Patterns and ecological implications of small hydropower development in Brazil

Thiago Belisário d'Araújo Couto

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Reading Committee:

Julian D. Olden, Chair

Gordon W. Holtgrieve

Timothy J. Beechie

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University of Washington

Abstract

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Thiago Belisário d`Araújo Couto

Chair of the Supervisory Committee:
Dr. Julian D. Olden
School of Aquatic and Fishery Sciences

The growing demand for renewable energy is fueling a global proliferation of Small Hydropower Plants (SHPs), which represents an emerging threat for freshwater ecosystems and aquatic biodiversity. The general perception that SHPs have low environmental impacts is implied in incentive policies for new contruction, although there is little evidence that “small” necessarily equates to lower impacts – especially if the ecological effects of the sheer number of SHPs are considered cumulatively. Here, I investigate the ecological effects of SHPs on river ecosystems, and potential cumulative changes to river connectivity, habitat loss and water quality. 1) I provide the first global overview of the proliferation of SHPs, summarizing status and trends of science and policy on SHPs. 2) Using Brazil as a case study, I explore the role of current and projected-future SHPs on cumulative river fragmentation, and further identify planned dam portfolios that maximize energy gains while minimizing river connectivity losses. 3) I dive into the case of the Chapecó River Basin (Brazil) to investigate in situ the ecological effects of habitat changes by SHPs on invertebrate and fish communities, and 4) the effects of different SHPs on downstream water thermal regimes. More than 80,000 SHPs are operating in 150 countries and the number of future SHPs are estimated to increase three-fold if all potential is explored. SHPs are the primary source of river fragmentation in Brazilian basins, which does not necessarily translate in significant gains in energy generation capacity. Thus, optimized choices of construction could meet future energy demands with much less river fragmentation. SHPs modify the composition of invertebrate and fish communities of the Chapecó Basin, but the magnitude of changes vary widely among dams and can be in some instances predicted by basic SHP attributes (e.g. dam height). Similarly, the effects of SHPs on water thermal regimes is quite variable among dams, but surface warm-water releases were the most common effect. Overall, these results show that the individual and cumulative ecological effects of SHPs are potentially high, highlighting the need for dissolving the “small” modifier from environmental policies and for strategic planning for future construction.

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DEDICATION

To my parents, Kátia and Rogério, who sacrificed a lot from their lives for my education. I also dedicate this work to my siblings, my wife Fernanda and my coming daughter, Marina.

Chapter 1. GLOBAL PROLIFERATION OF SMALL HYDROPOWER PLANTS – SCIENCE AND POLICY

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Large-scale electricity policies that embrace renewable resources have led to continued investments in hydropower. Despite evolving viewpoints regarding the sustainability of large hydropower installations, there has been a major increase in support for the widespread development of small hydropower plants (SHPs). A global synthesis reveals that 82,891 SHPs are operating or are under construction (11 SHPs for every one large hydropower plant) and that this number is estimated to triple if all potential generation capacity were to be developed. Fueled by considerable political and economic incentives in recent decades, the growth of SHPs has greatly outpaced available ecological science. We provide evidence for not only the lack of scientifically informed oversight of SHP development but also the limitations of the capacity-based regulations currently in use. The potential indiscriminate expansion of SHPs under the pretense of promoting sustainable energy is concerning, and we identify several important steps to help ensure new scientific advances, effective management, and policy reform in the future.

1.1 INTRODUCTION

One of the greatest challenges of this century is to support growing human societies with electricity policies that embrace environmentally sustainable resources. Hydropower, the power derived from the energy of flowing water, is the world’s primary source of renewable energy, contributing almost three-quarters of the global renewable supply and nearly one-fifth of all electricity production (REN21, 2015). Global investments in hydropower peaked in the second half of the 20th century, partly in response to a growing desire to diversify away from thermoelectric facilities and avoid greater dependency on fossil fuels (Oud, 2002). In response to the mounting demand for renewable energy, thousands of new hydropower installations are expected to be built and to commence operation in the coming decades (Zarfl, Lumsdon, Berlekamp, Tydecks, & Tockner, 2014). These activities are most prevalent in developing countries where governments are prioritizing new large dams as the centerpiece of energy plans (Winemiller et al., 2016). Unfortunately, these dams have substantial socioecological costs (Rosenberg, Bodaly, & Usher, 1995). Indeed, public support for new large dams has diminished as the high socioeconomic costs, emissions of greenhouse gases, and negative consequences for valued ecosystem services, including water quantity and quality, biodiversity, and fisheries, are increasingly realized (Ansar, Flyvbjerg, Budzier, & Lunn, 2014; Deemer et al., 2016; Kahn, Freitas, & Petrere, 2014).

Amid evolving conversations regarding large hydropower sustainability, a marked surge of support for small hydropower plants (SHPs) has emerged. The term SHP broadly refers to facilities that produce less electricity and operate in smaller rivers as compared to large hydropower plants (LHPs). However, because SHPs have a diversity of operation modes (e.g. diversion and non-diversion with or without storage), flow control structures, and environmental impacts (see section 1.1.1), this definition is ambiguous. Political and economic incentives for sustainable energy development initially fueled the growth of SHPs, particularly in Europe and China (Paish, 2002; Tang et al., 2012), but now countries across the world consider SHPs to be a critical component of future energy strategies. For instance, more than half of US states have renewable portfolio standards that disallow electricity from LHPs, yet they embrace power produced from SHPs (Kao et al., 2014), due in part to a perception that “smaller” equates to lower socioecological impact (Gleick, 1992). As this viewpoint continues to gain traction in public and political arenas, the lack of scientifically informed regulations (Erlewein, 2013; Gleick, 1992) and the potential indiscriminate expansion of SHPs under the pretext of sustainable energy promotion is concerning (Premalatha, Tabassum-Abbasi, Abbasi, & Abbasi, 2014).

We provide the first global overview of the expansion of SHPs, summarizing status and trends in an international context, and encouraging the bridging of critical science and policy gaps. First, we explore the rapid growth of the sector and quantify the current and future hotspots of SHP development based on a dataset compiled from the most up-to-date international energy policy reports. The dataset includes national definitions of SHPs, current numbers of SHPs and LHPs, and estimates of future SHP numbers based on countries’ untapped hydropower potential. Second, we expose the global inconsistencies in how SHPs are defined, and argue for the removal of the “small” modifier. Third, we discuss the environmental policies and the scientific knowledge that currently shape the management and regulatory practices of SHPs. Finally, we provide a synopsis of the scholarly literature examining the known or potential ecological impacts of SHPs, and identify the main challenges to ensure that environmental policies inform the worldwide expansion of SHPs.

1.1.1 *What is small hydropower?*

There is great diversity in the type and size of small hydropower plants (SHPs). The general principle of hydropower generation is the conversion of water pressure into mechanical shaft power by hydroturbines (Paish, 2002). Despite the diversity of operation modes, hydropower can be classified according to the level of flow control (i.e. proportion of water stored) and the presence of diversion structures (McManamay, Oigbokie, Kao, & Bevelhimer, 2016).

The degree of flow control ranges from installations that store water in reservoirs to schemes that do not retain much water (i.e. run-of-river) and are subjected to natural fluctuations of river discharge (Fig. 1.1). Some facilities are intermediate to these classes, in the sense that they control the flow only during specific periods of the year or times of day (McManamay et al., 2016). Reservoirs are created to minimize variability in water supply, storing water for periods of low flow or high electricity demand (Egré & Milewski, 2002). In multi-purpose hydropower dams, reservoirs are also used to supply other human needs such as irrigation, flood control, and urban consumption (Bernhard Lehner et al., 2011). Whereas operations involving water storage cause major alteration to natural flows of rivers, run-of-river schemes use weirs or small dams to block the river channel, in theory maintaining more natural flow conditions (S. Csiki & Rhoads, 2010).

Non-diversion schemes use water from an impoundment structure that is adjacent to the powerhouse, whereas diversion schemes transport water via canals or pipes to a distant powerhouse (Fig. 1.1). Thus, mainstream sections in diversion schemes are dewatered until the water is returned after passing through the turbines. The amount of water abstracted and transferred to the powerhouse dictates the magnitude of river dewatering, which can be very high during drought periods (N. C. Wu et al., 2010). Dewatered sections range from very short distances to several kilometers of reduced flow, depending on the SHP project (E. P. Anderson, Freeman, & Pringle, 2006). SHPs can be operated as complexes of multiple installations both in close proximity and distributed broadly within a watershed. For example, some run-of-river SHPs are located downstream of large reservoirs that are used to store water and control flow (McManamay et al., 2016). Others can use water transported from remote basins via interbasin transfer, causing major alterations to the ecosystem (E. P. Anderson et al., 2006; Cada & Zadroga, 1982). These examples emphasize the complexity of the environmental repercussions from the implementation of multiple SHP facilities.

1.2 METHODS

1.2.1 *Compiling information on the global status of SHPs*

We assembled a global dataset of SHPs by leveraging a rich literature on national energy policy that encompassed 203 countries and territories (the full dataset is available at https://figshare.com/s/e118ecae964140f16f80). We collected information on national definitions of SHP, and the number of SHP installations (operation, under construction, planned and potential), total SHP capacity (MW; planned and potential capacities), number of LHPs at the country level, and the installed capacity of hydropower in general. Information was first retrieved from the reports present in the World Small Hydropower Development Report (WSHPDR, 2016), representing the most recent global compendium on SHPs. The number of SHPs and capacities reported in WSHPDR (2016) correspond to each country-level current definition of SHP, except for Albania, Bolivia, Germany, Poland, and Russia that refers to plants up to 10 MW or 30 MW (in the case of Russia). Additional data on SHP numbers were obtained from other reputable sources for countries with incomplete information in WSHPDR (2016): including Ecuador (Agencia de Regulación y Control de Electricidad, 2015), Malaysia (SEDA, 2017), Czech Republic (ESHA, 2016), France (ESHA, 2016), Germany (ESHA, 2016), Hungary (ESHA, 2016) and Ireland (ESHA, 2016). In such cases, we gathered the most updated information available. Similarly, the number of existing SHPs in Canada was obtained for provinces from different agencies/energy providers/non-profits websites and reports because a national compilation could not be found (BC Hydro, 2017; Energy BC, 2017; Hydro-Québec, 2017; Ontario Power

Generation, 2017; TransAlta, 2017). The previous edition of the WSHPDR (Liu, Masera, & Esser, 2013) was used to obtain SHP numbers for 13 countries that numbers were not provided in WSHPDR (2016): including Afghanistan, Albania, Bangladesh, Bolivia, Chile, Croatia, Cyprus, Democratic Republic of Congo, Japan, Myanmar, Panama, and Vietnam. We estimated the current number of SHPs in operation or under construction for seven countries for which information was not readily available from the literature or any reputable electronic sources (Equatorial Guinea, Indonesia, Laos, Namibia, North Korea, Philippines, and Uzbekistan). To accomplish this, we divided the total installed capacity of each country (MW) by its upper limit of SHP definition and rounded to a whole number. We adopted 10 MW (the most common upper definition in the dataset) for countries that lack formal SHP definition.

Future numbers of SHPs were classified as either potential or planned. Potential SHPs represent new installations that are expected to be built if all current economically/technically feasible potential is explored. These levels may or may not be achievable in a long-term. Planned SHPs represent new installations that are included in national development plans for the next couple of decades (target years are not explicit for all countries, but generally ranged between 2020 and 2030). The number of potential and planned SHPs were retrieved from country reports contained in WSHPDR (2016), representing 61 and 72 countries, respectively. For country reports that do not provide future numbers of SHPs, we calculated estimates using the installed, planned and potential capacities retrieved from WSHPDR (2016). Installed capacity is the sum of all capacities of SHPs operating, potential capacity is the reported total capacity of a country that may be explored, and planned capacity is the reported capacity that is part of national plans for the next decades. The potential capacity reported in WSHPDR (2016) differ from the gross potential of a country because it only comprises the potential that is economically and technically feasible according to accessibility, proximity to the grid, permits, ownership, and other factors that can define investments in development. Not all countries have detailed studies to determine potential and planned capacities (lacking or incomplete assessments especially in developing countries) and some do not have any SHP potential (e.g. most countries of the Arabian Peninsula). Both cases were treated as missing values in the dataset for potential and planned SHPs (total of 52 and 113 countries, respectively). For the calculations of future numbers based on potential and planned capacities, we first estimated the remaining potential capacity by subtracting the installed capacity from the potential capacity for each country. Next, we divided the remaining capacity by the average capacity of current SHPs in the country; the latter was estimated by dividing the installed capacity by the number of SHPs currently in operation. Consequently, our estimates of future numbers of SHPs assume that the average capacity will be constant through time. To avoid unrealistic overestimations for countries with extremely small averages, we set 1 MW as the minimum average capacity. Eight countries have reported potential capacity, but yet to have any SHPs in operation (i.e., Botswana, Gambia, Ghana, Grenada, Guyana, Republic of Congo, South Sudan, and Uruguay); in these cases we assumed 10 MW as the average capacity or the reported definition if established. Numbers of potential and planned SHPs were estimated based on capacities for 90 and 18 countries, respectively.

The number of LHPs for each country was primarily retrieved from the International Commission of Large Dams dataset (ICOLD, 2017). This dataset contains a total of 9,664 hydropower plants, of which 7,239 plants have a generation capacity above the upper limits of country's classifications of SHPs. Hydropower plants below classification thresholds of SHPs of each country were not included, but plants with missing data on capacity were conservatively considered to be LHPs. We also considered two additional databases: WSHPDR (2016) and ESHA

(2016). The total number of LHPs per country (total = 7,721) was considered the largest reported number among these three data sources. We assumed that the number of LHPs is zero if no data for a country was reported in all three sources.

The expansion of SHPs through time was explored using installed capacities of hydropower from 2000 to 2016. Information on installed capacity per year was retrieved from reports by the International Renewable Energy Agency (IRENA, 2015, 2016, 2017). These reports contain estimates of hydropower installed capacity per year by country (continents grouping followed the same classification used in the global dataset of SHPs). Estimates represented in the reports are organized by capacity (< 1 MW, 1-10 MW, and > 10 MW). For this analysis, all hydropower plants below 10 MW were considered SHPs.

1.2.2 *Literature review of the ecological dimensions of SHPs*

We conducted our literature search on hydropower in two phases, which contributed to the statistics presented in the section “Science lags behind the rapid rise of small hydropower plants”. The most relevant literature found in these two phases and other literature found through readings were used in the section “Emerging evidence for the ecological impacts of small hydropower plants”.

First, we searched the Institute of Scientific Information (ISI; Thomson Reuters) Web of Science online database and identified peer-reviewed papers published through 2016 that contained the key-terms “Hydropower OR Hydroelectric OR Hydro-electric OR Hydro-Power” for the literature addressing all hydropower and then added the key-terms "Small Hydro*" OR "Micro Hydro*" OR "Mini Hydro*" OR "Pico Hydro*" OR "Small-scale Hydro*" for the literature addressing just SHPs. In each search, we filtered the research areas: “environmental sciences ecology”, “marine and freshwater biology” and “biodiversity conservation”. The papers published in the journal “Energy Policy” were excluded from the analysis because of their exclusive focus on policy instead of ecology and conservation. A total of 2,282 and 101 articles were identified for all hydropower and SHPs, respectively.

Second, we quantified the number of published papers that reported the generation capacity of the hydropower installations examined regardless of size. Using the same search terms described above for all hydropower, (n = 2,282 articles) we identified all papers published since 1980 (earlier publications were difficult access) that had titles classified as “marine and freshwater biology”. This resulted in a total of 539 articles. Next, we reviewed the entire text of each paper to identify those that reported the generation capacity of the studied installations, which resulted in 82 articles. In papers that examine more than one hydropower plant, we retrieved the mean capacity. We excluded from this analysis the small number of papers that bulked capacities for more than one dam without specifying the capacity of each facility. We also excluded from the analysis review papers, investigations of potential sites for construction and one paper that compared the cumulative effects of many large and small hydropower installations using modeling approaches. The hydropower capacities informed by the studies ranged between 0.03 to 4,800 MW.

1.3 RESULTS AND DISCUSSION

1.3.1 *Global proliferation of SHPs*

Our synthesis of the energy policy reports reveals that at least 82,891 SHPs are operating or are under construction in 150 countries (Fig. 2a). We estimate a tripling of this number to include an additional 181,976 plants that could be installed if all potential capacity were to be developed – of these, 10,569 new projects already appear in national plans (Fig. 2b). Distinct geographic patterns in SHP numbers are apparent, reflecting differences in socioeconomic conditions, varying regulations and incentives, and contrasting hydrologic potential. China is the global frontrunner, with 47,073 SHPs currently operating; the surge was propelled by private investments, technology leadership, and rural electrification programs (Tang et al., 2012; WSHPDR, 2016). An additional 26,877 SHPs are operating in Europe, where SHP development not only has a long history but has also received recent attention associated with the need to satisfy international agreements promoting clean energy (Hermoso, 2017; Paish, 2002). Future plans for SHPs are concentrated in Asia, the Americas, Southern and Eastern Europe, and East Africa. Considering that national dam inventories generally underestimate the number of smaller facilities, we expect that the numbers of SHPs reported here are conservative. Even based on these values, the current number represents more than 91% of the total hydropower installations, dwarfing the number of LHPs (n = 7,721; Fig. 3a). On average, there are close to 11 SHPs for every one LHP in the world. Although representing the vast majority of hydropower plants, SHPs contribute just 11% of the global electricity generation capacity based on hydropower.

SHPs are proliferating around the world (Fig. 3b; IRENA 2017) as a response to governmental incentives in the form of competitive tariffs, as well as investments by the private sector (i.e. trading energy with the grid or providing lower-cost energy for local industries), programs promoting rural electrification in remote regions, simplified licensing processes, and quick construction time (Egré & Milewski, 2002; Tang et al., 2012; WSHPDR, 2016). For example, aggressive policies in several European countries have encouraged rapid sector development since the 1980s (Jesus, Formigo, Santos, & Tavares, 2004; Paish, 2002). Numerous other countries are now duplicating such policy initiatives. In Brazil, a series of incentives and new regulations contributed to a fivefold increase in the number of SHPs over the past two decades (see section 1.3.2). Even in the US, where current conversations typically focus on the removal of aging dams (Bellmore et al., 2016), a recent US Department of Energy report highlights that 65 gigawatts of potential hydropower remain untapped (Kao et al., 2014), and that new hydropower would come predominantly in the form of hundreds of new SHPs.

1.3.2 *Brazil as a model of the rapid expansion of SHPs*

In Brazil, 1007 SHPs are currently operating, an additional 35 are under construction, and 156 are approved and awaiting final licensing (ANEEL, 2016). On average, 33 SHPs have been constructed per year from 2001 to 2016, a growth rate 14 times as fast as that witnessed in the 1990s (Fig. 1.4). By contrast, increases in LHPs have remained constant in recent decades. Massive investments by the private sector after 1995 were stimulated by new regulations within the energy market and by economic incentives, resulting in hundreds of new SHPs (Ferreira, Camacho, Malagoli, & Guimarães, 2016) (Appendix A. Fig. S1). Simultaneously, there was a reformulation of the environmental licensing process (CONAMA 237/1997) that later included the introduction of simplified licensing procedures for projects with perceived "little potential of causing environmental disturbance", which were defined as SHPs with less than 10 MW (CONAMA 279/2001). This licensing reform was followed by a re-definition in 2003 of what constitutes an SHP, which quadrupled the maximum reservoir area from 3 to 13 km^2 while maintaining the maximum limit of 30 MW (Ferreira et al., 2016). In November 2016, a new bill was approved, increasing the upper limit of SHPs from 30 to 50 MW (Law Nº 13.360/2016).

1.3.3 *Just how "small" are SHPs?*

Our assessment revealed notable disparities in how countries classify hydropower plants as "small" (Fig. 1.5). The size of a given hydropower facility is commonly defined according to its generation capacity in watts (hereafter termed "capacity"), which is the maximum capacity of hydropower production assuming optimal hydrologic conditions and turbine efficiency. Capacity-based definitions of SHPs vary substantially, ranging from up to 1 megawatt (MW) for facilities in Germany and Burundi to up to 50 MW for facilities in Canada, China, and Pakistan (Fig. 1.5). About 70% of countries with formal definitions classify SHPs as installations with less than 10 MW, which is increasingly recognized as the international standard (WSHPDR, 2016). Of those countries that currently operate SHPs, 15% do not have a formal (or widely accessible) national designation (Fig. 1.5). In addition to considerable variability in SHP definitions across countries, facilities designated as SHPs may have substantially different dam sizes, reservoir areas, storage capabilities, outlet structures, and operating procedures (Fig. 1.1; Fig. 1.6), all of which are strongly correlated with environmental impacts of dams (Gleick, 1992; Mbaka & Mwaniki, 2015; Poff & Hart, 2002). For instance, two SHPs in Brazil (São Sebastião, Braço Norte II) have the same capacity (10 MW), but differ thirtyfold with respect to their reservoir areas (0.2 and 6.0 km^2, respectively).

In addition to the limited and internationally inconsistent definitions of SHPs, there are considerable discrepancies in hydropower classification within countries; this means that some nations may support policies that are based on different capacity thresholds. In Russia, contrasting regulations classify SHPs as facilities generating less than either 25 or 30 MW (WSHPDR, 2016). SHP definitions remain highly dynamic elsewhere, including in the US and Brazil, where recent amendments increased the upper limits from 5 to 10 MW (Hydropower Regulatory Efficiency Act of 2013) and from 30 to 50 MW (Law 13.360/2016), respectively. Some countries further delineate SHPs into small, mini, micro, and pico hydro classifications. For the purposes of this review, we consider SHPs as all those with maximum capacity defined according to the country-specific definition or 10 MW for countries without national definitions.

1.3.4 *Limited environmental regulations for SHPs*

The rapid expansion of SHPs is due in part to weak regulatory oversight that has encouraged investments in the private and public sectors. According to a global policy compilation, at least 44 out of 160 countries require a formal environmental licensing process to construct and operate SHPs (WSHPDR, 2016), leaving potentially more than two-thirds of the countries without recognized environmental requirements. Certain countries have regulations requiring mitigation actions – such as fishway structures that attempt to facilitate fish movement and standards of minimum flow releases aimed at sustaining human and environmental water needs – when SHPs are constructed. In France, all hydropower installations, regardless of size, must have facilities that promote passage of migratory fish species (Larinier, 2008). By contrast, countries such as Costa Rica lack formal rules of minimal flow releases or else adopt arbitrary standards based on the policies of other countries (E. P. Anderson et al., 2006).

When present, environmental licensing processes associated with SHPs vary substantially among countries, but are often less rigorous than licensing of LHPs. Installation of LHPs frequently requires Environmental Impact Assessments (EIA), public consultations, management plans, and monitoring programs (Erlewein, 2013), whereas the installation of SHPs may involve only a simplified EIA if anything at all. In India, SHP construction (< 25 MW) does not necessitate an EIA; instead, only a Detailed Project Report must be approved, and even these evaluations are commonly inadequate (Erlewein, 2013). Similarly, in Brazil, only a Simplified Environmental Report is required for constructing SHPs with a capacity of less than 10 MW, but larger SHP projects (10 to 30 MW) are subjected to a licensing process similar to that of LHPs (see section 1.3.2). In the US, SHPs of less than 10 MW are exempt from some licensing requirements; similar exemptions are also granted to hydropower projects up to 40 MW placed along canals associated with agricultural, municipal, or industrial water consumption (Hydropower Regulatory Efficiency Act of 2013). Prior to 2008, EIAs were not mandatory in Turkey for installations up to 50 MW, but now projects between 0.5 and 20 MW require full EIAs (Dursun & Gokcol, 2011). These examples reinforce the notion that the "small" in SHP policies continues to be equated with negligible environmental impacts, and a streamlined approval process with inadequate regulatory oversight is applied in many instances.

Perhaps most concerning is that environmental regulations for SHPs are mandated on a single and arbitrary capacity threshold that does not necessarily equate with the magnitude of environmental impacts (Gleick, 1992). By pooling diverse hydropower projects, these regulatory frameworks overlook considerable variability in modes of operation, degree of flow modification, dam size, and inundated area, which ultimately define the ecological impacts of dams (Egré & Milewski, 2002; Poff & Hart, 2002). Brazil and Turkey include reservoir area in their SHP definitions (as large as 13 km^2 and 15 km^2, respectively) but these are exceptions (Dursun & Gokcol, 2011; Ferreira et al., 2016).

1.3.5 *Science lags behind the rapid rise of SHPs*

The widespread increase in SHPs continues to greatly outpace advances in scientific knowledge in the environmental field. Of the publications that we reviewed, fewer than 5% explicitly investigated SHPs, even though SHPs represent more than 91% of existing hydropower installations (Fig. 1.7). Research on SHPs grew in the 2000s when the sector flourished in many countries, a trend that started more than three decades earlier for LHPs. Research on LHPs has been fundamental to understanding, regulating, and attempting to mitigate the ecological impacts of hydropower (Kahn et al., 2014; J. Olden, 2016); however, we question whether such knowledge is transferable to inform policy and management decisions for SHPs. Furthermore, although generation capacity is the benchmark used to regulate and manage hydropower, only 15% of the papers included in our synthesis reported the capacities of the studied installations, making generalizations between SHPs and LHPs even more difficult. In short, the rise in the number of scholarly publications has not been commensurate with the rapid increase in SHPs observed globally.

1.3.6 *Emerging evidence for the ecological impacts of SHPs*

The diversity of sizes and operation modes of SHPs (Fig. 1.1) produce a myriad of ecological consequences that may not necessarily differ from what is expected from LHPs (Cada & Zadroga, 1982). Despite relatively slow progress in scientific research, evidence suggests that the environmental impacts of SHPs include those associated with the dam's construction and land inundation, as well as post-construction alteration to flow regimes, the loss of habitat connectivity, and the cumulative effects of multiple installations. However, the magnitude of such impacts may depend more on the attributes of individual projects and on the landscape context in which they are located, rather than merely being related to an arbitrary size classification (Gleick 1992). Perhaps the greatest difference between SHPs and LHPs is their position in the watershed; SHPs generally occur in smaller rivers as compared to LHPs (Kibler & Tullos, 2013). This is problematic given the importance of headwater streams in maintaining hydrologic connectivity, harboring biodiversity, and supporting ecosystem integrity at regional scales (Meyer et al., 2007).

The construction of the main (e.g. dams, reservoirs, diversion channels, penstock, powerhouses) and accompanying (e.g. roads, transmission lines, substations) structures are the initial sources of impact. During construction, there are direct impacts of deforestation and land preparation, as well as indirect impacts associated with transportation, construction materials, and energy requirements (Cada & Zadroga, 1982; Pang, Zhang, Wang, & Liu, 2015). When SHPs impound water, the newly created reservoirs produce greenhouse gases through mechanisms such as decomposition (Deemer et al., 2016) and inundate terrestrial habitats, potentially threatening biodiversity and compromising ecosystem functions (Benchimol & Peres, 2015).

Hydropower dams and diversions not only modify downstream flow regimes, channel morphology, and water temperature but also affect sediment transport and deposition (Baker, Bledsoe, Albano, & Poff, 2011; J. Olden, 2016). These modifications tend to be more evident in installations with greater storage capacities (Mbaka & Mwaniki, 2015), but they are also present for run-of-river SHP schemes (David Anderson, Moggridge, Warren, & Shucksmith, 2015; Meier, Bonjour, Wuest, & Reichert, 2003; Stanley, Luebke, Doyle, & Marshall, 2002). Among the various ecological effects (Table 1.1), artificially warm or cold water is released downstream from smaller dams, depending on the mode of operation and thermal profile of the reservoir, leading to impacts on biological communities (Hayes, Dodd, & Lessard, 2008). The transformation of flowing rivers to shallow, standing water can have numerous consequences that include effects on primary production and the alteration of algae, invertebrate, and fish communities (Jesus et al., 2004; Kubecka, Matena, & Hartvich, 1997; N. C. Wu et al., 2010).

Damming rivers can create major conservation problems by fragmenting critical habitats (Gido, Whitney, Perkin, & Turner, 2016). Dams and weirs associated with SHPs represent physical barriers for migratory species that rely on connected rivers to move upstream to spawn, to access floodplains, and for downstream migrations (Benstead, March, & Pringle, 1999; Dubois & Gloss, 1993; Ovidio & Philippart, 2002). Recent advances in fishway construction seek to mitigate some of these impacts; however, scholarly literature sources suggest that fishways often fail to support fish migrations (J. Olden, 2016). Several issues related specifically to fishways in SHPs remain to be resolved (Larinier, 2008). In Portugal, for example, over half of the SHP fishways are unsuitable for fish migration due to poor performance, insufficient structural integrity, and lack of maintenance (José Maria Santos et al., 2012).

Perhaps the most underappreciated issue is that the ecological impacts of SHPs may be detectable at the level of multiple installations, so SHPs should be considered collectively and not in isolation. In China, the cumulative effects of hydrologic modifications and reduced connectivity in rivers associated with 31 SHPs exceed those caused by four LHPs on a per unit of energy comparison (Kibler & Tullos, 2013). The presence of multiple obstacles also hinders the dispersal abilities of freshwater organisms. For instance, after facing a series of obstacles along their migration route, only 18% of tagged Atlantic salmon (*Salmo salar*) reached spawning grounds above a hydropower complex in Scotland (Gowans, Armstrong, Priede, & McKelvey, 2003). Given the huge number of current and planned SHPs, determining the ecological effects of multiple rather than individual installations deserves urgent attention.

1.3.7 *Management and policy*

Environmental policies must support robust and science-based licensing regulations for SHPs. Differentiating “small” from “large” hydropower is a primary decision node in most licensing frameworks when determining whether a full EIA (versus a simplified one, or none at all) is required for a new installation. However, a capacity-based criterion alone is insufficient to define expectations regarding the potential environmental impacts of hydropower, especially considering that the thresholds are often arbitrarily defined. Moreover, by focusing exclusively on a single attribute of an installation (namely its generation capacity, the desired product of hydropower), current environmental regulations could run the risk of favoring less productive, ecologically harmful projects over ones that are more productive and ecologically benign. SHP regulations must look beyond capacity and incorporate flow alteration, inundation area, impacts on habitat connectivity, and the cumulative effects of multiple installations.

We recognize the many challenges of conducting detailed EIAs for every potential hydropower project, especially in countries with limited financial and human resources. However, we advocate that EIAs should be compulsory during the licensing process for all SHP projects given the diversity of facility sizes, operation modes, and geographic locations. Adjusting site-specific characteristics to maximize electricity generation requires a wide range of technical solutions, which may create a myriad of environmental changes (Egré & Milewski, 2002). By considering SHPs as a homogeneous group in licensing frameworks, policy makers are likely to underestimate the facilities' potential environmental impacts. EIAs can help to identify specific impacts on the affected ecosystem, quantify threats to biodiversity, address disruptions to coupled human–natural systems (e.g. subsistence fishing, access to agricultural lands, social and cultural values), and propose mitigation actions. Environmental assessments must additionally go beyond individual SHP projects and consider the broader watershed context before new construction is approved. The continued proliferation of SHPs necessitates systematic evaluations that take into account the cumulative effects of current and planned installations, a process that is overlooked by most present-day environmental regulations. By incorporating elements of integrated watershed management (Wang et al., 2016) into hydropower strategic planning, managers can ensure adaptive, multifaceted, and watershed-scale evaluations based on the location and attributes of proposed projects.

