INSTITUTO NACIONAL DE PESQUISAS DA AMAZÔNIA - INPA
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IMPACTO DE BARRAGENS SOBRE A AVIFAUNA EM ÁREAS SAZONALMENTE ALAGÁVEIS DA AMAZÔNIA

GUSTAVO DE MELO MARTINS

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CAMILA CHEREM RIBAS
JULIANA DA SILVA MENGER

Dissertação apresentada ao Instituto Nacional de Pesquisas da Amazônia como parte dos requisitos para obtenção do título de Mestre em Ecologia.

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Aos 18 dias do mês de Maio do ano de 2023, às 13h00min, via videoconferência, reuniu-se a Comissão Examinadora de Defesa Pública, composta pelos seguintes membros: a Drª. **Maíra Benchimol de Souza**, da Universidade Estadual de Santa Cruz – UESC, o Dr. **Thiago Orsi Laranjeiras**, do Instituto Chico Mendes de Conservação da Biodiversidade – ICMBio e a Draª. **Jussara Santos Dayrell**, do Instituto Nacional de Pesquisas da Amazônia – INPA, tendo como suplentes a Drª. Luisa Maria Diele Viegas Costa Silva, da Universidade Federal da Bahia – UFBA e o Dr. Paulo Estefano Dineli Bobrowiec, do Instituto Nacional de Pesquisas da Amazônia – INPA, sob a presidência da orientadora, a fim de proceder a arguição pública do trabalho de **DISSERTAÇÃO DE MESTRADO** de **GUSTAVO DE MELO MARTINS**, intitulado: **"IMPACTO DE BARRAGENS SOBRE A AVIFAUNA EM ÁREAS SAZONALMENTE ALAGÁVEIS DA AMAZÔNIA"**, orientado pela Drª. Camila Cherem Ribas, do Instituto Nacional de Pesquisas da Amazônia – INPA e co-orientado pela Drª. Juliana da Silva Menger.

Após a exposição, o discente foi arguido oralmente pelos membros da Comissão Examinadora, tendo recebido o conceito final:

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Nada mais havendo, a presente ata foi lida, lavrada e assinada pelos membros da Comissão Examinadora.

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DR. THIAGO ORSI LARANJEIRAS ______________________

DRA. JUSSARA SANTOS DAYRELL ______________________

DRA. LUISA M. DIELE VIEGAS COSTA SILVA ______________________

DR. PAULO ESTEFANO DINELI BOBROWIEC ______________________

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Sinopse:

Estudou-se o impacto das hidrelétricas de Belo Monte e Santo Antônio sobre as comunidades de aves nos ambientes alagáveis da Amazônia. Aspectos como diversidade filogenética, funcional e taxonômica foram avaliados.

Palavras-chave: Amazônia, diversidade de aves, igapó, métricas de diversidade, várzea.

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“Sempre fui fascinado por diversidade biológica e imaginava ter uma vida cheia de aventuras científicas. A Amazônia era esse mundo selvagem inacreditável e tropical. Era como se eu tivesse morrido e chegado ao paraíso.” - Thomas Lovejoy

“Prezo insetos mais que aviões.
Prezo a velocidade
das tartarugas
mais que a dos mísseis.
Tenho em mim
esse atraso de nascença.
Eu fui aparelhado
para gostar de passarinhos.
Tenho abundância
de ser feliz por isso.
Meu quintal
É maior do que o mundo.”
- Manoel de Barros

RESUMO

A Amazônia é um bioma mega diverso devido, em grande parte, à sua heterogeneidade ambiental. As planícies aluviais da Amazônia têm uma fisionomia particular, variando em estrutura, composição florística e características ecológicas, cada uma associada ao tipo de água e geomorfologia dos rios que as formam. Estas planícies aluviais sustentam vários outros ecossistemas amazônicos e uma avifauna rica, especializada, e geograficamente estruturada. As planícies aluviais, contudo, foram diretamente afetadas pela construção de grandes barragens hidrelétricas, mas estes impactos são pouco compreendidos e mal quantificados. O objetivo desse estudo é compreender como a inundação permanente causada pelas grandes barragens afeta a diversidade taxonômica, funcional e filogenética de aves em dois sistemas distintos de planícies aluviais. Um total de 35 localidades foram amostradas nos rios Madeira e Xingu, abrangendo áreas controle e áreas impactadas pelas barragens hidrelétricas de Santo Antônio e Belo Monte, respectivamente. Utilizando monitoramento acústico passivo (PAM), foram registradas 202 espécies de aves no total e, embora diferenças na riqueza de espécies não tenha sido detectada, valores mais elevados para as diversidades filogenéticas e funcionais foram encontrados para as áreas impactadas. Também houve uma elevada substituição de espécies entre bacias hidrográficas distintas e entre áreas impactadas e áreas controle. As áreas não perturbadas apresentaram comunidade de aves filogeneticamente mais agrupadas e funcionalmente mais semelhantes do que comunidades de locais impactados. Foram observados padrões significativos de variação espacial na diversidade local e estruturação regional da avifauna das planícies de inundação amazônicas. Mais importante ainda, este estudo mostra que medidas de diversidade absoluta por si só não caracterizam os impactos causados por barragens fluviais na biodiversidade das planícies aluviais da Amazônia. Por conseguinte, o monitoramento dos impactos ambientais de barragens deve ser focado nos ambientes sazonalmente alagáveis e padronizado para incluir medidas de mudança na composição taxonômica, filogenética e funcional das comunidades biológicas desses ambientes.

ABSTRACT

Amazonia is a mega diverse biome due largely to its environmental heterogeneity. Amazonian floodplains have a particular physiognomy, varying in structure, floristic composition, and ecological characteristics, each associated with the type of water and geomorphology of the rivers that form them. These floodplains sustain several other Amazonian habitats and a rich, specialized, and geographically structured avifauna. Floodplains, however, have been directly impacted by the construction of large hydropower dams, but these impacts are little understood and poorly quantified. My goal was to understand how the permanent flooding caused by large dams affects taxonomic, functional and phylogenetic diversity in two distinct floodplain systems. Using passive acoustic monitoring (PAM), I sampled 35 sites in the Madeira and Xingu rivers, both undisturbed sites and sites affected by the Santo Antônio and Belo Monte hydroelectric dams, respectively. I recorded 202 species in total and although I did not find differences in species richness, I found higher values for phylogenetic and functional diversities in impacted areas. I also found high species turnover between distinct river basins and between impacted and control areas. Undisturbed areas were phylogenetically clustered and functionally more similar than impacted sites. I documented significant patterns of spatial variation in local diversity and regional assemblage structure of floodplain avifauna. Most importantly, we show that absolute diversity measures alone do not characterize the impacts caused by river damming on the biodiversity of Amazonian floodplains. Therefore, monitoring dams' environmental impacts should be standardized to include additional measures of community change and focus on seasonally flooded environments.

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INTRODUÇÃO

Devido às grandes proporções amazônicas, seja em relação às suas florestas contínuas ou bacia hidrográfica, a biodiversidade encontrada no bioma é a mais alta na região tropical. A heterogeneidade ambiental da Amazônia atua nos padrões de composição (Pomara et al. 2014), bem como os grandes rios funcionam como fatores limitantes na distribuição de espécies, principalmente aquelas comumente vistas nas florestas de terra-firme (Cracraft et al. 1985). Além disso, como grandes áreas da bacia amazônica são planas e a rede de rios drena lentamente o excesso de água da chuva, as planícies fluviais ficam alagadas ou inundadas durante a estação chuvosa (Junk et al. 2011). Essas áreas aluviais são formadas ao longo dos rios e abrigam uma fauna e flora adaptada e especializada ao pulso de inundação. Assim como espécies de terra-firme, as espécies encontradas nas áreas alagáveis apresentam padrões comuns de distribuição, e um dos fatores determinantes de sua ocorrência é a cor das águas (Cohn-Haft et al. 2007, Laranjeiras et al. 2020). Os ambientes alagáveis de rios da Amazônia apresentam uma grande variedade de cenários ambientais, dependendo da geologia e geomorfologia das bacias de onde os rios são originários (Wittmann et al. 2022). Sendo assim, a composição florística varia ao longo dos ambientes (Junk et al. 2011, 2015) e, consequentemente, a fauna associada à vegetação (Stotz et al. 1996).

As áreas alagáveis na Amazônia se formam devido à alta pluviosidade anual, que é distribuída de forma heterogênea entre as estações chuvosa e seca, e estima-se que 14% da bacia amazônica apresenta um período de alagamento ao longo do ano (L. L. Hess et al., 2015). As características pluviométricas e hídricas da bacia têm atraído projetos de implementação de usinas hidroelétricas, justificados pela crescente demanda por energia (Latrubesse et al., 2017). O número de usinas hidroelétricas operando na região amazônica tem crescido rapidamente nos últimos anos (Lees et al., 2016). Ainda assim, a região é considerada uma das principais para implementação de futuros projetos (Almeida et al., 2019; Latrubesse et al., 2017). A demanda por hidrelétricas contrasta com o atual conhecimento acerca das comunidades biológicas, as quais são diretamente impactadas por esses projetos (Latrubesse et al., 2020). Mudanças no transporte de sedimentos, fluxo de nutrientes, pulso de inundação e vazão são consequências da implementação de barragens que promovem impacto significante na biota aquática e aluvial (Forsberg et al., 2017; Latrubesse et al., 2017, 2020; Lees et al., 2016).

Dentre as barragens recentemente construídas e mais impactantes da Amazônia brasileira, destacam-se as Hidrelétricas de Santo Antônio e Jirau no Rio Madeira (água branca) e a megabarragem de Belo Monte, no Rio Xingu (água clara) (Cochrane et al., 2017; Latrubesse et al., 2017). A bacia do Rio Madeira drena uma área de 1380 km², o que corresponde a aproximadamente 20% da bacia Amazônica (Goulding et al., 2003). É o maior afluente do Rio Amazonas com vazão média de 32.000 m³/ s/ ano, tem 3.400 km de extensão e largura média de cerca de 2,4 km (Cochrane et al., 2017; Latrubesse et al., 2005). Santo Antônio e Jirau estão em operação desde 2012; inundando uma área de 870 km², e modificando a paisagem permanentemente (Cochrane et al., 2017). O Rio Xingu representa cerca de 7% da bacia hidrográfica da Amazônia, cobrindo uma área de aproximadamente 511.000 Km2 (Instituto Socioambiental, 2006). Além de ser um dos maiores rios de águas claras que drena os platôs cristalinos e planícies sedimentares do Escudo Brasileiro, é responsável por cerca de 5% da vazão do rio Amazonas (Oliveira, 2017). O Rio Xingu foi recentemente afetado pela construção da megabarragem de Belo Monte (Latrubesse et al., 2017). As obras da hidrelétrica iniciaram em junho de 2011 e apenas no ano de 2016 passou a gerar energia, alcançando mais de 11.000 MW/hora de potência (de Araújo et al., 2019; Lees et al., 2016).

Ambientes alagáveis da Amazônia abrigam uma avifauna rica, típica e especializada, com áreas de endemismo e geograficamente variável (Cohn-Haft et al., 2007; Laranjeiras et al., 2019; Remsen Jr & Parker III, 1983; Stotz et al., 1996). Muitas espécies de aves ocorrem somente em áreas alagadas por rios de água preta e clara (igapós) enquanto outras são especializadas em áreas alagadas por rios de água branca (várzeas) (Cohn-Haft et al., 2007; Laranjeiras et al., 2019). Thom et al. (2020) destacaram a existência de diversidade críptica em aves exclusivas das áreas alagáveis amazônicas, e Remsen Jr & Parker III (1983) levantaram que 15% das espécies das aves não-aquáticas da Amazônia são restritas à ambientes sazonalmente alagados. Há também registros de que aproximadamente 25% de todas as espécies de aves brasileiras ocorrem em várzeas, demostrando a importância desses ambientes para a conservação (Cohn-Haft et al., 2007).

Mesmo com a alta especialização da avifauna de áreas alagáveis da Amazônia, o conhecimento acerca dessas aves ainda apresenta lacunas consideráveis já que a maioria dos estudos, principalmente sobre a distribuição das espécies, foram realizados majoritariamente para as florestas de terra firme (Cohn-Haft et al., 2007; Thom et al., 2018). Tendo em vista os planos de construção de barragens, hidrovias e usinas hidrelétricas com potencial impactante para a região Amazônica (De Francesco & Carneirom, 2015), muitas espécies ou linhagens únicas podem ser extintas antes mesmo que conheçamos a sua biologia, ecologia e distribuição. Entender como a diversidade e a composição de espécies é distribuída em regiões naturalmente alagáveis é fundamental como linha de base para avaliar como o alagamento permanente dessas regiões impacta sua biota.

