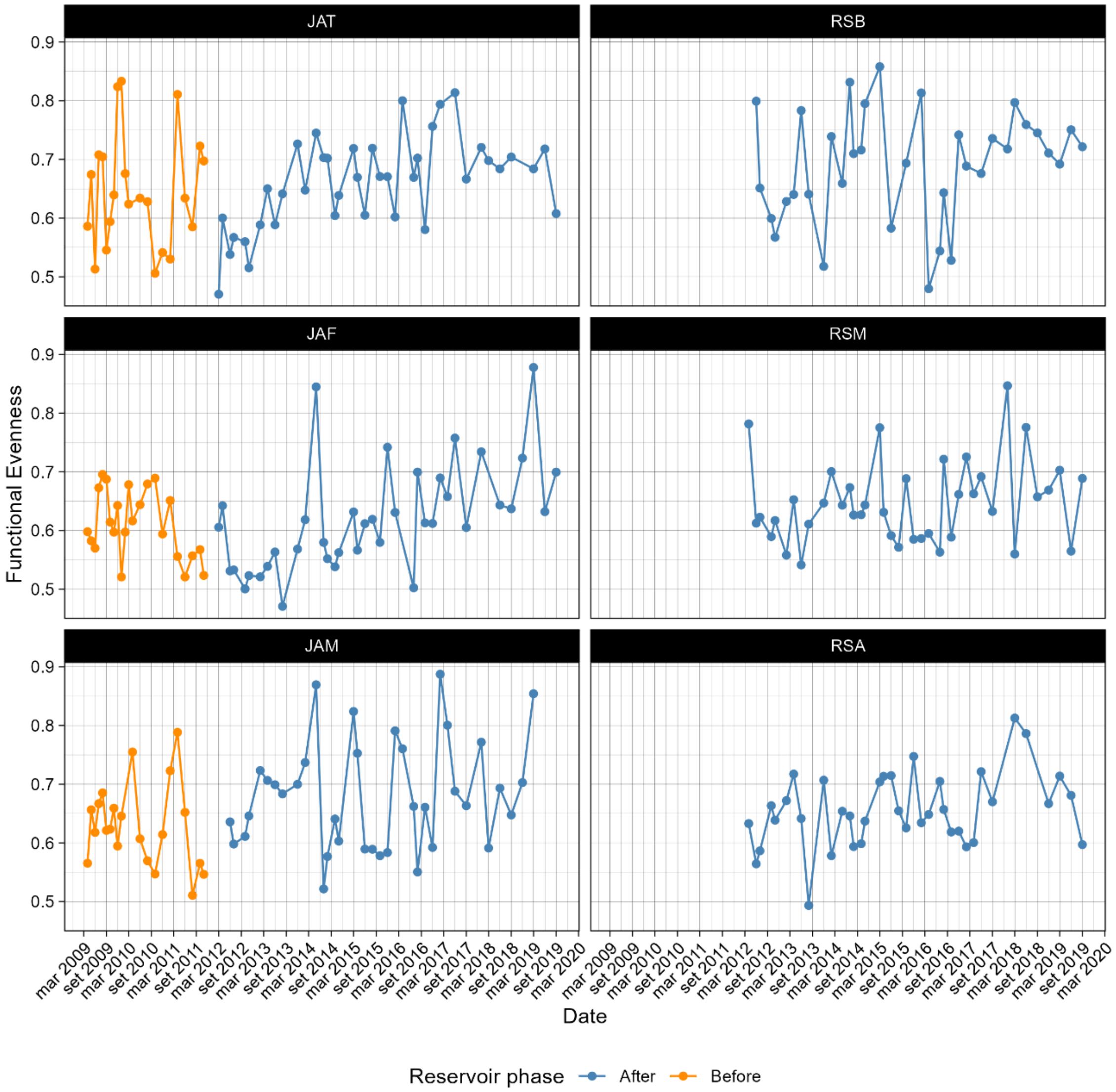

Figure 14 | Distribution of Functional Evenness (FEve) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

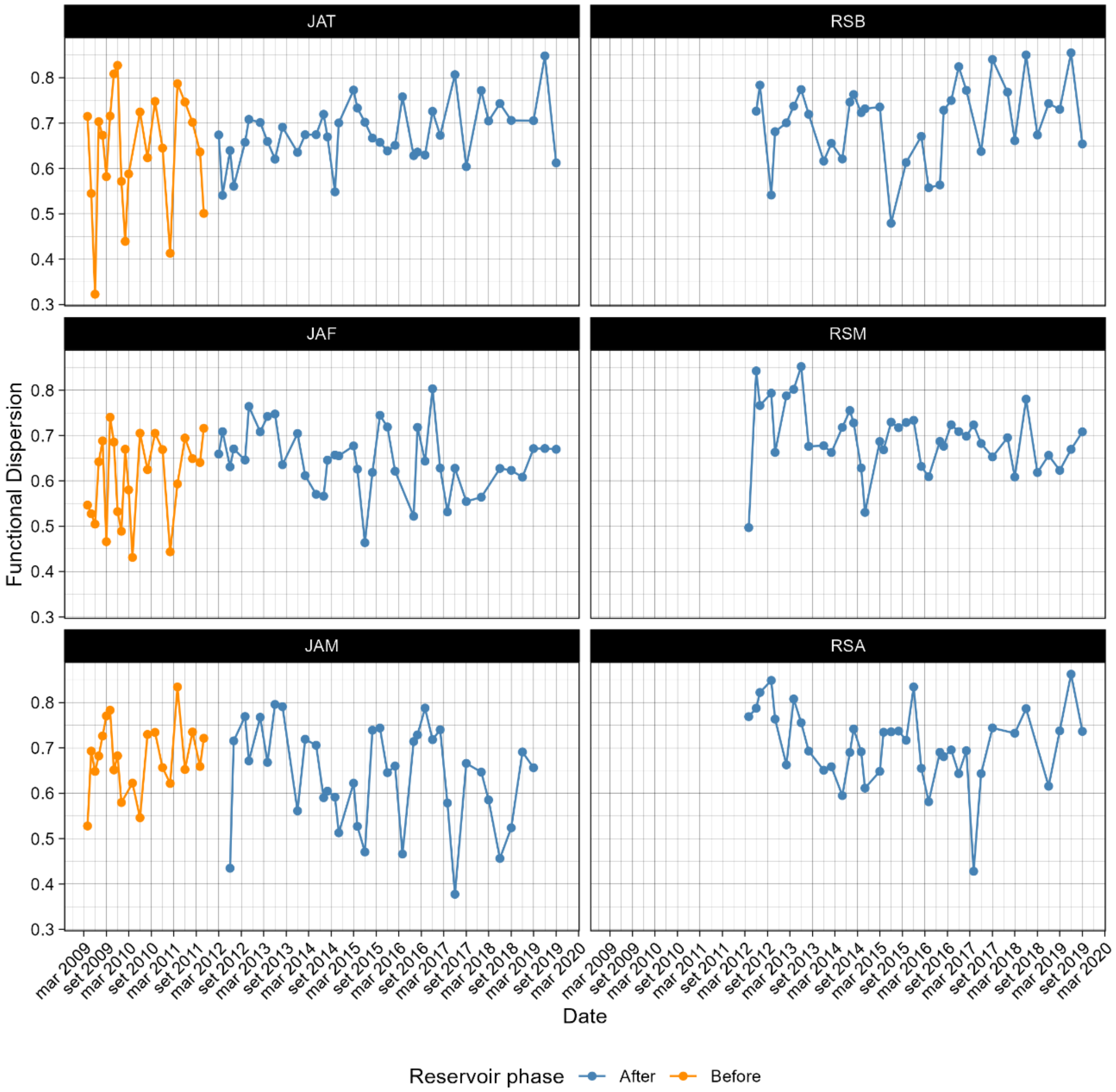

Figure 15 | Distribution of Functional Dispersion (FDis) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

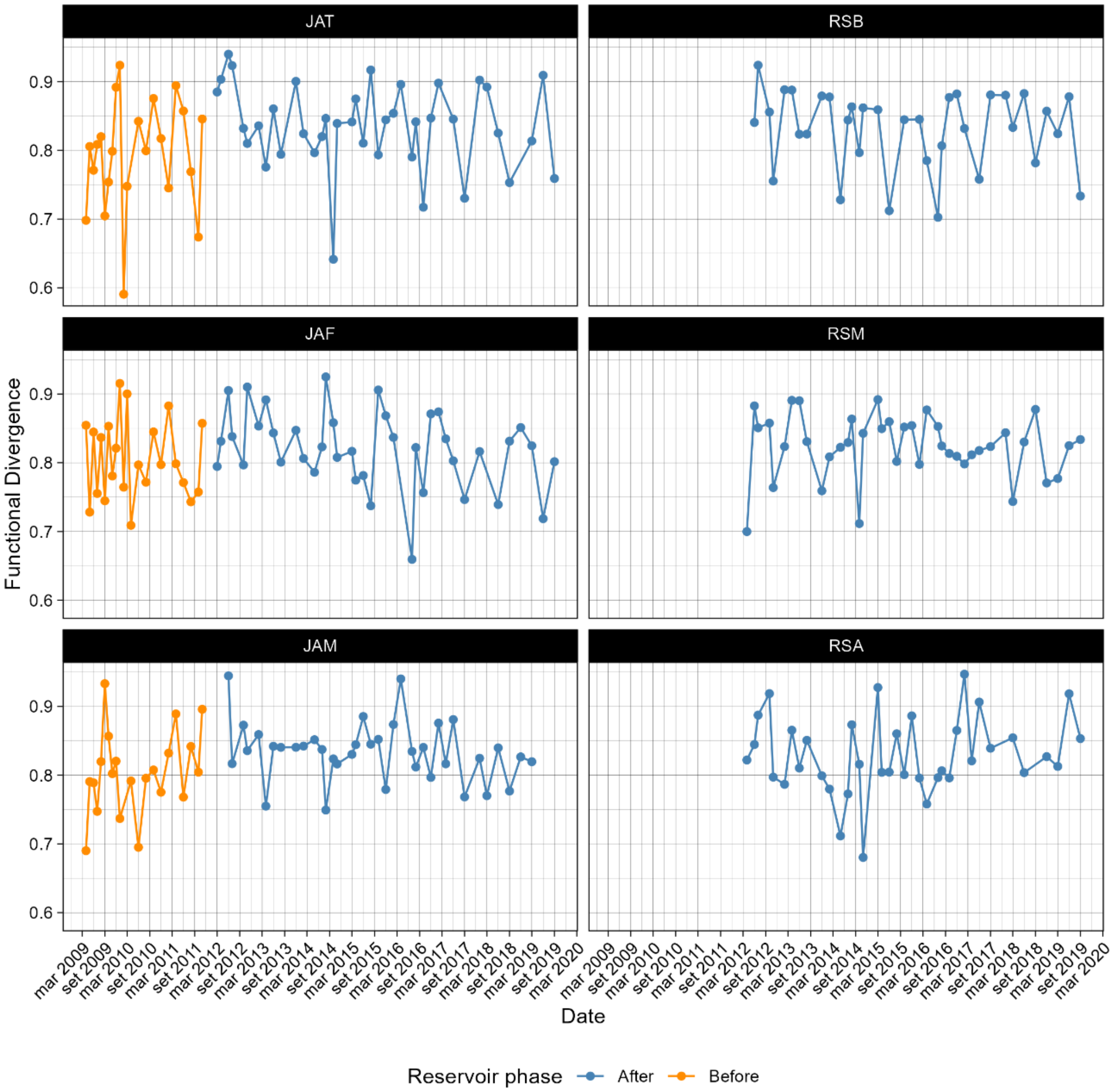

Figure 16 | Distribution of Functional Divergence (FDiv) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

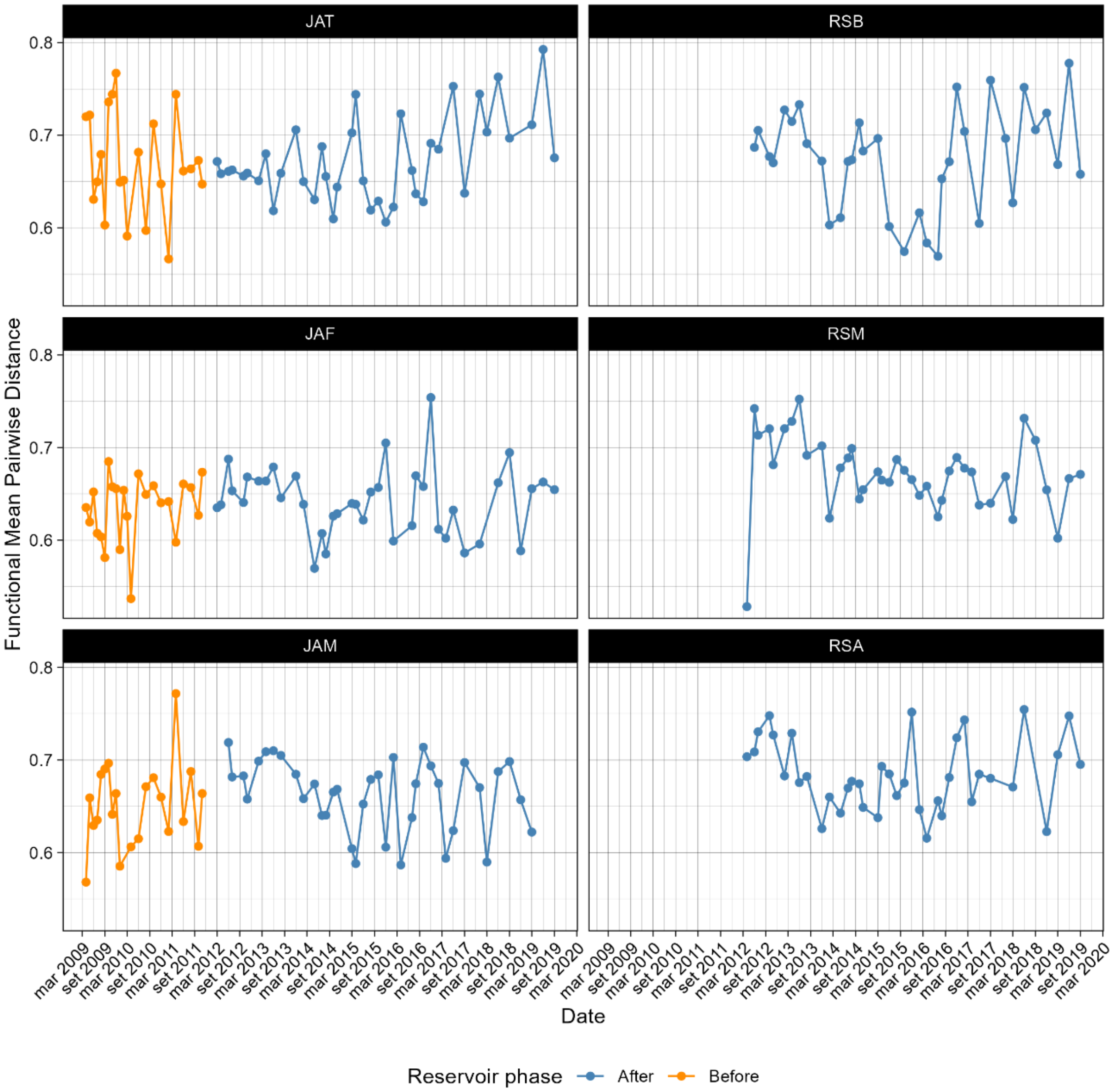

Figure 17 | Distribution of Functional Mean Pairwise Distance (FMPD) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

