UNIVERSIDADE DE BRASÍLIA
INSTITUTO DE CIÊNCIAS BIOLÓGICAS
PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA

Impactos de pequenas barragens sobre a comunidade de peixes na Amazônia

Amanda Santos Vasconcelos

Orientador: Murilo Sversut Dias

Brasília - DF
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Dissertação apresentada ao Programa de Pós-Graduação em Ecologia, do Instituto de Ciências Biológicas da Universidade de Brasília, como requisito parcial para a obtenção do título de Mestre em Ecologia. Orientador: Prof Dr. Murilo Sversut Dias.

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Dedicatória

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Resumo

A fragmentação de ecossistemas aquáticos por pequenas barragens e cruzamentos de estradas representa uma ameaça significativa à biodiversidade de ambientes lóticos, como riachos. Essas estruturas têm um impacto subestimado e pouco estudado devido ao seu tamanho reduzido. No Brasil, especialmente no Mato Grosso, sua rede de estradas exacerba o problema, muitas construções não autorizadas e não documentadas contribuem para a fragmentação dos corpos d'água e degradação do habitat. Compreender os efeitos dessas barreiras nos riachos e na biodiversidade de peixes é crucial, especialmente em regiões como a Amazônia. O presente estudo investigou o efeito dessas pequenas estruturas em riachos do Alto Xingu, Brasil, concentrando-se em avaliar mudanças ambientais e os impactos do isolamento ocasionado pelas barreiras na comunidade de peixes de riachos. Além disso, nós testamos o efeito dos atributos das barreiras no ambiente e na fauna de peixes. A amostragem foi realizada de forma pareada, com cada riacho amostrado duas vezes, a montante e a jusante de uma barreira, permitindo comparações. As barragens estudadas eram predominantemente pequenas, com menos de 6 metros de altura, construídas entre o final da década de 1980 e ao longo da década de 1990. As diferenças ambientais entre os tratamentos foram analisadas com Análise de Componentes Principais (PCA) e Análise de Variância Multivariada Permutacional (perMANOVA). Modelos lineares mistos foram usados para identificar diferenças significativas em variáveis ambientais entre os tratamentos. Abundância total, riqueza e composição de espécies de peixes foram analisadas usando modelos lineares de efeitos mistos, perMANOVAs e representados por Escalonamento multidimensional não métrico (NMDS). A Análise de Espécies Indicadoras (ISA) foi empregada para identificar espécies associadas a tratamentos específicos, e os efeitos das características das barragens (altura e idade) na dissimilaridade ambiental e da comunidade de peixes foram testados usando modelos lineares simples. Foram observadas diferenças significativas entre riachos de fluxo livre (sem barreira) e os desconectados (barrados). As análises mostraram variação ambiental marcante nos parâmetros de condutividade, oxigênio dissolvido e pH entre os trechos analisados. A fauna de peixes parece não ser afetada significativamente pelas barreiras, com abundância total, riqueza e composição de espécies feita com dados de abundância semelhantes entre os tratamentos analisados. Já a composição das espécies feita com dados de presença/ausência difere significativamente, com algumas espécies associadas a locais acima ou abaixo das barreiras. Surpreendentemente, não foram encontrados efeitos das características de barreira na dissimilaridade do habitat e da fauna de peixes. Nosso estudo mostra que as barreiras causam uma mudança sutil na qualidade da água e na composição

acima e abaixo das barreiras, mas não há indícios de drásticas alterações de hábitat ou extirpação local de espécies acima ou abaixo das barreiras. No geral, o estudo forneceu insights sobre os impactos de pequenas barreiras nos ambientes de riachos e nas comunidades de peixes, destacando a importância de considerar tanto os fatores ambientais quanto os ecológicos nos esforços de conservação de água doce.

Abstract

Aquatic ecosystem fragmentation by small dams and road crossings poses a significant threat to the biodiversity of lotic environments, such as streams. These structures have an underestimated and understudied impact due to their small size. In Brazil, especially in Mato Grosso, where road networks exacerbate the problem, many unauthorized and undocumented constructions contribute to the fragmentation of water bodies and habitat degradation. Understanding the effects of these barriers on streams and fish biodiversity is crucial, especially in regions like the Amazon, with vast road networks posing significant environmental threats. The present study investigated the effect of these small structures on streams in the Upper Xingu, Brazil, focusing on assessing environmental changes and the impacts of isolation caused by barriers on fish communities. Additionally, we tested the effect of barrier attributes on the environment and fish fauna. Sampling was conducted in pairs, with each stream sampled twice, upstream and downstream, allowing for comparisons. The studied dams were predominantly small, less than 6 meters in height, built between the late 1980s and throughout the 1990s. Environmental differences between treatments were analyzed using Principal Component Analysis (PCA) and Permutational Multivariate Analysis of Variance (perMANOVA). Mixed linear models were used to identify significant differences in environmental variables between treatments. Total abundance, richness, and species composition of fish were analyzed using mixed-effects linear models, perMANOVA, and represented by Non-metric Multidimensional Scaling (NMDS). Indicator Species Analysis (ISA) was employed to identify species associated with specific treatments, and the effects of dam characteristics (height and age) on environmental dissimilarity and fish community were tested using simple linear models. Significant differences were observed between free-flowing (unbarred) streams and disconnected (barred) streams. The analyses showed marked environmental variation in conductivity, dissolved oxygen, and pH parameters among the analyzed stretches. The fish fauna appears not to be significantly affected by the barriers, with total abundance, richness and species composition based on abundance data being similar between analyzed treatments. However, species composition

based on presence/absence data differed significantly, with some species associated with specific locations above or below the barriers. Surprisingly, no effects of barrier characteristics on habitat and fish fauna dissimilarity were found. Our study shows that barriers cause subtle changes in water quality and composition above and below the barriers, but there is no evidence of drastic habitat alterations or local extirpation of species above or below the barriers. Overall, the study provided insights into the impacts of small barriers on stream environments and fish communities, highlighting the importance of considering both environmental and ecological factors in freshwater conservation efforts.

Introdução Geral

Os ambientes de água doce ocupam apenas 0,8% da superfície terrestre, porém abrigam uma grande diversidade de organismos, especialmente de peixes (Dudgeon et al. 2006; Dawson 2012). Essa excepcional diversidade é denominada de paradoxo dos peixes de água doce (McDermott 2021), que representam mais de 20% das espécies de vertebrados descritas no mundo (Balian et al. 2008). A bacia Amazônica é particularmente rica, considerada a mais diversa bacia de água doce do mundo, com 2.411 espécies de peixes (Reis et al. 2016). No entanto, os ambientes dulcícolas estão entre os habitats mais ameaçados devido à importância da água doce para os seres humanos (Tickner et al. 2020).

A fragmentação dos ecossistemas aquáticos representa uma das principais ameaças aos ecossistemas e à biodiversidade de água doce, contribuindo significativamente para a crise da biodiversidade nesses ambientes, com muitas espécies de peixes extintas ou ameaçadas (Dudgeon et al. 2006; Vörösmarty et al. 2010; Costello 2015; Grill et al. 2019). Mudanças antrópicas nos cursos d'água, como açudes, barragens e travessias de estradas, resultam em declínios na biodiversidade aquática (Dudgeon et al. 2006; Castello et al. 2013; Costello 2015; Pocewicz and Garcia 2016; Dias et al. 2017; Wang et al. 2018; Dudgeon 2019). Essa fragmentação prejudica a conectividade longitudinal, da cabeceira até a foz, dos rios, criando barreiras físicas e comportamentais que bloqueiam a movimentação dos peixes no ambiente (Ward 1989; Agostinho, Pelicice, and Gomes 2008; Pelicice, Pompeu, and Agostinho 2015; Winemiller et al. 2016; Zarri et al. 2022).

Cerca de 40 mil grandes e médias barragens fragmentam aproximadamente 65% dos grandes rios do mundo, além de inúmeras estruturas em fluxos menores, principalmente em áreas de cabeceira (Nilsson et al. 2005; Grill et al. 2019; Barbarossa et al. 2020; Mulligan, Van Soesbergen, and Sáenz 2020; Zhang and Gu 2023). No entanto, pouco se sabe sobre as pequenas barreiras (menos de 05 metros de altura) presentes em riachos de primeira e terceira ordem, voltadas para a agricultura ou para travessias de estradas (Januchowski-Hartley et al. 2013). As travessias de estradas, embora representem a maioria das barreiras, recebem pouca atenção em estudos de fragmentação, representando apenas 4% das pesquisas realizadas (Zarri et al. 2022).

As pequenas barreiras oferecem uma gama de benefícios, como abastecimento de água, controle de inundações e servem como travessias de estradas. Predominantemente localizadas em riachos, principalmente afluentes de cabeceira, essas estruturas criam pequenos fragmentos isolados a montante (Poff and Hart 2002; Liermann et al. 2012; Pocewicz and Garcia 2016; Anderson et al. 2018; Jumani et al. 2020; Mulligan, Van Soesbergen, and Sáenz 2020; Freitas et al. 2022; Sun et al. 2023). No entanto, a falta de documentação de centenas de milhares ou até milhões dessas estruturas nos bancos de dados existentes resulta em desconhecimento de características essenciais, como localização espacial e volume de armazenamento (Liermann et al. 2012; Grill et al. 2019; Belletti et al. 2020; Mulligan, Van Soesbergen, and Sáenz 2020; Sun et al. 2023). Além disso, outros fatores relevantes como altura, forma, idade, tamanho e uso de estruturas não são conhecidos; Consequentemente, variações nos atributos podem ter diferentes efeitos na dispersão de indivíduos na rede hidrológica e na qualidade do hábitat (Poff and Hart 2002; Fuller, Doyle, and Strayer 2015; Grill et al. 2019). Essa falta de informações implica que os efeitos de fragmentação dos ecossistemas e na biodiversidade de água doce sejam subestimados, uma vez que muitas barreiras são ignoradas devido ao seu tamanho (Liermann et al. 2012; Couto and Olden 2018; Grill et al. 2019; Belletti et al. 2020; Sun et al. 2023).

