BALANCING HYDROPOWER AND FRESHWATER ENVIRONMENTS IN THE GLOBAL SOUTH

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BALANCING HYDROPOWER AND FRESHWATER ENVIRONMENTS IN THE GLOBAL SOUTH

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EDITORIAL
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Editorial: Balancing Hydropower and Freshwater Environments in the Global South

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Editorial on the Research Topic

Balancing Hydropower and Freshwater Environments in the Global South

The construction of hydropower dams is growing rapidly across the southern hemisphere and developing world (Winemiller et al., 2016), with most new dams being built in South America and Asia (Baumgartner et al., 2014). Freshwater ecosystems are tremendously impacted by dam construction and reservoir operation (Brown et al., 2014). For instance, the Living Planet Index indicates an 89% loss in biodiversity in freshwater environments globally arising from all forms of river development (Deinet et al., 2020). Dams alter flow (Timpe and Kaplan 2017) and sediment regimes (Wang et al., 2018), which impact ecosystem services, wetland conservation, water quality, land fertility, and fisheries productivity (Reilly et al., 2018).

Despite its known impacts, hydropower is generally considered a relatively cheap and climate-friendly source of energy (Athayde et al., 2019). It has been shown, however, that hydropower operations can have high green-house gas emissions, especially in the tropics (Almeida et al., 2019). Regardless, sustained economic and population growth are fuelling continued dam construction, often at the expense of other ecosystem services. Until recently, most research on the connections between dams and freshwater ecosystems has focused on the Northern hemisphere; this research topic seeks to address this gap. The 12 articles in the research topic ask several key questions related to the hydrological, ecological, social, and economic values of rivers and dams in the southern hemisphere: What ecosystem services are gained and lost with hydropower development? Over what time frame are impacts realized? Who "wins" and "loses" as these trade-offs are made?

Several studies presented evidence that hydropower operations caused substantial ecosystem impacts beyond the main river channel. Three papers quantified the ecological impacts of dam operations on connected wetland systems, such as the Pantanal (Ely et al.; Figueiredo et al.; Jardim et al.). Additionally, Fantin-Cruz et al. showed that dam-induced reductions in river flow reduced the frequency of wetland connectivity events. This disconnection had the additive effect of interrupting nutrient-rich sediment transport (Oliveira et al.) and reducing fisheries recruitment (Oliveira et al.). Taken together, these six papers connect hydrological alteration, sediment and nutrient dynamics, and fisheries impacts, highlighting the need for multi- and interdisciplinary approaches to fully understand dam-induced impacts on ecosystems.

Four papers addressed the impacts of different dam types and modes of operation. Developing operational protocols that reduce hydropeaking was identified as a straightforward way to mitigate the most undesirable hydrological, geomorphological, ecological, and social effects on downstream reaches (Almeida et al.). As noted by Doria et al., hydropeaking operations severely impact riverine (human) communities that are dependent on fisheries resources. In addition, there was a suggestion

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that converting conventional hydropower projects to "pumped hydropower" initiatives, while potentially beneficial economically, could create the unfavorable outcome of transferring invasive species (Doyle et al.). Finally, despite their relatively "small" scale, the planned proliferation of low-head hydropower dams is expected to have large social and ecological impacts in Uganda (O'Brien et al.), which these authors suggest may be partially mitigated by the adoption of locally relveant environmental flow practices.

Finally, two papers focused on dam planning. Campbell and Barlow and Gonzalez et al. suggested that improved pre-construction planning is fundamental to enhancing the ecological and social benefits of hydropower in tropical systems. Unfortunately, but perhaps unsurprisingly, stakeholders reported that dam companies prioritize decisions that maximize profits, as opposed to mitigating impacts. Providing economically sustainable outcomes, while minimizing environmental impacts, thus remains a major challenge (Silva et al., 2018). Regardless of region and dam type, it is clear that engineers, developers, planners, ecologists, and communities must work together and consider whole-catchment effects to bring about the best outcomes for people and rivers (Baumgartner L. et al., 2014).

AUTHOR CONTRIBUTIONS

All authors listed have made a substantial, direct, and intellectual contribution to the work and approved it for publication.