Ecological data are critically needed to support management strategies and policy reforms that attempt to minimize the impacts of SHPs. The vast and decentralized nature of SHPs imposes major challenges to the acquisition and compilation of fundamental information, especially when compared to LHPs (Bernhard Lehner et al., 2011; Zarfl et al., 2014). We argue that the collation of basic attributes of present and future SHPs is fundamental to guide any meaningful strategic planning. Information on geographic location, dam height, reservoir area and volume, capacity, and operation mode could foster a baseline inventory without requiring substantial investment. This information can be used to prioritize projects that meet electricity demands while minimizing harm to the environment.

Several research questions need to be addressed in order to understand the full impacts of SHPs. First, scientists must determine which factors, in addition to capacity, are able to reliably delineate the environmental impacts of hydropower. Second, given the recognized importance of headwater streams for watershed integrity (Meyer et al., 2007), strategic plans must identify the species and ecological processes likely to be at risk from SHP development. Third, the extreme density of SHPs in some areas highlights the need to more rigorously quantify their cumulative impacts (Kibler & Tullos, 2013). By addressing these knowledge gaps, the role of science in informing energy policy and watershed management will be strengthened.

SHPs and LHPs are not competing components of the hydropower sector. SHPs are broadly located in different parts of a basin (smaller rivers), financed by different entities (private investment), regulated by different policies, and respond to different energy demands (supplying out-of-grid consumers) as compared to LHPs. Consequently, in environmental policy making, there are political and social motivations to retain the SHP versus LHP classification. For instance, capacity classifications play a role in regulating the energy market and prices of electricity. By contrast, we argue for the removal of the “small” and “large” modifiers describing hydropower plants when conducting environmental assessments and mandating potential mitigation. A hydropower project artificially labeled as “small” should not automatically lead to a faster and less comprehensive licensing process.

1.4 CONCLUSIONS

Electricity is a basic human need in modern societies, and SHPs have a key role to play in supporting electricity policies that mandate the use of renewable resources. We have described the widespread proliferation of SHPs across the world, and highlighted the potential for continued growth of this sector into the future. The process by which hydropower plants are classified as “small” is based exclusively on the measure of generation capacity, which is arbitrarily chosen and inconsistently applied across countries. Additionally, scientific evidence indicates that the environmental impacts of SHPs are substantial, but existing policies and regulations appear to underestimate these impacts. We argue for stricter and more informed regulations, integrated watershed management plans that consider SHPs in aggregate, and readily available information systems to support strategic and impartial planning for new construction. Until such regulatory mechanisms are in place, it would be wise to reconsider the current pace of SHP expansion globally.

1.1 FIGURES

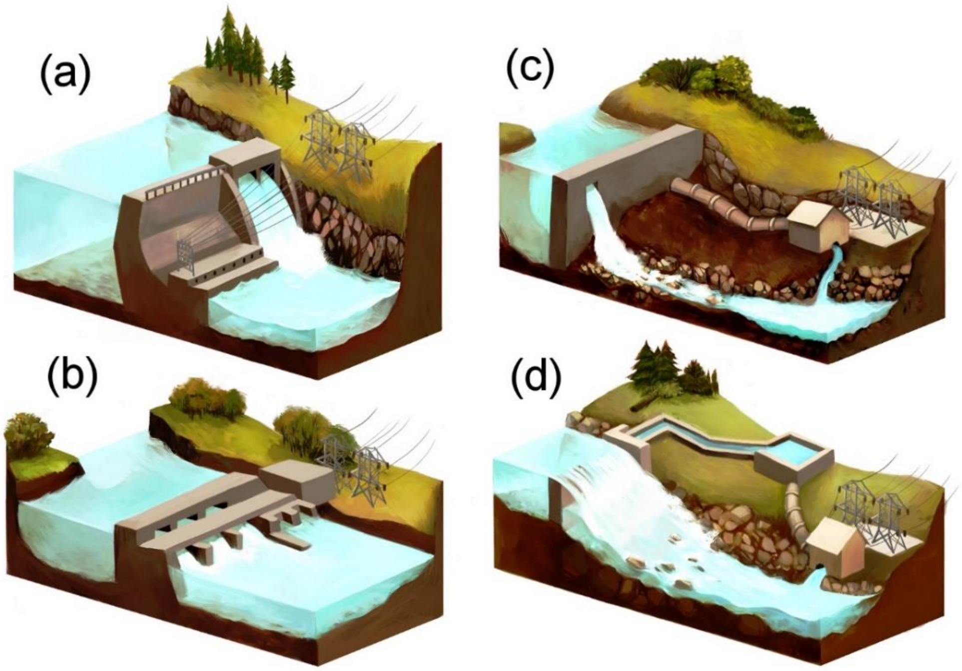

Figure 1.1. Schematic depicting the primary classification of operation modes of hydropower according to the presence of storage and diversion structures. Any of these operation modes can be found in installations classified as SHPs. Non-diversion with storage (a) and without storage (b). Diversion with storage (c) and without storage (d).

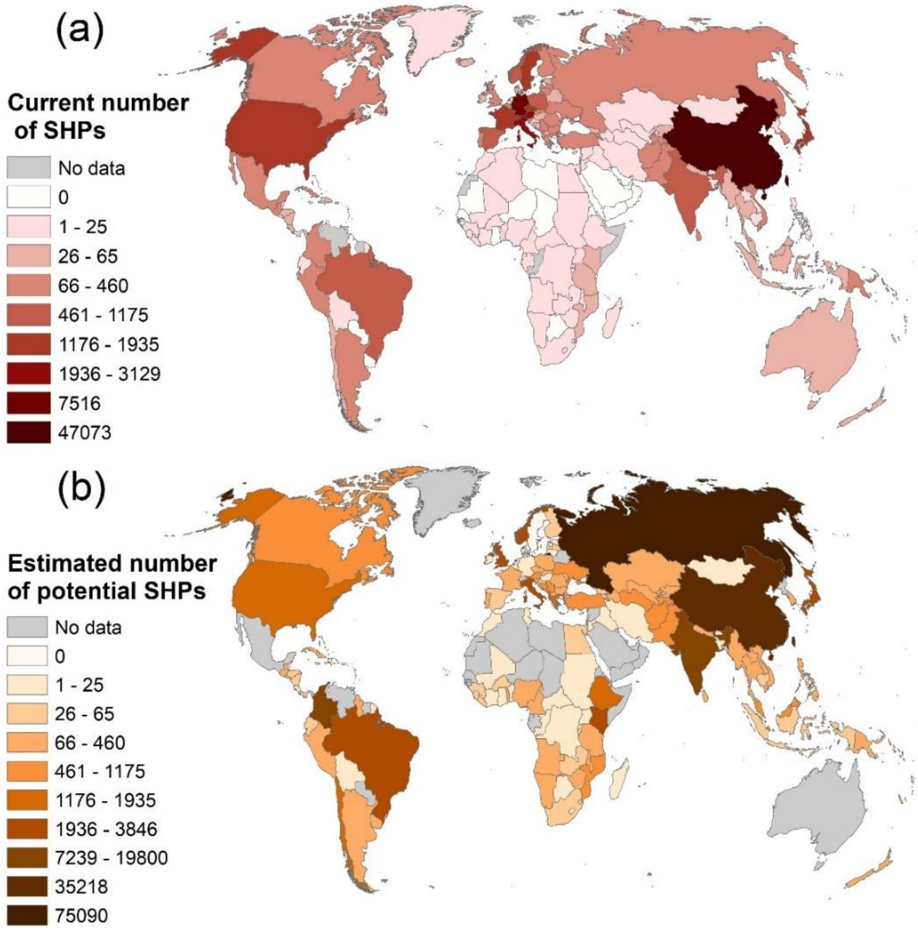

Figure 1.2. Map depicting (a) the current number of SHPs (82,891) operating or under construction across 150 countries and (b) the estimated number of potential SHPs (181,976), of which at least 10,569 are included in national plans to be implemented in the coming decades.

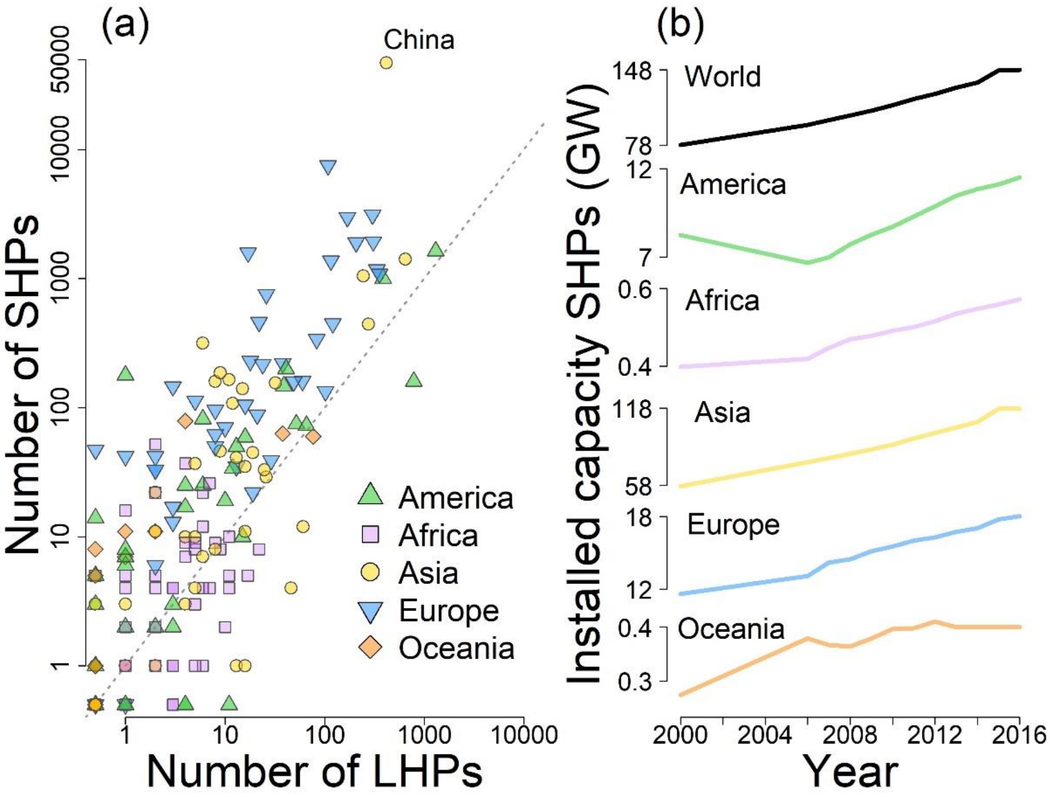

Figure 1.3. (a) Comparison of the number of SHPs and LHPs per country. Continents are represented by different colors and symbols. (b) Trends in installed capacity of SHPs (<10 MW) by continent from 2000 to 2016.

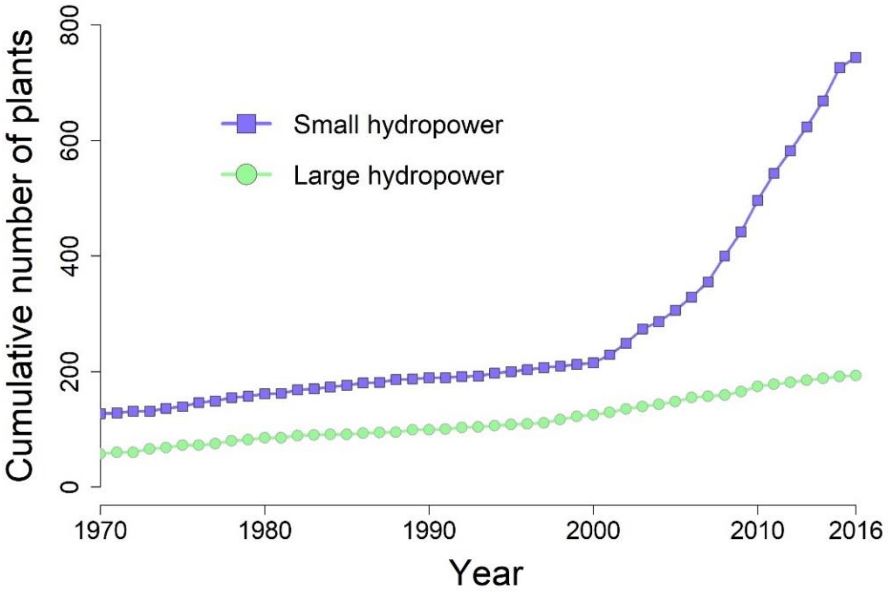

Figure 1.4. Number of Brazilian hydropower facilities in operation since 1970. Although LHPs have demonstrated a constant construction rate for decades, a rapid proliferation in the construction of SHPs commenced in the early 2000s following new regulations and incentives. This plot includes 937 hydropower facilities (out of 1,227 total) reporting the first date of operation (ANEEL 2016).

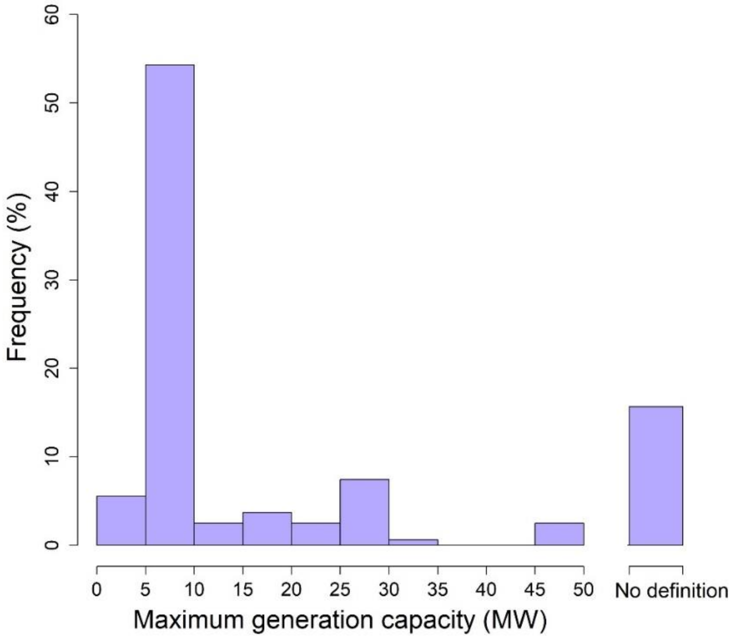

Figure 1.5. Frequency of upper limits of generation capacity used to define SHPs in 150 countries and territories (22 countries lack formal definitions).

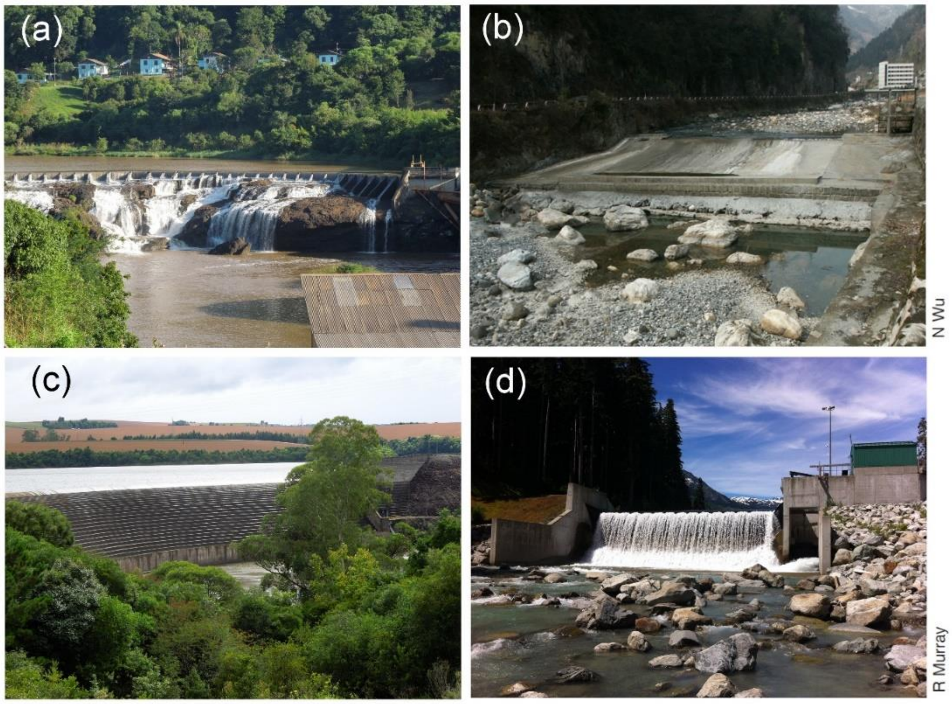

Figure 1.6. Examples of SHPs in (a) Brazil (CGH Abrasa, 1.5 MW), (b) China (Cangpinghe, 9.1 MW), (c) Brazil (PCH Ludesa, 30 MW), and (d) Canada (Rutherford, 49 MW).

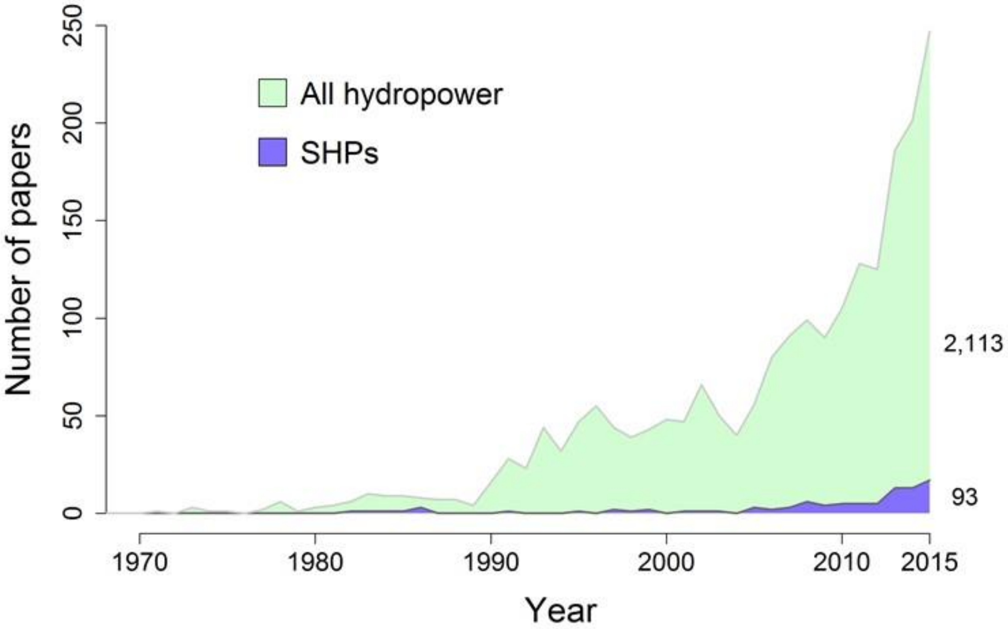

Figure 1.7. Number of scientific publications per year addressing the ecological consequences of SHPs compared to all forms of hydropower (only data for papers published up to, and including, 2015 were plotted).

1.2 TABLES

Table 1. 1. Examples of ecological effects of SHPs on freshwater biodiversity, organized according to flow control (storage versus non-storage) and presence of a diverting structure (diversion versus non-diversion).

Flow control	**Structure**	**Capacity (MW)**	**Ecological Response**	**Country**	**Reference**
Storage	Diversion	18	• Fish species composition modified in response to fragmentation by damming and dewatering. Opportunistic life-history strategists dominate over species exhibiting more complex reproductive requirements.	Costa Rica	Anderson *et al.* 2006
		< 10	• Downstream decrease in macroinvertebrates density and richness, with the replacement of high flow-adapted species by low-flow, generalist species.	Portugal	Jesus *et al.* 2004
		< 1	• Periphytic chlorophyll and biomass showed little difference between diversion canals and mainstream, suggesting minimum effect on nutrient uptake efficiency despite alterations to hydrology.	Spain	Izagirre *et al.* 2013
	Non-diversion	< 1	• Fish condition in reservoirs decreases with compromised integrity of nearshore vegetation.	Brazil	Cop-Ferreira *et al.* 2015
		Not reported	• The diversity of macroinvertebrate communities is reduced by altered downstream flow caused by hydropeaking dam operations.	Portugal	Cortes *et al.* 1998
	Not reported	Not reported	• Fish community composition modified by releases of warm water, with cold water species having decreased abundance downstream. Recovery of benthic invertebrate communities after one year of dam removal.	USA	Hayes *et al.* 2008
Non-storage and storage	Diversion	< 10	• Fish species composition and richness did not vary above and below the diversion, but discrepancies in body size structure were detected.	Portugal	Santos *et al.* 2006
Non-storage	Diversion	< 1.2	• Decrease in brown trout density and biomass. Trout density is inversely proportional to the amount of water diverted.	France	Demars 1985
		< 10	• Fish communities dominated by small-bodied species in dewatered sections of rivers. Average individual weight and total biomass of fish decreased four times in the reduced discharge sections, where biomass losses were proportional to the degree of abstraction.	Czech Republic	Kubecka *et al.* 1997
		< 5	• Slight decrease in fish species richness after the construction of SHPs, but no changes detected for counts of Atlantic salmon and brown trout.	UK	Bilotta *et al.* 2016

		< 1	• Decrease in population densities and biomass of brown trout, and change in age structure.	Spain	Almodóvar and Nicola 1999
		< 1	• Benthic macroinvertebrate communities showed modest downstream responses, where high flow-adapted mayfly and stonefly species exhibited reduced abundance in dewatered stream sections.	UK	Copeman 1997
		Not reported	• Microinvertebrate communities tend to be less dominated by riverine species, resulting in differences on density, species richness and community composition.	China	Zhou *et al.* 2008
		Not reported	• Benthic algae communities modified after SHP construction. Modification in algae communities and chlorophyll concentrations tend to be more pronounced in the dry season in response to a higher proportion of water abstraction.	China	Wu *et al.* 2009a, b, 2010
	Not reported	Not reported	• Macroinvertebrate communities modified, with the dominance of turbificid worms, hemoglobin-rich chironomids and burrowing mayflies over net-spinning caddisflies and heptageniid mayflies.	USA	Stanley *et al.* 2002
Not reported	Diversion	Not reported	• Fish community composition modified, especially in localities that suffer direct effects of diversions. Decrease in the abundance of an endangered perch-like species and other non-native salmonids.	Chile	Habit *et al.* 2007
	Not reported	Not reported	• Caddisfly assemblages modified, which is at least in part a result of hydropeaking dam operations.	Spain	Álvarez-Troncoso *et al.* 2015

Chapter 2. TRADE-OFFS BETWEEN SMALL HYDROPOWER DEVELOPMENT AND RIVER CONNECTIVITY REVEAL OPPORTUNITIES TO SAFEGUARD BRAZILIAN INLAND FISHERIES

The quest to increase the share of renewables on energy grids is fueling a global proliferation of Small Hydropower Plants (SHPs) – an emerging threat for freshwater ecosystems and the services they provide. Being both less regulated and considerably more abundant than Large Hydropower Plants (LHPs), SHPs pose a unique management challenge as they may cause cumulatively greater habitat fragmentation for valuable migratory fish. We quantify current and projected-future river fragmentation by thousands of dams across Brazil and assess the vulnerability of 191 migratory fish species to hydropower development. We further identify planned dam portfolios that maximize energy gains while minimizing river connectivity losses through a multi-objective optimization. We found that SHPs are the main source of fragmentation, occasioning average connectivity losses fourfold greater than LHPs. This imbalance will grow in the future with fragmentation by SHPs projected to increase by 21%, primarily driven by their expansion to currently undammed basins. Two-thirds of migratory species occupy basins that will be more fragmented in the future by SHPs than LHPs, including 16 red-listed and 13 economically important species. Optimal portfolios halve the number of both SHPs and LHPs compared to least-optimal, despite delivering the same total generation capacity. This amounts to hundreds of less dams being constructed, substantially lowering river fragmentation, and protecting numerous undammed basins. This study highlights the need for regulations and strategic planning that look

beyond individual projects to consider the cumulative effects of SHPs, an issue of high socioecological importance still overlooked by environmental and energy policies.

2.1 INTRODUCTION

Rapid expansion of hydropower dams threaten many remaining free-flowing rivers in biodiverse tropical regions of the world, interrupting the migrations of freshwater fishes on which millions of people rely directly for their livelihoods (G. Grill et al., 2019; Winemiller et al., 2016; Zarfl et al., 2014). Consequently, there is a growing need to carefully balance trade-offs between hydropower production and ecosystem functioning to ensure long-term regional food security and income (McIntyre, Reidy Liermann, & Revenga, 2016; Sabo et al., 2017). Considerable scientific activity and international scrutiny remain focused on Large Hydropower Plants (LHP), whereas the environmental repercussions of smaller projects have been largely overlooked in broad-scale energy policies and garnered little public attention. Despite, and perhaps due to this oversight, a series of political and economic incentives to renewable energies have been implemented across the world to benefit projects labeled as Small Hydropower Plants (SHPs) (Couto & Olden, 2018). These incentives assume that, in addition to alluding to the limited energy generation capacity of SHPs, “small” also equates to low environmental impacts. This has ultimately led to licensing exemptions, favored energy prices (e.g. feed-in tariffs), and subsidized loans that fuel increasing construction (Couto & Olden, 2018; Kelly-Richards, Silber-Coats, Crootof, Tecklin, & Bauer, 2017; Premalatha et al., 2014). However, emerging scientific evidence suggest that the ecological impacts of SHPs can be disproportionally high when compared to their societal benefits (Gleick, 1992; Kibler & Tullos, 2013).

The primary concern with rapid SHP development is that their number vastly dwarfs that of LHPs. Today, more than 80,000 SHPs are operating in at least 150 countries (11 times the number of LHPs) and future constructions are expected to be on the order of tens of thousands (Couto & Olden, 2018). Despite their sheer quantity, the combined contribution of SHPs to electrical grids are often quite low. For instance, SHPs represent over 80% of the 1,341 hydropower plants operating in Brazil, yet are responsible for only 5% of total generation capacity (ANEEL, 2019). The imbalance between the number of power plants and their generation capacity calls into question whether the marginal contributions to national-level energy requirements by SHPs are worth their potential environmental and other societal costs. In fact, the cumulative impacts of SHPs on basin-wide hydrology and habitat connectivity may exceed that of LHPs when standardized by hydropower generation (Kibler & Tullos, 2013; Timpe & Kaplan, 2017), thus challenging the vast majority of environmental policies that focus on the impacts of individual, and often large, dams (Athayde et al., 2019; Couto & Olden, 2018).

By virtue of their sheer number and geographic extent, the widespread proliferation of SHPs may be a significant, yet underappreciated, threat to ecosystem services such as the persistence of migratory fishes and the fisheries and diverse societal values they support (E. P. Anderson et al., 2018; Winemiller et al., 2016). Tropical migratory fish species comprise among the top-ranked inland fisheries according to market values and cultural preferences for consumption, but are also among the most vulnerable to hydropower development (Arantes, Fitzgerald, Hoeinghaus, & Winemiller, 2019; Goulding et al., 2019). Dams constrain the movement of migratory fish along river networks and isolate critical habitats necessary for their life history (e.g., spawning and feeding grounds), resulting in population declines, collapses of fishery stocks, and socioeconomic impacts (McIntyre et al., 2015). For instance, the fisheries yield of predominantly large-bodied migratory species decreased by half after the construction of Itaipu Dam (LHP) in Brazil, despite a three-fold increase in fishing effort (Hoeinghaus et al., 2009). Past and planned dam construction greatly exacerbates the loss and fragmentation of habitat in dendritic river networks through the cumulative effects of multiple barriers (Cote, Kehler, Bourne, & Wiersma, 2009; Tonkin et al., 2018) and therefore threatens the overall resilience of fisheries (Brennan et al., 2019).

Revealing optimal siting of new dam constructions is a key first step to balance the trade-offs between hydropower generation and river fragmentation that threatens migratory fish (Tickner et al., 2017), yet it is noticeably absent from many national energy plans. Managers and decision makers have a wide range of options to help guide where on the landscape prospective hydropower projects may be constructed (Jager, Efroymson, Opperman, & Kelly, 2015). Recent attention has focused on the utility of multi-objective optimization approaches to identify win-win management opportunities and provide more prescriptive recommendations — both highly desirable in decision-making (Chen & Olden, 2017). Optimization analyses can evaluate the performance of prospective hydropower plants portfolios by comparing their cumulative ecological impacts and generation capacity gains to achieve different scenarios of future energy demand (R. M. Almeida et al., 2019; Schmitt, Bizzi, Castelletti, & Kondolf, 2018; Ziv, Baran, Nam, Rodriguez-Iturbe, & Levin, 2012). Therefore, they offer an alternative that looks beyond the "more is better" perspective (Couto & Olden, 2018; Jager et al., 2015) to identify portfolios of SHPs and LHPs that maximize generation capacity while minimizing the loss of river connectivity.

The duality of growing hydropower demand for future energy supply and growing socioecological value of migratory fisheries in developing countries necessitates a robust examination of the trade-offs between future SHPs and LHPs development and the resulting fragmentation of rivers. We explore this issue in Brazil, where over 2,200 new hydropower projects are currently in different stages of the licensing and inventory processes. First, we compare the cumulative effects of current and projected-future SHPs and LHPs on network connectivity of Brazilian rivers – cumulative effects defined here as the fragmentation by dams in aggregate. Second, we quantify the resulting range-wide loss of connectivity for 191 migratory fish species, including both those that are nationally red-listed and those highly valuable for commercial or recreational fisheries. Third, we explore trade-offs between gains in energy capacity and loss in river connectivity to inform strategic planning for the development of new hydropower constructions in the country. We adopted the Dendritic Connectivity Index (DCI) as our metric of connectivity (Cote et al., 2009), which is a measurement of the probability of a fish being able to freely move between two random points in a given river network. Empirical research has shown that DCI can accurately describe the effects of river fragmentation and dispersal limitation on the spatial distribution of fish species (Jaeger, Olden, & Pelland, 2014; Perkin & Gido, 2012), and it has been employed in different parts of the world to assess the impacts of future fragmentation by dams (E. P. Anderson et al., 2018; Günther Grill, Ouellet Dallaire, Fluet Chouinard, Sindorf, & Lehner, 2014). We computed DCI for all Brazilian basins according to the present-day (2018) and projected-future (circa 2050) distribution of dams, contrasting the respective contributions of SHP vs. LHP over time and space. Our results compose a national-level assessment of present and future fragmentation by hydropower that informs future energy and environmental policies through a multi-objective optimization approach.

2.2 METHODS

2.2.1 *Dam distributions and attributes*

Our analysis combines hydrographic data with the spatial location of hydropower dams to identify fragmented river networks and quantify riverine connectivity across Brazil. We used the HydroSHEDS and HydroBASINS global hydrographic mapping products (B. Lehner, Verdin, & Jarvis, 2008; Bernhard Lehner & Grill, 2013), which contain 1.1 x 10^6 river reaches draining in and to Brazil, and polygons delimiting river basins at ten spatial levels of organization, respectively. Next we retrieved a comprehensive national dataset on hydropower dams from the repository of the Brazilian energy agency ANEEL (ANEEL, 2018). This dataset contains 3,795 dams with associated attributes that include size category (i.e. large, small, mini), generation capacity, opening date, and status (i.e. operating, under construction, decommissioned, inventory or licensing stage).

