UNIVERSIDADE FEDERAL DO PARÁ
INSTITUTO DE CIÊNCIAS BIOLÓGICAS
EMBRAPA AMAZÔNIA ORIENTAL
PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA

GILBERTO NEPOMUCENO SALVADOR

Barragens de rejeito e seus efeitos sobre a assembleia de peixes

Belém
2019

GILBERTO NEPOMUCENO SALVADOR

Barragens de rejeito e seus efeitos sobre a assembleia de peixes

Tese apresentada ao Programa de Pós-Graduação em Ecologia do convênio da Universidade Federal do Pará e Embrapa Amazônia Oriental, como requisito parcial para obtenção do título de Doutor em Ecologia. Área de concentração: Ecologia.
Linha de Pesquisa: Ecologia de Comunidades e Ecossistemas

Orientador: Dr. Raphael Ligeiro B. Santos (ICB/UFPA)
Co-orientador: Dr. Luciano F. de A. Montag (ICB/UFPA)

Belém
2019

Dados Internacionais de Catalogação na Publicação (CIP) de acordo com ISBD
Sistema de Bibliotecas da Universidade Federal do Pará
Gerada automaticamente pelo módulo Ficat, mediante os dados fornecidos pelo(a) autor(a)

S182b Salvador, Gilberto Nepomuceno.
Barragens de rejeito e seus efeitos sobre a assembleia de peixes / Gilberto Nepomuceno Salvador. — 2019.
xii, 101 f. : il. color.

Orientador(a): Prof. Dr. Raphael Ligeiro
Tese (Doutorado) - Universidade Federal do Pará, Instituto de Ciências Biológicas, Programa de Pós-Graduação em Ecologia, Belém, 2019.

1. Ictiofauna. 2. conservação. 3. mineração. 4. alterações ambientais. I. Título.

CDD 574.524

GILBERO NEPOMUCENO SALVADOR

Barragens de rejeito e seus efeitos sobre a assembleia de peixes

Tese apresentada ao Programa de Pós-Graduação em Ecologia do convênio da Universidade Federal do Pará e Embrapa Amazônia Oriental, como requisito parcial para obtenção do título de Doutor em Ecologia pela Comissão Julgadora composta pelos membros:

COMISSÃO JULGADORA

Prof. Dr. Raphael Barroso dos Santos Ligeiro
Universidade Federal do Pará (Presidente)

Profª. Drª. Bárbara Dunck Oliveira
Universidade Federal do Pará

Profª. Drª. Caroline Chaves Arantes
Texas A&M University, Wildlife and Fisheries Sciences

Prof. Dr. Fábio Cop Ferreira
Universidade Federal de São Paulo

Profª. Drª. Lilian Casatti
Universidade Estadual Paulista

Prof. Dr. Miguel Petrere Júnior
Universidade Federal do Pará

Prof. Dr. Paulo dos Santos Pompeu
Universidade Federal de Lavras

Prof. Dr. Tiago Octavio Begot Ruffeil
Universidade Federal do Pará

Aprovada em:
Local da defesa:

Dedico esse trabalho a todos aqueles
pescadores que, um dia, do rio tiraram
seu sustento, mas hoje tiram lembranças
e histórias de um passado recente

AGRADECIMENTOS

Quantas foram as pessoas que me ajudaram nesse caminho? Muitas. Mas algumas tiveram papel preponderante. O que dizer dos meus orientadores, Raphael Ligeiro e Luciano Montag (Miúdo)? Os dois me aceitaram de braços abertos, com a ideia inicial de trabalhar com Silvicultura na Amazônia. Tiveram paciência ao perceber que ainda havia em mim muito do mundo da consultoria, onde trabalhei até o dia que parti para Belém. Acreditaram quando parti para Macapá atrás de financiamento para o projeto de doutorado. Cobraram quando tinham que cobrar e me ajudaram quando senti necessidade. Outras pessoas também tiveram suma importância, como a Renata Frederico, que virou mais que companheira científica no meio do caminho e hoje estamos juntos, na ciência e na vida. Não há como esquecer do companheiro de trabalho e publicações Tiago Casarim (famoso Yoshi), e da dupla Guilherme Dutra e Bruno Prudente, companheiros de república e de discussões científicas. Falando no Dutra, devo a ele também o empurrão inicial desse doutorado. Há momentos em que a vida parece andar mais lenta e é necessário um empurrão pra ela voltar a acelerar. Esse empurrão, mesmo que pequeno, foi teu Dutra. A chegada em Belém foi pela rodoviária. Desci do ônibus para sentir o clima amazônico mais uma vez, mas dessa vez para morar. Seriam alguns anos morando na Sacramenta, escutando por osmose Bufalo Negro do Marajó. Faltava-me o queijo, mas sobravam novas amizades e os sabores da culinária paraense (e viva o Ver-o-Peso). Tiago Freitas, Ana Luiza, Naraiana Benone (que só fui descobrir que o nome era Naraiana lá pelo segundo ano), Tiago Begot, Thiago Barbosa, Cleo Lobato, Calebe Maia, e tantos outros do LABECO. Pessoal bom de trabalho e de prosa. Alguns viraram parceiros de publicações, outros parceiros de ideias e conversa. Viraram uma nova família. Falando nela, não poderia esquecê-la. As mães, o pai, e os irmãos Zé da rabeca, Loius Jasperito e Andret (e as respectivas de cada um). Por coincidência, um dos meus irmãos foi convocado para o 2º Batalhão de Infantaria de Selva em Belém. Quando chegaram, logo uns meses depois de mim, não sabíamos que a vida nos colocaria um presente e tantas tarefas – Lelê, guerreira, hoje sorri com os olhos mais lindos. No meio do caminho, aquele plano de trabalhar com silvicultura mudou, afinal, assistimos a um dos maiores desastres ambientais do país e tínhamos informações pretéritas. Carlucci foi quem percebeu essa possibilidade, e a ele agradeço esse novo empurrão. Quantos outros teria, eu, que agradecer? Muitos! Bob Hughes, Gustavo Ribeiro, Guilherme Santos, Leandro Brasil, Gabi ecóloga e confeiteira, Gabilichthys... Muitos. Hei de agradecer também aqueles que foram meus professores. Foram tantas matérias que fiz. À época de PUC me faltou um bom professor de ecologia. Foi em Belém que pude adentrar com mais ênfase nesse mundo que tanto gosto. Não poderia de esquecer o Rafa Leitão, que abriu as portas do seu laboratório para dar continuidade a algumas ideias. Agradecer ao pessoal que fez o curso Internacional de campo de Caxiuanã, especialmente à Híngara, à esquimó (Julie) e ao Cristian. Enfim, nada disso seria possível sem a bolsa de estudos provida pela CAPES e pela universidade pública, gratuita e de qualidade.

Separadamente, agradeço ao fotógrafo Felipe Werneck do IBAMA, por prover a imagem que ilustra a Sessão 3 dessa tese (Creative Communs: www.flickr.com/photos/ibamagov/32132222787).

O Rio? É doce.
A Vale? Amarga.
Ai, antes fosse
Mais leve a carga.

Entre estatais
E multinacionais,
Quantos ais!

A dívida interna.
A dívida externa
A dívida eterna.

Quantas toneladas exportamos
De ferro?
Quantas lágrimas disfarçamos
Sem berro?

Carlos Drummond de Andrade

Barragens de rejeito e seus efeitos sobre a assembleia de peixes

RESUMO

Considerada um dos principais impactos antrópicos sobre o meio ambiente, a mineração apresenta uma séria de atividades que vão desde a extração até o beneficiamento do minério. Nesse processo, as barragens de rejeito são construídas para estocar a parte não comercial do minério. Impactos ocasionados por esse tipo de empreendimento vão desde o seu rompimento, afetando rios inteiros, até impactos mais locais, como a fragmentação e a transformação de um ambiente lótico em lêntico. No presente estudo, abordaremos os impactos ocasionados por barragens de rejeito, tanto em uma escala de bacia, quanto em riachos localizados a montante dessas. Para isso, estruturamos essa tese em três sessões, sendo a primeira sessão relacionando o rompimento de uma barragem de rejeitos no município de Mariana, sobre a assembleia de peixes do rio Doce, a segunda sessão traz um crítica sobre os motivos que levaram ao novo desastre, dessa vez em Brumadinho, e a última sessão sobre às alterações ocasionadas pela implementação dessas estruturas nos riachos que deságuam no corpo do reservatório. Os estudos foram realizados em três áreas, sendo duas na bacia do rio Doce (Sessões 1 e 3), e uma na bacia do rio São Francisco (Sessão 2), sudeste do Brasil. Para a primeira sessão, coletas empregando redes de emalhar foram realizadas entre os anos de 2006 e 2018. Os tratamentos temporais foram definidos levando-se em consideração grandes alterações ambientais na bacia, como a construção da uma hidrelétrica e o rompimento da barragem de rejeitos em Mariana. Para a segunda sessão, a área de estudo foi definida como a bacia do rio Paraopeba, um dos principais afluentes do rio São Francisco. Nessa sessão, levantamos os porquês do rompimento de mais uma barragem. Já as coletas da sessão três ocorreram em uma área com grande concentração de minas de ferro. Foram selecionados dois tratamentos: riachos que deságuam em reservatórios de barragens de rejeito e riachos livres desse impacto, considerados tratamento controle. Ao final, conseguimos entender como esses empreendimentos afetam a ictiofauna, facilitando o diálogo com a conservação das espécies aquáticas em áreas afetadas diretamente pela atividade de mineração.

Palavras-chave: Ictiofauna; conservação; mineração; alterações ambientais

Tailing dams and their effects on the fish assemblage

ABSTRACT

Considered one of the main anthropic impacts on the environments, mining has a series of activities ranging from extraction to ore beneficiation. Along this process, tailings dams are built to stock the non-commercial part of the ore. Impacts caused by this type of structure range from its rupture, affecting whole rivers, to local impacts, such as the fragmentation and transformation of a lotic environment into a lentic one. In the present study, we will cover the impacts caused by tailings dams, both on a basin scale and on streams which flow into to the pounds. For this, we structured this thesis in three sessions, the first session relating the disruption of a tailings dam in the municipality of Mariana, on the Doce river fish assembly, the second session brings a critique about the reasons that led to a new disaster, now in Brumadinho, and the last session on the changes caused by the impact of those structures in the streams that drain into the tailings pounds. The studies were carried out in three areas, two in the Doce river basin (Sessions 1 and 3), and one in the São Francisco river basin (Session 2), southeast of Brazil. For the first session, collections using gillnets were carried out between the years 2006 and 2018. The temporal treatments were defined taking into account large environmental changes in the basin, such as the construction of a hydroelectric plant and the rupture of the tailings dam in Mariana. For the second session, the study area was defined as the Paraopeba river basin, one of the main tributaries of the São Francisco river. In this session, we raised the reasons for the rupture of one more dam. Already the collections of the session three occurred in an area with great concentration of iron mines. Two treatments were selected: streams which flow for tailings pounds and streams free of this impact, considered control treatment. In the end, we can understand how these ventures affect the ichthyofauna, facilitating the dialogue with the conservation of the aquatic species in areas directly affected by the mining activity.

***Keywords*:** Ichthyofauna; conservation; mining; environmental changes

SUMÁRIO

ÍNDICE DE FIGURAS

ÍNDICE DE TABELAS

ÍNDICE DE MATERIAL SUPLEMENTAR

1. INTRODUÇÃO GERAL

Os metais são uma classe de elementos químicos com inúmeras propriedades uteis, como força, maleabilidade e condutividade ao calor e elétrica (Hudson et al., 1999). O processo de obtenção desses elementos ocorre através da mineração, e ela foi um dos pilares para o desenvolvimento da humanidade (ICMM, 2012). Esse processo "compreende um conjunto de atividades destinadas a pesquisar, descobrir, mensurar, extrair, tratar ou beneficiar e transformar recursos minerais de forma a torná-los benefícios econômicos e sociais" (IBRAM, 2016). Uma mina pode ser vista como um mosaico de atividades que englobam a extração do minério na região cava, o beneficiamento desse na área de usinagem e a estocagem da porção não comercial e do estéril em barragens de rejeitos e pilhas de estéril, respectivamente (Figura 1-1).

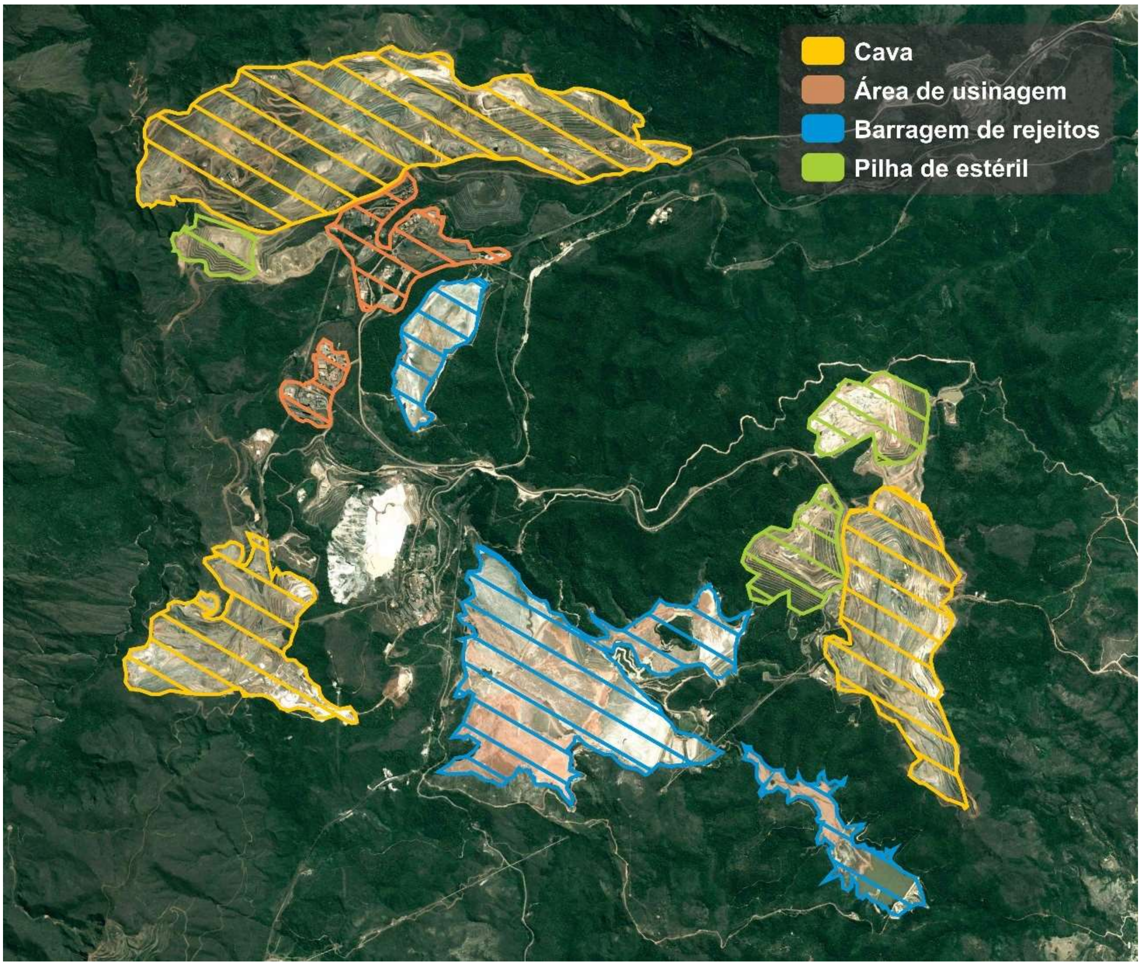


Figura 1-1. Exemplo de complexo minerário localizado no estado de Minas Gerais mostrando o mosaico de atividades que podem ser observadas em uma área de extração de minério de Ferro, como a cava, área de usinagem, barragem de rejeitos e pilha de estéril.

O minério de ferro é um dos principais produtos da mineração e a América Latina se destaca em sua produção (WEF, 2010). Ele é quase que exclusivamente extraído em operações na superfície

(U.S. DOI/ Bureau of Mines, 1983). Após o beneficiamento, a parte não comercial é separada do minério (Kossoff et al., 2014) e uma das vias de armazenamento desse material ocorre em barragens (Espósito and Duarte, 2010). No Brasil, os dois principais estados produtores desse mineral são Minas Gerais e Pará (IBRAM, 2011), e a região do quadrilátero ferrífero se destaca nesse cenário (Roeser and Roeser, 2010). O Quadrilátero Ferrífero é uma estrutura geológica de aproximadamente 7.200 km² e que apresenta uma grande heterogeneidade de paisagens (Jacobi and Carmo, 2008). Possui diversos complexos para exploração mineral, sendo o minério de ferro o principal material explorado (Figura 1-2).

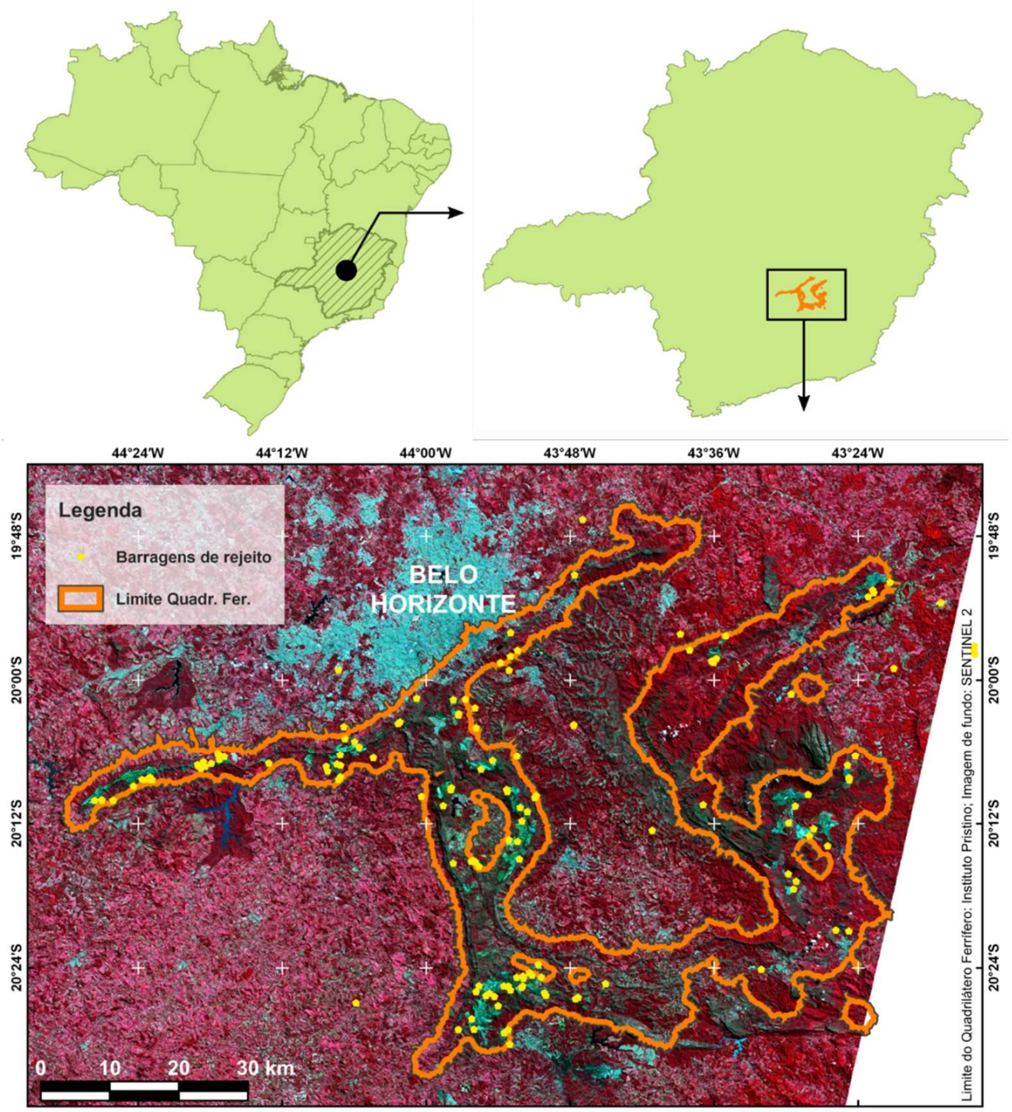


Figura 1-2. Localização do Quadrilátero Ferrífero em relação ao Brasil, mostrando seus limites e uso de solo (imagem SENTINEL 2 – NDVI), bem como suas barragens de rejeito (fonte: Agência Nacional de Mineração).

Apesar dos benefícios econômicos produzidos pela mineração, ela tem papel preponderante na degradação ambiental, incluindo nos sistemas aquáticos (Earle et al., 2018; Hughes et al., 2016).

Um dos distúrbios dessa atividade sobre o meio ambiente está na utilização das barragens de rejeito como forma de estocar a parte não comercial do minério (Espósito and Duarte, 2010). Somente no Brasil há registro de 600 barragens para essa finalidade, muitas delas instaladas em riachos do QF (ANA, 2016) (Figura 1-2). Seu principal impacto está relacionado ao rompimento dessas estruturas, e apesar de pouco noticiado ocorrem com dada frequência. São decorrentes de erros de engenharia e de inspeção de segurança, falha nas fundações do barramento e chuvas atípicas (Rico et al., 2008).

Mesmo havendo casos precedentes de ruptura de barragens de rejeito no Brasil (Abdelouas, 2019; Monteiro et al., 2014), somente após o rompimento da barragem de rejeito de Fundão, no município de Mariana, MG, que esse tipo de impacto passou a chamar atenção pública no Brasil. O rompimento e lançamento de 30 milhões de m^3 de rejeito de minério de ferro no dia 5 de novembro de 2015 na bacia do rio Doce foi considerado o pior desastre ambiental brasileiro (Escobar, 2015; Fernandes et al., 2016). A contaminação das águas utilizadas para abastecimento por diversas cidades ao longo dos rios afetados, a impressionante devastação ocorrida no trecho alto, a extensa mortandade de peixes e a possibilidade de contaminação de um dos principais sistemas marinhos brasileiros (Escobar, 2015) chocou uma nação. Dessa forma, se tornou essencial entender a magnitude e os impactos desse desastre sobre a fauna aquática. Somente com esse entendimento será possível montar um plano de recuperação ambiental e propostas de manejo para a bacia.

A magnitude do desastre ambiental causado pelo rompimento da barragem de Fundão parecia colocá-lo como algo único, que não se repetiria no Brasil. Planos de segurança de barragens de rejeito começaram a ser desenhados. Porém, mesmo diante de um desastre de tamanha magnitude como foi Fundão, a velocidade de atuação das empresas e do governo brasileiro para a não repetição desses acontecimentos foi consideravelmente lenta (Oliveira and Oliveira, 2019). Aquilo que parecia distante, impossível, aconteceu, um novo desastre. O rompimento da barragem B1 na mina de Córrego do Feijão, em Brumadinho, MG, lançou grande parte dos 12 milhões de m^3 de rejeito de minério de ferro no rio Paraopeba. Apesar de serem os impactos ambientais decorrentes desses desastres os mais proeminentes e agudos, também existem os impactos crônicos, menos visíveis aos olhos da opinião pública.

As barragens de rejeito apresentam impactos sobre o riacho semelhantes aqueles observados para grandes reservatórios. Elas podem causar a fragmentação e alterações físico-químicas na água em decorrência da transformação do ambiente lótico em lêntico (Agostinho et al., 2016, 2007). Supõe-se que tais transformações atuem de forma negativa sobre a fauna de peixes desses riachos, selecionando espécies generalistas em detrimento daquelas comumente encontradas em riachos. É interessante ponderar que o tipo de material estocado nas barragens pode ser determinante nessa transformação da ictiofauna, uma vez que os componentes químicos resultantes do beneficiamento de minério podem alterar ainda mais as características físico-químicas da água. Essa fragmentação, somada as substituições da fauna aquática no corpo do reservatório podem afetar as comunidades isoladas a montante desses barramentos (Brosse et al., 2011).

Diante disso, faz-se pertinente entender como as alterações ocasionadas por uma barragem de rejeitos pode afetar a fauna aquática, em especial a ictiofauna. Para isso, trabalharemos em duas frentes. A primeira voltada para os impactos do rompimento de barragens de rejeito, com um estudo de caso sobre o rompimento da barragem de Mariana (Sessão 2), e uma opinião sobre os caminhos que levaram ao rompimento da barragem de Brumadinho (Sessão 3). A segunda terá como objetivo estudar os efeitos de barragens de rejeito na assembleia de peixes em riachos localizados à montante de seus respectivos reservatórios (Sessão 4).

MATERIAL E MÉTODOS

Área de Estudo

O estudo foi realizado em duas bacias hidrográficas limítrofes, a do rio Doce e a do rio São Francisco (Figura 1-3). A bacia do rio Doce faz parte da ecorregião "Northeastern Mata Atlantica" (nº. 328) (Abell et al., 2008), e pode ser considerada uma das principais bacias dos estados de Minas Gerais e Espírito Santo, Sudeste do Brasil, drenando uma área de aproximadamente 83.400 km² (Vieira, 2009). Desde a confluência dos rios Carmo e Piranga, local onde passa a receber o nome de rio Doce, o rio percorre aproximadamente 600 km até sua foz no oceano atlântico, no município de Regência, Espírito Santo. Nesse percurso, recebe as águas de grandes afluentes, como os rios Piracicaba, Santo Antônio, Corrente Grande, Suaçuí-Grande, Guandu, Manhuaçu, Caratinga-Cuieté, Matipó e Casca. Pode ser dividida em três seções, sendo o trecho alto compreendido entre a nascente e a foz do rio Matipó, o médio entre a foz do Matipó e a divisa entre os estados do Espirito Santo e Minas Gerais, e o baixo entre essa divisa e o oceano Atlântico, onde o rio desagua em Regência/ES (Vieira, 2009). A sessão 2 foi amostrada no trecho médio da bacia do rio Doce, onde, entre os anos de 2007 e 2009, foi instalada uma hidrelétrica cujo reservatório se estende desde o município de Pedra Corrida/MG, até Governador Valadares/MG. A área de influência dessa hidrelétrica foi intensamente afetada pelo rompimento da barragem de rejeitos de Fundão (Escobar, 2015; Neves et al., 2016).

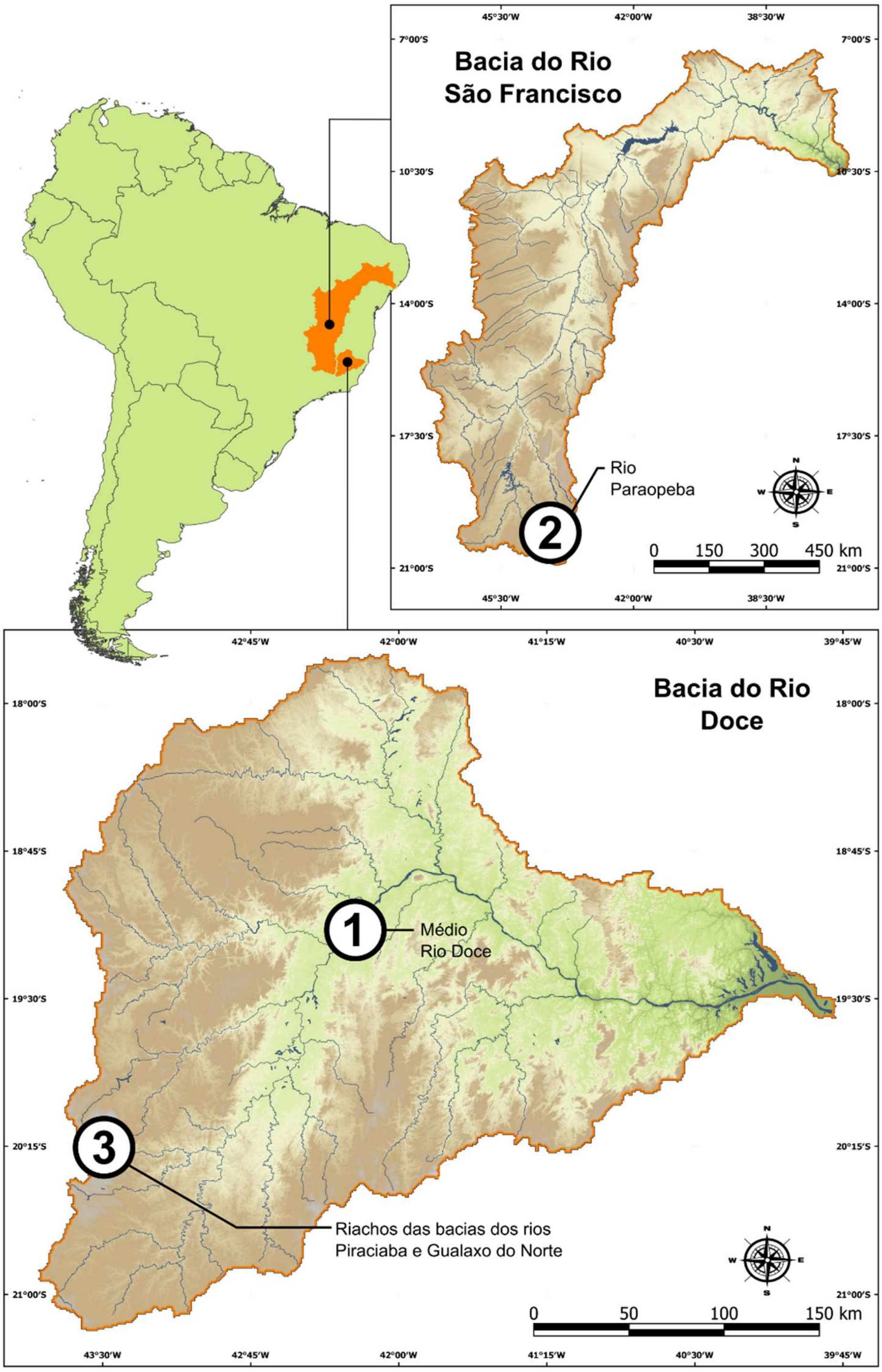


Figura 1-3. Localização espacial das bacias dos rios Doce e São Francisco em um contexto da América do Sul, e das áreas onde foram realizadas as amostragens (1/2) em relação à bacia dos rios Doce e São Francisco. Área 1: Área referente ao capítulo 1, sendo localizada no trecho médio do rio Doce, entre a confluência dos rios Santo Antônio e Suaçuí Grande. Área 2: Rio Paraopeba, desde a confluência do Cór. Do Feijão até a hidrelétrica de Três Marias. Área 3: Trecho alto da bacia do rio Doce, nas cabeceiras dos rios Gualaxo do Norte e Piracicaba.

No trecho alto das bacias dos rios Doce e São Francisco está localizado o quadrilátero ferrífero, com sua parte direita drenando para afluentes do rio Doce, e sua parte esquerda para afluentes do São Francisco. Como consequência do grande número de minas instaladas ao longo de sua área, também existem diversas barragens de rejeito (Figura 1-2). A sessão 3 se refere ao rompimento da barragem B1 no complexo minerário do Feijão. Essa mina drena para o rio Paraopeba, um dos principais afluentes do trecho alto da bacia do rio São Francisco (Alves and Leal, 2010).

A sessão 4 foi realizada em riachos que drenam a parte do Doce do QF. Drenam as cabeceiras dos rios Gualaxo do Norte e Piracicaba entre as cidades de Mariana e Itabira, MG. Eles apresentam um elevado declive, principalmente aqueles localizados ao sopé da serra do Caraça, um maciço rochoso com picos de mais de 2000 metros de altitude (Canelas and Bertoluci, 2007). Nessa região estão localizadas duas importantes unidades de conservação, o Parque Nacional do Gandarela e a Reserva Particular do Patrimônio Natural do Caraça.

2. Sessão 1

Influences of multiple anthropogenic disturbances coupled with a tailings dam rupture on spatiotemporal variation in fish assemblages of a tropical river

A primeira sessão desta tese foi publicada no periódico científico *Freshwater Biology*, disponível em: https://onlinelibrary.wiley.com/doi/abs/10.1111/fwb.13967

INFLUENCES OF MULTIPLE ANTHROPOGENIC DISTURBANCES COUPLED WITH A TAILINGS DAM RUPTURE ON SPATIOTEMPORAL VARIATION IN FISH ASSEMBLAGES OF A TROPICAL RIVER

ABSTRACT

The Mariana disaster, resulting from the rupture of the Fundão tailings storage facility (TSF) in 2015, is a large-scale environmental disaster, and it drastically affected the entire channel of the Rio Doce, southeastern Brazil. The disaster produced substantial channel siltation, increased turbidity, riparian damage, and a massive fish kill. However, the Rio Doce basin has a long history of environmental degradation, including hydroelectric dams. The latter are major threats to the ichthyofauna because they affect flow regimes, trap sediments and large wood, select generalist species, impair fish migrations, and favor non-native invasive species. In this study, we analyzed the effects of the rupture as well as impacts from the construction of a run-of-the-river hydroelectric dam on native and non-native fish abundances and species richness and composition in the middle Rio Doce. To do this, we examined fish survey data collected before and after each disturbance upriver of the reservoir, in the reservoir area, and downriver of the dam. The data were separated into three temporal intervals: free flowing river, post-filling reservoir, post-TSF disaster. To assess temporal changes in native and non-native fish species richness we used a rarefaction method and tested it with two-way ANOVA, which was also used to compare fish abundances. To test temporal changes in fish assemblage composition we used a PERMANOVA test. We observed no significant differences in total native or non-native species richness as a result of the dam or the disaster. However, we did observe increased non-native individual abundances after the TSF rupture as well as native individual abundances, depending on the location studied. The assemblage composition also changed after dam construction and the TSF rupture, except for the upriver reach after dam construction. We conclude that both the dam and the TSF rupture affected fish assemblages, favoring increased non-native abundances and also changed fish assemblage composition. Our results suggest that a century of landscape and riverscape pressures had extirpated the sensitive fishes from the Rio Doce prior to the TSF rupture. We conclude that multiple long-term chronic disturbances can be as harmful to aquatic biota as the acute effects of a major environmental disaster.The 2015 Mariana disaster resulting from the rupture of the Fundao Tailings Dam is considered the largest in Brazil and it drastically affected the entire channel of the Doce River. The disaster produced substantial channel siltation, turbidity, riparian damage, and a massive fish kill. In this study, we analyzed the temporal and spatial effects of the rupture as well as a run-of-the river hydroelectric dam in the middle Doce River through use of seasonal fish assemblage collections from 2006 to 2018 at pairs of sites upriver of the reservoir, in the reservoir area, and downriver of the dam. We observed no significant differences in total, native, or non-native species richness as a result of the dam or the disaster. However, we did observe differences in fish assemblage abundances and composition. We conclude that assessments based on entire fish assemblages are more sensitive than those based on fish species richness, that gillnetting misses some small benthic fishes, and that a century of previous landscape and riverscape pressures had already led to extirpation of sensitive fishes from the Doce River.