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The editors of this research topic hope you will enjoy reading these articles, which increase the overall awareness of hydropower and freshwater interactions in the Global South. We would like to thank all the contributors and the Frontiers staff who have helped to make this research topic a success.

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Hydropeaking Operations of Two Run-of-River Mega-Dams Alter Downstream Hydrology of the Largest Amazon Tributary

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Large storage dams have widely documented impacts on downstream aquatic environments, but hydroelectric dams with little or no capacity for storage of water inflows (i.e., run-of-river) have received less attention. Two of the world's largest run-of-river hydropower dams (Jirau and Santo Antônio, Brazil) are located on the Madeira River, the largest tributary to the Amazon River. Here we examine whether the Madeira dams have affected downstream seasonal flood pulses and short-term (daily and sub-daily) flow dynamics. We show that the combined effects of these dams on seasonal flood pulses were modest. However, dam operations significantly increased day-to-day and sub-daily flow variability. The increase in short-term flow variability is largely explained by rapid, short-term variations in river flow caused by fluctuations in energy demand (hydropeaking). Both the magnitude of hydropeaking and the mean absolute day-to-day change in discharge downstream of the dams doubled after dam closure. In addition, the median hourly rate of water level change downstream of the dams was three times higher than upstream. Our findings highlight that even run-of-river dams on very large rivers such as the Madeira—whose average discharge at the dam site is larger than that of the Mississippi River at its mouth—can alter downstream hydrology through hydropeaking. Although little studied in tropical floodplain rivers, hydropeaking by large run-of-river dams may be detrimental to downstream aquatic organisms and human populations that utilize the river for navigation and fisheries.

Keywords: Madeira River, Amazon, hydroelectricity, sub-daily discharges, environmental flow, run-of-the-river, hydrology, flood pulse

INTRODUCTION

Dams affect downstream ecosystems and their biodiversity through alteration of the frequency, magnitude, duration, timing, and rate of change of natural flow regimes (Richter et al., 1996; Poff et al., 1997; Nilsson and Berggren, 2000). Most of the existing knowledge on the downstream impacts of dams comes from storage dams with relatively large reservoirs, high flow regulation, and long water residence times, which cause significant disruption of downstream flow regimes (Lehner et al., 2011). Conversely, run-of-river hydroelectric facilities (i.e., hydroelectric generation with little or no active storage of water inflows) are generally thought to have lesser impacts on downstream hydrology (Csiki and Rhoads, 2014). However, run-of-river dams can cause short-term fluctuations in downstream flow as a result of daily and sub-daily variation in flow releases to meet short-term variation in demand for electricity (Ashraf et al., 2018; Greimel et al., 2018), a phenomenon commonly known as hydropeaking. Most studies on the downstream impacts of run-of-river dams have examined relatively small dams located in North America and Europe (Anderson et al., 2015; Bejarano et al., 2018). It is not known how modern, large run-of-river dams such as those newly constructed, under construction, and planned for the Amazon basin (Anderson et al., 2018; Almeida et al., 2019b) may affect downstream flow regimes. Understanding the hydrological effects of contemporary Amazon dam operations is especially important considering that past dam construction has caused substantial hydrological alterations in some Amazonian rivers (Timpe and Kaplan, 2017).

In unregulated large rivers, the natural flow regime is key to maintaining river and floodplain biodiversity, productivity, and ecosystem processes, supporting people through fisheries, harvest of other wild foods and products, and agriculture (Junk et al., 1989; Poff et al., 1997; McClain and Naiman, 2008; Lima et al., 2017). In recent years, new large dams have been proposed in many important tributaries of the Amazon River Basin (Winemiller et al., 2016; Anderson et al., 2018; Almeida et al., 2019b), raising concerns about downstream disruption of natural flows (Forsberg et al., 2017). One of the large Amazonian rivers with vast untapped hydroelectric potential is the Madeira