River connectivity in this study is an attribute of a river basin (Cote et al., 2009). A “river reach” is defined as a cartographic unit, represented by the line segment between two neighboring confluences. A ‘river fragment’ is a subset of a basin’s network (i.e. a set of fully connected reaches) that becomes disconnected from the rest of the network after the construction of a barrier (dam). We conducted our analysis for level eight-basins (containing an average of 314 km of rivers per basin), which corresponds to the scale of hydropower planning and natural resource management that accounts for the scales of migration of most freshwater fish species in Brazil (Carolsfield, Harvey, Ross, & Baer, 2003). River networks in coastal basins with two or more disconnected networks flowing directly to the ocean were analyzed separately (as “sub-basins”) to ensure fully connected networks.

Brazilian regulations classify hydropower dams as “small” when they produce < 30 MW and impound reservoirs with surface areas < 13 km^2, and as “mini” when generation capacity is < 1 MW; all other dams are labeled as “large” (Couto & Olden, 2018). These classifications are arbitrary choices not supported by scientific evidence – both small and mini dams receive licensing exemptions. Therefore, we combined them in a single category of SHPs. We assumed that all dams in operation in 2018 represent the current scenario, and dams under construction or in different stages of licensing or inventory (as of September 2019) represent the projected-future scenario. The future scenario reflects a time period from present-day to mid-century (circa 2050) that may change depending on future policies, governance, and other local factors (Empresa de Pesquisa Energética, 2018).

A series of spatial editing steps were performed on the hydropower dam dataset to ensure that it was accurately linked to the river network. This included the deletion of duplicate spatial records (hydropower plants within 300 m of each other) and the automatic snapping of dams within 100 m of the network to the nearest river reach. Dams beyond 100 m of the river network were manually repositioned based on visual reference to high-resolution imagery and to the Brazilian national cartographic dataset from IBGE (IBGE, 2019). Dams were discarded from the analysis in the rare occasion when they appeared to be disconnected from the main river network. Lastly, we also ensured the positional accuracy of all LHPs and inspected dams whose power generation capacity appeared high compared to the annual discharge of the river reach in which they were located. Following these steps, our analysis included 3,061 SHPs (1,032 current and 2,029 future) and 451 LHPs (214 current and 237 future).

2.2.2 *Migratory fish species*

We classified the 3,130 species of freshwater fish that are formally registered in Brazil as either migratory or non-migratory based on whether they demonstrate some level of longitudinal migrations in rivers to access feeding areas or to complete reproduction (i.e. upstream and downstream migrations along the main channel). Species whose migrations are restricted to lateral movements into floodplains were excluded (Carolsfield et al., 2003). Unfortunately, basic knowledge on natural history and migratory behavior is lacking for most fish species (Reis et al., 2016). Therefore, we compiled a list of taxonomic groups (Family and Genus levels) reported as longitudinally migratory in South America (Carolsfield et al., 2003), and extrapolated our definition of potentially migratory to individual species that belong to these groups, resulting in an initial list of 505 migratory species. This strategy is supported by strong associations between dispersal ability and morphological characteristics (e.g. body size and fecundity), and phylogenetic conservatism in parent-offspring dispersal distance reported in the literature (Comte & Olden, 2018a, 2018b). This initial list was subsequently reviewed by two expert ichthyologists in Brazil (Jansen Zuanon and Roberto Reis, personal communication), resulting in a final list of 365 species that are highly likely to be migratory (Appendix B, Table S1). Of these species, 24 are red-listed as Vulnerable, Threatened, Endangered or Critically Endangered according to national standards using IUCN criteria (ICMBio, 2018), and 24 are considered species of highest commercial and recreational value – species with annual yields of more than 10 tons and/or considered an important sport fisheries in Brazil (Freire, Machado, & Crepaldi, 2012; Goulding et al., 2019; MPA, 2011). Point datasets of occurrence records for 335 species were acquired from the environmental agency ICMBio (ICMBio, 2018). Species occurrence data were then intersected with all river basins that have or will have at least one hydropower dam, resulting in a basin-level distribution range for 191 migratory fish species.

2.2.3 *Quantifying river connectivity*

The Dendritic Connectivity Index (DCI) is a metric of river connectivity that reflects the probability that a mobile organism (fish) can move between two randomly selected points from a network (Cote et al., 2009). DCI ranges from 100 (i.e. completely free-flowing basin with no barriers) to zero and can be calculated for any size stream network. The DCI equations can be adjusted according to the movement behavior of the fish species (Cote et al., 2009). Here, we adopted the DCI_p developed for potamodromous fish — the guild of migratory fishes that complete their life cycle, and thus migratory movement, exclusively in freshwater habitats (Brönmark et al., 2013) — here assumed to be equally likely to move upstream as downstream. More specifically, DCI_p expresses the stream length-weighted average connectivity among all fragment pairs within the network, where each barrier is assigned a permeability value that represents the probability of a given fish passing through the barrier and connectivity is the joint probability that a fish can move between two fragments given the permeability of all barriers between the fragments. We adopted 10% as the permeability for all dams, which was informed by research reporting very low efficiencies of fish passage structures for non-salmonid fishes and for South American rivers, ranging between 0.7 and 21% (Noonan, Grant, & Jackson, 2012; Pompeu, Agostinho, & Pelicice, 2012). We attributed the same permeability to SHPs and LHPs due to the lack of evidence relating the performance of fish passage structures and size-categories of hydropower (José Maria Santos et al., 2012). Although smaller dams may seem easier to pass over or bypass compared to LHPs, there are three main reasons why this is unlikely. First, size classifications based on hydropower generation capacity do not necessarily reflect ecologically relevant attributes (e.g. dam height and flow regulation) that constrain fish movement (Angelo Antônio Agostinho et al., 2007; Couto & Olden, 2018; Januchowski-Hartley, Jézéquel, & Tedesco, 2019). Second, a high proportion of fish passages associated to SHPs have been reported to have critical structural and maintenance problems that completely eliminate their functionality (José Maria Santos et al., 2012). Third, SHPs are subjected to less rigorous requirements for environmental mitigation compared to LHPs, which means that the presence of fish passages and their monitoring and regular maintenance are also less likely (Couto & Olden, 2018). We did not include potential natural barriers to fish migration like waterfalls due to the lack of a comprehensive national dataset and to the high diversity of migratory fish in the analysis – species respond differently to different sorts of barriers (Torrente-Vilara, Zuanon, Leprieur, Oberdorff, & Tedesco, 2011). DCI analysis was conducted for scenarios containing SHPs alone, LHPs alone, and both dam types combined, in each of two time periods (current and future). Trends in DCI over the past century were also examined according to the opening dates of 1,211 (97%) dams from our dataset. We opted to separately explore scenarios of the existence of just SHPs or just LHPs because the contribution of each dam barrier to the basin-level DCI is non-additive. We only report DCI results (e.g. nationwide average) for the 1,216 Brazilian basins that have or will have at least one hydropower dam in the future.

2.2.4 *Selecting hydropower dam portfolios*

We estimated the unique contribution of each planned dam on basin-level DCI using a permutation analysis that calculated the difference between DCIs when a given dam is present and absent in each basin, repeated for all possible combinations of future 2,029 SHPs and 237 LHPs across the country. The overall contribution of each planned dam was summarized as the mean and the range of DCI loss when present in the basin. Next, we examined 70,000 future dam portfolios with respect to total nation-wide generation capacity and average basin-level DCI. Each portfolio represents a random selection of a future number of dams that were sampled from the pool of 2,266 planned dams. Analyzing all possible portfolios (all permutations of dam sites in each basin) would necessitate a massive computational effort, as the number of all portfolios equals 1.4 x 10^{682}, calculated as 2^n, where n is the number of planned dams. We determined the set of optimal and least-optimal dam portfolios (i.e., the Pareto frontier) that best and worst traded-off gains in hydropower generation capacity with losses in nation-wide river connectivity. Although we did not account for connection access to the national grid, construction and maintenance costs, and other economic factors, it is important to note that all dams are either under construction or in different stages of licensing or scoping, and therefore are considered highly feasible by the Brazilian energy agency. We computed the Pareto frontier for future optimal and least-optimal scenarios using the function ‘psel’ of the R package ‘rpref’, which is based on the top-k skyline operator and maximum or minimum values of an objective as a base preference (Roocks, 2016). Future demands for hydropower generation capacity was defined as 120-141 GW from 2030 to 2040 according to an international report that estimates energy policy scenarios (International Energy Agency, 2018).

2.3 RESULTS

2.3.1 *The role of SHPs on riverine fragmentation*

The construction of hydropower dams has resulted in the widespread fragmentation of Brazilian river basins over the past century, with an average basin-level loss in river connectivity (DCI) from 100 to 83, representing a decrease of 17 units (DCI of 100 indicates a completely free-flowing basin with no artificial barriers) (Fig. 2.1). SHPs alone contributed to an average decrease of 14 ± 21 DCI units, four times more than that observed for the LHP-only scenario (3 ± 11 DCI units). The decrease in river connectivity associated with historical SHP construction mirrors the overall trends, indicating a disproportional contribution of SHPs to river fragmentation over the last century. Fragmentation rates remained relatively steady through time until a marked acceleration in recent decades (Fig. 2.1). Whereas fragmentation rates due to LHPs remained constant, the loss of connectivity caused by SHPs increased by seven-fold after the year 2000 – fueled by numerous governmental incentives (Couto & Olden, 2018).

The 2,266 new hydropower projects currently in different stages of construction and potential approval will compromise future river connectivity throughout Brazil, primarily driven by new SHP constructions. Overall, future dam construction is expected to cause a 24% average loss in river connectivity (from a present-day average DCI of 83 to 62 projected by circa 2050), with decreases in individual basins ranging from 0 to 89% (Fig. 2.2a). This predicted loss is caused principally by anticipated SHP construction (average DCI decrease of 21%) compared to LHPs (4%). River fragmentation by hydropower has been historically more concentrated in basins located in the southeastern and southern portions of the country – in the Paraná, Uruguay and East Atlantic drainages (Fig. 2.3). Fragmentation by SHPs will continue to increase in these drainages, as well as expand in the Central-West and North regions over time.

The growing footprint of riverine fragmentation is predominantly the result of the expansion of planned SHP constructions, which are considerably more numerous and widespread compared to LHPs (Fig. 2.3). Consequently, 573 basins that are currently free of hydropower will have new SHPs in the future, representing the primary driver for future nation-wide river connectivity loss. A closer examination of basins that are currently free of hydropower reveals that river connectivity will decrease by over one-third (36% ± 22) in response to future dam construction – an outcome again driven by SHPs (Fig. 2.2b). Decreased connectivity of basins already impacted by hydropower is also predicted (Fig. 2.2c).

Further river fragmentation caused by planned SHPs is also expected across the geographic range of migratory fish, including species that are at-risk of extinction and those that support highly valuable commercial or recreational fisheries. The respective contributions of future SHPs and LHPs to river fragmentation in the ranges of migratory species are relatively similar in present-day (Fig. 2.4a), but the relative fragmenting effects of SHPs are expected to rise. Close to two-thirds (65%) of all migratory species occupy basins that will be fragmented more by SHP compared to LHP construction in the future (Fig. 2.4b). The detrimental effects of fragmentation by planned SHPs are manifested in the geographic range of 16 red-listed species and 13 species of highest value for fisheries (both out of 24 species), resulting in an expected DCI decrease of more than 40% for the majority of these species. For instance, a 60% decrease in river connectivity by SHPs is expected in the geographic range of *Brycon vermelha*, a red-listed species that is endemic to the upper portions of the rivers Mucuri and São Mateus (ICMBio, 2018). In addition, decreases by SHPs over 20% are expected in basins occupied by 12 species of high value, including *Prochilodus lineatus*, *Megaleporinus obtusidens*, *Pseudoplatystoma corruscans*, and *Salminus brasiliensis* – top-ranked in market prices, recreational importance and yields in the Paraná, Paraguay and Uruguay drainages (Carolsfield et al., 2003; Hoeinghaus et al., 2009). Results for all 191 species are found in SI Dataset S1. We also repeated the entire analysis with a modified measure of connectivity that accounts for situations where migratory fish populations are predominantly composed of external immigrants from downstream basins. All results were similar to those presented above – aside from greater loss of river connectivity due to both planned LHPs and SHPs construction (Appendix C, Fig. S1-S6).

2.3.2 *Meeting energy demands while safeguarding river connectivity*

Although it is implicit in regulations that generation capacity is an indicator of the potential environmental impacts of a hydropower plant, there is little association between generation capacity of individual future hydropower projects and their predicted effect on river connectivity (Fig. 2.5). This is true for all hydropower plants combined (Pearson R = 0.22), just SHPs (R = 0.17), and just LHPs (R = -0.01). Projected effect of each future hydropower project on river connectivity is provided in SI Dataset S2. We simulated 70,000 future scenarios to identify the portfolios of planned hydropower projects that optimize generation capacity gains while balancing nation-wide reductions in river connectivity. Although river connectivity and gains in generation capacity are inversely proportional, there is considerable variability in this relationship among dam portfolios (Fig. 2.6a). The Pareto frontier operator identified 198 optimal and 197 least-optimal dam portfolios (the remaining being considered sub-optimal), of which 30 and 32, respectively, fall inside the range of projected hydropower demands for the 2030-2040 period. A closer examination of this demand window reveals that least-optimal portfolios generally involve the excess construction of hundreds of SHPs to achieve the same total energy production (Fig. 2.6b). Optimal portfolios have on average 509 SHPs (± 215) and 66 LHPs (± 28), whereas least-optimal portfolios have twice these numbers, averaging 1,129 SHPs (± 256) and 121 LHPs (± 30). In addition, the number of basins that remain free of hydropower in optimal portfolios is two-fold larger than in least-optimal ones (Fig. 2.6c); on average, 228 (± 76) and 394 (± 61) basins will no longer be free of hydropower in optimal and least-optimal portfolios, respectively. As an example, new input of 24 gigawatts can optimally be achieved with 57% less SHPs (682), 45% less LHPs (51) and by damming 40% less free river basins (163), ultimately resulting in a nearly two-fold lower reduction in riverine connectivity ($DCI_{present\text{-}day}$ = 83, $DCI_{optimal}$ = 75, $DCI_{least\text{-}optimal}$ = 68) when compared to a least-optimal portfolio. These results highlight significant scope for optimal siting of planned new dams in coming decades, and support the importance of strategic planning and trade-off analysis to avoid excessive SHP construction that leads to needless loss in river connectivity.

2.4 DISCUSSION

Massive nation-wide investments in SHPs have caused substantial disruption of river connectivity in recent decades, threatening the persistence of migratory fish species upon which ecosystems and humans depend. Losses in connectivity are predominantly driven by SHPs, exceeding the impacts of LHPs in most basins across Brazil. This finding adds fuel to the growing notion that SHPs in high densities can collectively outweigh the effects of LHPs on river connectivity (Kibler & Tullos, 2013) despite the usual spatial centrality of larger dams in river networks (Couto & Olden, 2018). As conflict over water resources increases under growing population and energy demands, ecologically-sustainable hydropower development in developing countries will necessitate that rivers be managed for multiple co-benefits.

River fragmentation is on the rise in Brazil and has accelerated in the early 2000s, when construction rates of SHPs increased 13 times in comparison to the previous decade (Couto & Olden, 2018). This boom in SHP construction followed a series of policy incentives that included new regulations for the energy market, a relaxation of size-classification criteria, simplified environmental licensing, and economic encouragements such as feed-in tariffs and subsidized loans by public banks (Couto & Olden, 2018; Ferreira et al., 2016). For example, the first feed-in tariff policy in the country was implemented in 2004 and resulted in 59 new SHPs in just eight years (Eletrobras, 2019). Our results reveal that although basins located in the Paraná, Uruguay, and East Atlantic drainages have been more heavily fragmented by SHPs, river fragmentation in the northwestern Brazilian Highlands has increased dramatically in recent decades. This includes upper mainstem portions and tributaries of important rivers like Tocantins, Araguaia, Madeira, Tapajós, Xingu, São Francisco, Paraguay and Paraná.

Increased fragmentation of Brazilian rivers is imminent with thousands of SHPs currently undergoing planning, licensing and construction. These dams, as compared to LHPs, are expected to be the predominant source of future habitat fragmentation for two-thirds of the migratory fish species in Brazil, including species of high conservation priority and socioeconomic importance. In the Tapajós River Basin, among the most vulnerable rivers of the Amazon drainage for hydropower development (Latrubesse et al., 2017), growing conflicts and judicial battles have arisen over the proliferation of SHPs in some of its tributaries (Athayde et al., 2019). One of the most controversial cases is in the Juruena River in Mato Grosso, where the implementation of eight SHPs is affecting the fisheries of the Enawene Nawe people – an essential component of their culture, spiritual practices and food security (J. de Almeida, 2014). Here we report that 14 basins of the Tapajós drainage may lose more than 50% of river connectivity due to planned SHP construction (eight of them part of the Juruena Basin), and the projected loss is as high as 85% for Cupari Leste and Buriti basins.

Empirical evidence suggests that changes in the structural connectivity of rivers reduce species richness and affect the composition of fish communities through dispersal limitation (Perkin & Gido, 2012), highlighting the high vulnerability of migratory species to fragmentation (McIntyre et al., 2015). Our estimates indicate that there will be a 50% decrease in river connectivity caused by future SHPs in basins supporting the red-listed species: *Steindachneridion doceanum*, *B. vermelha*, and *B. opalinus*. Moreover, the species *P. lineatus* – one of the most harvested species in the Paraná, Paraguay and Uruguay drainages where fisheries yields in some portions reach 30 thousand tons annually (Baigún, Minotti, & Oldani, 2013; Carolsfield et al., 2003; Hoeinghaus et al., 2009) – occupy river basins expected to experience a 30% connectivity loss from SHPs alone. These examples call for heightened investigation of how rampant SHP development may compromise sustainable fisheries practices and threaten food security in the future.

Modern optimization approaches have provided renewed hope for reducing what have previously been considered inevitable environmental and social conflicts involving hydropower development (Chen & Olden, 2017; Sabo et al., 2017). Our study indicates that informed selection, or portfolios, of new dams can both meet future energy demands of Brazil and concurrently minimize the resulting loss of river connectivity. In fact, through a careful planning process, optimal dam portfolios almost halve the number of both SHPs and LHPs compared to least-optimal dam portfolios, despite delivering the same total energy generation capacity. This amounts to hundreds of less dams being constructed, substantially lowering degradation of river connectivity, and effectively protecting numerous basins that are currently free of hydropower. These results support previous investigations in other parts of the world that demonstrate the strength of optimization analyses in balancing hydropower production and dam-induced effects on floodplain-fisheries, greenhouse gas emissions, and sediment trapping (R. M. Almeida et al., 2019; Schmitt et al., 2018; Ziv et al., 2012). However, significant challenges persist. SHPs are still rarely considered in assessments of the cumulative ecological effects of future hydropower despite their global proliferation (Couto & Olden, 2018). For instance, 40% of the recent cumulative impact assessments for hydropower in Brazilian basins completely ignore SHPs (Athayde et al., 2019).

2.5 CONCLUSIONS

Managing rivers for multiple, sustainable benefits requires integrating scientific, social, and policy perspectives into operational decision frameworks (Poff et al., 2016; Tickner et al., 2017). This study challenges nation-wide policies for SHP development in Brazil that consistently assume their low potential for causing environmental impacts and generally ignore their cumulative ecological effects. We found that the expansion of SHPs poses a persistent and emerging threat to the connectivity of Brazilian rivers, resulting in important negative consequences for inland fisheries and migratory-fish conservation. In this regard, we demonstrate that strategic planning based on multi-objective optimization can provide critical guidance to which dams under different stages of planning should be constructed to meet energy needs while ensuring the smallest loss of river connectivity in the country. Environmentally-informed strategic planning, together with policy incentives to diversify renewable energy sources, are potential avenues to be explored (Opperman et al., 2019), and should minimize the adverse ecological effects of dam construction in tropical rivers.

2.6 FIGURES

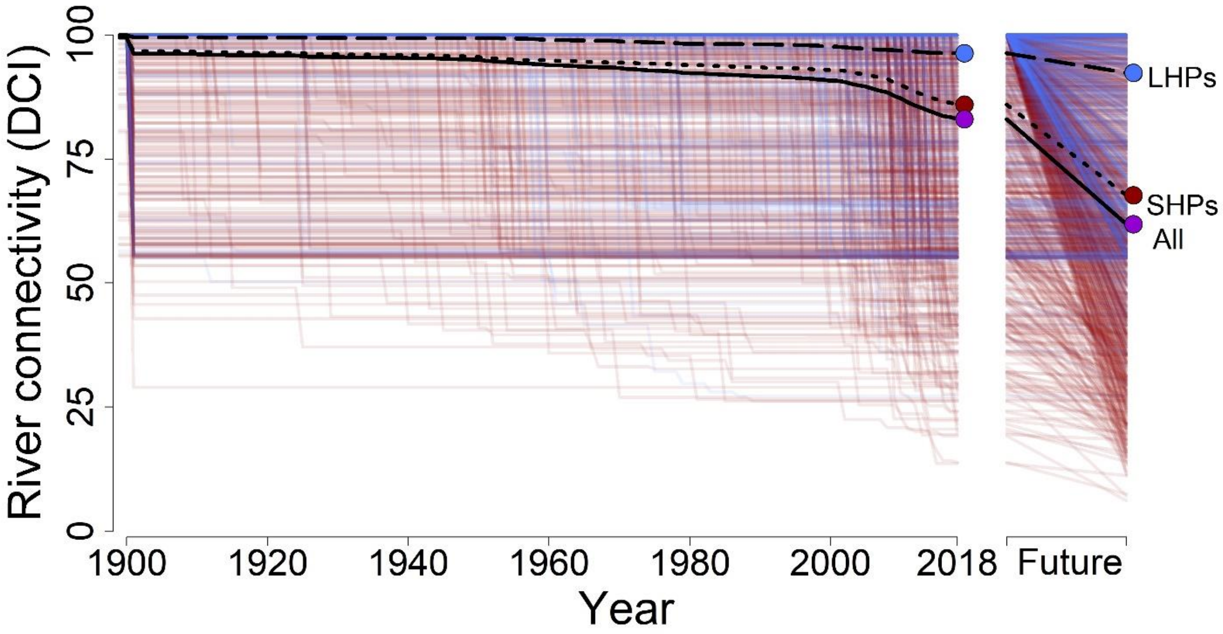

Figure 2.1. Temporal trends in river connectivity in Brazil according to changes in DCI over the past century and future projections due to ongoing and planned dam construction. Colored lines represent yearly estimates of DCI for 1,216 individual basins, with lines indicating the unique contribution of SHPs (red) and LHPs (blue). Black lines represent average values for SHPs (dotted), LHPs (dashed), and all types of hydropower combined (solid). “Future” reflects ongoing and planned construction that is realistically projected to occur in the mid-century (circa 2050); the exact year is subject to change depending on future policies and governance.

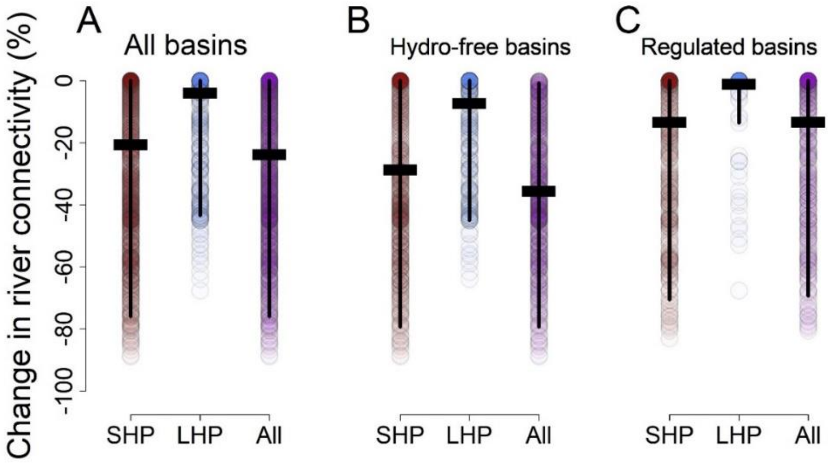

Figure 2.2. Predicted future change (%) in river connectivity (DCI) from present-day (2018) to mid-century (circa 2050) for the Brazilian basins according to the separate contributions of SHPs (red), LHPs (blue), or all types of hydropower combined (purple). Black bars indicate average values and black lines indicate 95% confidence intervals. Results are shown for (A) all 1,216 basins, (B) 573 basins that are currently free of hydropower, and (C) 643 basins currently regulated by hydropower.

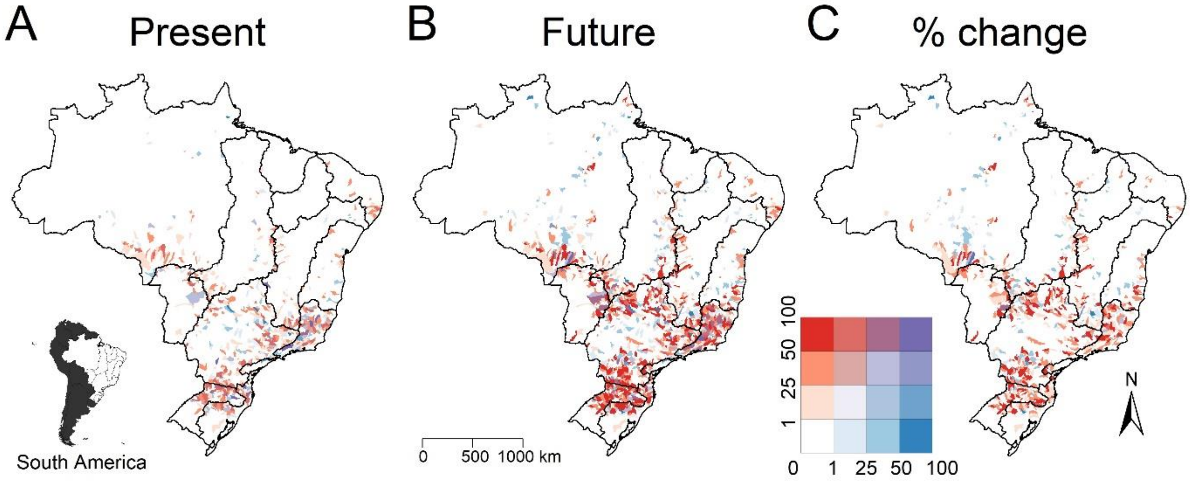

Figure 2.3. (A) Present (2018), (B) future (circa 2050) and (C) projected percentage change in river connectivity (DCI) for the Brazilian basins according to the contributions of SHPs (red shades) and LHPs (blue shades). Color pallets represent (A-B) values of absolute DCI loss from free-flowing state (i.e. 100 - DCI) or (C) percentage loss between periods (negative change in all instances). The black lines delimit the boundaries of the major Brazilian hydrographic units.

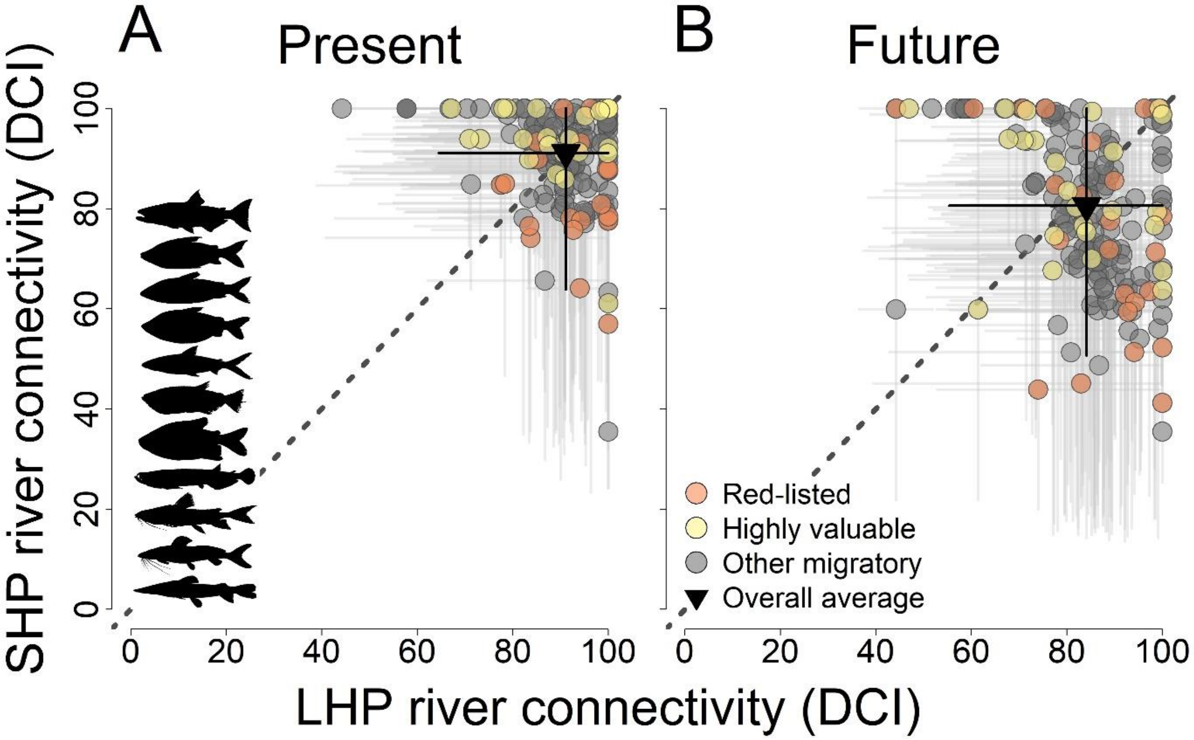

Figure 2.4. Contributions of SHPs and LHPs to river connectivity (DCI) in basins occupied by 191 migratory fish species according to (A) present-day and (B) projected future (mid-century) dam distributions. Average river connectivity across each species' geographic range are represented by circles (95% confidence intervals displayed as light gray lines). Colors differentiate the species that are red-listed (24, orange) or highly valuable for fisheries (24, yellow) from the remaining species (gray). Black triangles and black lines indicate the overall averages with associated 95% confidence intervals. The main taxonomic groups classified as migratory are illustrated in (A) (full species list available at Appendix B, Table S1).