Keywords: Fish communities, hydroelectric dam, mining activities, Neotropical Savanna, environmental disaster

INTRODUCTION

Mining activities are major sources of environmental disasters, either from chronic leakages (Woody *et al.*, 2010; Daniel *et al.*, 2015) or catastrophic failures (Luino & de Graff, 2012; Escobar, 2015). One of the most frequent catastrophes is the rupture of earthen tailings storage facilities (TSF) where non-commercial residues resulting from ore beneficiation are retained (Espósito & Duarte, 2010; Bowker & Chambers, 2017). Although not widely reported, TSF disasters are routine around the world, and their number is growing every year (Bowker & Chambers, 2017). Between 1915 and 2015, 308 such failures were reported globally (Carmo *et al.*, 2017), an average of three failures per year. In Brazil, 10 such failures were reported in the last 30 years, including six in the last six years (Salvador *et al.*, 2020). The immediate causes of these failures vary, but usually include inappropriate engineering, incorrect management and inspections, dam foundation failures, and atypical weather events (Rico, Benito & Díez-Herrero, 2008). The great majority of these failures occur at mines still in operation, and high metal prices lead to increased ore mining and processing and increased tailings storage without due care (Rico *et al.*, 2008; Bowker & Chambers, 2017). Following dam ruptures, the release of tailings to downstream waters jeopardizes water quality and physical habitat structure, with direct effects on aquatic biodiversity and human health. Those effects include species losses and changes in fish abundances and assemblage composition (Swales *et al.*, 1998; Daniel *et al.*, 2015). Another consequence of these disasters is long-term metal contamination of sediments, riparian vegetation, and aquatic fauna (Swales *et al.*, 1998; Hatje *et al.*, 2017).

In 2015, the Fundão TSF ruptured in southeastern Brazil, causing a large-scale environmental disaster. The direct cause of the failure was an improper axis retreat of the dam foundation and drainage problems, leading to dam collapse (Carmo *et al.*, 2017). The dam contained 63 million m^3 of iron ore tailings, of which 36 to 43 million m^3 spilled into the Rio Gualaxo do Norte (Morgenstern *et al.*, 2016; Carmo *et al.*, 2017). The tailings then flowed into the Rio Doce and traveled 633 km until reaching the Atlantic Ocean, from where they dispersed by tens of kilometers (Marta-Almeida *et al.*, 2016; Rudorff *et al.*, 2018). The disaster left 19 people dead, hundreds homeless, and directly affected the health and livelihoods of thousands of people (Escobar, 2015). The spill produced ferric oxide flocs on channel substrates, silted the Rio Doce channel, contaminated the river with metals (Hatje *et al.*, 2017) and produced a massive fish kill (Neves *et al.*, 2016; Grilo *et al.*, 2018). It also destroyed 457 ha of Atlantic Forest, a global biodiversity hotspot (Myers *et al.*, 2000; Omachi *et al.*, 2018).

Prior to the catastrophic spill of iron ore tailings into the Rio Doce, the aquatic ecosystems of this basin had experienced a long history of environmental degradation, which negatively affected their biodiversity and ecological processes from species to ecosystem levels (Vieira, 2009). In the first half of the 20th century, much of the basin was deforested for timber and agriculture, leaving only 27% of native forest cover (Espíndola *et al.*, 2016). The basin also suffers from insufficient sewage treatment, fragmentation by multiple hydropower dams, and widespread mining and non-native species introductions (Alves *et al.*, 2007; Vieira, 2009; FEAM, 2015).

Perhaps the major impact on fish assemblages in the Rio Doce basin has been the construction of multiple hydroelectric dams. Considered the main renewable energy source in the world (Matarazzo & Sgandurra, 2018), hydroelectric plants also are considered major anthropogenic pressures on fish faunas (Agostinho *et al.*, 2016; Winemiller *et al.*, 2016). The creation of an artificial lentic-like environment homogenizes the river ecosystem and changes its water quality, decreasing turbidity and dissolved oxygen, among other parameters (Dendy & Cooper, 1984; Agostinho, Gomes & Pelicice, 2007). Such changes can lead to reduced fish species richness and changes in fish assemblage composition, benefiting omnivorous, tolerant and non-native species (Agostinho *et al.*, 2007; Muniz et al., 2021). Fish assemblages also change in downriver sections following dam creation because of regulated flows, diminished flood pulses, and daily freshets (Agostinho *et al.*, 2007; Buenaga *et al.*, 2017). Dams and reservoirs act as barriers for upstream and downstream migratory species, reducing the stocks of those species (Pelicice & Agostinho, 2008; Pompeu, Agostinho & Pelicice, 2012). Although there are often accumulations of fish just downstream of the dam (increasing local species richness), those fishes are often more intensively fished at such sites by humans (Phelps *et al.*, 2016). The less-disturbed, upriver sections typically show few alterations in fish assemblages, except as an indirect result of the proliferation of lentic species from the reservoirs (Agostinho *et al.*, 2007).

Although reservoirs often affect native species negatively, they tend to favor non-native fishes (Agostinho *et al.*, 1999; Blackburn *et al.,* 2011; Becker *et al*., 2016). The translocation of a species is the first step of a biological invasion. Subsequently, a species needs to break through a series of barriers to establish, reproduce, and disperse in the new environment (Blackburn *et al*., 2011). Reservoirs facilitate breaking many of these barriers for fish (Agostinho *et al*., 1999; Loures & Pompeu, 2018; Daga *et al*., 2020), leading to increased abundance of non-native fishes over time (Franco et al., 2018; Loures & Pompeu, 2018). The Rio Doce has 40 non-native species, and reservoirs facilitated the establishment of many of them, such as *Cichla kelberi* and *Oreochromis niloticus* (Bueno *et al*., 2021). However, the first introductions of fish species preceded all the mainstem Rio Doce dams, and they occurred to increase the basin's fisheries for such species as *Salminus brasiliensis*, *Pimelodus maculatus* and *Prochilodus costatus* (Ruschi, 1965; Vieira, 2009).

Therefore in this paper, we report the findings from a survey of the middle Rio Doce after two major impacts on its fish fauna, a run-of-river impoundment and the TSF rupture. Following Nimmo *et al.* (2015), the first one is categorized as a press disturbance and the second one as a pulse disturbance. To do so, we evaluated the effect of both disturbances on fish abundance and species richness and composition. Comparing the river before and after the dam construction , we made three predictions. 1) Native fish species richness would decrease after dam creation in the reservoir and downriver sections as a result of the creation of an artificial lentic-like environment and altered river flow regimes; but it would not change native fish species richness immediately upriver of the reservoir. 2) Non-native species would increase in abundance because of the environments provided by reservoirs for this group of fishes. 3) Fish assemblage composition would change after dam implementation, but not upstream of the reservoir. Comparing the period after the dam implemantation and before and after the TSF rupture, we added one more prediction. 4) The fish assemblage indicators would decrease in all three sections (upriver, reservoir, downriver) as a result of fish mortalities and physicochemical changes in the river mainstem.

METHODS

Study area

The Rio Doce basin is one of the most important hydrographic basins in the northeastern Mata Atlântica ecoregion (Abell *et al.*, 2008), draining an area of approximately 83,400 km² and flowing for approximately 600 km before reaching the Atlantic Ocean (Vieira, 2009). Along its course, it receives water from large tributaries, such as the Piracicaba, Santo Antônio, Corrente Grande, Suaçuí Grande and Manhuaçu. The basin is sub-divided into three reaches: upper, middle and lower Rio Doce (Vieira, 2009).

There are 40 hydroelectric power plants throughout the major tributaries of the Rio Doce. They include 10 large-sized hydropower plants (LHP, with gross production > 30 MW/h) and 30 small hydropower plants (SHP, with gross production < 30 MW/h) (Zambaldi & Pompeu, 2020). There are another 97 such plants in the planning phase (22 LHP and 75 SHP) in this basin. In the Rio Doce mainstem, four of the 11 planned power plants are already operating (ANEEL, 2010).

Our survey was restricted to the middle Rio Doce, which flows 315 km between the Rio Matipó confluence, in the state of Minas Gerais, to the border with the state of Espírito Santo (Vieira, 2009). Our sampling sites encompassed 50 km of the middle Rio Doce between the municipalities of Naque and Governador Valadares. Two large tributaries enter the Rio Doce in this reach: the Rio Santo Antônio and the Rio Corrente Grande. In 2010, a run-off-the-river hydroelectric power plant (Baguari) was built in the Rio Doce, flooding an area of 16 km^2 along 22 km of the Rio Doce, and 5 km of the Rio Corrente Grande. The dam is 25 m tall and 478 m wide, built just downriver of the Fumaça Waterfall, a sequence of rapids and cascades with a 9-m drop. In November, 2015, this river section was seriously altered by the Fundão tailings storage facilities (TSF) rupture located 300 km upriver (Figure 2-1).

Acquisition of environmental data

We obtained water quality data from the Hidroweb project of the Brazilian environmental agency, which measures flow and water quality parameters from the main rivers in Brazil (ANA, 2018). In the studied reach, data from one station was available, and it is located at the middle Rio Doce near the Governador Valadares municipality (station code: 56849000) and downstream of Baguari Dam. Water quality measurements began in 1997 and stopped in 2010. After the TSF rupture, measurements were made daily for several months in the first year and monthly subsequently. Four variables were measured during our study period: temperature (°C), pH, turbidity (NTU), and dissolved oxygen (DO, mg/L).

Ichthyofaunal sampling

Our fish survey was split into three different temporal phases. 1) A pre-dam phase included the period before dam construction from May 2006 to February 2007 (hereafter called “River”). 2) A reservoir phase included the reservoir post-filling period from November 2009 to August 2013 (hereafter called “Dam”). 3) The TSF rupture phase included the post-TSF rupture period from April 2016 to March 2018 (hereafter called “Rupture”). There were two periods not sampled, ranging from March 2007 to October 2009 and from September 2013 to February 2016. Because of the differing spatial effects of dams (Agostinho et al., 2007, 2016), the river reach studied was spatially divided into three sections that were each sampled during every visit: UPRIVER, located upriver of the hydroelectric reservoir, RESERVOIR, located in the reservoir, and DOWNRIVER, located downriver of the hydroelectric dam. We chose not to evaluate the longitudinal gradient in the RESERVOIR section, because the run-of-river Baguari Reservoir has very little slope, low water residence time, and little increment in sedimentation rates or water transparency (Thornton et al., 1981; Carneiro & Bini, 2020).

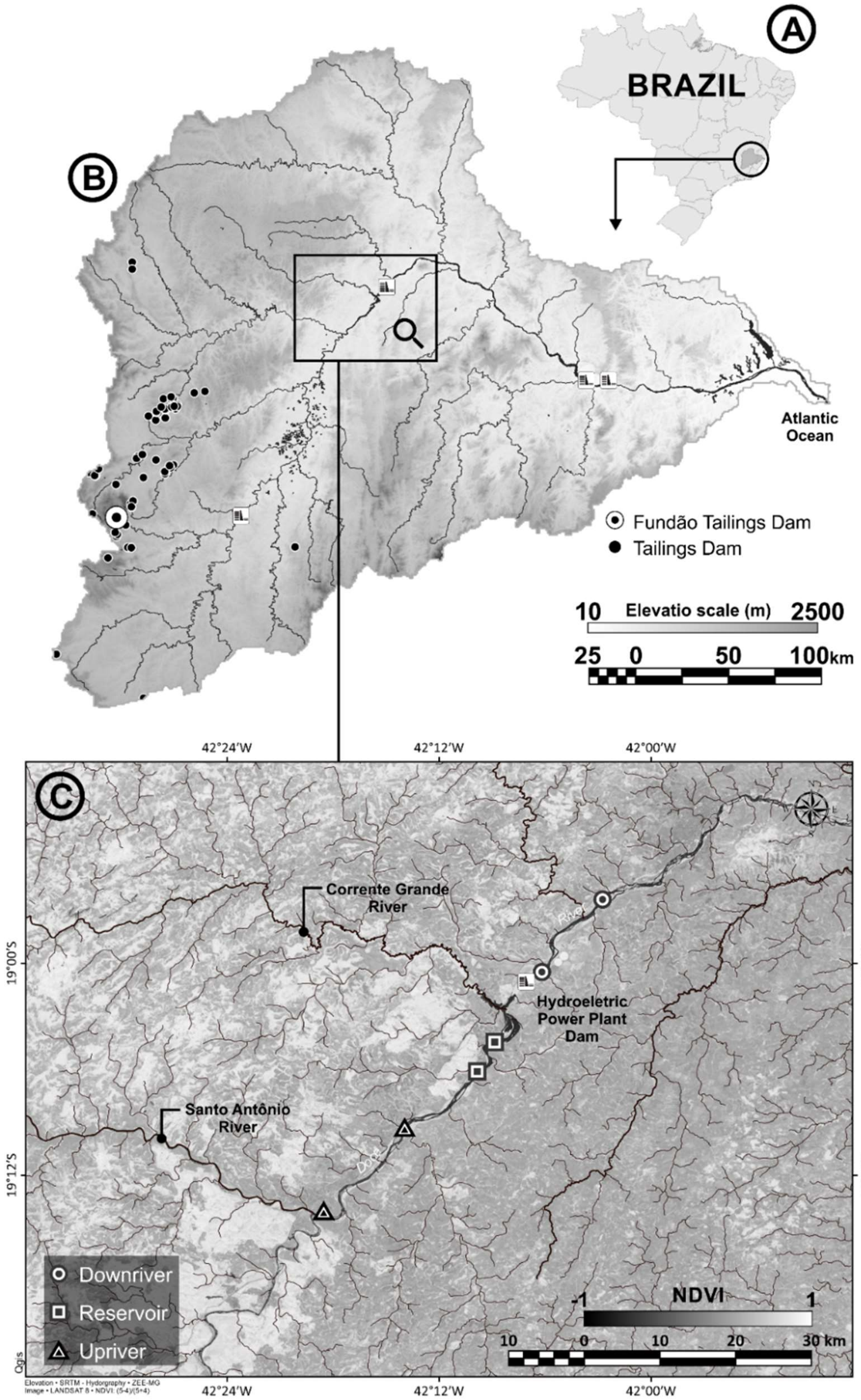


Figure 2-1. Locations of middle Doce River sampling sites. UPRIVER: sites located upriver of the hydroelectric reservoir; RESERVOIR: sites located in the run-of-river hydroelectric reservoir; DOWNRIVER: sites located downriver of the hydroelectric dam. Black represents water; dark-gray represents exposed soil, and white represents forest. It was not possible to distinguish between forest and forest plantation areas.

We sampled fish quarterly whenever conditions were safe, totaling 25 field visits (4 during River, 13 during Dam and 8 during Rupture phases). During each visit, we took three composite samples, each one coinciding with a section (UPRIVER, RESERVOIR and DOWNRIVER). We used two sets of gillnets per sample, 100 m long each (mesh sizes of 3, 4, 5, 6, 7, 8, 10, 12, 14, and 16 cm between opposite knots), varying from 1.5 to 1.8 m-high. The gillnets were fished overnight between 16:00 and 07:00 (15 hours). Although this method is widely used in the Neotropics (e.g., Pinto, Araujo & Hughes, 2006; Barbosa *et al.*, 2015; Sanches *et al.*, 2016), it is considered selective for more mobile larger-sized species and individuals (Kotwicki *et al.*, 2017). Collected specimens were fixed in 10% formalin and subsequently transferred to 70% alcohol. In the laboratory, fish species were identified to the lowest possible taxonomic level, usually species, and assigned as native or not for the basin following Viera (2009). Voucher specimens were deposited in five Brazilian ichthyological collections (Museu de Ciências Naturais da PUC Minas, Museu de Zoologia da UNICAMP, Museu Nacional – MNRJ, Coleção Ictiológica do Nupélia – UEM, and Museu de Biologia Professor Mello Leitão).

Data analyses

Water quality

We obtained 120 samples each of temperature, dissolved oxygen (DO), turbidity, and pH: 44 for the River phase, 8 for the Dam phase, and 68 for the Rupture phase from Hidroweb. We performed a principal component analysis (PCA) based on a Euclidean distance matrix using standardized data. This method is recommended to analyze metrics measured in different levels of resolution, and after standardization, data reflect the distance from the mean in units of standard deviation (Legendre & Legendre, 2012). To test for differences between phases (River, Dam, and Rupture), we used a permutational multivariate analysis of variance (PERMANOVA) (Anderson, 2001; Legendre & Legendre, 2012).

Fish assemblages

Seeking to understand the temporal variation in fish assemblages, the data from each section (UPRIVER, RESERVOIR and DOWNRIVER) were analyzed separately (Figure 2-1). We chose to test those data separately because the spatial effects of dams are already well documented in the literature (Agostinho *et al.*, 2007, 2016). Because abundance is affected by sampling effort (Harley, Myers & Dunn, 2001), abundance data were converted to catch per unit effort (CPUE), calculated by the number of fish caught in 100 m^2 of gillnets fished (100 m^2/15 h). To consider the abundance effect on the number of species collected, species richness for each site was assessed by applying an individual-based extrapolation method (n = 300) using the iNext package in R (Colwell *et al.*, 2012; Oksanen *et al.*, 2015; R Core Team, 2017). Because CPUE can be affected by seasonal influences, we assessed the difference in species richness between seasons (receding, low, rising and high water) using ANOVA. Once we observed no significant seasonal variation in species richness for any of the sections (UPRIVER: F = 2.080; d.f. = 3; *p* = 0.133/ RESERVOIR: F = 0.188; d.f. = 3; *p* = 0.903/ DOWNRIVER: F = 1.420; d.f. = 3; *p* = 0.265) (Supplementary 2-1), we combined data from all seasons in the analyses.

To assess differences in species richness and abundance, we used native and non-native extrapolated species richness and CPUE by phases/section. We assessed native and non-native data separately because anthropogenic impacts can favor the establishment and spread of non-natives, leading to different patterns than for natives (Martins *et al.*, 2015; Loures & Pompeu, 2018). To do so, both, richness and abundance values were log-transformed and examined using two-way ANOVA, with phases and sections the categorical variables. For significant differences, we performed an *a posteriori* multiple comparison test (Zar, 2010). All statistical tests were carried out considering a 5% significance level. We chose not to use a temporal analysis because we paused our collections for two years twice, between March 2007 to October 2009 and from September 2013 to February 2016, making this kind of analysis inappropriate.

Changes in fish assemblage composition among the three phases (River, Dam and Rupture) were analyzed using a PERMANOVA, and the results were summarized using a Principal Coordinates Analysis (PCoA) based on a Bray-Curtis dissimilarity index for each section (UPRIVER, RESERVOIR, DOWNRIVER; Anderson, 2001; Legendre & Legendre, 2012). To reduce the effects of dominant species, we log-transformed [$\log_{(10)}(x + 1)$] species abundance data. To assess how each species contributed to differences among treatments we conducted a similarity percentile analysis (SIMPER) (Clarke, 1993; Legendre & Legendre, 2012). Both analyses were performed in R (R Core Team, 2017).

RESULTS

Water quality

Among the water quality variables, turbidity showed considerable changes during the phases, decreasing in Dam (55 ± 70) when compared with River (85 ± 142), and increasing in Rupture (4,408 ± 18,073), when compared with both River and Dam (Table 2-1). Just after the rupture, turbidity reached a peak of 140,000 NTUs and then decreased gradually during the entire year until the next wet season, when it again reached a small peak of 2,003 NTUs (Supplementary 2-2). Dissolved oxygen remained near 7 mg/l in all three phases (River = 7.69 ± 0.11 mg/l; Dam = 7.73 ± 0.16 mg/l; Rupture = 6.89 ± 0.16 mg/l) (Table 2-1). However, just after the rupture, DO decreased to near anoxic (0.50 mg/l) in some days (Supplementary 2-2). Neither temperature nor pH showed marked changes during the temporal phases (Table 2-1; Supplementary 2-2).

The first two PCA axes explained 75.0 % of the water quality data variation, with the first axis explaining 47.5 %, and the second axis explaining 27.5 % (Figure 2-2). PCA-1 was positively correlated with turbidity (loading = 0.63), and negatively with DO (loading = - 0.69). PCA-2 was positively correlated with pH (loading = 0.84), and negatively with temperature (loading = - 0.47). We found no significant difference in water quality between phases (PERMANOVA: pseudo-F = 2.442; d.f. = 2; *p* = 0.06), although greater variability occurred in the Rupture phase.

Table 2-1. Water quality conditions of the main channel of the middle Rio Doce.

Variables	Phase	N	Min.	Mean	Max.
Temp. (°C)	River	44	19.0	26.4 ± 0.44	32.5
	Dam	8	24.3	27.9 ± 0.91	31.8
	Rupture	68	21.2	27.6 ± 0.34	34.3
pH	River	44	6.20	7.08 ± 0.06	8.20
	Dam	8	6.40	6.81 ± 0.13	7.50
	Rupture	68	6.00	6.96 ± 0.05	7.80
Turb. (NTU)	River	44	5	85 ± 141	794
	Dam	8	6	54 ± 70	206
	Rupture	68	3	4,407 ± 18,073	140,000
D.O. Mg/L	River	44	6.00	7.69 ± 0.11	9.30
	Dam	8	7.10	7.73 ± 0.16	8.50
	Rupture	68	0.50	6.89 ± 0.16	8.60

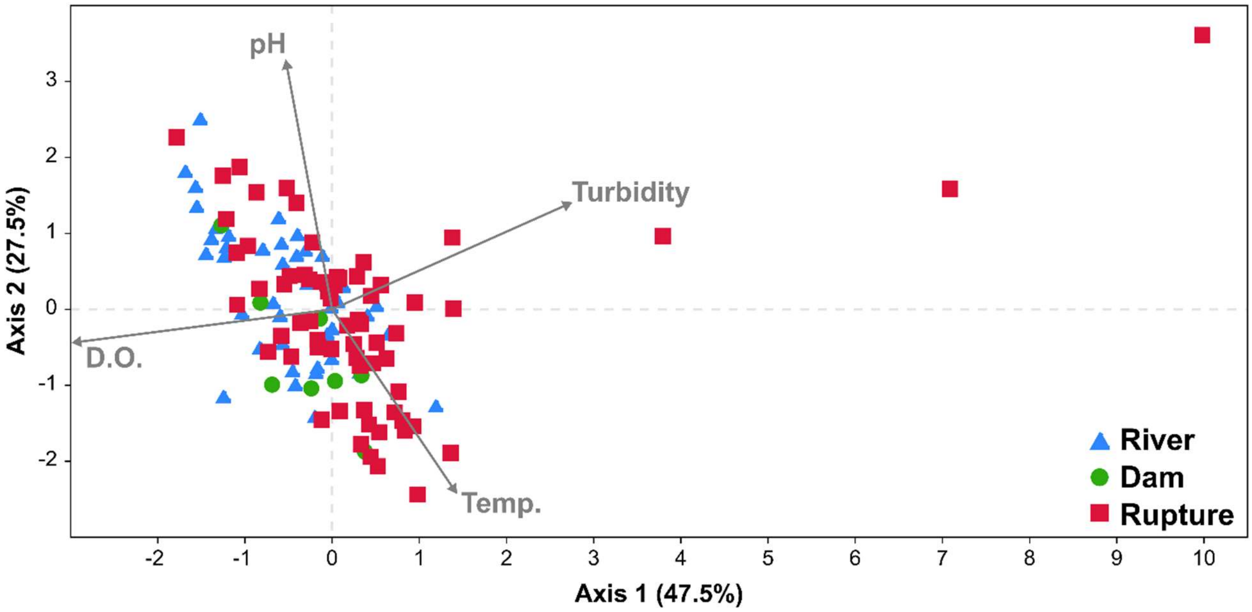


Figure 2-2. Water quality variables through the middle Rio Doce sampling phases showing the TSF disaster effect on turbidity and dissolved oxygen. River = pre-dam; Dam = reservoir post-filling; Rupture = TSF rupture. PERMANOVA was used to test the differences between phases (River x Dam x Rupture).

Fish assemblages

Our ichthyofaunal collections produced six orders, 18 families, and 42 species, 28 of which are native to the basin and 14 non-native (Table 2). The total richness in each section differed only slightly during the study: UPRIVER (39), RESERVOIR (32), DOWNRIVER (38). Thirty species occurred in all three sections; however, *Brycon dulcis*, *Australoheros* cf. *ipatinguensis* and *Synbranchus* sp. were collected only in the UPRIVER section, whereas the hybrid *Colossoma macropomum* x *Piaractus mesopotamicus* and *Gymnotus sylvius* occurred only in the DOWNRIVER section. Seven other species occurred in two sections (Table 2-2).

Table 2-2. Fish species collected in the main channel of the middle Rio Doce between 2006 and 2018. † Non-native species.

ORDER/Familly/*Species*	**SECTION**		
	UPRIVER	**RESERVOIR**	**DOWNRIVER**
CHARACIFORMES			
Anostomidae			
Hypomasticus copelandii (Steindachner 1875)	x	x	x
Megaleporinus conirostris (Steindachner 1875)	x	x	x
Megaleporinus obtusidens (Valenciennes 1837) †	x	x	x
Bryconidae			
Brycon dulcis Lima & Vieira 2017	x		
Salminus brasiliensis (Cuvier 1816) †	x	x	x
Characidae			
Astyanax lacustris (Lütken 1875)	x	x	x

ORDER/Familly/*Species*	**SECTION**		
	UPRIVER	**RESERVOIR**	**DOWNRIVER**
Astyanax sp.	x		x
Deuterodon aff. *taeniatus* (Jenyns 1842)	x		x
Oligosarcus acutirostris Menezes 1987	x	x	x
Psalidodon aff. *fasciatus* (Cuvier 1819)	x	x	x
Curimatidae			
Cyphocharax gilbert (Quoy & Gaimard 1824)	x	x	x
Erytrinidae			
Hoplias gr. *malabaricus* (Bloch 1794)	x	x	x
Hoplias intermedius (Günther 1864)	x	x	x
Prochilodontidae			
Prochilodus costatus Valenciennes 1850 †	x	x	x
Prochilodus vimboides Kner 1859	x	x	x
Serrasalmidae			
Colossoma macropomum x *Piaractus mesopotamicus* †			x
Metynnis cf. *lippincottianus* (Cope 1870) †	x		x
Pygocentrus nattereri Kner 1858 †	x	x	x
GYMNOTIFORMES			
Gymnotidae			
Gymnotus aff. *carapo* Linnaeus 1758	x	x	x
Gymnotus sylvius Albert & Fernandes-Matioli 1999			x
CICHLIFORMES			
Cichlidae			
Australoheros cf. *ipatinguensis* Ottoni & Costa 2008	x		
Cichla cf. *kelberi* Kullander & Ferreira 2006 †	x	x	x
Coptodon rendalli (Boulenger 1897) †	x	x	
Crenicichla lacustris (Castelnau 1855)	x	x	x
Geophagus brasiliensis (Quoy & Gaimard 1824)	x	x	x
Oreochromis niloticus (Linnaeus 1758) †	x	x	x
PERCIFORMES			
Scianidae			
Pachyurus adspersus Steindachner 1879	x	x	x
SILURIFORMES			
Auchenipteridae			
Glanidium botocudo Sarmento-Soares & Martins-Pinheiro 2013	x	x	x
Pseudauchenipterus affinis (Steindachner 1877)	x	x	x
Trachelyopterus striatulus (Steindachner 1877)	x	x	x
Callichthydae			
Hoplosternum littorale (Hancock 1828) †	x	x	x
Clariidae			
Clarias gariepinus (Burchell 1822) †	x	x	x
Heptapteridae			
Pimelodella sp.	x		x
Rhamdia quelen (Quoy & Gaimard 1824)		x	x
Loricariidae			
Delturus carinotus (LaMonte 1933)	x		x
Hypostomus affinis (Steindachner 1877)	x	x	x
Hypostomus luetkeni (Steindachner 1877)	x	x	x

ORDER/Familly/*Species*	**SECTION**		
	UPRIVER	**RESERVOIR**	**DOWNRIVER**
Loricariichthys castaneus (Castelnau 1855)	x	x	x
Pogonopoma wertheimeri (Steindachner 1867) †	x	x	x
Pimelodidae			
Pimelodus maculatus Lacepède 1803 †	x	x	x
Pseudopimelodidae			
Lophiosilurus alexandri Steindachner 1876 †	x	x	x
SYMBRANCHIFORMES			
Symbranchydae			
Synbranchus sp.	x		

The extrapolated total native and non-native species richness showed no significant differences among phases or sections (native/phase: F = 1.012, p = 0.369; native/sections: F = 1.155, p = 0.321; native/interactions: F = 0.489, p = 0.743) (non-native/phase: F = 0.274, p = 0.761; non-native/sections: F = 0.542, p = 0.583; non-native/interactions: F = 0.880, p = 0.480) (Figure 2-3; Supplementary 2-3). Unlike total species richness, native species CPUE was significant for interaction between phases and sections (native/phase: F = 6.359, p = 0.003; native/sections: F = 1.538, p = 0.222; native/interactions: F = 4.814, p = 0.001). The results also show distinct temporal patterns for sections, with higher native species CPUE values in River (23.75 ± 4.98) and Dam (23.20 ± 4.25), compared with Rupture (8.71 ± 3.15) in UPRIVER. Similarly, for all temporal phases, native species CPUE values in RESERVOIR (River = 10.73 ± 1.11; Dam = 13.48 ± 1.35) were higher than in Rupture = 9.61 ± 2.79). However, native species CPUE values were lower in River (4.71 ± 1.33) when compared with Dam (17.31 ± 3.35) and Rupture (16.96 ± 4.02) in DOWNRIVER (Figure 2-3; Supplementary 2-4 and Supplementary 2-5). Non-native fish species CPUE was significant for temporal phases, showing similar patterns between sections (non-native/phase: F = 14.650, p < 0.001; non-native/sections: F = 0.799, p = 0.454; non-native/interactions: F = 0.806, p = 0.525). Non-native fish species CPUE during Rupture (UPRIVER = 7.86 ± 0.74; RESERVOIR = 7.11 ± 0.78; DOWNRIVER = 6.18 ± 1.27) was higher than during both River (UPRIVER = 4.34 ± 2.78; RESERVOIR = 4.78 ± 3.51; DOWNRIVER = 1.77 ± 0.91) and Dam temporal phases (UPRIVER = 3.42 ± 0.64; RESERVOIR = 2.90 ± 0.43; DOWNRIVER = 3.54 ± 0.71) (Figure 2-3; Supplementary 2-4, and Supplementary 2-6).

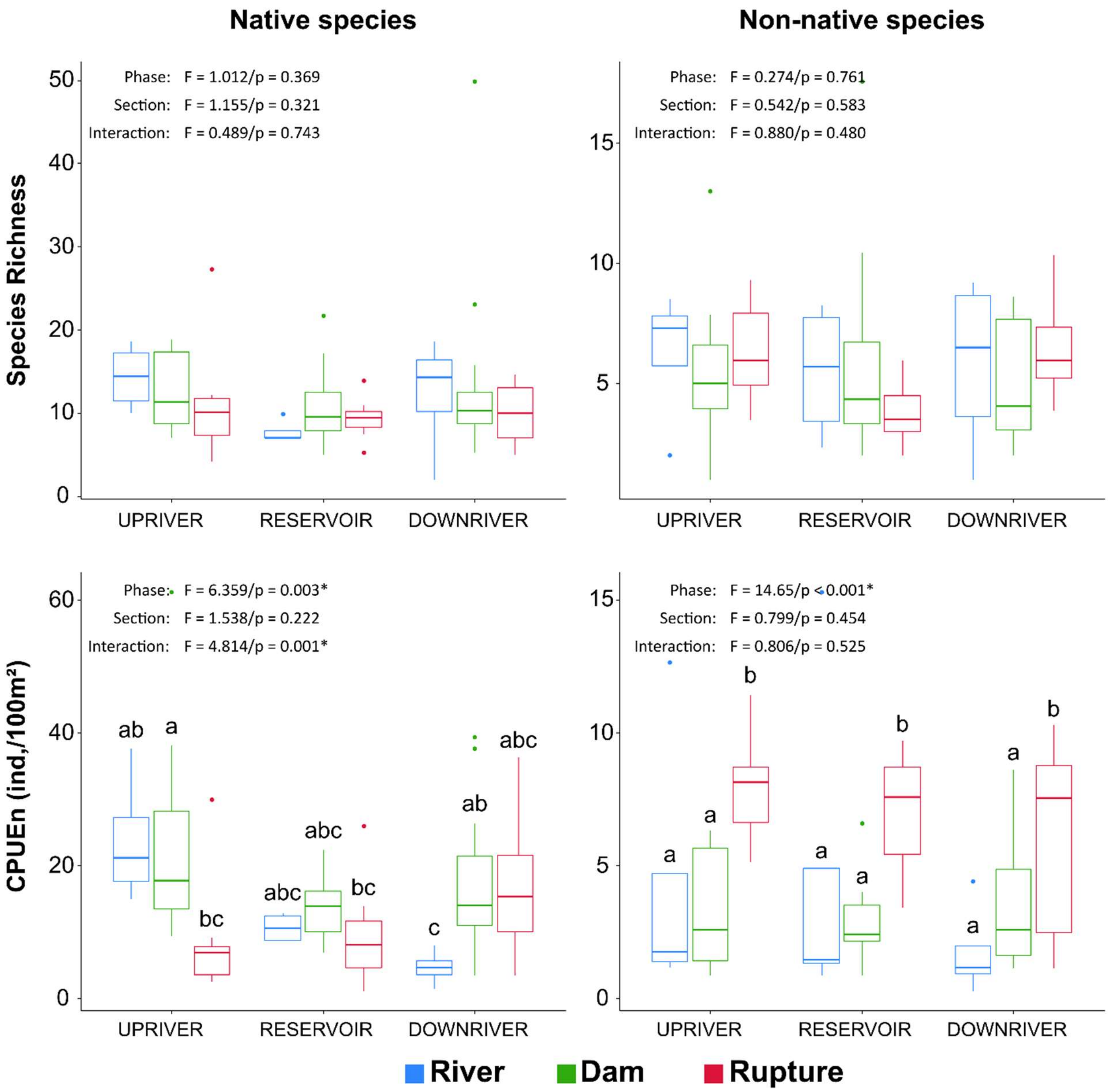


Figure 2-3. Native and non-native extrapolated species richness and CPUE variation in the middle Rio Doce during the three sampling phases (River = pre-dam; Dam = dam post-filling; Rupture = TSF rupture) by section (UPRIVER, RESERVOIR and DOWNRIVER). * indicates significative p values. Central lines represent medians, boxes represent quartiles (25% and 75%), external lines represent non-outlier ranges (95%), dots represent outliers.