Avaliar a resposta e estabilidade de ecossistemas a perturbações antrópicas tem sido uma área de interesse para o estudo de comunidades sob diferentes perspectivas (Hidasi-Neto et al. 2012;

Schmera et al. 2012; Mori et al. 2013). É possível compreender os processos que determinam o funcionamento e organização das comunidades após perturbações ambientais (Petchey et al. 2007). Além disso, o estudo das consequências das perturbações antrópicas na história evolutiva das comunidades pode ajudar a entender como elas respondem a esse tipo de impacto (Faith 1992; Swenson et al. 2011).

Por muito tempo, as investigações sobre a distribuição da biodiversidade se concentraram na diversidade taxonômica, com ênfase no número de espécies (ou seja, riqueza) e na abundância de indivíduos (Magurran, 2004). Entretanto, essa abordagem não consegue capturar adequadamente a amplitude da variação da biodiversidade, mesmo ao nível de espécies (Ricotta, 2005). As características ecológicas das espécies variam, expressando diversas respostas, interações e funções ecológicas (Tilman, 2001). Essas variações ecológicas resultam de processos variados de adaptação e diferenciação ao longo da história evolutiva das espécies. Portanto, avaliar a diversidade sob as perspectivas funcionais e evolutivas permite entender uma parcela maior da multiplicidade de processos envolvidos na geração e manutenção dos padrões de diversidade (Pavoine e Bonsall, 2011; Stevens e Tello, 2014), bem como seu papel nos processos ecossistêmicos (Hooper et al., 2005). Para investigar o efeito das barragens sobre a biodiversidade de aves, incluindo riqueza de espécies, diversidade funcional e filogenética, é possível quantificar as diferenças entre a diversidade observada em áreas de impactadas pelo alagamento das barragens e áreas naturais (diversidade alfa), e pela diferença na composição entre essas assembleias (diversidade beta).

Para ter uma ampla visão de como as barragens afetam a avifauna especializada em áreas sazonalmente alagáveis da Amazônia, é preciso avaliar não somente a diversidade taxonômica, mas também a diversidade funcional e a filogenética, quantificando as diferenças entre as diversidades observadas em sítios impactados pelas barragens e em sítios onde o alagamento ocorre sazonalmente (diversidade alfa), bem como quantificando as mudanças composicionais entre sítios (diversidade beta).

OBJETIVO GERAL

O objetivo do presente estudo foi avaliar o impacto das barragens sobre as comunidades de aves que habitam os ambientes naturalmente alagáveis amazônicos. A pergunta cerne deste trabalho foi: com as instalações dessas barragens, houve um impacto sobre a comunidade de aves especialistas de várzeas e igapós? A hipótese para esta pergunta é de que as comunidades de aves seriam diferentes nas áreas diretamente afetadas pelas barragens. Para isso, utilizei diferentes métricas de diversidade, incluindo a taxonômica, a funcional e a filogenética, caracterizando a avifauna de várzea e igapó.

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CAPÍTULO I

Martins, G.M.; Menger, J.; Melo, T. N. & Ribas, C.C. 2023. Using multiple facets of diversity to understand the impact of dams on bird communities in Amazonian floodplains. Manuscrito em preparação para *Biotropica*.

USING MULTIPLE FACETS OF DIVERSITY TO UNDERSTAND THE IMPACT OF DAMS ON BIRD COMMUNITIES IN AMAZONIAN FLOODPLAINS

Gustavo de Melo Martins¹, Juliana da Silva Menger[2], Tomaz Nascimento de Melo[3], Camila Cherem Ribas[1,2]

1 - Programa de Pós-Graduação em Ecologia, Instituto Nacional de Pesquisas da Amazônia, Manaus, AM, Brazil

2 - Coordenação de Biodiversidade, Instituto Nacional de Pesquisas da Amazônia, Manaus, AM, Brazil

3 – Rainforest Connection, Science Department, San Juan, Puerto Rico, USA

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Introduction

The floodplains of Amazonian rivers sustain biotic communities adapted to the annual flooding cycle (Ducke & Black, 1954; Junk et al., 2011; Piedade et al., 2013). It is estimated that 14% of the Amazon basin is partially flooded for a period of time throughout the year (Hess et al., 2015). The vegetation at Amazonian floodplains has a particular physiognomy, varying in structure, floristic composition, and ecological characteristics, driven by the type of water and geomorphology of the rivers that run through them (Junk et al., 2011; 2015).

The high rainfall and water availability in the Amazon basin have attracted projects to implement hydropower dams, justified by the growing demand for energy (Almeida et al., 2019; Latrubesse et al., 2017) (Figure 1). The number of hydropower dams operating in the Amazon region has grown rapidly , but the environmental impacts of reservoirs flooding large areas of forest has been greatly criticized (Flecker et al., 2022; Lees et al., 2016) (Figure 1). In an attempt to deal with this problem, run-of-the-river projects have been implemented in Amazonia. These hydropower complexes do not store large quantities of water in a reservoir, but instead they maintain large stretches of the river floodplains permanently flooded, disrupting the natural flooding cycle. However, the demand for run-of-the-river hydropower dams in Amazonia contrasts with the current knowledge about biological communities associated with seasonally flooded habitats, which are directly impacted by these projects (Latrubesse et al., 2020; Melo et al., 2021). These impacts include changes in sediment transport, nutrient flux, flood pulse, and river discharge, and affect the habitats, structure, connectivity, and functioning of seasonally flooded ecosystems, which in turn, affect biodiversity profoundly (Assahira et al., 2017; Bobrowiec, et al., 2021; Forsberg et al., 2017; Latrubesse et al., 2017, 2020; Lees et al., 2016).

Dams are a major infrastructure development project that can have a significant impact on biodiversity. Current levels of river fragmentation are highest in the United States, Europe, South Africa, India, and China. On the other hand, future dams are expected to fragment rivers even more, especially in the tropics, where connectivity could decrease by 20-40% for species in the Amazon, Niger, Congo, Salween, and Mekong basins (Barbarossa et al., 2020). They can fragment habitats, disrupt species movements, and alter hydrological regimes. Impacts of dams on biodiversity may include taxonomic changes (species loss or replacements), and also (Dayrell et al., 2021, McAllister et al., 2001, Melo et al., 2021) phylogenetic and functional changes, usually due to a substitution of specialists by generalist species (Devictor et al., 2008). Dams can have effects on vertebrate richness, especially birds and mammals (Bohada-Murillo et al., 2021), and small run-of-river dams may affect benthic diatom diversity and composition (Wang et al., 2022).

The impacts of dams on seasonally flooded habitats, however, are still vaguely documented (Bobrowiec et al., 2021; Dayrell et al., 2021; Latrubesse et al., 2020), as floodplains have been neglected in Environmental Impact Assessments (EIA) and in mitigation strategies (Cochrane et al. 2017, Li et al. 2020). The claims that run-of-the-river dams have lower environmental impacts and lower greenhouse gas emissions due to their smaller reservoirs, have been disputed recently (Latrubesse et al. 2017, Bertassoli Jr et al., 2021) based on the facts that these projects directly impact large areas of seasonally flooded environments (Latrubesse et al. 2017, Cochrane et al. 2017, Li et al. 2020), affecting the specialized biota and the people dependent on river productivity (Zuanon et al., 2019), and that they generate high greenhouse gas emissions (Bertassoli Jr et al., 2021).

Amazonian seasonally flooded environments harbor a rich, specialized, and geographically structured avifauna (Cohn-Haft et al., 2007; Laranjeiras et al., 2019; Remsen Jr & Parker III, 1983; Stotz et al., 1996; Thom et al., 2020). While some bird species occur only in forests flooded by black- and clear-water rivers (locally called igapós), others are specialized in forests flooded by white-water rivers (called várzeas) (Cohn-Haft et al., 2007; Laranjeiras et al., 2019). Even though Remsen and Parker III (1983) estimated that river-created habitats harbor 15% of all non-aquatic Amazonian birds, Thom et al. (2020) showed that cryptic diversity still exists and thus diversity in these habitats is likely underestimated. Given the ongoing impacts of existing dams and plans to build additional ones (De Francesco & Carneirom, 2015), many unique species or lineages may become extinct even before we know their identity, ecology, and distribution. Seasonal floodplain forests display varying structural, floristic, and ecological characteristics that are closely linked to different river types (Laranjeiras et al., 2019). Várzeas, igapós and river islands are essential for the maintenance of restricted functional groups, which are not adapted to other types of environments (Abreu et al., 2020). Disturbances on the flooding regime of these systems can affect population sizes and genetic diversity, contributing to local extinctions and population fragmentation (Pezzutti et al., 2018).

The three largest run-of-the-river hydropower dams in Amazonia are Santo Antônio and Jirau on the Madeira River and the Belo Monte complex on the Xingu river (Cochrane et al., 2017; Latrubesse et al., 2017). The Madeira River is the largest tributary of the Amazonas river and its basin drains an area in western Amazonia and base of the Andes that corresponds to approximately 20% of the Amazon basin (Cochrane et al., 2017; Goulding et al., 2003; Latrubesse et al., 2005). Santo Antônio and Jirau have been in operation since 2012, permanently flooding an area of 870 km² (Cochrane et al., 2017).

The Xingu River basin represents one of the largest clear-water rivers draining the crystalline plateaus and sedimentary plains of the Brazilian Shield. It is responsible for about 5% of the Amazon river's discharge (Oliveira, 2017). The construction of the Belo Monte hydropower complex began in June 2011 and its operation started in 2016 (de Araújo et al., 2019; Lees et al., 2016). The monitoring programs implemented by the consortium that operates the Belo Monte Plant have generally failed to detect significant environmental changes (Pezzutti et al., 2018), despite the major change in the landscape brought about by the damming of the Xingu river, the construction of an artificial channel, and the detour of much of the river's water into this channel. The impacts of the Belo Monte complex, thus, include both flooding above the first dam (Pimental) and drought along a 130 km stretch of the river between the two dams (Pimental and Belo Monte) which has reduced flow and is subjected to artificially controlled hydrograms that deeply affect the ecosystems and peoples living in the region (Zuanon et al., 2019).

Understanding how species diversity and composition are distributed in seasonally flooded habitats is a fundamental baseline for assessing the impacts of disruptions of the flooding cycle on biodiversity. Characterizing the avifauna of Amazonian seasonally flooded environments and assessing their response to the impacts of dam construction is of utmost importance for their conservation (Liang et al., 2019; Sax & Gaines, 2003). Thus, in this study we characterize the local diversity and composition of bird assemblages in undisturbed seasonally flooded areas and in areas that became permanently flooded due to dam construction at the Madeira and Xingu rivers, which sustain distinct types of seasonally flooded environments. Our goal is to understand how permanent flooding caused by large hydroelectric dams affects taxonomic, functional, and phylogenetic diversity in these different systems. We expect a higher diversity of birds in the Madeira river, due to its greater productivity in relation to the Xingu river. Moreover, we expect to find differences in the avifauna that occurs in undisturbed seasonally flooded environments on both rivers, with a higher number of floodplain specialized birds, and a substitution of specialists by generalist species in impacted sites. As permanent flooding creates new habitats that may be occupied by different lineages with divergent characteristics (e.g., aquatic species), we predict that functional and phylogenetic diversities will be higher in impacted sites.

Methodology

Study area

The flooded habitats along the Madeira and Xingu rivers are characterized by the contrasting fertility of their floodplains (Wittmann et al., 2022). The Madeira River is a sediment- and nutrient-rich white-water river and the vegetation along its floodplain is known as várzea, while Xingu is a sediment- and nutrient-poor clear water river, with floodplains covered by igapó vegetation (Junk et al., 2011; Prance, 1980). During the dry season, the exposed areas are colonized by early successional vegetation in várzeas due to the elevated fertility (Wittmann et al., 2022). However, this pattern is not observed in igapós, which have nutrient-poor soils when compared to várzeas. This heterogeneity in the vegetation results in a different composition of bird communities, associated with seasonally flooded habitats in these two basins.