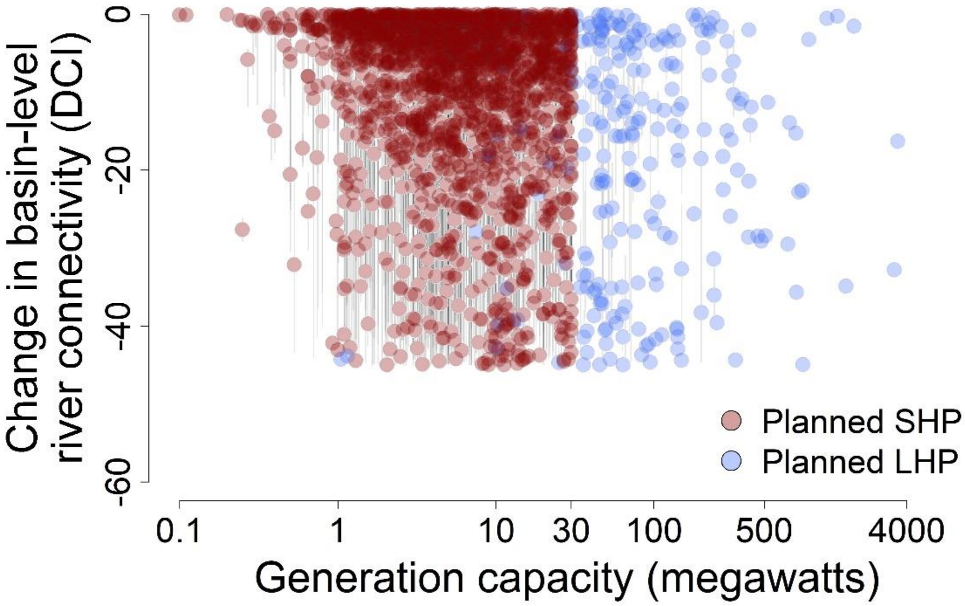

Figure 2.5. Relationship between the generation capacity of each future hydropower project and its effect on river connectivity (DCI) at the basin level. Estimates of average DCI loss (the average difference across all basin-level future scenarios with and without each dam) for planned SHPs (red) and LHPs (blue) are represented by circles, and the gray lines indicate the range. The 24 planned LHPs with future capacity below 30 MW were classified as such by the energy agency ANEEL, following other criteria such as reservoir area.

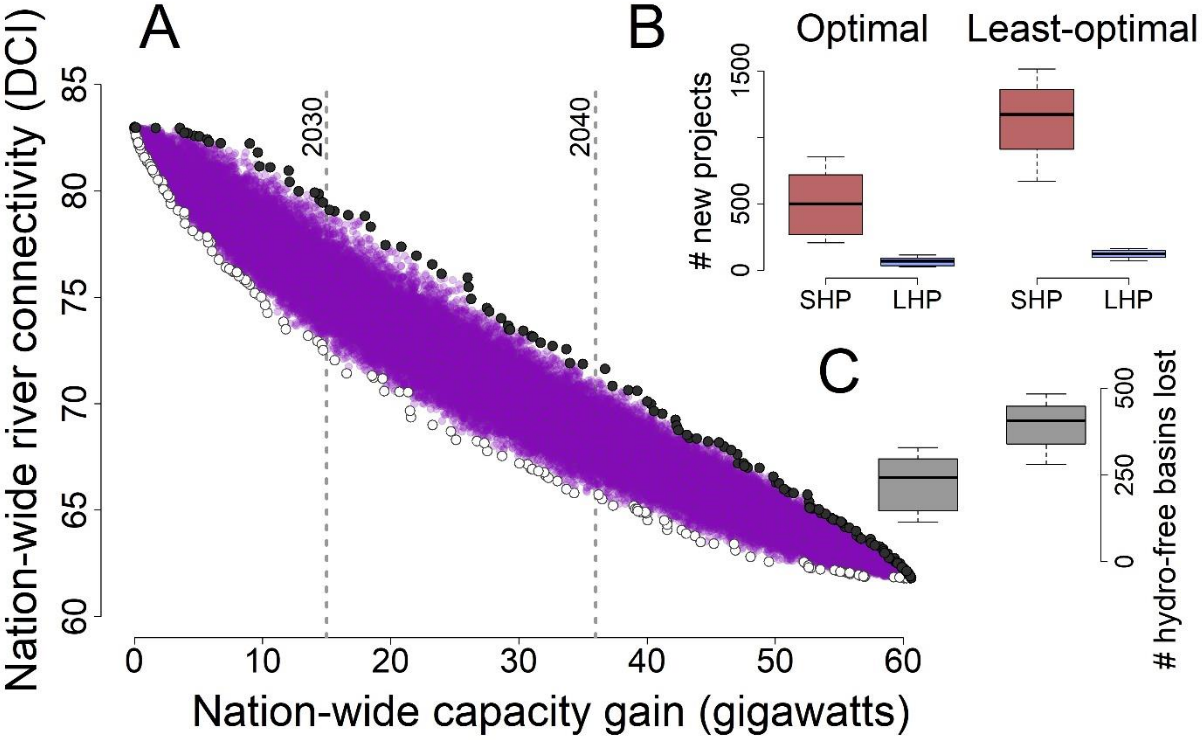

Figure 2.6. (A) Future projections of nation-wide river connectivity (average DCI across all basins) and total additional generation capacity (based on a 2018 baseline) for 70,000 bootstrap samples (circles) from the 2,266 planned dams in Brazil. Vertical dashed lines delineate the projected demands for additional hydropower generation capacity from 2030 to 2040. The colors of the circles indicate the optimal (black), least-optimal (white), and remaining (purple) dam portfolios computed using a Pareto frontier operator. The inset boxplots contrast (B) the number of new SHP and LHP projects to be built and (C) the number of basins that will no longer be free of hydropower in the optimal and least-optimal dam portfolios located within the 2030-2040 demand interval.

Chapter 3. FRESHWATER COMMUNITY RESPONSES TO HABITAT LENTIFICATION BY SMALL HYDROPOWER PLANTS IN SOUTHERN BRAZIL

1. Hydropower dams modify distribution patterns of freshwater organisms due to major changes in habitat characteristics above and below the impoundments. In recent decades, there has been a global proliferation of Small Hydropower Plants (SHPs) propelled by incentive policies and a general perception that small dams are less harmful to the environment. However, limited science is available on the effects of SHPs on freshwater communities, which is concerning considering their sheer numbers and the high heterogeneity of dams being classified as SHP.

2. We investigate the ecological effects of 12 SHPs on macroinvertebrate and fish assemblages of the Chapecó River Basin, Brazil. Our samples included sites located upstream of the SHPs and sites directly affected by their operations (i.e. reservoir, dewatering section and downstream). This design allowed us to estimate site-comparison shifts in assemblage taxonomic composition, and examine the role of different SHP attributes (e.g. dam height, reservoir area) on the magnitude of these shifts.

3. Our results reveal that SHPs can modify the composition of both macroinvertebrate and fish assemblages, but the magnitude of such changes may vary widely among dams. Both assemblages exhibited some level of directionality in taxonomic shifts, resulting in a gradual replacement of taxa associated to fast-flowing habitats by other groups more associated with standing waters. We also found that the magnitude of taxonomic shifts can be predicted by structural and spatial attributes of the SHPs, although the predictive power of the attributes vary between the two assemblages and among the position of sites relative to the dams.

4. Overall, we demonstrate that SHPs have important and quite heterogeneous effects on freshwater communities, which needs to be considered in order to implement effective environmental policies and regulations for the growing SHP sector.

3.1 INTRODUCTION

Accelerating losses of freshwater biodiversity is a global crisis (Tickner et al., 2020), and the rapid expansion of hydropower dams is considered a primary cause (G. Grill et al., 2019; Reid et al., 2019). The conservation challenge imposed by dam development is of particular concern in developing countries, where riverine ecosystems host an overwhelming number of fish and other aquatic species upon which millions of people rely directly for their livelihoods (Winemiller et al., 2016; Zarfl et al., 2014). While research attention often focuses on understanding and mitigating the ecological impacts of large hydropower dams, the vast majority of present-day and planned dams are, in fact, relatively smaller in size; these are collectively classified as Small Hydropower Plant (SHP) (Couto & Olden, 2018). The definition of SHP is ambiguous and quite inconsistent, but it broadly refers to dams with low generation capacity (< 10 megawatts) that tend to be constructed in low- to medium-order rivers. The “small” modifier has been used as a beacon to guide environmental policies that support incentives and licensing exemptions for SHPs, although little evidence exists that “small” necessarily equates to lower environmental impacts (Couto & Olden, 2018; Kelly-Richards et al., 2017; Kibler & Tullos, 2013; Lange et al., 2018). Furthermore, the sheer number of SHPs are far from small, with current and planned future constructions estimated in the order of hundreds of thousands globally (Couto & Olden, 2018).

Flow regulation by dams can impact essential geomorphological, hydrological and ecological processes in rivers like nutrient cycling, primary productivity, sediment transportation, and thermal and hydrologic regimes (J. Olden, 2016; J. D. Olden & Naiman, 2010; Poff et al., 1997). These effects are not restricted to large dams, and mounting evidence suggests that ecosystem effects are also associated with small dams (Hayes et al., 2008; Stanley et al., 2002; N. C. Wu et al., 2010). Environmental modification by SHPs may manifest above and below the impoundment through different mechanisms. First, dams and weirs reduce upstream flows (i.e. backwater effects) and accumulate water in reservoirs of varied sizes in a process termed river lentification (Falasco, Piano, Doretto, Fenoglio, & Bona, 2018; Sabater, 2008). The degree of conversion of flowing rivers to nearly standing-water varies from large storage structures to “run-of-river” schemes that have less storage capacity (David Anderson et al., 2015). Second, diversion SHPs reduce the flow immediately below the dam in long dewatered sections of the river, which run with low flows until they receive the water that was diverted to the powerhouse (Couto & Olden, 2018). The amount of water diverted dictates the degree of flow regulation in dewatering sections, which can be proportionally high for small dams (E. P. Anderson et al., 2006; Kibler, 2017). Third, flow regulation promoted by storage and diversion can reverberate further downstream, stabilizing peak and base flows well downstream of powerhouse outflows (Fencl, Mather, Costigan, & Daniels, 2015; Kibler, 2017; Meier et al., 2003).

Scientific studies have repeatedly demonstrated shifts in the composition of freshwater communities in response to dam-induced flow regulation. Lentification of riverine-habitats generally increase the abundance of generalist species at the expense of fluvial specialist species with more complex life histories (Freeman & Marcinek, 2006; Haxton & Findlay, 2008), and can often facilitate the establishment of invasive species (Angelo A. Agostinho, Gomes, Santos, Ortega, & Pelicice, 2016; Bunn & Arthington, 2002; Mims & Olden, 2013). Shifts in community composition are not restricted to newly created lentic habitats like reservoirs, but also occur downstream of dams. For example, water level fluctuations during peaking operations or water drawdown during periods of low flow may result in exposed or dewatering of the river channel, affecting algae, invertebrates and fish communities below SHPs (Jesus et al., 2004; Jumani et al., 2018; N. C. Wu et al., 2010). Yet, these ecological impacts have been shown to be variable (Haxton & Findlay, 2008; Mbaka & Mwaniki, 2015), and in some instances result in negligible or even positive effects on biodiversity (D Anderson, Moggridge, Shucksmith, & Warren, 2017; Izagirre et al., 2013; Lessard & Hayes, 2003).

Considering the vast heterogeneity of SHPs in terms of dam height, reservoir area, and dewatering distance, additional investigation into the direction and magnitude of potential effects on freshwater biodiversity is warranted (Couto & Olden, 2018; Poff & Hart, 2002). Enhancing this understanding has important repercussions for hydropower policy and regulations due to the central importance of differentiating “small” and “large” during the environmental licensing process (Couto & Olden, 2018; Kelly-Richards et al., 2017). Currently, size classifications of hydropower are arbitrary and primarily based on thresholds of energy generation capacity, thus failing to describe other meaningful descriptors of the likely ecological effects of these structures (Couto & Olden, 2018; Kibler, 2017). Moreover, it is expected that the spatial position and densities of dams play a major role on their ecological effects (Ward & Stanford, 1983), leading to concern that the lack of consideration of cumulative effects is a significant limitation of current SHP policies (Kibler & Tullos, 2013). Greater knowledge of the associations between dam attributes, their spatial positioning, and different dimensions of freshwater biodiversity is a necessary precursor to improve environmental policies and regulations currently being applied to SHPs.

The pace of SHP development in tropical and sub-tropical countries demands new science that informs rapidly evolving environmental policy. Here, we investigate the ecological impacts of SHPs on biological communities of the Chapecó River Basin (Brazil), a region where the SHP sector has been expanding fast in recent decades with dozens of SHPs operating or under way. First, we examine and compare how fish and benthic macroinvertebrate assemblages respond to lentification by SHPs via the mechanisms of water storage, dewatering, and downstream regulation, and test for consistent directionality in changes in taxonomic composition. Second, we explore whether structural attributes of each SHP and their longitudinal position in the watershed are predictive of observed shifts in assemblage composition. Overall, the results of this study reveal general patterns associated to the impacts of SHPs on sub-tropical freshwater communities, supporting policy changes and mitigation actions that target the growing SHP sector.

3.2 METHODS

3.2.1 *Study area*

This study was conducted in the mainstems of the Chapecó and Chapecozinho Rivers in the upper portions of the Chapecó River Basin, Brazil (Fig. 3.1a). The Basin is part of the Uruguay River System, and drains an area of 8,302 km^2 that correspond to a monthly average of 238 m^3/s of discharge (Estado de Santa Catarina, 2017). The Basin is part of an extensive basaltic flood province known as the Serra Geral formation, characterized by a combination of basaltic plateaus and acidic volcanic rocks in a highly dissected terrain (Crósta, Kazzuo-Vieira, Pitarello, Koeberl, & Kenkmann, 2012; MMA, 2010). The vegetation is classified as Araucaria moist forest, a critically threatened ecoregion of the Atlantic Forest Biome (WWF, 2020). The land use can be quite intense in the Basin, primarily driven by forestry activities and cattle ranching. However, riparian zones in the study area are fairly well preserved, especially in the surrounding areas of the Araucárias National Park – created in 2005 as an effort to protect the remaining fragments of the Araucaria forest in Santa Catarina State (MMA, 2010). The sinuous and steep valleys form a series of rapids and waterfalls in both rivers, creating highly desirable conditions for hydropower development. Together, both rivers have 23 hydropower installations already operating just in their mainstems, and additional 23 are under different licensing or construction stages (Desenvolver, 2016). Most of these installations are diversion schemes classified as SHPs (< 30 MW), of which 13 were sampled in this study (Fig. 3.1a; Table 3.1). The SHPs Cachoeirinha and Abrasa are so close to each other – two dams essentially connected by a reservoir – that they were considered a single functional dam in this study. Most SHPs sampled have moderate to low water storage, but some of them have considerably large reservoirs relative to the size of the rivers, and their dewatering sections can be as long as 8 km (Table 3.1). The SHP development is a source of ongoing conflicts in the region, being listed as one of the primary threats to conservation in the Basin (MMA, 2010).

3.2.2 *Sampling design*

The 12 SHPs were surveyed during the low-flow season in the Chapecó River (March 3 to April 15, 2018) and the Chapecozinho River (February 2 to March 22, 2019). For each SHP, four sampling sites were selected: one least-impacted site and three sites under influence of the SHP (Fig. 3.1b). The site upstream of the SHP is characterized by flowing water habitats and is the least impacted by dams. This site was usually located upstream of at least two rapids/riffles complexes to avoid any effects of the dams on site-level flows. Near the dam structure, the “reservoir site” is a lentic or quasi-lentic river section created by the SHP impoundment. The “dewatering site” is located in the dewatering section downstream of the dam structure and upstream of the powerhouse outlet. The “downstream site” represents a flowing-water habitat located at least two rapids/riffles below the powerhouse outlet where water is returned to the river. The exact locations of each sampling site depended on accessibility, and were selected to accommodate the wide range of micro- and mesohabitats available. At each site, we delimited a ~250-m river section, where benthic macroinvertebrates and fish were sampled.

We collected three replicate samples of benthic invertebrates in order to represent the diversity of microhabitats available at the site-level (i.e. water velocity, depth and subtract type). Each replicate was sampled with a Surber stream-bottom sampler with an area of 1-m^2 and a mesh size of 0.25 mm, and samples were preserved in the field in alcohol 70% (Pérez, 1988). The collected material was then carefully washed in laboratory using 0.50 mm sieves and processed using a stereomicroscope. The aquatic macroinvertebrates were placed in vials, and further identified at the lowest taxonomic resolution as possible (mostly Family level) using appropriate taxonomic keys (Hamada, Nessimian, & Querino, 2014; Pérez, 1988). All the processing and identification happened at the Laboratory of Ecology of Unochapeco.

Fish assemblages were sampled using three sets of gill nets and eight minnow traps deployed within the 250-m section of the river, always installed in both margins and covering varied microhabitats present in the site. Both nets and traps were placed overnight, normally from 4 pm to 10 am (mean set time of 17.9 ± 3 hours). We installed three sets of 25 m-long gill nets with multiple mesh sizes in parallel to the shores, following proposed methods for standardized fish sampling in large rivers (Oliveira, Gomes, Latini, & Agostinho, 2014; Tejerina-Garro & de Mérona, 2010). Each set was composed by a combination of 5 m-long nets of five different mesh-sizes (15, 30, 40, 55, 80 mm between opposed knots), varying from 1.44 to 2.00 m in height. Fish were also sampled with eight wire-mesh Gee minnow traps (0.42 m long x 0.21 m diameter with two 35-mm openings) baited with 120 mL volume of dry dog food. This technique targets small-sized fish, and has proven to be an efficient passive sampling method. Two sets of four regularly spaced minnow traps (2.0 m apart) connected by cords were placed near shore in depths that ranged from 0.1 to 2.0 m. The captured fish were measured, weighed, and identified in the field at the species level, based on field guides for the Upper Uruguay River (Zaniboni-Filho, 2004). Most specimens were released at the same sites they were captured, but some individuals were collected for further validation of identification. Collected specimens were euthanized with lethal doses of eugenol, fixed in formalin solution and then transferred to alcohol 70% (AVMA, 2020). The voucher specimens were deposited at the fish collection of the Universidade de Brasília (CIUnB).

3.2.3 *Data analysis*

Patterns in invertebrate and fish assemblage composition were investigated using unconstrained ordination techniques based on Principal Coordinates Analysis (PCoA) (Gower, 1966). Taxon-count data from the different methods and locations within a sampling site were bulked and log-transformed, and then summarized using a PCoA on Bray-Curtis dissimilarity distances. Ordination axes were overlaid with a broken-stick randomization model as a stopping rule (Jackson, 1993), and we retained just the first two PCoA axes (PC1 and PC2) for all subsequent invertebrate and fish analyses. Differences in assemblage composition between the four site groups (upstream, reservoir, dewatering, downstream) were tested using a Permutational Multivariate Analysis of Variance (PERMANOVA) with 9,999 permutations (M. J. Anderson, 2001) and a Bonferroni corrected statistical significance level. We also tested for differences in multivariate data dispersions between the site groups (M. J. Anderson, 2006). All multivariate analyses were conducted using the functions ‘cmdscale’, ‘envfit’, ‘adonis’ and ‘betadisper’ from the R package ‘vegan’ (Oksanen et al., 2019).

Circular statistics were used to quantify the magnitude and direction of assemblage compositional change from the least-impacted (upstream) site to each of the dam-affected sites (reservoir, dewatering, downstream) according to their position in the ordination space. These three groups of assemblage compositional shifts are referred in the results as Δ-Reservoir, Δ-Dewatering and Δ-Downstream. Based on the Cartesian coordinates of PC1 and PC2, we calculated Euclidian distances (i.e. magnitude) and θ angles (i.e. direction) for vectors connecting upstream to reservoir site, upstream to dewatering site, and upstream to downstream for each SHP. Angles were then converted to polar coordinates and submitted to Rayleigh’s test for circular uniformity (Jammalamadaka & Sengupta, 2001). This test assess whether the distribution of angles representing directional shifts in assemblage structure for reservoir, dewatering, and downstream sites (referenced to upstream) across SHPs significantly departed from uniformity (in which angles occur in all directions with equal frequency) (Landler, Ruxton, & Malkemper, 2018). Watson’s two-sample test of homogeneity (Jammalamadaka & Sengupta, 2001) was conducted to test for differences between mean angle of directional change between reservoir, dewatered, and downstream sites. Angular data were used to construct circular histograms that depict the mean and frequency of directional shifts in assemblage structure. All the analysis were performed using the R-package ‘CircStats’ (Lund & Agostinelli, 2018).

Relationships between shifts in taxonomic composition and basic attributes of SHPs were investigated through multiple linear regression models. We compiled a set of attributes of potential ecological relevance from energy/environmental agency databases, hydrologic data and licensing reports (Table 3.1). Multicollinearity among attributes were tested through Pearson correlations, and a threshold of 0.70 was adopted for variables selection (Dormann et al., 2013). Distance to upstream headwater, river discharge and number of dams upstream were highly correlated (> 0.95), so just distance to headwaters was adopted as a metric of longitudinal position. Dam length is fairly highly correlated with reservoir area (> 0.70), and was removed from the analysis. Dam height, dewatering distance and generation capacity were also collinear metrics (ranging from 0.83 to 0.87). In this case, we opted to keep just dam height due to its overall relevance in the ecological literature as a proxy of dam impacts on flow (Poff et al., 1997). All SHP attributes were then log-transformed to adjust to the regression models.

We tested the relative importance of different subsets of dam attributes for predicting magnitude and direction of assemblage compositional change from the least-impacted (upstream) site to each of the dam-affected sites (reservoir, dewatered, downstream) using an information-theoretic approach to model selection (Burnham & Anderson, 2002). We considered 32 candidate models containing all possible combination of variables. Regression diagnostics from the global model with all variables indicated a reasonable statistical fit with no significant deviations from the parametric assumptions. Consequently, we ranked our candidate models using Akaike information criterion (AICc) corrected for small sample sizes (Burnham & Anderson, 2002) to evaluate the relative support (given the data) for each model. We also calculated the difference between the best approximating model value and all remaining model values (ΔAIC_i). Maximum log-likelihood estimates and Akaike weights (w_i), representing the relative likelihood of a model against all candidate models, were calculated. Evidence ratios were computed to quantify the relative support for each model by dividing the w_i of the best approximating model by the w_i for each individual model. All the analyses were conducted in R 3.6.1 (R Core Team, 2019), and results were considered significant when p-values < 0.05.

3.3 RESULTS

3.3.1 *Macroinvertebrate assemblages*

Our surveys resulted in 71 macroinvertebrate taxa, belonging to Platyhelminthes, Nematoda, Annelida, Mollusca, Crustacea, Arachnida, Entognatha and Insecta (Appendix D, Table S1). The two first axes of the PCoA (PC1 and PC2) explained 31% of the total variation, and 16 invertebrate taxa demonstrated statistically significant eigenvectors on one or both axes (Fig. 3.2a, $p < 0.05$). Macroinvertebrate assemblages differed among groups of sites (PERMANOVA Pseudo-F = 2.77, $R^2 = 0.16$, $p < 0.001$), reflecting significant differences in species composition of upstream (least-impacted) versus reservoir sites (Pseudo-F = 6.38, $R^2 = 0.22$, $p = 0.001$), and upstream versus dewatering sites (Pseudo-F = 3.15, $R^2 = 0.13$, $p = 0.003$). Macroinvertebrate taxa associated to flowing waters (e.g. fluvial specialists like Baetidae, Leptophlebiidae, Philopotamidae, Elmidae) are more prevalent in upstream sites, and showed decreased abundance and occurrence in dam-affected sites (reservoir, dewatered, and downstream). Assemblage composition was less variable in upstream sites compared to dam-effected sites ($F = 7.28$, $p < 0.001$). Macroinvertebrate assemblage composition was the most variable in reservoir sites, and were generally devoid of fluvial specialists and dominated by taxa associated with standing waters (e.g. Caenidae, Euthyplociidae, Oligochaeta and Hirudinea). Dewatering and downstream sites showed intermediate variability in comparison to upstream and reservoir sites. In general, dewatering and downstream sites are both less depauperate of fluvial specialists in comparison to reservoir sites, but may also host taxa associated with standing waters in some SHPs.

We detected directional (non-uniform) shifts in macroinvertebrate species composition from the least-impacted (upstream) site to each of the dam-affected sites (reservoir, dewatered, downstream), generally moving from left to right along the first ordination axis (Fig. 3.3; Table 3.2). Directionalities were stronger for Δ-Reservoir (Rayleigh's test rho = 0.78, $p < 0.001$), followed by Δ-Dewatering (rho = 0.50, p = 0.05), not being significant for Δ-Downstream (rho = 0.36, p = 0.23). Although the magnitude of these changes (relative to upstream sites) are variable among SHPs, directionality of change for all dam-affected sites were not significantly different (Watson's test statistic $U_{\text{reservoir-dewatering}} = 0.06$, $p > 0.10$; $U_{\text{reservoir-downstream}} = 0.12$, $p > 0.10$; $U_{\text{dewatering-downstream}} = 0.02$, $p > 0.10$). Therefore, indicating an overall decrease in relative abundance of fluvial specialists and an increase in the abundance of taxa associated to standing-waters in dam-affected sites (Fig. 3.2a for reference).

The overall variance in magnitude and directionality of shifts in composition indicate heterogeneous effects of SHPs, which can be explained by some of their structural and spatial attributes (Table 3.3, Fig. 3.4a-b). The most supportive model for the magnitude of upstream-reservoir assemblage change incorporated dam height (AICc = 0.11, $w_i = 0.31$); this model was two times more likely given the data than the next most competitive model (Table 3.3). Taller dams were found to have larger effects on Δ-Reservoir (Fig. 3.4a; $R^2 = 0.30$, F = 5.67, p = 0.04). None of the candidate models outperformed the null model for Δ-Dewatering, indicating that the attributes tested are not good predictors of composition shifts or that there is little variability on the magnitude of shifts among SHPs (Table 3.3). The longitudinal position was the most supportive predictor of the magnitude of shifts in the sites located downstream of the dams (Fig. 3.4b; AICc = 6.40, $w_i = 0.34$), with more changes in Δ-Downstream happening further downstream of the headwaters ($R^2 = 0.28$, F = 5.24, p = 0.05).

3.3.2 *Fish assemblages*

We recorded a total of 16 fish species, belonging to five Families and four Orders (Appendix D, Table S2), all of them native to the Upper Uruguay System. Together, PC1 and PC2 explained 31% of the variation of the PCoA, which were significantly correlated with eigenvectors of eight fish species (Fig. 3.2b, $p < 0.05$). Fish composition did not change among groups of sites (Pseudo-F = 1.25, $R^2 = 0.08$, $p = 0.24$), and variation in data dispersion was homogeneous among all the groups ($F = 1.14$, $p = 0.35$). Although not significant, most reservoir sites were grouped within the top-left quadrant of the ordination space, corresponding to higher relative abundances of the cichlid *Geophagus brasiliensis* (Quoy & Gaimard, 1824), the erythrinid *Hoplias malabaricus* (Bloch, 1794) and the characin *Astyanax* cf. *lacustris* (Lütken, 1875) – three species often associated to low flow habitats or even standing waters in the South of Brazil (Borba, Latini, Baumgartner, Gomes, & Agostinho, 2019; Hirschmann, Majolo, & Grillo, 2008). Most upstream, dewatering and downstream sites were grouped outside of this quadrant, where other species associated to fluvial habitats were more abundant. For instance, loadings from the heptapterid *Rhamdia quelen* (Quoy & Gaimard, 1824), the cichlid *Chrenicichla igara* Lucena & Kullander (1992) and most loricariids attract the scores to the lower right quadrant, although the last two were not significantly correlated with PC1 nor PC2. We did not record any exotic fish species in our samples, but local anglers reported the presence in low densities of the common carp *Cyprinus carpio* Linnaeus (1758), grass carp *Ctenopharyngodon idella* (Valenciennes, 1844), and largemouth bass *Micropterus salmoides* (Lacepède, 1802).

Directional departures from upstream assemblage structure also occurred for fishes, but overall changes were much more variable in comparison to macroinvertebrate assemblages (Fig. 3.5; Table 3.2). We only detected significant directional changes in assemblage's composition for

Δ-Reservoir, generally moving towards the top-left panel of the ordination space (Fig. 3.5a; Rayleigh's test rho = 0.67, p = 0.003). This direction corresponds to the loadings of species associated to low-flow habitats such as *G. brasiliensis*, *H. malabaricus* and *A. lacustris* (Fig. 3.2b for reference). The distribution of angles for Δ-Dewatering and Δ-Downstream did not differ from uniform distributions (Fig. 3.5; Table 3.2), and directionality of change for all dam-affected sites were not significantly different (Watson's test statistic $U_{\text{reservoir-dewatering}} = 0.11$, $p > 0.10$; $U_{\text{reservoir-downstream}} = 0.13$, $p > 0.10$; $U_{\text{dewatering-downstream}} = 0.03$, $p > 0.10$).

SHP attributes also explained shifts in fish assemblage composition, but just for the upstream-reservoir site comparison. The most supportive model for the magnitude of upstream-reservoir assemblage change included reservoir area as the only response variable (Table 3.3); this model was five times more likely given the data than the next competitive model (AICc = -3.96, $w_i = 0.49$). Wider reservoirs were found to have larger effects on Δ-Reservoir (Fig. 3.4c; $R^2 = 0.41$, F = 5.67, p = 0.02). None of the candidate models outperformed the null models for Δ-Dewatering and Δ-Downstream (Table 3.3), indicating that the attributes tested are not good predictors of fish composition shifts or that there is little variability on the magnitude of shifts among SHPs. As observed for macroinvertebrates, the magnitude of taxonomic composition shifts in different site-comparisons are not predictable by the same SHP attributes, which also vary between the two assemblages (Table 3.3).

3.4 DISCUSSION

Habitat lentification caused by SHPs modifies taxonomic composition of both macroinvertebrate and fish assemblages, but the magnitude and directionality of changes were not homogeneous among dams and sites. We found evidence for directional shifts in taxonomic composition, reflecting a gradual replacement of taxa associated to fast-flowing habitats (i.e.

fluvial specialists) by habitat generalists that are more tolerant of standing waters. We also found that the magnitude of these shifts is predictable by some structural and spatial attributes of the dams, particularly those related to reservoir dimensions and the longitudinal position of the SHP. However, the predictive power of these attributes varied considerably between macroinvertebrates and fish assemblages. Overall, we demonstrate that SHPs may have a substantial and quite heterogeneous footprint on freshwater communities.

Water storage in reservoirs is the major driver of shifts in taxonomic composition of both macroinvertebrate and fish assemblages. Storage and backwater effects by small dams impact the hydrology and geomorphology of rivers above the impoundment, resulting in overall reductions of water velocity and on deposition of fine sediments (Shane Csiki & Rhoads, 2010). Such physical changes on basic habitat characteristics have important implications for freshwater communities, particularly for guilds of macroinvertebrates that rely on fast-flows, oxygenated waters and consolidated substrates (Linares et al., 2019; Ruocco, Portinho, & Nogueira, 2019; Stanley et al., 2002). We detected decreases in relative abundances of some families of mayflies, stoneflies and caddisflies (e.g. Baetidae, Leptophlebiidae, Perlidae, Philopotamidae), which are typical fluvial specialists (Bispo & Oliveira, 2007; Hart & Finelli, 1999). Conversely, generalist species tend to prevail in reservoirs with higher storage capacity and poor water quality (Freeman & Marcinek, 2006; Stanley et al., 2002). For instance, the pearl cichlid *G. brasiliensis* dominated fish assemblages in reservoirs, with juveniles being strikingly abundant in larger reservoirs. This species has omnivorous habits, sedentary behavior and an equilibrium life history strategy (Bastos, Condini, Varela, & Garcia, 2011), which explains its success in colonizing habitats with more stable flows. Other studies in the South of Brazil have also reported major increases in abundances of *G. brasiliensis* after dam closures (Borba et al., 2019; Hirschmann et al., 2008), reinforcing their high affinity to artificial lentic habitats.