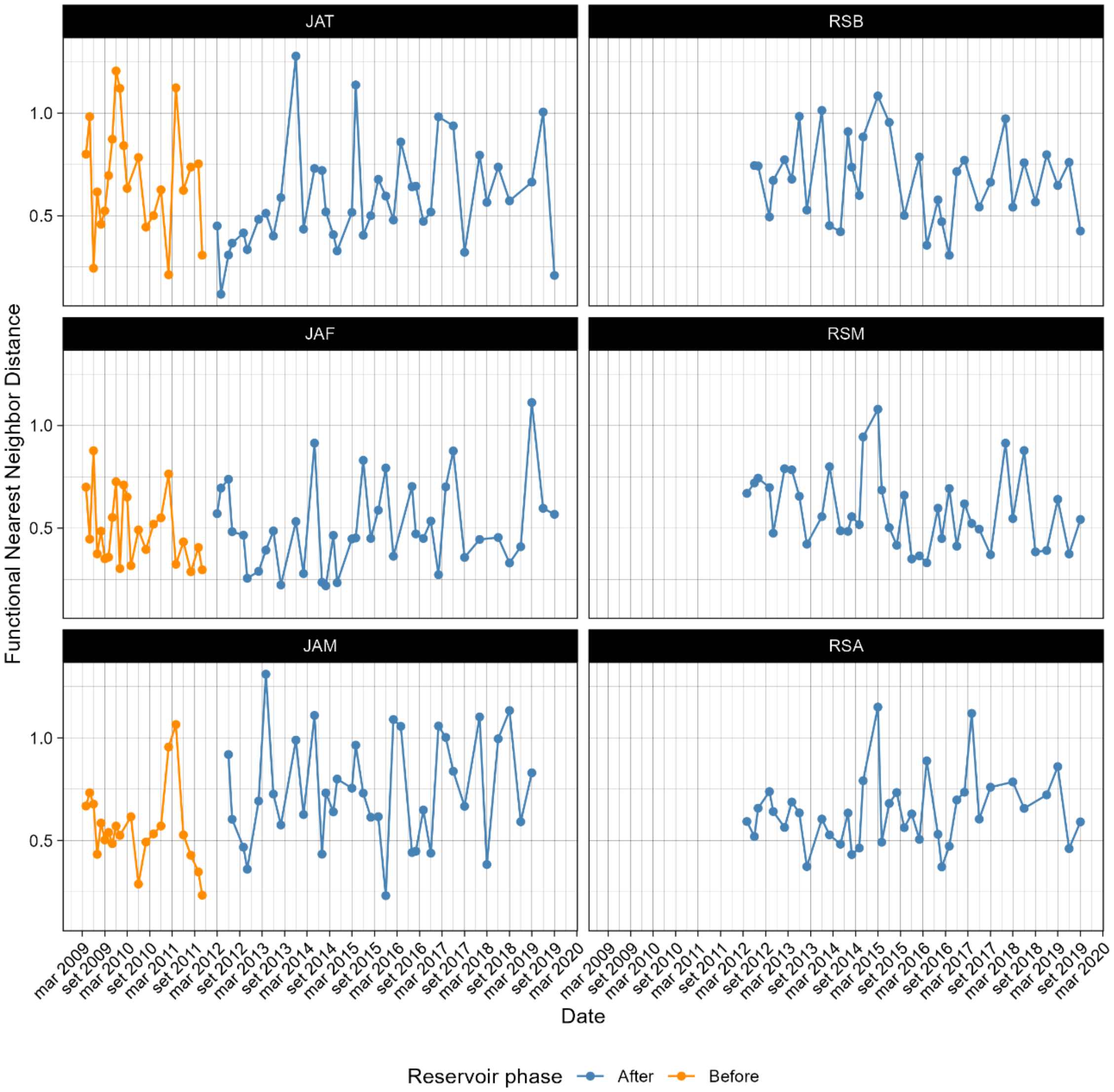

Figure 18 | Distribution of Functional Mean Nearest Neighbour Distance (FNND) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

Figure 19 | Distribution of Functional Originality (FOri) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

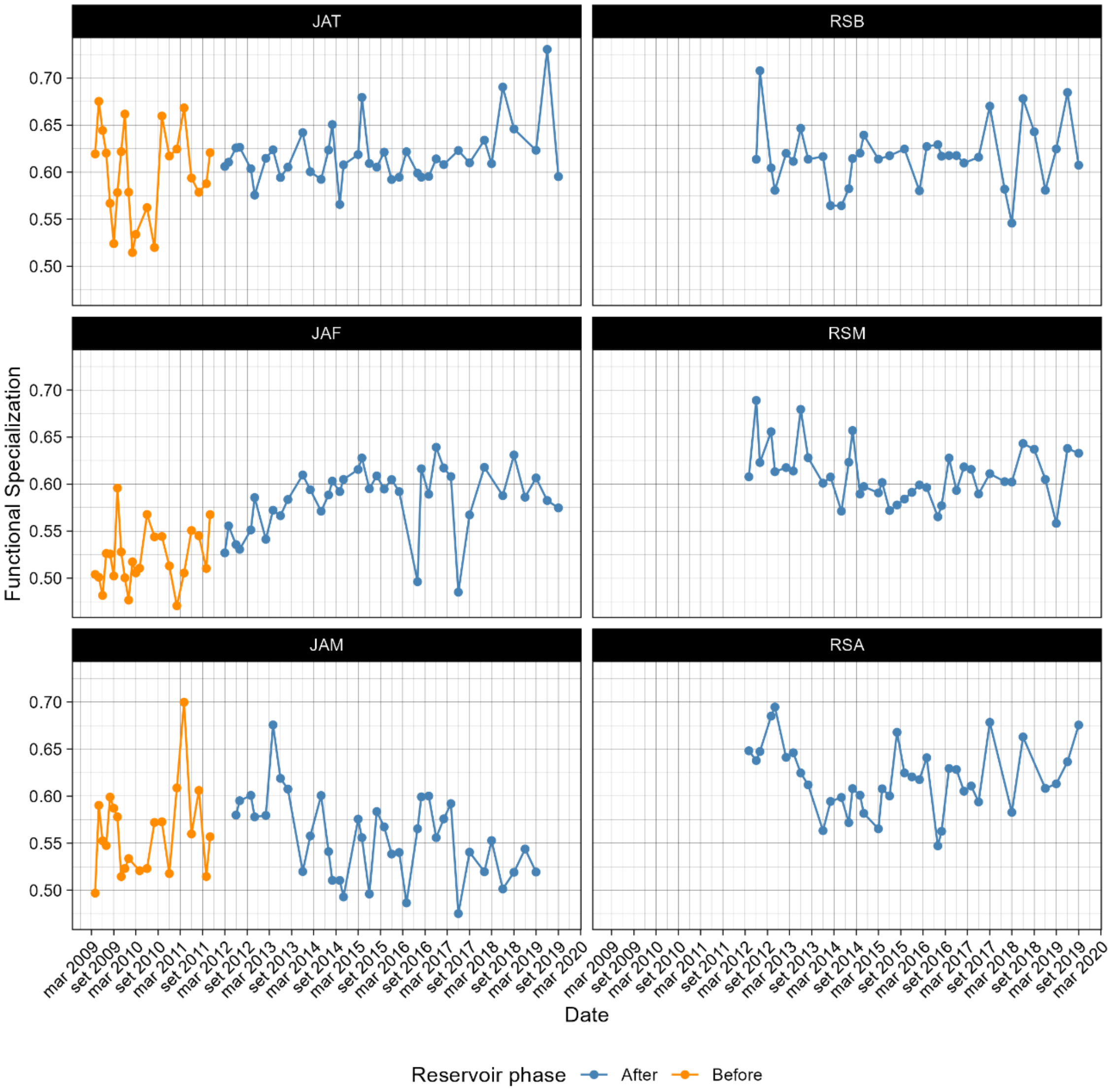

Figure 20 | Distribution of Functional Specialization (FSpe) recorded at each sampling site across the fish sampling events. Each point represents the value obtained during a specific sampling event (month/ year). Orange points represent the period before reservoir formation, and blue points represent the period after. Sampling sites included: the confluence of the Jaci-Paraná River with the Madeira River (JAF); an upstream site on the Jaci-Paraná River (JAM); the mouth of the Jatuarana Stream (JAT); and the upper (RSA), middle (RSM), and lower (RSB) sections of the Madeira River reservoir, which were added after reservoir impoundment. Fish sampling was carried out from April 2009 to September 2019.

Appendix E | Frequency of individuals in each functional trait category before and after impoundment. For each trait (trophic category, mouth position, water-column preference, body shape, caudal-fin type, reproductive season, migratory behaviour, and length class) the bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages. All length classes ≥52 cm (up to 451 cm) were combined into a single category.

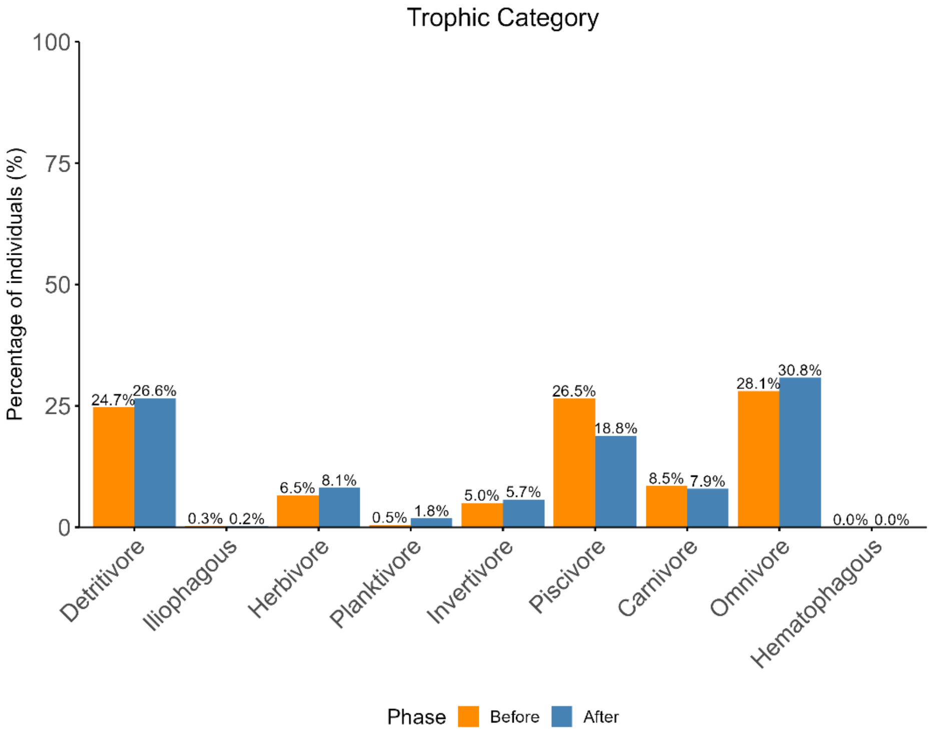

Figure 21 | Frequency of individuals by trophic category. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

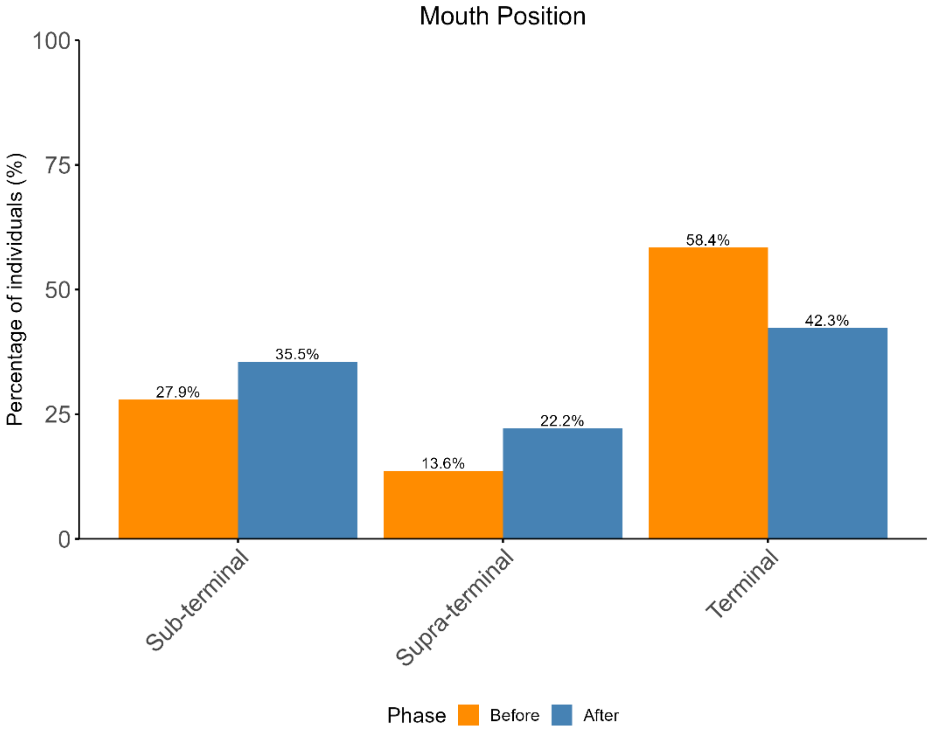

Figure 22 | Frequency of individuals by mouth position. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