Fish assemblage composition as measured by a Bray-Curtis dissimilarity index differed between River and Dam temporal phases versus the Rupture phase in the UPRIVER section (pseudo-F = 4.100; d.f. = 2; p = 0.001/ Pairwise test: River x Dam = 0.540; River x Rupture = 0.006; Dam x Rupture = 0.003) (Figure 2-4). This dissimilarity was influenced by decreased abundances of *Oligosarcus acutirostris, Astyanax lacustris, Trachelyopterus striatulus, Loricariichthys castaneus, Pogonopoma wertheimeri* and *Gymnotus* aff. *carapo,* and increased abundances of *Pimelodus maculatus, Hoplosternum littorale* and *Pygocentrus nattereri* (Table 2-3). On the other hand, fish assemblage composition changed between all three temporal phases for RESERVOIR and DOWNRIVER sections (RESERVOIR: pseudo-F = 5.015; d.f. = 2; $p = 0.001$/ Pairwise test: River x Dam = 0.003; River x Rupture = 0.009; Dam x Rupture = 0.003/ DOWNRIVER: pseudo-F = 5.170; d.f. = 2; $p = 0.001$/ Pairwise test: River x Dam = 0.006; River x Rupture = 0.018; Dam x Rupture = 0.003) (Figure 2-4). For the RESERVOIR section, *A. lacustris* and *Prochilodus costatus* were more abundant in the River phase; *L. castaneus, Megaleporinus conirostris, O. acutirostris* and *T. striatulus* were more abundant in the Dam phase; and *Pachyurus adspersus* and *P. nattereri* were more abundant in the Rupture phase (Table 2-3). For the DOWNRIVER section, *Hypostomus luetkeni* and *P. maculatus* were more abundant in the Dam phase and *A. lacustris, Hypostomus affinis, O. acutirostris, P. adspersus, P. nattereri* and *T. striatulus* were more abundant in the Rupture phase (Table 2-3).

Table 2-3. Most important species selected by SIMPER for distinguishing the UPRIVER section during the three study periods (River, Dam, Rupture) between 2006 and 2018. † Non-native species. The complete table for each section can be found in Supplementary 2-7.

Section	Taxon	Av. Dis.	Contr. (%)	Cum. (%)	Mean		
					River	Dam	Rupture
UPRIVER	*Oligosarcus acutirostris*	4.63	8.37	8.37	0.85	0.95	0.22
	Astyanax lacustris	4.53	8.19	16.55	1.47	1.53	0.82
	Pimelodus maculatus†	4.12	7.43	23.98	0.08	0.73	1.14
	Trachelyopterus striatulus	3.09	5.57	29.56	0.66	0.38	0.30
	Loricharichthys castaneus	2.97	5.36	34.92	0.82	0.62	0.32
	Pogonopoma wertheimeri†	2.59	4.68	39.59	0.51	0.44	0.40
	Hoplosternum littorale†	2.54	4.59	44.18	0.25	0.09	0.56
	Pygocentrus nattereri †	2.53	4.57	48.75	0.25	0.02	0.46
	Gymnotus aff. *carapo*	2.30	4.16	52.91	0.52	0.34	0.10
RESERVOIR	*Pygocentrus nattereri†*	5.43	9.46	9.46	0.25	0.26	1.15
	Astyanax lacustris	4.31	7.52	16.98	1.24	0.83	0.68
	Megaleporinus conirostris	3.80	6.61	23.59	---	0.62	0.20
	Trachelyopterus striatulus	3.76	6.55	30.14	0.18	0.72	0.49
	Oligosarcus acutirostris	3.65	6.37	36.51	0.48	0.68	0.35
	Pachyurus adspersus	3.37	5.86	42.37	---	---	0.69
	Loricariichthys castaneus	2.96	5.16	47.53	0.66	0.80	0.70
	Prochilodus costatus†	2.74	4.77	52.30	0.45	0.33	0.26
DOWNRIVER	*Hypostomus luetkeni*	5.89	9.34	9.34	---	1.05	0.29
	Astyanax lacustris	4.63	7.34	16.68	0.65	0.96	1.01
	Oligosarcus acutirostris	4.53	7.18	23.86	---	0.60	0.79
	Pygocentrus nattereri†	4.45	7.05	30.91	0.08	0.20	0.94
	Pachyurus adspersus	4.27	6.76	37.68	---	---	0.87
	Trachelyopterus striatulus	4.21	6.67	44.35	0.20	0.39	0.87
	Pimelodus maculatus†	3.18	5.03	49.38	0.41	0.69	0.52
	Hypostomus affinis	2.74	4.35	53.73	0.41	0.56	0.77

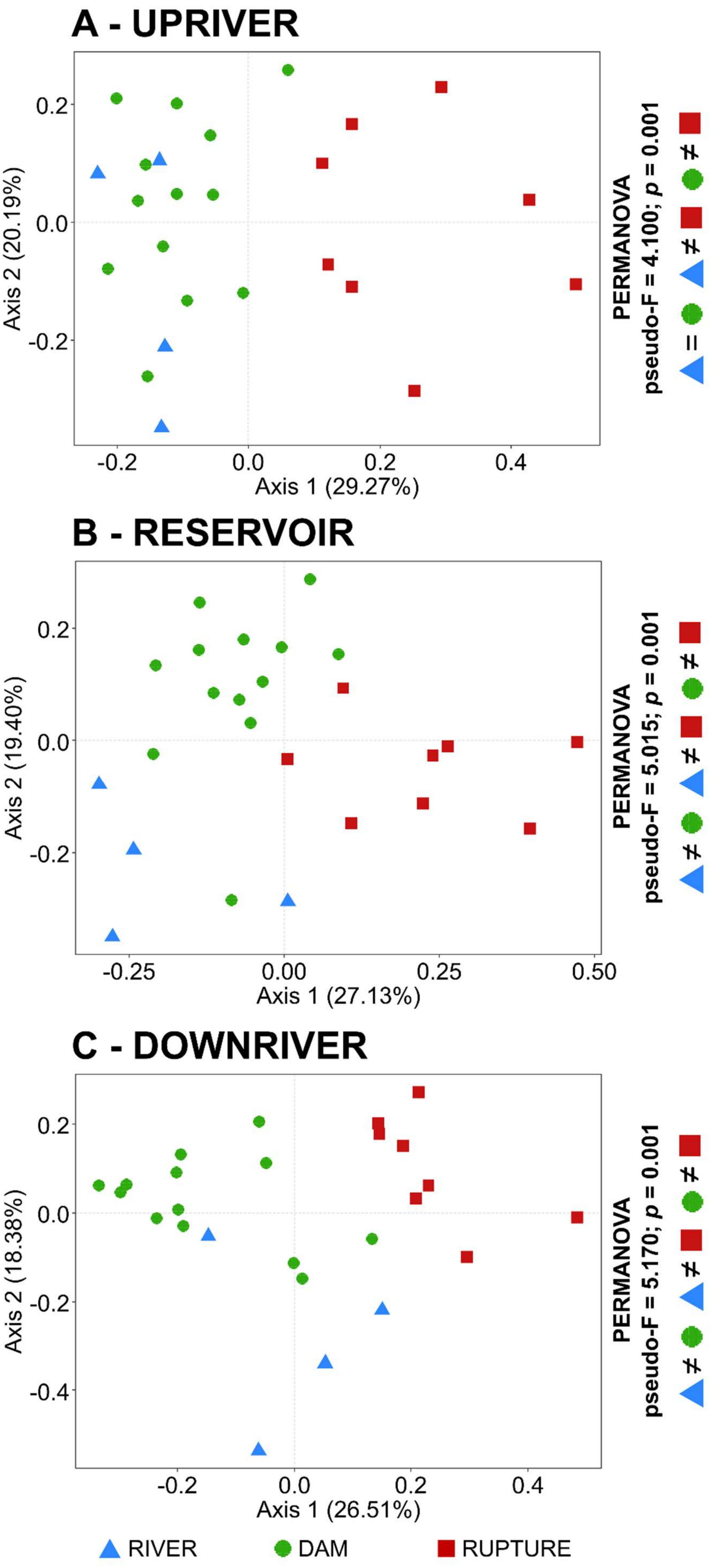


Figure 2-4. Principal Coordinates Analysis of middle Rio Doce fish assemblage composition. River = pre-dam; Dam = dam post-filling; Rupture = TSF rupture. PERMANOVA was used to test the differences between phases (River x Dam x Rupture).

DISCUSSION

Water quality

The Fundão TSF rupture was a critical environmental disaster in Brazil and one of the largest mining accidents in the world (Escobar, 2015; Abdelouas, 2019; Salvador *et al.*, 2020). The volume of tailings carried downriver, and the length of river affected by the tailings are the largest recorded in mining history (Abdelouas, 2019). As a direct consequence, we observed sudden changes in both turbidity and DO concentration soon after the disaster. Values above 1,000 NTU were measured throughout the first year after the disaster. Nearly anoxic DO levels were observed for a month after the disaster. The peak of turbidity, reduction of DO, and the deposit of ore tailings in the Rio Doce are believed to be the main causes of mass mortality of fish observed in the Rio Doce after the disaster (ANA, 2016a). After the rupture, iron ore tailings were deposited along the Rio Doce channel, changing its hydrology and coating substrates with ferric oxide flocs (Grilo et al., 2018). However, contrary to our expectations, no significant variation in those parameters was observed.

Turbidity would be expected to vary between all three phases. A reservoir tends to act as a sediment trap and it is common to observe a reduction in turbidity downstream of hydroelectric dams (de Cesare, Schleiss & Hermann, 2001). The location for measuring water quality variables was downstream from the Baguari hydroelectric plant and it did not receive any large effluent that could significantly alter turbidity. A plausible explanation is that Baguari HPP reservoir is a small run-of-river system that traps little sediment (Dendy & Cooper, 1984). This is reinforced by the fact that the lowest turbidity value recorded occurred during Dam phase. Also, the Fundão TSF spill produced a pulse of turbidity in the Doce River and a gradual return to pre-Rupture values (Rudorff et al., 2018), with turbidity oscillating around 3000 NTUs just one month after Rupture and DO returning to acceptable values after a few days (ANA 2018). This quick return, combined with the type of analysis (grouping the data into categories) and sample series used in our study, may have resulted in our missing short-term differences between Rupture and the River and Reservoir phases.

The returns of those parameters to prior disaster values do not indicate that the rupture did not affect the Rio Doce. Many parameters, such as metals and organic chemicals, were not measured throughout our entire study period by ANA (2018) and, consequently, were not analyzed here. However, studies carried out after the disaster (Rupture phase) showed increased metals concentrations along the river, leading to contamination of aquatic life (Hatje et al., 2017; Omachi et al., 2018; Weber et al., 2020; Vergilio et al., 2021). The contamination of water and aquatic life, especially fish, is considered one of the worst consequences of this disaster. The Rio Doce basin is home for more than 1 million people, and thousands of those are dependent on water supply from the Rio Doce, and other thousands lived on the river's fisheries (Viana, 2017; Santana et al., 2021).

Fish assemblages

Although we hypothesized that there would be River, Dam, and Rupture phase changes in native, and non-native species richness in river sections, we observed no significant differences. The reasons seem to differ for native versus non-native fish. Usually, a dam affects the fish fauna upriver, in the reservoir, and downriver of the dam (Agostinho *et al.,* 2007, 2016). The major effects are observed in the reservoir as a consequence of the creation of a lentic-like environment, with the replacement of many lotic fish species by lentic species. However, Baguari produced a run-of-river reservoir that filled the seasonal riverbed almost entirely in the fluvial zone, which usually shows lower impacts than in the lacustrine zone (Agostinho *et al.,* 2007, 2016; Baumgartner, Baumgartner & Gomes, 2018; Sanches *et al.,* 2016). River fragmentation also could extirpate migratory fishes (Pelicice & Agostinho, 2008). Nonetheless, there are many dam-free river reaches up- and down-river of Baguari, many of which are more than 100 km long and in which migratory fishes can reproduce (Santos *et al*., 2012). Another factor that helps explain the absence of differences between River and Dam phases for both native and non-native species richness was the previous extirpation of sensitive species (Ruschi, 1965; Alves *et al*., 2007; Vieira, 2009; Gomes *et al*., 2020) and the prior occurrences of resistant non-native species (Magurran, 2013; Soares *et al.*, 2017; Bueno *et al*., 2021). These factors may explain the absence of differences between River and Dam temporal phases, but not the absence of expected differences between Dam and Rupture. We expected reduced species richness in all sections as a result of the high fish mortality following the rupture (Neves *et al.*, 2016; Grilo *et al.*, 2018). However, recolonization occurred quickly, except for native species in DOWNRIVER and non-natives in RESERVOIR during the first year. In this case, the quick return of water quality (especially turbidity and DO) and refuges in large nearby tributaries provided conditions for relatively rapid fish recolonization of the Rio Doce mainstem. Those tributaries contain all the species recorded in the Rio Doce mainstem (Barbosa *et al.*, 1997; Vieira, 2006; 2009) and were not directly affected by the mudflow. Unfortunately, our method prevents detecting how this recolonization occurred.

Unlike fish species richness, we observed the expected differences in fish abundance (CPUE) among sampling phases. Native fish abundance showed distinct patterns between sections and phases, but non-native abundance showed a pattern only among sections. The absence of difference in native abundance between River and Dam in UPRIVER was expected given its minimal change following dam construction (Agostinho *et al.,* 2007, 2016; Mazzoni & Petito, 2012). For RESERVOIR, the maintenance of the native fish abundance likely resulted from the run-of-river or fluvial character of Baguari reservoir (Okada *et al*., 2005). The greater abundance of native species in the DOWNRIVER section during the Dam phase versus the River phase likely resulted from the creation of a lentic-like environment just downstream of the Baguari dam spillway. The 600-m reach between the spillway and the tailrace receives little water during most of the year, mostly occurring during the November-February wet season. Those conditions favored increased abundances of lentic species (Gomes *et al.*, 2015; Loures & Pompeu, 2018). Comparing Dam and Rupture phases, we observed a decline in native fish abundance only in the UPRIVER section, which could be linked to its sinuosity and substrate type. The UPRIVER section meanders, with sandy beaches inside the curves. Tailings were deposited on the beaches and remained after the wet season. The DOWNRIVER section is nearly straight and the principal substrate is bedrock, from which most of the tailings were washed during the wet season.

The results for abundance of non-native species also showed unexpected responses in the sections. Some non-native species increased in abundance in the Dam phase, other species experienced reduced abundance, resulting in a turnover of more abundant non-native species. Comparing Dam and Rupture phases, non-native abundance increased in all sections. *Hoplosternum littorale*, *Pygocentrus nattereri* and *P. maculatus* increased in abundance during the Rupture phase. Those species have traits that increase their capacities to colonize new environments, such as high fecundity and accessory respiration (Jucá-Chagas & Boccardo, 2006; Maia *et al*., 2007; Arantes *et al.*, 2019; Su, Villéger & Brosse, 2020). Such traits may have helped them establish populations in the Rio Doce faster than native species.

Although total species richness and native abundance were poor predictors of the dam and TSF rupture impacts, fish assemblage composition showed clear changes from both pressures, as we hypothesized. We observed distinct phase changes in fish assemblage composition for the RESERVOIR and DOWNRIVER sections, and between River and Dam phases versus the Rupture phase for the UPRIVER section. The similarity between fish assemblage composition in the River and Dam phases in the UPRIVER section is likely the result of minimal hydrologic alterations upriver of the run-of-river reservoir (Agostinho *et al.*, 2007, 2016; Baumgartner *et al.*, 2018). We expected differences in fish assemblage composition between the River and Dam phases in the RESERVOIR section because of the transformation of a lotic environment into a somewhat more lentic environment. Such changes usually lead to changes in both species composition and abundance, reflecting lost connectivity and altered hydrologic conditions (Pelicice & Agostinho, 2008; Petesse & Petrere, 2012; Agostinho *et al.*, 2016). However, in our study the proximity of a long free-flowing upriver section relative to the small run-of-the-river reservoir that we sampled led to few local extirpations or assemblage composition changes. In addition, some large migratory species had been introduced to the Rio Doce prior to dam construction. The abundances of those two species were reduced, but they were not extirpated, after the Dam phase. On the other hand, some small- and medium-sized species became more abundant following reservoir creation by providing them with improved spawning and rearing conditions (Gomes *et al.*, 2011; Ferreira *et al.*, 2013). The change in fish assemblage composition after the Rupture phase was associated with increases in non-native species that had previously established populations in the Rio Doce (Vieira, 2009). Some native generalist benthic invertivores also benefited from the disaster, probably because of increased amounts of tolerant macroinvertebrates (Linares *et al.*, 2019) and molluscs.

In all cases, it is important to consider our study in the context of earlier historical landscape and riverscape changes (Harding *et al.*, 1998; Fausch *et al.*, 2002). Although we assessed two major environmental changes in the Rio Doce basin, the degradation of this basin began in the eighteenth century, with the discovery and extraction of gold in its headwaters (Borba *et al.*, 2000). Today, most of the native Atlantic Forest has been replaced by human land uses (Pinto, 1945; ANA, 2016b). Rivers lacking riparian forests are more susceptible to bank erosion (Barrella *et al.*, 2001), which is occurring throughout the basin, and even small amounts of forest removal can eliminate sensitive taxa (Dala-Corte *et al.*, 2020; Martins *et al.*, 2021). Large continuous forests remain in only three protected areas: Rio Doce State Park, Serra do Brigadeiro State Park, and Gandarela National Park, lying in the upper and middle reaches of the Rio Doce basin. Elsewhere, high levels of soil erosion are common. Historical records show that the Rio Doce was once deep (Mascarenhas, 1898), but in recent years its average depth has been <3 m. Such massive changes in vegetation cover and soil erosion can substantively alter river flow regimes, bed substrate, and channel morphology (Poff, Bledsoe & Cuhaciyan, 2006; Wohl, Lininger & Baron, 2017).

The Rio Doce basin's previous history of deforestation, soil erosion, riverbank erosion, and industrial and domestic pollution limited the potential impact to fish assemblages from the dam and TSF rupture. Some species that once inhabited the river (e.g., *Henochilus wheatlandii, Steindachneridion doceanum*) are currently extirpated (Vieira, 2006). The sum of all these impacts, including those we studied, favored increased numbers of non-native fish species. This prevalence of non-native species may affect the long-term ecological stability of the river if they are more resilient than native species to climate change and further anthropogenic changes. The losses from those impacts can be even greater if we consider the possible loss of cryptic or spatially rare species present in the basin (Sales *et al.*, 2018; Hughes *et al.*, 2021).

It is important to note that gillnets are a poor sampling method for small-sized benthic species that move small distances such as *Trichomycterus* spp. (Kotwicki et al., 2017; Miranda-Chumacero *et al.*, 2015), which are present in the Rio Doce, but which we never collected. Gillnets also do a poor job of collecting eel-like fishes such as *Symbranchus* sp. and *Gymnotus* spp., which we rarely collected. Therefore, gillnetting, or any single river sampling method for riverine fishes, usually underestimates the entire fish assemblage (Curry *et al.*, 2009; Guy *et al.*, 2009; Dunn & Paukert, 2020). Such small-sized species usually are less resilient than medium- and large-sized species and have lower migratory capacities (Hitt & Angermeier, 2008; Matthews, 2012). Nonetheless, it would be interesting to understand how the TSF rupture influenced such species. Furthermore, although we observed fish recolonization in the study area after the TSF rupture, it is 300 km from the TSF. The middle Rio Doce has two hydroelectric reservoirs, but also has long free-flowing segments (Zambaldi & Pompeu, 2020). It would be interesting to evaluate the mainstem recovery from the TSF over the entire Rio Doce, thereby incorporating a study perspective based on Riverscape (Fausch *et al.*, 2002; Herlihy *et al.*, 2022) and Serial Discontinuity (Ward & Stanford, 1995) concepts. Nonetheless, we believe that our observed changes in fish assemblage composition indicated clear effects on middle Rio Doce fish assemblages resulting from both the dam and the TSF rupture.

Summary, Conclusions & Recommendations

Although our study design was not ideal to analyze a pulse disturbance, the changes in fish assemblage composition that we observed indicate clear effects on middle Rio Doce fish assemblages resulting from both the dam and the TSF rupture. Even a run-off-the-river reservoir may favor more generalist species while maintining lower abundances of riverine species. The presence of free-flowing reaches in the Doce basin also permitted the maintenance of migratory fishes up- and down-river of Baguari dam. Changes in substrate and water quality were rapid and relatively short-lived, but changed fish assemblage composition and favored non-native and generalist native species.

Because of the limited extent of our monitoring, we recommend additional TSF and dam studies in the Rio Doce basin based on four perspectives. Firstly, the survey should be conducted throughout the affected and unaffected segments of the basin, preferably based on a systematic probability survey design (e.g., LaVigne, Hughes & Herlihy, 2008; Hughes *et al.*, 2012; Herlihy *et al.*, 2022). Secondly, environmental parameters should include sedimentary metals, which are persistent contaminants after dam failures (Woody *et al.*, 2010; Hughes *et al.*, 2016), as well as quantitative measures of physical habitat structure (Herlihy *et al.*, 2022; Leal *et al.,* 2016; Leitão *et al.*, 2018). Thirdly, the fish assemblage indicators should include tolerant and non-native species abundances, fish assemblage composition and a multimetric index (Terra *et al.*, 2013; de Carvalho *et al.*, 2017). Fourthly, it is important to create baseline data for fish assemblages throughout the entire basin given the high probabilty of new ruptures. It would then be essential to be prepared to implement monthly fish assemblage collections just after the rupture to understand the degree to which the ichthyofauna recolonized and the resilience capacity of the river.

Nonetheless, we believe that the Doce basin native fish assemblages could eventually be rehabilitated as they were in the Willamette River, Oregon, and Scioto River, Ohio, both of which were once severely polluted (Hughes *et al.,* 2019; Yoder *et al.*, 2019). To increase the probability of doing so, it will be necessary to reforest much of the basin and large extents of its riparian forest (Pires *et al.*, 2017). It will also be necessary to increase riverscape connectivity (Fausch *et al.*, 2002), discouraging the construction of new hydroelectric plants in the basin and facilitating fish passage around dams and reservoirs and locating them more strategically (O'Hanley *et al.*, 2020; Twardek *et al.,* 2022). New methods of tailings storage are needed to reduce or eliminate the probability of further dam failures, including removal of existing tailings storage facilities (Davies, 2011; Franks *et al.*, 2011; Salvador *et al.*, 2020). Further recuperation of the Rio Doce basin requires sewage and industrial waste treatment, recovery of the river's flow regime, and reduced non-native species abundances, as has occurred in some USA and European basins (e.g., Schinegger *et al.*, 2016; Hughes *et al.*, 2019). In other words, to rehabilitate the Rio Doce, we must consider historical factors and the considerable investments needed throughout the entire basin to recover riverine ecosystem processes and functions (Pires *et al.*, 2017).

REFERENCES

Abdelouas, A. (2019). World Mine Tailings Failures - from 1915. Retrieved from https://www.wise-uranium.org/.

Abell, R., Thieme, M. L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., … Petry, A. (2008). Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. *BioScience,* 58, 403. DOI: 10.1641/B580507

Agostinho, A.A., Gomes, L.C. & Pelicice, F.M. (2007). Ecologia e Manejo de Recursos Pesqueiros em Reservatórios do Brasil. Maringá, PR: EDUEM.

Agostinho, A. A., Gomes, L. C., Santos, N. C. L., Ortega, J. C. G., & Pelicice, F. M. (2016). Fish assemblages in Neotropical reservoirs: colonization patterns, impacts and management. *Fisheries Research*, 173, 26–36. DOI:10.1016/j.fishres.2015.04.006

Agostinho A.A., Miranda L.E., Bini L.M., Gomes L.C., Thomaz S.M. & Suzuki H.I. (1999) Patterns of colonization in neotropical reservoirs, and prognoses on aging. Chapter 5. In: Theoretical Reservoir Ecology and its Application. (Eds: J.G. Tundisi & M. Straskraba). International Institute of Ecology, São Carlos.

Alves C.B.M., Vieira F., Magalhães L.B. & Brito M.F.G. (2007). Impacts of non-native fish species in Minas Gerais, Brazil: present situation and prospects. Chaptere 16. In: Ecological and Genetic Implications of Aquaculture Activities. (Ed: T.M. Bert). Springer, New York.

ANA (2016a). Encarte especial sobre a bacia do rio Doce: rompimento da barragem em Mariana. Agência Nacional de Águas/ Ministério do Meio Ambiente, Brasília.

ANA (2016b). Relatório de segurança de barragens 2015. Agência Nacional de Águas, Brasília.

ANA (2018). HidroWeb: Sistemas de Informações Hidrológicas. Retrieved from http://www.snirh.gov.br

Anderson, M. J. (2001). A new method for non-parametric multivariate analysis of variance. *Austral Ecology*, 26, 32–46. DOI: 10.1046/j.1442-9993.2001.01070.x

ANEEL (2010). Sistema de Informações Georreferenciadas do Setor Elétrico – SIGEL. Ministério das Minas e Energias, Brasilia.

Arantes, C. C., Fitzgerald D. B., Hoeinghaus D. J., & Winemiller K. O. (2019). Impacts of hydroelectric dams on fishes and fisheries in tropical rivers through the lens of functional traits. *Current Opinion in Environmental Sustainability*, 37, 28–40. DOI: 10.1016/j.cosust.2019.04.009

Barbosa F.A.R., Souza E.M.M., Vieira F., Renault G.P.C.P., Rocha L.A., Maia-Barbosa P.M., ... Mingoti S.A. (1997). Impactos antrópicos e biodiversidade aquática. Chapter 8. In: Biodiversidade, população e economia: uma região de mata atlântica. (Eds: J.A. Paula, A.F. Barbieri, C.B. Guerra, H.C. Landau, F. Vieira, F.A.R. Barbosa, ... T.M. Braga). UFMG/Cedeplar, Belo Horizonte.

Barbosa, T. A. P., Benone N. L., Begot T. O. R., Gonçalves A., Sousa L., Giarrizzo T., … Montag L. F. A. (2015). Effect of waterfalls and the flood pulse on the structure of fish assemblages of the middle Xingu River in the eastern Amazon basin. *Brazilian Journal of Biology*, 75, 78–94. DOI: 10.1590/1519-6984.00214BM

Barrella W., Petrere-Júnior M., Smith W.S., & Montag L.F.A. (2001). As relações entre as matas ciliares, os rios e os peixes. Chapter 12. In: Matas Ciliares: Conservação e Recuperação. (Eds: R.R. Rodrigues & H.F. Leitão-Filho). EDUSP, São Paulo.

Baumgartner, M. T., Baumgartner, G., & Gomes L.C. (2018). Spatial and temporal variations in fish assemblage: testing the zonation concept in small reservoirs. *Brazilian Journal of Biology*, 78, 487–500. DOI: 10.1590/1519-6984.170424

Becker, B., Galhardo, B. O. S., Macedo, D. R., Hughes, R. M., Callisto, M., & Santos, G. B. (2016). Influence of limnological zones on the spatial distribution of fish assemblages in three Brazilian reservoirs. *Journal of Limnology*, 75, 156–168. DOI: 10.4081/jlimnol.2015.1266

Blackburn, T. M., Pyšek, P., Bacher, S., Carlton, J. T., Duncan, R. P., Jarošík, V., … Richardson, D. M. (2011). A proposed unified framework for biological invasions. *Trends in Ecology & Evolution*, 26, 333–339. DOI: 10.1016/j.tree.2011.03.023

Borba, R. P., Figueiredo, B. R., Rawlins, B., & Martshullat, J. (2000). Arsenic in water and sediment in the Iron Quadrangle, State of Minas Gerais, Brazil. *Revista Brasileira de Geociências*, 30, 558–561.

Bowker, L., & Chambers, D. (2017). In the dark shadow of the supercycle tailings failure risk &

public liability reach all time highs. *Environments*, 4, 75. DOI: 10.3390/environments4040075

Buenaga, F. V. A. S. C., Espig, S. A., Castro, T. L. C., & dos Santos, M. A. (2017). Impactos ambientais do trecho de vazão reduzida em hidrelétricas. *Cadernos de Energia*, 4, 7-36.

Bueno, M. L., Magalhães, A. L. B., Andrade Neto, F. R., Alves, C. B. M., Rosa, D. M., Junqueira, N. T., ... Zenni, R.D. (2021). Alien fish fauna of southeastern Brazil: species status, introduction pathways, distribution and impacts. *Biological Invasions*, 23, 3021–3034. DOI: 10.1007/s10530-021-02564-x

Carmo, F. F., Kamino, L. H. Y., Junior, R. T., Campos, I. C., Carmo, F. F., Silvino, G., ... Pinto, C. E. F. (2017). Fundão tailings dam failures: the environment tragedy of the largest technological disaster of Brazilian mining in global context. *Perspectives in Ecology and Conservation*, 15, 145–151. DOI: 10.1016/j.pecon.2017.06.002

Carneiro, F. M., & Bini, L. M. (2020). Revisiting the concept of longitudinal gradients in reservoirs. *Acta Limnologica Brasiliensia*, 32, e8. DOI: 10.1590/s2179-975x1319

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology*, 18, 117–143. DOI: 10.1111/j.1442-9993.1993.tb00438.x

Colwell, R. K., Chao, A., Gotelli, N. J., Lin, S. Y., Mao, C. X., Chazdon, R. L., … Longino, J. T. (2012). Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. *Journal of Plant Ecology*, 5, 3–21. DOI: 10.1093/jpe/rtr044

Curry R.A., Hughes R.M., McMaster M., & Zafft D. (2009). Cold water fish in rivers. Chapter 8. In: Standard Methods for Sampling North American Freshwater Fishes. (Eds: S. Bonar, W. Hubert & D. Willis), American Fisheries Society, Bethesda.

Daga, V. S., Olden, J. D., Gubiani, E. A., Piana, P. A., Padial, A. A., & Vitule, J. R. S. (2020). Scale-dependent patterns of fish faunal homogenization in Neotropical reservoirs. *Hydrobiologia*, 847, 3759–3772. DOI: 10.1007/s10750-019-04145-5

Dala-Corte, R. B., Melo, A. S., Siqueira, T., Bini, L. M., Martins, R. T., Cunico, A. M., … Roque, F. O. (2020). Thresholds of freshwater biodiversity in response to riparian vegetation loss in the Neotropical region. *Journal of Applied Ecology*, 57, 1391-1402. DOI: 10.1111/1365-

2664.13657

Daniel, W. M., Infante, D. M., Hughes, R. M., Tsang, Y. P., Esselman, P. C., Wieferich, D., & Taylor, W. W. (2015). Characterizing coal and mineral mines as a regional source of stress to stream fish assemblages. *Ecological Indicators*, 50, 50–61. DOI: 10.1016/j.ecolind.2014.10.018

Davies M. (2011). Filtered dry stacked tailings–the fundamentals. Proceedings of the tailings and mine waste. pp. 1–9. The University of British Columbia, Vancouver, BC.

Dendy, F. E., & Cooper, C. M. (1984). Sediment trap efficiency of a small reservoir. *Journal of Soil and Water Conservation*, 39, 278–280.

de Carvalho, D. R., Leal, C. G., Junqueira, N. T., de Castro, M. A., Fagundes, D. C., Alves, C. B. M., ... Pompeu, P.S. (2017). A fish-based multimetric index for Brazilian savanna streams. *Ecological Indicators*, 77, 386–396. DOI: 10.1016/j.ecolind.2017.02.032

de Cesare, G., Schleiss, A., & Hermann, F. (2001). Impact of turbidity currents on reservoir sedimentation. *Journal of Hydraulic Engineering*, 127, 6–16.

Dunn, C. G., & Paukert, C. P. (2020). A flexible survey design for monitoring spatiotemporal fish richness in nonwadeable rivers: optimizing efficiency by integrating gears. *Canadian Journal of Fisheries and Aquatic Sciences*, 77, 978–990. DOI: 10.1139/cjfas-2019-0315

Escobar, H. (2015). Mud tsunami wreaks ecological havoc in Brazil. *Science*, 350, 1138–1139. DOI: 10.1126/science.350.6265.1138

Espíndola, H. S., Campos, R. B. F., Lamounier, K. C. C., & Silva, R. S. (2016). Desastre da Samarco no Brasil: desafios para a conservação da biodiversidade. *Fronteiras: Journal of Social, Technological and Environmental Science*, 5, 72–100. DOI: 10.21664/2238-8869.2016v5i3.p72-100.

Espósito, T. J., & Duarte, A. P. (2010). Classificação de barragens de contenção de rejeitos de mineração e de resíduos industriais em relação a fatores de risco. *Rem: Revista Escola de Minas*, 63, 393–398. DOI: 10.1590/S0370-44672010000200026

Fausch, K. D., Torgersen, C., Baxter, C. V., & Li H. (2002). Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience*, 52, 1–16. DOI: 10.1641/0006-3568(2002)052[0483:LTRBTG]2.0.CO;2

FEAM (2015). Minas Trata Esgoto. Fundação Estadual do Meio Ambiente, Belo Horizonte, MG.

Ferreira, G. L., Lopes, V. G., Gomes, J. H. C., & Branco, C. W. C. (2013). Condition factor and diet of the catfish *Loricariichthys castaneus* (Castelnau, 1855): gender differences in three tropical reservoirs. *Studies on Neotropical Fauna and Environment*, 48, 190–198. DOI: 10.1080/01650521.2014.890849

Franco, A. C. S., dos Santos, L. N., Petry, A. C., & García-Berthou, E. (2018). Abundance of invasive peacock bass increases with water residence time of reservoirs in southeastern Brazil. *Hydrobiologia*, 817, 155–166. DOI: 10.1007/s10750-017-3467-x

Franks, D. M., Boger, D. V., Côte, C. M., & Mulligan, D. R. (2011). Sustainable development principles for the disposal of mining and mineral processing wastes. *Resources Policy*, 36, 114–122. DOI: 10.1016/j.resourpol.2010.12.001

Gomes, I. D., Araújo, F. G., do Nascimento, A. A., & Sales, A. (2015). Equilibrium reproductive strategy of the armored catfish *Hypostomus auroguttatus* (Siluriformes, Loricariidae) in a tropical river in Southeastern Brazil. *Environmental Biology of Fishes*, 98, 249–260. DOI: 10.1007/s10641-014-0256-5

Gomes, I. D., Araújo, F. G., Uehara, W., & Sales, A. (2011). Reproductive biology of the armoured catfish *Loricariichthys castaneus* (Castelnau, 1855) in Lajes reservoir, southeastern Brazil. *Journal of Applied Ichthyology*, 27, 1322–1331. DOI: 10.1111/j.1439-0426.2011.01874.x

Gomes, J. P. C., Salvador, G. N., Casarim, R., Pompeu, P. S., Brito, M. F. G., Andrade, F. R., ... Vieira, F. (2020). Hydropower construction plans threaten the largest Brazilian national river. *Aquatic Conservation: Marine and Freshwater Ecosystems*, 30(12), 2464–2465. DOI: 10.1002/aqc.3485

Grilo, C. F., Quaresma, V. S., Amorim, G. F. L., & Bastos, A. C. (2018). Changes in flocculation patterns of cohesive sediments after an iron ore mining dam failure. *Marine Geology*, 400, 1–11. DOI: 10.1016/j.margeo.2018.03.004

Guy C.S., Braaten P.J., Herzog D.P., Pitlo J. & Rogers, R.S. (2009) Warmwater fish in rivers. Chapter 5. In: Standard Methods for Sampling North American Freshwater Fishes. (Eds: S. Bonar, W. Hubert & D. Willis). American Fisheries Society, Maryland.