In both Madeira and Xingu floodplains we sampled undisturbed sites and sites impacted the dams. In the Madeira river there are two sequential dams, Jirau and Santo Antônio, 120 km apart. In the Xingu River the Belo Monte complex also includes two dams, Pimental and Belo Monte, with a large stretch of rapids (Volta Grande do Xingu) between them. The impacted sites we studied have been permanently flooded by the Santo Antônio (8°48'3.6"S, 63°57'0"W) dam in the Madeira river and by the Pimental dam in the Xingu (3°28'30"S, 51°57'28.8"W) river (Figure 1). We define undisturbed sites as old-grown vegetation on sites that are still seasonally flooded, whereas impacted sites were seasonally flooded before the dams were built, but are now permanently flooded due to dam construction. Permanent flooding of impacted sites occurred since 2012 in the Madeira river and since 2016 in the Xingu river (Li et al., 2020; Latrubesse et al. 2017, Bertassoli Jr et al., 2021; Kalacska et al., 2020).

Sampling design

We sampled a total of 35 sites in the Madeira and Xingu rivers (Table S1, Figure 1). All sampling sites were at least 400 m apart from each other, to minimize sample independence (Campos-Cerqueira et al., 2020). We used a space-for-time approach (Blois et al., 2013), assuming that undisturbed sites in the same floodplain system sustain an avifauna similar to the avifauna that was present at currently permanently flooded sites before the dams were built (Cohn-Haft et al., 2007; Melo et al., 2021; Remsen Jr & Parker III, 1983; Stotz et al., 1996).

Among the 19 sites on the Madeira River (Melo et al., 2021), 10 were located in undisturbed seasonally flooded habitats and nine were located in areas permanently flooded by the Santo Antônio dam (Figure 1A). Similarly, among the 16 sites on the Xingu River (Figure 1B), 11 were at undisturbed seasonally flooded habitats, while six were located in areas permanently flooded by the Pimental dam. Sites were selected along the floodplains, both on riverbanks and on river islands.

Bird sampling

Avifauna was characterized through passive acoustic monitoring (Figure S1). This approach allows systematic sampling, improving the capacity to assess large-scale ecological processes (Sugai & Llusia, 2019). At each sampling site, one recorder was set up to record one minute every 10 minutes during 14.14 ± 5.21 consecutive days, as described by Melo et al. (2021), resulting in a soundscape of 47244 minutes for the Madeira river and 26516 minutes for the Xingu river. Recordings were obtained during the dry season (Figure S2) for both river systems (September and October 2017, on the Madeira and Xingu rivers, respectively), five years after dam construction for the Madeira River, and almost two years after dam construction for the Xingu River. All recordings were deposited in the RFCx-Arbimon platform, under the public projects “Birds of Xingu Flooded Habitats” and “Birds of Madeira Flooded Habitats”.

Soundscapes sampling

We sampled the obtained soundscapes by using a playlist with species' song models. We used expert knowledge and data from citizen-science platforms such as WikiAves and eBird, to build a comprehensive list of birds that may occur at Amazonian seasonally flooded habitats during the dry season (i.e. terrestrial species that can forage in floodplains, species that occupy environments that do not usually flood and even waterfowl that use these environments as perches and nest sites). The list included 280 bird species and was used to create our species song models. Then, we applied the Arbimon Pattern Matching feature, a tool to detect correlated signals between recordings and models, to our data. This correlation is calculated based on the time-frequency domain (LeBien et al. 2020). We selected the lowest threshold possible (i.e. 0.1), which yields a high number of matches, including several false positives (LeBien et al., 2020) and reducing the probability of false negatives. To validate the results, we confirmed matches through visual and auditive checks. We follow species nomenclature adopted by the Handbook of the Birds of the World (2022).

Data analyses

Considering the environmental differences between habitats seasonally flooded by white and black waters, we expected bird assemblages of Madeira and Xingu rivers to be distinct. Therefore, we calculated all diversity metrics for undisturbed sites and for impacted sites at each river separately and tested for differences between them. All analyses were performed using R software (Core Development Team 2016).

Alpha diversity

We analyzed the data in three separate sets (tab. S3): 1) “all species” (202 spp); 2) “without aquatic species” (182 spp), excluding species strictly associated with aquatic habitats; and 3) “floodplain specialists” (46 spp), which included only the species that are exclusively found in seasonally flooded habitats.

We calculated taxonomic diversity as the mean number of species per river and impact status (undisturbed vs. impacted), using the specnumber function of the VEGAN package (Oksanen et al., 2022). We used Jackknife, Chao, and Bootstrap estimators to estimate the total number of species per site using the BIODIVERSITY package (Kindt & Coe 2005).

We generated a phylogenetic tree by pruning the global phylogeny of all extant bird species (Jetz et al., 2012) to our species pool. We used the TreeAnnotator program in the package BEAST (Bouckaert et al. 2019) to derive the maximum clade credibility (MCC) tree from 1000 randomly selected trees provided by Jetz et al. (2012) available at http://www.birdtree.org. We estimated the MCC tree following a 10% burn in (i.e. the first 100 sampled trees were removed), with a posterior probability threshold of 0.5 and median node heights. The MCC tree is the best-supported tree from our sample of 1000 alternative trees. We then calculated Phylogenetic Diversity (PD) (Faith, 1992), Mean Nearest Taxon Distance (MNTD) and Mean Pairwise Distance (MPD) (Webb, 2000). We used the Standard Effect Size (SES) to observe the null distribution from the observed values of phylogenetic diversity (Swenson, 2014) using the PICANTE package (Kembel et al., 2010).

To calculate functional diversity, we selected nine bird traits or information related to the natural history of the species and their function on the environment. We compiled published data for each species reporting body mass, diet, hand-wind-index (HWI, a proxy for dispersion capability), range size, association with islands, preferred habitat type, territoriality, sensitivity to disturbances, and primary lifestyle (Table S2). We compiled these data from Sheard et al. (2020), Stotz et al. (1996), and Tobias et al. (2022) to build our trait dataset. We used continuous and categorical traits that were not correlated to each other (r < 0.19 or less). As such, we removed range size from the functional diversity analyses. We independently analyzed range size, HWI, and sensitivity to understand the overall pattern of bird occurrence in relation to the species' average area of occupancy, average proxy for dispersal ability, and sensitivity to disturbance for each impact status, respectively. For the other eight traits we computed multi-trait dissimilarity using the GAWDIS package (de Bello et al., 2021). The functional space was then represented by a principal coordinate analysis (PCoA), which we used to calculate functional richness and functional dispersion. Functional Richness (FRic) was calculated as the functional volume occupied by the species pool (Kuebbing et al., 2018; Mouchet et al., 2010) and the Functional Dispersion (FDis), as representing the mean distance of a species to the centroid of all species in the multidimensional trait space (Laliberté & Legendre, 2010). We calculated these two metrics using the mFD package (Magneville et al., 2022). We used metrics that are widely used in diversity studies, but which can be correlated with species richness (e.g. PD, FRic). Therefore, we added other metrics that are complementary and not correlated with species richness.

To test the hypothesis that alpha diversity indices were significantly different between bird communities at undisturbed seasonally flooded habitats and areas permanently flooded by the dams in each river, we employed Generalized Linear Mixed Models (GLMMs) to analyze the variation in alpha diversity metrics across different sampling sites and rivers. Models were selected using an information theory approach based on AIC (Akaike 1974). Models with Delta AIC <= 2 were considered equally plausible and we used the model weight (AIC) to contrast the best model to the constant (no-effect) model.

Beta diversity

We calculated a dissimilarity matrix using the Jaccard index based on species occurrences (Legendre & Legendre, 2012). We then partitioned beta taxonomic diversity into turnover and nestedness components using the beta.pair function of the R package BETAPART (Baselga & Orme, 2012). Turnover consists in the substitution of species between sites, while nestedness implies that the poorer assemblage is a subset of the richer one. We calculated phylogenetic beta diversity, which represents the differences in evolutionary history among lineages in a community. We carried out an MPD and MNTD extension using two functions of the PICANTE package: comdist calculates the mean phylogenetic distance between species in each site, while comdistnt, calculates the average phylogenetic distance between the closest taxon of a species in two communities. Functional beta diversity allows comparison of the functional characters of species from different localities.

To visualize beta diversity patterns, we performed a NMDS for each beta diversity metric and tested for differences between groups using a permutational analysis of variance (PERMANOVA) using the function adonis2 of the VEGAN package. All analyses were performed using R software (Core Development Team, 2016).

Results

Alpha diversity

Our analyses with the three separate datasets (all species, without aquatic and floodplain specialists) yielded similar results, thus we present only the results obtained using all sampled species (see Table S5 for results obtained using the other subsets). We recorded 202 species in total (Table S3). Estimates of total richness ranged from 217 to 232 (mean 224 ±7.27) species (Table S4), and the cumulative effort curve did not stabilize (Figure S2). Mean number of species per site was 35.29 ± 7.45. Total species richness tended to be higher on undisturbed sites (Table S4), but average species richness did not statistically differ (Estimate = -4.72, p = 0.07) (Figure 2A).

We registered 145 species from 40 bird families in the Madeira River, with 68 species from nine families found exclusively there. Psittacidae, Thamnophilidae, Thraupidae, and Tyrannidae represented more than one-third (35%) of all species registered in the Madeira River. We registered 130 species from 37 bird families in the Xingu River, with 59 species from six families unique to this system. Dendrocolaptidae, Thamnophilidae, Thraupidae, and Tyrannidae represented 40% of all species registered in the Xingu River.

At the Madeira River, we registered some island specialists and species commonly found in early successional habitats. Mazaria propinqua, Stigmatura napensis, Cranioleuca vulpina, Myrmochanes hemileucus, and Elaenia pelzelni were registered only on the islands of the impacted sites, while Conirostrum margaritae was found only on undisturbed islands. In addition, species typically associated with várzeas such as Pipra fasciicauda, Myrmoborus leucophrys, Cranioleuca gutturata, and Hemitriccus minor were registered only at undisturbed sites.

In the Xingu River, Colaptes punctigula, Crypturellus undulatus, Myrmeciza atrothorax, Ornithion inerme, and Picumnus aurifrons were registered just in undisturbed sites, while Inezia subflava, Sporophila minuta, Troglodytes aedon, Sublegatus obscurior, and Pitangus lictor were recorded only on impacted sites. Knipolegus orenocensis and Tolmomyias sulphurescens, two island specialists, were found in both undisturbed and impacted sites.

Undisturbed sites had significantly lower Phylogenetic Diversity (PD) (Estimate = -255.67, SE = 83.78, p = 0.00455) and Mean Nearest Taxon Distance (MNTD) (Estimate = -7.16, SE = 2.39, p = 0.0052) and marginally lower Mean Pairwise Distance (MPD) (Estimate = -3.859, SE = 2.237, p = 0.0943) than impacted sites (Figure 2B, 2C and 2D). Madeira and Xingu rivers showed no differences in PD (Estimate = 58.08, SE = 82.39, p = 0.48598), MPD (Estimate = -1.994, SE = 2.200, p = 0.3716) and MNTD mean values (Estimate = 2.30, SE = 2.35, p = 0.3341) (Figure 2B, 2C and 2D). These results indicate that species composition on undisturbed sites was more clustered in terms of evolutionary history than on the impacted sites.

Undisturbed sites also had lower Functional Richness (FRic) (Estimate = -0.12, SE = 0.02, p = 1.01e-08) and Functional Dispersion (FDis) (Estimate = -0.12, SE = 0.02, p = 7.28e-08) than impacted sites (Figure 2E and 2F). While FRic was higher on the Xingu river (Estimate = 0.047, SE = 0.02, p = 0.006), FDis estimates did not differ between Madeira and Xingu rivers (Estimate = 0.02, SE = 0.02, p = 0.342).

At impacted sites, 60% of recorded species had low sensitivity to disturbance, and only 6% had high sensitivity. At undisturbed sites the proportions are more similar, with low and high sensitivity species each representing about 30% of recorded species (Figure 3A). Average range size did not vary between river basins, but was lower for species recorded on undisturbed sites (Estimate = -1779243, SE = 348251, p = 1.45e-05) (Figure 3B, Table S5). Finally, we found lower species dispersal ability (HWI) in the undisturbed sites than in the permanently flooded sites (Estimate = -3.14, SE = 1.23, p = 0.03).

Beta diversity

Although taxonomic diversity was not significantly different (see above), our results show high species turnover among sites (Figure 4A). We found taxonomic composition to be significantly distinct between impacted and undisturbed sites (F = 3.92, r^2 = 0.18, p = 0.001), and between rivers (F = 7.7, r^2 = 0.20, p = 0.001). Mean taxonomic dissimilarity was 0.72 ± 0.14. The turnover component explained most of the dissimilarity between sites (0.68, ± 0.16).