Habitat-effects of dewatering and artificial downstream releases were also important drivers of shifts in taxonomic composition of macroinvertebrates. Other studies examining the effects of SHPs on dewatering and downstream sites also reported extirpations of fluvial specialists, dominance of generalists and even the facilitation of invasions by non-native mollusks (D Anderson et al., 2017; Jesus et al., 2004; Linares et al., 2019). Aquatic forms of macroinvertebrates are in general susceptible to fine-scale changes in habitat characteristics associated with dam operations reduced water velocity (Hart & Finelli, 1999; Haxton & Findlay, 2008; Mbaka & Mwaniki, 2015). Moreover, small dams can promote major downstream effects on macroinvertebrate assemblages through warm-water releases (Lessard & Hayes, 2003), movement constraints (Benstead et al., 1999) and changes in primary productivity and nutrient cycling (Arroita et al., 2017; Haxton & Findlay, 2008). By contrast, we did not detect significant shifts in fish assemblages downstream or in dewatering sites, contradicting studies that reported dewatering to be the most relevant impact of SHPs on fish assemblages (E. P. Anderson et al., 2006; Jumani et al., 2018; Kubecka et al., 1997). This mismatch is probably caused by the importance of migratory species in these studies, which are historically absent in the Upper Chapecó (Barradas, Silva, Harvey, & Fontoura, 2012). Another study in Southern Brazil examined the effects of dewatering on fish assemblages in a much larger dam (130 MW and dewatering section of 22 km) and also did not report major shifts in fish composition or losses of fluvial specialists – although the relative abundance of some species like *G. brasiliensis* increased four-fold (Borba et al., 2019). Their findings suggest that the patterns we observed are consistent with other studies on larger dams in the region.

Our results reveal that the magnitude of shifts in taxonomic composition can be quite variable among SHPs, which in some instances can be explained by structural dam attributes. Dam height is a simple metric that combines rough expectations for a series of biophysical impacts associated with water storage, such as water residence time, sediment transport and thermal modification (Poff & Hart, 2002). Although this approximation has limitations on describing the hydrologic effects of small dams by focusing on the impacts of storage and not of diversion (Kibler, 2017), the shifts in macroinvertebrate composition were fairly well predicted by dam height in reservoir assemblages in our study. Alternatively, shifts in the composition of fish species in reservoir sites were best predicted by reservoir area, probably reflecting the increased availability and stability of spawning and rearing habitats in littoral zones. More stable water bodies result in higher recruitments of the species *G. brasiliensis* and *H. malabaricus* that spawn in nests (de Lima, Reynalte-Tataje, & Zaniboni-Filho, 2017), a condition that is more likely to happen in larger reservoirs. The longitudinal position was the only predictor of taxonomic shifts below the SHPs, indicating that the magnitude of changes in macroinvertebrate assemblages is larger for dams that are more distant from the headwaters. River ecologists theorized that the spatial position of dams along the longitudinal gradient may result in distinct ecological effects, and that some of these effects can even accumulate downstream (Ward & Stanford, 1983). The degree of flow regulation and water residence time are expected to accumulate in the downstream direction in dam cascades (Bernhard Lehner et al., 2011), with macroinvertebrate assemblages being exposed to serial disturbances in flow regimes by serial SHPs.

One of the primary environmental concerns regarding the proliferation of SHPs is their potential for causing ecological changes that can manifest cumulatively across the riverscape (Kibler & Tullos, 2013; Lange et al., 2018). Direct lentification through reservoir storage and dewatering already affected 36% of the 390 km of the Chapecó and Chapecozinho mainstems (estimated here from their confluence to the headwaters based on images from Google Earth). An additional part of the remaining 64% is subjected to flow regulation and backwater effects. Therefore, it is reasonable to suppose that the shifts in taxonomic composition we observed are happening in freshwater communities in much of the Basin. For instance, cumulative habitat lentification can be a facilitator to the establishment of generalist and invasive species throughout the riverscape (Freeman & Marcinek, 2006; Johnson, Olden, & Vander Zanden, 2008). Anecdotally, anglers from different locations across the Chapecó Basin reported that *G. brasiliensis* was rare and not as widespread just a few decades ago, when most SHPs were not built. Our results indicate that *G. brasiliensis* has high affinity for SHP reservoirs, representing possible sources for their spread. In addition, cumulative lentification of fast-flowing habitats like rapids and riffles may result in local or even global extinctions of endemic species that are restricted to these habitats (De Lucena, 2007; Ruocco et al., 2019). These are potential examples of cumulative basin-wide effects of SHPs, but they still need specific assessments for validation.

3.5 CONCLUSIONS

Cumulative effects of habitat lentification and flow regulation are potential threats for the freshwater biodiversity of the Chapecó River Basin, but they have been largely underestimated in environmental assessments for new SHP construction (Desenvolver, 2016). Our results reveal that the magnitude of the ecological effects of SHPs on freshwater communities have quite heterogeneous magnitudes. Therefore, the “small” modifier alone is not sufficient to drive expectations on the potential of hydropower dams to promote environmental changes. Although our study shed light on the role of SHPs’ heterogeneity on their individual and cumulative effects on freshwater communities, there was not a universal variable able to predict the magnitude of shifts in composition for all site-locations and taxonomic groups examined. Therefore, it is unlikely that a priori classifications of dams based on their structural attributes are able to replace site-specific assessments of their impacts (e.g. Environmental Impact Assessments - EIAs). Currently, many countries do not require or require just a simplified version of EIAs on environmental licensing procedures for SHPs (Couto & Olden, 2018), which is clearly inadequate given our limited knowledge on the relationships between the heterogeneity of hydropower dams and their individual and cumulative ecological effects. Managers and decision-makers should look beyond hydropower-size classifications in order to implement effective policies and mitigation actions targeting the conservation of freshwater biodiversity.

3.6 FIGURES

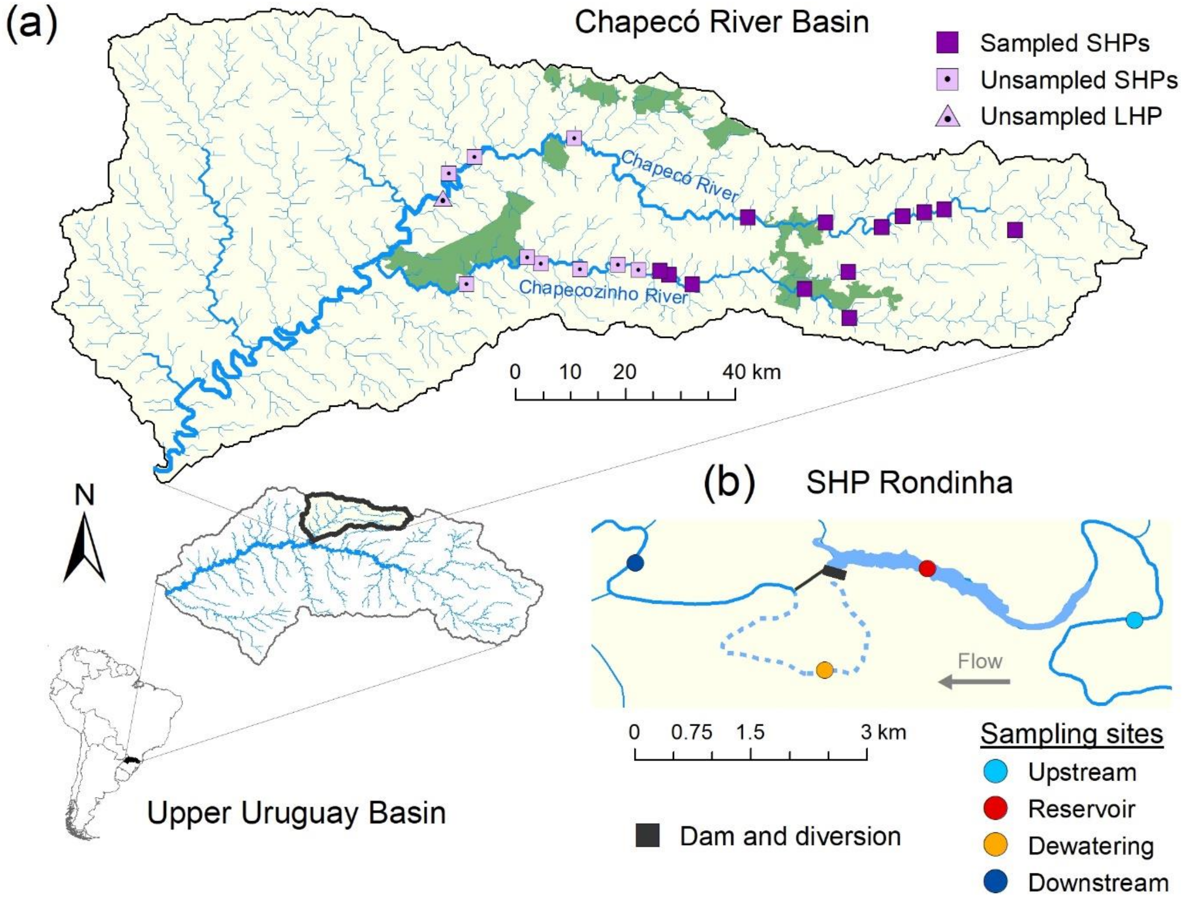

Figure 3.1. (a) Map depicting the 22 Small Hydropower Plants (SHPs) and the one Large Hydropower Plant (LHP) operating in the mainstems of Chapecó and Chapecozinho Rivers. The sampled SHPs are represented by purple squares and unsampled dams are represented in lilac. Green polygons represent the protected areas and lands owned by indigenous groups. (b) The inset on the SHP Rondinha illustrates the sampling design that included surveys of fish and benthic macroinvertebrates in upstream (light blue), reservoir (red), dewatering (orange) and downstream (dark blue) sites.

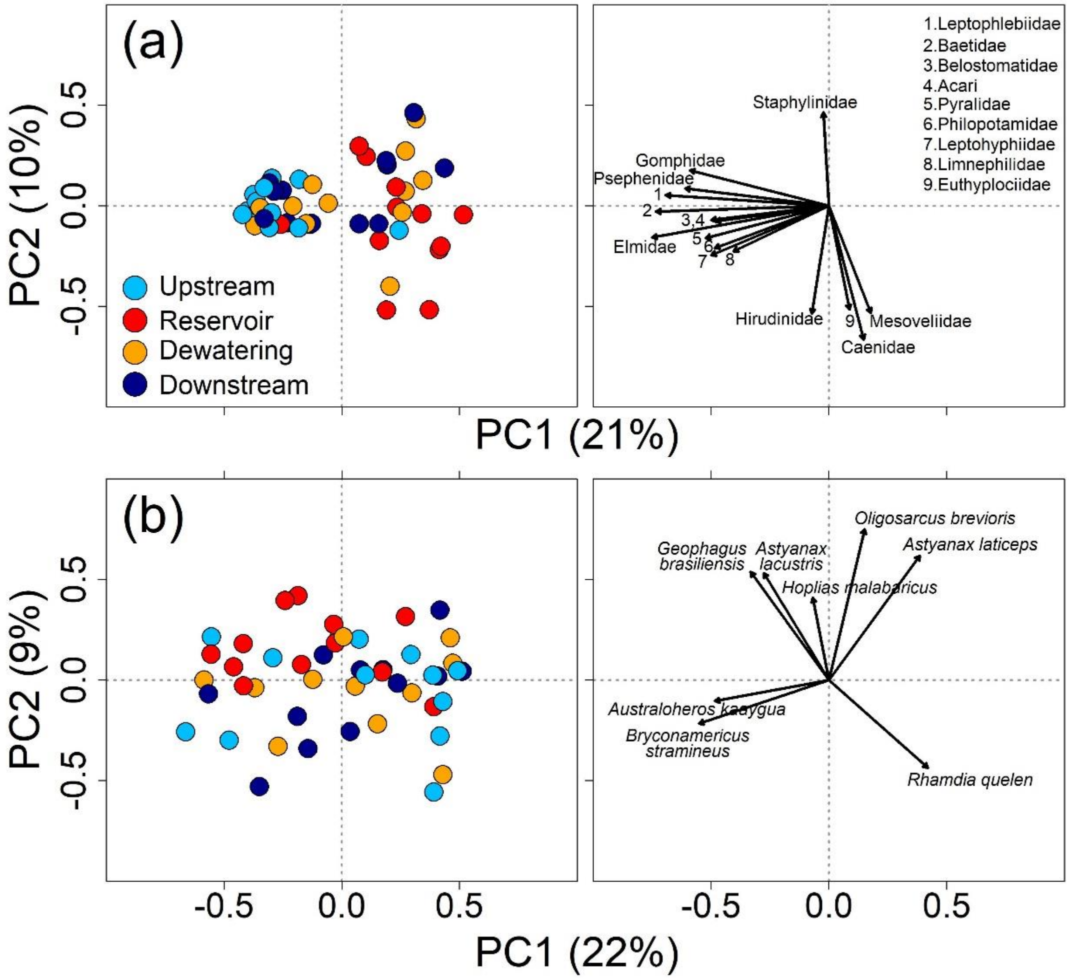

Figure 3.2. Ordination biplots of Principal Coordinate Analysis (PCoA) for (a) macroinvertebrate and (b) fish assemblages of the Chapecó River Basin, Brazil. PC1 and PC2 are the first two axes of each ordination (variation explained in parenthesis). Colored circles represent the site-scores, and the black arrows represent the correlation of each axis with taxon vectors (only significant correlations shown; $p < 0.05$). Colors distinguish upstream (light blue), reservoir (red), dewatering (orange), and downstream (dark blue) sites.

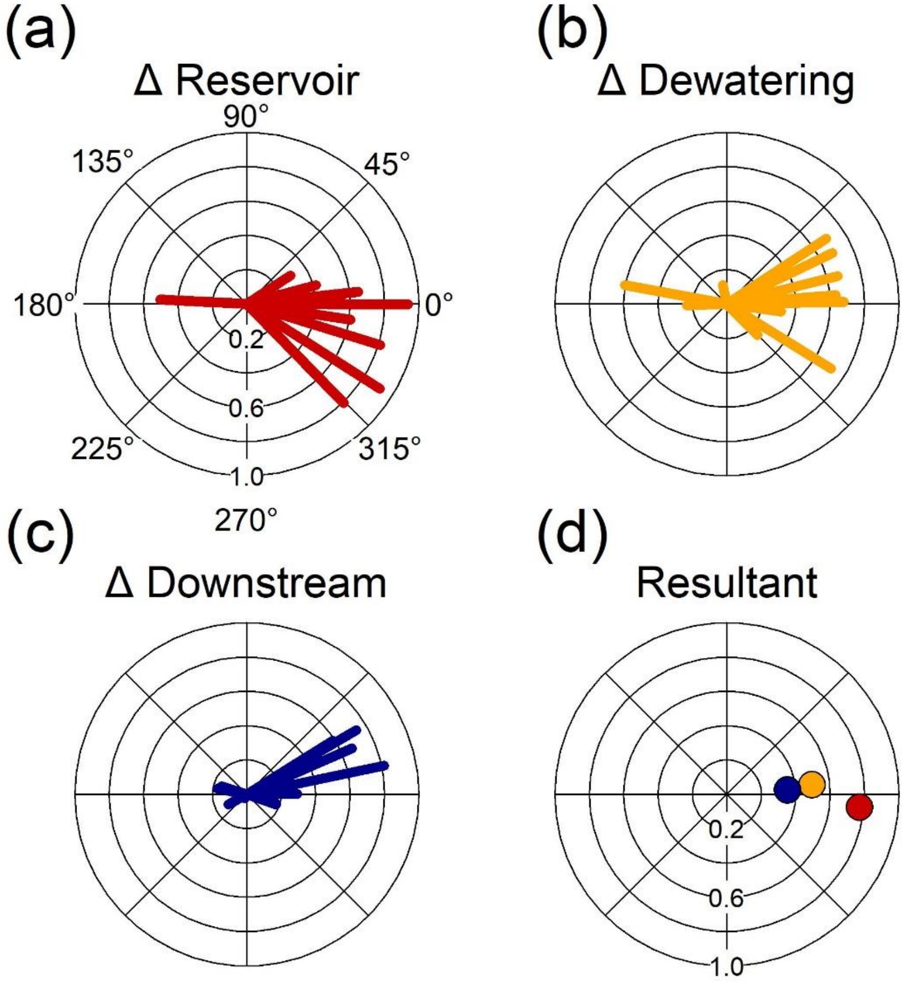

Figure 3.3. Polar plot depicting taxonomic shifts (Δ) in benthic macroinvertebrate assemblages between the least-impacted upstream site and the dam-affected sites under influence of the 12 SHPs (a-c). The length and angle of each colored bar represent respectively the magnitude (i.e. Euclidian distance) and direction of the shift in the ordination space (PC1 and PC2). (d) The resultant directionality (i.e. Rayleigh's test mean resultant length) for each site-comparison is represented by a colored circle: Δ-Reservoir (red), Δ-Dewatering (orange), and Δ-Downstream (dark blue).

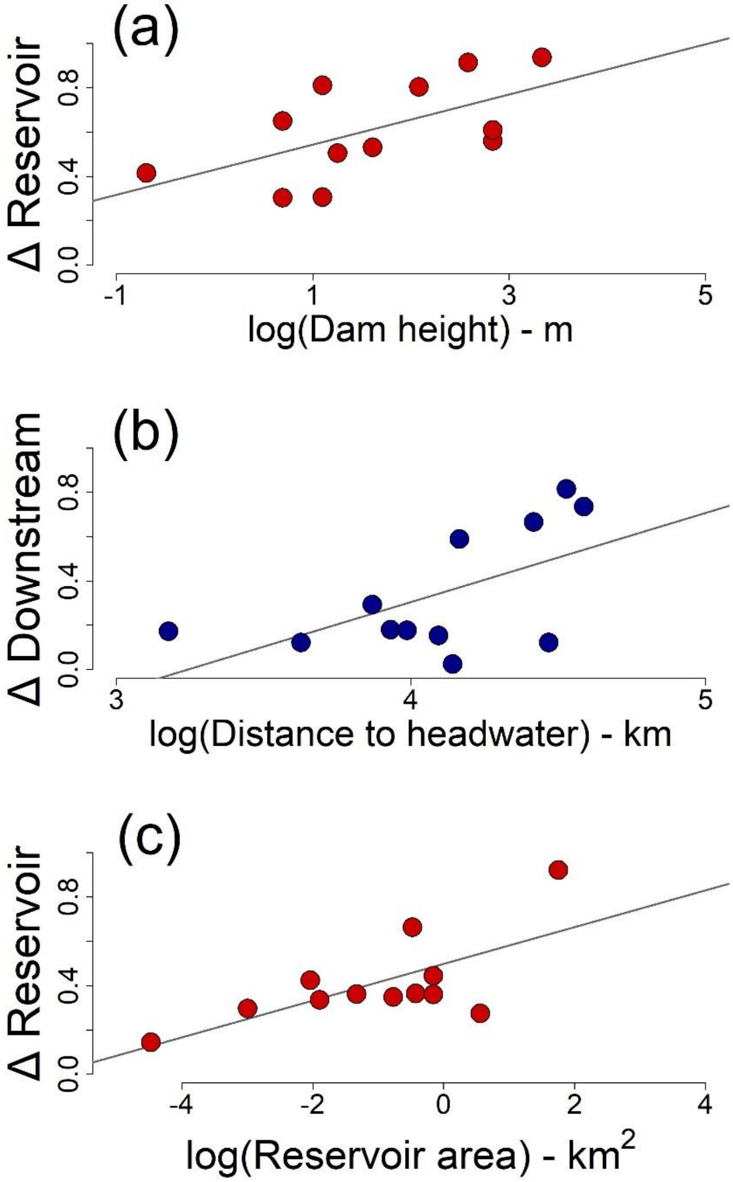

Figure 3.4. Relationships between the magnitude of shifts in site-comparison taxonomic composition (Δ) and SHP attributes for (a-b) macroinvertebrate and (c) fish assemblages. Results are based on the most supportive models selected using AICc.

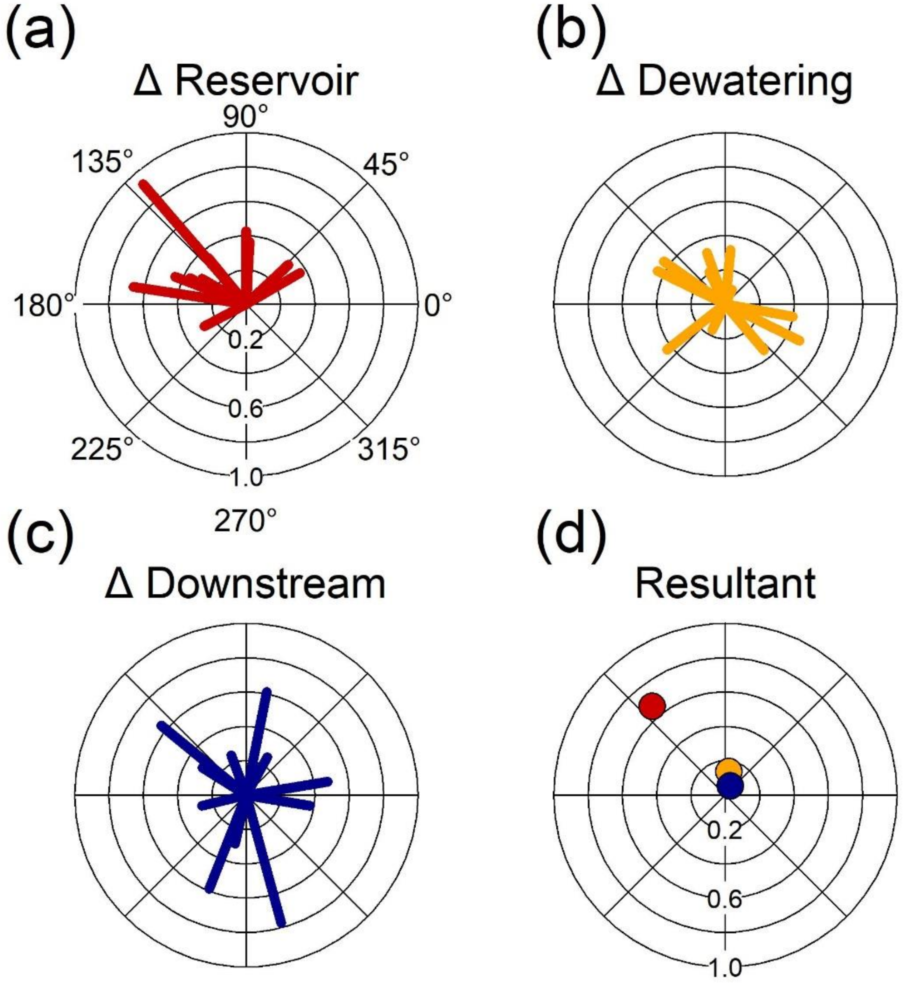

Figure 3.5. Polar plot depicting taxonomic shifts (Δ) in fish assemblages between the least-impacted upstream site and the dam-impacts sites of the 12 SHPs (a-c). The length and angle of each colored bar represent respectively the magnitude (i.e. Euclidian distance) and direction of the shift in the ordination space (PC1 and PC2). (d) The resultant directionality (i.e. Rayleigh's test mean resultant length) for each site-comparison is represented by a colored circle: Δ-Reservoir (red), Δ-Dewatering (orange), and Δ-Downstream (dark blue).

3.7 TABLES

Table 3. 1. List of attributes of the 12 Small Hydropower Plants (SHPs) sampled in this study. Average annual discharge and distance to headwater were sourced from Linke *et al.* (2019), dewatering distances were measured in Google Earth, and the remaining attributes were retrieved from federal and state-level databases and licensing reports (ANEEL 2020a-b).

SHP name	**Sub-basin**	**Generation capacity (MW)**	**Annual average discharge (m^3/s)**	**Distance to headwater (km)**	**Number of dams upstream**	**Dam height (m)**	**Dam length (m)**	**Reservoir area (km^2)**	**Dewatering distance (m)**	**Opening date**
Flor do Mato	Chapecozinho	4.80	11.9	50.933	0	13.2	267.3	5.77	280	6-Dec-1946
Adami Quimica	Chapecozinho	3.00	5.2	37.59	0	8.0	123.9	0.26	370	1-Jan-1970
Rio do Mato	Chapecozinho	1.20	14.7	62.896	1	0.5	38.0	0.01	360	26-Jun-2014
Dalapria	Chapecozinho	1,44	29.1	87.113	3	2.0	40.0	0.15	380	1-Feb-1985
Cachoeirinha-Abrasa	Chapecozinho	2.14	31.3	92.498	4	3.0	140.0	0.65	80	1-Jan-2007
Tonet	Chapecó	0.76	4.0	23.967	0	3.5	30.0	0.13	30	1-May-1975
Contestado	Chapecó	5.60	17.5	47.882	1	3.0	234.3	0.85	3,760	1-Nov-2007
Coronel Araujo	Chapecó	5.80	19.2	53.883	2	5.0	183.3	0.86	470	1-Nov-2007
Das Pedras	Chapecó	5.60	21.4	59.919	3	17.0	144.5	0.46	4,300	23-Dec-2017
Salto Santo Antonio	Chapecó	6.24	23.5	64.378	4	2.0	90.0	0.05	500	1-Jan-1963
Passos Maia	Chapecó	25.00	30.4	82.76	5	28.0	199.0	1.75	7,950	18-Feb-2012
Rondinha	Chapecó	9.60	38.3	98.112	6	17.0	172.0	0.62	4,350	4-Jun-2014

Table 3. 2. Directional statistics of shifts (Δ) in taxonomic composition between the least-impacted upstream site and the three types of dam-affected sites (i.e. reservoir, dewatering and downstream) for macroinvertebrate and fish assemblages. Resultant directions in degrees, resultant lengths (rho ± variance), and p-values are provided. Significant relationships are highlighted in bold ($p < 0.05$).

Assemblage	Site-comparison	Resultant direction (°)	rho	p-value
Macronivertebrates	Δ-Reservoir	-5.73	0.78 ± 0.2	**< 0.001**
	Δ-Dewatering	6.43	0.50 ± 0.5	**0.05**
	Δ-Downstream	4.11	0.35 ± 0.7	0.36
Fish	Δ-Reservoir	129.56	0.67 ± 0.3	**0.03**
	Δ-Dewatering	81.02	0.14 ± 0.9	0.80
	Δ-Downstream	60.23	0.06 ± 0.9	0.96

Table 3. 3. Model selection process for the relationships between magnitude of taxonomic shift (Δ) and SHP attributes based on Akaike information criterion (AIC). Global models include the river sub-basin (River), distance to the headwater (Longitudinal), dam height (Height), reservoir area (Area), and dam age (Age). All the possible combinations of variables from the global models were ranked according to AICc, and the three most supportive models for each site-comparison are shown. The sign of coefficients of the most supportive models are inside parenthesis and significant relationships are in bold ($p < 0.05$). Values of AICc, ΔAIC, weights (w_i) and evidence ratio (ER) of each model are provided.

Assemblage	**Site-comparison**	**Global model**	**Model**	**Rank**	**AICc**	**ΔAIC**	**w_i**	**ER**
Invertebrates	Δ-Reservoir	River + Longitudinal + Height + Area + Age	**Height(+)**	1	0.1	0.0	0.31	-
			River + **Height(+)**	2	1.6	1.5	0.15	2.1
			Null	3	1.8	1.7	0.13	2.4
			Area(+)	3	1.8	1.7	0.13	2.4
	Δ-Dewatering	River + Longitudinal + Height + Area + Age	Null	1	5.2	0.0	0.43	-
			Age(+)	2	7.1	1.9	0.17	2.5
			Height(-)	3	8.6	3.4	0.08	5.3
	Δ-Downstream	River + Longitudinal + Height + Area + Age	**Longitudinal(+)**	1	6.4	0.0	0.34	-
			Null	2	7.8	1.4	0.17	2.0
			River + **Longitudinal(+)**	3	8.8	2.4	0.11	3.6
Fish	Δ-Reservoir	River + Longitudinal + Height + Area + Age	**Area(+)**	1	-4.0	0.0	0.49	-
			Area(+) + Age(+)	2	-0.8	3.2	0.10	4.9
			Null	3	-0.1	3.8	0.07	7.0
	Δ-Dewatering	River + Longitudinal + Height + Area + Age	Null	1	-11.2	0.0	0.33	-
			Area(+)	2	-9.8	1.4	0.17	1.9
			Height(+)	3	-8.9	2.3	0.11	3.0
	Δ-Downstream	River + Longitudinal + Height + Area + Age	Null	1	1.5	0.0	0.24	-
			Age(+)	2	2.5	1.0	0.14	1.7
			River	3	3.0	1.5	0.11	2.2

Chapter 4. EFFECTS OF SMALL HYDROPOWER PLANTS WITH VARIED STORAGE CAPACITIES ON WATER TEMPERATURE REGIMES IN THE CHAPECÓ RIVER BASIN, BRAZIL

Small Hydropower Plants (SHPs) are proliferating in many developing countries across the world, representing major threats to the conservation of tropical and sub-tropical river ecosystems. Although the alteration of water temperature regimes has been described as one of the main ecological impacts of dams, very little is known about the potential effects of SHPs on stream temperatures. Here, we investigate how water temperature regimes vary across time and space in the Chapecó River Basin (Brazil), and how seven SHPs of varied dimensions affect downstream water temperatures through diversion and storage. We found that four SHPs modify daily thermal regimes, three of them by warmer downstream releases. The magnitude of changes was not homogeneous throughout the year for most SHPs, with larger effects happening in the transition between the fall and the winter, and during the summer. In addition, our results indicate that the effect of SHPs on daily stream temperatures is quite variable among dams, ranging from almost negligible to changes that can be as high as 2.6° C warmer, on average. Overall, the results of this research bring novel information about the effects of SHPs on water temperature regimes in sub-tropical rivers and shed light on the potential effects of the sheer number of SHPs on water temperatures basin wide.

4.1 INTRODUCTION

Hydropower is the most important renewable technology supplying global power grids (REN21, 2015), but this benefit is generally coupled to important social and environmental impacts. Although large dams are usually recognized as the most important source of environmental impacts of hydropower, the vast majority of dams operating in the world are in reality small-sized (Couto & Olden, 2018). More recently, there has been a remarkable global increase in new construction of Small Hydropower Plants (SHPs) – i.e. smaller dams with generation capacities below given thresholds (in most cases < 10 MW) that may include both storage dams and 'run-of-river' schemes (i.e. without considerable storage capacity) (David Anderson et al., 2015; Couto & Olden, 2018; Kelly-Richards et al., 2017). Therefore, SHPs compose a large proportion of hydrological changes in river ecosystems and can have major effects on freshwater biodiversity and water quality (Kibler & Tullos, 2013).

The alteration of water thermal regimes is one of the main ecological impacts of dams, and can result in very distinct effects on downstream water temperatures (i.e. cold vs warm releases) (J. D. Olden & Naiman, 2010; Webb, Hannah, Moore, Brown, & Nobilis, 2008). Water storage modifies different paths of heat exchange dynamics in rivers (Caissie, 2006), and reservoir characteristics and dam operation modes are in general primary determinants of modified downstream temperatures (J. D. Olden & Naiman, 2010). For instance, cold-water releases are usually associated with deeper and thermally stratified reservoirs, especially in cases where intakes draw water directly from the hypolimnetic zone (J. D. Olden & Naiman, 2010). Conversely, warm-water releases tend to occur when surface waters are selected from shallow reservoirs (Lessard & Hayes, 2003). Both cases – cold or warm water releases – have important ecological effects on river ecosystems, including changes in productivity, diversity and species composition (Haxton & Findlay, 2008; J. D. Olden & Naiman, 2010; Todd, Ryan, Nicol, & Bearlin, 2005).