Harding, J. S., Benfield, E. F., Bolstad, P. V, Helfman, G. S., & Jones E. B. (1998). Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Sciences of the United States of America*, 95, 14843–14847

Harley, S. J., Myers, R. A., & Dunn, A. (2001). Is catch-per-unit-effort proportional to abundance? *Canadian Journal of Fisheries and Aquatic Sciences*, 58, 1760–1772.

Hatje, V., Pedreira, R. M. A., de Rezende, C. E., Schettini, C. A. F., de Souza, G. C., Marin, D. C., ... Hackspacher, P. C. (2017). The environmental impacts of one of the largest tailing dam failures worldwide. *Scientific Reports*, 7, 10706. DOI: 10.1038/s41598-017-11143-x

Herlihy, A. T., Hughes, R. M., & Gerth, W. J. (2022). Longitudinal patterns in river ecological indicators in the Pacific Northwest: implications for river research, monitoring and management. *River Research and Applications*, 38, 548-560. DOI: 10.1002/rra.3916

Hitt N. P., & Angermeier P. L. (2008). Evidence for fish dispersal from spatial analysis of stream network topology. *Journal of the North American Benthological Society*, 27, 304–320. DOI: 10.1899/07-096.1

Hughes, R. M., Amezcua, F., Chambers, D. M., Daniel, W. M., Franks, J. S., Franzin, W., … Woody C. A. (2016). AFS position paper and policy on mining and fossil fuel extraction. *Fisheries,* 41, 12–15. DOI: 10.1080/03632415.2016.1121742

Hughes R.M., Bangs B.L., Gregory S.V., Scheerer P.D., Wildman R.C. & Ziller J.S. (2019). Recovery of Willamette River (Oregon, USA) fish assemblages: successes & remaining threats. Chapter 1.6. In: From Catastrophe to Recovery: Stories of Fish Management Success. (Eds: C.C. Krueger, W.W. Taylor & S.-J. Youn). American Fisheries Society, Bethesda.

Hughes, R. M., Herlihy, A. T., & Peck, D. V. (2021). Sampling efforts for estimating fish species richness in western USA river sites. *Limnologica*, 87, 125859. DOI: 10.1016/j.limno.2021.125859

Hughes, R. M., Herlihy, A. T., Gerth, W. J., & Pan, Y. (2012). Estimating vertebrate, benthic macroinvertebrate, and diatom taxa richness in raftable Pacific Northwest rivers for bioassessment purposes. *Environmental Monitoring and Assessment*, 184, 3185–3198. DOI: 10.1007/s10661-011-2181-9

Jucá-Chagas, R., & Boccardo, L. (2006). The air-breathing cycle of *Hoplosternum littorale* (Hancock, 1828) (Siluriformes: Callichthyidae). *Neotropical Ichthyology*, 4(3), 371–373. DOI: 10.1590/S1679-62252006000300009

Kotwicki, S., Lauth, R. R., Williams, K., & Goodman, S. E. (2017). Selectivity ratio: a useful tool for comparing size selectivity of multiple survey gears. *Fisheries Research*, 191, 76–86. DOI: 10.1016/j.fishres.2017.02.012

LaVigne, H. R., Hughes, R. M., & Herlihy, A. T. (2008). Bioassessments to detect changes in Pacific Northwest river fish assemblages: a Malheur River case study. *Northwest Science*, 82, 251–258. DOI: 10.3955/0029-344X-82.4.251

Leal, C. G., Pompeu, P. S., Gardner, T. A., Leitão, R. P., Hughes, R. M., Kaufmann, P. R., … Barlow, J. (2016). Multi-scale assessment of human-induced changes to Amazonian instream habitats. *Landscape Ecology*, 31(8), 1725–1745. DOI: 10.1007/s10980-016-0358-x

Legendre P. & Legendre L. (2012). Numerical Ecology. Oxford: Elsevier.

Leitão, R. P., Zuanon, J., Mouillot, D., Leal, C. G., Hughes, R. M., Kaufmann, P. R., … Gardner, T. A. (2018). Disentangling the pathways of land use impacts on the functional structure of fish assemblages in Amazon streams. *Ecography*, 41(1), 219–232. DOI: 10.1111/ecog.02845

Linares, M. S., Assis, W., de Castro Solar, R. R., Leitão, R. P., Hughes, R. M., & Callisto, M. (2019). Small hydropower dam alters the taxonomic composition of benthic macroinvertebrate assemblages in a neotropical river. *River Research and Applications*, 35, 725–735. DOI: 10.1002/rra.3442

Loures, R. C., & Pompeu, P. S. (2018). Long-term study of reservoir cascade in south-eastern Brazil reveals spatio-temporal gradient in fish assemblages. *Marine and Freshwater Research*, 69, 1983-1994. DOI: 10.1071/mf18109

Luino, F., & De Graff, J. V. (2012). The Stava mudflow of 19 July 1985 (Northern Italy): a disaster that effective regulation might have prevented. *Natural Hazards and Earth System Science*, 12, 1029–1044. DOI: 10.5194/nhess-12-1029-2012

Magurran, A.E. (2013). Measuring Biological Diversity. New York: John Wiley & Sons.

Maia, B. P., Ribeiro, S. M. F., Bizzotto, P. M., Vono, V., & Godinho, H. P. (2007). Reproductive activity and recruitment of the yellow-mandi *Pimelodus maculatus* (Teleostei: Pimelodidae) in the Igarapava Reservoir, Grande River, Southeast Brazil. *Neotropical Ichthyology*, 5(2), 147–152. DOI: 10.1590/S1679-62252007000200008

Marta-Almeida, M., Mendes, R., Amorim, F. N., Cirano, M., & Dias J. M. (2016). Fundão Dam collapse: oceanic dispersion of River Doce after the greatest Brazilian environmental accident. *Marine Pollution Bulletin*, 112, 359–364. DOI: 10.1016/j.marpolbul.2016.07.039

Martins, I., Sanches, B., Kaufmann, P. R., Hughes, R. M., Santos, G. B., Molozzi, J., & Callisto M. (2015). Ecological assessment of a southeastern Brazil reservoir. *Biota Neotropica*, 15, e20140061. DOI: 10.1590/1676-06032015006114

Martins, R. T., Brito, J., Dias-Silva, K., Leal, C. G., Leitão, R. P., Oliveira, V. C., ... Hughes R. M. (2021). Low forest-loss thresholds threaten Amazonian fish and macroinvertebrate assemblage integrity. *Ecological Indicators*, 127, 107773. DOI: 10.1016/j.ecolind.2021.107773

Mascarenhas, F. P. (1898). Sobre o Rio Doce: memória dos trabalhadores statisticos e topográficos das margens do Rio Doce, e esus principais confluentes, tirados pelo Alferes Francisco de Paula Mascarenhas, na viagem que fez ao arraial de Cuithe - Ouro Preto, 1832. *Imprensa Oficial de Minas Gerais*, 3, 53–64

Matarazzo, A., & Sgandurra, M. (2018). Hydropower as an important renewable energy source. *International Journal of Natural Resource Ecology and Management*, 3, 66. DOI: 10.11648/j.ijnrem.20180304.14

Matthews, W.J. (2012). Patterns in Freshwater Fish Ecology. New York: Springer.

Mazzoni R. & Petito J.T. (2012). Distribuição, abundância e estrutura das populações de peixes antes e depois do represamento. Chapter 4. In: Usina Hidrelétrica de Serra da Mesa: 15 Anos de Estudos da Ictiofauna do Alto Tocantins. (Eds: R. Mazzoni, E.P. Caramaschi & R. Iglesias-Rios), Furnas Centrais Hidrelétricas, Rio de Janeiro.

Miranda-Chumacero, G., Álvarez, G., Luna, V., Wallace, R. B., & Painter L. (2015). First observations on annual massive upstream migration of juvenile catfish *Trichomycterus* in an Amazonian River. *Environmental Biology of Fishes*, 98, 1913–1926. DOI: 10.1007/s10641-015-0407-3

Morgenstern, N. R., Vick, S. G., Viotti, C. B., & Watts, B. D. (2016). Relatório Sobre as Causas Imediatas da Ruptura da Barragem de Fundão. Fundação RENOVA, Belo Horizonte, MG.

Muniz, C. M., Frota, A., Ganassin, M. J. M., Agostinho, A. A., & Gomes, L. C. (2021). Do river basins influence the composition of functional traits of fish assemblages in Neotropical reservoirs? *Brazilian Journal of Biology*, 81, 765–775. DOI: 10.1590/1519-6984.230833

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. P., & Kent J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, 403, 853–858

Neves, A. C. O., Nunes, F. P., de Carvalho, F. A., & Fernandes, G. W. (2016). Neglect of ecosystems services by mining, and the worst environmental disaster in Brazil. *Natureza & Conservação*, 14, 24–27. DOI: 10.1016/j.ncon.2016.03.002

Nimmo, D. G., Mac Nally, R., Cunningham, S. C., Haslem, A., & Bennett, A. F. (2015). Vive la résistance: reviving resistance for 21st century conservation. *Trends in Ecology & Evolution*, 30, 516–523. DOI: 10.1016/j.tree.2015.07.008

O'Hanley, J. R., Pompeu, P. S., Louzada, M., Zambaldi, L. P., & Kemp, P. S. (2020). Optimizing hydropower dam location and removal in the São Francisco River basin, Brazil to balance hydropower and river biodiversity tradeoffs. *Landscape and Urban Planning*, 195, 103725. DOI: 10.1016/j.landurbplan.2019.103725

Okada, E. K., Agostinho, A. A., & Gomes, L. C. (2005). Spatial and temporal gradients in artisanal fisheries of a large Neotropical reservoir, the Itaipu Reservoir, Brazil. *Canadian Journal of Fisheries and Aquatic Sciences*, 62(3), 714–724. DOI: 10.1139/f05-015

Oksanen J., Blanchet F. G., Kindt R., Legendre P., Minchin P. R., O'Hara R. B., … Wagner H. (2015). VEGAN: Community Ecology Package. R package, version 2.5-7. Retrieved from https://cran.r-project.org/web/packages/vegan/vegan.pdf

Omachi, C. Y., Siani, S. M. O., Chagas, F. M., Mascagni, M. L., Cordeiro, M., Garcia, G. D., … Thompson, F. L. (2018). Atlantic Forest loss caused by the world's largest tailing dam collapse (Fundão Dam, Mariana, Brazil). *Remote Sensing Applications: Society and Environment*, 12, 30–34. DOI: 10.1016/j.rsase.2018.08.003

Pelicice, F. M., & Agostinho, A. A. (2008). Fish-passage facilities as ecological traps in large neotropical rivers. *Conservation Biology*, 22, 180–188. DOI: 10.1111/j.1523-1739.2007.00849.x

Petesse, M. L., & Petrere, M. (2012). Tendency towards homogenization in fish assemblages in the cascade reservoir system of the Tietê river basin, Brazil. *Ecological Engineering*, 48, 109–116. DOI: 10.1016/j.ecoleng.2011.06.033

Phelps Q.E., Baerwaldt K., Chen D., Du H., Shen L., Wang C., … Hoover J.J. (2016). Paddlefishes and sturgeons of the Yangtze and Mississippi Rivers: status, biology, and management. Chapter 2.1. In: Fisheries Resources, Environment, and Conservation in the Mississippi and Yangtze (Changjiang) River Basins. (Eds: Y. Chen, D.C. Chapman, J.R. Jackson, D. Chen, Z. Li, K.J. Kilgore, Q. Phelps & M.A. Eggleton), American Fisheries Society, Bethesda.

Pinto, B. C. T., Araujo, F. G., & Hughes, R. M. (2006). Effects of landscape and riparian condition on a fish index of biotic integrity in a large southeastern Brazil river. *Hydrobiologia*, 556, 69–83. DOI: 10.1007/s10750-005-9009-y

Pinto, O. (1945). Cinqüenta anos de investigação ornitológica: história das origens e do desenvolvimento da coleção ornitológica do Museu Paulista e de seu subseqüente progresso no departamento de zoologia da secretaria da agricultura. *Arquivso de Zoologia do Estado de São Paulo*, 4, 26–340.

Pires, A. P. F., Rezende, C. L., Assad, E. D., Loyola, R., & Scarano, F. R. (2017). Forest restoration can increase the Rio Doce watershed resilience. *Perspectives in Ecology and Conservation*, 15, 187–193. DOI: 10.1016/j.pecon.2017.08.003

Poff, N. L. R., Bledsoe, B. P., & Cuhaciyan, C. O. (2006). Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. *Geomorphology,* 79, 264–285. DOI: 10.1016/j.geomorph.2006.06.032

Pompeu, P. S., Agostinho, A. A., & Pelicice, F. M. (2012). Existing and future challenges: The concept of successful fish passage in South America. *River Research and Applications*, 28, 504–512. DOI: 10.1002/rra.1557

R Core Team (2017). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Rico, M., Benito, G., & Díez-Herrero, A. (2008). Floods from tailings dam failures. *Journal of Hazardous Materials*, 154, 79–87. DOI: 10.1016/j.jhazmat.2007.09.110

Rudorff, N., Rudorff, C. M., Kampel, M., & Ortiz, G. (2018). Remote sensing monitoring of the impact of a major mining wastewater disaster on the turbidity of the Doce River plume off the eastern Brazilian coast. *ISPRS Journal of Photogrammetry and Remote Sensing*, 145, 349–361. DOI: 10.1016/j.isprsjprs.2018.02.013

Ruschi, A. (1965). Lista dos tubarões, raias e peixes de água doce e salgada do estado do Espírito Santo e uma observação sobre a introdução do dourado no rio Doce. *Boletim do Museu de Biologia Professor Mello Leitão*, 25, 1–22.

Sales, N. G., Mariani, S., Salvador, G. N., Pessali, T. C., & Carvalho, D. C. (2018). Hidden diversity hampers conservation efforts in a highly impacted neotropical river system. *Frontiers in Genetics*, 9, 1–11. DOI: 10.3389/fgene.2018.00271

Salvador, G. N., Leal, C. G., Brejão, G. L., Pessali ,T. C., Alves, C. B. M., Rosa, G. R., ... Montag, L. F. A. (2020). Mining activity in Brazil and negligence in action. *Perspectives in Ecology and Conservation*, 18, 139–144. DOI: 10.1016/j.pecon.2020.05.003

Sanches, B. O., Hughes, R. M., Macedo, D. R., Callisto, M., & Santos, G. B. (2016). Spatial variations in fish assemblage structure in a southeastern Brazilian reservoir. *Brazilian Journal of Biology*, 76, 185–193. DOI: 10.1590/1519-6984.16614

Santana, F. C., Francelino, M. R., Schaefer, C. E. G. R., Veloso, G. V., Fernandes-Filho, E. I., Santana, A. J. P., ... Rosa, A. P. (2021). Water quality of the Gualaxo do Norte and Carmo rivers after the Fundão dam collapse, Mariana, MG. *Water, Air, & Soil Pollution*, 232, 1–13. DOI: 10.1007/s11270-021-05113-3

Santos G.B., Pompeu P. S., Salvador G.N., Pessali T.C., Rabelo N.V., Leal P.H.R. & Mesquita E.D.S. (2012). Marcação e recaptura da curimbatá-pioua (*Prochilodus costatus*) em um afluente do rio São Francisco e avaliação da necessidade de transposição de peixes na UHE Gafanhoto. Chapter 2. In: Série Peixe Vivo: transposição de peixes. (Eds: J.M. Lopes & F.O. Silva). Companhia Energética de Minas Gerais, Belo Horizonte.

Schinegger, R., Palt, M., Segurado, P., & Schmutz, S. (2016). Untangling the effects of multiple human stressors and their impacts on fish assemblages in European running waters. *Science of* *the Total Environment*, 573, 1079–1088. DOI: 10.1016/j.scitotenv.2016.08.143

Soares, B. E., Cabral, G. L., Estrella F., & Caramaschi, E. P. (2017). Two-decade remaining effects of bauxite tailings on the fish taxonomic structure of a clear-water floodplain lake in central Amazon (Batata lake, Pará state, Brazil). *Oecologia Australis*, 21, 311–322. DOI: 10.4257/oeco.2017.2103.08

Su, G., Villéger, S., & Brosse, S. (2020). Morphological sorting of introduced freshwater fish species within and between donor realms. *Global Ecology and Biogeography*, 29, 803–813. DOI: 10.1111/geb.13054

Swales, S., Storey, A. W., Roderick, I. D., Figa, B. S., Bakowa, K. A., & Tenakanai, C. (1998). Biological monitoring of the impacts of the Ok Tedi copper mine on fish populations in the Fly River system, Papua New Guinea. *Science of The Total Environment*, 214, 99–111. DOI: 10.1016/S0048-9697(98)00057-6

Terra, B. D. F., Hughes, R. M., Francelino, M. R., & Araújo, F. G. (2013). Assessment of biotic condition of Atlantic Rain Forest streams: a fish-based multimetric approach. *Ecological Indicators*, 34, 136–148. DOI: 10.1016/j.ecolind.2013.05.001

Thornton K.W., Kennedy R.H., Carroll J.H., Walker W.W., Gunkel R.C. & Ashby S. (1981). Reservoir sedimentation and water quality: an heuristic model. Pages 654-661. In: Proceedings of the Symposium on Surface Water Impoundments. (Ed: H.G. Stefan). American Society of Civil Engineers, Minneapolis.

Twardek, W. M., Cowx, I. G., Lapointe, N. W. R., Paukert, C., Beard, T. D., Bennett, E. M., … Cooke, S. J. (2022). Bright spots for inland fish and fisheries to guide future hydropower development. *Water Biology and Security*, 1, 100009. DOI: 10.1016/j.watbs.2022.100009

Vergilio, C. S., Lacerda, D., Souza, T. S., de Oliveira, B. C. V., Fioresi, V. S., de Souza, V. V., ... de Resende, C. E. (2021). Immediate and long-term impacts of one of the worst mining tailing dam failure worldwide (Bento Rodrigues, Minas Gerais, Brazil). *Science of The Total Environment*, 756, 143697. DOI: 10.1016/j.scitotenv.2020.143697

Viana, J. P. (2017). As atividades de pesca e aquicultura na bacia do rio Doce: subsídios para a mitigação dos impactos socioeconômicos do desastre da Samarco em Mariana, Minas Gerais. *IPEA - Boletim Regional, Urbano e Ambiental*, 16, 103–114

Vieira, F. (2006). A ictiofauna do Rio Santo Antônio, bacia do Rio Doce, MG: proposta de conservação. PhD Thesis, Universidade Federal de Minas Gerais, Belo Horizonte.

Vieira, F. (2009). Distribuição, impactos ambientais e conservação da fauna de peixes da bacia do rio Doce. *MG – Biota*, 2, 5–22.

Ward, J. V., & Stanford, J. A. (1995). The serial discontinuity concept: extending the model to floodplain rivers. *Regulated Rivers: Research & Management*, 10, 159–168.

Weber, A. A., Sales, C. F., Faria, F. S., Melo, R. M. C., Bazzoli, N., & Rizzo, E. (2020). Effects of metal contamination on liver in two fish species from a highly impacted neotropical river: A case study of the Fundão dam, Brazil. *Ecotoxicology and Environmental Safety*, 190, 110165. DOI: 10.1016/j.ecoenv.2020.110165

Winemiller, K. O., McIntyre, P. B., Castello, L., Fluet-Chouinard, E., Giarrizzo, T., Nam, S., … Sáenz, L. (2016). Balancing hydropower and biodiversity in the Amazon, Congo, and Mekong. *Science*, 351, 128–129. DOI: 10.1126/science.aac7082

Wohl, E., Lininger, K. B., & Baron, J. (2017). Land before water: the relative temporal sequence of human alteration of freshwater ecosystems in the conterminous United States. *Anthropocene*, 18, 27–46. DOI: 10.1016/j.ancene.2017.05.004

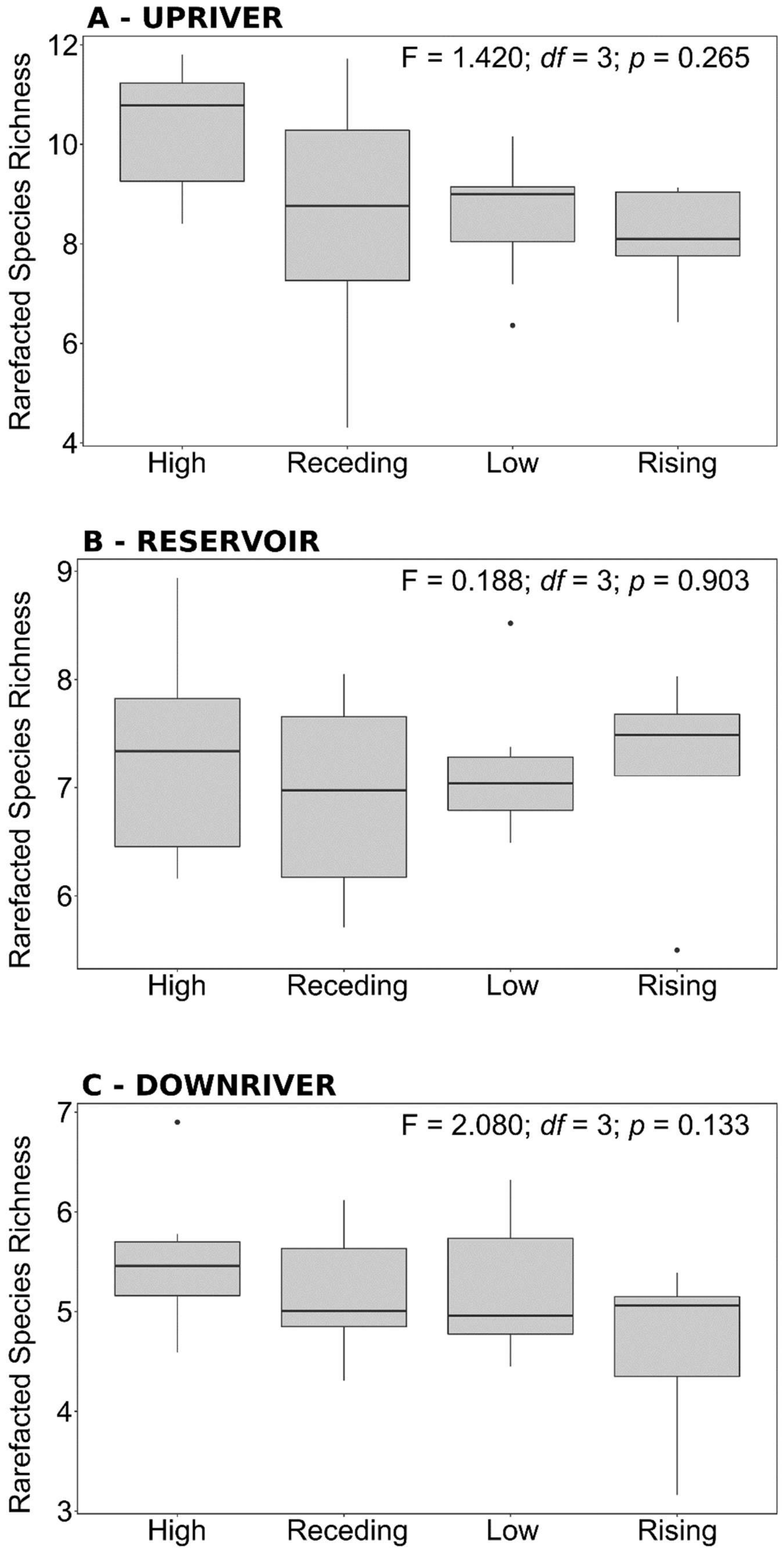
Woody, C. A., Hughes, R. M., Wagner, E. J., Quinn, T. P., Roulson, L. H., Martin, L. M., … Griswold, K. (2010). The mining law of 1872: change is overdue. *Fisheries*, 35, 321–331. DOI: 10.1577/1548-8446-35.7.321

Yoder C.O., Rankin E.T., Gordon V.L., Hersha L.E., & Boucher C.E. (2019). Degradation and recovery of the Scioto River (Ohio, USA) fish assemblage from presettlement to present-day conditions. Chapter 1.9. In: From Catastrophe to Recovery: Stories of Fishery Management Success. (Eds: C. Krueger, W. Taylor & S.-J. Youn). American Fisheries Society, Bethesda (Maryland).

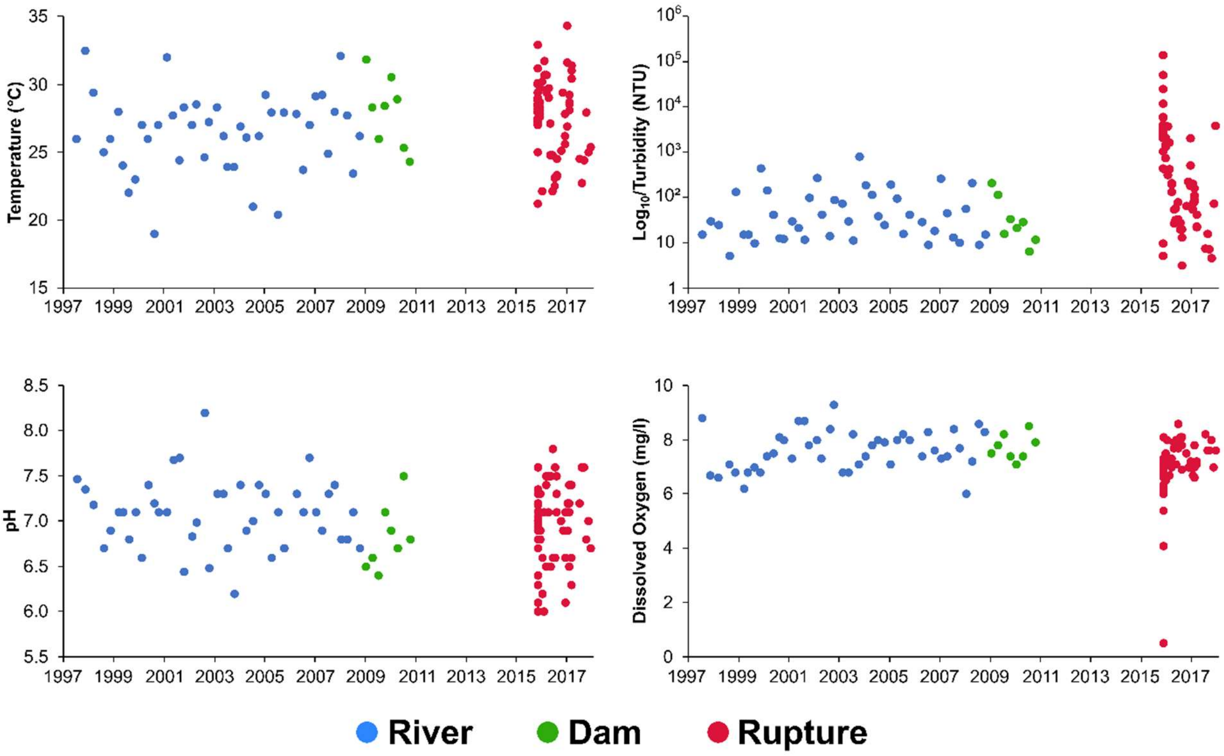
Zambaldi, L., & Pompeu, P. S. (2020). Evaluation of river fragmentation and implications for the conservation of migratory fish in Southeastern Brazil. *Environmental Management*, 65, 702–709. DOI: 10.1007/s00267-020-01266-9

Zar, J.H. (2010). Biostatistical Analysis. Englewood Cliffs: Prentice Hall.

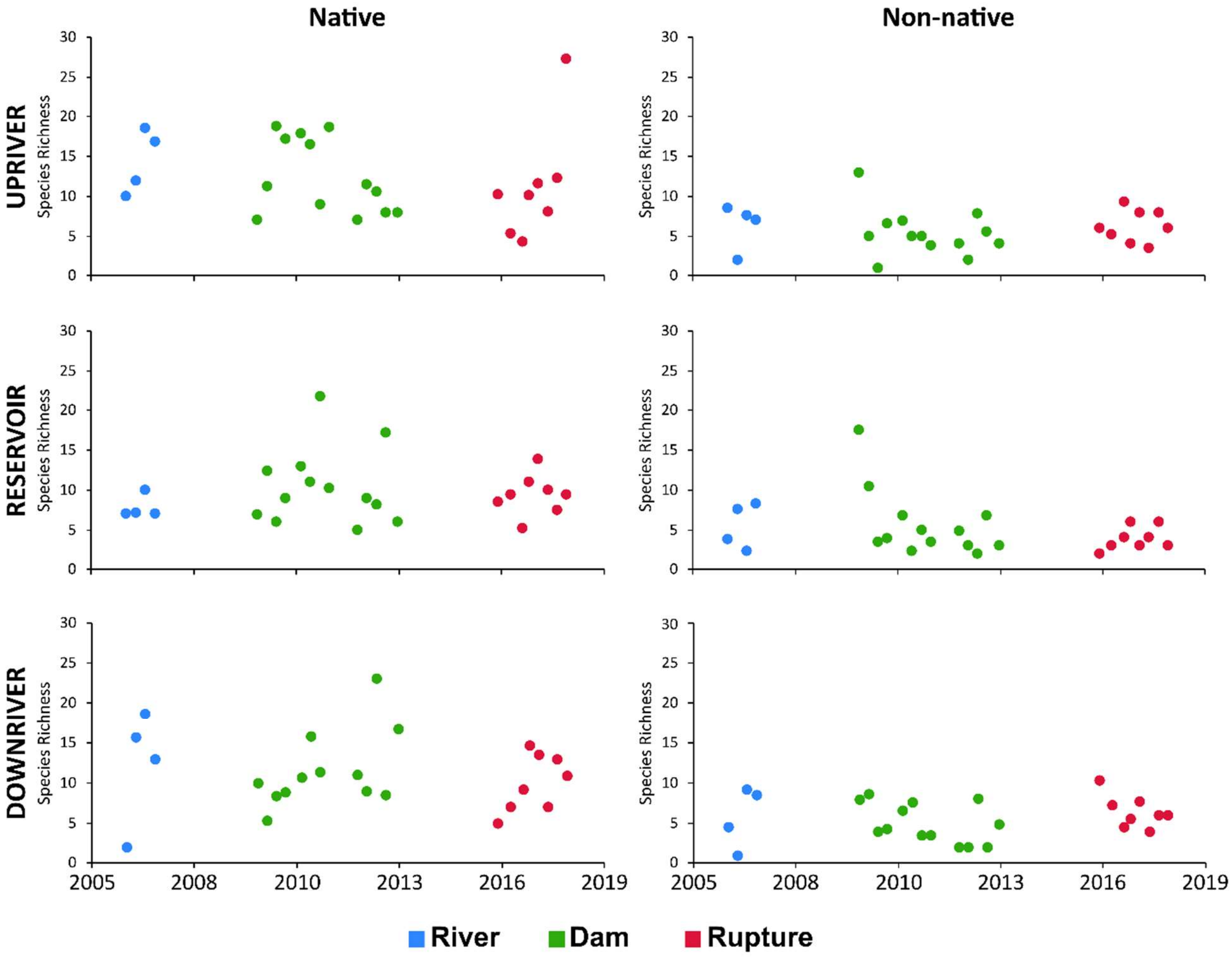
SUPPLEMENTARY MATERIAL



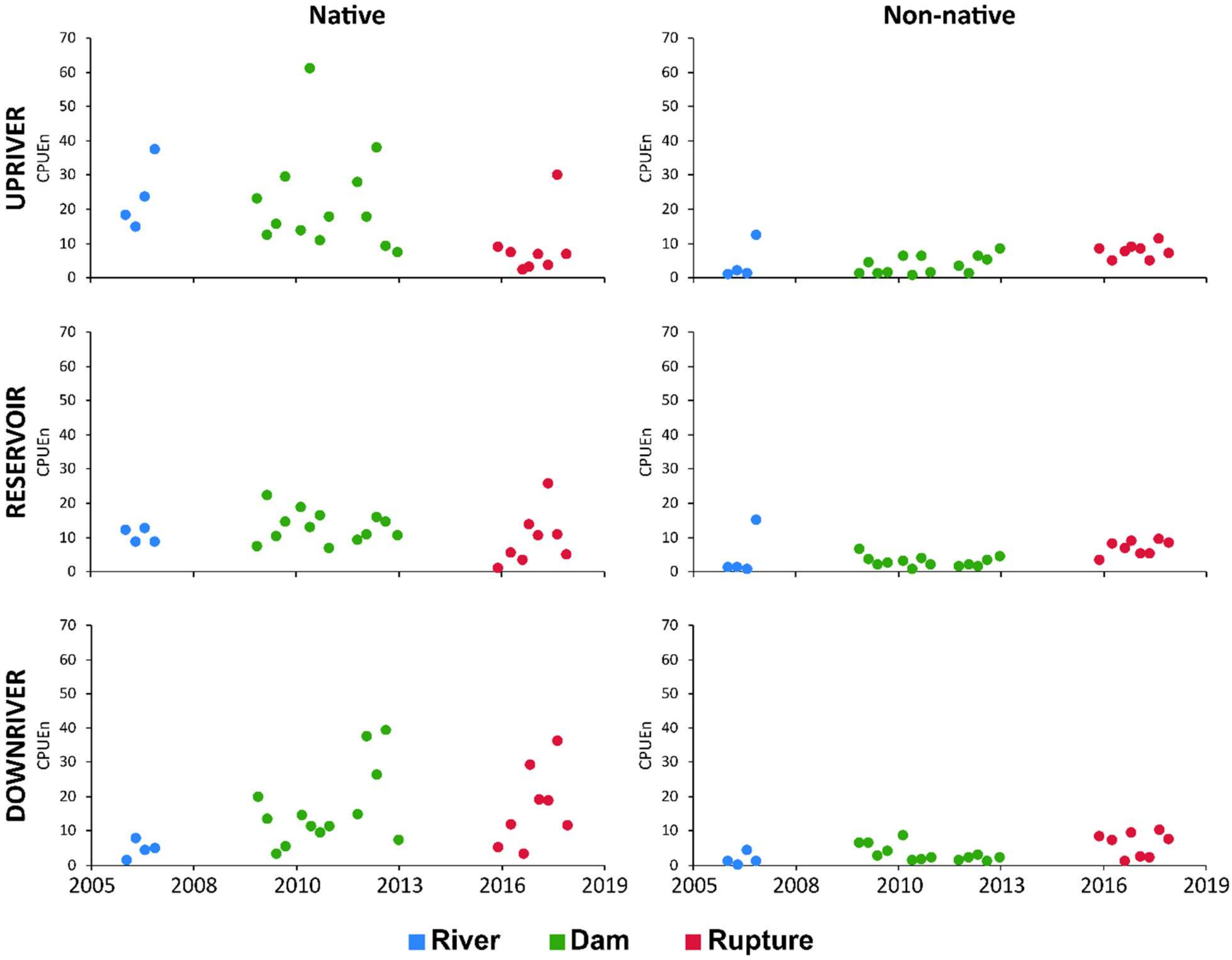
Supplementary 2-1. Species richness variation through seasons in the middle Rio Doce.



Supplementary 2-2. Water quality variables through time in the middle Rio Doce sampling phases.