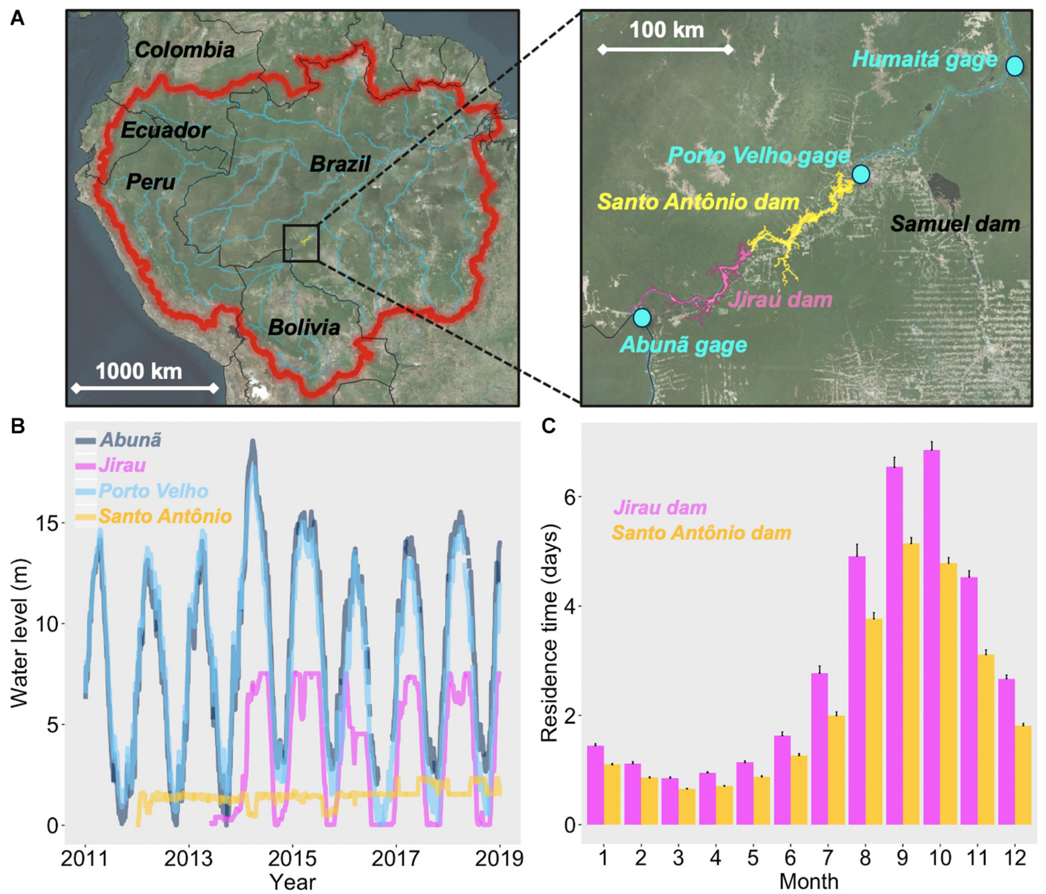

FIGURE 1 | Study site characterization. **(A)** Location of Jirau and Santo Antônio dams and the Abunã (upstream), Porto Velho (5 km downstream of Santo Antônio dam), and Humaitá (250 km downstream) gaging stations; the red outline indicates the Amazon basin limits. **(B)** Water level variation at Jirau dam, Santo Antônio dam, Abunã (upstream of both dams), and Porto Velho (downstream of both dams); data for Jirau and Santo Antônio are shown after the reservoirs were filled to full pool. Operational rules at Jirau are more variable to avoid inundation of Bolivian territory during low flows. **(C)** monthly average (±standard error) of theoretical water residence times within both reservoirs between 2012 and 2018. The annual average residence time is 3.0 days at Jirau and 2.2 days at Santo Antônio.

River, the largest source of water, sediments, and nutrients to the Amazon River mainstem (McClain and Naiman, 2008; Almeida et al., 2015).