The impacts of dams on water temperatures have been predominantly studied in systems of which dams have large storage capacity and cold-water releases (J. D. Olden & Naiman, 2010). However, there is mounting evidence that small dams can also have important effects on thermal regimes (E. P. Anderson et al., 2006; Hayes et al., 2008; Lessard & Hayes, 2003; Meier et al., 2003) – most of them reporting warm-water releases. For instance, small dams with surface water releases in Michigan increased downstream water temperatures in 2.7 °C on average during the summer, which resulted in shifts in macroinvertebrates community composition and decreased abundances of cold-water fish species (Lessard & Hayes, 2003). In addition, operation modes and storage capacity of small dams are directly related to their effects on stream temperatures, and 'run-of-river' schemes usually have less effects on downstream temperatures than dams with storage (Maheu, St-Hilaire, Caissie, & El-Jabi, 2016). Therefore, it is expected that the effects of SHPs on temperature regimes can be quite diverse, and directly related to their operation modes (i.e. 'run-of-river' vs storage).

Many SHPs operate diverting water through pipes or canals to a distant powerhouse, creating dewatering sections that can last for many kilometers (Couto & Olden, 2018). Although the ecological and hydrological impacts of dams are usually perceived as a direct response to water storage, the hydrological effects of dewatering can be quite high for small dams (Kibler, 2017). For instance, dewatering may have significant effects on downstream thermal regimes, especially in longer dewatered reaches that run in lower slope channels (Meier et al., 2003). Changes in water temperatures in dewatering sections may also have implications for biota. For instance, increased water temperatures and reduced dissolved oxygen through dewatering are major drivers of shifts in fish community composition in tropical rivers impacted by SHPs (E. P. Anderson et al., 2006; Jumani et al., 2018).

Multiple SHPs are very often implemented in the same river (i.e. as dam cascades), and their ecological effects are not necessarily restricted to a single location, being manifested cumulatively across the riverscape (Kibler & Tullos, 2013). Modifications of thermal regimes by large dams were reported to reverberate downstream beyond the immediate range of the impact source, being detected in some cases many kilometers downstream (Ellis & Jones, 2013; Long, Ji, Liu, Yang, & Lorke, 2019; Preece & Jones, 2002). Similarly, the effects of small dams on water temperatures can be maintained for a few kilometers downstream of the impoundment (Lessard & Hayes, 2003; Maheu et al., 2016). Therefore, changes in water temperatures by multiple SHPs may be manifested cumulatively in the river when they are implemented in high densities, which is concerning considering the sheer number of prospective SHP construction.

Tens of thousands new SHPs are expected to start operating in the coming decades, most of them in developing countries (Couto & Olden, 2018). However, very little is known about the ecological consequences of dam-induced changes in water thermal regimes in tropical and sub-tropical rivers – particularly for small dams (E. P. Anderson et al., 2006; J. D. Olden & Naiman, 2010; Webb et al., 2008). Here, we investigate the thermal water regimes of the Chapecó River Basin (Brazil), where dozens of SHPs have been installed in the last decades. We examined how water temperature regimes vary in unregulated and SHP-regulated sites across time and space in the Basin, and how seven different SHPs may affect downstream water temperatures. We predict that SHPs release warmer waters downstream due to the effects of surface water intake and dewatering. We also expect that the effects of SHPs on thermal regimes vary due to their high diversity of structural characteristics and operation modes. Overall, the results of this research bring novel information about the effects of SHPs on downstream water temperatures and shed light on the potential basin-wide effects of the sheer number of SHPs on stream temperatures.

4.2 METHODS

This study was conducted in the Chapecó and Chapecozinho Rivers, Chapecó River Basin, Uruguay drainage, Brazil (Fig. 4.1). Together, both rivers have 23 hydropower installations already operating just in their mainstems, and additional 23 are under different licensing or construction stages (Desenvolver, 2016). Most of these installations are diversion schemes classified as SHPs (< 30 MW) and all of them have surface water outlets. Most SHPs sampled have moderate to low water storage, but some of them have considerably larger reservoirs relative to the size of the rivers, and their dewatering sections can be as long as 8 km (Table 4.1). The land use can be quite intense in the Basin, primarily driven by forestry activities and cattle ranching. However, riparian zones in the study area are fairly well preserved, especially in the surrounding areas of the Araucárias National Park – created in 2005 as an effort to protect the remaining fragments of the Araucaria forest in Santa Catarina State (MMA, 2010). The SHP development is a source of ongoing conflicts in the region, being listed as one of the primary threats to conservation in the Basin (MMA, 2010).

A total of 28 temperature loggers were placed and recovered across the Basin between the summers of 2018 and 2019 (Fig. 4.1). The temperature loggers (models HOBO pendant or TidBit, Onset®, U.S.A.) were placed in sites of flowing waters (i.e. not in the reservoirs) and installed in submerged boulders with steel cables. The exact siting of each logger was selected to be as much distant from the margin as possible to decrease chances of desiccation, but on depths of less than 1.5 m to increase chances of visual location. Each logger was synchronized to record temperature data in a four-hour interval (at 2:00, 6:00 and 10:00 am/pm). We tried to spread the loggers throughout the basin as much as possible, which allowed us to have an overall snapshot of the whole Basin. A set of 16 loggers were installed in sites that are not directly affected by SHPs (i.e. unregulated sites), which we define here as sites located upstream of a reservoir or at least 5 km downstream of a dam. Loggers were also installed immediately upstream and downstream of seven SHPs in flowing waters, and three loggers were installed in dewatering sections in three of these SHPs (Table 4.1).

All the time series were visually inspected for errors and discrepancies (Sowder & Steel, 2012), which were treated as missing values in the few sites that needed some correction. Final datasets were converted to mean daily water temperatures, and then basic parameters on magnitude, variability, frequency and timing of river thermal regimes were estimated with the R package ‘streamThermal’ (Arismendi, Johnson, Dunham, & Haggerty, 2013; Tsang et al., 2016). The effects of SHPs were inferred comparing temporal trends and the differences between upstream and regulated sites (i.e. downstream and dewatering). Spatial synchrony between thermographs were estimated using Pearson cross-correlations, and Pearson’s r coefficients were used to explore the relationship between synchrony and attributes of the SHPs (i.e. reservoir area and dewatering distance).

4.3 RESULTS

Water temperatures varied considerably across time and space in unregulated sites, ranging from a minimum of 7.5 °C to a maximum of 28.8 °C (sub-daily records). Although there was a lot of variation on daily mean temperatures among sites, June and August were consistently the coldest months of the year, and December and January the warmest (Fig. 4.2). The magnitude and variability of daily mean temperatures differed among sites according to their longitudinal position, following a general pattern of warming and increased stability towards the river mouth

(Fig. 4.2). For instance, the closest site to the headwater of the Chapecó River (9.1 km) had an average daily mean temperature of 12.1 °C during the winter (min of 8.7 °C) and of 19.5 °C during the summer (max of 20.8 °C). On the other side of the spectrum, the most downstream site (212 km) had an average daily mean temperature of 17.0 °C during the winter (min of 13.7 °C) and of 26.3 °C during the summer (max of 28.1 °C). The variability of daily temperatures also varied between these two sites, with the range of daily temperatures per month being on average a 0.6 °C (SD ± 0.3) larger for the headwater site. The frequency of cold-water events (i.e. < 12 °C) were concentrated in the winter period, being more common in headwater sites and becoming gradually rare further downstream (Fig. 4.2).

The effects of SHPs on downstream water temperatures were highly variable (Fig. 4.3), ranging from almost negligible changes (e.g. SHP Tonet) to pronounced changes in both magnitude and variability (e.g. SHPs Flor do Mato). Four SHPs modified the magnitudes of daily thermal regimes, three of them by warmer downstream releases. On average, the SHPs Flor do Mato, Contestado and Passos Maia increased daily temperatures by 2.6, 0.9 and 0.7 °C, respectively (Fig. 4.4, Table 4.1). Coronel Araújo was the only SHP with cold-water releases, with the site downstream being on average 1.0 °C colder than the upstream (Fig. 4.4). The dewatering section of the SHP Contestado was on average 1.1 °C warmer than the upstream site, which is even warmer than the records for the downstream site (Fig. 4.4). However, we did not detect major changes in magnitude at the dewatering sections of other SHPs.

Differences between upstream and downstream thermal regimes were not homogeneous throughout the year for most SHPs, with larger differences happening in the transition between the fall and the winter, and during the summer (Fig. 4.4). For instance, daily temperatures downstream of SHP Flor do Mato were 3.0 °C higher than upstream during 118 days of the year, of which 89 were between April and August. The largest difference in daily temperatures was recorded in Flor do Mato (6.8 °C) in May, when the mean temperature of the upstream site dropped to 9.0 °C and the downstream site maintained in 15.8 °C. Water temperatures in most unregulated sites across the basin had higher variability during this exact same period (Fig. 4.2; Fig. 4.3), indicating a possible role of storage and surface area on thermal stabilization. However, no clear pattern emerged in the relationships between reservoir area or dewatering distance and the cross-correlation between upstream and downstream thermographs (Fig. 4.5).

4.4 DISCUSSION

Overall, our results indicate that SHPs may have important effects on thermal regimes of the Chapecó River Basin, but these effects are quite variable across time and space. Thermal regimes of the Chapecó Basins are characterized by two marked periods, one warmer and more stable in the summer, and other colder and highly variable during the winter. In addition, we detected a general pattern of increase in water temperature and a decrease in variability in the downstream direction.

We identified variable effects of the SHPs on downstream water temperature regimes, which included dams with almost negligible effects and dams with releases that are on average 2.6 °C warmer than upstream. Warmer surface releases were the most prevalent effect of the SHPs, consistent with findings from a growing body of scientific literature on the ecological effects of small dams (Lessard & Hayes, 2003). The dewatering section of SHP Contestado was significantly warmer than both upstream and downstream sites, suggesting that diversion can be an important mechanism of stream warming. In fact, a study in the Swiss Alps found that water diversion can increase stream temperatures by 3.7 (± 0.9) °C during the summer, with magnitudes of the effect of different diversions schemes being highly dependent on their slope (Meier et al., 2003).

Although water diversion can be an important driver of temperature changes (Jumani et al., 2018; Meier et al., 2003), our results suggest that water storage and longitudinal position should be the most important factors affecting the release temperatures in the Chapecó Basin. Direct solar radiation is the most important component of the total energy flux in rivers (Caissie, 2006), and reservoirs with larger surface areas have increased exposure to it – particularly during the summer. For instance, the SHPs Flor do Mato and Tonet had very distinct effects on their downstream releases, although they both operate closer to the headwaters. The SHP Tonet is a typical 'run-of-river' scheme with negligible storage, while SHP Flor do Mato has a reservoir of almost 6 km^2 (Table 4.1).

In addition, differences between downstream and upstream thermographs tended to be higher during the transition between the fall and the winter, indicating that water storage slows seasonal decreases in downstream temperatures. However, the longitudinal position of the SHP determines the amount of cold-water influxes it receives, with dams close to the headwaters being more susceptible to effects of water storage on downstream thermal regimes. This would explain why thermographs in SHP Rondinha varied less than other SHPs located closer to headwaters. These findings match with early predictions by the Serial Discontinuity Concept (Ward & Stanford, 1983), but regarding warm-water releases in headwaters.

The potential cumulative effects of SHPs on basin-wide ecosystem processes is an issue of high concern in the ecological literature (Couto & Olden, 2018; Kibler & Tullos, 2013; Lange et al., 2018). Our results demonstrate that modifications of thermal regimes by SHPs can be quite high in some cases, shedding light on a potential path of cumulative ecological effects of aggregates of small dams. The effects of warm-water releases by small dams have been reported to persist for a few kilometers downstream of small dam outlets (Jumani et al., 2018; Lessard &

Hayes, 2003). Therefore, modifications in thermal regimes by multiple SHPs have the potential of promoting major changes in magnitude, variation and synchrony of water temperatures at the basin level. Such changes were already observed for large dam cascades (Long et al., 2019), but not yet for studies on small dams.

4.5 CONCLUSIONS

The sheer number of SHPs in developing countries has the potential of promoting major changes in thermal regimes in tropical and sub-tropical rivers. Our results indicate that the effects of SHPs on stream temperatures are quite variable and can be considerably high for SHPs with high water storage capacity. Although information on the effects of altered temperature regimes on the biota is scarce for tropical and sub-tropical rivers, there is evidence that increased or decreased water temperatures can change freshwater community composition and fish spawning activities (Jumani et al., 2018; Sato, Bazzoli, Rizzo, Boschi, & Miranda, 2005). Therefore, understanding the effects of modified temperature regimes on freshwater organisms and river ecosystems is a necessary next step to inform better management practices and regulations for SHPs, especially in basins where SHPs have been implemented in high densities.

4.6 FIGURES

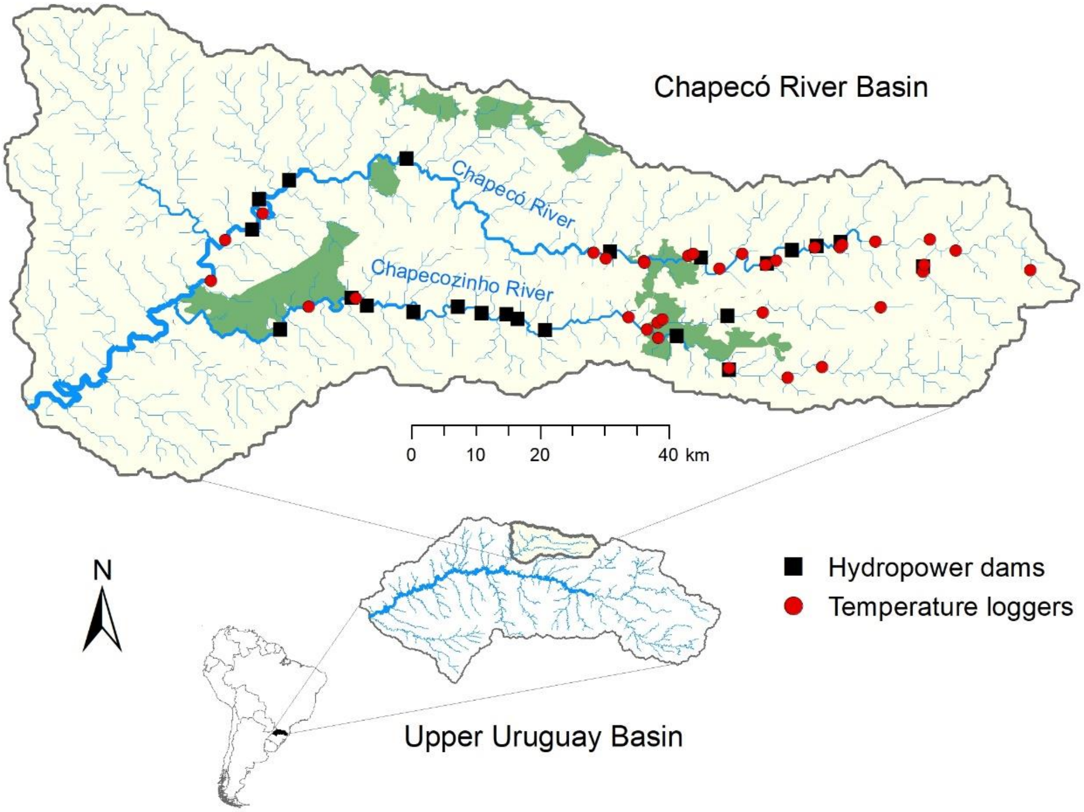

Figure 4.1. Map depicting the 23 hydropower plants (black symbols) operating in the mainstems of Chapecó and Chapecozinho Rivers (22 of them are classified as SHPs), and the location of the 28 temperature loggers placed and recovered across the Basin (red circles). Green polygons represent the protected areas and lands owned by indigenous groups.

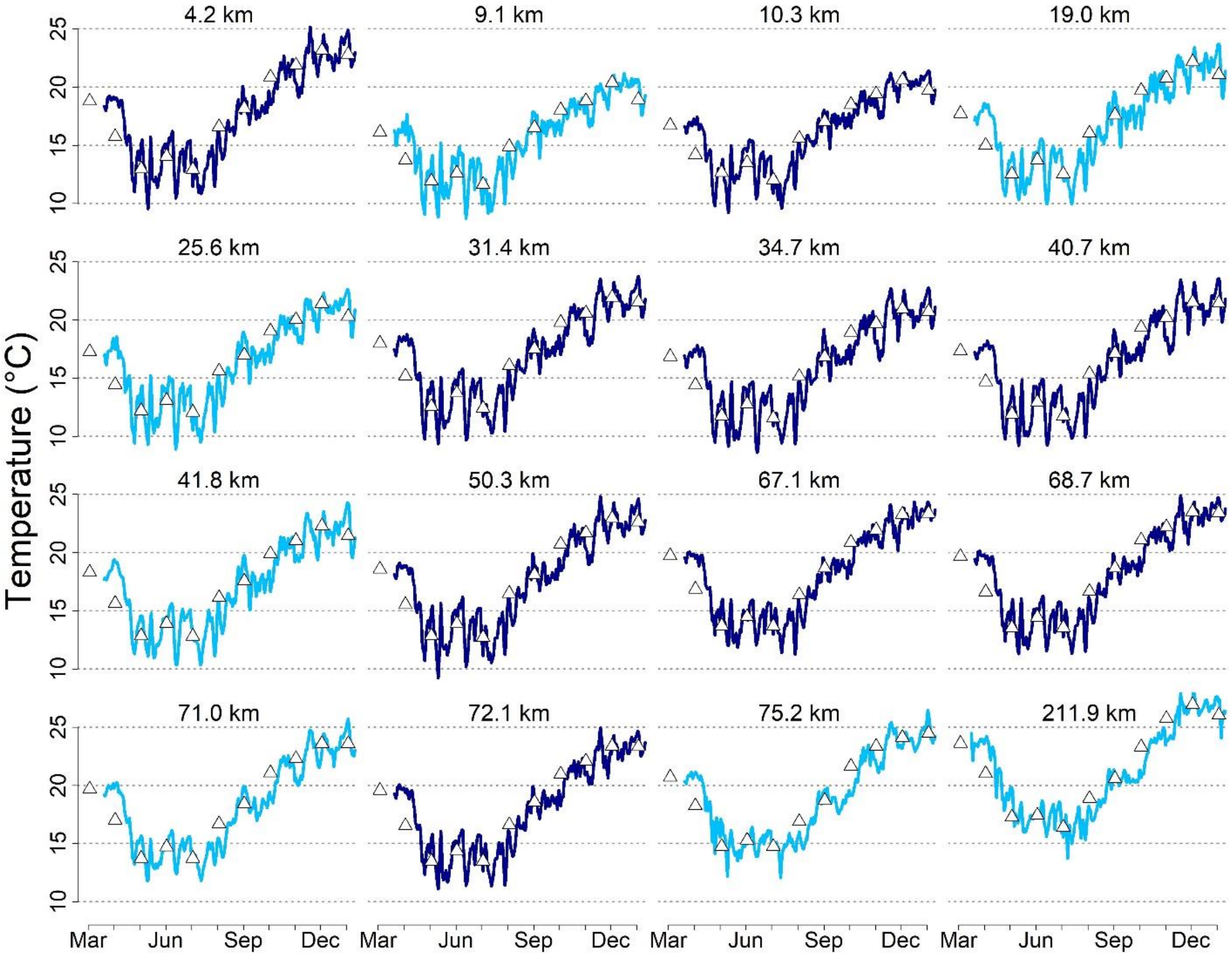

Figure 4.2. Daily mean water temperatures registered in unregulated sites from March 2018 to February 2019 in the Chapecó River Basin. Each panel represents and depicts thermographs of unregulated sites with their respective distance to the headwater in the Chpecó (light blue) and Chapecozinho (dark blue) Rivers. White triangles represent monthly averages.

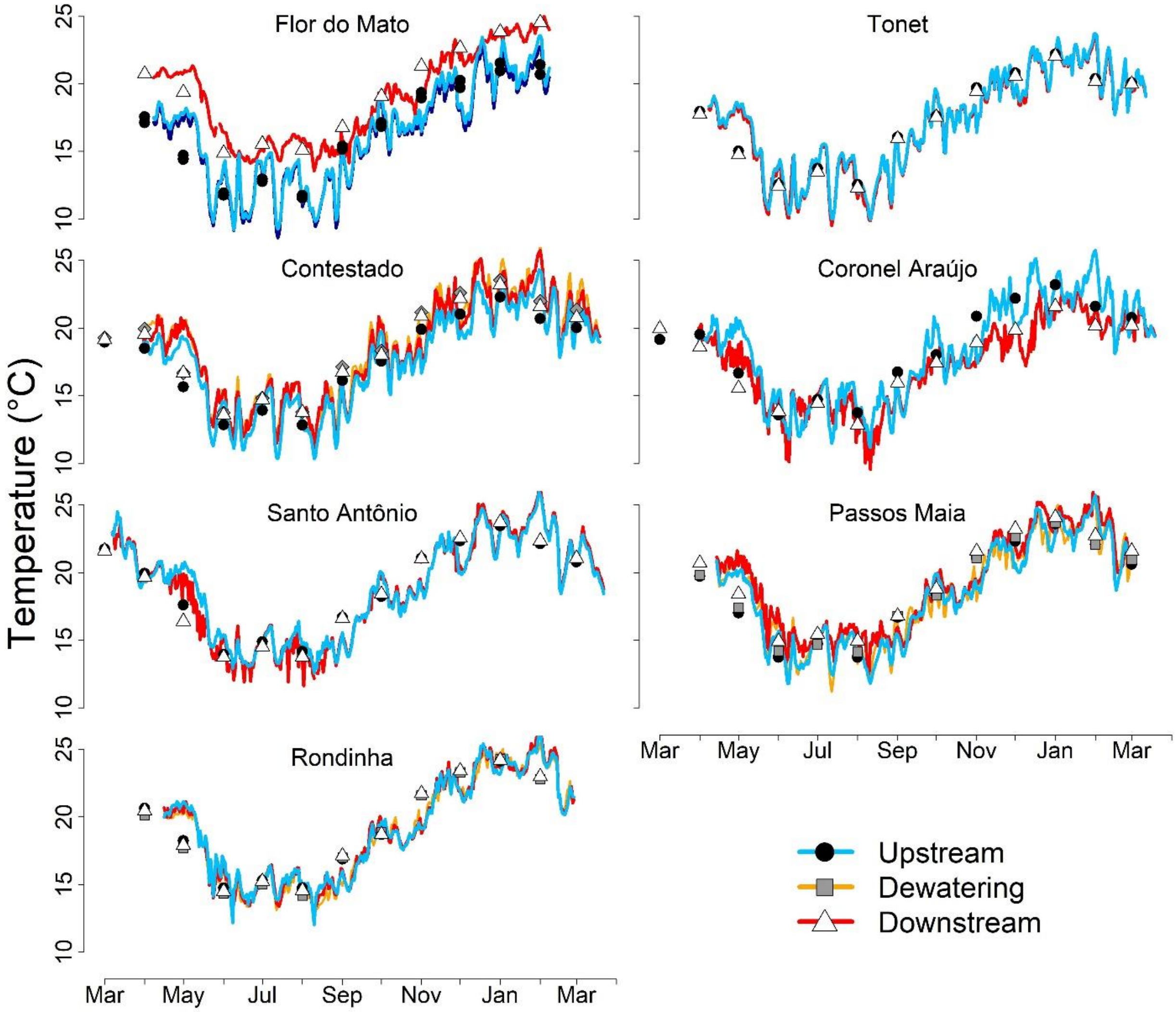

Figure 4.3. Daily mean water temperatures registered near seven SHPs from March 2018 to March 2019 in the Chapecó River Basin. Each panel represents a SHP and depicts thermographs of sites immediately upstream (blue) and downstream (red) of the dam. Thermographs of sites in dewatering sections (orange) are also provided for the SHPs Contestado, Passos Maia, and Rondinha. Moreover, thermographs of two upstream sites were provided for SHP Flor do Mato. Symbols represent monthly averages for upstream (black circle), dewatering (gray square) and downstream sites (white triangles).

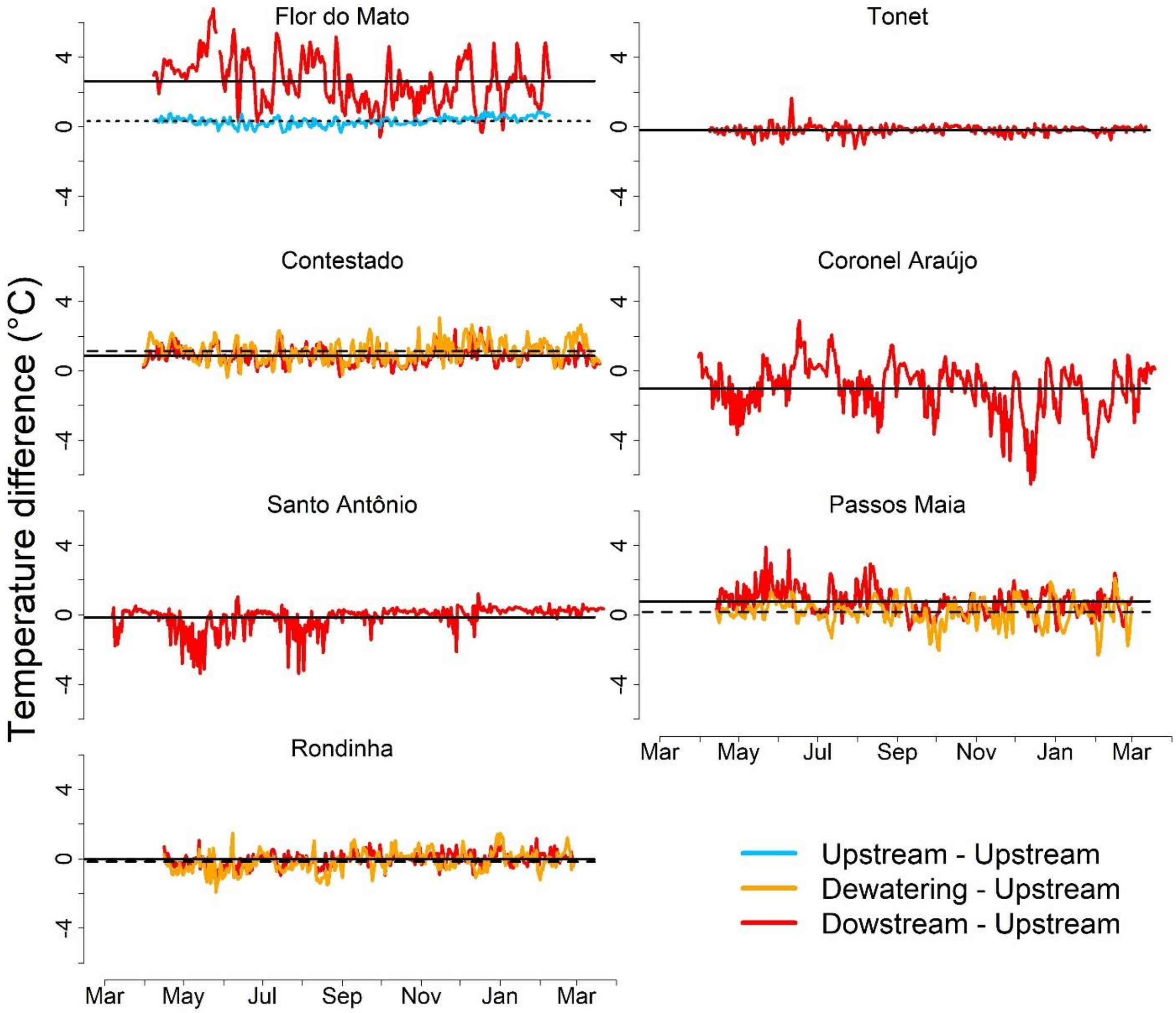

Figure 4.4. Differences in daily mean water temperatures between downstream and upstream sites for seven SHPs of the Chapecó River Basin. Each panel represents a SHP and depicts differences in thermographs between downstream and upstream sites (red), and between dewatering and upstream sites (orange). The difference between the upstream site and another site further upstream is also provided for the SHP Flor do Mato (blue). Mean differences for these three site-comparisons are represented by continuous, dashed and dotted lines, respectively.

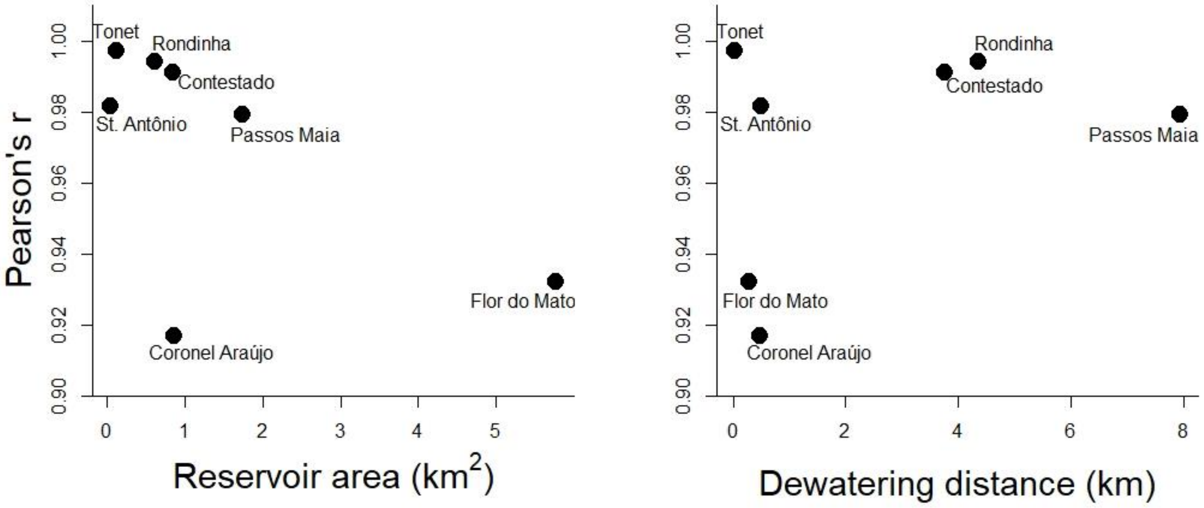

Figure 4.5. Relationships between thermographs synchrony (i.e. Pearson correlations) and reservoir area and dewatering distance for six SHPs of the Chapecó River Basin. Correlations were calculated between thermographs located upstream and downstream of each SHP.

4.7 TABLES

Table 4. 1. List of attributes of the seven SHPs of the Chapecó River Basin that have upstream and downstream thermographic records. The SHPs Contestado, Passos Maia and Rondinha also include thermographs in the dewatering section.