Supplementary 2-3. Native and non-native extrapolated species richness variation through time in the middle Rio Doce during the three sampling phases (River = pre-dam; Dam = dam post-filling; Rupture = TSF rupture) by section (UPRIVER, RESERVOIR and DOWNRIVER).



Supplementary 2-4. Native and non-native abundance in CPUEn variation through time in the middle Rio Doce during the three sampling phases (River = pre-dam; Dam = dam post-filling; Rupture = TSF rupture) by section (UPRIVER, RESERVOIR and DOWNRIVER).

Supplementary 2-5. Post-hoc Tukey test for native CPUE between sections and phases.

	Sections	**UPRIVER**			**RESERVOIR**			**DOWNRIVER**		
Sections	**Phase**	**River**	**Dam**	**Rupture**	**River**	**Dam**	**Rupture**	**River**	**Dam**	**Rupture**
UPRIVER	**River**	#								
	Dam	1.000	#							
	Rupture	0.073	**0.011**	#						
RESERVOIR	**River**	0.805	0.767	0.948	#					
	Dam	0.866	0.765	0.386	1.000	#				
	Rupture	0.107	**0.021**	1.000	0.977	0.521	#			
DOWNRIVER	**River**	**0.015**	**0.003**	0.963	0.509	0.084	0.924	#		
	Dam	0.947	0.920	0.226	0.998	1.000	0.332	**0.045**	#	
	Rupture	0.932	0.910	0.432	1.000	1.000	0.559	0.100	1.000	#

Supplementary 2-6. Post-hoc Tukey test for non-native CPUE between phases.

	RIVER	**DAM**	**RUPTURE**
RIVER	#		
DAM	0.251	#	
RUPTURE	**0.000**	**0.000**	#

Supplementary 2-7. Most important species selected by SIMPER for distinguishing the UPRIVER, RESERVOIR and DOWNRIVER sections during the three study periods (River, Dam, Rupture) between 2006 and 2018. † Non-native species.

UPRIVER

Taxon	Av. Dis.	Contr. (%)	Cum. (%)	Mean		
				River	Dam	Rupture
Oligosarcus acutirostris	4.63	8.37	8.37	0.85	0.95	0.22
Astyanax lacustris	4.53	8.19	16.55	1.47	1.53	0.82
Pimelodus maculatus†	4.12	7.43	23.98	0.08	0.73	1.14
Trachelyopterus striatulus	3.09	5.57	29.56	0.66	0.38	0.30
Loricariichthys castaneus	2.97	5.36	34.92	0.82	0.62	0.32
Pogonopoma wertheimeri†	2.59	4.68	39.59	0.51	0.44	0.40
Hoplosternum littorale†	2.54	4.59	44.18	0.25	0.09	0.56
Pygocentrus nattereri †	2.53	4.57	48.75	0.25	0.02	0.46
Gymnotus aff. *carapo*	2.30	4.16	52.91	0.52	0.34	0.10
Hypostomus affinis	2.30	4.15	57.06	0.71	0.46	0.71
Prochilodus costatus†	2.22	4.01	61.06	0.40	0.27	0.25
Cyphocharax gilbert	2.13	3.85	64.92	0.29	0.34	0.08
Pseudauchenipterus affinis	2.07	3.73	68.65	0.18	0.26	0.27
Hoplias intermedius	1.50	2.71	71.36	0.35	0.17	0.11
Megaleporinus conirostris	1.34	2.42	73.78	0.08	0.15	0.21
Hoplias gr. *malabaricus*	1.29	2.33	76.11	0.23	0.15	0.17
Salminus brasiliensis †	1.22	2.21	78.32	0.18	0.14	0.10
Psalidodon aff. *fasciatus*	1.21	2.18	80.50	0.08	0.20	---
Clarias gariepinus†	1.12	2.02	82.52	0.35	0.09	---
Pachyurus adspersus	1.05	1.89	84.41	---	---	0.23
Lophiosilurus alexandri†	0.91	1.65	86.05	0.15	0.11	0.04
Prochilodus vimboides	0.90	1.63	87.69	0.28	0.02	0.08
Oreochromis niloticus†	0.82	1.48	89.17	---	0.07	0.12
Glanidium botocudo	0.77	1.39	90.55	0.25	---	0.04
Geophagus brasiliensis	0.73	1.32	91.87	0.15	0.07	0.04
Cichla cf. *kelberi*†	0.72	1.29	93.17	0.12	0.08	0.04
Coptodon rendalli†	0.57	1.04	94.20	---	---	0.11
Brycon dulcis	0.56	1.01	95.21	---	0.05	0.11
Australoheros cf. *ipatinguensis*	0.48	0.86	96.07	---	---	0.10
Pimelodella sp.	0.33	0.59	96.66	0.12	---	---
Hypostomus luetkeni	0.33	0.59	97.25	---	0.06	---
Metynnis cf. *lippincottianus*†	0.31	0.57	97.82	---	0.02	0.04
Delturus carinotus	0.31	0.56	98.38	0.08	0.02	---
Synbranchus sp.	0.21	0.37	98.75	0.08	---	---
Megaleporinus obtusidens†	0.19	0.35	99.10	---	0.04	---
Deuterodon aff. *taeniatus*	0.15	0.27	99.37	0.08	---	---
Crenicichla lacustris	0.14	0.25	99.62	---	---	0.04
Astyanax sp.	0.11	0.21	99.83	---	0.02	---
Hypomasticus copelandii	0.10	0.17	100.00	---	0.02	---

RESERVOIR

Taxon	Av. Dis.	Contr. (%)	Cum (%)	Mean		
				River	Dam	Rupture
Pygocentrus nattereri†	5.43	9.46	9.46	0.25	0.26	1.15
Astyanax lacustris	4.31	7.52	16.98	1.24	0.83	0.68
Megaleporinus conirostris	3.80	6.61	23.59	---	0.62	0.20
Trachelyopterus striatulus	3.76	6.55	30.14	0.18	0.72	0.49
Oligosarcus acutirostris	3.65	6.37	36.51	0.48	0.68	0.35
Pachyurus adspersus	3.37	5.86	42.37	---	---	0.69
Loricariichthys castaneus	2.96	5.16	47.53	0.66	0.80	0.70
Prochilodus costatus†	2.74	4.77	52.30	0.45	0.33	0.26
Hoplosternum littorale†	2.60	4.53	56.83	0.31	0.14	0.34
Pimelodus maculatus†	2.42	4.22	61.05	0.37	0.59	0.71
Hypostomus affinis	2.36	4.11	65.16	0.20	0.64	0.38
Pseudauchenipterus affinis	2.33	4.07	69.22	0.08	0.32	0.26
Gymnotus aff. *carapo*	2.29	3.99	73.21	0.60	0.23	0.14
Hoplias gr. *malabaricus*	1.85	3.23	76.44	0.18	0.15	0.26
Pogonopoma wertheimeri†	1.60	2.79	79.23	---	0.16	0.20
Prochilodus vimboides	1.26	2.19	81.41	0.33	0.05	0.08
Clarias gariepinus†	1.21	2.11	83.52	0.27	0.10	---
Hoplias intermedius	1.19	2.07	85.59	0.08	0.19	0.08
Lophiosilurus alexandri†	1.13	1.97	87.56	0.27	0.08	---
Psalidodon aff. *fasciatus*	0.99	1.73	89.29	0.15	0.11	---
Hypostomus luetkeni	0.94	1.64	90.93	---	0.14	0.04
Glanidium botocudo	0.91	1.58	92.51	0.23	0.02	---
Salminus brasiliensis†	0.89	1.55	94.06	0.20	0.07	0.04
Coptodon rendalli†	0.85	1.48	95.54	0.23	0.05	---
Cichla cf. *kelberi*†	0.71	1.23	96.77	---	0.08	0.08
Geophagus brasiliensis	0.53	0.92	97.69	0.15	---	---
Hypomasticus copelandii	0.37	0.64	98.32	---	0.02	0.04
Oreochromis niloticus†	0.25	0.43	98.76	---	0.05	---
Megaleporinus obtusidens†	0.24	0.42	99.18	---	0.04	---
Cyphacharax gilbert	0.19	0.34	99.52	---	---	0.04
Crenicichla lacustris	0.16	0.28	99.80	---	---	0.04
Rhamdia quelen	0.12	0.20	100.00	---	0.02	---

DOWNRIVER

Taxon	Av. Dis.	Contr. (%)	Cum. (%)	Mean		
				River	Dam	Rupture
Hypostomus luetkeni	5.89	9.34	9.34	---	1.05	0.29
Astyanax lacustris	4.63	7.34	16.68	0.65	0.96	1.01
Oligosarcus acutirostris	4.53	7.18	23.86	---	0.60	0.79
Pygocentrus nattereri†	4.45	7.05	30.91	0.08	0.20	0.94
Pachyurus adspersus	4.27	6.76	37.68	---	---	0.87
Trachelyopterus striatulus	4.21	6.67	44.35	0.20	0.39	0.87
Pimelodus maculatus†	3.18	5.03	49.38	0.41	0.69	0.52
Hypostomus affinis	2.74	4.35	53.73	0.41	0.56	0.77
Prochilodus costatus†	2.45	3.89	57.62	0.20	0.33	0.40
Delturus carinotus	2.38	3.76	61.38	0.15	0.36	---
Megaleporinus conirostris	2.17	3.44	64.82	0.20	0.26	0.17
Hoplias intermedius	2.12	3.37	68.19	0.15	0.26	0.53
Gymnotus aff. *carapo*	1.92	3.04	71.23	0.31	0.15	0.11
Clarias gariepinus†	1.92	3.04	74.27	0.27	0.20	0.06
Lophiosilurus alexandri†	1.73	2.75	77.01	0.15	0.29	0.04
Hoplosternum littorale†	1.67	2.64	79.66	0.08	0.02	0.32
Salminus brasiliensis†	1.41	2.23	81.89	0.08	0.12	0.22
Loricariichthys castaneus	1.39	2.19	84.08	0.08	0.24	0.08
Glanidium botocudo	1.30	2.06	86.14	0.36	---	---
Hoplias gr. *malabaricus*	1.23	1.95	88.09	---	0.15	0.16
Psalidodon aff. *fasciatus*	1.04	1.65	89.74	0.12	0.12	---
Pseudauchenipterus affinis	0.96	1.52	91.26	0.08	0.08	0.06
Cichla cf. *kelberi*†	0.95	1.50	92.76	---	0.15	0.04
Pogonopoma wertheimeri†	0.55	0.87	93.63	---	0.04	0.08
Oreochromis niloticus†	0.50	0.80	94.43	---	---	0.11
Prochilodus vimboides	0.48	0.76	95.19	0.08	---	0.04
Hypomasticus copelandii	0.42	0.66	95.85	---	0.02	0.08
Crenicichla lacustris	0.38	0.60	96.45	---	0.04	0.04
Geophagus brasiliensis	0.28	0.44	96.90	0.08	---	---
Rhamdia quelen	0.27	0.43	97.33	---	0.02	0.04
Deuterodon aff. *taeniatus*	0.27	0.43	97.76	0.08	---	---
Astyanax sp.	0.26	0.42	98.17	0.08	---	---
Cyphocharax gilbert	0.26	0.41	98.58	---	---	0.06
Colossoma macropomum x *Piaractus mesopotamicus*†	0.23	0.36	98.94	---	---	0.06
Megaleporinus obtusidens†	0.21	0.33	99.28	---	0.04	---
Pimelodella sp.	0.16	0.26	99.53	---	0.02	---
Metynnis cf. *lippincottianus*†	0.15	0.23	99.77	---	0.02	---
Gymnotus sylvius	0.15	0.23	100.00	---	---	0.04

3. Sessão 2

Mining activity in Brazil and negligence in action: when the cheap is expensive

A primeira sessão desta tese foi publicada no periódico científico *Perspective in Ecology and Conservation*, disponível em: https://www.sciencedirect.com/science/article/pii/S2530064420300249

MINING ACTIVITY IN BRAZIL AND NEGLIGENCE IN ACTION: WHEN THE CHEAP IS EXPENSIVE

ABSTRACT

The number of tailings dam failures has increased in Brazil. Only in the last six years there have been four disasters involving this type of structure. Despite the great socio-environmental impacts of these disasters, little has been done to mitigate the damage caused. The inaction of mining companies and the absence of river basin-level management approaches have greatly hampered efforts for environmental recovery. In addition, there has been a slowdown in Brazilian environmental legislation, leaving us as a society to question which path we intend to follow in the future, one which prioritizes socio-environmental health and the recovery of degraded ecosystems, or the maintenance of a system that so far has caused untold damage to Brazilian society and the environment.

Keywords: Brumadinho; Mariana; Fundão; Disaster; Tailings dam; Rupture.

Brazil is the second largest producer of mineral ores in the world (National Minerals Information Center/ US Geological Survey, 2017). This prominent position in the global market comes with a downside, the immense environmental liabilities generated along the production chain (Mechi and Sanches, 2010). Tailings dams are one of the main liabilities associated with mining activities, and approximately 600 of these structures are found in Brazil (ANA, 2016). When they fail, the environmental impacts can be huge, and recently the number of collapsed dams has increased both globally and in Brazil (Abdelouas, 2019; Escobar, 2015; Fernandes et al., 2016; Pereira et al., 2019).

Since 1986, eight tailings dams have failed in Brazil, four in just the last six years (Abdelouas, 2019, 2006) (Figure 3-1). The most recent occurred this January when the rupture of the B1 dam at the Córrego do Feijão mine complex in Brumadinho, Minas Gerais, dumped much of the 12 million m3 of stored waste into the surrounding landscape, killing hundreds of people in the process. The magnitude of the disaster and the fact that it happened just over three years after another major dam collapse, the Fundão dam disaster in Mariana (Escobar, 2015), also in Minas Gerais, has reopened discussions about the social and environmental impacts of ore tailings storage. With 700 m in length and 86 m in height, Vale S.A. built Brumadinho's dam to receive waste produced during the enrichment of iron ore. The number of deaths in Brumadinho (231 dead and 41 missing, Defesa Civil de Minas Gerais, 2019) is incomparably higher than that observed in the Mariana disaster, when 19 people lost their lives. These numbers will be near to the records for the dam collapse in Stava, Italy, considered one of the world's worst mining tragedies (Luino and De Graff, 2012). The waste from Brumadinho's dam wiped out the Ferro Carvão stream and its surroundings, reaching the Paraopeba River, which is one of the main tributaries of the Upper São Francisco River basin. Since 1998, the Paraopeba River is considered a high priority for fish conservation, due to the presence of endangered species and the high abundance of migratory species (Alves and Leal, 2010; Costa et al., 1998; Drummond et al., 2005). The region is not only important for fish conservation, the Paraopeba River also shelters the endangered Black-collared swallow, *Pygochelidon melanoleuca*, a river-dwelling bird species (Silva et al., 2017). The river also provides other essential ecosystem services for millions of citizens, being an important source of water for human consumption and agriculture irrigation (da Silva et al., 2015). After the disaster, water contamination occurred along a 271 km stretch downstream, up to the Retiro Baixo Hydroelectric Plant (IBAMA/IEF, 2019) (Figure 3-2).

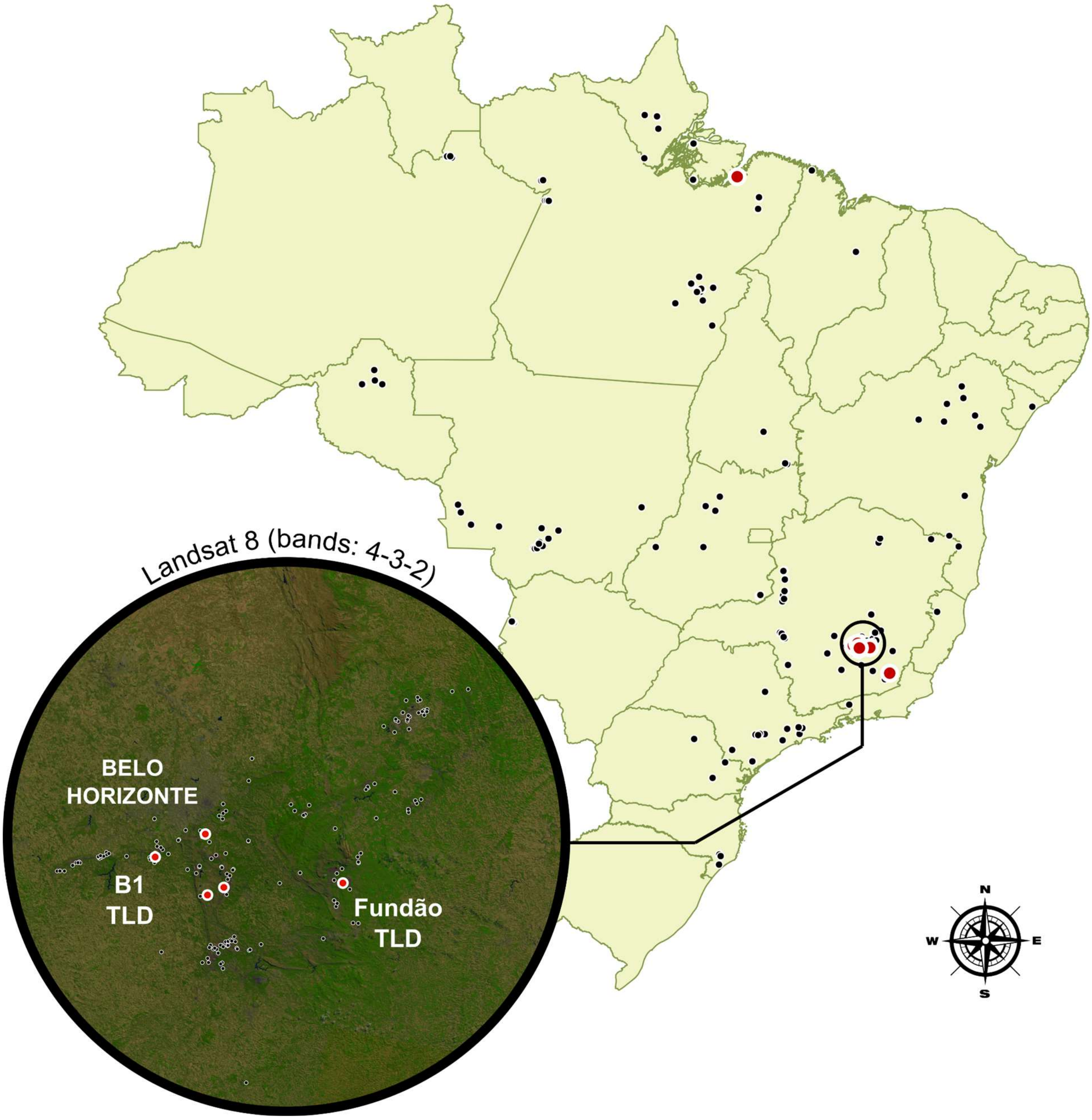


Figure 3-1. Distribution of all tailing's dams along Brazilian territory. Red dots: tailing dams failures; Black dots: tailing dams. Source: Agência Nacional de Mineração (ANM).

Despite the increasing number of collapsed dams in Brazil, little has been done to mitigate the environmental damage. Little has been learned as well. The disaster in Brumadinho join to the Fundão tailings dam failure in Mariana as one of Brazil's worst environmental disaster (Escobar, 2015; Fernandes et al., 2016). Following the Mariana disaster, most of the studies that were made available to society resulted from initiatives other than those carried out by the Renova Foundation (for details see Fundação Renova, 2019), an entity created by the mining company Samarco to understand the environmental impacts and propose solutions. In contrast to the studies of the Renova Foundation, these independent studies included data from prior to the collapse, using a before-after control-impact approach, enabling a more accurate estimate of the environmental impacts. However, these studies could only evaluate a short time and were limited in their scope (e.g. focusing on single taxonomic groups) or covered only a small proportion of the overall affected area (e.g. Hatje et al., 2017; Marta-Almeida et al., 2019; Omachi et al., 2016). These spatial and temporal limitations should have been met quickly by the mining company as a mandatory counterpart, in the form of a robust integrated assessment of all affected ecosystems, freshwater, terrestrial and, when necessary, marine. However, these limitations were not addressed by the mining company.

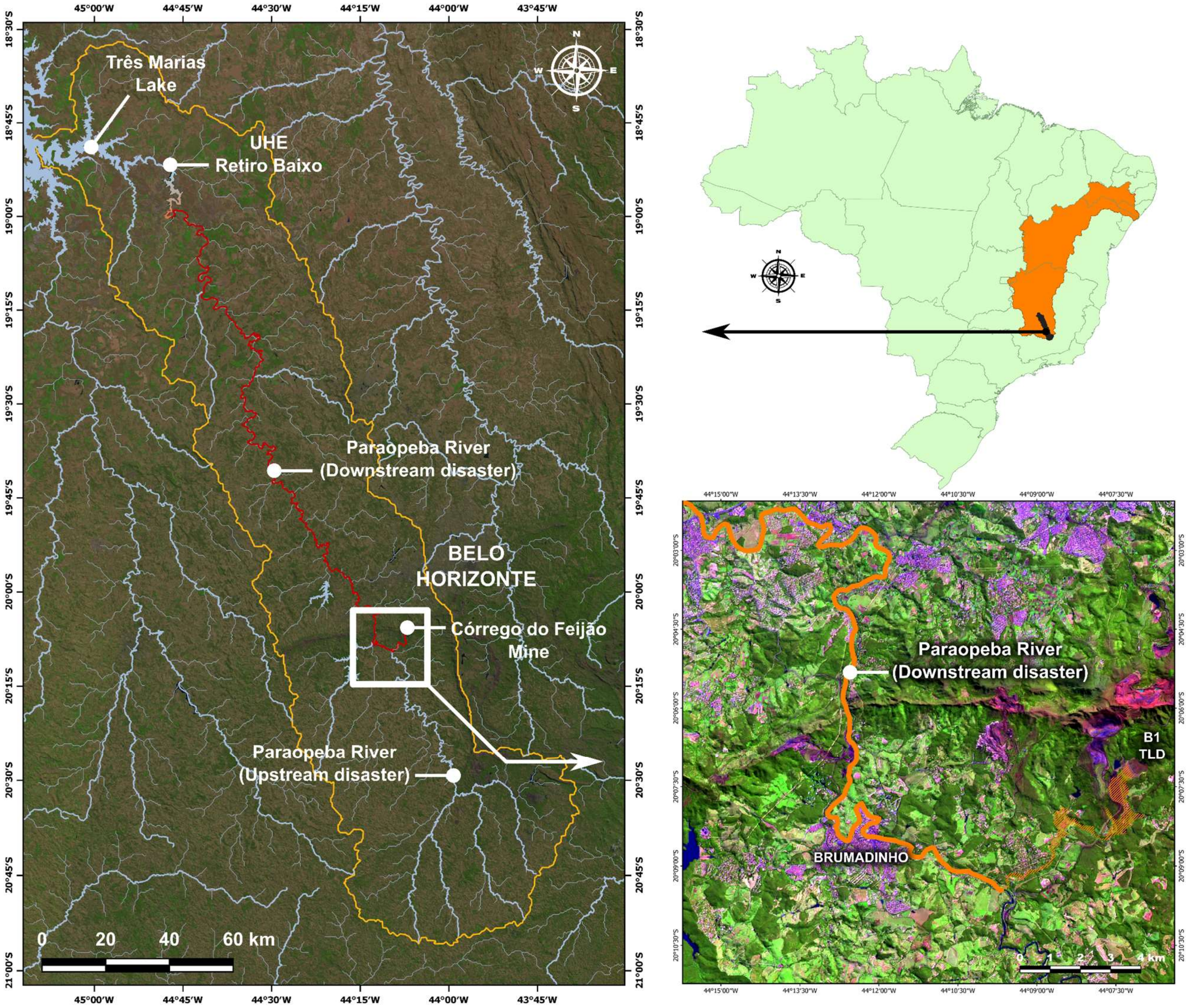


Figure 3-2. Iron ore waste way along Rio Paraopeba basin after the rupture of B1 dam at the municipality of Brumadinho, Minas Gerais, Brazil.

The lack of organization and access to information maintained by environmental agencies on studies about the impacts of mining activities hampers our ability to learn from the past. Data generated in studies carried out along river basins required by the Brazilian government for the licensing and monitoring of mining activities could provide this information. However, although the companies allocate large amounts of money for this purpose, often, this expense does not return good quality data, and this is not an exclusive problem from mines companies (Dias et al., 2017). Even when the results obtained in the monitoring are reliable and could be published in scientific journals, it is necessary to comply with confidentiality agreements, which prevents publication without the consent of the companies involved. And mining companies are generally not keen on making available data that could damage their public image. It would be expected that at least these data would be used for decision-making by Brazilian environmental agencies. However, after being delivered to government agencies, these data are rarely systematically compiled, or used for the development of evidence-based management practices.

In a historical context, all these disasters were not enough to bring governments and society to a serious debate about the risks of mining activities in Brazil, including the problem of tailing dams. What we observe is a process of privatization of regulatory powers in this sector (from Santos and Milanez, 2017), where the whole process of licensing, monitoring, and the environmental recovery following disasters is made by the companies themselves. Since the approval of the provisional measure 790 (Brasil, 2017), this situation seems to have moved towards a darker scenario, seen by the simplification of the environmental licensing and allowance of mineral exploitation inside conservation units, and the territories of indigenous and “quilombola” communities (Lopes and Oliveira, 2018). Given the history of mining activities in Brazil, it is likely that companies will use tailings dams to store ore in those important areas for biodiversity conservation. Other large-scale measures, carried out by executive and legislative powers, also have the potential to relax environmental licensing for these activities (see Azevedo-Santos et al., 2017), leaving the path open for natural resource exploitation without proper socio-environmental responsibilities.

Several Brazilian states, in particular, Minas Gerais and Pará, are economically-dependent on mining activities (Reis and Silva, 2015). The complex relationships between individual states’ and municipalities’ financial rewards, policies, and liabilities for environmental damage from mining makes this an extremely thorny issue. For instance, Mariana disaster was not enough to stop or engender in-depth discussions about the MP 790, which simplified environmental licensing in Brazil (Brasil, 2017). Even after Brumadinho disaster, the discussions were still superficial, only being based on the decommissioning of dams raised by the upstream method (for larger details of construction methods, see Cardozo et al., 2016). Although important, this decision will lead to the decommissioning of only 20% of tailings dams included in the Brazilian national dam safety plan

(ANM, 2019). With the development of more efficient methods of storing mining tailings, or even their recycling (Edraki et al., 2014; Franks et al., 2011; Kinnunen et al., 2018), it remains unclear whether the maintenance of existing tailings dams, and the risks these structures pose to society and the environment, is in fact necessary at all. Are arguments of higher operational costs for mining companies enough for the continuity of such a serious problem? Even with so many human lives lost, the financial costs of rebuilding affected urban areas, and the destruction of the natural ecosystems?

However, some actions in the sector have taken different paths, and show the companies' commitment to environmental problems. After media reports of ore tailings overflow because of a high precipitation event in the state of Pará, the company responsible for the ore beneficiation has partnered with NGOs and public universities to assess the environmental impacts (pers. obs.: Salvador, G. N.). These studies, if truly independent of the company involved, are essential to understand the real extent of the socio-environmental impacts of such disasters. An example of the absence of this was the plume of waste that reached the mouth of the Doce River in 2015 due to the Mariana disaster (Escobar, 2015). Preliminary measurements using satellite images showed that their effect would not reach the Abrolhos archipelago (Marta-Almeida et al., 2016). However, recent studies, three years on from the disaster, indicated the contamination of corals in Abrolhos by tailings (Floresti, 2019). Regarding the Brumadinho disaster, data generated both by independent and official organizations in Paraopeba River are conflicting, and the misinformation produces more apprehension among the general public (IBAMA/IEF, 2019). An example was the extent of fish mortality recorded after Brumadinho, which unlike Mariana where the entire range of Doce River was affected, was concentrated in the first few kilometers downstream from the collapsed dam (IBAMA/IEF, 2019). However, it is plausible to expect further medium and long-term consequences for the ichthyofauna in the Paraopeba River, for instance changes in the species composition and increases in contamination by heavy metals, especially near the dam collapse.

Governmental negligence occurs around the world (Hyndman, 2001; Luino and De Graff, 2012; Smith and Morris, 1992), and Brazil is no different. Little has been learned from the disasters of the last two decades, on both how to avoid problems and the monitoring of environmental restoration efforts. The warnings prior to the Brumadinho disaster were ignored, which led to the loss of hundreds of lives, the pollution of rivers and fisheries by heavy metals, and a change in the biota of the affected watercourses. Maybe now, Brazilian society should wake up and demand tougher regulation of governments and corporations. In this context, it is necessary to reflect on which paths to follow: 1. One which prioritizes socio-environmental health and the recovery of degraded ecosystems, and that incorporates the hard lessons learnt from past mistakes, or 2. To continue on the path traveled until now, with the reduction of operating costs of mining companies trumping socio-environmental impacts. For us, the latter option is a dangerous and irresponsible route where the possibility of new disasters is well-known and extremely high. It is also necessary to go further than close some endangered tailing dams. It is necessary to change the rules for new mining activities and choose recycling methods of ore tailings.

REFERENCES

Abdelouas, A., 2019. World Mine Tailings Failures - from 1915 [WWW Document]. URL https://worldminetailingsfailures.org/ (accessed 2.2.19).

Abdelouas, A., 2006. Uranium Mill Tailings: and Environmental Impact. Elements 2, 335–342. doi:10.2113/gselements.2.6.335

Alves, C.B.M., Leal, C.G., 2010. Aspectos da conservação da fauna de peixes da bacia do rio São Francisco em Minas Gerais. MG Biota 2, 26–44.

ANA, 2016. Relatório de segurança de barragens 2015. Agência Nacional de Águas, Brasília.

ANM, 2019. Nota explicativa sobre tema de segurança de barragens, focado nas barragens construídas ou alteadas pelo método a montante, além de outras especificidades referentes. Brasília, DF. Agência Nacional de Águas, Brasília.

Azevedo-Santos, V.M., Fearnside, P.M., Oliveira, C.S., Padial, A.A., Pelicice, F.M., Lima, D.P., Simberloff, D., Lovejoy, T.E., Magalhães, A.L.B., Orsi, M.L., Agostinho, A.A., Esteves, F.A., Pompeu, P.S., Laurance, W.F., Petrere, M., Mormul, R.P., Vitule, J.R.S., 2017. Removing the abyss between conservation science and policy decisions in Brazil. Biodivers. Conserv. 26, 1745–1752. doi:10.1007/s10531-017-1316-x

Brasil, 2017. Medida Provisória no 790, de 2017. Altera o Decreto-Lei no 227, de 28 de fevereiro de 1967 - Código de Mineração, e a Lei no 6.567, de 24 de setembro de 1978, que dispõe sobre regime especial para exploração e aproveitamento das substâncias minerais que especifica e dá outras providências. Congresso Nacional, Brasília, DF.

Cardozo, F.A.C., Pimenta, M.M., Zingano, A.C., 2016. Métodos construtivos de barragens de rejeitos de mineração – uma revisão. Holos 8, 77–85. doi:10.15628/holos.2016.5367

Costa, C.M.R., Hermann, G., Martins, C.S., 1998. Biodiversidade em Minas Gerais: um atlas para sua conservação. Fundação Biodiversitas, Belo Horizonte.

da Silva, B.M.B. da, da Silva, D.D., Moreira, M.C., 2015. Influência da sazonalidade das vazões nos critérios de outorga de uso da água: estudo de caso da bacia do rio Paraopeba. Ambient. Água - An Interdiciplinary J. Appl. Sci. 10, 623–634. doi:10.4136/1980-993X

Defesa Civil de Minas Gerais, 2019. INFORMAÇÕES: Desastre Barragem de rejeitos de Brumadinho [WWW Document]. URL http://www.defesacivil.mg.gov.br/index.php/component/gmg/page/618-informacoes2703 (accessed 3.28.19).

Dias, A.M. da S., Fonseca, A., Paglia, A.P., 2017. Biodiversity monitoring in the environmental impact assessment of mining projects: a (persistent) waste of time and money? Perspect. Ecol. Conserv. 15, 206–208. doi:10.1016/j.pecon.2017.06.001

Drummond, G.M., Martins, C.S., Machado, A.B.M., Sebaio, F.A., Antonini, Y., 2005. Biodiversidade em Minas Gerais: um Atlas para sua conservação, 2nd ed. Fundação Biodiversitas, Belo Horizonte.

Edraki, M., Baumgartl, T., Manlapig, E., Bradshawb, D., Franks, D.M., Moran, C.J., 2014. Designing mine tailings for better environmental, social and economic outcomes: a review of alternative approaches. J. Clean. Prod. 84, 411–420. doi:10.1016/j.jclepro.2014.04.079

Escobar, H., 2015. Mud tsunami wreaks ecological havoc in Brazil. Science (80-.). 350, 1138–1139. doi:10.1126/science.350.6265.1138

Fernandes, G.W., Goulart, F.F., Ranieri, B.D., Coelho, M.S., Dales, K., Boesche, N., Bustamante, M., Carvalho, F.A., Carvalho, D.C., Dirzo, R., Fernandes, S., Galetti, P.M., Millan, V.E.G., Mielke, C., Ramirez, J.L., Neves, A., Rogass, C., Ribeiro, S.P., Scariot, A., Soares-Filho, B., 2016. Deep into the mud: ecological and socio-economic impacts of the dam breach in Mariana, Brazil. Nat. e Conserv. 14, 35–45. doi:10.1016/j.ncon.2016.10.003

Floresti, F., 2019. Lama da Samarco afetou os corais de Abrolhos, comprova estudo. Rev. Galileu.

Franks, D.M., Boger, D. V., Côte, C.M., Mulligan, D.R., 2011. Sustainable development principles for the disposal of mining and mineral processing wastes. Resour. Policy 36, 114–122. doi:10.1016/j.resourpol.2010.12.001

Fundação Renova, 2019. Arquívos e Relatórios [WWW Document]. URL https://www.fundacaorenova.org/arquivos-e-relatorios/ (accessed 3.20.19).

Hatje, V., Pedreira, R.M.A., de Rezende, C.E., Schettini, C.A.F., de Souza, G.C., Marin, D.C., Hackspacher, P.C., 2017. The environmental impacts of one of the largest tailing dam failures worldwide. Sci. Rep. 7, 10706. doi:10.1038/s41598-017-11143-x

Hyndman, D., 2001. Academic responsibilities and representation of the Ok Tedi crisis in postcolonial Papua New Guinea. Contemp. Pac. 13, 33–54. doi:10.1353/cp.2001.0014

IBAMA/IEF, 2019. Nota técnica no 5/2019/NUBIO-MG/DITEC-NG/SUPES-MG. Instituto Brasileiro do Meio Ambiente e dos Recursos Naturais Renováveis (IBAMA) - Instituto Estadual de Florestas de Minas Gerais (IEF), Belo Horizonte, MG.