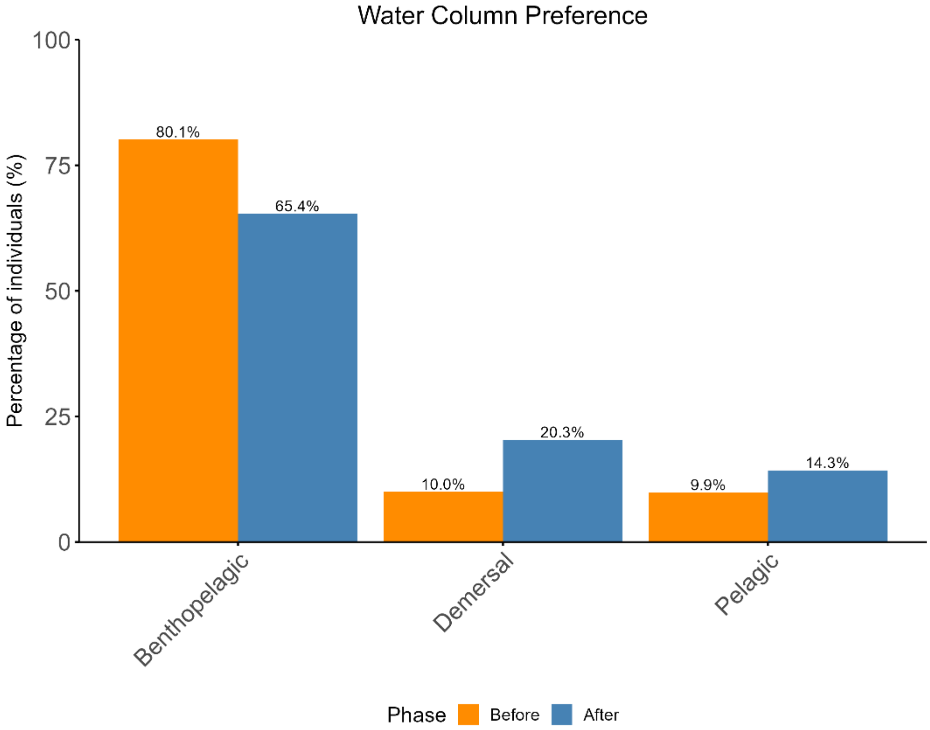

Figure 23 | Frequency of individuals water-column preference. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

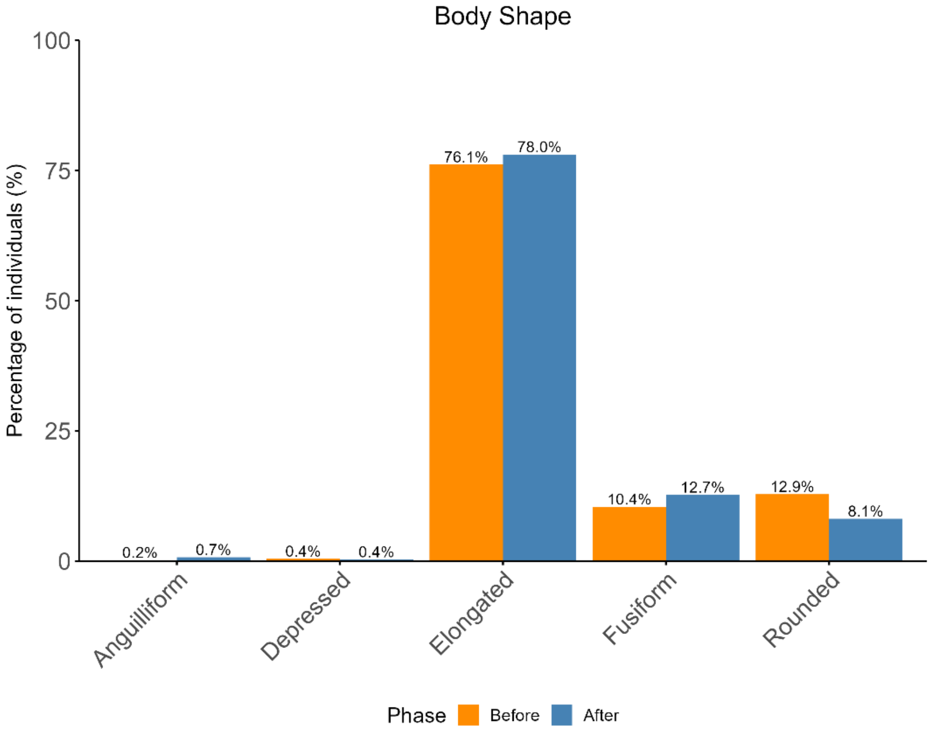

Figure 24 | Frequency of individuals by body shape. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

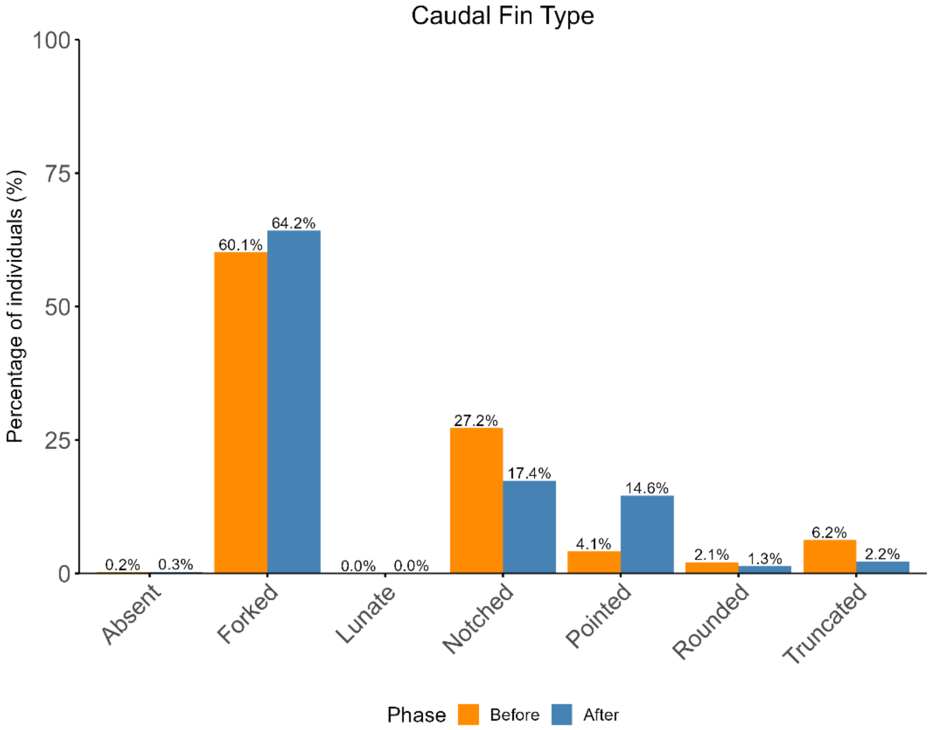

Figure 25 | Frequency of individuals by caudal-fin type. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

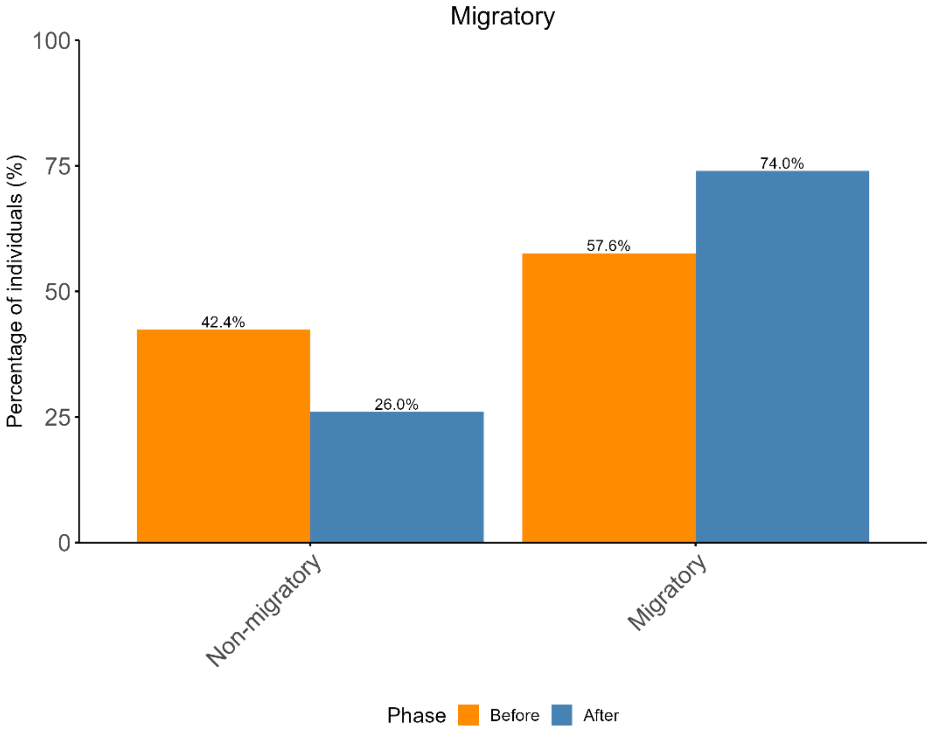

Figure 26 | Frequency of individuals by migratory behaviour. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

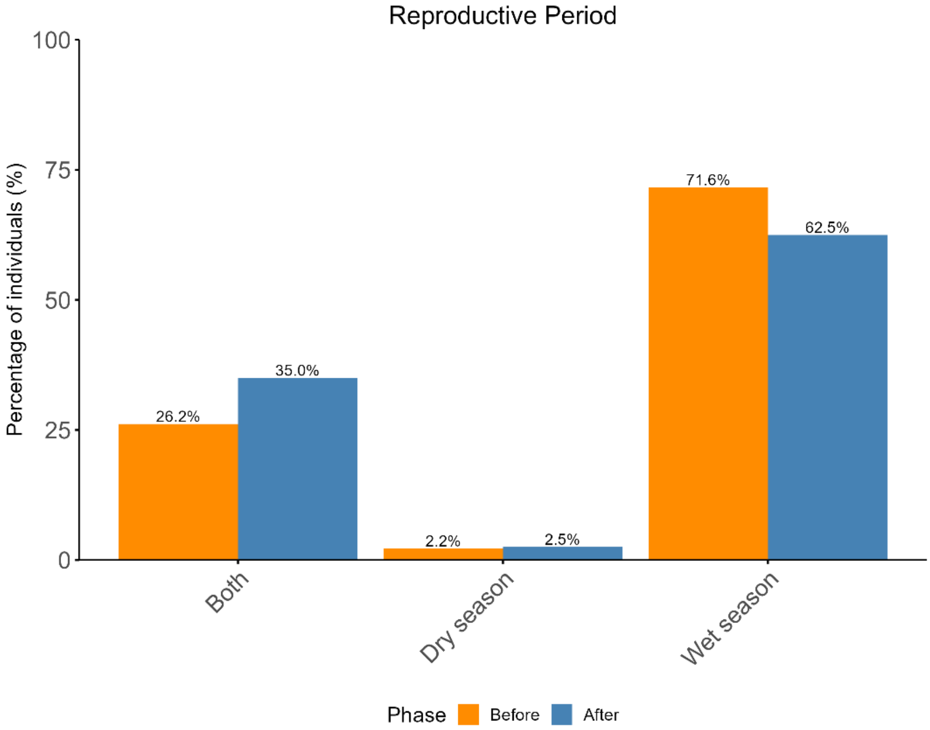

Figure 27 | Frequency of individuals by reproductive season. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages.

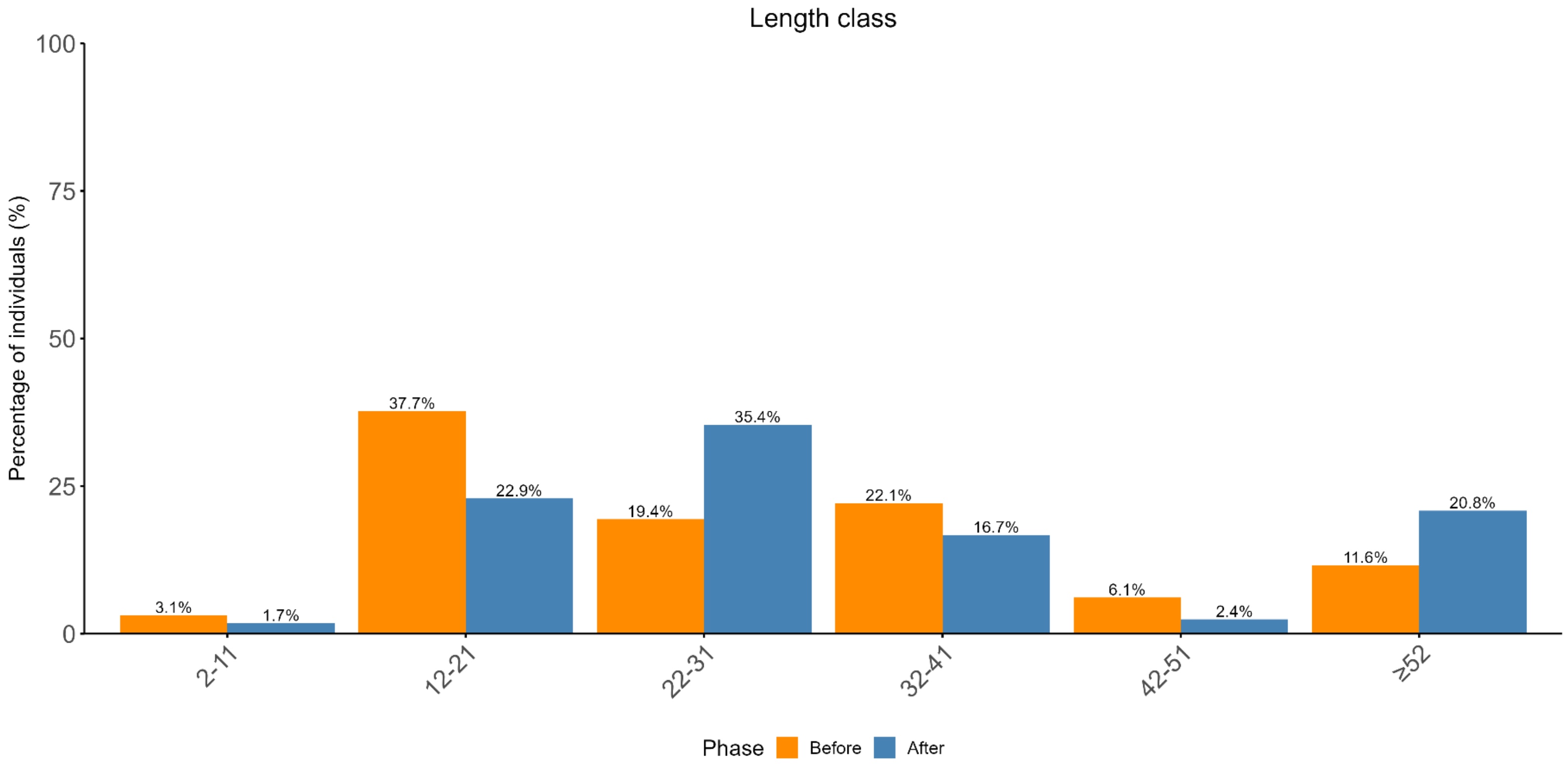

Figure 28 | Frequency of individuals by length class. The bars show the percentage (%) of individuals in each category for before (orange) and after (blue) phases. Percentages are calculated within phase (summing to 100%); values above the bars report the phase-specific percentages. All length classes ≥52 cm (up to 451 cm) were combined into a single category.