Name	**River**	**Capacity (MW)**	**Distance to headwater (km)**	**Dam height (m)**	**Dam length (m)**	**Reservoir area (km²)**	**Volume (hm³)**	**Dewatering distance (km)**	**Opening date**
Flor do Mato	Chapecozinho	4.8	50.9	13.2	267.3	5.77	43.6	0.28	12/6/1946
Tonet	Chapeco	0.8	24.0	3.5	30	0.13	0.01	0.03	5/1/1975
Contestado	Chapeco	5.6	47.9	3	234.29	0.8526	1.4	3.76	11/1/2007
Coronel Araújo	Chapeco	5.8	53.9	5	183.3	0.8556	1.4	0.47	11/1/2007
Santo Antônio	Chapeco	6.2	64.4	2	90	0.05	0.09	0.5	1/1/1963
Passos Maia	Chapeco	25.0	82.8	28	199	1.75	13.82	7.95	2/18/2012
Rondinha	Chapeco	9.6	98.1	17	172	0.62	3.02	4.35	6/4/2014

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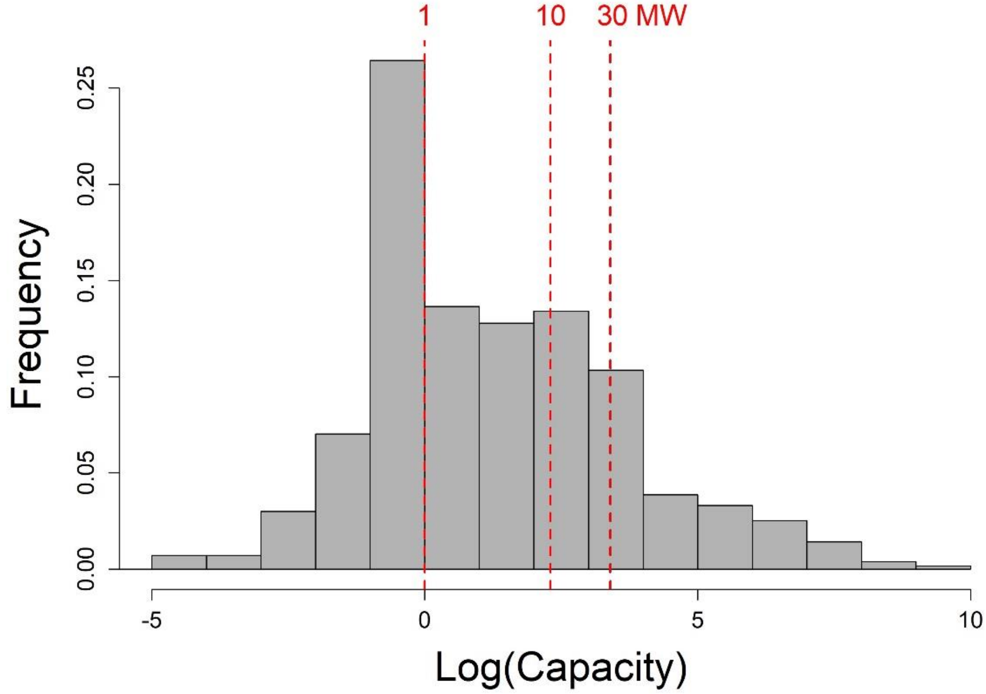
APPENDIX A



Appendix A, Fig. S1. Frequency of size distributions (MW) of hydropower dams in Brazil (n = 1,267). Most hydropower dams in Brazil are classified as small (median = 2.4 MW), in which 31% have capacities below 1 MW, 68% below 10 MW, and 85% below 30 MW (ANEEL 2016).

APPENDIX B

Appendix B, Table S1. Full list of migratory fish species from Brazil compiled for this study and their respective vulnerability (i.e. percentage of future DCI loss) to river fragmentation by SHPs and LHPs.

Order	**Family**	**Species**	**% Change SHP**	**% Change LHP**
Characiformes	Anostomidae	Abramites hypselonotus (Günther, 1868)	NA	NA
Characiformes	Anostomidae	Anostomoides laticeps (Eigenmann, 1912)	NA	NA
Characiformes	Anostomidae	Anostomoides passionis Dos Santos & Zuanon, 2006	NA	NA
Characiformes	Anostomidae	Anostomus anostomus (Linnaeus, 1758)	NA	NA
Characiformes	Anostomidae	Anostomus ternetzi Fernández-Yépez, 1949	NA	NA
Characiformes	Anostomidae	Gnathodolus bidens Myers, 1927	NA	NA
Characiformes	Anostomidae	Hypomasticus garmani (Borodin, 1919)	-22.61	-15.3
Characiformes	Anostomidae	Hypomasticus julii (Santos, Jegu & Lima, 1996)	NA	NA
Characiformes	Anostomidae	Hypomasticus lineomaculatus Birindelli, Peixoto, Wosiacki & Briski, 2013	NA	NA
Characiformes	Anostomidae	Hypomasticus megalepis (Günther, 1863)	NA	NA
Characiformes	Anostomidae	Hypomasticus mormyrops (Steindachner, 1875)	-48.68	-19.2
Characiformes	Anostomidae	Hypomasticus pachycheilus (Britski, 1976)	NA	NA
Characiformes	Anostomidae	Hypomasticus thayeri (Borodin, 1929)	-38.7	-5.7
Characiformes	Anostomidae	Laemolyta fernandezi Myers, 1950	NA	NA
Characiformes	Anostomidae	Laemolyta garmani (Borodin, 1931)	NA	NA
Characiformes	Anostomidae	Laemolyta proxima (Garman, 1890)	NA	NA
Characiformes	Anostomidae	Laemolyta taeniata (Kner, 1858)	NA	NA
Characiformes	Anostomidae	Leporellus retropinnis (Eigenmann, 1922)	-18.3	0
Characiformes	Anostomidae	Leporellus vittatus (Valenciennes, 1850)	-30.05	-12.3
Characiformes	Anostomidae	Leporinus affinis Günther, 1864	NA	NA
Characiformes	Anostomidae	Leporinus agassizii Steindachner, 1876	NA	NA
Characiformes	Anostomidae	Leporinus amae Godoy, 1980	-36.94	-7.86
Characiformes	Anostomidae	Leporinus amazonicus Dos Santos & Zuanon, 2008	NA	NA
Characiformes	Anostomidae	Leporinus amblyrhynchus Garavello & Britski, 1987	-33.38	-13.29
Characiformes	Anostomidae	Leporinus aripuanaensis Garavello & Santos, 1981	NA	NA
Characiformes	Anostomidae	Leporinus bahiensis Steindachner, 1875	-29.18	-8.33
Characiformes	Anostomidae	Leporinus bistriatus Britski, 1997	NA	NA
Characiformes	Anostomidae	Leporinus bleheri Géry, 1999	NA	NA
Characiformes	Anostomidae	Leporinus brinco Birindelli, Britski & Garavello, 2013	-31.84	0
Characiformes	Anostomidae	Leporinus britskii Feitosa, Santos & Birindelli, 2011	NA	NA

Characiformes	Anostomidae	Leporinus brunneus Myers, 1950	NA	NA
Characiformes	Anostomidae	Leporinus conirostris Steindachner, 1875	-40.27	-13.6
Characiformes	Anostomidae	Leporinus copelandii Steindachner, 1875	-39.31	-14.9
Characiformes	Anostomidae	Leporinus cylindriformis Borodin, 1929	NA	NA
Characiformes	Anostomidae	Leporinus desmotes Fowler, 1914	NA	NA
Characiformes	Anostomidae	Leporinus elongatus Valenciennes, 1850	-16.17	-12.33
Characiformes	Anostomidae	Leporinus falcipinnis Mahnert, Géry & Muller, 1997	NA	NA
Characiformes	Anostomidae	Leporinus fasciatus (Bloch, 1794)	NA	NA
Characiformes	Anostomidae	Leporinus friderici (Bloch, 1794)	-37.74	-14.18
Characiformes	Anostomidae	Leporinus geminis Garavello & Santos, 2009	NA	NA
Characiformes	Anostomidae	Leporinus gomesi Garavello & Santos, 1981	NA	NA
Characiformes	Anostomidae	Leporinus granti Eigenmann, 1912	NA	NA
Characiformes	Anostomidae	Leporinus guttatus Birindelli & Britski, 2009	-21.55	0
Characiformes	Anostomidae	Leporinus jamesi Garman, 1929	NA	NA
Characiformes	Anostomidae	Leporinus klausewitzi Géry, 1960	NA	NA
Characiformes	Anostomidae	Leporinus lacustris Amaral Campos, 1945	-15.42	-11.83
Characiformes	Anostomidae	Leporinus macrocephalus Garavello & Britski, 1988	-33.9	-7.91
Characiformes	Anostomidae	Leporinus maculatus Müller & Troschel, 1844	NA	NA
Characiformes	Anostomidae	Leporinus marcgravii Lütken, 1875	0	-48.32
Characiformes	Anostomidae	Leporinus melanopleura Günther, 1864	NA	NA
Characiformes	Anostomidae	Leporinus melanopleurodes Birindelli, Britski & Garavello, 2013	-39.59	-8.36
Characiformes	Anostomidae	Leporinus melanostictus Norman, 1926	NA	NA
Characiformes	Anostomidae	Leporinus microphthalmus Garavello, 1989	-32.56	-13.77
Characiformes	Anostomidae	Leporinus microphysus Birindelli & Britski, 2013	0	-33.2
Characiformes	Anostomidae	Leporinus moralesi Fowler, 1942	NA	NA
Characiformes	Anostomidae	Leporinus nattereri Steindachner, 1876	NA	NA
Characiformes	Anostomidae	Leporinus obtusidens (Valenciennes, 1837)	-24.67	-15.96
Characiformes	Anostomidae	Leporinus octofasciatus Steindachner, 1915	-27.44	-16.18
Characiformes	Anostomidae	Leporinus octomaculatus Britski & Garavello, 1993	NA	NA
Characiformes	Anostomidae	Leporinus ortomaculatus Garavello, 2000	NA	NA
Characiformes	Anostomidae	Leporinus parae Eigenmann, 1908	NA	NA
Characiformes	Anostomidae	Leporinus paranensis Garavello & Britski, 1987	-20.39	-16.84
Characiformes	Anostomidae	Leporinus parvulus Birindelli, Britski & Lima, 2013	0	-40.41
Characiformes	Anostomidae	Leporinus piau Fowler, 1941	-7.39	-11.29
Characiformes	Anostomidae	Leporinus piavussu Britski, Birindelli & Garavello, 2012	-24.23	-10.9
Characiformes	Anostomidae	Leporinus pitingai Santos & Jégu, 1996	NA	NA
Characiformes	Anostomidae	Leporinus reinhardti Lütken, 1875	-9.11	-11.42
Characiformes	Anostomidae	Leporinus reticulatus Britski & Garavello, 1993	NA	NA
Characiformes	Anostomidae	Leporinus santosi Britski & Birindelli, 2013	NA	NA
Characiformes	Anostomidae	Leporinus sexstriatus Britski & Garavello, 1980	NA	NA
Characiformes	Anostomidae	Leporinus steindachneri Eigenmann, 1907	-32.25	-11.46
Characiformes	Anostomidae	Leporinus striatus Kner, 1858	-24.8	-14.25

Characiformes	Anostomidae	Leporinus taeniatus Lütken, 1875	-11.98	-10.15
Characiformes	Anostomidae	Leporinus taeniofasciatus Britski, 1997	NA	NA
Characiformes	Anostomidae	Leporinus tigrinus Borodin, 1929	NA	NA
Characiformes	Anostomidae	Leporinus trifasciatus Steindachner, 1876	NA	NA
Characiformes	Anostomidae	Leporinus trimaculatus Garavello & Santos, 1992	NA	NA
Characiformes	Anostomidae	Leporinus tristriatus Birindelli & Britski, 2013	-13.38	-11.07
Characiformes	Anostomidae	Leporinus unitaeniatus Garavello & Santos, 2009	NA	NA
Characiformes	Anostomidae	Leporinus vanzoi Britski & Garavello, 2005	NA	NA
Characiformes	Anostomidae	Leporinus venerei Britski & Birindelli, 2008	NA	NA
Characiformes	Anostomidae	Petulanos intermedius (Winterbotton, 1980)	NA	NA
Characiformes	Anostomidae	Petulanos plicatus (Eigenmann, 1912)	NA	NA
Characiformes	Anostomidae	Pseudanos gracilis (Kner, 1858)	NA	NA
Characiformes	Anostomidae	Pseudanos trimaculatus (Kner, 1858)	NA	NA
Characiformes	Anostomidae	Pseudanos varii Birindelli, Lima & Britski, 2012	-1.24	0
Characiformes	Anostomidae	Pseudanos winterbottomi Sidlauskas & Santos, 2005	NA	NA
Characiformes	Anostomidae	Rhytiodus argenteofuscus Kner, 1858	NA	NA
Characiformes	Anostomidae	Rhytiodus lauzannei Géry, 1987	NA	NA
Characiformes	Anostomidae	Rhytiodus microlepis Kner, 1858	NA	NA
Characiformes	Anostomidae	Sartor elongatus Santos & Jégu, 1987	NA	NA
Characiformes	Anostomidae	Sartor respectus Myers & Carvalho, 1959	NA	NA
Characiformes	Anostomidae	Sartor tucuruiense Santos & Jégu, 1987	-15.26	-22.47
Characiformes	Anostomidae	Schizodon altoparanae Garavello & Britski, 1990	-31.87	-17.62
Characiformes	Anostomidae	Schizodon australis Garavello, 1994	NA	NA
Characiformes	Anostomidae	Schizodon borellii (Boulenger, 1900)	-21.59	-11.92
Characiformes	Anostomidae	Schizodon dissimilis (Garman, 1890)	NA	NA
Characiformes	Anostomidae	Schizodon fasciatus Spix & Agassiz, 1829	NA	NA
Characiformes	Anostomidae	Schizodon intermedius Garavello & Britski, 1990	-13.19	-12.52
Characiformes	Anostomidae	Schizodon isognathus Kner, 1858	-30.62	-13.47
Characiformes	Anostomidae	Schizodon jacuiensis Bergmann, 1988	-2.25	-1.92
Characiformes	Anostomidae	Schizodon knerii (Steindachner, 1875)	-10.95	-0.14
Characiformes	Anostomidae	Schizodon nasutus Kner, 1858	-28.4	-15.49
Characiformes	Anostomidae	Schizodon platae (Garman, 1890)	-43.25	-21.92
Characiformes	Anostomidae	Schizodon rostratus (Borodin, 1931)	0	-29.37
Characiformes	Anostomidae	Schizodon vittatus (Valenciennes, 1850)	NA	NA
Characiformes	Anostomidae	Synaptolaemus latofasciatus (Steindachner, 1910)	NA	NA
Characiformes	Bryconidae	Brycon amazonicus (Spix & Agassiz 1829)	NA	NA
Characiformes	Bryconidae	Brycon devillei (Castelnau, 1855)	-36.53	-2.73
Characiformes	Bryconidae	Brycon falcatus Müller & Troschel, 1844	-14.71	-26.6
Characiformes	Bryconidae	Brycon ferox Steindachner, 1877	-44.38	-7.06
Characiformes	Bryconidae	Brycon gouldingi Lima, 2004	-17.33	-16.88
Characiformes	Bryconidae	Brycon hilarii (Valenciennes, 1850)	-24.38	-6.39
Characiformes	Bryconidae	Brycon insignis Steindachner, 1877	-48.65	-5.94
Characiformes	Bryconidae	Brycon melanopterus (Cope, 1872)	0	-1.57

Characiformes	Bryconidae	Brycon nattereri (Günther, 1864)	-37.08	-7.94
Characiformes	Bryconidae	Brycon opalinus (Cuvier, 1819)	-54.91	-17.06
Characiformes	Bryconidae	Brycon orbignyanus (Valenciennes, 1850)	-28.2	-11.15
Characiformes	Bryconidae	Brycon orthotaenia (Günther, 1864)	-17.27	-0.14
Characiformes	Bryconidae	Brycon pesu Müller & Troschel, 1845	-21.95	-20.5
Characiformes	Bryconidae	Brycon polylepis Mosco Morales, 1988	-1.22	-21.15
Characiformes	Bryconidae	Brycon vermelha Lima & Castro, 2000	-58.8	0
Characiformes	Bryconidae	Salminus brasiliensis (Cuvier, 1816)	-19.7	-18.15
Characiformes	Bryconidae	Salminus franciscanus Lima & Britski, 2007	-36.01	-4.38
Characiformes	Bryconidae	Salminus hilarii Valenciennes, 1850	-33.53	-4.62
Characiformes	Curimatidae	Ctenocheirodon pristis Malabarba & Jerep, 2012	-43.96	-0.88
Characiformes	Curimatidae	Curimata acutirostris Vari & Reis, 1995	NA	NA
Characiformes	Curimatidae	Curimata aspera (Günther, 1868)	NA	NA
Characiformes	Curimatidae	Curimata cisandina (Allen, 1942)	NA	NA
Characiformes	Curimatidae	Curimata cyprinoides (Linnaeus, 1766)	NA	NA
Characiformes	Curimatidae	Curimata inornata Vari, 1989	NA	NA
Characiformes	Curimatidae	Curimata knerii Steindachner, 1876	NA	NA
Characiformes	Curimatidae	Curimata macrops Eigenmann & Eigenmann, 1889	-19.12	-20.71
Characiformes	Curimatidae	Curimata ocellata Eigenmann & Eigenmann, 1889	NA	NA
Characiformes	Curimatidae	Curimata roseni Vari, 1989	NA	NA
Characiformes	Curimatidae	Curimata vittata (Kner, 1858)	NA	NA
Characiformes	Curimatidae	Curimatella alburna (Müller & Troschel, 1844)	NA	NA
Characiformes	Curimatidae	Curimatella dorsalis (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Curimatella immaculata (Fernández-Yépez, 1948)	NA	NA
Characiformes	Curimatidae	Curimatella lepidura (Eigenmann & Eigenmann, 1889)	-1.14	-23.28
Characiformes	Curimatidae	Curimatella meyeri (Steindachner, 1882)	NA	NA
Characiformes	Curimatidae	Curimatopsis crypticus Vari,1982	NA	NA
Characiformes	Curimatidae	Curimatopsis evelynae Géry, 1964	NA	NA
Characiformes	Curimatidae	Curimatopsis macrolepis (Steindachner, 1876)	NA	NA
Characiformes	Curimatidae	Curimatopsis microlepis Eigenmann & Eigenmann, 1889	NA	NA
Characiformes	Curimatidae	Curimatopsis myersi Vari, 1982	-30.35	-13.47
Characiformes	Curimatidae	Cyphocharax abramoides (Kner, 1858)	NA	NA
Characiformes	Curimatidae	Cyphocharax aninha Wosiacki & da Silva Miranda, 2014	NA	NA
Characiformes	Curimatidae	Cyphocharax festivus Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax gangamon Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax gilbert (Quoy & Gaimard, 1824)	-33.65	-9.94
Characiformes	Curimatidae	Cyphocharax gillii (Eigenmann & Kennedy, 1903)	-40.09	-10.8
Characiformes	Curimatidae	Cyphocharax gouldingi Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax helleri (Steindachner, 1910)	NA	NA
Characiformes	Curimatidae	Cyphocharax leucostictus (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Cyphocharax mestomyllon Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax modestus (Fernández-Yépez, 1948)	-26.8	-14.18
Characiformes	Curimatidae	Cyphocharax multilineatus (Myers, 1927)	NA	NA

Characiformes	Curimatidae	Cyphocharax nagelii (Steindachner, 1881)	-13.49	-20.87
Characiformes	Curimatidae	Cyphocharax nigripinnis Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax notatus (Steindachner, 1908)	NA	NA
Characiformes	Curimatidae	Cyphocharax pantostictos Vari & Barriga S., 1990	NA	NA
Characiformes	Curimatidae	Cyphocharax pinnilepis Vari, Zanata & Camelier, 2010	-19.62	-8.32
Characiformes	Curimatidae	Cyphocharax platanus (Günther, 1880)	NA	NA
Characiformes	Curimatidae	Cyphocharax plumbeus (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Cyphocharax saladensis (Meinken, 1933)	-6.2	-11
Characiformes	Curimatidae	Cyphocharax santacatarinae (Fernández-Yépez, 1948)	-27.66	-14.16
Characiformes	Curimatidae	Cyphocharax signatus Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax spilotus (Vari, 1987)	-14.7	-14.09
Characiformes	Curimatidae	Cyphocharax spiluropsis (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Cyphocharax spilurus (Günther, 1864)	NA	NA
Characiformes	Curimatidae	Cyphocharax stilbolepis Vari, 1992	NA	NA
Characiformes	Curimatidae	Cyphocharax vanderi (Britski, 1980)	-17.99	-5.64
Characiformes	Curimatidae	Cyphocharax voga (Hensel, 1870)	-29.36	-14.26
Characiformes	Curimatidae	Potamorhina altamazonica (Cope, 1878)	NA	NA
Characiformes	Curimatidae	Potamorhina latior (Spix & Agassiz, 1829)	NA	NA
Characiformes	Curimatidae	Potamorhina pristigaster (Steindachner ,1876)	NA	NA
Characiformes	Curimatidae	Potamorhina squamoralevis (Braga & Azpelicueta, 1983)	-16.55	-15.57
Characiformes	Curimatidae	Psectrogaster amazonica Eigenmann & Eigenmann, 1889	NA	NA
Characiformes	Curimatidae	Psectrogaster ciliata (Müller & Troschel, 1844)	NA	NA
Characiformes	Curimatidae	Psectrogaster curviventris Eigenmann & Kennedy, 1903	-17.97	-15.19
Characiformes	Curimatidae	Psectrogaster essequibensis (Günther, 1864)	NA	NA
Characiformes	Curimatidae	Psectrogaster falcata (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Psectrogaster rhomboides Eigenmann & Eigenmann, 1889	0	-2.15
Characiformes	Curimatidae	Psectrogaster rutiloides (Kner, 1858)	NA	NA
Characiformes	Curimatidae	Psectrogaster saguiru (Fowler, 1941)	NA	NA
Characiformes	Curimatidae	Steindachnerina amazonica (Steindachner, 1911)	NA	NA
Characiformes	Curimatidae	Steindachnerina bimaculata (Steindachner, 1876)	NA	NA
Characiformes	Curimatidae	Steindachnerina biornata (Braga & Azpelicueta, 1987)	-35.79	-12.44
Characiformes	Curimatidae	Steindachnerina brevipinna (Eigenmann & Eigenmann, 1889)	-31.08	-8.64
Characiformes	Curimatidae	Steindachnerina conspersa (Holmberg, 1891)	-9.54	0
Characiformes	Curimatidae	Steindachnerina corumbae Pavanelli & Britski, 1999	-29.61	-11.2
Characiformes	Curimatidae	Steindachnerina dobula (Günther, 1868)	NA	NA
Characiformes	Curimatidae	Steindachnerina elegans (Steindachner, 1875)	NA	NA
Characiformes	Curimatidae	Steindachnerina fasciata (Vari & Géry, 1985)	NA	NA
Characiformes	Curimatidae	Steindachnerina gracilis Vari & Williams Vari, 1989	NA	NA
Characiformes	Curimatidae	Steindachnerina guentheri (Eigenmann & Eigenmann, 1889)	NA	NA
Characiformes	Curimatidae	Steindachnerina hypostoma (Boulenger, 1887)	NA	NA
Characiformes	Curimatidae	Steindachnerina insculpta (Fernández-Yépez, 1948)	-27.26	-12.4
Characiformes	Curimatidae	Steindachnerina leucisca (Günther, 1868)	NA	NA

Characiformes	Curimatidae	Steindachnerina nigrotaenia (Boulenger, 1902)	-24.35	0
Characiformes	Curimatidae	Steindachnerina notograptos Lucinda & Vari, 2009	NA	NA
Characiformes	Curimatidae	Steindachnerina notonota (Miranda Ribeiro, 1937)	-37.99	0
Characiformes	Curimatidae	Steindachnerina planiventris Vari & Williams Vari, 1989	NA	NA
Characiformes	Curimatidae	Steindachnerina quasimodoi Vari & Williams Vari, 1989	NA	NA
Characiformes	Curimatidae	Steindachnerina seriata Netto-Ferreira & Vari, 2011	NA	NA
Characiformes	Cynodontidae	Rhaphiodon vulpinus Agassiz, 1829	-17.79	-22.86
Characiformes	Hemiodontidae	Anodus elongatus Agassiz, 1829	NA	NA
Characiformes	Hemiodontidae	Anodus orinocensis (Steindachner, 1887)	NA	NA
Characiformes	Hemiodontidae	Argonectes longiceps (Kner, 1858)	NA	NA
Characiformes	Hemiodontidae	Argonectes robertsi Langeani, 1999	NA	NA
Characiformes	Hemiodontidae	Bivibranchia bimaculata Vari, 1985	NA	NA
Characiformes	Hemiodontidae	Bivibranchia fowleri (Steindachner, 1908)	NA	NA
Characiformes	Hemiodontidae	Bivibranchia notata Vari & Goulding, 1985	NA	NA
Characiformes	Hemiodontidae	Bivibranchia velox (Eigenmann & Myers, 1927)	NA	NA
Characiformes	Hemiodontidae	Hemiodus argenteus Pellegrin, 1908	NA	NA
Characiformes	Hemiodontidae	Hemiodus atranalis (Fowler, 1940)	NA	NA
Characiformes	Hemiodontidae	Hemiodus goeldii Steindachner, 1908	NA	NA
Characiformes	Hemiodontidae	Hemiodus gracilis Günther, 1864	NA	NA
Characiformes	Hemiodontidae	Hemiodus immaculatus Kner, 1858	NA	NA
Characiformes	Hemiodontidae	Hemiodus iratapuru Langeani & Moreira, 2013	NA	NA
Characiformes	Hemiodontidae	Hemiodus jatuarana Langeani, 2004	NA	NA
Characiformes	Hemiodontidae	Hemiodus langeanii Beltrão & Zuanon, 2012	NA	NA
Characiformes	Hemiodontidae	Hemiodus microlepis Kner, 1858	NA	NA
Characiformes	Hemiodontidae	Hemiodus orthonops Eigenmann & Kennedy, 1903	-16.55	-15.57
Characiformes	Hemiodontidae	Hemiodus parnaguae Eigenmann & Henn, 1916	-9.34	-18.33
Characiformes	Hemiodontidae	Hemiodus quadrimaculatus Pellegrin, 1908	NA	NA
Characiformes	Hemiodontidae	Hemiodus semitaeniatus (Kner, 1858)	NA	NA
Characiformes	Hemiodontidae	Hemiodus sterni (Géry, 1964)	NA	NA
Characiformes	Hemiodontidae	Hemiodus ternetzi Myers, 1927	NA	NA
Characiformes	Hemiodontidae	Hemiodus thayeria Böhlke, 1955	NA	NA
Characiformes	Hemiodontidae	Hemiodus tocantinensis Langeani, 1999	NA	NA
Characiformes	Hemiodontidae	Hemiodus unimaculatus (Bloch, 1794)	NA	NA
Characiformes	Hemiodontidae	Hemiodus vorderwinkleri (Géry, 1964)	NA	NA
Characiformes	Hemiodontidae	Micromischodus sugillatus Roberts, 1971	NA	NA
Characiformes	Prochilodontidae	Prochilodus argenteus Spix & Agassiz, 1829	-8.64	-10.32
Characiformes	Prochilodontidae	Prochilodus brevis Steindachner, 1875	-36.29	0
Characiformes	Prochilodontidae	Prochilodus britskii Castro, 1993	0	-39.59
Characiformes	Prochilodontidae	Prochilodus costatus Valenciennes, 1850	-20.66	-1.12
Characiformes	Prochilodontidae	Prochilodus hartii Steindachner, 1875	-32.35	-23.01
Characiformes	Prochilodontidae	Prochilodus lacustris Steindachner, 1907	-32.49	0
Characiformes	Prochilodontidae	Prochilodus lineatus (Valenciennes, 1837)	-30.05	-14.87
Characiformes	Prochilodontidae	Prochilodus nigricans Spix & Agassiz, 1829	NA	NA