Phylogenetic beta diversity was also distinct among sites. MPD was significantly different between impacted and undisturbed sites (F = 1.16, r^2 = 0.034, p = 0.001), and between rivers (F = 1.23, r^2 = 0.036, p = 0.001). Results were similar for MNTD, with significant differences between impacted and undisturbed sites (F = 8.06, r^2 = 0.16, p = 0.001), and between rivers (F = 11.12, r^2 = 0.22, p = 0.001).

Finally, functional beta diversity analyses showed the same pattern as for taxonomic and phylogenetic beta diversity: functional composition was distinct between impacted and undisturbed sites (F = 1.98, r^2 = 0.17, p = 0.001) and between rivers (F = 1.45, r^2 = 0.10, p = 0.002), although undisturbed sites were functionally more similar than impacted sites (Figure 4B).

Discussion

We documented significant patterns of variation among bird assemblages at undisturbed and dam-impacted floodplains of two large Amazonian rivers. Although we found no change in species richness, permanent flooding due to river damming affected species composition, phylogenetic and functional diversity. Bird assemblages at impacted sites are composed of generalist, widespread, disturbance-tolerant species, while, bird assemblages of undisturbed sites include more sensitive species, with smaller range sizes, and specialized on seasonally-flooded environments. Results obtained corroborate that species richness, the diversity metric most used in impact assessments (Chiarucci et al., 2011), is not sufficient to characterize the impacts caused by river damming on the biodiversity of Amazonian floodplains. Moreover, we emphasize that evaluating how and in which direction species replacements take place and what characteristics allow species to remain or to migrate after disturbance are more informative for conservation than comparing alpha diversity values alone.

Taxonomic diversity

Although species richness did not vary among sites, the identity of the species did, as revealed by our beta diversity analyses. *Myrmoborus leucophrys*, *Thamnomanes caesius*, and *Lathrotriccus euleri* are species commonly found in floodplain habitats, and were only found in undisturbed areas in both rivers. *Turdus fumigatus*, *Sakesphorus luctuosus*, *Xiphorhynchus obsoletus*, *Hylophylax punctulatus* were found only in igapós, whereas *Brotogeris cyanoptera*, *Hypocnemis peruviana*, *Myrmotherula hauxwelli*, and *Cranioleuca gutturata* were found only in várzeas. These species specialized in the dynamic seasonally flooded environments, were replaced by generalist species in impacted sites, such as *Crotophaga ani*, *Butorides striata*, *Ardea cocoi*, and *Pitangus lictor*. Thus, species loss promoted by the permanent flooding is partially compensated by the colonization by birds tolerant or favored by habitat modification, leading to little if any change in species richness, but large differences in the bird communities considering sensitivity to disturbance and range size (Figure 3A and 3B). Bird communities in impacted sites include species less sensitive to disturbance and species with larger ranges, indicating that habitat specialists are replaced by generalist species.

The absolute value for species richness tended to be higher in the Madeira River, corroborating previous studies that have shown a richness gradient decreasing from western to eastern Amazonian floodplains (Cohn-Haft et al., 1997; Ribas & Aleixo, 2019; Rosenberg, 1990). This is partially explained by greater habitat variability in western floodplains (Rosenberg, 1990). In addition to richness patterns, Amazonian bird communities show large dissimilarity across the basin, which we also corroborate in our results. This variation is due to the heterogeneity of vegetation found along rivers, current and historical variation in habitat connectivity, and species characteristics (Pomara et al., 2014). Although patterns of endemism have been much more discussed for upland terra-firme birds, recent studies have shown that the avifauna associated to seasonally flooded environments is also heterogeneously distributed, with clear patterns of regionalization (Cohn-Haft et al., 2007; Laranjeiras et al., 2019).

Taxonomic composition

Anthropogenic impacts, such as dam construction, often result in species replacements (Olden, 2006) and impacted assemblages tend to become more similar (McKinney & Lockwood, 1999). Our results indicate a high taxonomic turnover among studied sites with a dominance of more resilient species in impacted sites, and a larger number of sensitive species on undisturbed sites (Figure 3A). These results point to a loss of unique characteristics of unique Amazonian habitats. In this process, the differences between bird communities adapted to habitats seasonally flooded by distinct water types (várzeas and igapós) accumulated over evolutionary time are lost due to permanent flooding and the invasion of generalist species.

Floodplains harbor a high diversity that goes beyond the specialist taxa. Cohn-Haft et al. (2007) recorded nearly a quarter of all Brazilian avifauna in the Amazon river floodplains, highlighting the importance of conserving these environments. Due to the seasonal flooding, specialist birds are expected to be good dispersers, able to leave sites and recolonize them later (Cohn-Haft et al., 2007). However, we conducted our sampling during the dry season, a period that naturally has the lowest water level, and still failed to find several specialist species in the impacted areas. The threat to these species is in progress and they may be undergoing a process of local extinction, as they do not seem to move into upland forests due to competition with other species (Rowedder et al., 2021).

Four of the undisturbed sites clustered together with impacted sites in the Madeira River (Figure 4A). Although not affected by permanent flooding, these sites are located on islands covered mostly by early successional vegetation. These habitats occur only on the várzeas of the Madeira River and support a unique avifauna, including specialists such as *Mazaria propinqua* and *Knipolegus orenocensis* (Melo et al., 2021; Rosenberg, 1990). Birds specialized in early successional vegetation seem to benefit from the first stages of vegetation change due to flooding, leading to the observed similarity. However, it is not yet clear how these habitats will support these populations in the long run. Early successional habitats in river islands have been identified as potentially vulnerable to damming due to their dependency on sedimentation patterns, which are highly modified by the permanent flooding (Latrubesse et al., 2020).

Two species classified as vulnerable by IUCN (2022) - *Pionites leucogaster* and Conirostrum margaritae - were recorded. *C. margaritae* is a specialist of short-lived successional vegetation created by and dependent on seasonal flooding. Other species recorded on undisturbed sites that nest near or directly on the ground (e.g. *Crypturellus undulates*, *Tinamus major*, *Cercomacroides nigrescens*, *Myrmoborus leucophrys*) are also vulnerable to permanent flooding (Silva et al., 2022). It has been shown that specialized taxa nesting on fluvial islands have a reproductive cycle highly correlated with the hydrological cycle and are susceptible to flood pulse alterations (Correa et al., 2022). In contrast, transforming the lotic environment into lentic (Alho, 2020; Latrubesse et al., 2020) seems to favor widespread species, such as representatives of the family Ardeidae, detected only in the impacted sites.

Phylogenetic Diversity

Anthropogenic impacts may have varying effects on phylogenetic diversity. Some authors suggest that impacts function as environmental filters, generating less stable and more homogeneous environments and thus favoring communities composed of more closely related species (Helmus et al., 2010). Bird species from younger clades would generally benefit from impacts (Frishkoff et al., 2014; Liang et al., 2019). In central Brazil, for example, dam construction caused a marked reduction in bird phylogenetic diversity (Abreu et al., 2020). Our results, however, show the opposite pattern, with lower phylogenetic diversity in undisturbed sites. Similar results were found for Amazonian terra-firme forest birds, where secondary forests presented higher phylogenetic diversity than undisturbed forests (Borges et al., 2020). This is likely related to the replacement of specialized species by more disturbance-tolerant, generalist species from phylogenetically distant clades on impacted sites.

Indeed, we recorded representatives of lineages broadly associated to aquatic environments (i.e. Anseriformes, Charadriiformes, Suliformes, and Pelecaniformes), with most species found exclusively in the impacted sites. In agreement with our results, Jiang et al. (2019) showed that phylogenetic diversity increased with the invasion of species that did not occur in the assemblages before the impact.

Undisturbed sites had lower MNTD values when compared to impacted sites, and also were the only ones to show significant phylogenetic clustering (Table S5). These results indicate that Amazonian floodplain bird assemblages include closely related species, probably as a consequence of evolutionary specialization to these environments. Phylogenetic beta diversity differed significantly between river types and between impact status. Environmental heterogeneity correlates strongly with phylogenetic dissimilarity (Swenson, 2011; Swenson et al., 2011).

Functional diversity

As expected, our analyses showed lower FRic and FDis for undisturbed sites (Figure 3). Undisturbed floodplains sustain a specific functional group of highly specialized species and may function as an environmental filter causing a phenotypic attraction (Webb et al., 2002). Permanent flooding creates new habitats that may be occupied by species with different characteristics, resulting in an increase in functional diversity. Interestingly, in contrast to the taxonomic turnover, functional composition of undisturbed sites is nested within the functional composition of the impacted sites, suggesting that new traits are added to the community with the arrival of invasive species that benefit from the permanent flooding.

Higher functional diversity in impacted environments is not unprecedented in comparative studies of environments that have undergone some type of change. For example, change in water quality generates a disturbance that results in increased functional diversity (Barbosa et al., 2020), just as deforestation near streams leads to the same pattern (Teresa & Casatti, 2012). However, analyzes that track turnover, in this case functional, can detect greater changes in the composition of assemblages (Hillebrand et al., 2018). In addition, specialized function may be lost with impact, as functional groups comprising generalist species seem to be resilient to variation in vegetation cover, evidencing adaptability to diverse types of environments (Arantes et al., 2019).

Bird assemblages in environments that have suffered some type of disturbance (i.e., logged forests) also show high levels of functional diversity (Chapman et al., 2018). However, these values decrease when exploitation becomes a conversion of forest to monoculture. Therefore, a similar process may happen in sites impacted by permanent flooding. Functional diversity may increase during a certain post-impact period but, as the trees die and the habitat changes, only a few of the current species remain and the multifunctional space may be reduced.

Conclusions

Our results show that the landscape change caused by permanent flooding allowed disturbance-tolerant, generalist species, from different phylogenetic clades and functional groups, to occupy the disturbed environment, in detriment of the original avifauna specialized in seasonally flooded habitats. This impact, however, is not apparent if only mean values for the distinct alpha diversity metrics are compared. These results also point to the importance of long-term monitoring of impacted floodplains, as impacts may change in the period following the initiation of disturbance.

The biota associated with floodplains is an important component of overall Amazonian diversity, but is threatened by dams due to reduction and fragmentation of available habitat, aggravated by the fact that many specialist species have significantly smaller range sizes when compared to species that are benefited by the impacts. As our results show, some specialized species will not occupy these permanently flooded areas, and thus their populations will be disconnected along the riverine habitats above and below the reservoir.

Monitoring of environmental impacts caused by dams should be standardized to include seasonally flooded environments as a unique category. In the post-impact period, monitoring must consider the long-term changes in the landscape. Importantly, the sensitivity of Amazonian seasonally flooded habitats to flood pulse alterations, and their importance for the whole Amazonian system must be considered when evaluating the supposedly smaller impacts attributed to run-of-the-river dams in Amazonia.

Author Contribution Statement

GMM, CCR and JSM conceived the study. GMM and TNM processed and validated data. GMM and JSM performed the analyses. CCR was responsible for funding acquisition and project administration. All authors wrote the manuscript.

Conflict of Interest

The corresponding author confirms on behalf of all authors that there have been no involvements that might raise the question of bias in the work reported or in the conclusions, implications, or opinions stated.

Data Availability Statement

The data supporting the results of this study will be openly available in Dryard with a DOI and reference number after the journal submission process.