CAPÍTULO II

THE FISH CAPTURE DEVICE (FISHCD): A STANDARDISED MONITORING TOOL FOR ASSESSING FISH PASSAGE SUCCESS AT HYDROPOWER FACILITIES

Abstract

Assessing the performance of fishways remains a global challenge, largely due to the absence of standardised monitoring approaches. Effective fishway monitoring in large tropical rivers calls for standardised, species-level assessment with consistent sampling. The Fish Capture Device (FishCD), a current-driven rotating device adapted from traditional fishwheel designs, was deployed for continuous monitoring at the fishway exit of the Santo Antônio Hydropower Plant, Madeira River, Brazil. Over an eight-month sampling period (September/2024 to April/2025), the FishCD captured 149 individuals (18 Species, 7 Families and 2 Orders), evidencing passage across diverse ecological guilds of Amazonian fishes. All recorded species had been previously reported upstream and downstream of the dam. Characiformes contributed the greatest species richness (11 species), whereas Siluriformes contributed the greatest total abundance (106 individuals). Abundance peaked during the rising-water period, coinciding with the upstream movements typical of migratory fishes, and capture events were more frequent at night. Outflow discharge had a significant negative effect on capture rates, while diel period affected only the probability of zero-capture events. The FishCD demonstrated multiple operational advantages over conventional monitoring approaches, including continuous sampling without external power requirements, standardised capture protocols, and responsiveness under a variety of environmental and operational conditions, whilst providing complementary evidence on fishway effectiveness relative to established technologies. The FishCD proved effective as a reproducible tool for quantifying species-specific passage success in tropical fishways, providing empirical data for evidence-based fishway assessment and adaptive management in regulated Amazonian River systems.

Keywords: Amazonian fishes; Fishway; Hydropower Plant; Madeira River; Migratory fishes.

Introduction

Effective monitoring of fish passages to assess whether they have achieved the intended outcome of restoring viable population connectivity remains a persistent methodological challenge (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017; Roscoe & Hinch, 2010). Notwithstanding their long-standing use, fishway performance evaluation suffers either from a lack of standardized monitoring protocols, preventing meaningful comparisons across sites, designs, and species assemblages (Castro-Santos, 2009), or from the complete absence of evaluation, particularly in South American reservoirs where many fishways have never been systematically assessed (Pompeu et al., 2012). Furthermore, current monitoring approaches are constrained by a lack of standardisation in metrics and methods, alongside significant logistical and technological constraints, which collectively limit data comparability and meta-analytical potential (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017; Roscoe & Hinch, 2010).

This methodological fragmentation, driven by a global lack of standardised methods for monitoring fish passage, has hindered evidence-based improvements in fishway design (Bunt et al., 2011) and contributed to persistent uncertainties in the evaluation of passage effectiveness. These challenges are particularly pronounced for megadiverse fish assemblages in tropical rivers (Albert et al., 2020; Winemiller et al., 2016), where monitoring frameworks originally developed for salmonid species are of limited utility in tropical systems and require context-specific approaches (Silva et al., 2018). Fish passage assessment employs a range of technological approaches, each designed to address specific monitoring requirements while offering distinct operational capabilities and constraints. Fish trapping systems utilize funnel-net configurations or cage traps strategically positioned at fishway entrances and exits to enable direct species identification and quantitative abundance comparisons between entering and exiting assemblages; however, behavioural avoidance, size-selective bias and discontinuous sampling can under-represent some taxa (Jones & O'Connor, 2017; Mekong River Commission, 2023). Video monitoring deploys underwater cameras within passage pools, enabling continuous behavioural observation and automated counts without handling, but performance declines under the high turbidity common in tropical rivers and during low-light periods (Haas et al., 2024; Xu et al., 2024). Biotelemetry approaches, comprising passive integrated transponder (PIT), radio, acoustic and hybrid radio-acoustic tagging, involve surgically implanting or externally attaching transmitters to track individual movements via strategically positioned receiver arrays (antennas or hydrophones) (Cooke et al., 2012). These methods yield high-resolution spatial temporal data on fish behaviour and movement patterns; however, scaling individual-level observations to population-level inference typically requires substantial tagging effort and prolonged monitoring (Iaia et al., 2025; Nakai et al., 2024), and specialised infrastructure, which substantially increases operational and financial costs (Cooke et al., 2012). Advanced hydroacoustic systems, including split-beam sonar and infrared scanners, automate detection and enumeration at fixed stations, yet their performance is sensitive to flow turbulence, equipment-generated noise and gas-bubble formation, particularly during high-flow periods (Cui et al., 2024). While collectively informative for fish passage assessment, each technology remains limited under hydrodynamically complex river conditions, reinforcing the need for integrated, standardised approaches that combine methods and mitigate their limitations. A comparative synthesis of the main fish passage monitoring approaches, highlighting their applications, advantages and limitations, is provided in Table 7.

Table 7. Overview of fish passage monitoring approaches used worldwide in fishways, comparing their main applications, advantages and limitations.

Monitoring approach	Main applications	Advantages	Limitations
Fish trapping systems (e.g. cast nets, seine nets)	Species identification; quantitative comparisons between entering and exiting assemblages	Direct capture allows taxonomic resolution at species level; enables abundance-based comparisons; relatively low technological requirements	Behavioural avoidance; size- and species-selective bias; discontinuous sampling; labour-intensive handling; under-representation of some taxa
Video monitoring (underwater cameras)	Behavioural observation; automated counting at fixed locations	Non-invasive; continuous monitoring; enables behavioural analyses without handling	Performance strongly reduced under high turbidity and low-light conditions; limited taxonomic resolution; high data-processing demand
Biotelemetry (PIT, radio, acoustic, hybrid tags)	Individual movement tracking; passage routes; estimation of passage proportions for tagged fish	High-resolution spatiotemporal data; allows inference on movement pathways, timing and passage efficiency for marked individuals	High operational and financial costs; limited sample sizes; tag-related handling and mortality risks; scaling to population-level inference requires extensive tagging effort and long-term monitoring; infrastructure-dependent
Hydroacoustic systems (split-beam sonar, infrared scanners)	Automated detection and enumeration at fixed stations	Non-invasive; continuous operation; suitable for high throughput counting	Sensitive to turbulence, entrained air, suspended sediments and acoustic noise; reduced performance during high-flow conditions; limited taxonomic discrimination

In response to these methodological challenges and the need for standardized biological assessment of fishway effectiveness, this study evaluates the Fish Capture Device (FishCD), experimentally implemented at the Santo Antônio Hydropower Plant

(SA-HPP) fishway exit on the Madeira River, Brazil, under the Research, Development, and Innovation Program of National Agency of Electric Energy (ANEEL; PDI nº PD-06683-0320/2020). The Santo Antônio Hydropower Plant is part of the Madeira River Hydropower Complex, located in Porto Velho, state of Rondônia. The Madeira River Hydropower Complex acts as a physical constraint, potentially altering longitudinal connectivity among Madeira River reaches and with the Amazon River (Anderson, et al., 2018).

Assessing fishway effectiveness on the Madeira River presents significant challenges beyond conventional monitoring due to the exceptional biodiversity and unique ecological features of this system. The Madeira River, the second largest tributary of the Amazon, is a whitewater river characterised by turbidity and high suspended-sediment concentrations (Bogotá-Gregory et al., 2020; Graça et al., 2025; Junk et al., 2011). Historically, the river featured a 300 km stretch of rapids and waterfalls that functioned as selective biogeographical filters shaping fish fauna distribution patterns (Torrente-Vilara et al., 2011). These rapids, particularly the steep Teotônio and Jirau sections, acted as selective filters (Torrente-Vilara et al., 2011) rather than absolute barriers: during the wet season they permitted passage by migratory species while restricting others, thereby maintaining biogeographical complexity and supporting basin-wide connectivity for populations able to pass them (Anderson, et al., 2018; Arantes et al., 2023; Torrente-Vilara et al., 2011).

Among these migrants are the goliath catfish (*Brachyplatystoma rousseauxii*), which undertake the longest documented freshwater migrations in the world (Barthem et al., 2017; Duponchelle et al., 2016) and is part of Amazonian fisheries (Doria et al., 2012; Isaac & Barthem, 1995). This historical biogeographical heterogeneity presents a major challenge for managing river connectivity at the Madeira River Hydropower Complex: must facilitate passage for species adapted to the former rapids while preventing artificial homogenisation caused by the upstream spread of lowland species whose natural distributions were once limited by these natural barriers.

The FishCD is a technological adaptation of traditional fishwheel designs, which are current-driven rotating devices historically employed for monitoring upstream-migrating salmonids in Alaska, Canada, and Japan (Begich & Wilburn, 2022; Reid & Ban, 2025; Saito et al., 2021), specifically engineered for continuous operation at fishway exits in tropical river systems. This study presents an experimental deployment of the FishCD prototype at the SA-HPP to evaluate the FishCD's performance and operational capabilities, with the aim of advancing evidence-based fish passage assessment and establishing a reproducible monitoring tool for megadiverse Neotropical rivers. Over eight months operation (September 2024-April 2025), trials tested whether the device delivered standardised captures across diverse morphologies and ecological guilds under variable hydrological regimes and diel periods, and whether it produced quantitative data suitable for statistical analysis of exit abundance and passage frequency of fish exiting the fishway.

Material and Methods

Study Site

The SA-HPP is located on the Madeira River in Porto Velho, Rondônia, Brazil. The Madeira River extends 3,240 km in total length, of which 1,425 km flows through the Brazilian states of Rondônia and Amazonas. The SA-HPP is one of two run-of-the-river hydropower facilities constructed at Santo Antônio Falls, operating with an installed capacity of 3,568 MW across 50 bulb-type turbines distributed in four powerhouses (Eletrobras, 2025).

The SA-HPP fishway is a semi-natural channel design (Clay, 1995) built on the site of the former Presidio Island and excavated into the island's bedrock. It comprises two entrance arms (approximately 200 m each) that converge into a single channel of about 1 km, 10 m wide and at least 2.5 m deep, with a 2.5% gradient and a discharge between 4 and 10 m³/s (Hahn et al., 2020). The channel terminates at a single fish exit opening to the reservoir between Powerhouses 1 and 4 (Figure 29) and its operation is regulated by the reservoir water elevation. Each entrance is 6 m wide and 14.5 m high, with a water depth of about 10 m (Hahn et al., 2020). Deflectors, previously used to control water force and velocity, were replaced by large reinforcement boulder deflectors in 2016.

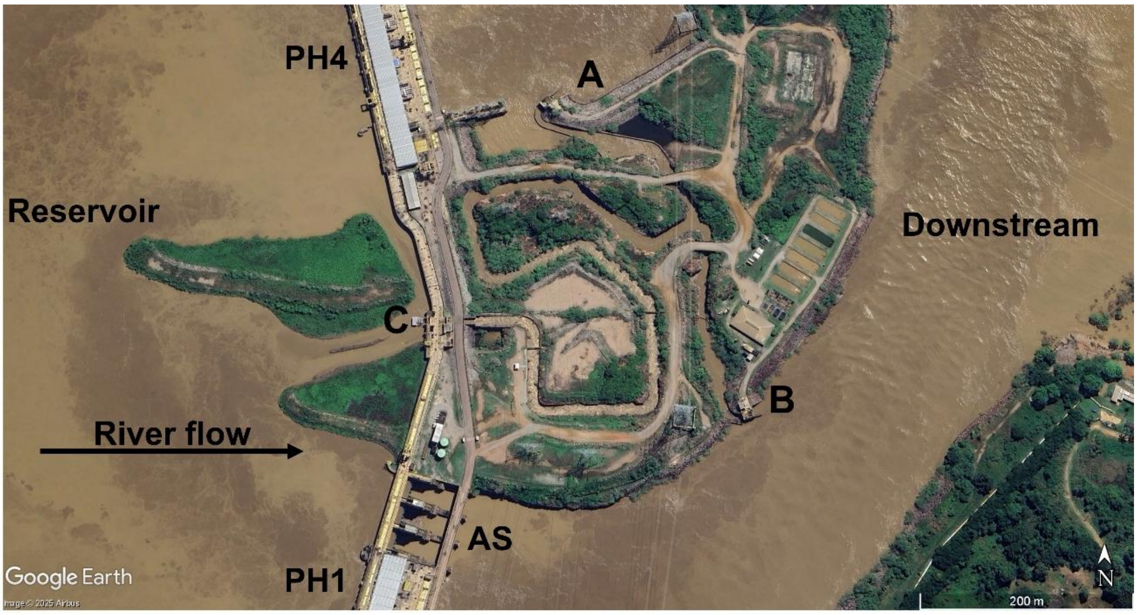

Figure 29. The SA-HPP fishway is a semi-natural channel design featuring two fish entrances: on the left bank (A) and on the right bank (B). These entrances converge before the single fish exit (labeled as C), which leads toward the reservoir. The locations of Powerhouses 1 (PH1) and 4 (PH4), as well as the Auxiliary Spillway (AS), are also shown. The black arrow indicates the natural flow of the Madeira River, moving from the reservoir to downstream. Image source: Google Earth © 2025 Airbus Retrieved July 14, from https://earth.google.com.

Fish Capture Device Design and Operation

The FishCD prototype was designed to address methodological limitations in fishway monitoring by enabling standardised, non-invasive capture at fishway exits. The device is adapted from the design of a fishwheel, a current-driven rotating device historically used to monitor and harvest salmon migrating upstream in rivers of the Northern Hemisphere rivers (Begich & Wilburn, 2022; Reid & Ban, 2025; Saito et al., 2021; Zuray, 2005). Whereas traditional fishwheels have been used almost exclusively for salmonids in free-flowing river stretches, the FishCD was specifically designed to operate at the fishway exit and to address the ecological context of Neotropical rivers, where high fish diversity encompasses both migratory and non-migratory species with distinct ecological strategies. The device was developed based on the size and swimming behaviour of fish species expected to use the SA-HPP fishway, as identified in previous studies (Torrente-Vilara et al., 2011; Torrente-Vilara, 2009; Cella-Ribeiro et al., 2017; Queiroz et al., 2013), thereby accommodating their ecological traits and reflecting the expectation of upstream-migrating assemblages.