Kinnunen, P., Ismailov, A., Solismaa, S., Sreenivasan, H., Räisänen, M.-L., Levänen, E., Illikainen, M., 2018. Recycling mine tailings in chemically bonded ceramics – A review. J. Clean. Prod. 174, 634–649. doi:10.1016/j.jclepro.2017.10.280

Lopes, V.M.C., Oliveira, M.L.R., 2018. Novo marco legal para a mineração e suas implicações para a atividade minerária no Brasil sob a luz da justiça ambiental. Polêm!Ca 18, 33–53. doi:10.12957/polemica.2018.39422

Luino, F., De Graff, J. V., 2012. The Stava mudflow of 19 July 1985 (Northern Italy): A disaster that effective regulation might have prevented. Nat. Hazards Earth Syst. Sci. 12, 1029–1044. doi:10.5194/nhess-12-1029-2012

Marta-Almeida, M., Mendes, R., Amorim, F.N., Cirano, M., Dias, J.M., 2016. Fundão Dam collapse: Oceanic dispersion of River Doce after the greatest Brazilian environmental accident. Mar. Pollut. Bull. 112, 359–364. doi:10.1016/j.marpolbul.2016.07.039

Mechi, A., Sanches, D.L., 2010. Impactos ambientais da mineração no estado de São Paulo. Estud. Avançados 24, 209–220.

National Minerals Information Center/ US Geological Survey, 2017. Global iron ore production data; clarification of reporting from the USGS. Min. Eng. 69, 20–23.

Omachi, C.Y., Siani, S.M.O., Chagas, F.M., Mascagni, M.L., Cordeiro, M., Garcia, G.D., Thompson, C.C., Siegle, E., Thompson, F.L., 2018. Atlantic Forest loss caused by the world's largest tailing dam collapse (Fundão Dam, Mariana, Brazil). Remote Sens. Appl. Soc. Environ. 12, 30–34. doi:10.1016/j.rsase.2018.08.003

Pereira, L.F., Cruz, G. de B., Guimarães, R.M.F., 2019. Impactos do rompimento da barragem de rejeitos de Brumadinho, Brasil: uma análise baseada nas mudanças de cobertura da terra. J. Environ. Anal. Prog. 4, 122–129. doi:10.24221/JEAP.4.2.2019.2373.122-129

Reis, J.C., Silva, H., 2015. Mineração e desenvolvimento em Minas Gerais na década 2000-2010. Novos Cad. NAEA 18, 73–100.

Santos, R.S.P., Milanez, B., 2017. The construction of the disaster and the "privatization" of mining regulation: reflections on the tragedy of the Rio Doce Basin, Brazil. Vibrant Virtual Brazilian Anthropol. 14, 127–149. doi:10.1590/1809-43412017v14n2p127

Silva, G.A., Salvador, G.N., Malacco, G.B., Nogueira, W., Almeida, S.M., 2017. Range and conservation of the regionally Critically Endangered Black-collared Swallow, Pygochelidon melanoleuca (Wied, 1820) (Aves, Hirundinidae), in Minas Gerais, Brazil. Check List 13, 455–459. doi:10.15560/13.5.455

Smith, R.E.W., Morris, T.F., 1992. The impacts of changing geochemistry on the fish assemblages of the Lower Ok Tedi and Middle Fly river, Papua New Guinea. Sci. Total Environ. 125, 321–344. doi:10.1016/0048-9697(92)90399-D

4. Sessão 3

Tailings dams and reservoirs change upstream fish assemblages in Atlantic Forest streams

A terceira sessão desta tese foi elaborada e formatada conforme as normas do periódico científico *Water, Beiology and Security*, disponível em: https://www.keaipublishing.com/en/journals/water-biology-and-security/guide-for-authors/

TAILINGS DAMS AND RESERVOIRS CHANGE UPSTREAM FISH ASSEMBLAGES IN ATLANTIC FOREST STREAMS

ABSTRACT

Riverscape connectivity is a critically important component determining the ecological condition of lotic ecosystems. We evaluated the changes in fish assemblages caused by the loss of connectivity by mine tailings storage dams (TSDs), hypothesizing that headwater fish assemblages are restructured by TSDs located downstream, even though the upstream habitats are not altered. We used standard methods to collect fish in 24 first to third order sites, with half draining to TSDs (dammed) and the other half free from this impact (undammed). To identify differences between treatments, we used PERMANOVA to test both environmental variables and ichthyofauna composition (Bray-Curtis) and evaluate the biological metrics that most influenced assemblage composition change. As expected, we observed no difference between treatments for environmental variables, but we did observe differences in fish assemblage composition. We also observed five metrics with lower values in dammed streams (richness and abundance of intolerant species, siluriform richness, and abundance of *Pareiorhaphis proskynita* and *Trichomycterus brasiliensis*) and two metrics with higher values in dammed streams (perciform richness and abundance of *Oligosarcus argenteus*). We postulated that these changes can be an effect of stream fragmentation by TSDs plus source-sink mechanisms and conclude that mine TSDs located downstream change headwater fish assemblages, a factor often neglected in biomonitoring studies.

Keywords: Habitat fragmentation; Source-Sink dynamics; Iron ore mining; Rio Doce Basin

INTRODUCTION

Headwater streams are complex and dynamic ecosystems, representing over 90% of the channel length in hydrographic networks (Welcomme, 1985; Meyer et al., 2007; Colvin et al., 2019). They incorporate a diverse set of microhabitats and zones along elevation gradients (Vannote et al., 1980; Winemiller et al., 2008), which are reflected in their biological diversity patterns (Winemiller et al., 2008; Colvin et al., 2019). However, headwaters have become increasingly impaired (Hancock, 2002) because they are very susceptible to anthropogenic disturbances because of their small sizes (Meyer et al., 2007) and their dependency on riparian and catchment conditions (Hughes et al. 2006; 2019).

In many regions of the world, mining processes greatly alter stream fish faunas (Mechi and Sanches, 2010; Daniel et al., 2015; Hughes et al., 2016). Those impacts include increased concentrations of pollutants, metals, fine sediments and turbidity (Malm et al., 1997; Mechi and Sanches, 2010; Barros et al., 2014), and reduced flows (Hughes et al., 2016). These changes degrade stream habitat structure and water quality (Wallace et al., 1996; Snyder et al., 2003; Pirhalla, 2004), leading to altered fish assemblages (Brosse et al., 2011; Soares et al., 2017; Salvador et al., 2022).

The negative effects of mining can be perceived regionally (Freund and Petty, 2007; Lobo et al., 2018, Salvador et al. 2020). Nonetheless, most studies aiming to understand the environmental consequences of mining in watercourses focus on their downstream impacts (Barros et al., 2014). This may result from the common assumption that pollution of upstream reaches affects downstream reaches (Allan, 2004). However, aquatic communities are influenced by the whole river network and not only unidirectionally (Fausch et al., 2002).

Tailing storage dams (TSDs) also influence riverscapes in mining areas (Ward and Stanford, 1983; Cooper et al., 2017). These structures aim to store the non-commercial portion of processed ores, and usually they are built on small streams (Espósito and Duarte, 2010). Because they fragment river networks, they reduce aquatic biodiversity (Vörösmarty et al., 2010; Brasil et al., 2014; Leitão et al., 2018). Such fragmentation impairs the dispersal of organisms (Perkin and Gido, 2012) by isolating upstream populations from downstream ones, precluding some fish species from completing their life cycles (Colvin et al., 2019). Many neotropical fishes require connected reaches and spatiotemporal segregation to complete the spawning and rearing components of their life cycles (Abilhoa et al., 2011; Mazzoni and Iglesias-Rios, 2012; Mazzoni et al., 2018). It is common for many adult fishes of Atlantic Forest streams to pass the fast currents of rapids to reach river headwaters (Mazzoni and Iglesias-Rios, 2012; Mazzoni et al., 2018). However, after reproduction, eggs and larvae of many of those species drift downstream, reside there as juveniles, and migrate upstream again as adults (Abilhoa et al., 2011; Mazzoni and Iglesias-Rios, 2012).

Besides stream fragmentation, reservoirs may lead to local restructuring of the ichthyofauna when generalist taxa prevail over specialists (Rahel, 2002; Poff et al., 2007). Moreover, the word "tailings" is generic and can apply to a series of compounds resulting from mining processes (Kossoff et al., 2014). These compounds, often found in high concentrations in TSDs, can increase the deleterious effect of the reservoir by favoring water-quality tolerant species. Those species can influence assemblage structure in reaches upstream from the dams through mechanisms like source-sink dynamics, in which population performances are influenced by population movements from refuges into less suitable habitats and vice versa (Pulliam, 1988).

One way to assess the impact of anthropogenic changes on the environment is through use of multimetric indices (MMIs; Carvalho et al., 2017; Ruaro et al., 2019; Vadas et al., 2022). An MMI is composed of multiple metrics or variables. Those metrics may be indices themselves (e.g., a diversity index or a tolerance index) or simpler variables (e.g., species richness, species abundance, sensitive or tolerant species richness or abundance, or % sensitive or tolerant taxa, etc.). MMIs indices were first proposed by Karr (1981) to indicate biological condition and they have been used in the neotropical region for several purposes, such as assessing the effect of landscape changes and assessing the environmental quality of an area (Terra et al., 2013; Carvalho et al., 2017; Ferreira et al., 2018; Prudente et al., 2018). To generate an index, it is necessary to compare disturbed streams with reference streams and then, incorporate biological information through use of biological metrics that can be related to species richness, composition, abundance, diversity, and function (Karr et al., 1986; Fausch et al., 1990; Carvalho et al., 2017).

In the Atlantic Forest, pristine streams are dominated by Characiformes and Siluriformes (Casatti et al., 2001; Uieda and Uieda, 2001; Abilhoa et al., 2008; Terra et al., 2013), which is typical of other South American ichthyofauna (Reis et al., 2016). Some groups of Siluriformes are considered indicators of good stream environmental quality, for both Atlantic Forest and Brazilian Savanna (Cerrado) streams. For example, these include Trichomycteridae species richness and the proportion of invertivorous individuals, such as those belonging to the *Neoplecostomus* genus (Braga and Gomiero, 2009; Terra et al., 2013; Carvalho et al., 2017).

Conversely, Perciformes, the third most representative order of Neotropical fish species, are associated with environments of calmer waters and low energy, such as pools, lagoons and lower river reaches (Kullander, 2003). Species of this group also have been introduced around the world

(Alves et al., 2007; Mohanty et al., 2015; Hasan and Tamam, 2019). Such non-native species are scarce or absent in pristine Atlantic Forest streams (Casatti et al., 2001; Uieda and Uieda, 2001; Terra et al., 2013). On the other hand, perciforms and non-natives are commonly associated with hydroelectric reservoirs in the neotropical region (Alves et al., 2007; Garcia et al., 2017; Casimiro et al., 2018).

The effects of isolating headwater streams by artificial dams are poorly understood (Pringle, 2001) and downstream modifications can cause negative effects upstream, which are often difficult to associate with those modifications (Pringle, 1997). Therefore, our objectives were to 1) evaluate the effect of downstream dams on fish assemblage structure and 2) develop an MMI calculated from multiple fish assemblage structural variables, such as those mentioned above, for comparing neotropical fish assemblages in dammed and undammed headwater streams. We also evaluated the ability of the MMI to discriminate changes in assemblage structure. Our hypothesis was that headwater fish assemblages are restructured by downstream TSDs, even though the local physical and chemical habitats are not influenced. Specifically, we expected a decline of Siluriformes and dam intolerant species as well as a prevalence of dam-tolerant species, Perciformes and non-native species in dammed stream sites.

MATERIAL AND METHODS

Study area

The Rio Doce Basin drains approximately 83,000 km^2 (Vieira, 2009) and is located in the northeastern Atlantic Forest biome (Abell et al., 2008). The headwaters of some principal streams in the upper basin drain the Quadrilátero Ferrífero geologic formation (QF). The QF covers approximately 7,200 km^2 and has a very diverse landscape with peaks up to 2,000 m tall (Jacobi and Carmo, 2008). It is named for its abundance of iron ore (Roeser and Roeser, 2010), and is the location of many iron mines and TSDs. Despite the many mines, the QF is mostly forested (Lamounier et al., 2011). We selected 24 headwater stream sites distributed near large iron mines within the Rio Doce Basin portion of the Quadrilátero Ferrífero, 12 sites upstream of TSDs and 12 sites lacking downstream TSDs (Figure 4-1). Sites were sampled where they were most easily and safely accessed. Because of low map resolution and the proximities of some unmapped streams (Hughes & Omernik 1981; 1983), it appears that some streams were sampled at multiple sites. That was not the case. Each stream was sampled at only one site (Figure 4-1). The sampled streams share the same set of regional species, similar climate, native vegetation, catchment conditions, topography, geology and local habitat, but differ by the presence or absence of TSDs downstream.

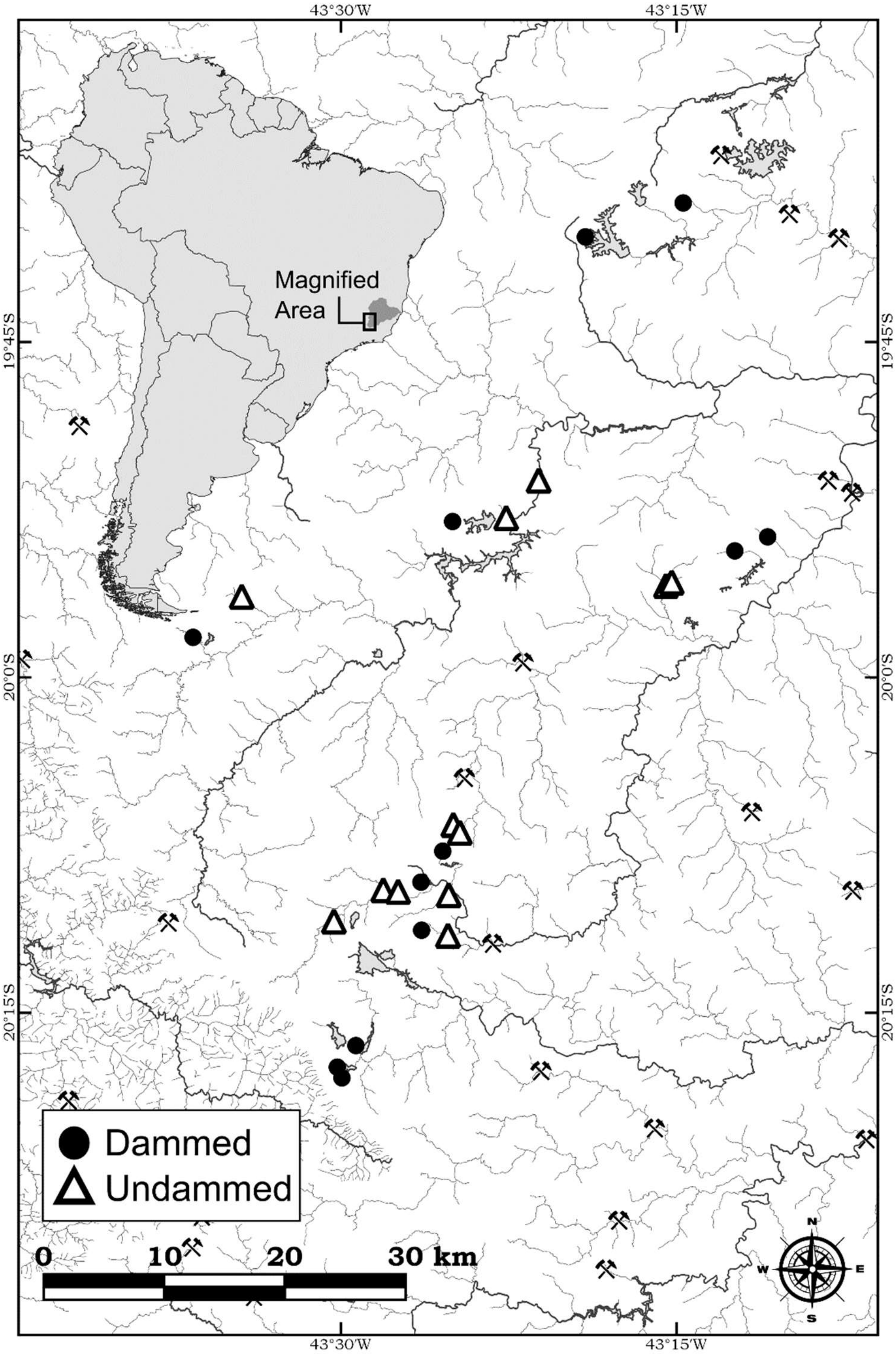


Figure 4-1. Spatial distribution of sites collected in streams dammed and undammed in Rio Doce Basin, one coastal drainage of "Northeastern Mata Atlantica" ecoregion.

Environmental variables

To evaluate the effect of the dam on the local habitat, we measured three groups of environmental variables at all 24 of the 50-m long stream sites. The first group included local physical habitat variables. We measured mean stream width (X_WIDTH) and mean depth (X_DEPTH) before fish collection at the beginning, middle and end of the sites. We also estimated the proportion of primary riparian forest (RIP-FOR) along the site and stream flow (STR-FLOW) by using ecologic-economic zonation data of the Minas Gerais State project (ZEE; Mello et al., 2008).

The second group of environmental variables comprised hydrographic variables: upstream basin area (BAS-AREA), downstream distance to a major watercourse (DIST-RIVER), topographic difference (SLOPE-RANGE) and stream slope (SLOPE). To do so, we used georeferenced information system (GIS) tools. First, we delimitated the catchment areas upstream of the sites using the hydrography present in ZEE (Mello et al., 2008). We did this manually because some basins are very small, and computational algorithms generate high levels of error. We also used the hydrography information provided by ZEE to calculate stream length, i.e., the distance between the collection site and stream spring, and the distance between the collection site and the TSD, or the first major watercourse downstream for undammed sites. We defined major watercourse as a river of fourth order or higher. Basin boundaries were used to obtain the highest and lowest values of altitude from the Shuttle Radar Topography Mission (SRTM) database; that information also was used to calculate the topographic difference and stream slope. The first was the difference between the highest and lowest values of altitude; the second variable was determined by dividing the topographic difference by stream length.

Land-use variables composed the last group. Such variables affect watercourses by changing stream metabolic rates, hydrologic parameters (Poff et al., 2006; Silva-Junior, 2016), and ichthyofauna (Molina et al., 2017; Hughes et al., 2006; 2019). To calculate the proportion of land-use, we used the entire catchment upstream of the sites, classifying land use into five categories: natural cover (%_NAT-COV), mining (%_MINE), road (%_ROAD), forestry (%_FORESTRY) and pasture (%_PAST). Again, because of the small size of some basins, we classified the images manually, using images available from Google Earth (Ayach et al., 2012).

Fish collection

We collected fish from all 24 of the 50-m long sites during the dry season in both 2011 and 2012 through the use of two distinct and complementary methods, one passive and the other active. For the passive method, we placed three traps in each site in the afternoon, baited with meat, and removed them the following morning for a fishing time of 16 hours. For the active method, we used a 5-mm mesh hand-net with a mouth size of 0.25 m^2. At each site, we dipped at 100 randomly chosen stations, thoroughly disturbing the bed and vegetation substrates. The fish were euthanized using a lethal dose of eugenol and fixed in 10% formalin and taken to the laboratory where they were identified to species. We deposited voucher specimens in the Natural History Museum of the Catholic

University of Minas Gerais (MCNIP), the Ichthyological Collection of the Taxonomic Collection Center of the Federal University of Minas Gerais (ICT-UFMGMHN), and the Zoology Museum of the University of Campinas (ZUEC).

Data analyses

To test if TSDs were the principal predictor for altered fish assemblages in the streams, we first determined whether environmental variables differed between dammed and undammed sites. To do so, we performed a series of metric screens to ensure the effectiveness of variables. First, we excluded variables with coefficients of variation (CV) <30%. We also examined variable Spearman correlations ($|rs| \geq 0.7$; $p \leq 0.05$) to identify multicollinearity, retaining those variables with higher predictive importance for tropical stream fish assemblages. With the retained variables, we performed a principal component analysis (PCA) using standardization data transformation, which aims to facilitate the comparison between variables. To test for difference between treatments, we used a permutational multivariate analysis of variance (PERMANOVA) (Legendre and Legendre, 2012).

To assess changes in assemblage structure between the two treatments, we tested species richness and abundance using a t-test after ensuring normality and homoscedasticity. We also performed a PERMANOVA based on a Bray-Curtis similarity matrix for log-transformed ($Log_{(x+1)}$) abundance of fish species, followed by a principal coordinates analysis (PCoA) to visualize differences between the fish assemblage composition of the two treatments. To assess how each species contributed to the difference between treatments, we conducted a similarity percentile analysis (SIMPER). These analyses were performed in R (R Core Team, 2017). We also calculated 78 candidate biological metrics based on richness and abundance (number and percentage; Supplementary 4-1).

To calculate the MMI, we first classified fish species as native or non-native following Vieira (2009) and by their ability to survive in reservoirs (dam-tolerant or dam-intolerant) based on data from other mine monitoring programs in the region (personal observation: G.N. Salvador). We eliminated metrics with CV < 30% to remove metrics with a low range (Klemm et al., 2003). We also eliminated metrics with 80% or more of 0 values and examined variable Spearman correlations ($|rs| \geq 0.7$; $p \leq 0.05$) to identify multicollinearity, retaining the variables with higher predictive importance for tropical stream fish assemblages (Whittier et al., 2007a). We tested the remaining metrics for their relationship with the environmental variables to eliminate metrics that are strongly affected by natural gradients (Whittier et al., 2007a). To assess fish metrics differing between treatments, we used the quartile overlap method of Barbour et al. (1996), comparing the two treatments through a t-test after ensuring normality and homoscedasticity of the data.

Finally, to calculate MMI scores, each metric was scored continuously from 0 (poor) to 1 (good) using the metric upper and lower limits (Hughes et al., 1998). We used the 5th and 95th percentiles of the raw values to exclude extreme values, which can interfere with metric interpretation. Metrics that decreased in dammed sites compared with undammed sites were scored with 1 based on the 95th percentile of observed raw values and 0 based on the 5th percentile of observed raw values. For metric values decreasing in dammed sites, raw observed values that fell at the 25th, 50th and 75th percentiles were scored as 0.25, 0.5, and 0.25, respectively. Metrics that increased in dammed sites compared with undammed sites were scored with 0 based on the 95th percentile of raw observed values and with 1 based on the 5th percentile of raw observed values. This meant that for those metric values that increased in dammed sites, raw observed values that fell at the 25th, 50th and 75th percentiles were scored as 0.75, 0.5, and 0.25, respectively. The scores of each of the 7 variables (each ranging from 0 to 1) were added to produce a final MMI score ranging from 0 (very poor) to 7 (good).

RESULTS

Environmental variables

From 16 environmental variables, we excluded one because of its low coefficient of variation and four because of multicollinearity, leaving eleven variables (Supplementary 1). The PCA generated using those variables explained 45.83% of data variation, with the first axis explaining 24.53%, and the second axis explaining 21.31%. Four variables had loads higher than 0.5 on the PCA: X_WIDTH and BAS-AREA with the first axis, %_NAT-COVER and SLOPE with the second axis. X_WIDTH varied from 6.0 m to 0.5 m with a mean of 2.1±1.5 m, and BAS-AREA varied from 36.4 ha to 0.2 ha with a mean of 5.5±7.7 ha. The % NAT-COVER varied from 100% to 14% with a mean of 82±27 %, and SLOPE varied from 421 m/km to 19 m/km with a mean of 169±113 m/km (Table 4-1). The PERMANOVA test did not show a significant difference between treatments for those variables (F = 1.505, p = 0.138) (Figure 4-2).

Table 4-1. Environmental conditions of dammed and undammed sites.

Environmental variable	DAMMED				UNDAMMED			
	Mean	**SD**	**Max.**	**Min.**	**Mean**	**SD**	**Max.**	**Min.**
%_NAT-COVER	85%	25%	100%	21%	79%	30%	100%	14%
%_MINE	7%	23%	79%	0%	9%	18%	62%	0%
%_ROAD	0%	1%	3%	0%	0%	0%	1%	0%
%_CATTLE	5%	12%	42%	0%	4%	12%	42%	0%
RIP-FOR	62%	32%	100%	0%	74%	35%	100%	0%
X_DEEP	0.63	0.35	1.30	0.30	0.93	0.60	2.50	0.30
X_WIDTH	1.88	1.42	4.60	0.50	2.32	1.60	6.00	0.70
BAS-AREA	2.41	2.34	7.97	0.16	7.81	10.13	36.40	0.52
DIST-RIVER	0.59	0.68	2.53	0.02	2.72	2.48	6.16	0.10
SLOPE	192	98	421	79	145	126	377	19
SLOPE-RANGE	434	245	868	133	500	303	919	163

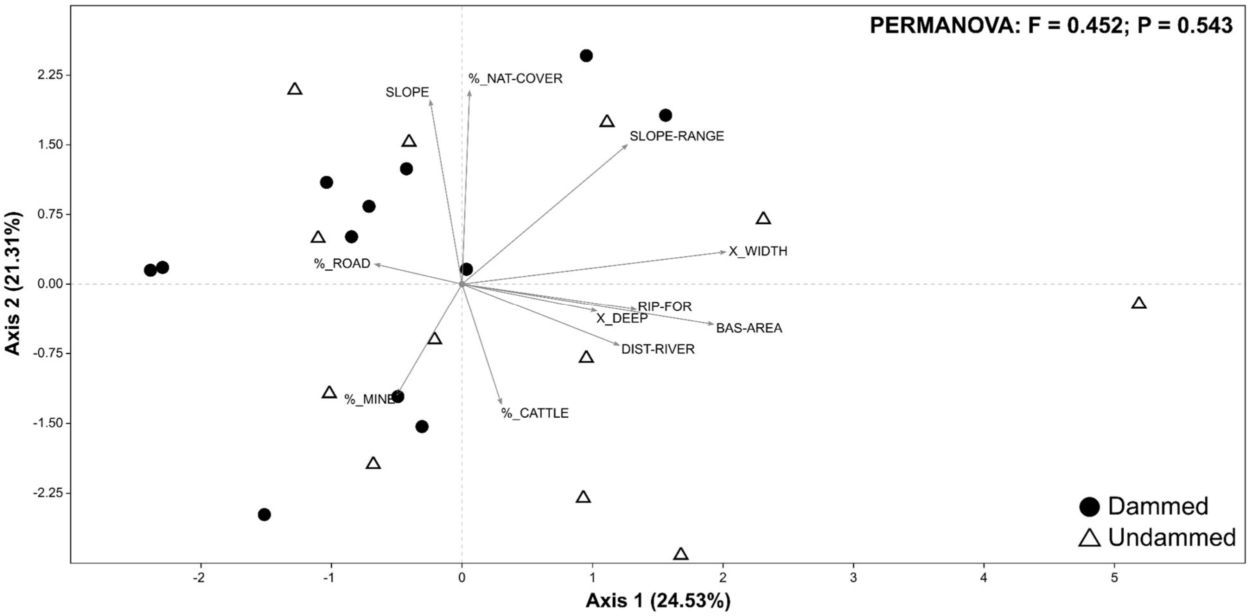


Figure 4-2. Bidimensional projection of environmental variables showing similarity between the treatments.

Fish assemblages

Assemblage composition

We collected a total of 1,431 fishes, and the average abundance was 60±14 fishes per site, with the range varying from 263 fishes to 1 fish per site. We found 21 species, distributed in four orders and seven families (Table 2). Eleven of those species were classified as dam-tolerant because of their occurrence in TSDs. The dam- tolerant species included two non-natives (*Oreochromis niloticus*, *Poecilia reticulata*) and nine natives (*Astyanax* gr. *lacustris*, *Astyanax* gr. *scabripinnis*, *Oligosarcus argenteus, Hoplias intermedius, Rhamdia quelen, Hypostomus affinis, Australoheros* *ipatinguensis*, *Geophagus brasiliensis, Phalloceros* aff. *uai)*. One non-native species, *Knodus* aff. *moenkhausii* was classified as dam-intolerant because it had not been recorded as occurring in tailings dams.

Table 4-2. Fish species collected from 24 dammed and undammed sites, voucher number from Natural History Museum of PUC-Minas.

TAXON	VOUCHER (MCNIP)	DAM-TOLERANT	NON-NATIVE
CHARACIFORMES			
CHARACIDEA			
Astyanax fasciatus (Cuvier 1819)	520		
Astyanax lacustris (Linnaeus 1758)	522	x	
Astyanax gr. *scabripinnis* (Jenyns 1842)	217	x	
Astyanax taeniatus (Jenyns 1842)	523		
Hyphessobrycon aff. *santae* (Eigenmann 1907)	273		
Knodus cf. *moenkhausii* (Eigenmann & Kennedy 1903)	220		x
Oligosarcus argenteus Günther 1864	236	x	
ERYTHRINIDAE			
Hoplias intermedius (Günther 1864)		x	
SILURIFORMES			
HEPTAPTERIDAE			
Rhamdia quelen (Quoy & Gaimard 1824)	224	x	
TRICHOMYCTERIDAE			
Trichomycterus aff. *alternatus* (Eigenmann 1917)	280		
Trichomycterus brasiliensis Lütken 1874	214		
Trichomycterus immaculatus (Eigen. & Eigen. 1889)	530		
LORICARIIDAE			
Hypostomus affinis (Steindachner 1877)	223	x	
Neoplecostomus doceensis Roxo et al. 2014	239		
Pareiorhaphis scutula Pereira, Vieira & Reis 2010	512		
Pareiorhaphis cf. *proskynita* Pereira & Britto 2012	216		
PERCIFORMES			
CICHLIDAE			
Australoheros ipatinguensis Ottoni & Costa 2008	227	x	
Geophagus brasiliensis (Quoy & Gaimard 1824)	219	x	
Oreochromis niloticus (Linnaeus 1758)	---	x	x
CYPRINODONTIFORMES			
POECILIIDAE			
Phalloceros cf. *uai* Lucinda 2008	221	x	
Poecilia reticulata Peters 1859	218	x	x

In undammed sites, average fish richness was 4.3±2.5, and abundance was 54±17 individuals, whereas in dammed sites, average fish richness was 2.7±2.0, and abundance was 65±22 individuals. The differences in species richness and abundance seemed to be related to chance because there was no statistical difference between treatments (Species Richness: U = 1.669, p = 0.095; Abundance: U = 0.318, p = 0.751). Fish assemblage structure differed between treatments (PERMANOVA; F = 2.101; p = 0.033), but the degree of segregation was weak ($R^2 = 0.09$), as summarized by PCoA (Figure 4-3). The undammed sites supported 17 species, six of which were exclusive to that treatment (*Astyanax fasciatus*, *A. lacustris, H. intermedius, Hyphessobrycon* aff. *santae, Neoplecostomus doceensis* and *P.* aff. *uai*). The dammed sites contained 15 species, four restricted to that treatment (*A. ipatinguensis, Hypostomus affinis, O. niloticus* and *P. reticulata*). The contribution of each species to the difference between treatments (SIMPER) indicated that approximately 50% of the variation was related to the abundance of five species. The two most frequent in undammed sites were *Pareiorhaphis* cf. *proskynita* and *Trichomycterus brasiliensis* and the three most frequent in dammed sites were *Oligosarcus argenteus*, *A.* gr. *scabripinnis* and *R. quelen* (Table 4-3).

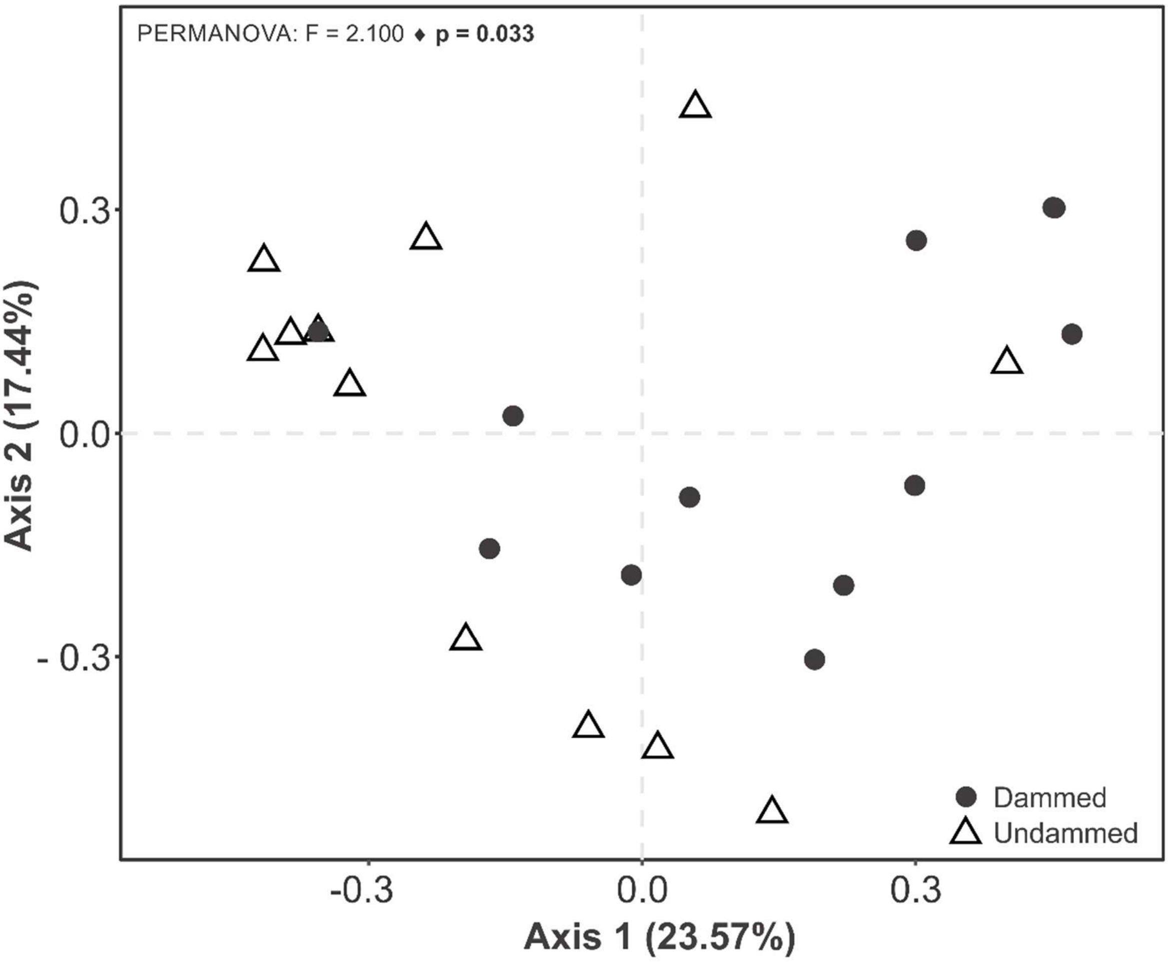


Figure 4-3. Bidimensional projection of Bray-Curtis fish assemblage similarities of dammed and undammed sites.

Table 4-3. Results of SIMPER analysis showing the percent contributions of the species that influenced the dissimilarity between fish assemblages of dammed and undammed sites.