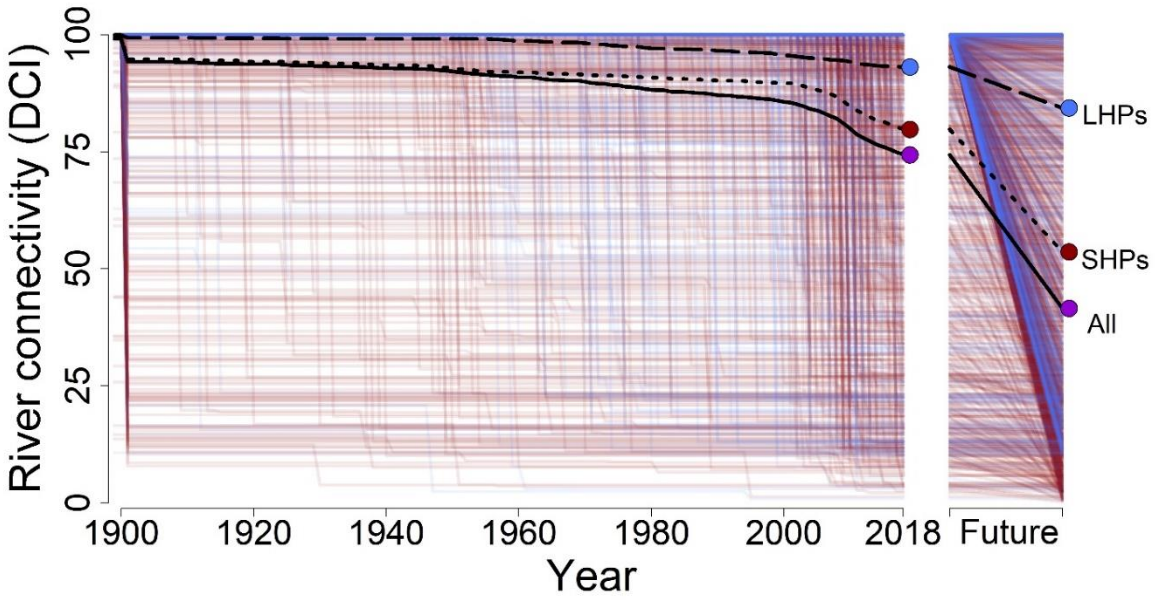
Characiformes	Prochilodontidae	Prochilodus rubrotaeniatus Jardine & Schomburgk, 1841	NA	NA
Characiformes	Prochilodontidae	Prochilodus vimboides Kner, 1859	-40.57	-7.26
Characiformes	Prochilodontidae	Semaprochilodus brama Valenciennes, 1850	NA	NA
Characiformes	Prochilodontidae	Semaprochilodus insignis (Jardine & Schomburgk, 1841)	NA	NA
Characiformes	Prochilodontidae	Semaprochilodus taeniurus (Valenciennes, 1821)	NA	NA
Characiformes	Serrasalmidae	Colossoma macropomum (Cuvier, 1816)	-23.42	-16.57
Characiformes	Serrasalmidae	Myloplus asterias (Müller & Troschel, 1844)	-40.1	-38.63
Characiformes	Serrasalmidae	Myloplus rubripinnis (Müller & Troschel, 1844)	-0.62	-14.76
Characiformes	Serrasalmidae	Myloplus torquatus (Kner, 1858)	-16.49	-19.83
Characiformes	Serrasalmidae	Mylossoma aureum (Spix & Agassiz, 1829)	0	-53.07
Characiformes	Serrasalmidae	Mylossoma duriventre (Cuvier, 1818)	NA	NA
Characiformes	Serrasalmidae	Piaractus brachypomus (Cuvier, 1818)	-9.88	-22.83
Characiformes	Serrasalmidae	Piaractus mesopotamicus (Holmberg, 1887)	NA	NA
Characiformes	Triportheidae	Triportheus albus Cope, 1872	-6.52	-28.47
Characiformes	Triportheidae	Triportheus angulatus (Spix & Agassiz, 1829)	-3.74	-21.49
Characiformes	Triportheidae	Triportheus auritus (Valenciennes, 1850)	0	-43.41
Characiformes	Triportheidae	Triportheus brachipomus (Valenciennes, 1850)	0	-55.8
Characiformes	Triportheidae	Triportheus culter (Cope, 1872)	NA	NA
Characiformes	Triportheidae	Triportheus curtus (Garman,1890)	NA	NA
Characiformes	Triportheidae	Triportheus guentheri (Garman, 1890)	-2.85	-0.28
Characiformes	Triportheidae	Triportheus nematurus (Kner, 1858)	-13.77	-15.29
Characiformes	Triportheidae	Triportheus pantanensis Malabarba, 2004	0	-0.39
Characiformes	Triportheidae	Triportheus pictus (Garman, 1890)	NA	NA
Characiformes	Triportheidae	Triportheus rotundatus (Jardine, 1841)	-0.62	-22.5
Characiformes	Triportheidae	Triportheus signatus (Garman, 1890)	0	-41.41
Characiformes	Triportheidae	Triportheus trifurcatus (Castelnau, 1855)	-21	-14.48
Siluriformes	Doradidae	Pterodoras granulosus (Valenciennes, 1821)	-30.71	-14.07
Siluriformes	Heptapteridae	Rhamdia branneri Haseman, 1911	-40.45	-6.33
Siluriformes	Heptapteridae	Rhamdia foina (Müller & Troschel, 1848)	-40.1	-55.83
Siluriformes	Heptapteridae	Rhamdia itacaiunas Silfvergrip, 1996	-10.17	-15.57
Siluriformes	Heptapteridae	Rhamdia jequitinhonha Silfvergrip, 1996	-47.71	0
Siluriformes	Heptapteridae	Rhamdia laukidi Bleeker, 1858	0	-29.52
Siluriformes	Heptapteridae	Rhamdia muelleri (Günther, 1864)	NA	NA
Siluriformes	Heptapteridae	Rhamdia poeyi Eigenmann & Eigenmann, 1888	-4.89	-12.71
Siluriformes	Heptapteridae	Rhamdia quelen (Quoy & Gaimard, 1824)	-34.18	-13.55
Siluriformes	Heptapteridae	Rhamdia voulezi Haseman, 1911	-32.66	-1.74
Siluriformes	Loricariidae	Rhinelepis aspera Spix & Agassiz, 1829	-20.23	-17
Siluriformes	Loricariidae	Rhinelepis strigosa Valenciennes, 1840	NA	NA
Siluriformes	Pimelodidae	Aguarunichthys inpai Zuanon, Rapp Py-Daniel & Jégu, 1993	0	-42.22
Siluriformes	Pimelodidae	Aguarunichthys tocantinsensis Zuanon, Rapp Py-Daniel & Jégu, 1993	-14.45	-10.09
Siluriformes	Pimelodidae	Bagropsis reinhardti Lütken, 1874	-22.44	-10.94
Siluriformes	Pimelodidae	Bergiaria westermanni (Lütken, 1874)	-7.39	-0.14

Siluriformes	Pimelodidae	Brachyplatystoma capapretum Lundberg & Akama, 2005	0	-42.22
Siluriformes	Pimelodidae	Brachyplatystoma filamentosum (Lichtenstein, 1819)	-6.35	-26.79
Siluriformes	Pimelodidae	Brachyplatystoma juruense (Boulenger, 1898)	0	-42.22
Siluriformes	Pimelodidae	Brachyplatystoma platynemum Boulenger, 1898	-0.62	-0.33
Siluriformes	Pimelodidae	Brachyplatystoma rousseauxii (Castelnau, 1855)	0	-0.66
Siluriformes	Pimelodidae	Brachyplatystoma tigrinum (Britski, 1981)	NA	NA
Siluriformes	Pimelodidae	Brachyplatystoma vaillantii (Valenciennes, 1840)	0	-32.89
Siluriformes	Pimelodidae	Calophysus macropterus (Lichtenstein, 1819)	0	-24.58
Siluriformes	Pimelodidae	Cheirocerus eques Eigenmann, 1917	NA	NA
Siluriformes	Pimelodidae	Cheirocerus goeldii (Steindachner, 1908)	NA	NA
Siluriformes	Pimelodidae	Conorhynchos conirostris (Valenciennes, 1840)	0	-0.57
Siluriformes	Pimelodidae	Duopalatinus emarginatus (Valenciennes, 1840)	-19.79	-0.14
Siluriformes	Pimelodidae	Duopalatinus peruanus Eigenmann & Allen, 1942	0	-2
Siluriformes	Pimelodidae	Exallodontus aguanai Lundberg, Mago-Leccia & Nass, 1991	0	-42.22
Siluriformes	Pimelodidae	Hemisorubim platyrhynchos (Valenciennes, 1840)	-28.1	-8.8
Siluriformes	Pimelodidae	Hypophthalmus edentatus Spix & Agassiz, 1829	-1.24	0
Siluriformes	Pimelodidae	Hypophthalmus fimbriatus Kner, 1858	-0.41	-28.47
Siluriformes	Pimelodidae	Hypophthalmus marginatus Valenciennes, 1840	-6.35	-29.1
Siluriformes	Pimelodidae	Iheringichthys labrosus (Lütken, 1874)	-23.94	-18
Siluriformes	Pimelodidae	Iheringichthys megalops Eigenmann & Ward, 1907	NA	NA
Siluriformes	Pimelodidae	Iheringichthys syi Azpelicueta & Britski, 2012	0	-0.51
Siluriformes	Pimelodidae	Leiarius marmoratus (Gill, 1870)	-12.53	-15.12
Siluriformes	Pimelodidae	Leiarius pictus (Müller & Troschel ,1849)	-41.24	0
Siluriformes	Pimelodidae	Luciopimelodus pati (Valenciennes, 1835)	-23.14	-20.43
Siluriformes	Pimelodidae	Megalonema amaxanthum Lundberg & Dahdul, 2008	NA	NA
Siluriformes	Pimelodidae	Megalonema platanum (Günther, 1880)	-31.33	-18.72
Siluriformes	Pimelodidae	Megalonema platycephalum Eigenmann, 1912	-6.17	-18.04
Siluriformes	Pimelodidae	Parapimelodus nigribarbis (Boulenger, 1889)	-27.12	-28.74
Siluriformes	Pimelodidae	Parapimelodus valenciennis (Lütken, 1874)	-15.02	-16.61
Siluriformes	Pimelodidae	Phractocephalus hemioliopterus (Bloch & Schneider, 1801)	-6.1	-32.24
Siluriformes	Pimelodidae	Pimelodina flavipinnis Steindachner, 1876	-15.44	-27.23
Siluriformes	Pimelodidae	Pimelodus absconditus Azpelicueta, 1995	-38.39	-11.58
Siluriformes	Pimelodidae	Pimelodus albofasciatus Mees, 1974	-1.15	-22.72
Siluriformes	Pimelodidae	Pimelodus altissimus Eigenmann & Pearson, 1942	NA	NA
Siluriformes	Pimelodidae	Pimelodus argenteus Perugia, 1891	NA	NA
Siluriformes	Pimelodidae	Pimelodus atrobrunneus Vidal & Lucena, 1999	-36.19	-12.26
Siluriformes	Pimelodidae	Pimelodus blochii Valenciennes, 1840	-14.96	-26.59
Siluriformes	Pimelodidae	Pimelodus britskii Garavello & Shibatta, 2007	-35.89	-7.28
Siluriformes	Pimelodidae	Pimelodus fur (Lütken, 1874)	-7.39	-22.33
Siluriformes	Pimelodidae	Pimelodus halisodous Ribeiro, Lucena & Lucinda, 2008	0	-55.75
Siluriformes	Pimelodidae	Pimelodus joannis Ribeiro, Lucena & Lucinda, 2008	0	-29.12
Siluriformes	Pimelodidae	Pimelodus luciae Rocha & Ribeiro, 2010	NA	NA

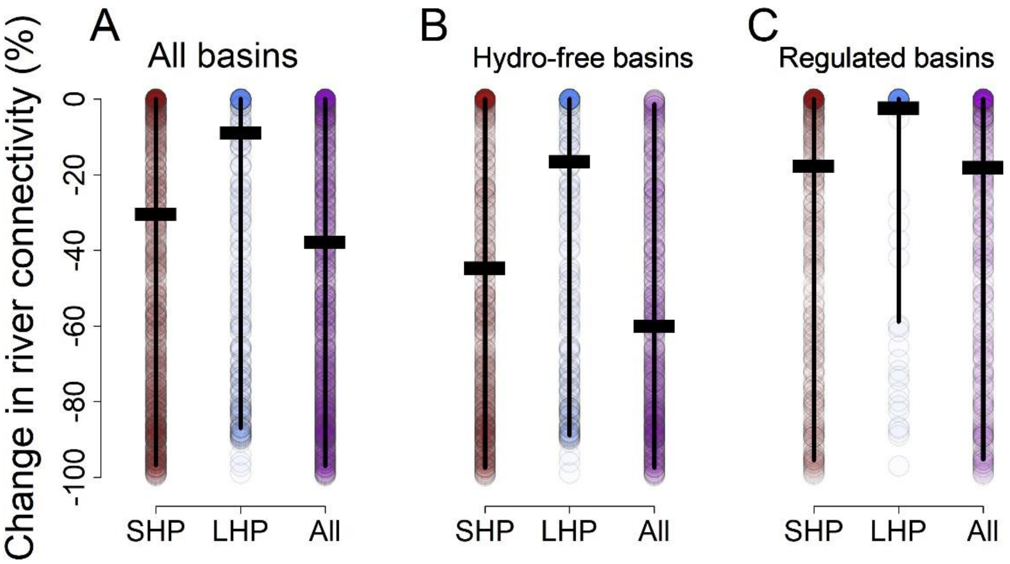
Siluriformes	Pimelodidae	Pimelodus maculatus Lacepède, 1803	-27.9	-16.53
Siluriformes	Pimelodidae	Pimelodus microstoma Steindachner, 1877	-45.98	-4.78
Siluriformes	Pimelodidae	Pimelodus multicratifer Ribeiro, de Lucena & Oyakawa, 2011	NA	NA
Siluriformes	Pimelodidae	Pimelodus mysteriosus Azpelicueta, 1998	-3.16	0
Siluriformes	Pimelodidae	Pimelodus ornatus Kner, 1858	-17.23	-16.64
Siluriformes	Pimelodidae	Pimelodus ortmanni Haseman, 1911	-28.02	-10.6
Siluriformes	Pimelodidae	Pimelodus pantaneiro Souza-Filho & Shibatta, 2007	NA	NA
Siluriformes	Pimelodidae	Pimelodus paranaensis Britski & Langeani, 1988	-29.4	-19.27
Siluriformes	Pimelodidae	Pimelodus pictus Steindachner, 1876	NA	NA
Siluriformes	Pimelodidae	Pimelodus pintado Azpelicueta, Lundberg & Loureiro, 2008	-51.3	-13.29
Siluriformes	Pimelodidae	Pimelodus platicirris Borodin, 1927	-64.52	0
Siluriformes	Pimelodidae	Pimelodus pohli Ribeiro & Lucena, 2006	0	-0.57
Siluriformes	Pimelodidae	Pimelodus stewartii Ribeiro, Lucena & Lucinda, 2008	0	-24.48
Siluriformes	Pimelodidae	Pimelodus tetramerus Ribeiro & Lucena, 2006	-21.31	-21.27
Siluriformes	Pimelodidae	Pinirampus pirinampu (Spix & Agassiz, 1829)	-25.39	-22.65
Siluriformes	Pimelodidae	Platynematichthys notatus (Jardine & Schomburgk, 1841)	0	-40.79
Siluriformes	Pimelodidae	Platysilurus mucosus (Vaillant, 1880)	NA	NA
Siluriformes	Pimelodidae	Platystomatichthys sturio (Kner, 1858)	0	-29.49
Siluriformes	Pimelodidae	Propimelodus araguayae Rocha, de Oliveira & Rapp Py-Daniel, 2007	0	-43.41
Siluriformes	Pimelodidae	Propimelodus caesius Parisi, Lundberg & DoNascimiento, 2006	NA	NA
Siluriformes	Pimelodidae	Propimelodus eigenmanni (Van der Stigchel, 1946)	0	-17.34
Siluriformes	Pimelodidae	Pseudoplatystoma corruscans (Spix & Agassiz, 1829)	-20.41	-10.56
Siluriformes	Pimelodidae	Pseudoplatystoma punctifer (Castelnau, 1855)	-10.74	-22.33
Siluriformes	Pimelodidae	Pseudoplatystoma reticulatum (Eigenmann & Eigenmann, 1889)	-23.36	-1.65
Siluriformes	Pimelodidae	Pseudoplatystoma tigrinum (Valenciennes, 1840)	-2.58	-19.27
Siluriformes	Pimelodidae	Sorubim elongatus Littmann, Burr, Schmidt & Isern, 2001	-1.77	-23.39
Siluriformes	Pimelodidae	Sorubim lima (Bloch & Schneider, 1801)	-30.08	-17.65
Siluriformes	Pimelodidae	Sorubim maniradii Littmann, Burr & Buitrago-Suarez, 2001	NA	NA
Siluriformes	Pimelodidae	Sorubim trigonocephalus Miranda Ribeiro, 1920	-17.9	-16.6
Siluriformes	Pimelodidae	Sorubimichthys planiceps (Spix & Agassiz, 1829)	-15.26	-22.47
Siluriformes	Pimelodidae	Steindachneridion amblyurum (Eigenmann & Eigenmann, 1888)	0	-3.72
Siluriformes	Pimelodidae	Steindachneridion doceanum (Eigenmann & Eigenmann, 1889)	-56.1	-26.05
Siluriformes	Pimelodidae	Steindachneridion melanodermatum Garavello, 2005	-28.71	-1.4
Siluriformes	Pimelodidae	Steindachneridion parahybae (Steindachner, 1877)	-6.65	-14.85
Siluriformes	Pimelodidae	Steindachneridion punctatum (Miranda Ribeiro, 1918)	NA	NA
Siluriformes	Pimelodidae	Steindachneridion scriptum (Miranda Ribeiro, 1918)	-26.25	-21.61
Siluriformes	Pimelodidae	Zungaro jahu (Ihering, 1898)	-24.58	-20.05
Siluriformes	Pimelodidae	Zungaro zungaro (Humboldt, 1821)	-5.09	-30.6

APPENDIX C

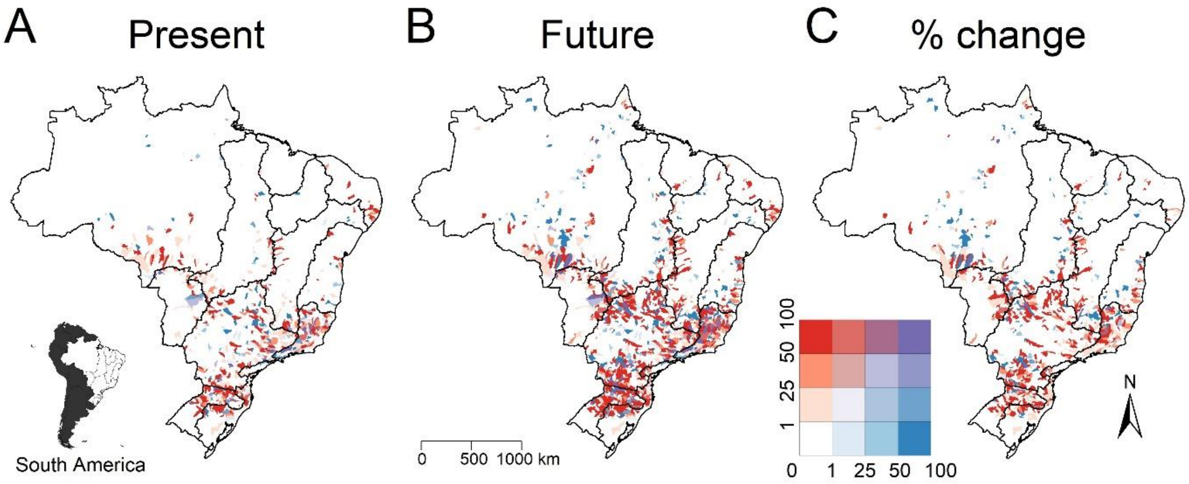
We also conducted all analyses using the DCI_i equation that simulate cases in which migratory fish populations are predominantly composed of external immigrants that arrive from downstream basins. The DCI_i is essentially the equation developed for diadromous species in the original paper by Cote *et al.* (2009), but the spatial reference of the ocean is replaced by the river basin immediately downstream. Overall, the results from DCI_i analysis suggest an increased role of LHPs on river fragmentation across the country compared to the DCI_p analysis, especially in the future scenario. However, the role of SHPs on fragmentation also increased, and SHPs remained the main source of present and projected-future connectivity loss in the country.



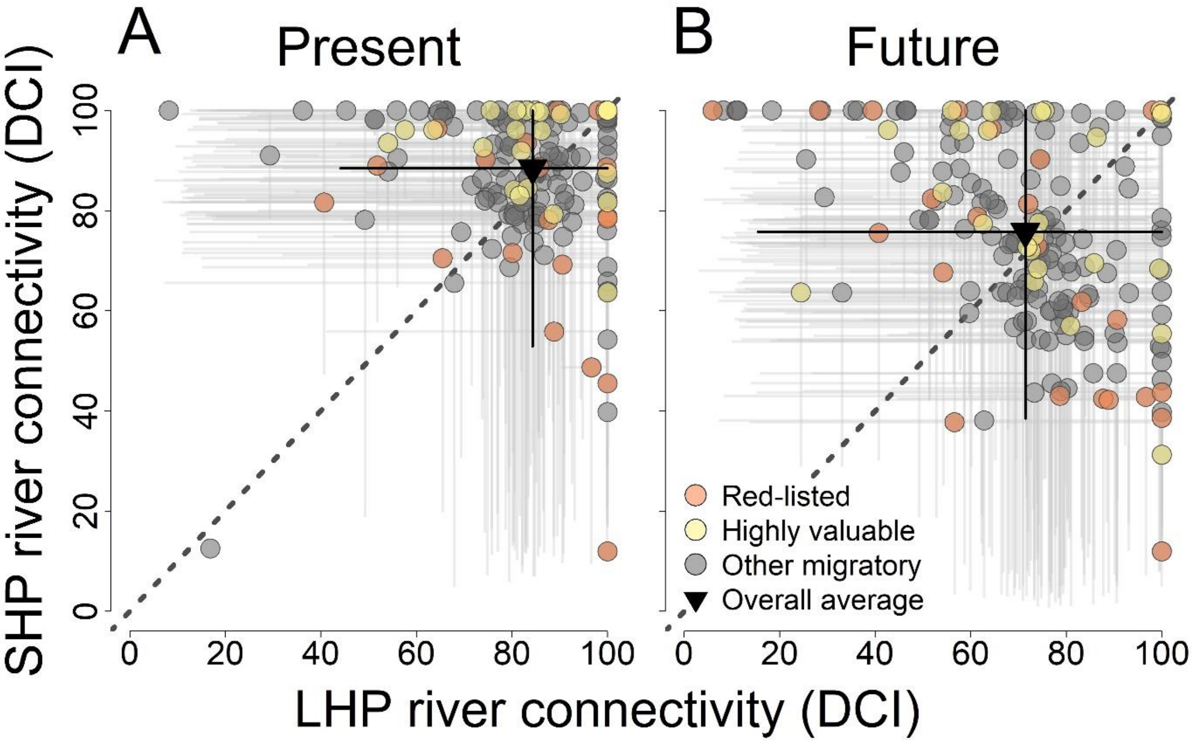
Appendix C, Fig. S1. Temporal trends in river connectivity in Brazil according to changes in DCIi over the past century and future projections due to ongoing and planned dam construction. Colored lines represent yearly estimates of DCI_i for 1,216 individual basins, with lines indicating the unique contribution of SHPs (red) and LHPs (blue). Black lines represent average values for SHPs (dotted), LHPs (dashed), and all types of hydropower combined (solid). “Future” reflects ongoing and planned construction that is realistically projected to occur in the mid-century (circa 2050); the exact year is subject to change depending on future policies and governance.



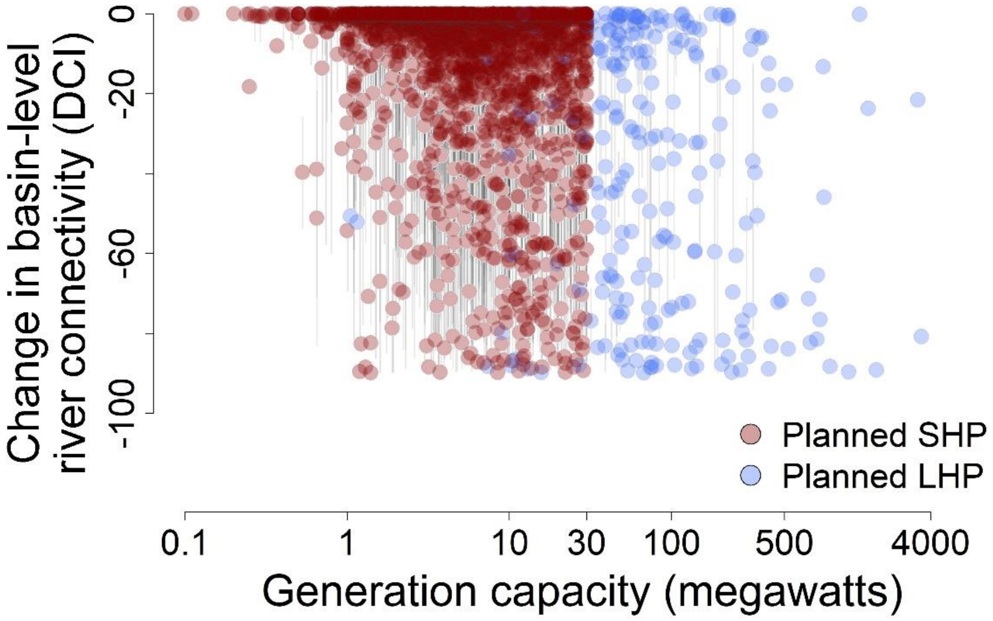
Appendix C, Fig. S2. Predicted future change (%) in river connectivity (DCI_i) from present-day (2018) to mid-century (circa 2050) for the Brazilian basins according to the separate contributions of SHPs (red), LHPs (blue), or all types of hydropower combined (purple). Black bars indicate average values and black lines indicate 95% confidence intervals. Results are shown for (A) all 1,216 basins, (B) 573 basins that are currently free of hydropower, and (C) 643 basins currently regulated by hydropower.



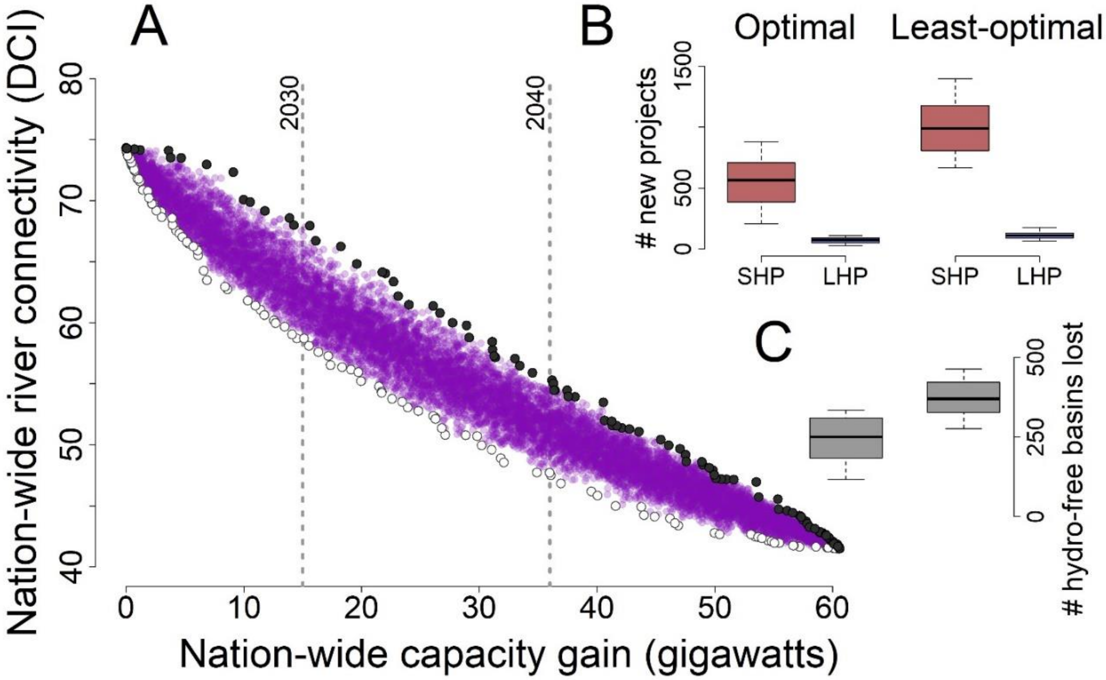
Appendix C, Fig. S3. (A) Present (2018), (B) future (circa 2050) and (C) projected percentage change in river connectivity (DCI_i) for the Brazilian basins according to the contributions of SHPs (red shades) and LHPs (blue shades). Color pallets represent (A-B) values of absolute DCI_i loss from free-flowing state (i.e. 100 - DCI$_i$) or (C) percentage loss between periods (negative change in all instances). The black lines delimit the boundaries of the major Brazilian hydrographic units.



Appendix C, Fig. S4. Contributions of SHPs and LHPs to river connectivity (DCI$_i$) in basins occupied by 191 migratory fish species according to (A) present-day and (B) projected future (mid-century) dam distributions. Average river connectivity across each species' geographic range are represented by circles (95% confidence intervals displayed as light gray lines). Colors differentiate the species that are red-listed (24, orange) or highly valuable for fisheries (24, yellow) from the remaining species (gray). Black triangles and black lines indicate the overall averages with associated 95% confidence intervals.



Appendix C, Fig. S5. Relationship between the generation capacity of each future hydropower project and its effect on river connectivity (DCI_i) at the basin level. Estimates of average DCI loss (the average difference across all basin-level future scenarios with and without each dam) for planned SHPs (red) and LHPs (blue) are represented by circles, and the gray lines indicate the range. The 24 planned LHPs with future capacity below 30 MW were classified as such by the energy agency ANEEL, following other criteria such as reservoir area.



Appendix C, Fig. S6. (A) Future projections of nation-wide river connectivity (average DCI_i across all basins) and total additional generation capacity (based on a 2018 baseline) for 10,000 bootstrap samples (circles) from the 2,266 planned dams in Brazil. Vertical dashed lines delineate the projected demands for additional hydropower generation capacity from 2030 to 2040. The colors of the circles indicate the optimal (black), least-optimal (white), and remaining (purple) dam portfolios computed using a Pareto frontier operator. The inset boxplots contrast (B) the number of new SHP and LHP projects to be built and (C) the number of basins that will no longer be free of hydropower in the optimal and least-optimal dam portfolios located within the 2030-2040 demand interval.

APPENDIX D

Appendix D, Table S1. List of macroinvertebrate taxa recorded in the Chapecó River Basin.

Taxa

ANNELIDA
Clitellata
Hirudinea
Oligochaeta

ARTHROPODA
Crustacea
Amphipoda
Amphipoda A
Hyalellidae

Branchiopoda
Branchiopoda

Decapoda
Aeglidae

HEXAPODA
Entognatha
Collembola

Insecta
Coleoptera
Curculionidae
Dytiscidae
Elmidae
Gyrinidae
Hydrophilidae
Psephenidae
Staphylinidae

Diptera
Ceratopogonidae
Chironomidae
Dixidae
Dolichopodidae
Empididae
Psychodidae
Simullidae
Tabanidae
Tipulidae

Ephemeroptera
Baetidae
Caenidae
Ephemeridae
Euthyplociidae
Leptohyphiidae
Leptophlebiidae

Hemiptera
Belostomatidae
Corixidae
Mesoveliidae
Naucoridae
Notonectidae
Veliidae

Lepidoptera
Pyralidae

Megaloptera
Corydalidae
Sialidae

Neuroptera
Sisyridae

Odonata
Aeshnidae
Calopterygidae
Coenagrionidae
Dicteriadidae
Gomphidae
Libellulidae
Megapodagrionidae

Plecoptera
Perlidae
Gripopterygidae

Trichoptera
Calamoceratidae
Glossosomatidae
Hydrobiosidae
Hydropsychidae
Hydroptilidae
Leptoceridae
Limnephilidae
Odontoceidae
Philopotamidae
Polycentropodidae

MOLLUSCA
Gastropoda
Architaenioglossa
Ampullariidae

Basommatophora
Ancylidae
Physidae
Planorbidae

Littorinimorpha
Hydrobiidae

Bivalvia
Bivalvia A

Unionida
Hyriidae

Sphaeriida
Sphaeriidae

NEMATODA
Nematoda

PLATYHELMINTHES
Platyhelminthes A
Turbellaria

Appendix D, Table S2. List of fish species recorded in the Chapecó River Basin.

Taxa
CHARACIFORMES
Characidae
Astyanax lacustris (Lütken, 1875)
Astyanax cf. *laticeps* (Cope, 1894)
Bryconamericus stramineus Eigenmann, 1908
Moenkhausia sp. "losango"
Oligosarcus brevioris Menezes, 1987
Erythrinidae
Hoplias malabaricus (Bloch, 1794)
SILURIFORMES
Heptapteridae
Heptapterus mustelinus (Valenciennes, 1835)
Rhamdia quelen (Quoy & Gaimard, 1824)
Loricariidae
Ancistrus taunayi Miranda Ribeiro, 1918
Hemiancistrus cf. *fuliginosus* Cardoso & Malabarba, 1999
Hypostomus commersoni Valenciennes, 1836
Hypostomus uruguayensis Reis, Weber & Malabarba, 1990
Rineloricaria anhanguapitan Ghazzi, 2008
Rineloricaria stellata Ghazzi, 2008
CICHLIFORMES
Cichlidae
Australoheros kaaygua Casciotta, Almirón & Gómez, 2006
Crenicichla igara Lucena & Kullander, 1992
Geophagus brasiliensis (Quoy & Gaimard, 1824)

VITA

Thiago B. A. Couto is a freshwater ecologist who is passionate about rivers and tropical fishes. Couto has led and participated in numerous research projects and scientific expeditions in Brazilian rivers throughout his career, including rivers and floodplains in the Amazon, Paraná, and Uruguay Basins. His work addresses scientific questions that have on-the-ground management and conservation implications for freshwater environments. He believes that science plays a major role in informing political decisions and that environmental scientists should get involved with the decision-making process.