A presença de barreiras, principalmente as grandes (> 5 m), causam impactos ecológicos significativos no ambiente. Ao reduzirem ou interromperem o fluxo natural de água, transformam os sistemas de água corrente em corpos d'água parados, resultando em mudanças físicas e químicas (Cote et al. 2009; Wang et al. 2018; Grill et al. 2019; Zaidel et al. 2021; Zhang and Gu 2023). Deste modo, ocorrem alterações nos padrões de transporte de sedimentos e consequentemente retenção de substratos, alterações nos regimes de inundações, mudanças na velocidade do fluxo de água, alterações da condutividade e de temperatura de água (Poff and Hart 2002; Anderson, Freeman, and Pringle 2006; Fuller, Doyle, and Strayer 2015; Timpe and Kaplan 2017; Chandesris et al. 2019; Jumani et al. 2020; Flecker et al. 2022). Assim, essas alterações podem afetar a qualidade dos ambientes aquáticos, principalmente a jusante, e consequentemente, a biodiversidade local (Wang et al. 2018).

Para grandes barragens com grandes reservatórios, é relatado que estas atuam como uma barreira física, química e comportamental para espécies de peixes, favorecendo o aumento de espécies lênticas e funcionando como armadilhas para larvas e ovos destes (Agostinho, Pelicice, and Gomes 2008; Pelicice, Pompeu, and Agostinho 2015). As alterações locais nos corpos d'água podem criar barreiras para o deslocamento das espécies, limitando sua capacidade de lidar com alterações ambientais e rastrear locais adequados (Radinger et al. 2018; Pelicice, Pompeu, and Agostinho 2015). A redução na possibilidade de movimentação dos peixes apresenta resultados persistentes, deletérios e imprevisíveis que podem ser exacerbados por outras perturbações humanas como desmatamento (Haddad et al. 2015). As barreiras afetam as espécies de diferentes maneiras e seus efeitos podem variar de negativos, neutros a positivos conforme a espécie em questão (Puijenbroek et al. 2021; Franklin, Noon, and George 2002). Para algumas espécies, a ocorrência ou abundância são pouco afetadas, porém outras são severamente afetadas, podendo resultar em aumento ou diminuição na ocorrência e na abundância, ou até extinção local (Yan et al. 2013; Wang et al. 2018; Li et al. 2022).

A restrição na movimentação dos peixes entre locais não apenas afeta a dinâmica das populações das espécies, mas também influencia toda a estrutura da assembleia a montante da barreira, podendo resultar em alterações significativas na diversidade e na composição das assembleias de peixes, com consequente perda de espécies e redução do fluxo gênico (Anderson, Freeman, and Pringle 2006; Sá‑Oliveira et al. 2015; Costea et al. 2021; Zarri et al. 2022). Assim, a alta densidade de barreiras nos riachos de cabeceira podem afetar negativamente as populações de peixes devido aos efeitos da fragmentação dos habitats (Perkin and Gido 2012; Zarri et al. 2022). Consequentemente, o aumento do isolamento de populações a montante pode limitar a movimentação de organismos aquáticos entre os habitats, reduzir a recolonização acima da barreira após eventos de extinção local (Fuller, Doyle, and Strayer 2015). Além disso, os indivíduos e espécies isoladas acima das barreiras podem enfrentar uma pressão de seleção maior do que aquelas abaixo, uma vez que podem não receber novos indivíduos na população (Fuller, Doyle, and Strayer 2015; Puijenbroek et al. 2021).

Apesar da região Amazônica ser considerada moderadamente fragmentada, com a maioria de seus longos rios ainda livres de barragens, existem milhares de pequenas barreiras não consideradas na maioria dos estudos (Nilsson et al. 2005; Grill et al. 2019). A Bacia do Rio Xingu, na região Amazônica, é pouco afetada por grandes barragens com índice de integridade acima de 60%, porém possui ao menos 10 mil barreiras em riachos de cabeceira de segunda e terceira ordem (Macedo et al. 2013; Latrubesse et al. 2017). Apesar do impacto significativo e negativo que as barreiras exercem no ambiente fluvial, como discutido acima, a falta de documentação adequada sobre essas estruturas e a escassez de estudos de impactos e planos de mitigação ressaltam a necessidade de pesquisas mais aprofundadas no assunto. Portanto, investigar os efeitos das pequenas barreiras em pequenos corpos d'água e na biodiversidade de peixes do Rio Xingu é fundamental.

Assim, nosso objetivo é avaliar os efeitos de pequenas estruturas agrícolas que servem como travessias de estrada em características ambientais dos riachos, na riqueza de peixes, na abundância e na composição de espécies de riachos amazônicos. Nós esperamos mudanças ambientais nos riachos a jusante das barreiras, principalmente relacionadas à temperatura, e diminuição na abundância e na riqueza de espécies nas seções a montante da barragem devido ao seu isolamento e potencial extirpação local de espécies. Também testamos os efeitos da altura e da idade das barreiras na dissimilaridade ambiental e de peixes entre trechos a montante e a jusante. Neste sentido, esperamos que barreiras maiores e mais antigas tenham maiores impactos na estrutura da comunidade de peixes de riachos.

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Impact of Small Barriers on Environmental and fish assemblage from Amazonian Streams

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Abstract

The fragmentation of aquatic ecosystems poses a significant threat to aquatic biodiversity, primarily due to dams for agriculture and road crossings. However, small barriers are often overlooked in studies, despite their considerable impact on watercourse fragmentation. In Brazil, particularly in the Amazon region, there are thousands of undocumented barriers, underscoring the need to investigate their effects on environmental characteristics and fish biodiversity. We evaluated the impacts of small barriers on habitat characteristics, abundance, richness, and species composition of Amazonian stream fishes. Additionally, we tested whether the height and age of barriers affect streams and fish assemblages. Using standardized sampling in 50 m sections of streams, we conducted paired sampling, comparing upstream and downstream sections of barriers, and included sampling in free-flowing streams. We found significant differences in dissolved oxygen, pH, and water conductivity between upstream and downstream sections. Fish richness and total abundance were similar regardless of the treatments analyzed. The species composition differed significantly between upstream and downstream streams with presence/absence data, but these differences appear to be due to the presence of large river species downstream rather than local extinction of species upstream. The age and height of barriers had no effect on environmental dissimilarity or species composition. Overall, our results suggest than small barriers have an effect on the environment and fish fauna, but the synergistic effects of multiple barriers and intensive land use need further investigation in the future.

Key words: culverts; deforestation arc; neotropical fishes, road crossing; small dams

Introduction

Aquatic fragmentation is a major cause of biodiversity crisis in lotic environments, causing extinctions and threats to many aquatic species (Dudgeon et al. 2006; Vörösmarty et al. 2010; Castello et al. 2013; Costello 2015; Pocewicz and Garcia 2016; Dias et al. 2017; Dudgeon 2019; Pelicice et al. 2021). Fragmentations are usually caused by the constructions of dams and/or road crossings along streams/rivers, significantly changing local aquatic habitats and ecosystems (Januchowski-Hartley et al. 2013; Pocewicz and Garcia 2016; Vörösmarty et al. 2010; Grill et al. 2019). Despite the predominance of small dam structures in streams for water supply or road crossings, detailed quantification about them is scarce and incomplete (Januchowski-Hartley et al. 2013; Zhang and Gu 2023). The number of road crossings surpasses 38 times that of dams in the North American lakes region (Januchowski-Hartley et al. 2013; Zarri et al. 2022). However, although the lack of data spans various countries in Europe, there are over a million small barriers reported in streams and rivers, including 110,944 culverts (Belletti et al. 2020). At the global scale, 11 small hydroelectric dams in small rivers and streams are estimated for each large dam worldwide (Couto and Olden 2018), along with numerous small agriculture dams and road stream crossings. Understanding the effects of those small fragmentation agents on aquatic biodiversity is fundamental to minimizing the aquatic biodiversity crisis.

Most barriers are relatively small structures located on streams, mainly headwater tributaries, which create small isolated upstream fragments from the tributaries (Poff e Hart 2002; Belletti et al. 2020; Jumani et al. 2020; Sun et al. 2023; Freitas et al. 2022). These dams vary according to their structural characteristics such as height and length, and impact the natural flow of rivers based on the volume of water stored (Poff and Hart 2002). Due to their limited storage capacity and size, their characteristics, such as spatial location, storage volume, and operational rules, are not well documented (Poff and Hart 2002; Liermann et al. 2012; Chandesris et al. 2019; Grill et al. 2019; Belletti et al. 2020; Mulligan, Van Soesbergen, and Sáenz 2020; Sun et al. 2023). European data show that most barriers in freshwaters are less than five meters in height, contributing to their underrepresentation in studies and inventories (Belletti et al. 2020). The lack of information about millions of small dams or road crossings means that the effects on freshwater ecosystems are unknown and possibly underestimated, as many structures are ignored due to their small size (Liermann et al. 2012; Belletti et al. 2020).