Taxon	Mean dissim.	%	Cum. %	Mean	
				Dammed	Undammed
Oligosarcus argenteus	12.22	13.9	13.9	0.588	0.232
Pareiorhaphis cf. *proskynita*	11.94	13.6	27.6	0.158	0.574
Trichomycterus brasiliensis	8.02	9.2	36.7	0.050	0.401
Astyanax scabripinnis	7.14	8.1	44.9	0.247	0.319
Rhamdia quelen	6.05	6.9	51.8	0.155	0.284
Poecilia reticulata	4.98	5.7	57.5	0.314	---
Pareiorhaphis scutula	4.95	5.6	63.1	0.096	0.277
Astyanax taeniatus	4.88	5.6	68.7	0.151	0.238
Geophagus brasiliensis	4.51	5.1	73.8	0.308	0.090
Phalloceros cf. *uai*	4.13	4.7	78.5	---	0.354
Trichomycterus aff. *alternatus*	3.68	4.2	82.7	0.065	0.100
Trichomycterus immaculatus	3.59	4.1	86.8	0.025	0.198
Australoheros ipatinguensis	3.55	4.1	90.9	0.279	---
Hyphessobrycon aff. *santae*	2.25	2.6	93.4	---	0.172
Astyanax fasciatus	1.52	1.7	95.2	---	0.140
Knodus cf. *moenkhausii*	1.51	1.7	96.9	0.040	0.075
Hypostomus affinis	1.11	1.3	98.1	0.070	---
Astyanax lacustris	0.65	0.7	98.9	---	0.050
Oreochromis niloticus	0.40	0.5	99.3	0.040	---
Hoplias intermedius	0.33	0.4	99.7	---	0.025
Neoplecostomus doceensis	0.25	0.3	100.0	---	0.025

Multimetric Index determinations

From 78 candidate metrics, we excluded 35 metrics with a high frequency of zero values, 29 were redundant, and seven did not distinguish treatments, leaving seven metrics (defined below). Three of the metrics were richness based (R_DIN; R_SIL; R_PER), and the other four were abundance based (A_DIN; A_OL-AR; A_PA-PR; A_TR-BR). The remaining metrics were checked for correlation with environmental variables and four of those showed significant results. Stream width was correlated with both R_SIL ($p = 0.037$; $r_s = 0.429$) and A_PA-PR ($p = 0.007$; $r_s = 0.532$). Basin area was correlated with R_DIN ($p = 0.046$; $r_s = 0.188$) and slope was correlated with A_TR-BR ($p = 0.049$; $r_s = -0.407$). Although those variables were correlated with the fish metrics, the correlations were weak, explaining <30% of metric variability.

Of the metrics selected, five had higher values in undammed sites (Figure 4-4). We collected an average of 2.8 ± 0.4 dam intolerant species (R_DIN), and 2.4 ± 0.5 siluriform species (R_SIL) in undammed sites, versus 0.7 ± 0.3 intolerant species, and 0.7 ± 0.3 siluriform species in dammed sites.

The average abundance of dam intolerant individuals (A_DIN) was 29.1 ± 9.7 in undammed sites versus 10.5 ± 6.6 in dammed sites. The abundance of *P. proskynita* (A_PA-PR) was 6.3 ± 2.8 individuals in undammed sites versus 3.3 ± 3.2 in dammed sites. The abundance of *T. brasiliensis* (A_TR-BR) was 4.3 ± 2.2 individuals in undammed sites versus 0.3 ± 0.3 in dammed sites. Two metrics had higher values in dammed sites (Figure 4-5). In dammed sites, we collected an average of 0.7 ± 0.3 perciform species (R_PER), versus 0.2 ± 0.1 species in undammed sites. We also caught an average of 13.9 ± 7.7 individuals of *O. argenteus* (A_OL-AR) in dammed sites versus 3.5 ± 2.8 individuals in undammed sites.

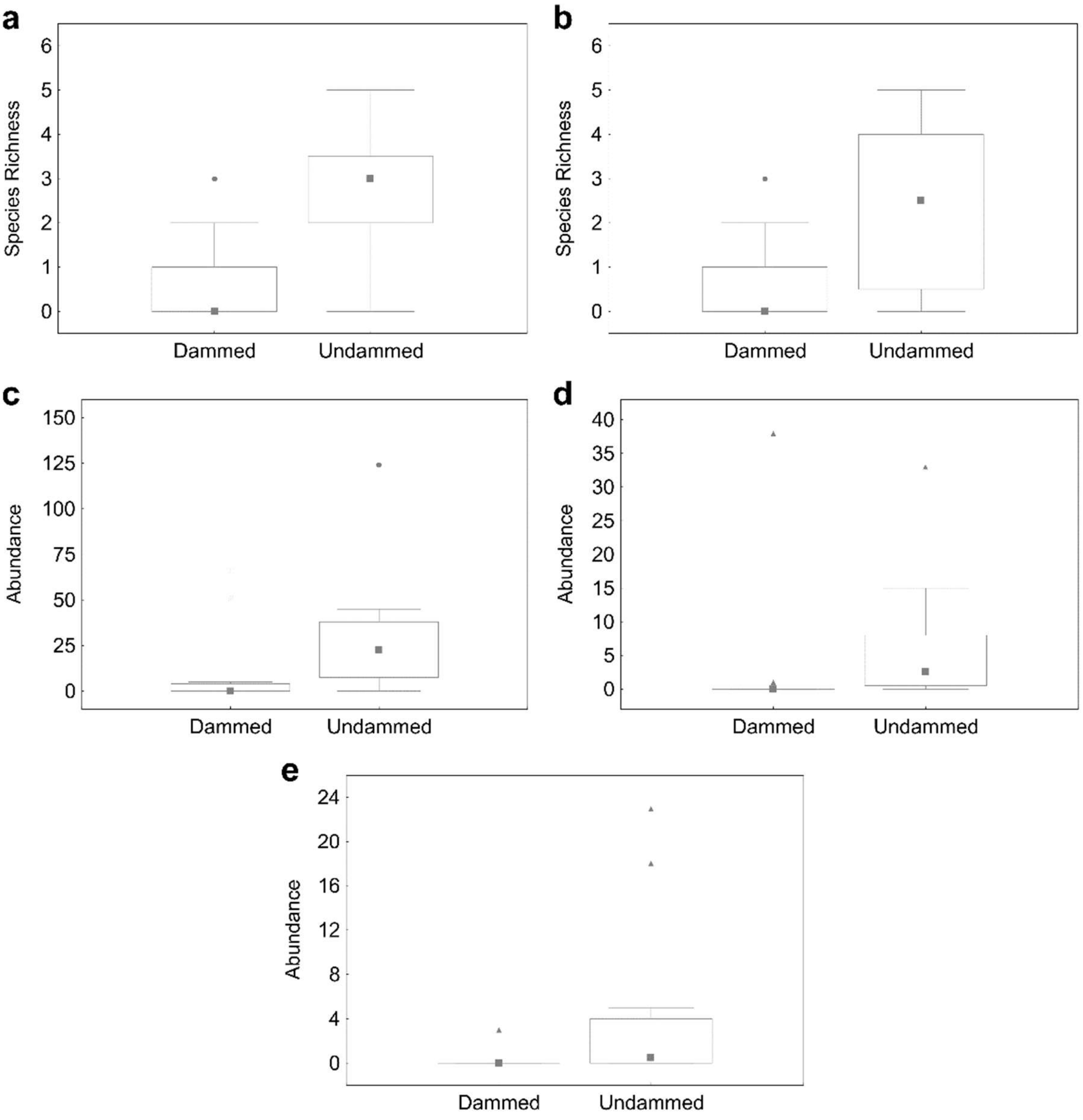


Figure 4-4. Fish assemblage metrics with values higher in undammed sites: dam intolerant species richness (a), siluriform species richness (b), dam intolerant abundance (c), *Pareiorhaphis proskynita* abundance (d), and *Trichomycterus brasiliensis* abundance (e). Central square represents median, box represent quartiles (25% and 75%), lines represent non-outlier range, circle represent outlier and star represent extremes values.

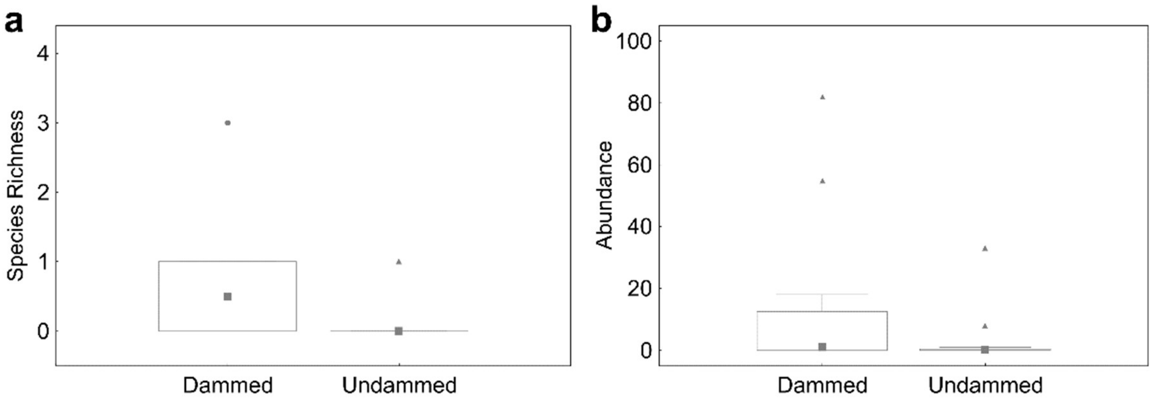


Figure 4-5. Fish assemblage metrics with values higher in dammed sites: perciform species richness (a), and Oligosarcus argenteus abundance (b). Central square represents median, box represent quartiles (25% and 75%), lines represent non-outlier range, circle represent outlier and star represent extremes values.

The mean MMI score for the 12 dammed sites (2.6 ± 0.5) was significantly lower than that in the 12 undammed sites (5.4 ± 0.6) (t = 3.669; DF = 21; p = 0.001). All high scores (>6) occurred in the undammed sites, whereas all low scores (< 3) were in dammed sites (Figure 4-6).

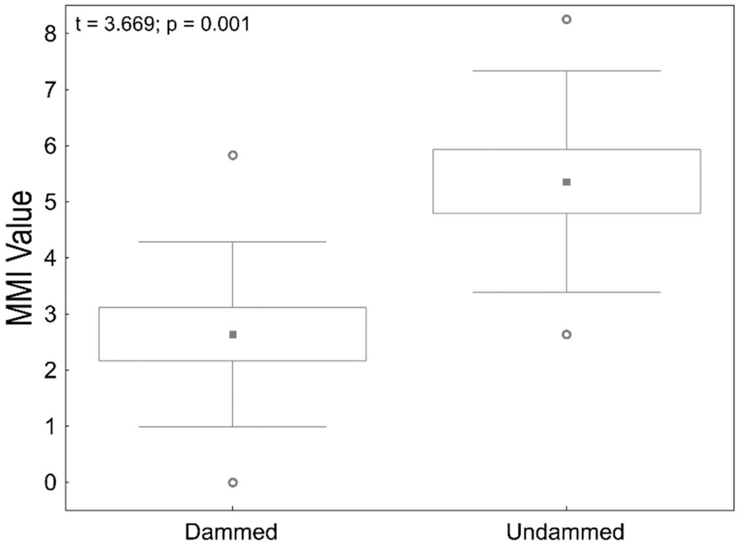


Figure 4-6. Multimetric index value generated for sites in dammed and undammed streams. Central square represents mean of the sites, box represents mean plus standard error, lines represent mean plus standard deviation, and circle represents outliers.

DISCUSSION

Mining is considered a cornerstone of human development (Ghose, 2009). However, it often impairs the structure and function of downstream aquatic ecosystems (Hughes et al., 2016) by degrading water quality (Smolders et al., 2003; Lin et al., 2005; Besser et al., 2007, 2009). Although mining has the capacity to degrade environmental characteristics, we did not detect any difference in environmental characteristics between upstream reaches of dammed and undammed sites. This similarity led us to believe that those variables are poor predictors for describing fish assemblage composition in our treatments and led us to question why the fish assemblages were dissimilar. We believe there are at least four explanations: life-cycle disruption, species intolerance or tolerance to lentic conditions, and non-native species.

The presence of a TSD was not enough to significantly change species richness and abundance values at the sites. Although average species richness was lower in dammed streams than undammed streams, the variability of this parameter was high in both treatments, leading to no statistical significance. Other studies have shown that species richness is often not a good parameter for assessing environmental disturbances (Soares et al., 2017; Ferreira et al., 2018; Vadas et al., 2022). One explanation for this is the substitution of some species groups, like dam intolerant species by Perciformes species. The abundance of individuals also did not differ significantly between treatments, again because of the high variablility of abundance. For this reason, total abundance is rarely used in MMIs (Terra et al., 2013; Carvalho et al., 2017; Ruaro et al., 2019).

Despite similarities in species richness and abundance of individuals, fish assemblage composition was not similar between treatments. This difference is mostly related to the species exclusively recorded in each treatment. Nearly half of the species were caught only in one or the other treatment. Some of those species exclusively recorded in undammed streams are rare in the Rio Doce basin, and they live in rarely occurring habitats. *Neoplecostomus doceensis* is found in clear running water (Roxo et al., 2014), and *H.* aff. s*antae*, is found in marshland (a similar habitat of the closely related *Hyphessobrycon uaiso* (Carvalho and Langeani, 2013). Other species are abundant along small and medium sized rivers, like *A. fasciatus*, *A. lacustris* and *H. intermedius*, but they are uncommon in small streams.

The species exclusively in dammed sites were linked to the presence of TSD reservoirs. Two of those species (*O. niloticus* and *P. reticulata*) are non-native to the Rio Doce basin. The occurrence of some non-native species is strongly correlated with degraded habitats (Casatti et al., 2006). However, the presence of those species in some pristine upstream environments is related to stream fragmentation by TSDs. A non-native species must overcome several challenges to establish a viable population in a new environment (Blackburn et al., 2011). The creation of a reservoir can facilitate this process to species adopted to live in lentic-like environments, once new opportunities for establishment are generated by the many environmental changes caused by the reservoir, which some native species cannot overcome (Agostinho et al., 1999, 2007). After its establishment in a TSD, the non-native species usually experiences high population growth (Agostinho and Júlio Jr., 1996; Sanches et al., 2016) favoring dispersal (Leibold et al., 2004) and consequent colonization of the upstream reaches of dammed streams.

The other two species exclusively recorded in dammed streams were the native species *H. affinis* and *A. ipatinguensis*. *H. affinis* is a relatively large-growing species, reaching up to 31-cm standard length (Albieri et al., 2014). It is usually found in Rio Doce basin rivers in lentic environments, including reservoirs (Freitas and Salvador, 2022). *A. ipatinguensis* also prefers lentic environments, although it reaches small sizes as adults (Barbosa et al., 1997; Salvador et al., 2018). For both species, our collections were unexpectedly made in high-slope and rapidly flowing sites, but not in undammed sites. These results indicate that TSDs function as refuges and sources of these species, which find TSDs ideal locations to establish their populations.

Although the exclusive species are part of the explanation of the difference in fish assemblage composition, they were not common species. Thus, another explanation is found in the MMI metrics. In undammed sites, mean siluriform richness was greater than in dammed sites. This pattern in minimally disturbed sites, is commonly observed for the fish fauna of the neotropical region (Lowe-McConnell, 1999; Reis et al., 2016), and has also been observed in other Rio Doce basin streams (Vieira et al., 2005, 2015). However, the life cycle of some Atlantic Forest fishes is an important factor to understand this pattern, with higher values of R_DIN, R_SIL and A_DIN observed in undammed sites. Many small fishes of the Atlantic Forest show an ontogenetic segregation among downstream and upstream reaches, with juveniles living in downstream reaches and adults living in upstream reaches (Menezes and Caramaschi, 2000; Mazzoni et al., 2004; Braga et al., 2007). The presence of rapids and cascades preclude juveniles with low swimming performance from upstream reaches (Abilhoa et al., 2011). Streams fragmented by TSDs block passage of adults of these species, as observed in larger rivers fragmented by hydropower dams (Pelicice et al., 2015).

Dam-intolerant species (R_DIN and A_DIN) and the abundance of *P.* aff. *proskynita* (A_PA-PR) and *T. brasiliensis* (A_TR-BR) also have been negatively affected in dammed streams. This can be related to the specialist-disturbance hypothesis, which states that specialist species are more affected by habitat disturbance than generalists (Vázquez and Simberloff, 2002). These benthic species can colonize high-altitude environments (>1,300 m) (Pereira and Britto, 2012; Vieira et al., 2015), and are associated with riffle areas, like many other siluriform species (Casatti, 2005). Upstream areas more frequently experience hydraulic disturbances (Peres-Neto et al., 1995), leading to those species being carried downstream during peak flow events, which frequently occur in the rainy season. In undammed streams, those species can recolonize upstream areas after recession of peak flows (Buckup et al., 2000; Carvajal-Quintero et al., 2015; Miranda-Chumacero et al., 2015). However, in dammed streams, the TSDs act as ecological traps, eliminating individuals that are confined in them. Possible reasons for their elimination include predation and inability to survive in lentic environments, like that observed for reservoirs above hydroelectric dams (Agostinho et al., 2012). In addition, recolonization by dam-intolerant species following disturbances is hindered by the presence of TSDs, which isolate upstream from downstream reaches.

The greater richness of perciforms (R_PER) in dammed sites is related to the ability of those species to survive in the TSDs and disperse from them. This group is not abundant in high energy streams (Casatti, 2005; Fagundes et al., 2015), like our sampled sites. However, the TSDs can maintain large populations of these species, creating a mass-effect in dammed streams. The ponds act as source of perciforms, donating individuals to lotic upstream reaches, even when conditions are unfavorable for the establishment of those species. Some perciforms can also alter conditions in the new environment (Blackburn et al., 2014) through increased competition, predation, and genetic degradation of native species, thereby reducing or extirpating wild stocks (Agostinho and Júlio Jr., 1996; Simberloff, 2003). In such situations, the Perciformes (all of which were cichlids in our study) benefited in the dammed streams because cichlids are highly resilient in lentic environments (Kullander, 2003). Therefore, the mean perciform richness was increased in dammed streams compared with undammed streams.

Although individual metrics did not always show significant differences between dammed and undammed sites (Fig. 4 and 5), when combined into an MMI the average scores for the dammed sites were significantly lower than those of the undammed sites (Fig. 6). This indicates the usefulness of the MMI approach for assessing biological condition (Karr, 1981; Ruaro and Gubiani, 2013; Terra et al., 2013; Ruaro et al. 2019; Vadas et al., 2022). Sites with MMI scores <3 were almost always observed in dammed streams, and dammed sites with minimally disturbed environmental conditions still had MMI scores lower than those in undammed sites. This reflects the negative upstream effects of TSDs on fish assemblages. These results can be extrapolated to other mountainous Atlantic Forest streams, because the general patterns of assemblage structure are similar. However, additional studies on the effects of other types of landscape fragmenting agents on neotropical fish assemblages in small streams and other biomes are needed. For example, poorly constructed road crossings were found to alter stream habitats and fish assemblages in small Amazonian streams (Leal et al., 2016; Leitão et al., 2018). Plus, it is important to investigate how small headwater dams affect the functional ecology of streams and their fish assemblage traits (Leitão et al. 2018).

Our results also have important implications for biomonitoring. In most biomonitoring approaches, the measure of how much a site deviates from integrity is usually made by comparing sites to a reference condition (Hughes et al., 1986; Hawkins et al., 2010), i.e., sites minimally or least-disturbed in a given region (Stoddard et al., 2006). The methods to select those sites often consider the ecological conditions of the catchment area upstream of the sites (Collier et al., 2007; Whittier et al., 2007b; Ligeiro et al., 2013), downstream conditions being mostly neglected. However, we showed that downstream conditions can be decisive, and the longitudinal riverscape profile should be investigated when selecting reference sites.

We conclude that TSDs and other dams work as strong filters for species inhabiting upstream reaches, which lead to changes in the composition of the ichthyofauna. They do so by negatively affecting the richness and abundance of species whose environmental requirements are negatively affected by their impacts and positively affecting those that benefit from their impacts. Therefore, despite the apparent similarity of local habitat characteristics, upstream reaches of dammed and undammed streams do not contain the same fish assemblages, which has important implications for biomonitoring, aquatic management and conservation planning.

REFERENCES

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S.C., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J. V., Heibel, T.J., Wikramanayake, E., Olson, D., López, H.L., Reis, R.E., Lundberg, J.G., Sabaj Pérez, M.H., Petry, P., 2008. Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. Bioscience 58, 403–414. https://doi.org/10.1641/B580507

Abilhoa, V., Duboc, L.F., Azevedo-Filho, D.P., 2008. A comunidade peixes de um riacho de Floresta Araucária, Alto Rio Iguaçu, Sul do Brasil. Rev. Bras. Zool. 25, 238–246. https://doi.org/10.1590/S0101-81752008000200012

Abilhoa, V., Braga, R.R., Bornatowski, H., Vitule, J.R.S., 2011. Fishes of the Atlantic Rain Forest streams: Ecological patterns and conservation. In: Grillo, O., Venora, G., eds. Changing diversity in changing environment. InTech Open, London, UK, pp. 259–282. https://doi.org/10.5772/24540

Agostinho, A.A., Agostinho, C.S., Pelicice, F.M., Marques, E.E., 2012. Fish ladders: Safe fish passage or hotspot for predation? Neotrop. Ichthyol. 10, 687–696. https://doi.org/10.1590/S1679-62252012000400001

Agostinho, A.A., Gomes, L.C., Pelicice, F.M., 2007. Ecologia e manejo de recursos pesqueiros em reservatórios do Brasil. EDUEM, Maringá, Brazil, pp. 501.

Agostinho, A.A., Júlio Jr., H.F., 1996. Peixes de outras águas: ameaça ecológica. Ciência Hoje 21, 36–44.

Agostinho, A.A., Miranda, L.E., Bini, L.M., Gomes, L.C., Thomaz, S.M., Suzuki, H.I., 1999. Patterns of colonization in neotropical reservoirs, and prognoses on aging. In: Tundisi, J.G., Straskraba, M., eds. Theoretical reservoir ecology and its aplication. International Institute of Ecology, São Carlos, Brazil, pp. 227–264.

Allan, J.D., 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. Annu. Rev. Ecol. Evol. Syst. 35: 257–284. https://doi.org/10.1146/annurev.ecolsys.35.120202.110122

Albieri, R.J., Costa, M.R., Santos, A.B.I., Albieri, R.C., Araújo, F.G., 2014. Weight-length relationships of 22 fish species from Paraíba do Sul River in Rio de Janeiro State, Southeastern Brazil. J. Appl. Ichthyol. 30, 431–433. https://doi.org/10.1111/jai.12374

Alves, C.B.M., Vieira, F., Magalhães, L.B., Brito, M.F.G., 2007. Impacts of non-native fish species in Minas Gerais, Brazil: Present situation and prospects. In: Bert, T.M., ed. Ecological and genetic implications of aquaculture activities. Springer Science & Business Media, Berlim, Germany, pp. 291–314. https://doi.org/10.1007/978-1-4020-6148-6

Ayach, R.L., Cunha, E.R., da Silva, L.F., Bacani, V.M., 2012. Utilização de imagens Google Earth para mapeamento do uso e cobertura da terra da bacia hidrográfica do córrego Indaiá, MS. Rev. Geonorte 2, 1801–1811.

Barbosa, F.A.R., Souza, E.M.M., Vieira, F., de Paula, C.P.G., Rocha, L.A., Maia-Barbosa, P.M., Oberdá, S.M., Mingoti, S.A., Renault, G.P.C.P., Rocha, L.A., Maia-Barbosa, P.M., Oberdá, S.M., Mingoti, S.A., 1997. Impactos antrópicos e biodiversidade aquática. In: Paula, J.A., Barbieri, A.F., Guerra, C.B., Landau, H.C., Vieira, F., Barbosa, F. A. R., Costa, H.S.M., Guerra, L.P., Monte-Mor, R.L.M., Simões, R.F., Braga, T.M., eds. Biodiversidade, População e Economia: Uma Região de Mata Atlântica. Cedeplar, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil, pp. 345–454.

Barbour, M.T., Gerritsen, J., Griffith, G.E., Frydenborg, R., McCarron, E., White, J.S., Bastian, M.L., 1996. A Framework for biological criteria for Florida streams using benthic macroinvertebrates. J. North Am. Benthol. Soc. 15, 185–211. https://doi.org/10.2307/1467948

Barros, D.A., Silva, B.M., Silva, R.A., Pereira, J.A.A., Borges, L.A.C., Pereira, E.L., Nascimento, G.D.O., 2014. Atributos físicos e pH de um fragmento do rio Taquari sob a mineração de bauxita no planalto de Poços de Caldas - Mg. Holos 4, 78. https://doi.org/10.15628/holos.2014.1378

Besser, J.M., Brumbaugh, W.G., Allert, A.L., Poulton, B.C., Schmitt, C.J., Ingersoll, C.G., 2009. Ecological impacts of lead mining on Ozark streams: Toxicity of sediment and pore water. Ecotoxicol. Environ. Saf. 72, 516–526. https://doi.org/10.1016/j.ecoenv.2008.05.013

Besser, J.M., Brumbaugh, W.G., May, T.W., Schmitt, C.J., 2007. Biomonitoring of Lead, Zinc, and Cadmium in streams draining Lead-mining and non-mining areas, Southeast Missouri, USA. Environ. Monit. Assess. 129, 227–241. https://doi.org/10.1007/s10661-006-9356-9

Blackburn, T.M., Essl, F., Evans, T., Hulme, P.E., Jeschke, J.M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D.M., Sendek, A., Vilà, M., Wilson, J.R.U., Winter, M., Genovesi, P., Bacher, S., 2014. A unified classification of alien species based on the magnitude of their environmental impacts. PLoS Biol. 12, e1001850. https://doi.org/10.1371/journal.pbio.1001850

Blackburn, T.M., Pyšek, P., Bacher, S., Carlton, J.T., Duncan, R.P., Jarošík, V., Wilson, J.R.U., Richardson, D.M., 2011. A proposed unified framework for biological invasions. Trends Ecol. Evol. 26, 333–339. https://doi.org/10.1016/j.tree.2011.03.023

Braga, F.M.S., Gomiero, L.M., 2009. Alimentação de peixes na microbacia do Ribeirão Grande, Serra da Mantiqueira oriental, SP. Biota Neotrop. 9, 207–212. https://doi.org/10.1590/S1676-06032009000300021

Braga, M.R., Vitule, J.R.S., Aranha, J.M.R., 2007. Estrutura populacional de *Mimagoniates microlepis* (Steindachner, 1876) (Characidae, Glandulocaudinae) em um riacho de Floresta Atlântica, Paraná (Brasil). Acta Biol. Parana. 36, 67–81. https://doi.org/10.5380/abpr.v36i0.9804

Brasil, L.S., Giehl, N.F.S., Almeida, S.M., Valadão, M.B.X., dos Santos, J.O., Pinto, N.S., Batista, J.D., 2014. Does the damming of streams in the southern Amazon basin affect dragonfly and damselfly assemblages (Odonata: Insecta)? A preliminary study. Int. J. Odonatol. 17, 187–197. https://doi.org/10.1080/13887890.2014.963712

Brosse, S., Grenouillet, G., Gevrey, M., Khazraie, K., Tudesque, L., 2011. Small-scale gold mining erodes fish assemblage structure in small neotropical streams. Biodivers. Conserv. 20, 1013–1026. https://doi.org/10.1007/s10531-011-0011-6

Buckup, P.A., Zamprogno, C., Vieira, F., Teixeira, R.L., 2000. Waterfall climbing in *Characidium* (Crenuchidae: Characidiinae) from eastern Brazil. Ichthyol. Explor. Freshw. 11: 273–278.

Carvalho, D.R., Leal, C.G., Junqueira, N.T., de Castro, M.A., Fagundes, D.C., Alves, C.B.M., Hughes, R.M., Pompeu, P.S., 2017. A fish-based multimetric index for Brazilian savanna streams. Ecol. Indic. 77, 386–396. https://doi.org/10.1016/j.ecolind.2017.02.032

Carvalho, F.R., Langeani, F., 2013. *Hyphessobrycon uaiso*: new characid fish from the Rio Grande, upper Rio Paraná basin, Minas Gerais State (Ostariophysi: Characidae), with a brief comment about some types of *Hyphessobrycon*. Neotrop. Ichthyol. 11, 525–536. https://doi.org/10.1590/S1679-62252013000300006

Carvajal-Quintero, J.D., Maldonado-Ocampo, J.A., Urbano-Bonilla, A., 2015. Climbing behavior of *Cordylancistrus* sp. in the Colombian Andes. Univ. Sci. 20: 209–215. https://doi.org/10.11144/Javeriana.SC20-2.cbcc

Casatti, L., 2005. Fish assemblage structure in a first order stream, southeastern Brazil: longitudinal distribution, seasonality, and microhabitat diversity. Biota Neotrop. 5, 75–83. https://doi.org/10.1590/S1676-06032005000100009

Casatti, L., Langeani, F., Castro, R.M.C., 2001. Peixes de riacho do parque estadual Morro do Diabo, Bacia do Alto Rio Paraná, SP. Biota Neotrop. 1, 1–15. https://doi.org/10.1590/s1676-06032001000100005

Casatti, L., Langeani, F., Ferreira, C.P., 2006. Effects of physical habitat degradation on the stream fish assemblage structure in a pasture region. Environ. Manage. 38, 974–982. https://doi.org/10.1007/s00267-005-0212-4

Casimiro, A.C.R., Garcia, D.A.Z., Vidotto-Magnoni, A.P., Britton, J.R., Agostinho, A.A., de Almeida, F.S., Orsi, M.L., 2018. Escapes of non-native fish from flooded aquaculture facilities: the case of Paranapanema River, southern Brazil. Zoologia 35, 1–6. https://doi.org/10.3897/zoologia.35.e14638

Collier, K.J., Haigh, A., Kelly, J., 2007. Coupling GIS and mulitvariate approaches to reference site selection for wadeable stream monitoring. Environ. Monit. Assess. 127, 29–45. https://doi.org/10.1007/s10661-006-9256-z

Colvin, S.A.R., Sullivan, S.M.P., Shirey, P.D., Colvin, R.W., Winemiller, K.O., Hughes, R.M., Fausch, K.D., Infante, D.M., Olden, J.D., Bestgen, K.R., Danehy, R.J., Eby, L., 2019. Headwater streams and wetlands are critical for sustaining fish, fisheries, and ecosystem services. Fisheries 44, 73–91. https://doi.org/10.1002/fsh.10229

Cooper, A.R., Infante, D.M., Daniel, W.M., Wehrly, K.E., Wang, L., Brenden, T.O., 2017. Assessment of dam effects on streams and fish assemblages of the conterminous USA. Sci. Total Environ. 586, 879–889. https://doi.org/10.1016/j.scitotenv.2017.02.067

Daniel, W.M., Infante, D.M., Hughes, R.M., Tsang, Y.-P., Esselman, P.C., Wieferich, D., Herreman, K., Cooper, A.R., Wang, L., Taylor, W.W., 2015. Characterizing coal and mineral mines as a regional source of stress to stream fish assemblages. Ecol. Indic. 50, 50–61. https://doi.org/10.1016/j.ecolind.2014.10.018

Espósito, T.J., Duarte, A.P., 2010. Classificação de barragens de contenção de rejeitos de mineração e de resíduos industriais em relação a fatores de risco. REM - Rev. Esc. Minas 63, 393–398. https://doi.org/10.1590/S0370-44672010000200026

Fagundes, D.C., Leal, C.G., Carvalho, D.R., Junqueira, N.T., Langeani, F., Pompeu, P.S., 2015. The stream fish fauna from three regions of the Upper Paraná River basin. Biota Neotrop. 15, e20140187. https://doi.org/10.1590/1676-06032015018714

Fausch, K.D., Lyons, J., Karr, J.R., Angermeier, P.L., 1990. Fish communities as indicators of environmental degradation. Am. Fish. Soc. Symp. 8, 123–144.

Fausch, K.D., Torgersen, C., Baxter, C.V., Li, H., 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. Bioscience 52, 1–16.

Ferreira, M.C., Begot, T.O., Prudente, B.S., Juen, L., Montag, L.F.A., 2018. Effects of oil palm plantations on the habitat structure and biota of streams in Eastern Amazon. Environ. Biol. Fishes 101, 547–562. https://doi.org/10.1007/s10641-018-0716-4

Freitas, T.M.S., Salvador, G.N., 2022. Biological aspects of *Hypostomus affinis* (Siluriformes: Loricariidae) in Brazilian coastal rivers. Acta Limnol. Bras. 34, e3. https://doi.org/10.1590/s2179-975x1121

Freund, J.G., Petty, J.T., 2007. Response of fish and macroinvertebrate bioassessment indices to water chemistry in a mined Appalachian watershed. Environ. Manage. 39, 707–720. https://doi.org/10.1007/s00267-005-0116-3

Garcia, D.A.Z., Casimiro, A.C.R., Costa, A.D.A., Yabu, M.H.S., Leme, G.L.A., Orsi, M.L., 2017. New occurrence of a non-native ornamental cichlid in the Paranapanema River, southern Brazil. Neotrop. Biol. Conserv. 12, 159–163. https://doi.org/10.4013/nbc.2017.122.10

Ghose, A.K., 2009. Technology vision 2050 for sustainable mining. Procedia Earth Planet. Sci. 1: 2–6. https://doi.org/10.1016/j.proeps.2009.09.003

Hancock, P.J., 2002. Human impacts on the stream-groundwater exchange zone. Environ. Manage. 29, 763–781. https://doi.org/10.1007/s00267-001-0064-5

Hasan, V., Tamam, M.B., 2019. First record of the invasive Nile Tilapia, *Oreochromis niloticus* (Linnaeus, 1758) (Perciformes, Cichlidae), on Bawean Island, Indonesia. Check List 15, 225–227. https://doi.org/10.15560/15.1.225

Hawkins, C.P., Olson, J.R., Hill, R.A., 2010. The reference condition: predicting benchmarks for ecological and water-quality assessments. J. North Am. Benthol. Soc. 29, 312–343. https://doi.org/10.1899/09-092.1

Hughes, R.M., Omernik., J.M., 1981. Use and misuse of the terms, watershed and stream order. In: Krumholz, L.A., ed. The warmwater streams symposium. Southern Division American Fisheries Society, Bethesda, MD, pp. 320-326.

Hughes, R.M., Omernik., J.M., 1983. An alternative for characterizing stream size. In: Fontaine III, T.D., Bartell, S.M., eds. Dynamics of lotic ecosystems. Ann Arbor Press, Ann Arbor, MI, pp. 87-102.