Two of the world's largest run-of-river dams were built on the mainstem of the Madeira River in Brazil this decade (Jirau and Santo Antônio). As of 2020, several more dams have been proposed for upstream reaches (Almeida et al., 2019b), including large storage dams on tributaries (Forsberg et al., 2017). An analysis using environmental vulnerability indices has identified the Madeira as the Amazonian river system that is most threatened by dam construction (Latrubesse et al., 2017). Recent studies on the impacts of the Jirau and Santo Antônio dams report decreases in downstream fishery yields (Santos et al., 2018; Lima et al., 2020) and suspended sediment concentrations (Latrubesse et al., 2017)—although the attribution of suspended sediment changes to the dams has been questioned because concentrations have also decreased upstream of both reservoirs (Ayes et al., 2019). A remote sensing analysis has revealed that the area inundated by the Jirau and Santo Antônio dams is 60% larger than initially predicted in pre-dam environmental impact assessments (Cochrane et al., 2017), which may be in part related to changes in project design. Although the residence time of the Madeira dams is short (**Figure 1**), drowned tributary valleys created by the dams show significant limnological alterations, including thermal stratification and increased availability of organic matter (De Faria et al., 2015; Almeida et al., 2019a).

Understanding the environmental effects of the Madeira dams is critical to better document impacts, guide mitigation measures, and inform decisions on the siting and design of future Amazonian hydropower facilities. Here we use pre- and post-dam flow data from above and below the Madeira dams to examine whether they have affected downstream seasonal flood pulses and short-term flow dynamics. We hypothesized that minimal changes to seasonal flood pulses would be observed given the run-of-river design of the dams. In contrast, we expected that examination of sub-daily and day-to-day changes in discharge would reveal the existence and magnitude of hydropeaking.

MATERIALS AND METHODS

Study Site

With an area of 1.4 million km^2, the Madeira River basin extends through Brazil, Peru, and Bolivia, covering ∼25% of the Amazon basin. The Madeira River flows into the Amazon River downstream of Manaus, Brazil. The Jirau and Santo Antônio dams are two run-of-river dams built in the municipality of Porto Velho, about 1,000 km upstream of the Madeira River mouth (**Figure 1**). These dams are ∼100 km apart and were designed so that water inflows approximately equal outflows (i.e., run-of-river), but damming has created reservoirs that are operated at a relatively constant water level throughout the year—particularly at Santo Antônio dam (**Figure 1C**). The downstream dam is Santo Antônio (installed capacity: 3,568 MW, reservoir area: 471 km^2, total reservoir volume: 2075 × 10^6 m^3, reservoir length: 130 km, average depth: 11 m), and the upstream dam is Jirau (installed capacity: 3,750 MW, reservoir area: 362 km^2, reservoir volume: 2747 × 10^6 m^3, average depth: 11 m). The Santo Antônio reservoir started filling in September 2011, reaching full pool in January 2012; filling of the Jirau reservoir started in October 2012 and reached full pool in May 2013. Because the Jirau dam is immediately upstream of the reservoir of Santo Antônio dam and it was filled shortly afterward, our observations speak to the combined effects of the two dams on the downstream river. **Figure 2** illustrates how inflows are managed distinctly in run-of-river versus storage dams by comparing water inflows and outflows at the Santo Antônio (run-of-river) and the nearby Samuel dam (storage), located on a Madeira tributary (Jamari River; see **Figure 1A**); both dams are used to generate hydroelectricity.

Hydrological Data

Data on river stage and discharge between 2006 and 2018 were obtained from the Abunã, Porto Velho and Humaitá gaging stations (codes 15320002, 15400000, and 15630000, respectively), which are maintained by Brazil's National Water Agency[1]. The