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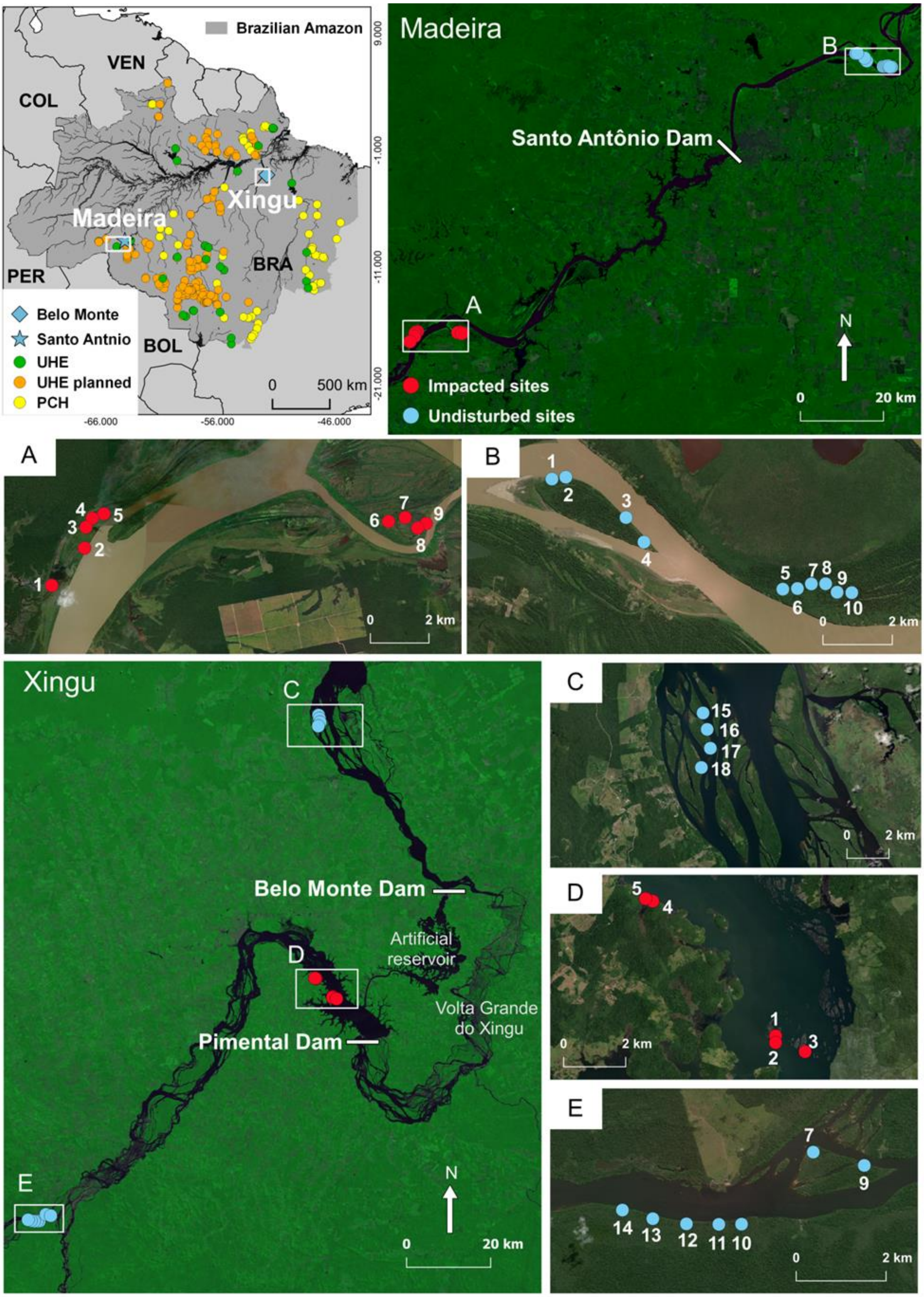
Figures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BRA
PER
BOL
Belo Monte
Santo Antnio
UHE
UHE planned
PCH
0
500 km
9.000
-1.000
-11.000
-21.000
-66.000
-56.000
-46.000
Madeira
B
Santo Antônio Dam
A
N
Impacted sites
Undisturbed sites
0
20 km
A
1
2
3
4
5
6
7
8
9
0
2 km
B
1
2
3
4
5
6
7
8
9
10
0
2 km
Xingu
C
Belo Monte Dam
Artificial reservoir
D
Volta Grande do Xingu
Pimental Dam
E
N
0
20 km
C
15
16
17
18
0
2 km
D
5
4
1
2
3
0
2 km
E
7
9
14
13
12
11
10
0
2 km

Fig 1: Map of South America highlighting the Amazon, its Brazilian hydrography and respective hydropower dams, both in operation and planned.

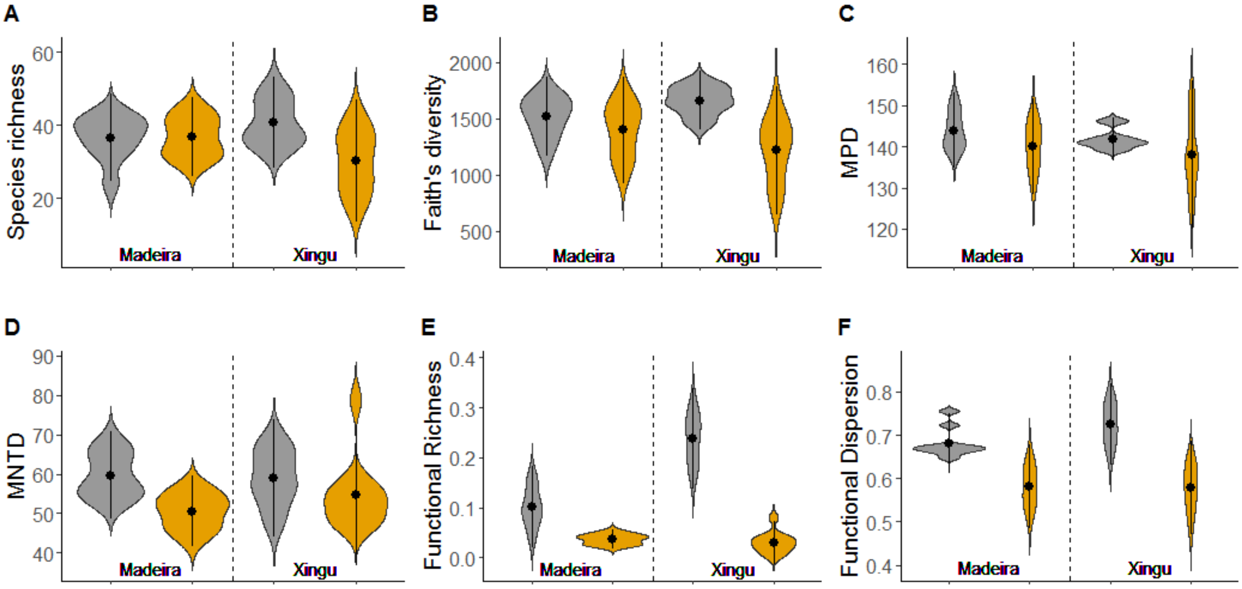

Fig 2: Violin plot showing the distribution patterns of different alpha diversity metrics in impacted (gray – disturbed Madeira and Xingu sites) and undisturbed (orange – undisturbed Madeira and Xingu) sites. The first two violins represent the Madeira River, while the other two represent the Xingu River. The plots are Mean Species Richness (A), Phylogenetic Diversity - PD (B), Mean Pairwise Distance (C), Mean Nearest Taxon Distance (D), Functional Richness - FRic (E) and finally Functional Dispersion - FDis (F).

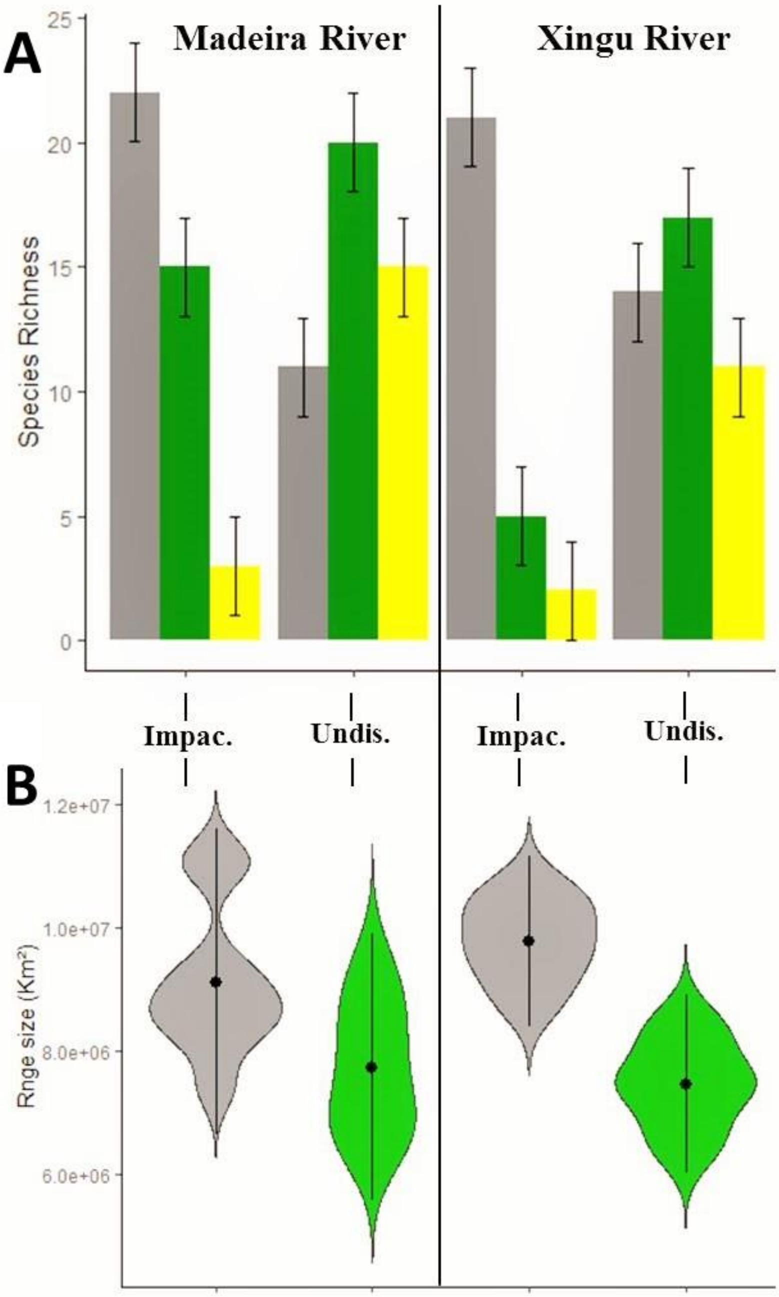

Fig 3: Degree of species sensitivity to disturbance. The bar plot shows the number of species with low (gray), medium (green), and high (yellow) sensitivity to disturbance for each river basin and impact status . (D) Average range size (km²) of species found in sites at each river basin and impact status, showing that species found in undisturbed sites have smaller ranges than those recorded in impacted sites. Colors represent sites in undisturbed (green) and impacted (gray) areas.

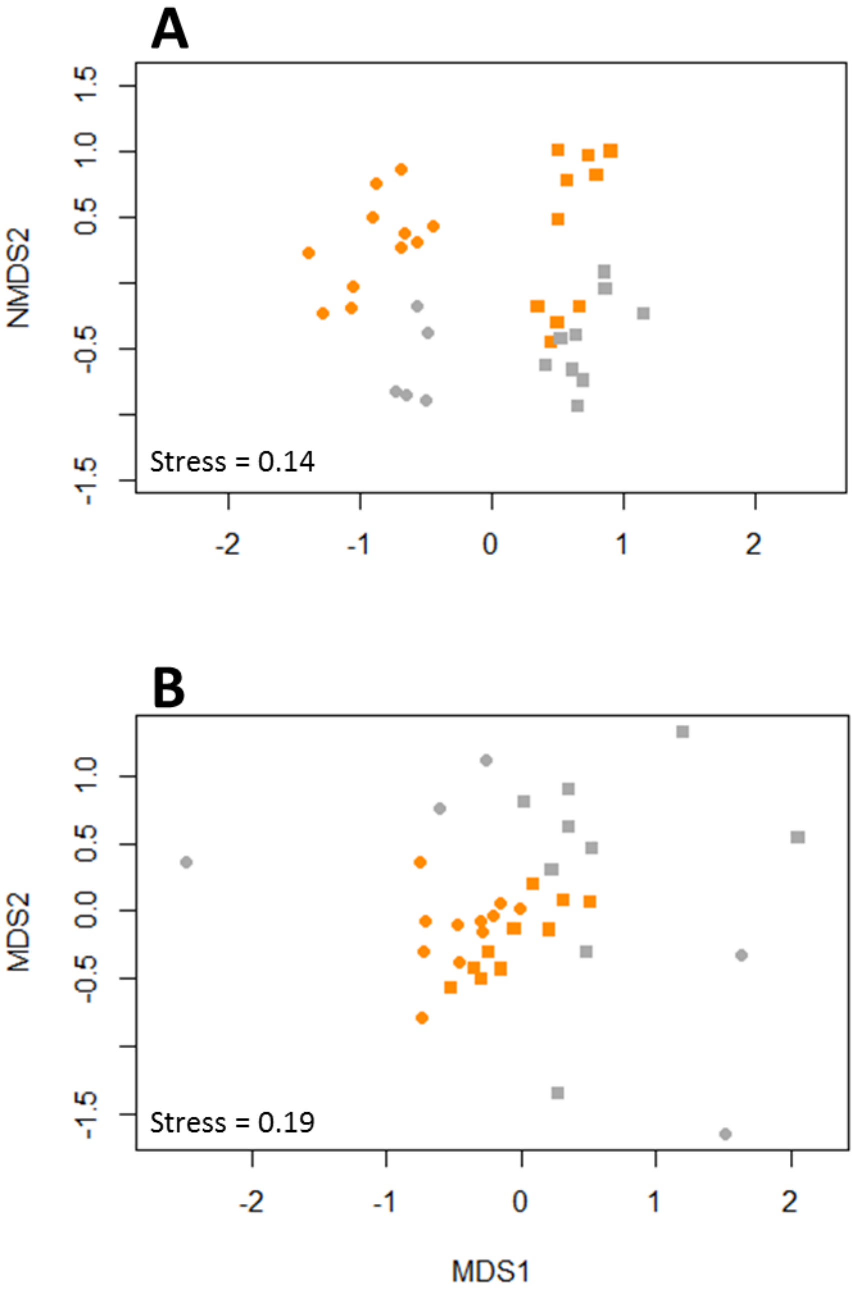

Fig 4: Non-metric multidimensional scaling (NMDS) shows dissimilarity in taxonomic composition **(A)** and in functional composition **(B)** between sites. For both **(A)** and **(B)** colors represent sites in undisturbed (orange) and impacted (gray) areas, while squares represent sites in the Madeira River and dots represent sites in the Xingu River.