Small dams and road crossings significantly impact aquatic ecosystems by altering the hydrological regime and the physical and chemical characteristics of streams, affecting habitats and species (Poff and Hart 2002; Daigle 2010; Yan et al. 2013; Fuller, Doyle, and Strayer 2015; Chandesris et al. 2019; Zaidel et al. 2021; Jumani et al. 2020). Road crossings over watercourses with undersized culverts leads to perched channel, increased water flow, and changes in substrate, with silting and light penetration, resulting in impaired downstream habitats and connectivity affecting fish assemblages (Gibson, Haedrich, and Wernerheim 2005; Brejão, Teresa, and Gerhard 2020). The fragmentation of watercourses restricts fish movement and impairs longitudinal connectivity from headwaters to the river mouth (Ward

1989; Warren e Pardew 1998; Gibson, Haedrich, and Wernerheim 2005; Grill et al. 2019; Tickner et al. 2020). Decreased longitudinal connectivity creates physical and physiological barriers that block fish movement upstream and hinders downstream movement, resulting in species loss and reduced gene flow (Daigle 2010; Fuller, Doyle, and Strayer 2015; Zarri et al. 2022). Evidence suggests that the high density of barriers in headwaters negatively affects fish populations due to the effects of habitat fragmentation and simplification (Perkin and Gido 2012; Zarri et al. 2022). In Brazil, dams built to control water flow for irrigation and road crossings are often overlooked in inventories and river fragmentation studies (Castello et al. 2013; Pocewicz and Garcia 2016). The density of impoundment associated with road crossings and agriculture use is 15 times higher in deforested areas than in native forest areas, which represents one impoundment every 7.5 km of stream compared to more than 100 km of forested streams (Pocewicz and Garcia 2016). In the Amazon region, the Xingu River basin is minimally affected by large dams, with an integrity index above 60% (Latrubesse et al. 2017). However, there are at least 10,000 barriers in second and third-order streams in the headwaters of the upper Xingu River basin (Macedo et al. 2013).

Investigating the effects of barriers, such as road crossings, on stream integrity and fish biodiversity in the Amazon is crucial due to the extensive road network in the region, many of which are constructed without authorization in private areas, crossing thousands of water bodies (Pocewicz and Garcia 2016; Botelho et al. 2022). The state of Mato Grosso, Brazil, situated within the deforestation arc, features the largest road network in the Amazon region (Levy et al. 2018; Botelho et al. 2022). Evidence indicates that these barriers cause significant changes in the riverine environment, directly affecting the quality of habitats and local biodiversity (Leal et al. 2016; Pocewicz e Garcia 2016; Leitão et al. 2018; Brejão, Teresa, and Gerhard 2020).

Therefore, we evaluate the effects of small agriculture dams and road crossing on stream characteristics, fish richness, abundance, and species composition of small Amazonian streams. We expected environmental changes downstream of the dams mostly related to temperature and a decrease in species abundance and richness in upstream dam sections due to its isolation. We also tested the effects of dam age, and height on environmental and fish dissimilarity between up- and downstream reaches. In this sense, we expect larger and older barriers to have greater impacts on the fish community structure.

Methods

Study area

We conducted this study in streams from the upper Xingu River basin in August 2022 at Tanguro Farm (Querência - Mato Grosso State, Brazil). This region is a transition zone between the Cerrado and the Amazon biomes (Balch et al. 2008; Maracahipes-Santos et al. 2020), the annual precipitation is 1,800 mm, the mean annual temperature varies between 24° and 26°C (Balch et al. 2008; Alvares et al. 2013), and the elevation is about 400 meters above the sea level (Alvares et al. 2013). The Tanguro Farm (13.428759 S; 51.944721W) is located in the region known as the Brazilian deforestation arc and encompasses a total area of 82,000 hectares. Originally intended for cattle ranching in the 1980s, the farm has recently transitioned to the cultivation of soybeans, cotton, and corn. The property consists of a mosaic of forested areas (nearly 60% of the territory) and croplands (nearly 40%) (Oliveira, Santos, and Santos-Costa 2010).

Our sampling occurred during the dry season (usually between June and September 2022) in streams of first to third order sensu Strahler (1957). Overall, the streams drain to Darro and Tanguro rivers (tributaries of the upper Xingu River basin), are mainly composed of sand substrate, have acid water, and are typical Amazonian *terra firme* streams (Mendonça, Magnusson, and Zuanon 2005; Espírito‑Santo et al. 2009). Even though we sampled streams in both native vegetation and croplands, they all have riparian forests along their stream courses, and canopy cover was overall high in all sampling sites (Supplementary Table 1).

Study design

We sampled four reaches in two streams without dams, considered free-flowing (free_flow), and 16 reaches in six streams with dams (disconnected). Half of the sampled streams drained into the Darro River and half into the Tanguro River (Figure 1). Additionally, we sampled pairs of reaches upstream (disconnected_up) and downstream (disconnected_down) of the assessed barriers, forming one pair per disconnected stream. This paired approach was also applied in the free-flowing streams, with one up- and downstream reach in the same stream, but without barriers. This paired design was crucial for experimentally controlling the variability of other unmeasured factors and better assessing the effects of dams on stream environmental characteristics and fish fauna. Thus, the sampling sites were divided into three treatments with four free-flowing reaches, seven disconnected_up reaches, and nine disconnected_down reaches (Figure 1). As our goal was to evaluate the effects of isolation caused by dams and not the local effects of the created reservoirs, we emphasize that all sampled reaches exhibited characteristics of lotic streams, with flowing water, and forest cover. Sampling between reaches within the same stream was conducted while maintaining the shortest possible distance between them, with a maximum of two kilometers.

Some streams had more than one dam in sequence (CascavelR1 and CascavelR4), so some of our reaches considered downstream may also be reaches upstream of un-sampled dams. This was the case in at least two sampled sub-basins that had two dams in sequence. In these cases, we sampled two areas between dams; the most upstream reach was considered down, and the most downstream reach was considered up, maintaining the nomenclature for comparison between pairs with different treatments. In the other case, we did not sample more downstream reaches because the area was heavily impacted by deforestation, which could introduce confounding effects in the results, and access to the stream was hindered by the amount of lianas in the area.

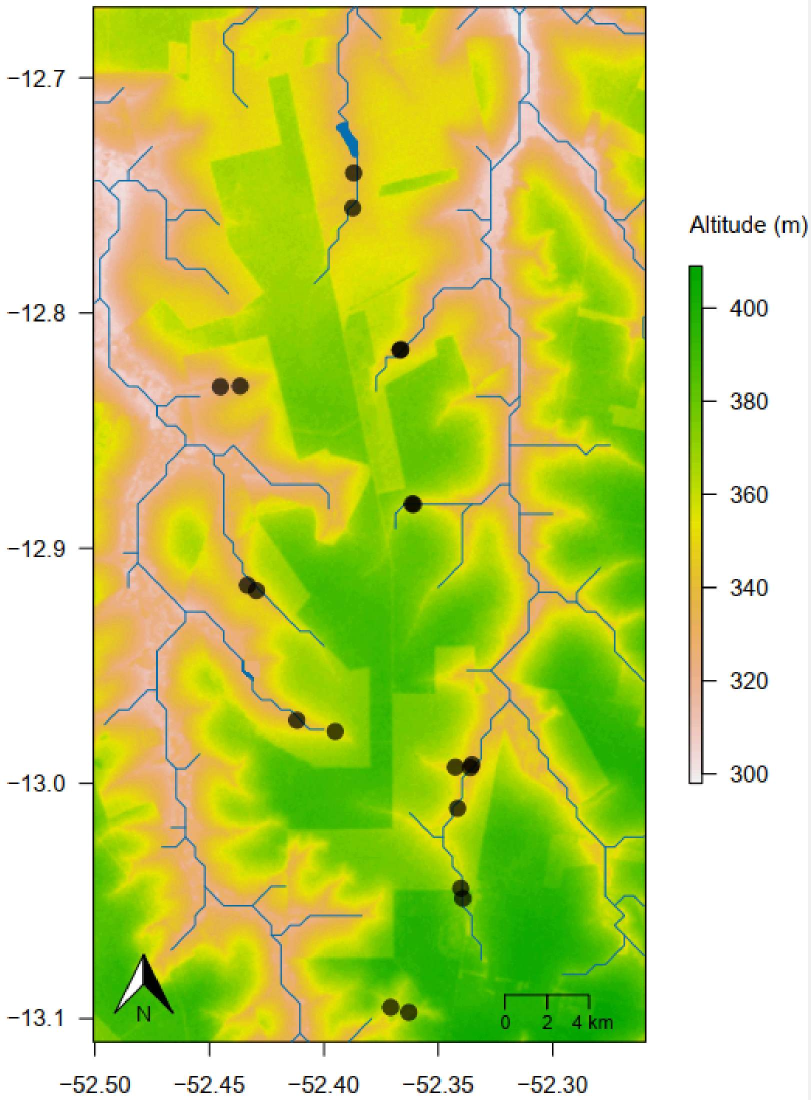

Figure 1: Study area with the 20 sampled reaches divided into eight streams in two sub-basins, Darro and Tanguro rivers, upper Xingu River basin, Mato Grosso State.

Dam's age and height

Dams associated with road crossings are closely linked to changes in land use, primarily serving to regulate water supply for livestock, irrigation of crops, and the construction of pathways for vehicular passage (Macedo et al. 2013; Pocewicz and Garcia 2016). The Tanguro farm features over 20 small dams distributed across the sub-basins of the Darro and Tanguro rivers (table 1). These dams were predominantly constructed using earth embankments with a lateral concrete pipe for water passage, beginning in the early 1980s and extending until the mid-1990s. For this reason, by documenting the year of deforestation within the farm, we were able to determine the construction age of each sampled dam.