[1] https://www.snirh.gov.br/hidroweb/

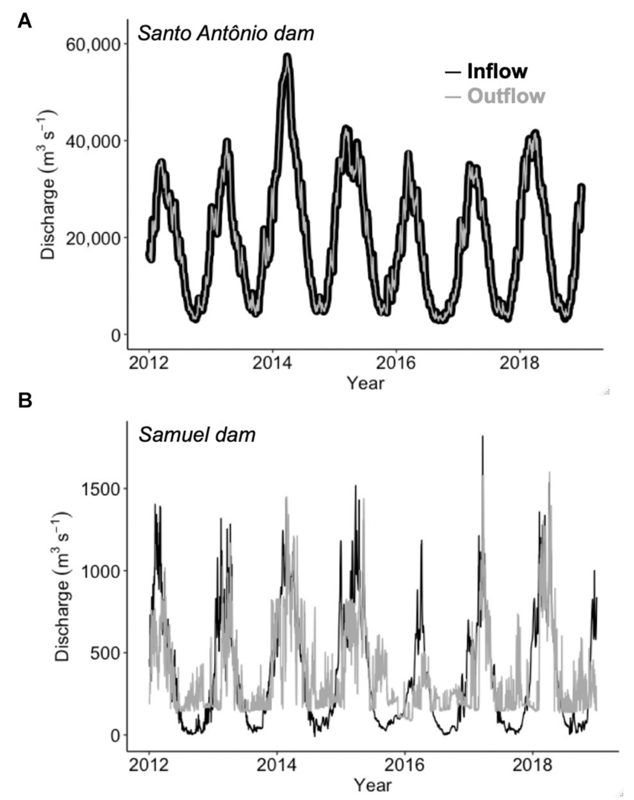

FIGURE 2 | Water inflow and outflow in run-of-river and storage dams. **(A)** Inflow and outflow discharges nearly matched each other at Santo Antônio, a run-of-river dam, between 2012 and 2018. **(B)** Inflow and outflow discharges were very different from each other at Samuel, a nearby storage dam.

Abunã station, located about 5 km upstream of the Jirau reservoir (i.e., upstream of both dams; drainage area 921,000 km^2), is used as a reference station. The Porto Velho station, located about 5 km downstream of the Santo Antônio dam (i.e., downstream of both dams; drainage area 976,000 km^2), was used to assess direct hydrologic effects. The Humaitá station, located ∼250 km downstream from the Porto Velho station, was used to assess downstream attenuation of the observed hydrologic effects. We also analyzed hourly discharge and water level data for Abunã and Porto Velho between 2015 and 2018 (post-dam)—hourly data were not available for the pre-dam period. We estimated daily water residence times within the Jirau and Santo Antônio reservoirs by dividing daily river discharge (at Abunã for Jirau and at Porto Velho for Santo Antônio) by reservoir volume (**Figure 1C**); we call this the theoretical water residence time because it assumes complete mixing of river water within the entire reservoir.

The magnitude and duration of annual extreme water conditions were evaluated by calculating lowest and highest average daily flows over 3, 7, and 30-day periods for each year