CONCLUSÃO

Foi realizado um estudo sobre a diversidade regional de aves em áreas de planície de inundação, tanto perturbadas quanto não perturbadas por barragens, em dois grandes rios amazônicos. Observou-se que as diversidades taxonômica, filogenética e funcional das aves foram afetadas pela interrupção do ciclo de inundação causado pelas barragens. As mudanças na diversidade foram similares nas áreas de igapós e várzeas, e a riqueza de espécies por si só não foi suficiente para entender os impactos da barragem sobre a biodiversidade das florestas alagáveis. Como mostram os resultados, algumas espécies especializadas não ocuparão estas áreas permanentemente inundadas, e assim suas populações serão fragmentadas ao longo dos habitats ribeirinhos acima e abaixo do reservatório, afetando o tamanho das populações e a diversidade genética. Considerando a complexidade dos impactos a estes sistemas e os muitos processos ecológicos e sociais que deles dependem, os custos a longo prazo de tais projetos têm sido muito subestimados até agora. Finalmente, as análises de diversidade fornecem diversas opções de interpretação, e cabe a nós utilizá-las em suas diferentes facetas e interpretar os resultados voltados para o mundo biológico real, além dos números.

APÊNDICE

APÊNDICE A - SUPPORTING INFORMATION

IMPACTS OF LARGE DAMS ON AMAZONIAN FLOODPLAIN BIRD COMMUNITIES

Gustavo de Melo Martins[1]*, Juliana da Silva Menger2, Tomaz Nascimento de Melo3, Camila Cherem Ribas1,2

1 - Programa de Pós-Graduação em Ecologia, Instituto Nacional de Pesquisas da Amazônia, Manaus, AM, Brazil
2 - Coordenação de Biodiversidade, Instituto Nacional de Pesquisas da Amazônia, Manaus, AM, Brazil
3 – Rainforest Connection, Science Department, 440 Cobia Drive, Suite 1902, Katy, TX, 77494, USA

Figure S1: Model of a passive autonomous recorder installed in the impacted region of the Madeira River, September 2017. Photo captured by TNM.

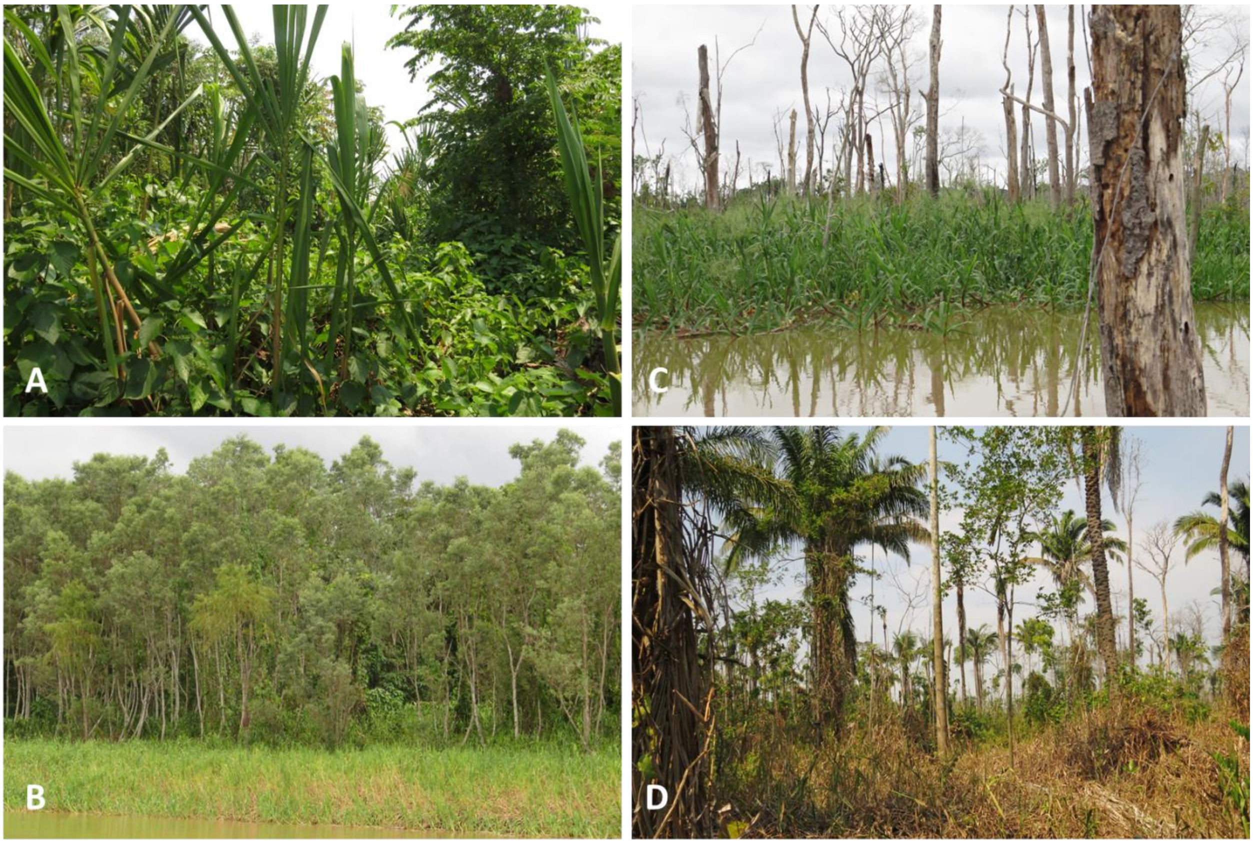

Figure S2: Areas were sampled along the Madeira River, including **(A)** a forest understory dominated by Cecropia trees, **(B)** a floodplain unaffected by permanent flooding, and **(C,D)** areas that have already been impacted by permanent flooding and are now exhibiting changes in their landscape. Photos captured by TNM

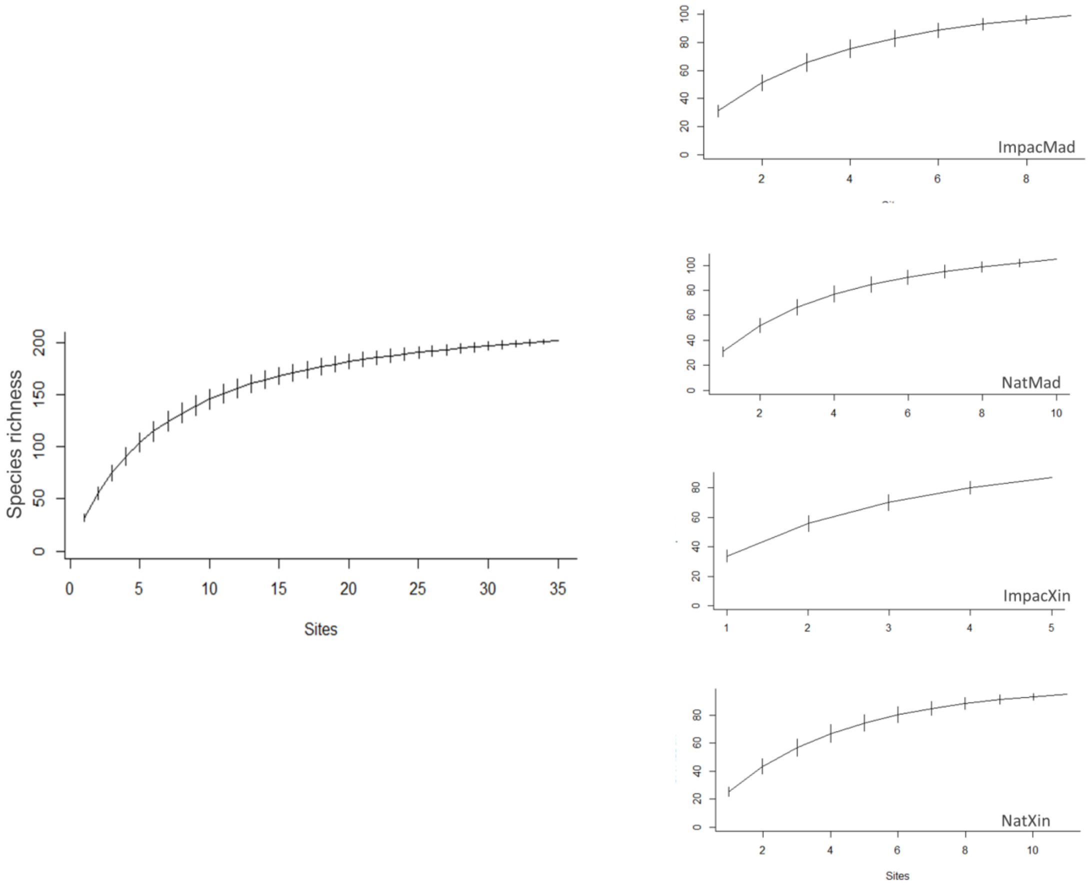

Figure S3: Smoothed collector's curve including all locations in both rivers (the left one). We used 19 sites in Madeira River and 16 in Xingu River, nine, ten, five, and eleven following the ones on the right, from top to bottom.

Table S1: Sampled sites in the Madeira and Xingu rivers. Column Name represents the names used in Arbimon projects. Sites marked with an asterisk showed phylogenetic clustering tested using the null model

River	Name	Region	Local	Code	Latitude	Longitude	Recording counts (min)
Madeira	Jusante 01*	Downstream	Island	NatMad	-8.58169	-63.62578	2440
	Jusante 02*	Downstream	Island	NatMad	-8.57525	-63.63050	2871
	Jusante 03*	Downstream	Island	NatMad	-8.56517	-63.64992	2868
	Jusante 04*	Downstream	Island	NatMad	-8.56467	-63.64628	2862
	Jusante 05	Downstream	Riverbank	NatMad	-8.59403	-63.58936	2832

	Jusante 06*	Downstream	Riverbank	NatMad	-8.59386	-63.58553	2859
	Jusante 07	Downstream	Riverbank	NatMad	-8.59264	-63.58189	900
	Jusante 08	Downstream	Riverbank	NatMad	-8.59261	-63.57814	2860
	Jusante 09	Downstream	Riverbank	NatMad	-8.59483	-63.57514	2521
	Jusante 10*	Downstream	Riverbank	NatMad	-8.59494	-63.57128	677
	Montante 01	Upstream	Riverbank	ImpacMad	-9.17558	-64.61267	2859
	Montante 02	Upstream	Riverbank	ImpacMad	-9.17697	-64.61631	2841
	Montante 03	Upstream	Riverbank	ImpacMad	-9.17975	-64.61825	2816
	Montante 04	Upstream	Riverbank	ImpacMad	-9.18622	-64.61869	2728
	Montante 05	Upstream	Riverbank	ImpacMad	-9.19781	-64.62889	2718
	Montante 07	Upstream	Island	ImpacMad	-9.18506	-64.51908	2659
	Montante 08	Upstream	Island	ImpacMad	-9.18314	-64.52425	2618
	Montante 09	Upstream	Island	ImpacMad	-9.18417	-64.51522	1745
	Montante 10	Upstream	Island	ImpacMad	-9.18100	-64.51256	2570
Xingu	Xingu 01	Upstream	Island	ImpacXin	-3.34746	-52.01844	1521
	Xingu 02	Upstream	Island	ImpacXin	-3.34943	-52.01844	1870
	Xingu 03	Upstream	Island	ImpacXin	-3.35203	-52.00988	1882
	Xingu 04	Upstream	Riverbank	ImpacXin	-3.30840	-52.05407	1723
	Xingu 05	Upstream	Riverbank	ImpacXin	-3.30768	-52.05615	1881
	Xingu 07	Upstream	Island	NatXin	-3.81355	-52.63248	1469
	Xingu 09	Upstream	Island	NatXin	-3.81622	-52.62225	1470
	Xingu 10*	Upstream	Riverbank	NatXin	-3.82799	-52.64687	1458
	Xingu 11	Upstream	Riverbank	NatXin	-3.82803	-52.65143	1457
	Xingu 12	Upstream	Riverbank	NatXin	-3.82793	-52.65793	1452
	Xingu 13*	Upstream	Riverbank	NatXin	-3.82690	-52.66468	1450
	Xingu 14	Upstream	Riverbank	NatXin	-3.82517	-52.67078	1446
	Xingu 15*	Downstream	Island	NatXin	-2.74465	-52.04837	1010
	Xingu 16*	Downstream	Island	NatXin	-2.75169	-52.04659	1007
	Xingu 17	Downstream	Island	NatXin	-2.75977	-52.04522	1008
	Xingu 18*	Downstream	Island	NatXin	-2.76803	-52.04908	1003

Table S2: Traits used to calculate the functional diversity indices. All sources were used from Tobias et al., 2022 **(I)**, Sheard et al., 2020 **(II)**, and Stotz et al., 1996 **(III)**.