Another attribute easily estimated from field measurements is the dam height, which is an indicative of dam size. We estimated the hypotenuse (in meters) and the angle (α, in degrees) of the slope opposing the dam wall; these measurements allowed us to calculate the dam's height using the sine equation from a right triangle: height (opposite side) = hypotenuse * sin α.

Overall, the studied dams are less than 6 meters in height (mean = 2.7 meters, sd = 1.65) and were constructed between the late 1980s and throughout the 1990s (Table 1). Some dams are no longer operational, and consequently, do not receive regular maintenance to prevent potential breaches. Some are even intentionally opened with the assistance of tractors. An example is the Anta R6 dam, which we sampled, where the piping is absent. However, due to the residual presence of water in the reservoir and the lack of additional measures to mitigate damage at the site, we classify this stream as fragmented.

Table 1: Attributes of sampled dams in the Darro and Tanguro River basins, Tanguro Farm, municipality of Querência, Mato Grosso State, Brazil.

Dam name	Stream ID	River basin	Height	Age	Construction year	Elevation
CascavelR1	1	Darro	1.03	34	1990	364
CascavelR4	1	Darro	1.4	32	1992	345
TanguroR2	2	Darro	1.55	38	1986	332
TanguroR4	3	Darro	2.9	38	1986	340
MutumR2	4	Tanguro	3.01	28	1998	339
MutumR4	5	Tanguro	4.24	29	1997	345
Anta R6	6	Tanguro	5.37	32	1992	326

Environmental variables

In order to characterize the environment and obtain physical and chemical measurements of the habitat, we followed the sampling protocol described by Mendonça, Magnusson, and Zuanon (2005). Each sample site was a 50 m-long stream reach where we quantified environmental variables and fish communities. In brief, we recorded the elevation and geographical coordinates for each section and dam. At the beginning of each sampled reach, we used two multiparameter probes (the Metrohm 914 and YSI Professional Optical Dissolved Oxygen, ProODO) to measure water pH, temperature (°C), electrical conductivity (mS/cm), and dissolved oxygen (mg/L), respectively. Then, we recorded the stream width at four equidistant points (at 0, 16, 32, and 50m positions of the stream reach) and took ten depth and substrate characteristics along each of the four width transects (resulting in 40 depth and substrate measurements). Additionally, we recorded three measurements of surface water velocity and took three photos in each of the four transects to estimate canopy coverage (for further details, see Mendonça, Magnusson, and Zuanon 2005). From the width and depth data, we computed the cross-sectional area of the sampled stream channel and multiplied it by the average water velocity to derive the stream flow (Supplementary Table 1).

We registered the frequency of occurrence of 13 substrate types and summarized them in a single habitat diversity metric by calculating the Shannon diversity index using their frequencies for each reach. In addition to substrate diversity, we also used the three most frequent substrate types (sand, fine litter, and course litter) as metrics to characterize habitats in each sampling site (Supplementary Table 1).

Fish sampling

We conducted fish sampling during the daytime in the 50-meter reaches blocked in the same four equidistant positions (0, 16, 32, and 50m) using a fine mesh net (with 5 mm between knots) in order to avoid fish escaping and maximize our sampling effort (Mendonça, Magnusson, and Zuanon 2005). Fish capture was carried out by four peoples over a 45 - 60 minute period, employing a combination of active techniques, such as hand-netting for species associated with leaf litter, roots, and aquatic vegetation, and seine netting for species associated with sandy bottoms and open waters. Sampling was conducted upstream of the water flow (from downstream to upstream) (Uieda and Castro 1999).

We euthanized the captured fish by immersion in clove oil (eugenol, Griffiths 2000). Most of the biological material was fixed in a 10% formalin solution for at least 24 hours, and then preserved in 70% alcohol (Uieda and Castro 1999); whereas some individuals were fixed and preserved in 100% alcohol for future genetic analyses. The identification of the specimens was carried out based on literature and inputs from specialists. The collected fish were deposited in the Ichthyological Collection of the University of Brasília (CIUnB), Brasília, Brazil.

Data Analysis

Preparatory analyses

Due to technical issues with one of the multiparameter probes, it was not possible to record pH and electrical conductivity at only four sampling sites. By using the "namiar" package, we found that the missing data represented only 0.3% of the dataset. In order to avoid the exclusion of sites or predictors, we opted to impute the missing data using the "missForest" package, which employs the Random Forest algorithm. This choice provides greater flexibility in imputing different types of data, whether categorical or quantitative, and the error associated with our imputation was low (16%).

Environmental data

We standardized (mean zero and unity standard deviation) all environmental data using the decostand function from the "vegan" package, and then performed a Principal Component Analysis (PCA) by correlation, aiming to visualize the environmental distinctions among treatments (free_flow, disconnected up and down) using the first two axes (which captured 53% of total variability).

To test for mean differences in environmental data between all treatments while controlling for the effects of other explanatory variables, we performed a Permutational Multivariate Analysis of Variance (perMANOVA), utilizing the adonis2 function from the "vegan" package. The environmental data was transformed in a distance matrix based on the

Euclidean distance and we tested the effect of predictors (treatment, stream ID, and the identity of the sampled pairs) using 4,999 permutations. Both the stream and the pair identity were useful to control for such extra variability and statistically determine a paired comparison in the multivariate space. We also employed a Bonferroni test on the perMANOVA results using the “pairwise.adonis2” function and the sample previous conditions (model description, permutations, and distance) to identify the differences among treatment groups. Subsequently, to identify which environmental variables exhibited differences among treatments, we constructed mixed linear models using the "lmer" function from the "lme4" package. We performed this analysis individually for each of the environmental predictors (dependent variable), considering treatment and sampled sub-basin as independent variables, and pairs as a random effect. The significance of our models was assessed using the Likelihood Ratio Test (LRT) implemented in the drop1 function. To assess the differences between treatments, a Generalized Linear Hypotheses Testing (GLHT) analysis was conducted following the modeling using the mcp (multiple comparison procedures) function from the "multcomp" package with the Tukey test. This approach allowed for pairwise treatment comparisons to identify significant differences between them.

Fish fauna

We conducted analyses using mixed-effect linear models to test for differences in species abundance and richness between treatments. In these models, treatment and sub-basin were treated as independent variables, the sampled pairs as a random effect, and we used LRTs for testing the significance of individual predictors. Then, we performed (GLHT; see above) to test the differences between treatments.

To evaluate community composition, we standardized species abundance data by dividing them by the total site abundance using the decostand function from the "vegan" package, hence generating relative abundances (in %). Then, we applied perMANOVA using the Bray-Curtis distance method and 4,999 permutations to test for differences in species composition between treatments. Similar to the environmental data, treatment, sub-basins, and the pair's identity were considered independent variables. We also rerun the same analyses with the presence/absence of fish data in order to give more weight to rare species. In both cases, we also applied a Bonferroni test on the perMANOVA output with the pairwise.adonis2 function to identify the differences among treatment groups. Finally, to graphically summarize species composition data, we performed two Non-metric Multidimensional Scaling (NMDS) analyses with abundance and presence/absence data, and we conducted correlations between them by using the "envfit" function with 4,999 permutations for both abundance and presence/absence data to investigate the relationship between NMDS axes and environmental predictors.

To determine if any species were associated with one of the treatments, we conducted an Indicator Species Analysis (ISA) using abundance and presence/absence data. The ISA is a tool that identifies species from each treatment level. We employed the "multipatt" function from the "indicspecies" package with 4999 permutations. The analysis was performed twice: first considering only a single treatment, and then analyzing pairs of treatments. The parameter ‘duleg’ was set to TRUE in the first step, allowing for the identification of species indicative of a single specific treatment.

Dam parameters

To test the effects of dam characteristics (height and age) on the environment and fish fauna, we calculated environmental dissimilarity (EnvDis) between the up and down sections of each pair, community dissimilarity using abundance data (AbunDis), and presence/absence data (PADis). We then tested simple linear models to assess the effects of dam height and age separately on the three dependent variables (EnvDis, AbunDis, and PADis).

Results

Treatment effects on stream environment

We found significant differences between dam treatments (perMANOVA: Df= 2; F= 3.37; p = 0.001), with free_flow sites separated from disconnected (Figure 1; Table 2). The first PCA axis (38% variation) was positively linked to depth, dissolved oxygen, flow rate, sand, and temperature, and negatively to conductivity and coarse litter. The second axis (16% variability) was positively linked to pH, substrate diversity, and temperature, and negatively related to elevation and depth (Figure 1). The difference in treatments was between the disconnected_down and the free_flow treatments (pairwise perMANOVA: DF = 1; F = 4.876; p =0.02), and disconnected_up and the free_flow treatments (pairwise perMANOVA: Df = 1; F = 2.716; p = 0.02), while no significant differences were found between disconnected_up - disconnected_down treatments (Supplementary Table 2).

Linear Mixed-Effect models for individual habitat metrics as response variables detected significant differences in conductivity, dissolved oxygen, and pH (Supplementary Table 3), confirming perMANOVA results. These three variables were significantly different between disconnected_down and disconnected_up, showing lower conductivity (LMM, Tukey contrasts: Estimate = 0.77 ± 0.41, z value = 3.803, p = 0.001), higher dissolved oxygen (LMM, Tukey contrast: Estimate = -0.87 ± 0.36, z value = -2.415, p = 0.03), and higher pH (LMM, Tukey contrast: Estimate = -0.14 ± 0.06, z value = -2.344, p = 0.04) downstream compared to upstream. Dissolved oxygen was also significantly different between disconnected_down and free_flow (LMM, Tukey contrast: Estimate = -2.09 ± 0.73, z value = -2.861, p = 0.01). Overall, free-flow sites exhibited high conductivity, elevated pH, along low oxygen levels compared to the other groups (Supplementary Table1); whereas disconnected_down sites had conductivity lower than 0.8 µS/cm, oxygen higher than 0.9 mg/L and pH 0.1 higher than the upstream sites (Supplementary Table 1). Although not significantly different, the temperature was slightly higher at the disconnected_down compared to disconnected_up sites (Supplementary Table 1).