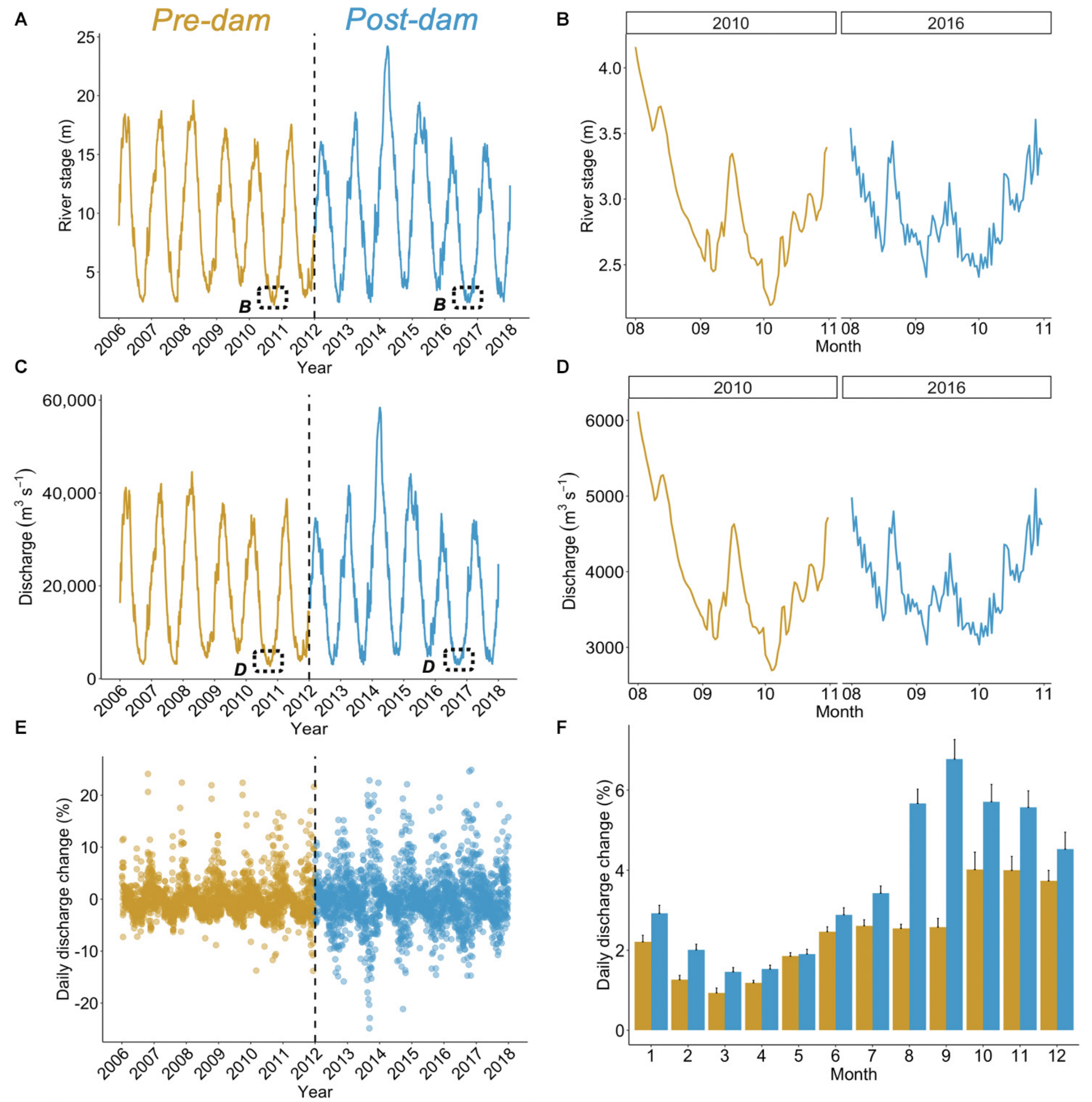

FIGURE 3 | Seasonal variation of water level and discharge before and after damming. Daily variation in **(A)** stage and **(C)** discharge of the Madeira River at Porto Velho (5 km downstream of the Santo Antônio dam) between 2006 and 2018. Close-up view of daily variation in **(B)** stage and **(D)** discharge between August and October of 2010 (pre-dam) and 2016 (post-dam). **(E)** Day-to-day percent change in discharge at Porto Velho between 2006 and 2018. **(F)** Mean (±standard error) day-to-day percent change in discharge for each month pre- and post-dam installation. The dashed vertical lines in panels **(A)**, **(C)**, and **(E)** indicate the date of closure of the Santo Antônio dam, the first to be concluded.