Data type	Functional component	Traits	Traits values
Continuous	Size	Body mass (g) (I)	Trait with many aspects of species biology/ecology
	Dispersal ability	Hand-wing Index (II)	A metric of flight efficiency and dispersal ability in birds
	Habitat specialization	Island (I)	Association with riverine islands

	Biogeographical information	Range size (km^2) (I)	Geographical range of the species
Categorical	Social interaction	Territoriality (I)	Regarding the level of species tolerance to other individuals
	Environmental disturbance	Sensitivity (III)	Level of sensitivity regarding the alteration of the natural landscape
	Foraging patterns	Lifestyle (I)	Environment where species spend the most time foraging
	Breeding biology	Habitat type (I)	Types of environment where the breeding sites are usually found
	Dietary niche	Diet (I)	Trophic niches that encompass the main types of resources used by birds.

Table S3: Birds registered in each of the four environments. We followed the Birds of the World (2022) nomenclature. We used the IUCN (2022) classification of threatened species. The species with an asterisk (n=20) were removed from the analyses in the subset of non-aquatic species. Highlighted species (n=46) were considered in the subset floodplain specialist birds.

Category	**Taxa**	**IUCN**	NatMad	ImpacMad	NatXin	ImpacXin
Order	**Tinamiformes Huxley, 1872**					
Family	Tinamidae Gray, 1840					
	Tinamus major (Gmelin, 1789)	LC	x			
	***Crypturellus cinereus* (Gmelin, 1789)**	LC			x	x
	***Crypturellus undulatus* (Temminck, 1815)**	LC	x	x	x	
Order	**Anseriformes Linnaeus, 1758**					
Family	Anhimidae Stejneger, 1885					
	**Anhima cornuta* (Linnaeus, 1766)	LC				x
Order	**Galliformes Linnaeus, 1758**					
Family	Cracidae Rafinesque, 1815					
	Pipile cujubi (Pelzeln, 1858)	VU		x		
	Ortalis guttata (Spix, 1825)	LC	x	x		
Order	**Columbiformes Latham, 1790**					
Family	Columbidae Leach, 1820					
	Patagioenas cayennensis (Bonnaterre, 1792)	LC	x		x	x

	Leptotila rufaxilla (Richard & Bernard, 1792)	LC	x	x	x	
Order	**Cuculiformes Wagler, 1830**					
Family	Cuculidae Leach, 1820					
	Crotophaga major Gmelin, 1788	LC		x	x	x
	*_Crotophaga ani_ Linnaeus, 1758	LC		x		x
	Coccycua minuta (Vieillot, 1817)	LC	x	x	x	
	Piaya cayana (Linnaeus, 1766)	LC	x		x	
Order	**Nyctibiiformes Yuri, Kimball, Harshman, Bowie, Braun, Chojnowski, Hackett, Huddleston, Moore, Reddy, Sheldon, Steadman, Witt & Braun, 2013**					
Family	Nyctibiidae Chenu & Des Murs, 1851					
	Nyctibius grandis (Gmelin, 1789)	LC	x	x		
Order	**Caprimulgiformes Ridgway, 1881**					
Family	Caprimulgidae Vigors, 1825					
	Nyctiprogne leucopyga (Spix, 1825)	LC			x	x
	Nyctidromus albicollis (Gmelin, 1789)	LC		x	x	
Order	**Apodiformes Peters, 1940**					
Family	Apodidae Olphe-Galliard, 1887					
	*_Chaetura chapmani_ Hellmayr, 1907	LC		x		x
	*_Chaetura brachyura_ (Jardine, 1846)	LC		x		x
Family	Trochilidae Vigors, 1825					
	Glaucis hirsutus (Gmelin, 1788)	LC	x			x
	Phaethornis ruber (Linnaeus, 1758)	LC			x	x
	Phaethornis hispidus (Gould, 1846)	LC	x	x		
	Phaethornis superciliosus (Linnaeus, 1766)	LC			x	
	Amazilia fimbriata (Gmelin, 1788)	LC		x		x
	Hylocharis cyanus (Vieillot, 1818)	LC			x	x
Family	Rallidae Rafinesque, 1815					
	Laterallus exilis (Temminck, 1831)	LC		x		
	Aramides cajaneus (Statius Muller, 1776)	LC	x			x
Order	**Charadriiformes Huxley, 1867**					
Family	Charadriidae Leach, 1820					
	Vanellus chilensis (Molina, 1782)	LC		x		
Family	Jacanidae Chenu & Des Murs, 1854					
	*_Jacana jacana_ (Linnaeus, 1766)	LC		x		
Family	Laridae Rafinesque, 1815					
	*_Phaetusa simplex_ (Gmelin, 1789)	LC		x		
Order	**Eurypygiformes Fürbringer,**					

	1888					
Family	Eurypygidae Selby, 1840					
	Eurypyga helias (Pallas, 1781)	LC			x	x
Order	**Suliformes Sharpe, 1891**					
Family	Anhingidae Reichenbach, 1849					
	**Anhinga anhinga* (Linnaeus, 1766)	LC				x
Order	**Pelecaniformes Sharpe, 1891**					
Family	Ardeidae Leach, 1820					
	**Butorides striata* (Linnaeus, 1758)	LC		x		x
	**Ardea cocoi* Linnaeus, 1766	LC		x		x
	**Egretta thula* (Molina, 1782)	LC		x		
Order	**Accipitriformes Bonaparte, 1831**					
Family	Pandionidae Bonaparte, 1854					
	**Pandion haliaetus* (Linnaeus, 1758)	LC			x	x
Family	Accipitridae Vigors, 1824					
	Leptodon cayanensis (Latham, 1790)	LC	x	x		
	***Spizaetus tyrannus* (Wied, 1820)**	LC	x			
	**Rostrhamus sociabilis* (Vieillot, 1817)	LC				x
	***Leucopternis schistaceus* (Sundevall, 1850)**	LC			x	x
	Buteogallus urubitinga (Gmelin, 1788)	LC	x		x	x
	Rupornis magnirostris (Gmelin, 1788)	LC	x	x	x	x
	Buteo nitidus (Latham, 1790)	LC		x		
Order	**Strigiformes Wagler, 1830**					
Family	Strigidae Leach, 1820					
	Megascops choliba (Vieillot, 1817)	LC		x	x	x
	Megascops usta (Sclater, 1858)	LC	x			
	Lophostrix cristata (Daudin, 1800)	LC	x		x	
	Ciccaba virgata (Cassin, 1849)	LC	x			
	Glaucidium hardyi Vielliard, 1990	LC	x			
	Glaucidium brasilianum (Gmelin, 1788)	LC	x			
Order	**Trogoniformes A. O. U., 1886**					
Family	Trogonidae Lesson, 1828					
	Trogon melanurus Swainson, 1838	LC	x			
	Trogon viridis Linnaeus, 1766	LC	x			
Order	**Coraciiformes Forbes, 1844**					
Family	Alcedinidae Rafinesque, 1815					
	**Megaceryle torquata* (Linnaeus, 1766)	LC		x	x	x

	***Chloroceryle amazona* (Latham, 1790)	LC		x		x
Order	**Galbuliformes Fürbringer, 1888**					
Family	Galbulidae Vigors, 1825					
	Galbula ruficauda Cuvier, 1816	LC		x	x	
	Galbula cyanescens Deville, 1849	LC		x		
	Galbula dea (Linnaeus, 1758)	LC	x		x	
Family	Bucconidae Horsfield, 1821					
	Chelidoptera tenebrosa (Pallas, 1782)	LC				x
	Monasa nigrifrons (Spix, 1824)	LC	x	x	x	x
	Bucco tamatia (Gmelin, 1788)	LC	x		x	x
	Bucco capensis Linnaeus, 1766	LC			x	
Order	**Piciformes Meyer & Wolf, 1810**					
Family	Capitonidae Bonaparte, 1838					
	Capito auratus (Dumont, 1816)	LC	x			
Family	Ramphastidae Vigors, 1825					
	Ramphastos tucanus Linnaeus, 1758	LC	x	x	x	x
	Ramphastos vitellinus Lichtenstein, 1823	LC	x	x		
	***Pteroglossus aracari* (Linnaeus, 1758)**	LC			x	
	Pteroglossus castanotis Gould, 1834	LC	x			
Family	Picidae Leach, 1820					
	***Picumnus aurifrons* Pelzeln, 1870**	LC		x	x	
	Melanerpes cruentatus (Boddaert, 1783)	LC	x	x		
	Veniliornis passerinus (Linnaeus, 1766)	LC	x	x		
	Campephilus melanoleucos (Gmelin, 1788)	LC	x	x	x	x
	Dryocopus lineatus (Linnaeus, 1766)	LC	x	x	x	x
	Celeus torquatus (Boddaert, 1783)	NT			x	
	***Celeus flavus* (Statius Muller, 1776)**	LC			x	x
	Colaptes punctigula (Boddaert, 1783)	LC	x	x	x	
Order	**Falconiformes Bonaparte, 1831**					
Family	Falconidae Leach, 1820					
	Herpetotheres cachinnans (Linnaeus, 1758)	LC	x	x	x	
	Micrastur ruficollis (Vieillot, 1817)	LC			x	
	Ibycter americanus (Boddaert, 1783)	LC	x	x		
	Daptrius ater Vieillot, 1816	LC	x		x	x

	**Milvago chimachima* (Vieillot, 1816)	LC	x	x		x
	Falco rufigularis Daudin, 1800	LC		x		x
Order	**Psittaciformes Wagler, 1830**					
Family	Psittacidae Rafinesque, 1815					
	Brotogeris cyanoptera (Pelzeln, 1870)	LC	x	x		
	Pionus menstruus (Linnaeus, 1766)	LC	x	x	x	x
	Amazona ochrocephala (Gmelin, 1788)	LC	x			
	Amazona farinosa (Boddaert, 1783)	LC	x	x		
	***Amazona amazonica* (Linnaeus, 1766)**	LC			x	x
	Pionites leucogaster (Kuhl, 1820)	VU	x	x		
	Aratinga weddellii (Deville, 1851)	LC	x	x		
	Orthopsittaca manilatus (Boddaert, 1783)	LC		x		
	Ara severus (Linnaeus, 1758)	LC	x	x	x	x
	Ara macao (Linnaeus, 1758)	LC	x	x	x	
	Aratinga leucophthalmus (Statius Muller, 1776)	LC			x	x
Order	**Passeriformes Linnaeus, 1758**					
Family	Thamnophilidae Swainson, 1824					
	Epinecrophylla amazonica (Ihering, 1905)	LC	x			
	Myrmeciza atrothorax (Boddaert, 1783)	LC			x	
	***Myrmochanes hemileucus* (Sclater & Salvin, 1866)**	LC		x		
	Myrmotherula brachyura (Hermann, 1783)	LC	x			
	Myrmotherula sclateri Snethlage, 1912	LC			x	
	***Myrmotherula multostriata* Sclater, 1858**	LC			x	x
	Myrmotherula axillaris (Vieillot, 1817)	LC	x		x	x
	Myrmotherula hauxwelli (Sclater, 1857)	LC	x			
	Myrmotherula menetriesii (d'Orbigny, 1837)	LC			x	
	Thamnomanes caesius (Temminck, 1820)	LC	x		x	
	***Sakesphorus luctuosus* (Lichtenstein, 1823)**	LC			x	x
	Thamnophilus doliatus (Linnaeus, 1764)	LC		x		
	Thamnophilus schistaceus d'Orbigny, 1835	LC	x		x	
	Thamnophilus aethiops Sclater, 1858	LC				x
	***Thamnophilus amazonicus* Sclater, 1858**	LC			x	x