Hughes, R.M., Amezcua, F., Chambers, D.M., Daniel, W.M., Franks, J.S., Franzin, W., MacDonald, D., Merriam, E., Neall, G., Pompeu, P.S., Reynolds, L., Woody, C.A., 2016. AFS position paper and policy on mining and fossil fuel extraction. Fisheries 41, 12–15. https://doi.org/10.1080/03632415.2016.1121742

Hughes, R.M., Infante, D.M., Wang, L., Chen, K., Terra B.F. 2019. Advances in understanding landscape influences on freshwater habitats and biological assemblages. American Fisheries Society, Symposium 90, Bethesda, Maryland, USA, pp. 523. https://doi.org/10.47886/9781934874561

Hughes, R.M., Kaufmann, P.R., Herlihy, A.T., Kincaid, T.M., Reynolds, L., Larsen, D.P., 1998. A process for developing and evaluating indices of fish assemblage integrity. Can. J. Fish. Aquat. Sci. 55, 1618–1631. https://doi.org/10.1139/f98-060

Hughes, R.M., Larsen, D.P., Omernik, J.M., 1986. Regional reference sites: a method for assessing stream potentials. Environ. Manage. 10, 629–635. https://doi.org/10.1007/BF01866767

Hughes, R.M., Wang, L., Seelbach, P.W. 2006. Landscape influences on stream habitat and biological assemblages. American Fisheries Society, Symposium 48, Bethesda, Maryland, USA, pp. 698. https://doi.org/10.47886/9781888569766

Jacobi, C.M., Carmo, F.F., 2008. Diversidade dos campos rupestres ferruginosos no Quadrilátero Ferrífero , MG. Megadiversidade 4: 25–33.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6, 21–27. https://doi.org/10.1577/1548-8446(1981)006<0021:AOBIUF>2.0.CO;2

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J., 1986. Assessing biological integrity in running waters: A method and its rationale. Illinois Natural History Survey Special Publication 5, Illinois, Michigan, USA, pp. 1-28.

Klemm, D.J., Blocksom, K.A., Fulk, F.A., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Peck, D. V., Stoddard, J.L., Thoeny, W.T., Griffith, M.B., Davis, W.S., 2003. Development and evaluation of a Macroinvertebrate Biotic Integrity Index (MBII) for regionally assessing Mid-Atlantic Highlands streams. Environ. Manage. 31, 656–669. https://doi.org/10.1007/s00267-002-2945-7

Kossoff, D., Dubbin, W.E., Alfredsson, M., Edwards, S.J., Macklin, M.G., Hudson-Edwards, K.A., 2014. Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. Appl. Geochemistry 51, 229–245. https://doi.org/10.1016/j.apgeochem.2014.09.010

Kullander, S.O., 2003. Family Ciclhidae (Cichlids). In: Reis, R.E., Kullander, S.O., Ferraris-Jr., C.J., eds. Check List of the Freshwater Fishes of South and Central America. EDIPUCRS, Porto Alegre, Brazil, pp. 605–654.

Lamounier, W.L., Carvalho, V.L.M., Salgado, A.A.R., 2011. Serra do Gandarela: possibilidade de ampliação das unidades de conservação no Quadrilátero Ferrífero - MG. Rev. do Dep. Geogr. - USP 22, 171–192. https://doi.org/10.7154/RDG.2011.0022.0009

Leal, C.G., Pompeu, P.S., Gardner, T.A., Leitão, R.P., Hughes, R.M., Kaufmann, P.R., Zuanon, J., de Paula, F.R., Ferraz, S.F.B., Thomson, J.R., Mac Nally, R., Ferreira, J., Barlow, J., 2016. Multi-scale assessment of human-induced changes to Amazonian instream habitats. Landsc. Ecol. 31, 1725–1745. https://doi.org/10.1007/s10980-016-0358-x

Legendre, P., Legendre, L., 2012. Numerical ecology, 3rd ed. Elsevier B.V., Oxford, UK, pp. 1006.

Leibold, M.A., Holyoak, M., Mouquet, N., Amarasekare, P., Chase, J.M., Hoopes, M.F., Holt, R.D., Shurin, J.B., Law, R., Tilman, D., Loreau, M., Gonzalez, A., 2004. The metacommunity concept: A framework for multi-scale community ecology. Ecol. Lett. 7, 601–613. https://doi.org/10.1111/j.1461-0248.2004.00608.x

Leitão, R.P., Zuanon, J., Mouillot, D., Leal, C.G., Hughes, R.M., Kaufmann, P.R., Villéger, S., Pompeu, P.S., Kasper, D., de Paula, F.R., Ferraz, S.F.B., Gardner, T.A., 2018. Disentangling the pathways of land use impacts on the functional structure of fish assemblages in Amazon streams. Ecography 41, 219–232. https://doi.org/10.1111/ecog.02845

Ligeiro, R., Hughes, R.M., Kaufmann, P.R., Macedo, D.R., Firmiano, K.R., Ferreira, W.R., Oliveira, D., Melo, A.S., Callisto, M., 2013. Defining quantitative stream disturbance gradients and the additive role of habitat variation to explain macroinvertebrate taxa richness. Ecol. Indic. 25, 45–57. https://doi.org/10.1016/j.ecolind.2012.09.004

Lin, C., Tong, X., Lu, W., Yan, L., Wu, Y., Nie, C., Chu, C., Long, J., 2005. Environmental impacts of surface mining on mined lands, affected streams and agricultural lands in the Dabaoshan Mine region, southern China. L. Degrad. Dev. 16, 463–474. https://doi.org/10.1002/ldr.675

Lobo, F.L., Novo, E.M.L.M., Barbosa, C.C.F., Vasconcelos, V.H.F., 2018. Monitoring water siltation caused by small-scale gold mining in Amazonian rivers using multi-satellite images. In: Gökçe, D., eds. Limnology. InTech Open, London, UK, pp. 2-17. https://doi.org/10.5772/intechopen.79725

Lowe-McConnell, R.H., 1999. Estudos ecológicos em comunidades de peixes tropicais. EDUSP, São Paulo, Brazil, pp. 536.

Malm, O., Guimarães, J.R.D., Castro, M.D., Bastos, W.R., Viana, J.P., Branches, F.J.P., Silveira, E.G., Pfeiffer, W.C., 1997. Follow-up of mercury in fish, human hair and urine in the Madeira and Tapajós basins, Amazon, Brazil. Water, Air Soil Pollut. 97, 45–51. https://doi.org/10.1007/BF02409643

Mazzoni, R., Iglesias-Rios, R., 2012. Movement patterns of stream-dwelling fishes from Mata Atlântica, Southeast Brazil. Rev. Biol. Trop. 60, 1837–1846. https://doi.org/10.15517/rbt.v60i4.2184

Mazzoni, R., Pinto, M.P., Iglesias-Rios, R., Costa, R., 2018. Fish movement in an Atlantic Forest stream. Neotrop. Ichthyol. 16, 1–12. https://doi.org/10.1590/1982-0224-20170065

Mazzoni, R., Schubart, S.A., Iglesias-Rios, R., 2004. Longitudinal segregation of *Astyanax janeiroensis* in Rio Ubatiba: A Neotropical stream of south-east Brazil. Ecol. Freshw. Fish 13, 231–234. https://doi.org/10.1111/j.1600-0633.2004.00062.x

Mechi, A., Sanches, D.L., 2010. Impactos ambientais da mineração no estado de São Paulo. Estud. Avançados 24, 209–220.

Mello, C.R., Silva, A.M., Coelho, G., Marques, J.J.G., Campos, C.M.M., 2008. Recursos Hídricos, In: Scolforo, J.R.S., de Oliveira, A.D., de Carvalho, L.M.T., eds. Zoneamento Ecologico-Economico de Minas Gerais: Componentes Geofísico e Biótico. Editora UFLA, Lavras, Brazil, pp. 103–136.

Menezes, M.S., Caramaschi, E.P., 2000. Longitudinal distribution of *Hypostomus punctatus* (Osteichthyes, Loricariidae) in a coastal stream from Rio de Janeiro, Southeastern Brazil. Brazilian Arch. Biol. Technol. 43, 229–233. https://doi.org/10.1590/S1516-89132000000200013

Meyer, J.L., Strayer, D.L., Wallace, J.B., Eggert, S.L., Helfman, G.S., Leonard, N.E., 2007. The contribution of headwater streams to biodiversity in river networks. J. Am. Water Resour. Assoc. 43, 86–103. https://doi.org/10.1111/j.1752-1688.2007.00008.x

Miranda-Chumacero, G., Álvarez, G., Luna, V., Wallace, R.B., Painter, L., 2015. First observations on annual massive upstream migration of juvenile catfish *Trichomycterus* in an Amazonian River. Env. Biol. Fish. 98, 1913–1926. https://doi.org/10.1007/s10641-015-0407-3

Mohanty, S.K., Mishra, S.S., Khan, M., Mohanty, R.K., Mohapatra, A., Pattnaik, A.K., 2015. Ichthyofaunal diversity of Chilika Lake, Odisha, India: an inventory, assessment of biodiversity status and comprehensive systematic checklist (1916-2014). Check List 11, 1817. https://doi.org/10.15560/11.6.1817

Molina, M.C., Roa-Fuentes, C.A., Zeni, J.O., Casatti, L., 2017. The effects of land use at different spatial scales on instream features in agricultural streams. Limnologica 65, 14–21. https://doi.org/10.1016/j.limno.2017.06.001

Pelicice, F.M., Pompeu, P.S., Agostinho, A.A., 2015. Large reservoirs as ecological barriers to downstream movements of Neotropical migratory fish. Fish Fish. 16, 697–715. https://doi.org/10.1111/faf.12089

Pereira, E.H.L., Britto, M.R., 2012. A New Distinctively Colored Catfish of the Genus *Pareiorhaphis* (Siluriformes: Loricariidae) from the Rio Piracicaba, Upper Rio Doce Basin, Brazil. Copeia 2012, 519–526. https://doi.org/10.1643/cg-11-129

Peres-Neto, P.R., Bizerril, P.R., Iglesias, R., 1995. An overview of some aspects of river ecology: a case study on fish assemblages distribution in an eastern Brazilian coastal river, in: Esteves, F.A., ed. Estrutura, Funcionamento e Manejo de Ecossistemas Brasileiros. Editora da Universidade Federal do Rio de Janeiro, Rio de Janeiro, Brazil, pp. 317–334.

Perkin, J.S., Gido, K.B., 2012. Fragmentation alters stream fish community structure in dendritic ecological networks. Ecol. Appl. 22, 2176–2187. https://doi.org/10.1890/12-0318.1

Pirhalla, D.E., 2004. Evaluating fish–habitat relationships for refining regional indexes of biotic integrity: development of a tolerance index of habitat degradation for Maryland stream fishes. Trans. Am. Fish. Soc. 133, 144–159. https://doi.org/10.1577/t01-145

Poff, N.L., Olden, J.D., Merritt, D.M., Pepin, D.M., 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. Proc. Natl. Acad. Sci. 104, 5732–5737. https://doi.org/10.1073/pnas.0609812104

Poff, N.L.R., Bledsoe, B.P., Cuhaciyan, C.O., 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology 79, 264–285. https://doi.org/10.1016/j.geomorph.2006.06.032

Pringle, C.M., 1997. Exploring how disturbance is transmitted upstream: going against the flow. J. North Am. Benthol. Soc. 16, 425–438. https://doi.org/10.2307/1468028

Pringle, C.M., 2001. Hydrologic connectivity and the management of niological reserves: A global perspective. Ecol. Appl. 11, 981–998. https://doi.org/10.1890/1051-0761(2001)011[0981:HCATMO]2.0.CO;2

Prudente, B.S., Pompeu, P.S., Montag, L., 2018. Using multimetric indices to assess the effect of reduced impact logging on ecological integrity of Amazonian streams. Ecol. Indic. 91, 315–323. https://doi.org/10.1016/j.ecolind.2018.04.020

Pulliam, H.R., 1988. Sources, Sinks, and Population Regulation. Am. Nat. 132: 652–661.

R Core Team, 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Viena, Austria.

Rahel, F.J., 2002. Homogenization of freshwater faunas. Annu. Rev. Ecol. Syst. 33, 291–315. https://doi.org/10.1146/annurev.ecolsys.33.010802.150429

Reis, R.E., Albert, J.S., Di Dario, F., Mincarone, M.M., Petry, P., Rocha, L.A., 2016. Fish biodiversity and conservation in South America. J. Fish Biol. 89, 12–47. https://doi.org/10.1111/jfb.13016

Roeser, H.P., Roeser, P.A., 2010. O Quadrilátero Ferrífero - Mg, Brasil: Aspectos Sobre Sua História, Seus Recursos Minerais E Problemas Ambientais Relacionados. Geonomos 18, 33–37. https://doi.org/10.18285/geonomos.v18i1.67

Roxo, F., Costa e Silva, G., Zawadzki, C., Oliveira, C., 2014. *Neoplecostomus doceensis*: a new loricariid species (Teleostei, Siluriformes) from the Rio Doce basin and comments about its putative origin. Zookeys 440, 115–127. https://doi.org/10.3897/zookeys.440.8203

Ruaro, R., Gubiani, É.A., 2013. A scientometric assessment of 30 years of the Index of Biotic Integrity in aquatic ecosystems: Applications and main flaws. Ecol. Indic. 29, 105–110. https://doi.org/10.1016/j.ecolind.2012.12.016

Ruaro, R., Gubiani, E.A., Hughes, R.M., Mormul, R.P., 2019. Global trends and challenges in multimetric indices of ecological condition. Ecol. Ind. 110, 105862. https://doi.org/10.1016/j.ecolind.2019.105862.

Salvador, G.N., Frederico, R.G., Pessali, T.C., Vieira, F., Freitas, T.M.S., 2018. Length-weight relationship of 21 fish species from Rio Doce River basin, Minas Gerais, Brazil. J. Appl. Ichthyol. 34, 1198–1201. https://doi.org/10.1111/jai.13734

Salvador, G.N., Leal, C.G., Brejão, G.L., Pessali, T.C., Alves, C.B.M., Rosa, G.R., Ligeiro, R., Montag, L.F.A., 2020. Mining activity in Brazil and negligence in action. Perspect. Ecol. Conserv. 18, 139–144. doi:10.1016/j.pecon.2020.05.003

Salvador, G.N., Montag, L.F.A., Hughes, R.M., Almeida, S.M., Prudente, B.S., Pessali, T.C., Barroso, T.A., Cianciaruso, M.V., Ligeiro, R., Juen, L., Carlucci, M.B., 2022. Influences of multiple anthropogenic disturbances coupled with a tailings dam rupture on spatiotemporal variation in fish assemblages of a tropical river. Freshw. Biol. In press.

Sanches, B.O., Hughes, R.M., Macedo, D.R., Callisto, M., Santos, G.B., 2016. Spatial variations in fish assemblage structure in a southeastern Brazilian reservoir. Brazilian J. Biol. 76, 185–193. https://doi.org/10.1590/1519-6984.16614

Silva-Junior, E.F., 2016. Land use effects and stream metabolic rates: a review of ecosystem response. Acta Limnol. Bras. 28, e10. https://doi.org/10.1590/S2179-975X0716

Simberloff, D., 2003. Invasive species: what everyone needs to know. Oxford University Press, Oxford, UK, pp. 352.

Smolders, A.J.P., Lock, R.A.C., Van der Velde, G., Medina Hoyos, R.I., Roelofs, J.G.M., 2003. Effects of mining activities on heavy metal concentrations in water, sediment, and macroinvertebrates in different reaches of the Pilcomayo River, South America. Arch. Environ. Contam. Toxicol. 44, 314–323. https://doi.org/10.1007/s00244-002-2042-1

Snyder, C.D., Young, J.A., Villella, R., Lemarié, D.P., 2003. Influences of upland and riparian land use patterns on stream biotic integrity. Landsc. Ecol. 18, 647–664. https://doi.org/10.1023/B:LAND.0000004178.41511.da

Soares, B.E., Cabral, G.L., Estrella, F., Caramaschi, E.P., 2017. Two-decade remaining effects of bauxite tailings on the fish taxonomic structure of a clear-water floodplain lake in central Amazon (Batata lake, Pará state, Brazil). Oecologia Aust. 21, 311–322. https://doi.org/10.4257/oeco.2017.2103.08

Stoddard, J.L., Larsen, D.P., Hawkins, C.P., Johnson, R.K., Norris, R.H., 2006. Setting expectations for the ecological condition of running waters: the concept of reference condition. Ecol. Appl. 16, 1267–1276. https://doi.org/10.1890/1051-0761(2006)016[1267:SEFTEC]2.0.CO;2

Terra, B.D.F., Hughes, R.M., Francelino, M.R., Araújo, F.G., 2013. Assessment of biotic condition of Atlantic Rain Forest streams: a fish-based multimetric approach. Ecol. Indic. 34, 136–148. https://doi.org/10.1016/j.ecolind.2013.05.001

Uieda, V.S., Uieda, W., 2001. Species composition and spatial distribution of a stream fish assemblage in the east coast of Brazil: Comparison of two field study methodologies. Brazilian J. Biol. 61, 377–388. https://doi.org/10.1590/S1519-69842001000300006

Vadas, R.L., Jr., Hughes, R.M., Bello-Gonzales, O., Callisto, M., Carvalho, D., Chen, K., Davies, P.E., Ferreira, M.T., Fierro, P., Harding, J.S., Kleynhans, C.J., Macedo, D.R., Mercado-Silva, N., Moya, N., Nichols, S.J., Pompeu, P.S., Ruaro, R., Stevenson, R.J., Terra, B.F., Thirion, C., Ticiani, D., Yoder, C.O., 2022. Assemblage-based biomonitoring of freshwater ecosystem health via multimetric indices: a critical review and suggestions for improving their applicability. Water Biology & Security doi.org/10.1016/j.watbs.2022.100054

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The River Continuum Concept. Can. J. Fish. Aquat. Sci. 37, 130–137. https://doi.org/10.1139/f80-017

Vázquez, D.P., Simberloff, D., 2002. Ecological specialization and susceptibility to disturbance: Conjectures and refutations. Am. Nat. 159, 606–623. https://doi.org/10.1086/339991

Vieira, F., 2009. Distribuição, impactos ambientais e conservação da fauna de peixes da bacia do rio Doce. MG - Biota 2, 5–22.

Vieira, F., Gomes, J.P.C.G., Maia, B.P., Silva, L.G.M., 2015. Peixes do Quadrilátero Ferrífero: guia de identificação. Fundação Biodiversitas, Belo Horizonte, Brazil, pp. 206.

Vieira, F., Santos, G.B., Alves, C.B.M., 2005. A ictiofauna do Parque Nacional da Serra do Cipó (Minas Gerais, Brasil) e áreas adjacentes. Lundiana 6, 77–87.

Vörösmarty, C.J., McIntyre, P.B., Gessner, M.O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S.E., Sullivan, C.A., Liermann, C.R., Davies, P.M., 2010. Global threats to human water security and river biodiversity. Nature 468, 334–334. https://doi.org/10.1038/nature09549

Wallace, J.B., Grubaugh, J.W., Whiles, M.R., 1996. Biotic indices and stream ecosystem processes: Results from an experimental study. Ecol. Appl. 6, 140–151. https://doi.org/10.2307/2269560

Ward, J., Stanford, J., 1983. The serial discontinuity concept of lotic ecosystems. In: Fontaine, T.D., Bartell, S.M., eds. Dynamics of lotic ecosystems. Ann Arbor Science, Ann Arbor, Michigan, USA, pp. 29–42.

Welcomme, R., 1985. River fisheries. FAO Fisheries Technical Paper, Rome, Italy, pp. 330.

Whittier, T.R., Hughes, R.M., Stoddard, J.L., Lomnicky, G.A., Peck, D. V., Herlihy, A.T., 2007a. A structured approach for developing indices of biotic integrity: Three examples from streams and rivers in the Western USA. Trans. Am. Fish. Soc. 136, 718–735. https://doi.org/10.1577/T06-128.1

Whittier, T.R., Stoddard, J.L., Larsen, D.P., Herlihy, A.T., 2007b. Selecting reference sites for stream biological assessments: Best professional judgment or objective criteria. J. North Am. Benthol. Soc. 26, 349–360. https://doi.org/10.1899/0887-3593(2007)26[349:srsfsb]2.0.co;2

Winemiller, K.O., Agostinho, A.A., Caramaschi, E.P., 2008. Fish ecology in tropical streams. In: Dudgeon, D., ed. Tropical stream ecology. Academic Press, Cambridge, UK, pp. 107–146. https://doi.org/10.1016/B978-012088449-0.50007-8

SUPPLEMENTARY MATERIAL

Supplementary 4-1. Candidate and final metrics (in bold), its description and exclusion reason.

Métric	Description	Exclusion reason
R_TOT	Total species richness	Excluded by quartis
R_CHA	Species richness of Characiformes	Excluded by quartis
R_SIL	**Species richness of Siluriformes**	
R_PER	**Species richness of Perciforme**	
R_CYP	Species richness of Cyprinodontiformes	Excluded by the high number of 0
R_NON	Non-native species richness	Excluded by the high number of 0
R_NAT	Native species richness	Excluded by correlation
R_DTO	Dam tolerant species richness	Excluded by quartis
R_DIN	**Dam intolerant species richness**	
R_%_TOT	Percent of total species richness	Excluded by correlation
R_%_CHA	Percent of species richness of Characiformes	Excluded by correlation
R_%_SIL	Percent of species richness of Siluriformes	Excluded by correlation
R_%_PER	Percent of species richness of Perciforme	Excluded by correlation
R_%_CYP	Percent of species richness of Cyprinodontiformes	Excluded by correlation
R_%_NON	Percent of non-native species richness	Excluded by the high number of 0
R_%_NAT	Percent of native species richness	Excluded by correlation
R_%_DTO	Percent of dam tolerant species richness	Excluded by correlation
R_%_DIN	Percent of dam intolerant species richness	Excluded by correlation
A_TOT	Total abundance	Excluded by quartis
A_CHA	Abundance of Characiformes	Excluded by correlation
A_SIL	Abundance of Siluriformes	Excluded by correlation
A_PER	Abundance of Perciforme	Excluded by correlation
A_CYP	Abundance of Cyprinodontiformes	Excluded by correlation
A_NON	Abundance of non-native species	Excluded by the high number of 0
A_NAT	Abundance of native species	Excluded by quartis
A_DTO	Abundance of dam tolerant species	Excluded by quartis
A_DIN	**Abundance of dam intolerant species**	
A_AS.FA	Abundance of *Astyanax fasciatus*	Excluded by the high number of 0
A_AS.LA	Abundance of *Astyanax lacustris*	Excluded by the high number of 0
A_AS.SC	Abundance of *Astyanax scabripinnis*	Excluded by correlation
A_AS.TA	Abundance of *Astyanax taeniatus*	Excluded by the high number of 0
A_AU.IP	Abundance of *Australoherus ipatinguensis*	Excluded by the high number of 0
A_GE.BR	Abundance of *Geophagus brasiliensis*	Excluded by correlation
A_HO.IN	Abundance of *Hoplias intermedius*	Excluded by the high number of 0
A_HY.SA	Abundance of *Hyphessobrycon* aff. *santae*	Excluded by the high number of 0
A_HY.AF	Abundance of *Hypostomus affinis*	Excluded by the high number of 0
A_KN.MO	Abundance of *Knodus moenkhausii*	Excluded by the high number of 0
A_NE.DO	Abundance of *Neoplecostomsu doceensis*	Excluded by the high number of 0
A_OL.AR	**Abundance of *Oligosarcus argenteus***	
A_OR.NI	Abundance of *Oreochromis niloticus*	Excluded by the high number of 0
A_PA.PR	**Abundance of *Pareiorhaphis* aff. *proskynita***	
A_PA.SC	Abundance of *Pariorhaphis scutula*	Excluded by the high number of 0
A_PH.UA	Abundance of *Phalloceros* aff. *uai*	Excluded by the high number of 0
A_PO.RE	Abundance of *Poecilia reticulata*	Excluded by the high number of 0
A_RH.QU	Abundance of *Rhamdia quelen*	Excluded by quartis
A_TR.AL	Abundance of *Trychomycterus alternatus*	Excluded by the high number of 0
A_TR.BR	**Abundance of *Trichomycterus brasiliensis***	

Métric	Description	Exclusion reason
A_TR.IM	Abrundance of *Trichomycterus immaculatus*	Excluded by the high number of 0
A_%_TOT	Percento of total abundance	Excluded by correlation
A_%_CHA	Percent of abundance of Characiformes	Excluded by correlation
A_%_SIL	Percent of abundance of Siluriformes	Excluded by correlation
A_%_PER	Percent of abundance of Perciforme	Excluded by correlation
A_%_CYP	Percent of abundance of Cyprinodontiformes	Excluded by correlation
A_%_NON	Percent of abundance of non-native species	Excluded by the high number of 0
A_%_NAT	Percent of abundance of native species	Excluded by correlation
A_%_TOL	Percent of abundance of dam tolerant species	Excluded by correlation
A_%_INTOL	Percent of abundance of dam intolerant species	Excluded by correlation
A_%_AS.FA	Percent of abundance of *Astyanax fasciatus*	Excluded by the high number of 0
A_%_AS.LA	Percent of abundance of *Astyanax lacustris*	Excluded by the high number of 0
A_%_AS.SC	Percent of abundance of *Astyanax scabripinnis*	Excluded by correlation
A_%_AS.TA	Percent of abundance of *Astyanax taeniatus*	Excluded by the high number of 0
A_%_AU.IP	Percent of abundance of *Australoherus ipatinguensis*	Excluded by the high number of 0
A_%_GE.BR	Percent of abundance of *Geophagus brasiliensis*	Excluded by correlation
A_%_HO.IN	Percent of abundance of *Hoplias intermedius*	Excluded by the high number of 0
A_%_HY.SA	Percent of abundance of *Hyphessobrycon* aff. *santae*	Excluded by the high number of 0
A_%_HY.AF	Percent of abundance of *Hypostomus affinis*	Excluded by the high number of 0
A_%_KN.MO	Percent of abundance of *Knodus moenkhausii*	Excluded by the high number of 0
A_%_NE.DO	Percent of abundance of *Neoplecostomsu doceensis*	Excluded by the high number of 0
A_%_OL.AR	Percent of abundance of *Oligosarcus argenteus*	Excluded by correlation
A_%_OR.NI	Percent of abundance of *Oreochromis niloticus*	Excluded by the high number of 0
A_%_PA.SP	Percent of abundance of *Pareiorhaphis* aff. *proskynita*	Excluded by correlation
A_%_PA.SC	Percent of abundance of *Pariorhaphis scutula*	Excluded by the high number of 0
A_%_PH.UA	Percent of abundance of *Phalloceros* aff. *uai*	Excluded by the high number of 0
A_%_PO.RE	Percent of abundance of *Poecilia reticulata*	Excluded by the high number of 0
A_%_RH.QU	Percent of abundance of *Rhamdia quelen*	Excluded by correlation
A_%_TR.AL	Percent of abundance of *Trychomycterus alternatus*	Excluded by the high number of 0
A_%_TR.BR	Percent of abundance of *Trichomycterus brasiliensis*	Excluded by correlation
A_%_TR.IM	Percent of abundance of *Trichomycterus immaculatus*	Excluded by correlation

CONCLUSÕES GERAIS

As barragens de rejeito são um grande passivo ambiental da mineração. Seus impactos, apesar de ainda pouco estudados, são significativos sobre a fauna aquática, especialmente sobre os peixes. Muitas vezes pautada como um impacto pontual, que gerará efeito tão somente sobre a área alagada do reservatório criado para armazenar o rejeito da mineração, elas possuem a capacidade de alterar as assembleias de peixes em uma escala longitudinal. Esse impacto, em alguns casos, pode ser sentido a quilômetros do local de inserção da mesma, como mostrado na sessão 1. As alterações provenientes do rompimento da barragem de rejeitos de Fundão, apesar de não ter levado o rio Doce a morte, levaram a uma alteração na composição de sua ictiofauna em seu trecho médio, beneficiando principalmente espécies não nativas. Justamente esse, que é considerado um dos maiores problemas para a conservação da biodiversidade atual, se beneficiou das alterações promovidas pelo desastre. Mostramos também que pouco aprendemos com o desastre com a barragem de rejeitos de Fundão. A ocorrência de um novo desastre em Brumadinho expôs o quão falho os sistemas de prevenção foram. Novamente pagamos um preço ambiental alto, dessa vez na bacia do rio São Francisco. Além disso, mostramos como a burocracia e os conflitos de interesse podem trazer problemas para os trabalhos que visam entender os impactos desses desastres e as propostas de solução. Por fim, mostramos que a construção desse tipo de empreendimento pode afetar até mesmo as comunidades isoladas a montante do reservatório de rejeitos, levando a uma alteração da composição da ictiofauna e ao beneficiamento de espécies generalistas e não nativas. Apesar de ser ainda um pequeno passo frente aos impactos dessas estruturas, concluímos que, sim, elas apresentam efeitos negativos sobre a fauna aquática.

REFERÊNCIAS GERAIS

Abdelouas, A., 2019. World Mine Tailings Failures - from 1915 [WWW Document]. URL https://worldminetailingsfailures.org/ (accessed 2.2.19).

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Balderas, S.C., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J. V., Heibel, T.J., Wikramanayake, E., López, H.L., Reis, R.E.., Lundberg, J.G., Pérez, M.H.S., Petry, P., 2008. Freshwater ecoregions of the world : a new map of biogeo- graphic units for freshwater biodiversity conservation. Bioscience 58, 403–414.

Agostinho, A.A., Gomes, L.C., Pelicice, F.M., 2007. Ecologia e manejo de recursos pesqueiros em reservatórios do Brasil. EDUEM, Maringá.

Agostinho, A.A., Gomes, L.C., Santos, N.C.L., Ortega, J.C.G., Pelicice, F.M., 2016. Fish assemblages in Neotropical reservoirs: Colonization patterns, impacts and management. Fish. Res. 173, 26–36. doi:10.1016/j.fishres.2015.04.006

Alves, C.B.M., Leal, C.G., 2010. Aspectos da conservação da fauna de peixes da bacia do rio São Francisco em Minas Gerais. MG Biota 2, 26–44.

ANA, 2016. Encarte especial sobre a bacia do rio Doce: rompimento da barragem em Mariana. Agência Nacional de Águas/ Ministério do Meio Ambiente, Brasília.

Brosse, S., Grenouillet, G., Gevrey, M., Khazraie, K., Tudesque, L., 2011. Small-scale gold mining erodes fish assemblage structure in small neotropical streams. Biodivers. Conserv. 20, 1013–1026. doi:10.1007/s10531-011-0011-6

Canelas, M.A.S., Bertoluci, J., 2007. Anurans of the Serra do Caraça, southeastern Brazil: species composition and phenological patterns of calling activity. Iheringia. Série Zool. 97, 21–26. doi:10.1590/s0073-47212007000100004

Earle, A.R., Redmann, R.E., Crowder, A.A., Ariano, T.C., Corrigan, C.A., Farmer, R.J., Jackson, L.M., 2018. Iron Ore, in: Redmann, R.E. (Ed.), Environmental Effects of Mining. Routledge Taylor & Francis Group, Abingdon, pp. 225–233.

Escobar, H., 2015. Mud tsunami wreaks ecological havoc in Brazil. Science (80-.). 350, 1138–1139.

doi:10.1126/science.350.6265.1138

Espósito, T. de J., Duarte, A.P., 2010. Classificação de barragens de contenção de rejeitos de mineração e de resíduos industriais em relação a fatores de risco. Rem Rev. Esc. Minas 63, 393–398. doi:10.1590/S0370-44672010000200026

Fernandes, G.W., Goulart, F.F., Ranieri, B.D., Coelho, M.S., Dales, K., Boesche, N., Bustamante, M., Carvalho, F.A., Carvalho, D.C., Dirzo, R., Fernandes, S., Galetti, P.M., Millan, V.E.G., Mielke, C., Ramirez, J.L., Neves, A., Rogass, C., Ribeiro, S.P., Scariot, A., Soares-Filho, B., 2016. Deep into the mud: ecological and socio-economic impacts of the dam breach in Mariana, Brazil. Nat. e Conserv. 14, 35–45. doi:10.1016/j.ncon.2016.10.003

Hudson, T.L., Fox, F.D., Plumlee, G.S., 1999. Metal Mining and the Environment. American Geological Institute, Alexandria, Virginia.

Hughes, R.M., Amezcua, F., Chambers, D.M., Daniel, W.M., Franks, J.S., Franzin, W., MacDonald, D., Merriam, E., Neall, G., dos Santos Pompeu, P., Reynolds, L., Woody, C.A., 2016. AFS Position Paper and Policy on Mining and Fossil Fuel Extraction. Fisheries 41, 12–15. doi:10.1080/03632415.2016.1121742

IBRAM, 2016. Gestão e manejo de rejeitos da mineração. Intituto Brasileiro de Mineração, Brasília, DF.

IBRAM, 2011. Sistema de Informações e Análises da Economia Mineral Brasileira, 6th ed. Instituto Brasileiro de Mineração, Brasília, DF.

ICMM, 2012. Trends in the mining and metals industry, Mining's contribution to sustainable development. London.

Jacobi, C.M., Carmo, F.F., 2008. Diversidade dos campos rupestres ferruginosos no Quadrilátero Ferrífero , MG. Megadiversidade 4, 25–33.

Kossoff, D., Dubbin, W.E., Alfredsson, M., Edwards, S.J., Macklin, M.G., Hudson-Edwards, K.A., 2014. Mine tailings dams: Characteristics, failure, environmental impacts, and remediation. Appl. Geochemistry 51, 229–245. doi:10.1016/j.apgeochem.2014.09.010

Monteiro, V.S., Umbelino, A.I., Damato-Neto, J., Cusati, G.G., Rosado, V.B., 2014. Qualidade da água após impacto do rompimento da barragem São Francisco, in: Congresso Brasileiro de Gestão

Ambiental. Instituto Brasileiro de Estudos Ambientais e de Saneamento, belo, pp. 1–5.

Neves, A.C. de O., Nunes, F.P., de Carvalho, F.A., Fernandes, G.W., 2016. Neglect of ecosystems services by mining, and the worst environmental disaster in Brazil. Nat. Conserv. 14, 24–27. doi:10.1016/j.ncon.2016.03.002

Oliveira, V.D.C., Oliveira, D.D.C., 2019. A semântica do eufemismo: mineração e tragédia em Brumadinho. Rev. Eletrônica Comun. Informação e Inovação em Saúde 13, 13–38. doi:10.29397/reciis.v13i1.1783

Rico, M., Benito, G., Díez-Herrero, A., 2008. Floods from tailings dam failures. J. Hazard. Mater. 154, 79–87. doi:10.1016/j.jhazmat.2007.09.110

Roeser, H.P., Roeser, P.A., 2010. O Quadrilátero Ferrífero -Mg, Brasil: Aspectos Sobre Sua História, Seus Recursos Minerais E Problemas Ambientais Relacionados. Geonomos 18, 33–37.

U.S. DOI/ Bureau of Mines, 1983. Development guidelines for closing underground mines: Michigan case histories. Washington DC.

Vieira, F., 2009. Distribuição, impactos ambientais e conservação da fauna de peixes da bacia do rio Doce. MG - Biota 2, 5–22.

WEF, 2010. Mining & metals: Scenarios to 2030. Cologry, Geneva.