The FishCD consists of a pontoon-mounted structure measuring 12 m (length) × 8 m (width), supporting a rotating wheel with dimensions of 8 m (height) × 5 m (width). Constructed from aluminium for its durability, corrosion resistance, and ease of installation in strong river currents. The structure contains three capture baskets with 6 cm mesh (opposite-knot distance), designed to retain fish as the wheel turns. Each basket guides captured fishes through three slides into lateral holding tanks (live boxes) with continuous water circulation), where they remain unharmed for identification, tagging, and release with minimal stress (Figure 30). A dedicated laboratory area integrated into the FishCD enables immediate fish identification, morphometric measurements, and data recording under standardised conditions.

The FishCD was positioned at the single exit point of the SA-HPP fishway, where the fishway channel connects to the reservoir. This placement ensures that fish completing passage through the fishway are captured, providing direct quantification of successful passage events. After positioning the FishCD at the fishway exit using a tugboat and small aluminium motorboats, the unit was secured with four manual lever-operated wire-rope hoists (Tirfor®-type): two lateral hoists attached to permanent anchorage points on the dam structure, and two stern hoists were led to deadweight moorings upstream. This arrangement stabilises the device in plan while allowing vertical adjustment as reservoir water elevation varies.

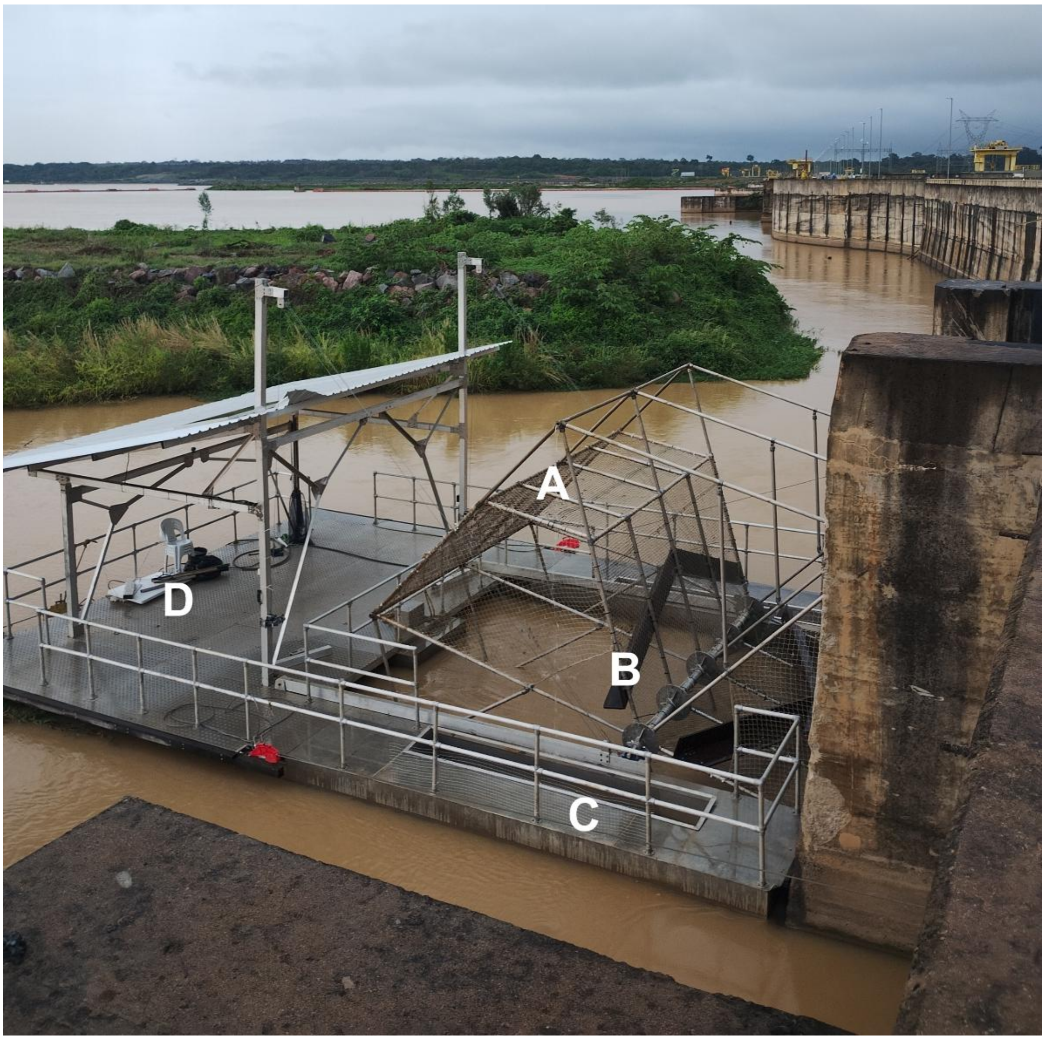

Figure 30. Operational Fish Capture Device (FishCD) prototype installed at the fishway exit, measuring 12 m (length) × 8 m (width), with an 8 m (height) × 5 m (width) rotating wheel. The structure consists of three baskets (A) that capture fish as the wheel turns, three slides (B) that guide the fish into one of the lateral tanks (C), which temporarily hold them alive, and a laboratory area (D) for identification, measurement, tagging, and data recording. Photo: Marina Granai.

Sampling Protocol, Data Acquisition and Fish Processing

Fish sampling was conducted from September 2024 to April 2025 under permits issued by the Brazilian Institute of Environment and Renewable Natural Resources (IBAMA Permit ABIO nº 1618/2024 - RET nº 1/2024). Daily data on reservoir water elevation (m), along with natural flow (m^3/s) of the Madeira River and outflow (m^3/s) of SA-HPP were obtained from the Reservoir Performance Monitoring System database – SAR (Agência Nacional de Águas, 2025). These variables were used to characterise environmental conditions over the study period and to examine their influence on fish-passage patterns.

The sampling protocol comprised three daytime inspections of the FishCD live boxes at 08:00, 12:00 and 16:00. In accordance with safety regulations applied by SA-HPP, night-time on-board activities were not permitted. Consequently, no nocturnal inspection was conducted. Nevertheless, the FishCD operated autonomously overnight, with water circulating continuously in the live boxes, and any overnight captures were retrieved at the first inspection the following morning.

During each inspection, captured fish were transferred from the live boxes to the on-board laboratory using fine-mesh dip nets. Individuals were placed immediately into separate containers with oxygenated river water to maintain condition during processing. The protocol minimised handling time and stress: identification and measurements were completed rapidly. Processing time did not exceed 1 minute per fish from netting to release. All captured fish were identified to species according to taxonomic literature (Lundberg & Akama, 2005; Ohara et al., 2017; Queiroz et al., 2013; Santos et al., 2006). Standard and total lengths (in centimeters) and body weights (in grams) were recorded, and individuals were released immediately after processing from the stern platform, upstream of the device.

Fish condition was assessed prior to release. Each individual was inspected for telemetry devices (radio or radio–acoustic transmitters) and conventional marks (dart tags), and any signs of stress or injury were recorded. These checks were standardised because monitoring programmes on the Madeira River routinely use radio/radio–acoustic telemetry and dart tagging. No mortality occurred during handling, supporting the effectiveness of the capture and handling procedures. One representative per species was retained as a voucher specimen, sedated with clove oil (eugenol) and euthanised before fixation in 40% formalin, then transferred to 70% ethanol for long-term preservation.

Data analysis

Community Structure Analysis

Sampling sufficiency was evaluated using an individual-based rarefaction curve, with total expected species richness estimated via Chao 1 estimator (Chao, 1984). Curves were generated using the *iNEXT* package (Hsieh et al., 2024) in R (R Core Team, 2024). Extrapolation continued until the rarefaction curve approached the Chao1 asymptote, providing an estimate of the effort required to approach the putative species pool. Species composition was compared with previous studies from the Madeira River (Cella-Ribeiro et al., 2017; Queiroz et al., 2013; Torrente-Vilara, 2009; Torrente-Vilara et al., 2011), to assess the FishCD performance in capturing a representative range of the regional ichthyofauna.

Four functional traits related to feeding, locomotion, and reproduction were selected based on previous studies (Erős et al., 2012; Luck et al., 2012; Villéger et al., 2017; Terra et al., 2016; Zhang et al., 2023). The trait dataset comprised four categorical traits (feeding behaviour, habitat preference, reproductive period, and migration behaviour). The functional traits data were compiled from available literature (Alcalá & Betancourt, 2022; Carolsfeld et al., 2003; Cella-Ribeiro et al., 2016; Ohara et al., 2017; Queiroz et al., 2013; Reis et al., 2003; Röpke et al., 2024; Santos et al., 2006) and the FishBase database (Froese & Pauly, 2024). The functional traits assigned to each species are provided in Supporting Information (Table S1).

Capture Rate Modelling

Fish capture patterns were analysed using a zero-inflated negative binomial (ZINB) regression model to account for overdispersion and excess zeros characteristic of ecological count data (Zuur et al., 2009). The ZINB model addresses two distinct processes: (1) the probability of structural zeros (no captures due to absence of fish), and (2) the magnitude of captures when fish are present. This approach is particularly appropriate for fishway monitoring data, where zero captures may result from genuine absence of fish exiting the fishway. The ZINB model comprises two components: a conditional negative binomial count model (Equation 1) and a zero-inflation model (Equation 2).

$$log(\mu_i) = \beta_0 + \beta_1 Period_i + \beta_2 OutflowDischarge_i + log(SamplingEffort_i) \quad (1)$$

$$(\pi_i) = \Upsilon_0 + \Upsilon_1 Period_i + \Upsilon_2 OutflowDischarge_i \quad (2)$$

In these components μ_i represents the expected count given that the observation arises from the count process, and π_i is the probability that the observation belongs to the structural zero state.

The response variable was the number of individuals captured per inspection. The conditional component of the ZINB model followed a negative binomial distribution. Diel period (day and night) and outflow discharge (SA-HPP discharge, 1,000 m^3/s; scaled and centred) was included as fixed effect and covariate, respectively. Outflow discharge was scaled and centred to improve model convergence. Sampling effort was incorporated as an offset term by including the natural logarithm of the operational time fraction (hours of operation/24 hours) for each inspection period. This standardisation enables interpretation of results as catch per unit effort (CPUE), facilitating comparison across inspection periods with varying operational durations. Model predictions were compared against observed values to assess predictive accuracy. Model results were interpreted alongside statistical significance to assess biological or operational relevance of observed patterns.

All statistical analyses were conducted using R software (R Core Team, 2024). Statistical significance was evaluated at a threshold of $p \leq 0.05$.

Results

Environmental conditions and FishCD operational performance

From September 2024 and April 2025, both reservoir water elevation and outflow displayed significant seasonal variability, as depicted in Figure 31. In the late dry season in September and October, outflow from the Madeira River remained relatively low, ranging from approximately 2,300 to 2,400 m^3/s. From November onward, outflow increased steadily, peaking at 39,500 m^3/s in April. Reservoir water elevation exhibited a comparable trend, remaining stable between September and November, with values rising gradually from 70.5 m in September to 71.3 m in January, before declining in the following months. This decline, despite sustained high river flow, stems from short-term operational adjustments. Although the run-of-river reservoir lacks long-term storage capacity, its finite water residence time permits limited operational coordination by the Brazilian National Electric System Operator (ONS) and the Brazilian National Water Agency (ANA) during peak hydrological periods, particularly in relation to upstream water level management near adjacent infrastructure.

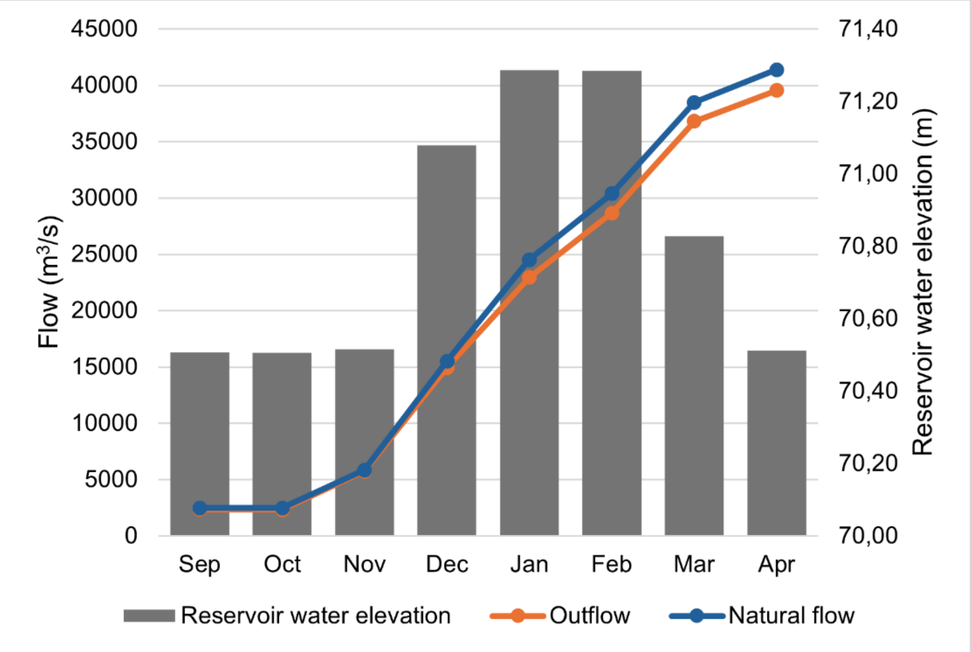

Figure 31. Monthly variation in reservoir water elevation (gray bars, in meters), outflow (orange line, in m³/s), and natural flow of the Madeira River (blue line, in m³/s) from September 2024 to April 2025. Flow values are shown on the primary y-axis, while reservoir water elevation is displayed on the secondary y-axis.

FishCD operational time varied considerably across months, ranging from 3.5 hours in February to 113 hours in January, with sampling conducted over a total of 26 days. Sampling effort comprised four days in September-October, three days each in November, December and April, five days in January, and two days in March. Total operational time was distributed between day (225 hours) and night (266.5 hours) periods, with longer nocturnal operations particularly evident from September to December 2024. No captures were recorded in February as sampling was paused for necessary unscheduled maintenance on the FishCD.

Fish assemblage composition and temporal patterns

Over the test period, captures at the fishway exit resulted in 149 individuals from 18 species (7 Families, 2 Orders) (Figure 32). Characiformes dominated species richness (11 species), while Siluriformes dominated total abundance (106 individuals). All had been previously reported both upstream and downstream of the Santo Antônio and Teotônio Rapids. These 18 species represent only a small fraction (4.4%) of the fish assemblage previously recorded in the Madeira River rapids upstream of the dam (Cella-Ribeiro et al., 2017; Queiroz et al., 2013; Torrente-Vilara, 2009; Torrente-Vilara et al., 2011). Species richness fluctuated monthly, with the highest values recorded in October 2024 (8 species), followed by September 2024 (7) and March 2025 (5). The lowest richness values occurred in December 2024 and January 2025 (3 species each), while November and April also showed reduced richness (4 species).