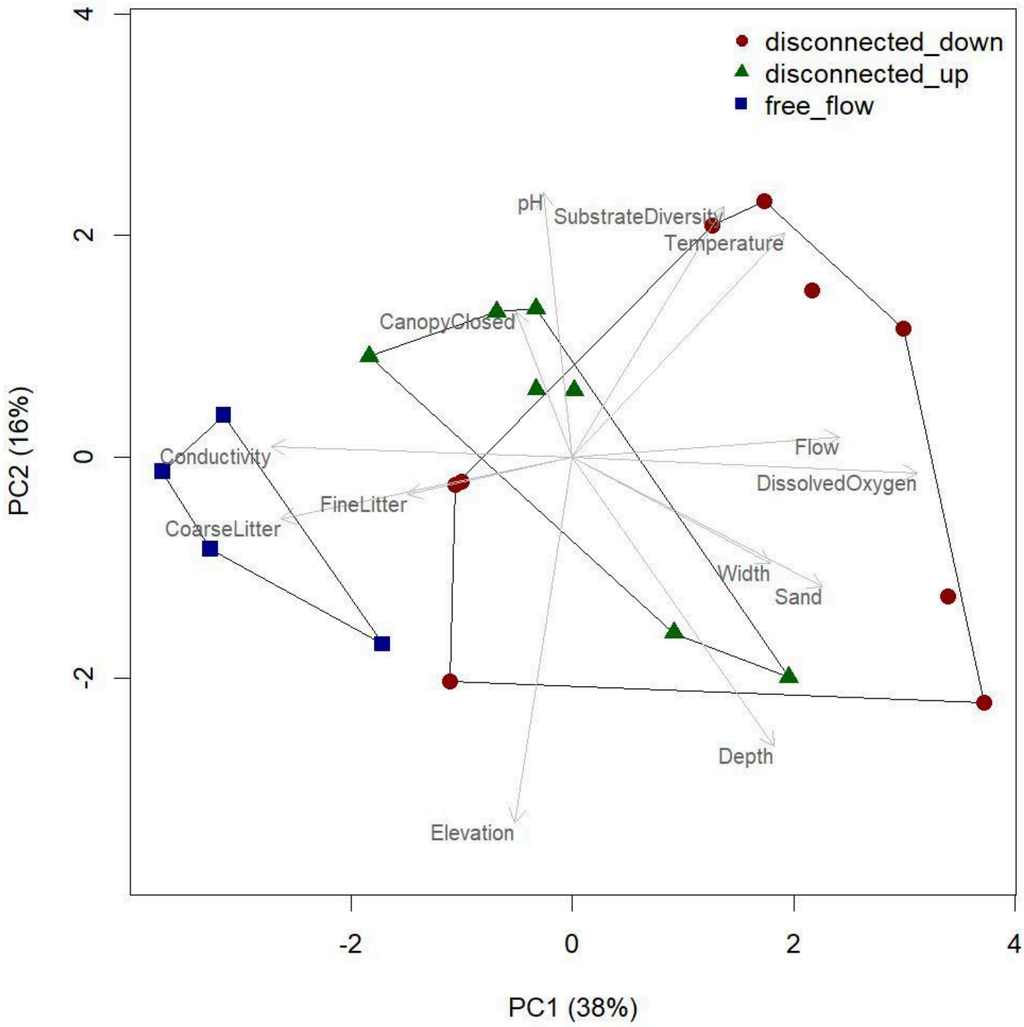

Figure 2: Principal Component Analysis (PCA) representing 53% of the environmental variation in streams of upper Xingu River basin, Mato Grosso State. Treatments are represented by symbols.

Tabela 2: Results of perMANOVA analysis showing the effects of Treatment, Stream, and Pairs on the environmental factors, and the fish species composition (both with abundance and presence/absence data) in amazonian streams from the upper Xingu River basin, Mato Grosso State. Significance level: $p < 0.05$. Treatment: Free flow, Downstream and Upstream; Stream: identity of eight streams; and Pairs: ten pairs of sampled sections. Pairwise comparisons for Treatment groups are available in Supplementary Table 2.

	Environmental			Abundance data			Presence/Absence data		
	Df	F	p	Df	F	p	Df	F	p
Treatment	2	3.37	**0.001**	2	1.86	**0.03**	2	5.77	**0.0002**
Stream	6	1.47	0.08	6	1.23	0.1	6	1.95	**0.02**
Pairs	2	1.30	0.2	2	1.65	0.06	2	3.54	**0.002**

Treatment effects on fish fauna

A total of 2,764 individuals belonging to six orders, 18 families, and 38 fish species were captured (Supplementary Table 4). The most abundant and widely distributed species were *Hyphessobrycon mutabilis* (27% of total abundance), and *Moenkhausia phaeonota* (18%). Additionally, the species *Aequidens michaeli*, *Melanorivulus megaroni*, *Pyrrhulina* cf. *australis, Gymnotus* cf. *carapo*, and *Helogenes marmoratus* were recorded in at least 65% of the sampled sites and, together, represented 21% of the total abundance. Most of these species were present in all treatments, except *P.* cf. *australis*, not found in free flow sites (see Supplementary Table 4). Of the 38 species collected, 14 were common (found in more than 45% of the sampled sites), 17 were found in up to seven sites (occurrence between 10% and 35% of the sites), and finally, seven species were rare with a single occurrence.

Disconnected_up and down sites shared 29 species, representing 76% of the sampled species, while free_flow shared 16 species with the other treatments, corresponding to 94% of their species. Although there were these small differences, we detected no significant effects in total abundance (Treatment: LRT= 1.850; p = 0.1; Stream: LRT= 11.411; p = 0.07) nor in species richness (Treatment: LRT= 0.236; p = 0.6; Stream: LRT= 5.140; p = 0.5).

Species composition with both data types differed significantly among treatments (Table 2; Figure 3). Based on abundance data, we found differences only between free_flow and disconnected_down sites (pairwise perMANOVA: Df= 1; F= 2.857; p = 0.03), and no differences between disconnected_up and down (pairwise perMANOVA: Df= 1; F= 1.629; p = 0.1) or between disconnected_up and free_flow sites (pairwise perMANOVA: Df= 1; F= 1.390; p = 0.2) (Figure 3A). The abundance of *Bryconops* sp.*, Rhinotocinclus acuen*, and *Rhinotocinclus kwarup* were positively associated with the first axis, whereas *Hypressobrycon mutabilis* was negatively associated with it. *Cetopsidium sp., Melanorivulus megaroni,* and *Pimelodella* sp. were positively associated with the second axis, whereas *Moenkhausia phaeonota* was negatively associated (Figura 3A).

Using Presence-Absence data, there were significant differences among all treatments (Table 2 and further pairwise perMANOVA): free_flow and disconnected_up (Df= 1; F= 10.087; p = 0.004), free_flow and disconnected_down (Df= 1; F= 8.533; p = 0.003), and disconnected down and up (Df= 1; F= 2.204; p = 0.04) (Figure 2B). *Bryconops* sp. and *Rhinotocinclus acuen* were positively associated with the first axis, whereas *Cetopsidium sp.*, *Melanorivulus megaroni*, *Pimelodella sp.*, and *Rhinotocinclus kwarup* were positively associated with the second axis, and *Jupiaba* cf. *anterior* was negatively related to it (Figure 2B).

The ISA was significant for several species, notably *Hypopygus lepturus* (0.04) and *Brachyhypopomus* sp. (0.03) for the disconnected_up sections, and *Pimelodella* sp. (0.03) for the disconnected_down sections based on abundance data. Using presence/absence data, *Brachyhypopomus* sp. (0.02) and *Pyrrhulina australis* (0.001) were identified as indicators of disconnected_up.