	Cymbilaimus lineatus (Leach, 1814)	LC	x		x	
	Hypocnemoides maculicauda (Pelzeln, 1868)	LC			x	x
	Hylophylax punctulatus (Des Murs, 1856)	LC			x	x
	Sclateria naevia (Gmelin, 1788)	LC		x	x	x
	Myrmoborus leucophrys (Tschudi, 1844)	LC	x		x	
	Cercomacra cinerascens (Sclater, 1857)	LC			x	x
	Cercomacroides nigrescens (Cabanis & Heine, 1859)	LC			x	
	Hypocnemis peruviana Taczanowski, 1884	LC	x	x		
Family	Dendrocolaptidae Gray, 1840					
	Sittasomus griseicapillus (Vieillot, 1818)	LC	x	x		
	Dendrocincla fuliginosa (Vieillot, 1818)	LC	x		x	
	Dendrexetastes rufigula (Lesson, 1844)	LC	x	x	x	
	Nasica longirostris (Vieillot, 1818)	LC	x	x	x	x
	Dendrocolaptes certhia (Boddaert, 1783)	LC	x		x	x
	Dendrocolaptes picumnus Lichtenstein, 1820	LC	x			
	Xiphorhynchus obsoletus (Lichtenstein, 1820)	LC			x	x
	Xiphorhynchus guttatus (Lichtenstein, 1820)	LC	x	x	x	x
	Dendroplex picus (Gmelin, 1788)	LC	x	x	x	x
Family	Xenopidae Bonaparte, 1854					
	Xenops minutus (Sparrman, 1788)	LC			x	x
Family	Furnariidae Gray, 1840					
	Furnarius figulus (Lichtenstein, 1823)	LC				x
	Automolus paraensis Hartert, 1902	LC			x	
	Cranioleuca vulpina (Pelzeln, 1856)	LC		x		
	Cranioleuca gutturata (d'Orbigny & Lafresnaye, 1838)	LC	x			
	Synallaxis gujanensis (Gmelin, 1789)	LC	x	x		
	Synallaxis propinqua (Pelzeln, 1859)	LC		x		
Family	Pipridae Rafinesque, 1815					
	Pipra fasciicauda Hellmayr, 1906	LC	x			
	Pipra rubrocapilla (Temminck, 1821)	LC			x	
Family	Tityridae Gray, 1840					
	Pachyramphus castaneus (Jardine & Selby, 1827)	LC	x	x		

	Pachyramphus polychopterus (Vieillot, 1818)	LC	x	x		
Family	Rhynchocyclidae Berlepsch, 1907					
	Tolmomyias sulphurescens (Spix, 1825)	LC	x	x	x	
	Tolmomyias poliocephalus (Taczanowski, 1884)	LC	x	x		
	Tolmomyias flaviventris (Wied, 1831)	LC	x		x	x
	Todirostrum maculatum (Desmarest, 1806)	LC	x	x	x	x
	Todirostrum chrysocrotaphum Strickland, 1850	LC	x		x	x
	Myiornis ecaudatus (d'Orbigny & Lafresnaye, 1837)	LC	x	x	x	
	Hemitriccus minor (Snethlage, 1907)	LC	x		x	x
Family	Tyrannidae Vigors, 1825					
	Stigmatura napensis Chapman, 1926	LC		x		
	Inezia subflava (Sclater & Salvin, 1873)	LC				x
	Ornithion inerme Hartlaub, 1853	LC			x	
	Camptostoma obsoletum (Temminck, 1824)	LC	x	x	x	x
	Elaenia flavogaster (Thunberg, 1822)	LC				x
	Elaenia pelzelni Berlepsch, 1907	LC		x		
	Myiopagis gaimardii (d'Orbigny, 1839)	LC	x	x	x	x
	Tyrannulus elatus (Latham, 1790)	LC	x	x		
	Capsiempis flaveola (Lichtenstein, 1823)	LC			x	
	Attila cinnamomeus (Gmelin, 1789)	LC	x	x	x	x
	Attila spadiceus (Gmelin, 1789)	LC	x			
	Legatus leucophaius (Vieillot, 1818)	LC	x			
	Myiarchus tuberculifer (d'Orbigny & Lafresnaye, 1837)	LC	x	x		
	Myiarchus ferox (Gmelin, 1789)	LC	x	x	x	x
	Pitangus sulphuratus (Linnaeus, 1766)	LC	x	x	x	x
	Pitangus lictor (Lichtenstein, 1823)	LC		x		x
	Myiodynastes maculatus (Statius Muller, 1776)	LC				x
	Megarynchus pitangua (Linnaeus, 1766)	LC		x		
	Myiozetetes similis (Spix, 1825)	LC		x		
	Tyrannus melancholicus Vieillot, 1819	LC	x	x	x	x
	Sublegatus obscurior Todd, 1920	LC				x

	***Cnemotriccus fuscatus* (Wied, 1831)**	LC	x	x		
	Lathrotriccus euleri (Cabanis, 1868)	LC	x		x	
	***Knipolegus orenocensis* Berlepsch, 1864**	LC		x	x	x
Family	Vireonidae Swainson, 1837					
	Vireolanius leucotis (Swainson, 1838)	LC			X	
	***Hylophilus semicinereus* Sclater & Salvin, 1867**	LC			x	x
Family	Hirundinidae Rafinesque, 1815					
	Pachysylvia hypoxantha Pelzeln, 1868	LC			x	x
	*****_Progne tapera_ (Linnaeus, 1766)	LC				x
	*****_Progne chalybea_ (Gmelin, 1789)	LC	x	x		x
	*****_Tachycineta albiventer_ (Boddaert, 1783)	LC		x		x
Family	Troglodytidae Swainson, 1831					
	Troglodytes aedon Naumann, 1823	LC	x	x		x
	Campylorhynchus turdinus (Wied, 1831)	LC	x			
	Thryothorus genibarbis (Swainson, 1838)	LC			x	
	***Thryothorus leucotis* (Lafresnaye, 1845)**	LC	x	x	x	x
Family	Polioptilidae Baird, 1858					
	Ramphocaenus melanurus Vieillot, 1819	LC	x		x	x
	Polioptila plumbea (Gmelin, 1788)	LC			x	x
Family	Donacobiidae Aleixo & Pacheco, 2006					
	Donacobius atricapilla (Linnaeus, 1766)	LC		x		
Family	Turdidae Rafinesque, 1815					
	Turdus fumigatus Lichtenstein, 1823	LC			x	
	Turdus hauxwelli Lawrence, 1869	LC		x		
Family	Fringillidae Leach, 1820					
	Euphonia chrysopasta Sclater & Salvin, 1869	LC		x		
	Euphonia violacea (Linnaeus, 1758)	LC			x	x
	***Euphonia laniirostris* d'Orbigny & Lafresnaye, 1837**	LC	x	x		
Family	Passerellidae Cabanis & Heine, 1850					
	Ammodramus aurifrons (Spix, 1825)	LC	x	x	x	x
	Arremon taciturnus (Hermann, 1783)	LC			x	x
Family	Icteridae Vigors, 1825					
	Psarocolius decumanus (Pallas, 1769)	LC			x	x

	Psarocolius viridis (Statius Muller, 1776)	LC	x	x		
	Cacicus cela (Linnaeus, 1758)	LC	x	x	x	
	Icterus cayanensis (Linnaeus, 1766)	LC			x	x
Family	Thraupidae Cabanis, 1847					
	Nemosia pileata (Boddaert, 1783)	LC	x		x	x
	Hemithraupis guira (Linnaeus, 1766)	LC				x
	Dacnis cayana (Linnaeus, 1766)	LC			x	
	Saltator coerulescens Vieillot, 1817	LC	x	x	x	x
	Coereba flaveola (Linnaeus, 1758)	LC	x		x	x
	***Eucometis penicillata* (Spix, 1825)**	LC	X			
	Ramphocelus carbo (Pallas, 1764)	LC	x	x	x	x
	Sporophila minuta (Linnaeus, 1758)	LC				x
	Sporophila castaneiventris Cabanis, 1849	LC		x		
	Conirostrum speciosum (Temminck, 1824)	LC			x	
	***Conirostrum margaritae* (Holt, 1931)**	VU	x			
	Tangara episcopus (Linnaeus, 1766)	LC	x	x		x
	Tangara palmarum (Wied, 1821)	LC	x	x	x	x
	Tangara mexicana Linnaeus1766	LC		x		

Table S4: Estimates of species richness for different groups of sites

		Estimators		
	Bootstrap	**Chao**	**Jackknife**	**Species Richness observed**
General	217.58 ± 5.36	224.23 ± 10.46	232.11 ± 8.13	**202**
Madeira River	158.97 ± 5.69	168.68 ± 11.38	173.42 ± 9.25	**145**
Xingu River	132.38 ± 4.96	137.06 ± 3.01	138.95 ± 5.31	**130**
Impacted Madeira	109.61 ± 6.27	107.96 ± 5.19	118.55 ± 8.92	**99**
Undisturbed Madeira	116.37 ± 5.31	132.34 ± 14.15	129.3 ± 9.5	**105**
Impacted Xingu	98.42 ± 7.49	98.61 ± 5.99	109.4 ± 11.74	**87**
Undisturbed Xingu	103.84 ± 4.69	100.06 ± 3.54	105.7 ± 5.76	**95**

Table S5: GLMM results for all species, without aquatic species, and floodplain specialist. Results show the best model (i.e.: alternative or null model) including AIC Weight, estimate, t-value, and p-value for each subset. Values for metrics that showed significant differences are represented by the p-value in bold. AIC values less than 2 indicate no difference from the null model (represented by the asterisk)

		Weight	Intercept	Undisturbed	STD. Error	t value	p value
All birds 202 species included	Species Richness (SR)	0.91	38.107	-4.720	2.475	-1.907	0.066
	Phylogenetic Diversity (PD)	1.00	1567.930	-265.350	82.020	-3.235	**0.003**
	Mean Pairwise Distance (MPD)	0.90	143.206	-4.191	2.201	-1.904	0.066
	Mean Nearest Taxon Distance (MNTD)	0.98	59.409	-6.772	2.352	-2.879	**0.007**
	Functional Richness (FRic)	1.00	0.156	-0.124	0.016	-7.610	**1.400e-08**
	Functional Dispersion (FDis)	1.00	0.697	-0.118	0.016	-7.204	**4.223e-08**
Without aquatic species 182 species included	Species Richness (SR) *	0.64	32.863	0.196	2.514	0.078	0.938
	Phylogenetic Diversity (PD)	0.68	1325.580	-43.400	84.100	-0.516	0.609
	Mean Pairwise Distance (MPD) *	0.63	139.745	-0.981	2.178	-0.450	0.656
	Mean Nearest Taxon Distance (MNTD)	0.97	58.259	-6.406	2.581	-2.482	**0.019**
	Functional Richness (FRic)	0.81	0.103	-0.057	0.015	-3.832	**5.81e-04**
	Functional Dispersion (FDis)	1.00	0.613	-0.070	0.012	-5.800	**2.17e-06**
Floodplain specialists 46 species included	Species Richness (SR) – null model	0.53	9.211	0.623	1.095	0.569	0.573
	Phylogenetic Diversity (PD)	0.67	532.069	-0.369	50.782	-0.007	0.994
	Mean Pairwise Distance (MPD) – null model	0.76	135.547	0.3058	4.665	0.066	0.948
	Mean Nearest Taxon Distance (MNTD)	0.98	92.391	-12.504	5.435	-2.30	**0.028**
	Functional Richness (FRic) – null model	0.78	0.030	-0.022	0.021	-1.071	0.092
	Functional Dispersion (FDis)	0.76	0.401	-0.034	0.025	-1.363	0.083

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