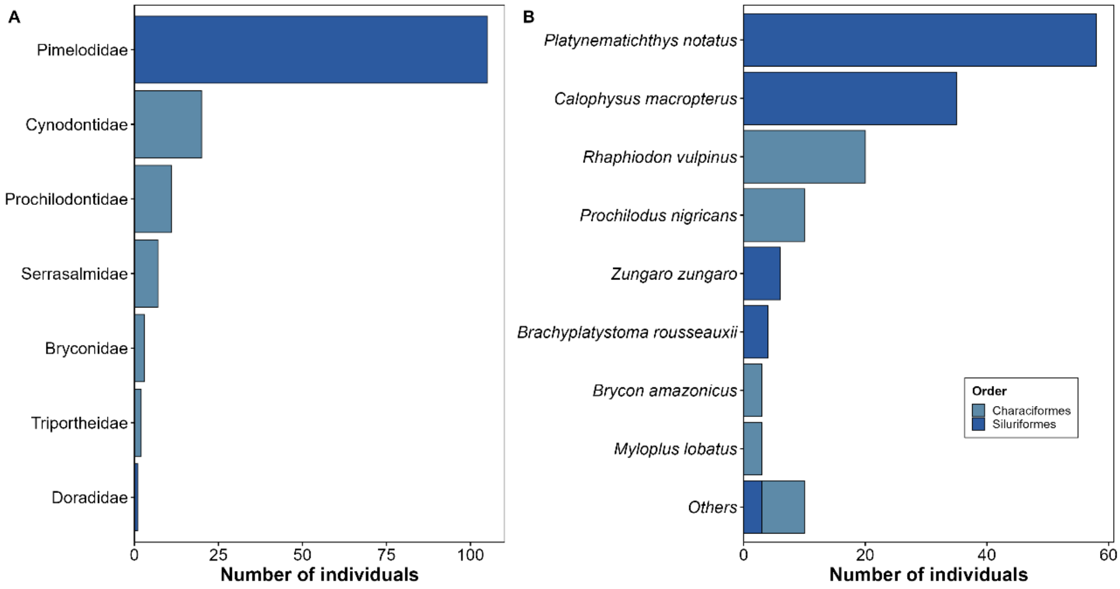

Figure 32. (A) Abundance of fish Families and (B) species captured by the experimental Fish Capture Device (FishCD) during the testing phase conducted between September 2024 and April 2025. 'Others' grouped species represented by a unique individual each (*Brachyplatystoma platynema, Colossoma macropomum, Mylossoma aureum, Pterodoras granulosus, Semaprochilodus insignis, Serrasalmus* sp.1, *Serrasalmus* sp.2, *Sorubim elongatus, Triportheus albus, Triportheus* sp.1).

The fish assemblage exhibited a wide range of body sizes, with total lengths varying from 8 to 120 cm and body weights ranging from 5 g to 32.150 g. This size range encompasses small-bodied species (e.g. *Triportheus* spp.) and small juveniles of larger species (e.g. *Pterodoras granulosus* (Valenciennes 1821)) to large migratory catfish, including *Brachyplatystoma rousseauxii* (Castelnau, 1855) and *Zungaro zungaro* (Humboldt 1821), demonstrating the FishCD performance to capture fish across a broad morphological spectrum.

Species captured at the fishway exit were demersal (S = 7, n = 106, 71.14%), followed by benthopelagic species (S = 10, n = 23, 15.44%), while pelagic species represented only a single species with 20 individuals (S = 1, n = 20, 13.42%). Trophic strategies were dominated by piscivores (S = 6, n = 124, 83.22%), with detritivores (S = 2, n = 11, 7.38%), omnivores (S = 3, n = 5, 3.36%), herbivores (S = 2, n = 4, 2.68%), carnivores (S = 3, n = 3, 2.01%), and invertivores (S = 2, n = 2, 1.34%) collectively contributing the remaining assemblage. Most captured species exhibited migratory behaviour (S = 16; n = 147; 98.66%) and reproduced during the wet season (S = 16; n = 126; 84.56%), confirming the fishway's use by the target migratory guild (Figure 33). Notable captures included long-distance migrants *B. rousseauxii*, known for the longest freshwater migrations in the Neotropics, and *B. platynema*, both representing critical species for assessing fishway effectiveness in supporting large-bodied catfish migrations.

Species list with abundance (N) and traits is provided in Supporting Information Table S1.

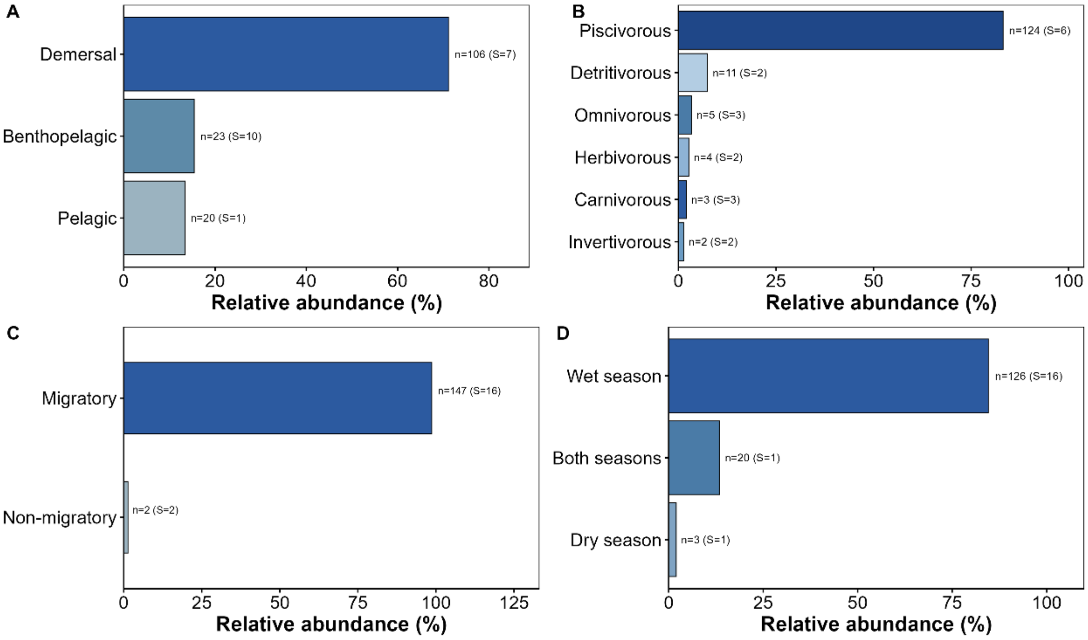

Figure 33. Functional traits composition of fish captured by the Fish Capture Device (FishCD) at Santo Antônio fishway (Madeira River, Brazilian Amazon) between September 2024 and April 2025. (A) Water column position; (B) Feeding guilds; (C) Migratory behaviour; (D) Reproductive season.

Estimated richness, counts and capturability

An expected species richness of 63 species was estimated by the Chao 1 estimator. This estimate is approximately 3.5 times higher than the observed richness, although the wide confidence interval (95% CI: 18-110 species) indicates considerable uncertainty. The rarefaction curve shows no signal of stabilization, indicating that additional sampling would likely reveal more species. The extrapolation curve extends to 3,000 individuals (20 times the current sample size) to approximate the Chao 1 estimate (Figure 34), suggesting that substantially more sampling effort would be required to capture the full diversity of fishway users.

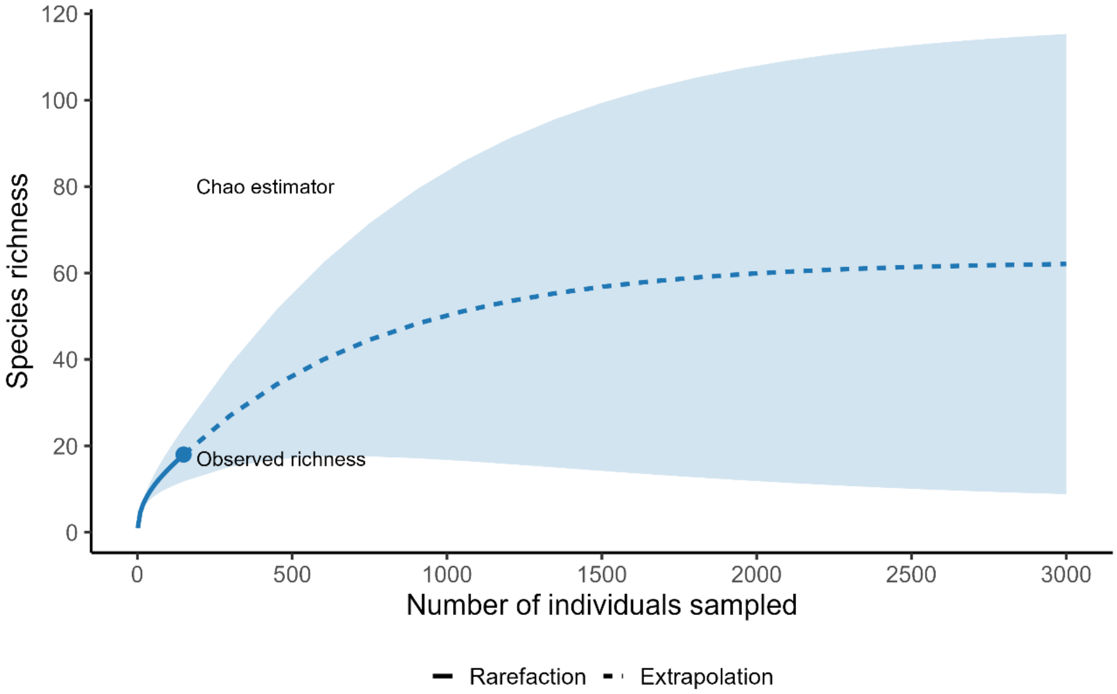

Figure 34. Individual-based rarefaction curve for the FishCD prototype. The rarefaction curve, based on 149 individuals and 18 observed species, was extrapolated to 3,000 individuals to approximate the Chao estimator of 63 species. The shaded area represents 95% confidence intervals.

During the eight-month of FishCD operation, there were 66 inspection events with 48 (72.7%) rendering no fish capture, 41 at daily and 7 at night periods. From the non-zero events, the number of individuals captured ranged from 1 to 36 per event, with temporal variation evident throughout the study period. Higher capture frequencies were recorded from September to November, particularly at night, while reduced activity was observed from January to April.

The baseline capture rate predicted by the abundance model during daytime under average outflow conditions was approximately 13 fish per 24 hours (e^{β_0} = 13.17). Diel period had a negligible effect on capture magnitude (β_1 = -0.084, p = 0.850), suggesting similar catch per unit effort (CPUE) between day and night periods when fish were present (Table 8). Higher absolute counts observed at night were largely attributable to longer operation times of the FishCD during nocturnal periods (Figure 35).

Table 8. Parameter estimates for the zero-inflated negative binomial (ZINB) model - conditional (CM) and zero-inflated (ZI) components - of fish captures at the fishway. SE = standard error. Results were considered significant at p < 0.05.

Parameter	CM - Estimate (SE)	CM - p-value	ZI - Estimate (SE)	ZI - p-value
Intercept	2.578 (0.366)	<0.001	1.222 (0.475)	0.010
Night period	-0.084 (0.443)	0.850	-1.692 (0.687)	0.014
Outflow discharge	-0.058 (0.015)	<0.001	-0.012 (0.025)	0.633

Outflow discharge was the only significant predictor of capture success in the count model. Each 1,000 m³/s increase in discharge was associated with a 5.8% decline in CPUE ($1 - e^{\beta_2} = 0.058$, $p < 0.001$, SE: 0.015; Table 8). This negative relationship indicates that elevated discharge reduced fish exit at fishway, accounting for the temporal decline in captures observed between September 2024 and April 2025.

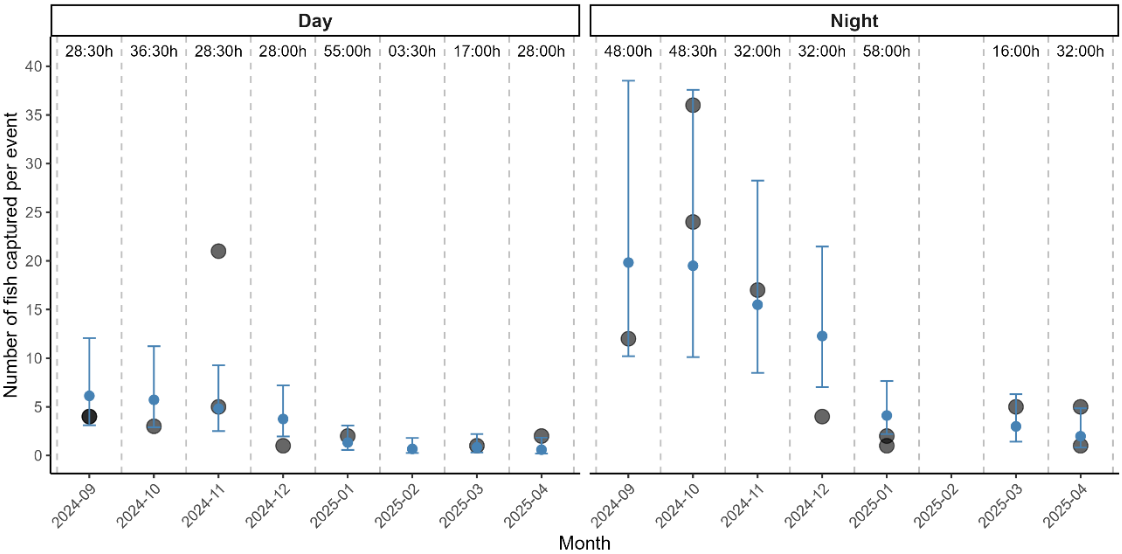

Figure 35. Number of individuals captured during FishCD inspections, separated by time of day from September 2024 to April 2025. Each gray dot represents a single inspection event. The total operational time (in hours) for both day and night periods are shown at the top. Blue dots and lines represent the monthly counts and the 95% predictive intervals, conditioned on monthly operational time and outflow discharge.

While diel period did not affect average fish abundance, it significantly influenced the probability of zero-capture events (Table 8; Figure 36). The likelihood of recording no fish was consistently higher during daytime, with a baseline probability of 77.2% ($\Upsilon_0 = 1.222$, $p = 0.010$), compared to 38.5% at night ($\Upsilon_1 = -1.692$, $p = 0.014$). Outflow discharge had no detectable effect on the probability of zero captures (Υ_2 = --0.012, $p = 0.633$). This pattern suggests that although average fish abundances at fishway exit passage rates remain constant across diel cycles, the probability exit events increase at night, suggesting more frequent dispersal events during this period.