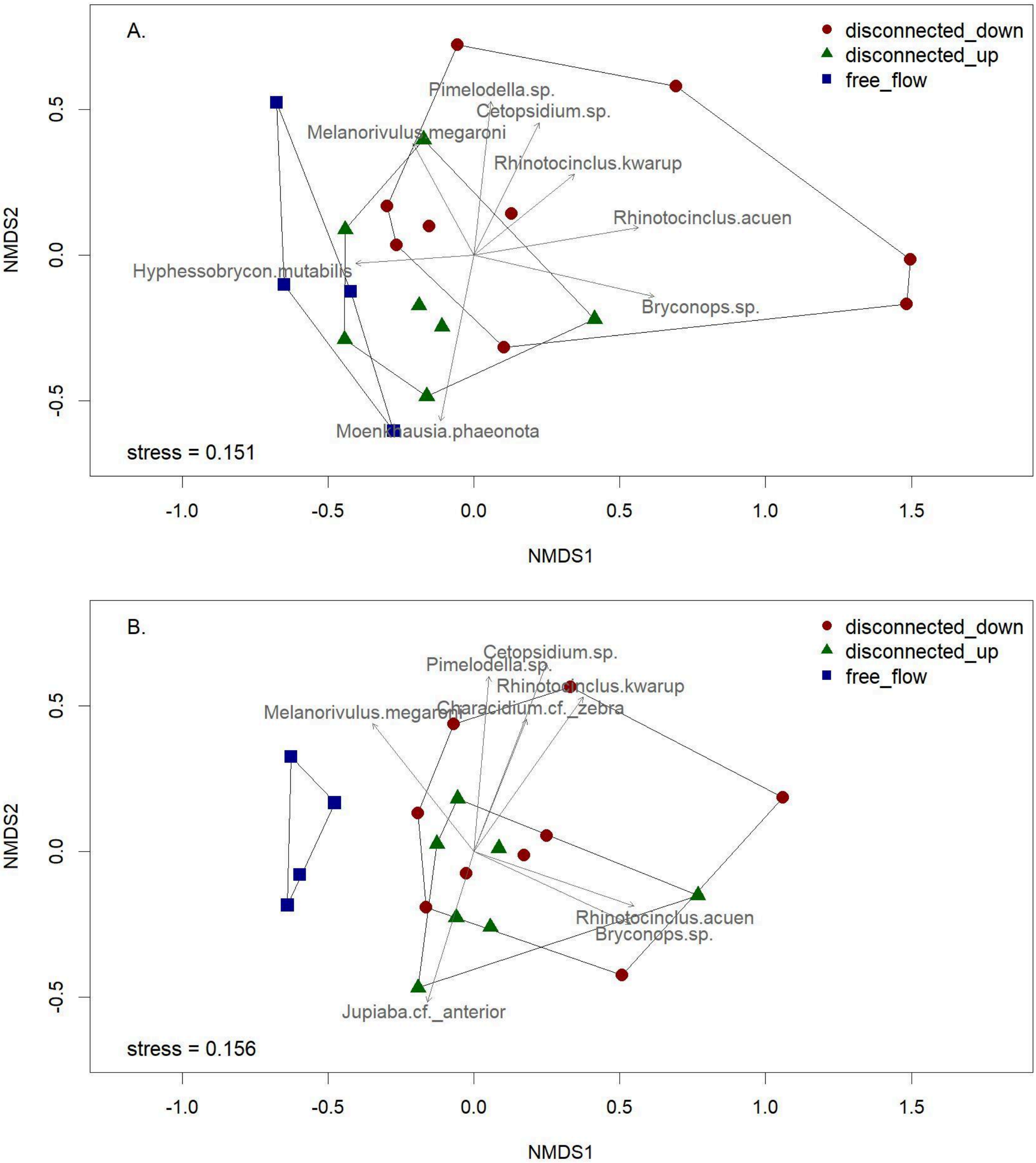

Figure 3: NMDS plot showing the distribution of sites according to a) abundance and b) presence/absence of fish species composition from streams in Upper Xingu, Mato Grosso. Treatments are represented by symbols and their sites are connected by the hull line. Based on non-parametric correlations among abundance and axis (envfit function - The function fits environmental vectors or factors onto an ordination), species are shown associated with the first or second axes.

Environment and fish fauna vs dam attributes

Linear models did not show significant effects of dam age or height on dissimilarity fish (both based on abundance and presence/absence data) nor on habitat dissimilarity (Figure 4, Supplementary Table 5).

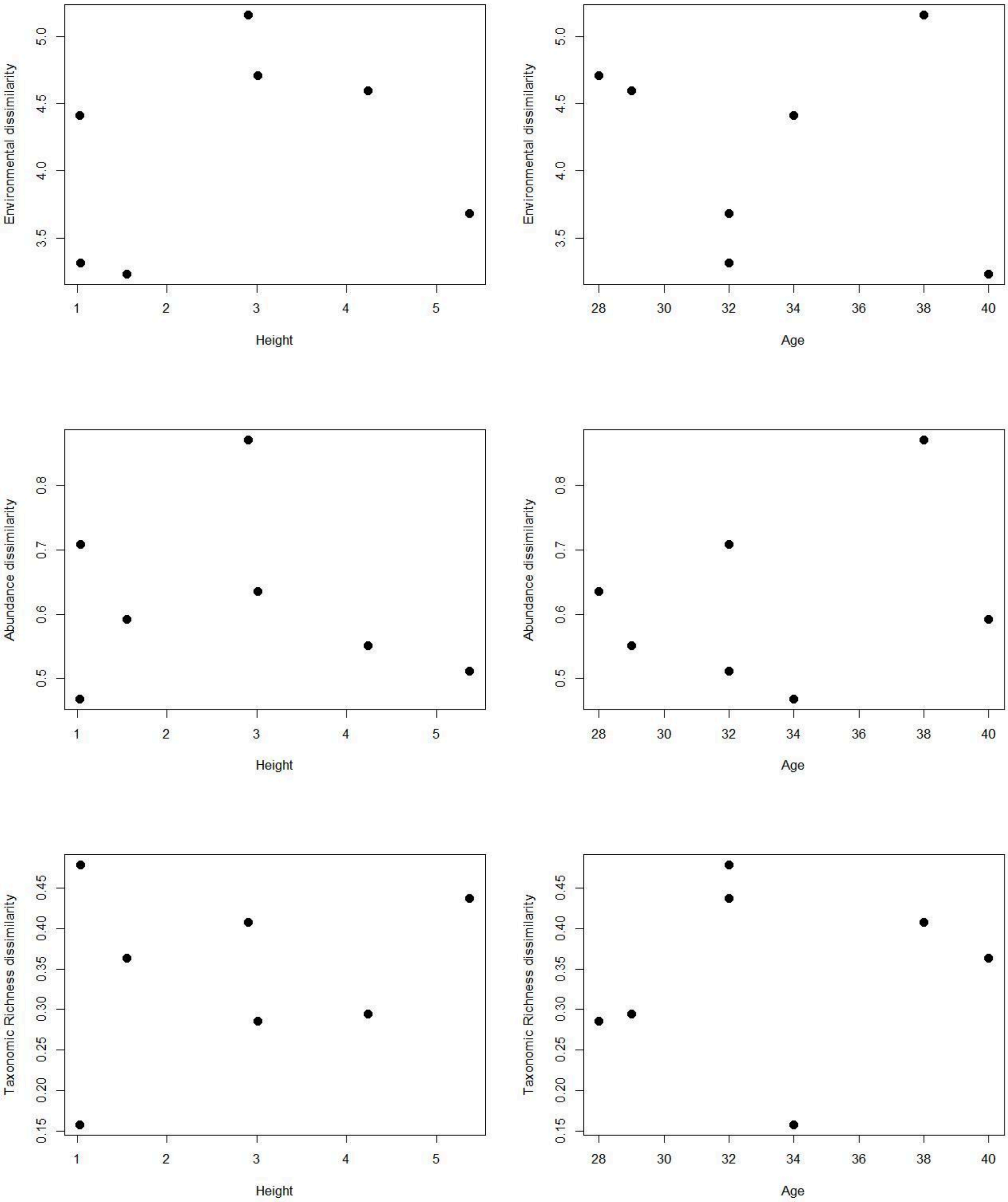

Figure 4: Relationships between environmental dissimilarity and species composition (calculated using abundance and presence/absence data) and the attributes (height and age) of the sampled dams.

Discussion

Our study stands out from others conducted in the Amazon region because it was carried out in a highly modified transition area for agriculture, characterized by a high density of barriers and the longest stretch of secondary roads opened in the Amazon. Our sampling was conducted in a paired design, both downstream and upstream, in headwater streams obstructed by small dams assessed whether two attributes (age and height) influenced the ecosystem and aquatic fauna, and included sampling in free-flowing streams as controls. Our results demonstrate that the main differences are evident in environmental characteristics and species composition between free-flowing and disconnected sites. However, we found subtle differences in chemical water condition and species composition (based on presence/absence data) among down and upstream sections, and we did not identify any effects of barrier height or age on the evaluated metrics.

Distinct water chemical conditions

Our findings suggest strong differences in environmental variables across the free-flow, disconnected upstream, and downstream segments. Although highly similar to each other, downstream reaches differ from upstream ones due to lower conductivity values, and higher levels of dissolved oxygen and pH. This outcome suggests that the dams' effects are primarily confined to water chemical attributes with negligible alterations in downstream physical habitat. The observed pattern resembles that identified by Yan et al. (2013) in segments of streams fragmented by low-head dams, where water flows over the dam wall rather than through a culvert, as in our study. Previous studies have yielded similar findings, demonstrating decreased conductivity between fragmented and free-flowing river stretches (Vasconcelos et al. 2021). However, some of our results conflict with this study, as we do detect differences in pH, for instance. This conflicting outcome likely stems from the small reservoir surface area found in our study, as the aforementioned study investigated large rivers with hydroelectric dams and extensive reservoir areas (Vasconcelos et al. 2021). Our findings suggest that the detected changes are likely attributed to water retention by the reservoirs created by the barriers and the reduction in longitudinal transport of suspended sediments, as evidenced in other studies (Jumani et al. 2020; Da Cruz et al. 2021). Thus, the presence of a small barrier in an Amazonian stream appears to impact water chemical attributes, such as conductivity and pH, regardless of watercourse size and barrier dimensions. The environmental repercussions on water chemical parameters of larger streams and rivers might be profound, given the extensive fragmentation in the region's water bodies.

A temperature increase downstream a barrier has been reported by several studies, both in tropical and temperate regions (Chandesris et al. 2019; Da Cruz et al. 2021; Zaidel et al. 2021), whereas temperature was not significantly different downstream and upstream. Forested streams in the region have an average temperature of 25°C compared to an average of 27°C0 in agricultural streams (Macedo et al. 2013). In agricultural streams having dams, water temperature takes an average of 2.74 km downstream to return to cooler temperatures due to riparian shading and the input of cooler groundwater (Macedo et al. 2013). Although our results did not show a significant temperature increase, it is important to note that thermal changes can be intense in highly fragmented areas and significantly affect downstream aquatic ecosystems and fauna worldwide (Vörösmarty et al. 2010).