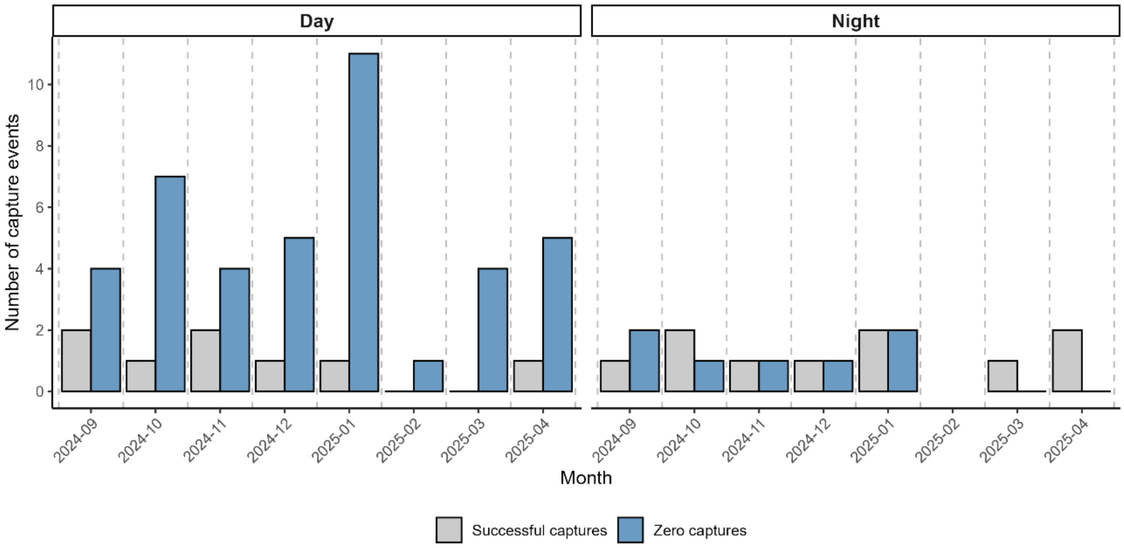

Figure 36. Monthly frequency of capture events recorded using the FishCD between September 2024 and April 2025 (formatted as year-month), separated by diel period (Day/Night). Bars represent the number of successful capture events (gray) and zero-capture events (blue).

Device performance and validation

The FishCD successfully captured morphologically and ecologically diverse species, including both small-bodied (*Triportheus* spp.) and large migratory catfish (*Brachyplatystoma* spp.), covering the full-size range expected in Amazonian fishways. No bite marks or signs of trauma were observed in captured individuals, indicating minimal stress and confirming the device's non-harmful capture mechanism. Also, no telemetry tags (radio or radio-acoustic) or dart tags were observed on any captured individual, nor were tag-related scars noted.

The device captured commercially and culturally important species, as well as taxa prioritised in conservation of populations and fisheries, including *Colossoma macropomum*, *Brycon amazonicus*, and two *Brachyplatystoma* species, representing apex predators in Amazonian River systems. Captures also included benthopelagic species (*B. amazonicus*, *Mylossoma aureum*, *Semaprochilodus insignis*) and pelagic species (*Rhaphiodon vulpinus*) demonstrates that the fishway is use by multiple migratory guilds and confirms the FishCD's ability to detect passage across different behavioural and morphological groups.

Discussion

Across eight months of operation, the FishCD performed consistently under variable environmental conditions, capturing 149 individuals from 18 Species, seven Families and two Orders. Although this eight-month period limits broader ecological inference about annual assemblage dynamics. However, the dataset was sufficient to demonstrate device functionality and to detect biologically meaningful temporal patterns in passage. Under average flows, CPUE was approximately 13 fish per day. For every increase of 1,000 m³/s in discharge, the CPUE declined by approximately 6%. Passage events were predominantly nocturnal; correspondingly, daytime inspections were about twice as likely to find non-empty traps when the catch accumulated overnight, compared to catches from daylight periods alone.

The device recorded species ranging from small-bodied characiforms (*Triportheus* spp.) to large migratory catfishes (*Brachyplatystoma* spp.), and reflected assemblages previously reported both upstream and downstream of the Madeira rapids (Cella-Ribeiro et al., 2017; Queiroz et al., 2013; Torrente-Vilara et al., 2011; Torrente-Vilara, 2009). The device is therefore suitable for monitoring the full-size range of fish expected to use the fishway. Compliance with welfare considerations is supported by the lack of visible trauma, with no bite marks or external injuries recorded.

Effectiveness extended across multiple migratory guilds and habitat preferences. It successfully captured demersal (*Brachyplatystoma platynema, Zungaro zungaro*), benthopelagic (*Brycon amazonicus*, *Mylossoma aureum* and *Semaprochilodus insignis*), and pelagic (*Rhaphiodon vulpinus*) species, including long-distance migrants like *Brachyplatystoma rousseauxii*. This taxonomic and ecological breadth suggests the device is not limited by specific swimming modes or body forms. Its design therefore appears suitable for monitoring complex, megadiverse fish assemblages, providing a representative sample of the assemblage structure.

The FishCD overcomes key limitations of conventional fishway monitoring (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017; Roscoe & Hinch, 2010). Unlike trap-based systems affected by behavioural avoidance, size selectivity and intermittent sampling (Jones & O'Connor, 2017), it operates continuously and samples all fish exiting the structure irrespective of species-specific responses to monitoring gear. Its passive, current-driven mechanism avoids reliance on external power and maintains constant sampling effort across environmental states.

The FishCD operation remains effective under the high turbidity typical of tropical rivers - Madeira River has high concentrations of suspended sediment (Graça et al., 2025; Junk et al., 2011) that prevent sufficient water clarity for reliable video monitoring. It also operates efficiently in low-light conditions, enabling the detection of nocturnal migrants, unlike video-based methods. The fishway exit conveys high-velocity flows that emulate the former rapids, which generating bubble plumes that compromise split-beam sonar and infrared scanning. In contrast, the FishCD operates reliably under these high-velocity, turbulent, and bubble-rich conditions. Its performance, including the capture and counting of fish, remains unimpaired by air entrainment and turbulence.

An operational challenge during deployment was the unusually high load of large woody debris (LWD). During the wet season on the Madeira River, large amounts of LWD are transported downstream; while most is diverted to the log spillway, some escapes and accumulates around the FishCD and others hydropower structures. At times, LWD collided with the device, causing damage and necessitating suspension of sampling in February. The episode was atypical, likely linked to two preceding years of drought (Espinoza et al., 2024; Laureanti et al., 2024) that mobilised exceptional volumes of LWD, and to debris-retention infrastructure being compromised by the extraordinary load.

A further constraint was related to SA-HPP´s safety protocols that prevent night-time processing. However, integrated holding tanks (live boxes) maintained fish in good condition until the first morning inspection, mitigating this limitation and protecting both data integrity and welfare.

Integration with other monitoring approaches can broaden evaluation. Biotelemetry yields high-resolution movement data for tagged individuals but is constrained by low detectability at fishway exits, where depth reduces antenna efficiency, and has produced low detection rates for *Brachyplatystoma* at SA-HPP (Hahn et al., 2020). Despite these limitations, biotelemetry allows estimation of passage proportions for tagged fish. By enumerating all individuals completing the fishway irrespective of tagging status, the FishCD records passage events directly. Used together, these approaches enable cross-validation of telemetry-based inferences and support evaluation across biological scales.

FishCD outputs enable quantitative analysis of passage patterns. A negative flow between discharge and capture rate suggests sensitivity to environmental drivers -likely reflecting attraction flows and hydraulics within the passage - and aligns with previous evidence that hydraulic conditions influence passage success (Baek et al., 2015; Silva et al., 2018; Wilkes et al., 2018).

Temporal patterns aligned with known migration dynamics in Amazonian assemblages, with elevated captures during September-November coincided with the dry-to-wet season transition (Arias et al., 2021, 2021; Espinoza et al., 2019), a recognised trigger for upstream movements (Barthem et al., 2017; Barthem & Goulding, 1997). The pattern of captures supports the device's value to monitoring programmes aimed at resolving species-specific responses to environmental variation.

Diel patterns further illustrated temporal variation that would be difficult to detect with intermittent sampling. Although larger absolute night-time counts partly reflect longer unattended intervals, the higher frequency of exits at night suggests a behavioural preference for night-time exit, a pattern also reported in studies of fishway passage and diel activity in Neotropical fishes (Bizzotto et al., 2009; Lopes et al., 2018).

As a prototype developed under the Brazilian Electricity Regulatory Agency's Research, Development and Innovation Programme (ANEEL; PD-06683-0320/2020), the FishCD remains experimental, and these initial results provide a foundational assessment of its performance for monitoring fishway efficacy. The catch over eight months underscores the value of multi-season or multi-year operation to encompass the full diversity of potential users. Furthermore, exceptional drought conditions in 2022 and 2023 (Espinoza et al., 2024; Laureanti et al., 2024) may also have influenced the reproduction (Röpke et al., 2024) the recruitment dynamics of migratory species, and overall passage rates in the Amazon Basin. This context reinforces the necessity for longer-term monitoring to account for interannual climate variation.

Finally, positioning at a single exit ensures complete enumeration of successful passages but does not resolve entrance rates or within-structure behaviour. This constraint is shared by other exit-based methods. For comprehensive assessment, FishCD data could be combined with entrance monitoring to derive efficiency metrics (Castro-Santos, 2009; Hershey, 2021), although challenging hydraulics at many entrances may limit the feasibility of some technologies (Hahn et al., 2020).

Methodological Advances and Future Directions

The scalable design of the FishCD enables deployment at fishway exits across diverse dam types, providing the foundation for standardised monitoring protocols that facilitate comparative studies of fishway performance and support evidence-based improvements in passage design. Its ability to generate quantitative, species-specific data addresses a fundamental requirement for biological assessment of passage effectiveness in regulated rivers. Further development should prioritise extended monitoring across full hydrological cycles, integration with complementary approaches, and refinement of protocols that incorporate environmental drivers of fish movement. This approach standardises sampling and directly enumerates successful passage, reducing uncertainty and yielding robust species-specific data that support adaptive management while maintaining a non-invasive, comprehensive monitoring approach.

By addressing the persistent challenge of standardising biological assessments across diverse contexts (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017), the FishCD represents a methodological advance in fishway monitoring. Its demonstrated performance under variable operational conditions positions it as a valuable tool for generating comparable datasets across river systems, ultimately supporting conservation strategies and fishway design improvements worldwide.

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Supporting Information

Table S1. Taxonomic classification and functional traits of fish species captured by the FishCD during the testing phase between September 2024 and April 2025. Taxa are organized by Order, Family, and Species. Functional traits were classified according to feeding preference (FD; Carnivorous - Car, Detritivorous - Det, Herbivorous - Heb, Invertivorous - Inv, Omnivorous - Omn, Piscivorous - Pis), water column preference (WCP; Benthopelagic - BPe, Demersal - Dem, Pelagic - Pel), reproductive period (RP; Dry season - Dry, Wet season - Wet, Both seasons - Both), migratory behaviour (MB; Migratory - M, Non-migratory - N-M), and total length (TL; in centimeters). All species were recorded upstream and downstream of the Santo Antônio and Teotônio Rapids between 2009 and 2011 (Cella-Ribeiro et al., 2017; Queiroz et al., 2013; Torrente-Vilara, 2009; Torrente-Vilara et al., 2011), before the dam closure.

Order	Family	Species	N	FD	WCP	RP	MB
Characiformes	Bryconidae	*Brycon amazonicus*	3	Omn	Bpe	Both	M
Characiformes	Cynodontidae	*Rhaphiodon vulpinus*	20	Pis	Pel	Both	M
Characiformes	Prochilodontidae	*Prochilodus nigricans*	10	Det	Bpe	Wet	M
Characiformes	Prochilodontidae	*Semaprochilodus insignis*	1	Det	Bpe	Wet	M
Characiformes	Serrasalmidae	*Colossoma macropomum*	1	Omn	Bpe	Wet	M
Characiformes	Serrasalmidae	*Myloplus lobatus*	3	Her	Bpe	Dry	M
Characiformes	Serrasalmidae	*Mylossoma aureum*	1	Omn	Bpe	Wet	M
Characiformes	Serrasalmidae	*Serrasalmus* sp.1	1	Car	Bpe	Wet	N-M
Characiformes	Serrasalmidae	*Serrasalmus* sp.2	1	Car	Bpe	Wet	N-M
Characiformes	Triportheidae	*Triportheus albus*	1	Inv	Bpe	Wet	M
Characiformes	Triportheidae	*Triportheus* sp.1	1	Inv	Bpe	Wet	M
Siluriformes	Pimelodidade	*Brachyplatystoma platynema*	1	Pis	Dem	Wet	M
Siluriformes	Pimelodidade	*Brachyplatystoma rousseauxii*	4	Pis	Dem	Wet	M
Siluriformes	Pimelodidade	*Calophysus macropterus*	35	Pis	Dem	Wet	M
Siluriformes	Pimelodidade	*Platynematichthys notatus*	58	Pis	Dem	Wet	M
Siluriformes	Pimelodidade	*Sorubim elongatus*	1	Car	Dem	Wet	M
Siluriformes	Pimelodidade	*Zungaro zungaro*	6	Pis	Dem	Wet	M
Siluriformes	Doradidae	*Pterodoras granulosus*	1	Her	Dem	Wet	M

DISCUSSÃO GERAL

Alterações na estrutura das assembleias de peixes são esperadas após a formação de reservatórios, uma vez que o barramento modifica a conectividade fluvial e altera as condições ambientais (A. A. Agostinho et al., 2016; Eloranta et al., 2018; Fráguas & Pompeu, 2021; Loures & Pompeu, 2018; Orsi & Britton, 2014; Pelicice et al., 2018; Perônico et al., 2020; C. Zhang et al., 2020).

No presente estudo, entretanto, os índices de diversidade taxonômica (e.g., riqueza) não indicaram que a reorganização da ictiofauna estivesse associada à formação do reservatório. Por outro lado, a vazão do Rio Madeira e a distância das assembleias em relação à barragem permaneceram como os principais fatores explicativos das variações na estrutura da assembleia após o represamento.

De forma complementar, a abordagem funcional indicou que as alterações na ictiofauna estiveram associadas a todos os preditores ambientais considerados, incluindo a formação do reservatório, a vazão do rio Madeira e a distância das assembleias em relação à barragem. Considerando que os atributos funcionais das espécies estão associados aos processos ecológicos, as alterações ambientais decorrentes do represamento levaram a alterações nos índices da diversidade funcional da assembleia (Chen et al., 2023; Mouillot et al., 2013; Sagouis et al., 2016). Esses resultados corroboram a hipótese de que a diversidade funcional é mais sensível às mudanças ambientais associadas à fase de colonização do reservatório do que os índices tradicionais de diversidade taxonômica.