Minor effects on stream fish assemblages

Our results show that total fish abundance and fish richness are similar among all treatments, and are comparable with other studies in Amazonian streams under natural (Mendonça, Magnusson, and Zuanon 2005; Espírito‑Santo et al. 2009; Dias et al. 2021) or impacted conditions (Ilha, Rosso, and Schiesari 2019). The species composition based on abundance data is similar between up- and downstream reaches. Those results suggest that the major fish assemblage pattern is not influenced by the barriers or that upstream sites are not losing species or individuals due to the stream isolation. These minor differences in species composition are supported by another study in the amazon region dealing with fragmentation caused by road crossings (Brejão, Teresa, and Gerhard 2020). Based on underwater visual sampling, the authors found similar species composition between up- and downstream reaches and distinct composition between impoundment and both up/downstream reaches despite higher fish taxonomic and functional diversity downstream dams (Brejão, Teresa, and Gerhard 2020). Stream fish species from Neotropical regions usually have restricted geographical distributions due to dispersal limitations and environmental constraints (Reis 2013; Castro and Polaz 2020) and those found here seem not to be highly impacted by the isolation effects caused by small dams.

We did find, however, a difference in species composition between up- and downstream reaches based on presence/absence data, but this result seems to be linked to colonization of downstream sites rather than small dam effects upstream. Upstream reaches had species usually sampled in first to second order streams and common in these areas (*Brachyhypopomus* sp., *Hypopygus lepturus*, and *Pyrrhulina australis*) (Ilha, Rosso, and Schiesari 2019; Schiesari et al. 2020). On the other hand, *Pimelodella* sp. was identified as an indicative of downstream reaches, and five other species were only present in downstream reaches (including *Cetopsis* sp., *Imparfinis* sp., *Myleus* sp., and *Sternopygus macrurus*; all from the same second-order stream) (Suplementar Table 4), which are mostly found in high order streams and rivers. As downstream reaches are inevitably closer to major Darro and Tanguro rivers (the main species pool source) than upstream reaches, fish composition downstream may be more distinct from upstream reaches due to the high flux of immigrants (Henriques-Silva et al. 2019; Stegmann et al. 2019). Although they did not find differences in fish species composition, Brejão et al. (2020)used proximity to a main source to explain the higher taxonomic and functional diversity in downstream sites.

It is worth noting that free-flow sites differed significantly both in environment and species composition (abundance and presence/absence data) between treatments. There is a strong link between land conversion and stream fragmentation by small dams (i.e., there are no dams in forested areas), all free-flow sites in our study are in distinct streams compared to those having barriers. Hence, this difference in terms of species composition may be due to the high beta diversity in environment and species composition among streams (Mendonça et al. 2005; Dias et al. 2021) as also evidenced by stream effects controlled in our study. This high beta diversity justifies and emphasizes the strict paired sampling adopted here and in other studies (Dias, Magnusson, and Zuanon 2010; Brejão, Teresa, and Gerhard 2020) to control for potential confounding effects when evaluating the fragmentation effects on streams and fauna. Another reason for free-flow differences compared to the other treatments is that all disconnected sites are within a matrix of cropland cover, but they all hold riparian vegetation cover. The integrity of the riparian vegetation cover was unrelated to fish richness and abundance, but strongly linked to functional diversity in the same area (Freitas et al. 2022).

Dams attributes on stream and fish assemblages

Dam age or height had no effect on habitat and fish dissimilarity between up- and downstream reaches. This suggests that stream fish and environment are not intensely altered due to the attributes of the dams or that other factors may be influencing dissimilarity between pairs. Even though the presence of small dams has only subtle effects on streams and fish as those detected here, dam age or height are presumably related to the persistence of the impact over time and to the size of the barrier. Previous studies show that the timing and the magnitude of dam effects on assemblages depend both on attributes of the fish fauna, of the watershed, and/or the dam (Arantes et al. 2019). The location of dams on the watershed and dam height directly affect fish ability to pass through barriers (Jumani et al. 2020; Zarri et al. 2022). The studied dams are positioned in the middle of the longitudinal stream course and their size are up to 5 m in height, which could be a strong barrier to dispersal. The absence of differences found here may be due to the stabilization of assemblage after three decades after dam's construction, as most changes in diversity patterns occur in the first years after impoundments (Perônico et al. 2020) and the environment and diversity metrics tend to stabilize with a new composition (Agostinho, Pelicice, and Gomes 2008; Perônico et al. 2020). Other factors such as stream size, land use, water quality, and the number of dams affect fish communities in fragmented streams (Holcomb, Nichols, and Gangloff 2016), but our study virtually controlled for most of these in the experimental design, the statistical analyses or both. Another important, though not tested, metric is the reservoir surface area, as dams of less than three meters height can flood up to one kilometer upstream (Brejão, Teresa, and Gerhard 2020), which may pose a significant barrier to fish less than 10 centimeters in size.

Perspectives and limitations

The differences in environmental parameters and species composition among the different treatments may have significant implications for the conservation and management of freshwater ecosystems fragmented by small agriculture dams and road crossings. Given that biodiversity inventories of freshwater fish are incomplete, especially in the tropics, the status of most populations is unknown in terms of geography, habitat, and taxonomy (Dudgeon et al. 2006). Additionally, at least two streams had multiple barriers, which may have created environments with new characteristics and isolation for fauna between barriers. It is also important to evaluate the area of the reservoir created by the barrier and the height of the outflow pipe, especially when it is higher than the water surface. Further research is needed to assess land use in areas adjacent to streams and the interaction between dams and deforestation.

Many landowners have implemented thousands of small impoundments, like those assessed in this study, with the expectation of minimal environmental impacts (Freitas et al. 2022). Culverts and small dams are the most common structures fragmenting aquatic environments, but together they represent only 4% and 18%, respectively, of the studies on this subject (Zarri et al. 2022). Three bills under discussion in the Brazilian legislature aim to simplify the construction of small dams for irrigation in small rivers (Azevedo-Santos et al. 2024). However, the approval of such projects may intensify negative impacts on aquatic ecosystems and biodiversity due to the lack of technical assessments and rigorous authorization processes (Azevedo-Santos et al. 2024). Despite assumptions that small dams have limited impacts, we did not evaluate the effects of multiple small barriers, or their effects in combination with intensive riparian forest degradation or other type of land use conversion. These synergistic effects may have profound impact on stream and aquatic biodiversity given the large portion of Neotropical freshwater fish species inhabiting headwaters (Tickner et al. 2020; Fróis et al. 2021).

Conclusion

Our study permitted to evaluate the effects of small dams (constructed for agriculture purposes or due to road crossing) in stream habitat and fish biodiversity in an Amazonian region of intense land use change. The higher pH, conductivity and oxygen levels downstream dams, are linked to changes reservoirs promote in water chemistry. On the other hand, only species composition based on presence/absence data differ between up- and downstream sections, and there is no indicative of species being extirpated upstream due to the damming. This minor effect on fish fauna may be due to the resilience of stream fish fauna to small damming structures or, due no significant effects of barrier age or height on the environment or species composition, the barriers may not prevent fish from dispersing from up and downstream. We emphasize the importance of preserving fragmented streams in areas of native vegetation to maintain population remnants even under high stream fragmentation levels. Furthermore, we highlight the need for further research to better understand the effects of fragmentation and develop effective conservation strategies for these vulnerable ecosystems.

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Supplementary Material

Supplementary Table 1: Comparison of environmental variables among the free_flow, downstream, and upstream treatments in streams of the Upper Xingu Region, Mato Grosso.

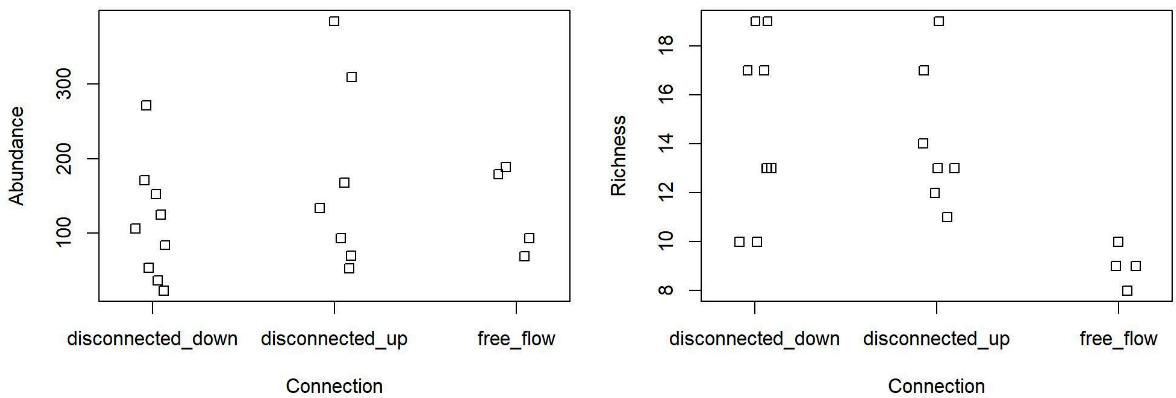
	Free_flow		Upstream		Downstream	
	Mean	sd	Mean	sd	Mean	sd
Canopy Open (%)	7.17	± 2.55	8.29	± 2.85	7.60	± 2.55
Conductivity (µS/ cm)	4.79	± 0.67	4.32	± 0.40	3.50	± 0.35
Depth (m)	0.19	± 0.1	0.30	± 0.19	0.45	± 0.19
Elevation (m)	366	± 21	350	± 13	349	± 19
Flow (m^3/s)	6.69	± 2.36	17.60	± 6.03	19.48	± 7.93
OD (mg.L)	4.60	± 1.11	5.51	± 0.55	6.40	± 0.77
pH	5.00	± 0.12	4.91	± 0.24	5.01	± 0.20
Substrate shan	1.17	± 0.15	1.42	± 0.17	1.48	± 0.24
Temperature (°C)	21.85	± 0.98	23.33	± 0.91	23.86	± 1.8
Width (m)	1.43	± 0.33	2.08	± 0.92	2.50	± 0.79
Fine litter (%)	21	± 17	16	± 12	18	± 14
Coarse litter (%)	47	± 14	24	± 18	22	± 13
Sand (%)	6	± 12	23	± 15	24	± 21

Supplementary Table 2: Results Bonferroni test on the effects among treatment pairs, streams, and pairs for environmental factors, species composition (both with abundance and presence/absence data) of stream fish in the Amazonian streams of the Upper Xingu region, Mato Grosso. Significance level: $p < 0.05$. Treatment: Free flow, Downstream, and Upstream; Stream: identity of eight streams; and Pairs: ten pairs of sampled sections.

		Down + Up			Free + Down			Free + Up		
		Df	F	p	Df	F	p	Df	F	p
Environmental	Treatment	1	1.6	0.1	1	4.9	**0.02**	1	2.7	**0.02**
	Stream	5	1.6	**0.04**	6	0.9	0.5	6	1.1	0.4
	Pairs	2	1.32	0.2	2	1.1	0.4	1	0.6	0.8
Abundance data	Treatment	1	1.6	0.1	1	2.8	**0.02**	1	1.4	0.2
	Stream	5	1.1	0.3	6	1.4	0.1	6	1.3	0.1
	Pairs	2	2.5	0.09	2	1.7	0.1	1	1.3	0.3
Presence/ Absence data	Treatment	1	2.2	**0.05**	1	8.5	**0.003**	1	10	**0.002**
	Stream	5	1.8	**0.001**	6	1.6	0.1	6	2.1	0.1
	Pairs	2	3.2	**0.001**	2	2.6	0.06	1	2.4	0.1

Supplementary Table 3: Results of the mixed linear model demonstrating the effect of treatment and stream with pairs as a random factor for the environmental variables of streams in the Upper Xingu River, Mato Grosso. Likelihood-ratio test (LRT) and Significance level: $p < 0.05$. Treatment: Free flow, Downstream, and Upstream; Stream: identity of eight streams; and Pairs: ten pairs of sampled sections.

	Treatment		Stream	
	LRT	p	LRT	p
Canopy Open (%)	0.173	0.6	2.175	0.9
Conductivity (µS/ cm)	16.788	**0.0001**	12.794	**0.04**
Coarse Litter (%)	0.132	0.7	10.751	0.09
Depth (m)	3.502	0.06	15.296	**0.01**
Dissolved Oxygen (mg.L)	8.510	**0.004**	12.602	**0.04**
Elevation (m)	0.886	0.34	17.016	**0.009**
Fine Litter (%)	0.138	0.7	10.765	0.09
Flow (m^3/s)	0.306	0.5	12.383	0.06
pH	8.103	**0.004**	26.643	**0.0001**
Sand (%)	0.032	0.8	5.483	0.4
Substrato Diversity	1.145	0.2	9.720	0.13
Temperature (°C)	1.489	0.2	5.404	0.4
Width (m)	1.318	0.2	9.095	0.1



Supplementary Figure 1: Abundance and Species Richness of Stream Fish in Upper Xingu, Mato Grosso.

Supplementary Table 4: List of species separated by hydrographic basin, Darro or Tanguro, and separated by the position of the stretch: Free_flow (Ff); upstream (Up); and downstream (Do), with four, seven, and nine stretches, respectively.

Order	Family	Specific Name	Darro Basin			Tanguro Basin		
			Ff	Up	Do	Ff	Up	Do
Characiformes	Crenuchidae	*Characidium cf. zebra*	0	5	14	0	0	2
	Erythrinidae	*Hoplerythrinus unitaeniatus*	0	4	1	0	0	0
		Hoplias malabaricus	2	4	8	2	4	2
	Serrasalmidae	*Myleus sp.*	0	0	1	0	0	0
	Lebiasinidae	*Pyrrhulina cf. australis*	0	11	31	0	26	26
	Iguanodectidae	*Bryconops sp.*	0	4	183	0	2	7
	Characidae	*Hemigrammus parana*	0	3	14	0	163	2
		Hemigrammus sp.	33	4	4	0	0	0
		Hyphessobrycon mutabilis	50	56	58	145	281	128
		Hyphessobrycon sp.	0	0	0	1	0	0
		Jupiaba cf. anterior	0	0	0	6	15	3
		Microschemobrycon elongatus	0	8	0	0	0	0
		Moenkhausia collettii	0	7	6	0	0	61
		Moenkhausia cotinho	0	0	0	1	3	3
		Moenkhausia phaeonota	44	14 4	24	84	107	81
		Moenkhausia pirauba	0	0	1	5	8	7
		Thayeria boehlkei	0	0	0	0	1	0
Gymnotiformes	Sternopygidae	*Eigenmannia cf. trilineata*	14	12	7	0	9	6
		Sternopygus macrurus	0	0	0	0	0	1
	Gymnotidae	*Gymnotus cf. carapo*	5	18	10	7	45	13
	Hypopomidae	*Brachypopomus sp.*	0	6	4	0	18	2
	Rhamphichthyida e:	*Gymnorhamphichthys rondoni*	0	27	10	0	1	32
		Hypopygus lepturus	0	10	5	0	20	6

		Steatogenys elegans	0	4	2	0	0	0
Siluriformes	Cetopsidae	*Cetopsidium sp.*	0	1	5	0	0	1
		Cetopsis sp.	0	0	1	0	0	0
		Helogenes marmoratus	9	11	26	20	3	8
	Callichthydae	*Megalechis sp.*	2	1	0	0	1	0
	Loricariidae	*Rhinotocinclus acuen*	0	13	79	0	2	7
		Rhinotocinclus kwarup	0	5	14	0	0	0
	Heptaperidae	*Imparfinis sp.*	0	0	1	0	0	1
		Pimelodella sp.	1	3	22	0	3	5
		Rhamdia quelen	1	0	0	0	0	1
Synbranchiformes	Synbranchidae	*Synbranchus marmoratus*	0	2	0	0	1	3
Cichliformes	Cichlidae	*Aequidens michaeli*	18	38	22	5	48	25
		Lugubria rosemariae	0	3	2	0	0	0
		Mesonauta acora	0	0	4	0	0	0
Cyprinodontiformes	Rivulidae	*Melanorivulus megaroni*	69	21	16	6	25	15

Supplementary Table 5: Results of dissimilarity models of pairs (downstream and upstream) for environmental variables, abundance, and presence/absence in relation to the age and height of barriers. The values of Df, F, and p are provided for each model.

	Age			Height		
	Df	F	p	Df	F	p
Environmental	1	0.257	0.2	1	0.317	0.3
Abundance	1	0.537	0.4	1	0.239	0.6
Presence/Absence	1	0.176	0.6	1	0.113	0.7

Conclusão geral

Nossos resultados mostraram que as características ambientais dos riachos estudados são semelhantes, com apenas pequenas diferenças nas variaveis químicas, condutividade, oxigênio dissolvido e pH, entre os trechos à jusante e à montante das barragens. Portanto, embora a análise conjunta das variáveis não tenha revelado divergências no espaço multidimensional, individualmente elas diferiram entre os tratamentos. Consequentemente, podemos destacar que as principais diferenças identificadas são relacionadas às características químicas da água, assim ressaltamos a importância de considerar essas variáveis, além das variáveis físicas associadas a estrutura do canal ou do habitat.

As espécies *Hyphessobrycon mutabilis* e *Moenkhausia phaeonota* foram identificadas como as mais comuns e amplamente distribuídas nos riachos estudados. Isso sugere que, apesar da fragmentação, algumas espécies adaptadas a riachos correntes persistem nos trechos de fluxo lótico dos riachos fragmentados, independentemente da posição em relação a barreira. A abundância total e riqueza de espécies entre os trechos amostrados não foram diferentes entre os tratamentos. Nós detectamos diferenças entre os tratamentos na composição das espécies de peixes feitos com dados de abundância, mas as divergências mais marcantes foram observadas entre os trechos de fluxo livre e os trechos a montante/jusante da barreira. Adicionalmente, ao considerar dados de presença/ausência, identificamos diferenças na composição entre todos os tratamentos. As espécies *Brachypopomus* sp., *Hypopygus lepturus*e *Pyrrhulina australis* foram identificadas como indicadoras de ambientes fragmentados acima de barreira e *Pimelodella* sp. como abaixo. Por fim, nossos resultados não mostraram efeitos significativos da idade ou da altura das barreiras na dissimilaridade ambiental ou na composição de espécies entre os locais amostrados acima e abaixo das barreiras, mas sugerimos que futuras pesquisas explorem mais a fundo essa relação, assim como a influência da área do lago e da altura da tubulação usada na construção das passagens de rodovias.

Desta forma, nossos resultados indicam que as pequenas barragens analisadas parecem não ter um grande efeito na fauna de peixes de riachos e não há um indicativo forte de extirpação local acima ou abaixo das barreiras. Em suma, nosso estudo ressalta que a fauna de peixes parece ser bastante resiliente ao impacto das pequenas barragens, mas reforçamos a necessidade de estudos futuros explorando a questão em cenários mais complexos entre as características das barreiras e os diferentes cenários de uso da terra, a fim de criação políticas de manejo e conservação de ecossistemas aquáticos.