Como esperado, a perturbações ambientais ocasionadas pelo reservatório mostraram baixa sensibilidade quando avaliadas por meio dos índices taxonômicos (Mouillot et al., 2013), sendo mais claramente evidenciadas pelos índices de diversidade funcional. Relações semelhantes entre diversidade taxonômica e funcional da ictiofauna têm sido observadas em outros reservatórios (Jia et al., 2020; Lin et al., 2021; Queiroz-Sousa et al., 2019; Sagouis et al., 2016; C. Zhang et al., 2020, 2020, 2022), reforçando que métricas funcionais tendem a responder de forma mais precoce e direta às alterações ambientais, uma vez que refletem modificações no conjunto de atributos associados ao desempenho ecológico das espécies.

As mudanças na estrutura da assembleia após a formação do reservatório foi conduzida predominantemente por variação balanceada na abundância das espécies, e não por gradientes unidirecionais de abundância. Esse padrão indica uma reconfiguração da assembleia baseada na substituição compensada de indivíduos entre espécies, sem evidência de perdas direcionais consistentes ao longo dos gradientes espacial ou temporal. A predominância variação balanceada sugere ausência de homogeneização associada à redução sistemática de abundâncias (Baselga, 2010, 2012, 2013) e pode estar relacionada à elevada diversidade taxonômica do Rio Madeira (Graça et al., 2025; Queiroz et al., 2013; Ohara, 2018), que disponibiliza um amplo conjunto de espécies funcionalmente distintas, capazes de substituir-se em termos de abundância relativa. Adicionalmente, variações na abundância das espécies são esperadas em ambientes amazônicos, fortemente influenciados pela sazonalidade hidrológica (Barthem & Goulding, 1997; Goulding, 1980; Junk et al., 1989). Nesse contexto, a operação do reservatório a fio d’água tende a preservar parte dessa dinâmica hidrológica natural, permitindo que oscilações sazonais na abundância relativa continuem atuando como mecanismo estruturador da assembleia.

O gradiente espacial longitudinal ao longo do reservatório também contribuiu de forma significativa para a diferenciação das assembleias, como relatado em outros reservatórios neotropicais (A. A. Agostinho et al., 2016; Baumgartner et al., 2018). Historicamente, a área do rio atualmente adjacente à barragem situava-se entre a corredeira de Santo Antônio, a jusante, e as corredeiras do Macaco e do Teotônio, a montante, uma configuração que já impunha restrições naturais à distribuição das espécies antes do represamento. Nesse contexto, embora não tenha sido detectado um gradiente longitudinal nítido no reservatório, observou-se um contraste marcado entre a área próxima à barragem e os trechos mais a montante.

De forma consistente com essa condição historicamente mais restritiva, o ponto mais próximo da barragem (RSB) apresentou assembleias comparativamente menos diversas e mais uniformes, tanto em termos taxonômicos quanto funcionais. Essa estrutura deve ser interpretada como uma caracterização espacial da área no período pós-formação do reservatório, uma vez que esse trecho não dispunha de monitoramento prévio. O padrão local está provavelmente associado às condições hidrodinâmicas predominantes na região adjacente à barragem. Mesmo sob um regime a fio d’água, caracterizado por fraca estratificação limnológica e tempos de residência relativamente curtos, as diferenças ambientais tendem a ser mais pronunciadas nas proximidades da barragem, o que pode restringir o conjunto de táxons capazes de ocupar esse setor e resultar em um espaço funcional mais limitado.

A vazão do Rio Madeira, intrinsecamente associada à sua sazonalidade, manteve-se como determinante da estruturação da ictiofauna mesmo após a formação do reservatório. A permanência da vazão como fator estruturador da assembleia sugere que a operação a fio d'água promoveu alterações menos pronunciadas no regime hidrológico, mantendo características lóticas do trecho. Este cenário contrasta com o observado em reservatórios de acumulação, que tipicamente transformam sistemas fluviais em ambientes predominantemente lênticos.

Ao contrário do previsto para reservatórios de acumulação neotropicais durante a fase de colonização (A. A. Agostinho et al., 1999), nos quais se observa aumento de piscívoros e carnívoros e maior participação de espécies de pequeno porte, a assembleia analisada apresentou uma reorganização trófica distinta, caracterizada por redução relativa dessas categorias funcionais e por incremento na representatividade de classes de tamanho intermediário após a formação do reservatório. Ainda assim, de forma consistente com o esperado para a fase de colonização, houve aumento na abundância de espécies com preferências alimentares mais generalistas, como onívoros e detritívoros, possivelmente associado ao aporte de matéria orgânica terrestre ao sistema. Paralelamente, espécies com corpos alongados ou fusiformes também aumentaram após o represamento. Tais características estão associadas à exploração de recursos dispersos e à manutenção da posição em fluxos d' água (Blake, 2004), reforçando que a vazão do Rio Madeira permanece como fator estruturador da assembleia. A redução relativa de piscívoros combinada ao aumento de espécies generalistas indica uma reestruturação trófica compatível com estágios mais avançados do período de instabilidade descrito para reservatórios, o qual pode se estender por até 30 anos (A. A. Agostinho et al., 1999). Contudo, essa estimativa baseia-se em estudos conduzidos em reservatório de acumulação, enquanto o presente estudo avalia um reservatório a fio d'água, caracterizado por apresentar espelho d'água reduzido e vazão próxima à natural. Esses resultados reforçam que não há um padrão único de colonização e reorganização da ictiofauna, sendo tais processos fortemente influenciados pelas características específicas de cada reservatório (A. A. Agostinho et al., 1999).

Após a formação do reservatório, três novos pontos de amostragem foram incorporados ao programa de monitoramento. Esses pontos foram alocados em áreas próximas às antigas corredeiras, anteriormente de difícil acesso, nas quais os petrechos de amostragem usualmente empregados (e.g. redes de espera) não eram aplicáveis devido à forte correnteza e à presença de obstáculos rochosos. A inclusão dessas unidades amostrais, associada ao maior tempo de monitoramento no período pós-formação do reservatório (sete anos, em comparação a três anos no período anterior), pode ter ampliado a detecção de espécies (A. G. Oliveira et al., 2018) e, consequentemente, contribuído para as diferenças observadas na estrutura da assembleia.

Mesmo operando sob regime a fio d'água, foram observadas variações nos índices de diversidade funcional da assembleia de peixes, sem alterações na diversidade taxonômica. A variação desses índices esteve associada principalmente à vazão do Rio Madeira, cujo regime foi mantido próximo ao natural pela operação, e ao gradiente espacial em relação à barragem, cuja área adjacente já apresentava restrições ambientais previamente descritas, resultando em valores relativamente menores. O padrão observado foi predominantemente conduzido por variação balanceada na abundância das espécies, indicando redistribuição na abundância relativa entre táxons, sem evidência de homogeneização da assembleia (Baselga, 2013). Como consequência, houve maior representatividade relativa de espécies com estratégias alimentares generalistas (onívoros e detritívoros) e de morfotipos alongados ou fusiformes associados a ambientes lóticos. Nesse contexto, os atributos funcionais responderam de forma mais sensível às condições ambientais do que os índices taxonômicos.

A assembleia do reservatório é influenciada por fatores ambientais, como a distância em relação à barragem e a vazão do rio (mantida próxima à condição natural pelo regime de operação a fio d'água), bem como pelo contexto de conectividade longitudinal do sistema, associado à presença do STP. A implantação de STPs constitui uma das principais medidas de mitigação adotadas mundialmente em empreendimentos hidrelétricos, com o objetivo de restabelecer a conectividade longitudinal e viabilizar a passagem de peixes entre os trechos a jusante e a montante da barragem. Entretanto, avaliar a eficiência desses sistemas permanece um desafio mundial, em grande parte devido à ausência de protocolos padronizados e de métricas comparáveis, o que limita a avaliação do sucesso de passagem e a comparação entre sistemas (Bunt et al., 2016; Hershey, 2021; Kemp, 2016; Lira et al., 2017).

Nesse contexto, desenvolveu-se e testou-se um dispositivo experimental de captura (PDI PD-06683-0320/2020) no STP da UHE SA (Rio Madeira), com o objetivo de aprimorar o monitoramento da passagem de peixes e propor uma ferramenta replicável para avaliação do desempenho de STPs em rios neotropicais.

O dispositivo capturou espécies com diferentes estratégias migratórias e ecológicas, incluindo desde pequenos caracídeos (e.g. *Triportheus* spp.) até grandes siluriformes migradores (*Brachyplatystoma* spp.). Entre estas, registraram-se espécies pelágicas (*Rhaphiodon vulpinus*), bentopelágicas (*Brycon amazonicus*, *Mylossoma aureum*, *Semaprochilodus insignis*) e demersais (*Brachyplatystoma platynema*, *Zungaro zungaro*), incluindo *Brachyplatystoma rousseauxii*. Essa diversidade taxonômica e ecológica sugere baixa seletividade do dispositivo em relação a diferentes estratégias de natação e formatos corporais. Além disso, a ausência de lesões externas e sinais de predação nos espécimes capturados, indica que o dispositivo atende a critérios básicos de bem-estar animal. Dessa forma, o dispositivo mostrou-se adequado para o monitoramento de assembleias megadiversas, fornecendo uma amostra representativa da estrutura da assembleia que efetivamente completa o trajeto pelo STP.

O dispositivo apresenta vantagens operacionais significativas frente aos desafios metodológicos recorrentes na avaliação da passagem de peixes. Diferentemente de armadilhas que podem induzir evasão comportamental ou introduzir seletividade por classe de comprimento ou morfotipo (Jones & O'Connor, 2017; Mekong River Commission, 2023), seu desenho passivo e contínuo, acionado pelo fluxo de água, proporciona esforço amostral estável e tende a capturar os indivíduos que concluem a passagem, independentemente da espécie. Além disso, o equipamento mantém sua funcionalidade em condições de alta turbidez, baixa luminosidade e sob escoamento turbulento, fatores que frequentemente comprometem outras técnicas, como sistemas de vídeo e o uso de sonares (Cui et al., 2024; Haas et al., 2024; Xu et al., 2024).

Seu desempenho contrasta com as limitações de métodos frequentemente empregados. A biotelemetria (Cooke et al., 2012; Hahn et al., 2020) envolve custos operacionais elevados e geralmente se restringe a poucas espécies, enquanto os sistemas de hidroacústica têm seu desempenho prejudicado em rios turbulentos devido à formação de bolhas e ao ruído do escoamento (Cui et al., 2024). Além disso, a descontinuidade das amostragens, outro desafio comum na avaliação da passagem de peixes (Jones & O'Connor, 2017; Mekong River Commission, 2023), é minimizada pela operação contínua do dispositivo. Como consequência, o método viabiliza a quantificação direta e comparável dos peixes que completaram o trajeto pelo sistema, reduzindo a incerteza inerente às amostragens intermitentes e fornecendo base empírica mais robusta para a avaliação da efetividade da passagem, o que o posiciona como ferramenta promissora para o monitoramento padronizado de sistemas de transposição de peixes.

A operação do dispositivo apresentou dois principais contratempos: o elevado volume de troncos flutuantes e a impossibilidade de realizar vistorias noturnas nos tanques laterais. No Rio Madeira, durante os períodos de enchente e cheia, grandes quantidades de troncos de madeira de grande porte são transportadas a jusante. Em fevereiro, colisões de troncos com o dispositivo provocaram danos ao equipamento e resultaram na suspensão temporária das amostragens. As secas atípicas registradas em 2022 e 2023 (Espinoza et al., 2024; Laureanti et al., 2024) possivelmente mobilizaram volumes excepcionais de troncos, contribuindo para o evento observado no período de 2024-2025. Em relação aos protocolos de segurança, cada empreendimento hidrelétrico adota diretrizes próprias. No caso da UHE SA, atividades embarcadas no período noturno não eram autorizadas pela equipe de Saúde e Segurança do Trabalho (SST). Ainda assim, o dispositivo manteve funcionamento contínuo durante a noite, e os tanques laterais preservaram os peixes capturados em boas condições até a primeira inspeção matinal, assegurando a integridade dos dados e o atendimento a critérios de bem-estar animal.

Finalmente, como o reservatório ainda se encontra na fase de colonização, novas alterações na estrutura da ictiofauna podem ocorrer ao longo do tempo, processo adicionalmente influenciado pela dinâmica sazonal do pulso hidrológico (Barthem & Goulding, 1997; Goulding, 1980; Junk et al., 1989). Nesse contexto, a integração de diferentes dimensões da diversidade, taxonômica e funcional, e com dados de passagem de peixes pelo STP é fundamental para avaliar as respostas da ictiofauna às mudanças ocasionados pelo empreendimento. A manutenção de um programa de monitoramento de longo prazo torna-se, portanto, essencial para detectar mudanças ecológicas graduais e aprimorar a precisão na avaliação de impactos em rios neotropicais. Os dados gerados por esses programas fornecem base científica para orientar futuras tomadas de decisão e subsidiar estratégias de manejo e conservação diante das mudanças nas condições ambientais.

CONCLUSÃO GERAL

Com base nos resultados apresentados nos Capítulos I e II, conclui-se que:

- Considerando os índices de diversidade taxonômica, a reorganização da assembleia esteve associada principalmente à vazão do Rio Madeira e à distância em relação à barragem; por sua vez, os índices de diversidade funcional responderam também à formação do reservatório, além da vazão e do gradiente espacial;
- A variação balanceada na abundância das espécies explicou predominantemente as mudanças observadas nos índices da assembleia, sem evidência de homogeneização biótica;
- A vazão do Rio Madeira e a distância à barragem configuraram-se como fatores estruturadores da assembleia no reservatório;
- Observou-se um gradiente longitudinal de diversidade, ainda que discreto, caracterizado por assembleias menos diversas e mais uniformes nas proximidades da barragem;
- A diversidade de guildas registradas demonstra que o dispositivo (FishCD) é capaz de amostrar ampla variedade de espécies, com diferentes estratégias ecológicas;
- As capturas foram influenciadas negativamente pela vazão defluente, indicando que o funcionamento do STP depende das regras operativas e da vazão do rio, especialmente em sistemas que mantêm variações sazonais do regime hidrológico;
- O dispositivo (FishCD) pode ser aplicado em diferentes empreendimentos hidrelétricos que possuam STPs com saídas de peixes, viabilizando a coleta padronizada de dados. Essa padronização permite comparações entre diferentes modelos de STPs, contribuindo para a avaliação da eficiência de passagem, o desenvolvimento de estratégias de conservação de espécies migradoras e o embasamento de decisões em projetos futuros;
- O monitoramento de longo prazo, com abordagem multidimensional que integre diversidade taxonômica e funcional, é fundamental para detectar respostas ecológicas da assembleia e orientar decisões de manejo e conservação diante das alterações geradas por reservatórios.

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