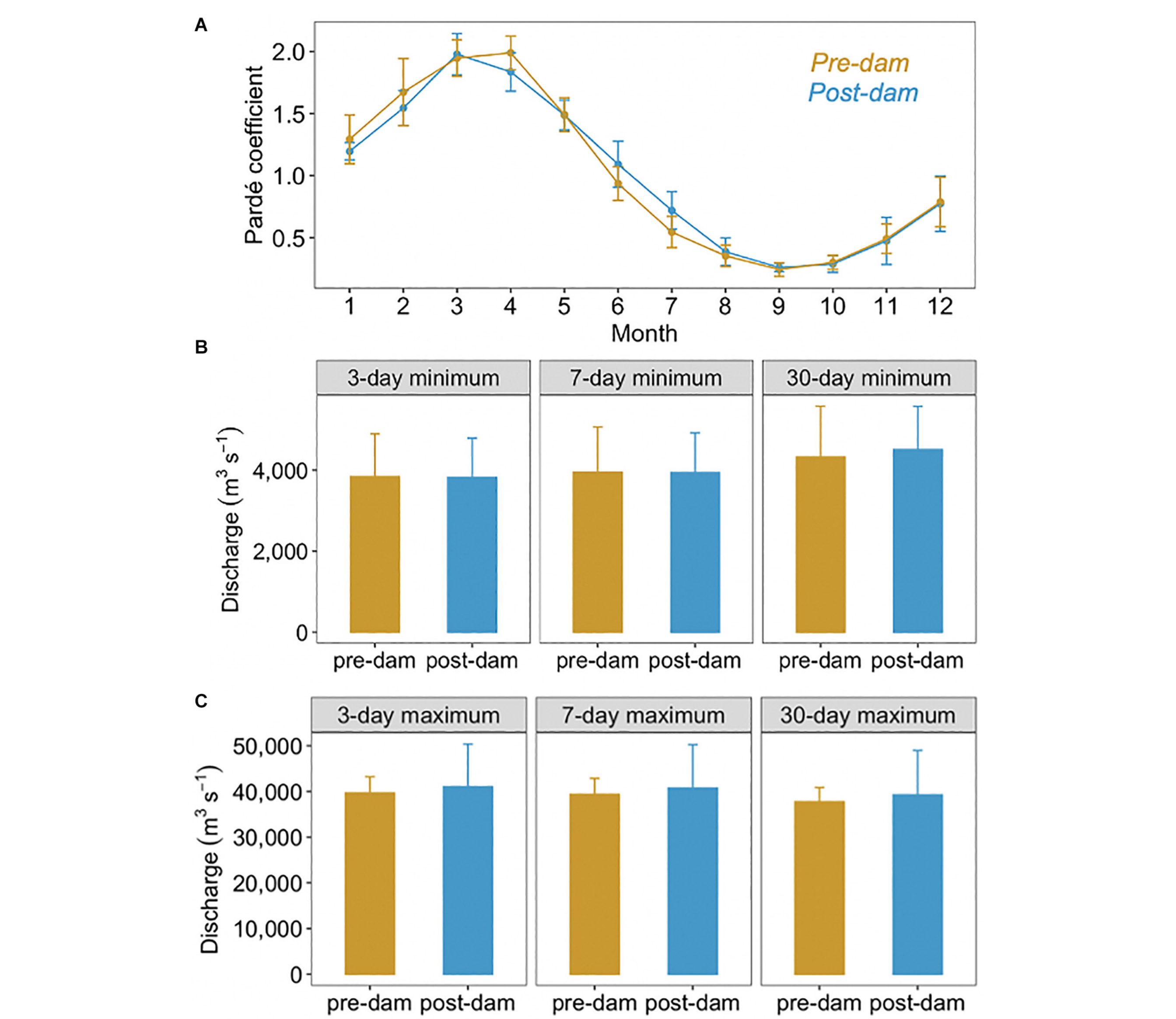

FIGURE 4 | Indicators of variability in seasonal flows based on daily discharge between 2006 and 2018. **(A)** Mean (± standard deviation) monthly Pardé coefficients (i.e., mean monthly discharge divided by the mean annual discharge) of the Madeira River at Porto Velho (5 km downstream of the Santo Antônio dam). **(B,C)**, Mean (± standard deviation) annual 3, 7, and 30-day minimum and maximum discharges of the Madeira River at Porto Velho.

(The Nature Conservancy, 2009). We assessed the seasonality of the flow regime by calculating monthly Pardé coefficients, which are defined as mean monthly discharge divided by the mean annual discharge (Meile et al., 2011). Thus, comparison of pre- versus post-dam monthly Pardé coefficients allowed us to determine whether the dams have modified seasonal flood pulses. In addition to indicators of seasonal changes in the flow regime, we used several indicators of short-term hydrological alterations, namely the Richards–Baker (R–B) flashiness index, daily discharge fall and rise rate (m^3 s^{-1} day^{-1}), number of reversals, hourly rate of water level change (cm h^{-1}), and hourly discharge change rate (HP1; m^3 s^{-1} h^{-1}), which is a dimensionless indicator of the magnitude of hydropeaking based on hourly discharge data. The R–B flashiness index is the sum of absolute daily change in discharge divided by the sum of average daily discharges (Baker et al., 2004). The rise rate is the daily change in discharge when it is increasing, whereas the fall rate is the daily change when discharge is decreasing. Reversals are changes from a rising period to a falling period or vice versa; here, a change in the sign of the difference between two consecutive days is considered as a reversal event. Daily discharge rise and fall rates and the number of reversals were calculated using Indicators of Hydrologic Alteration version 7.1, a freely available software (The Nature Conservancy, 2009). We used hourly discharge data to calculate HP1 for each day by dividing the difference between maximum and minimum discharge by

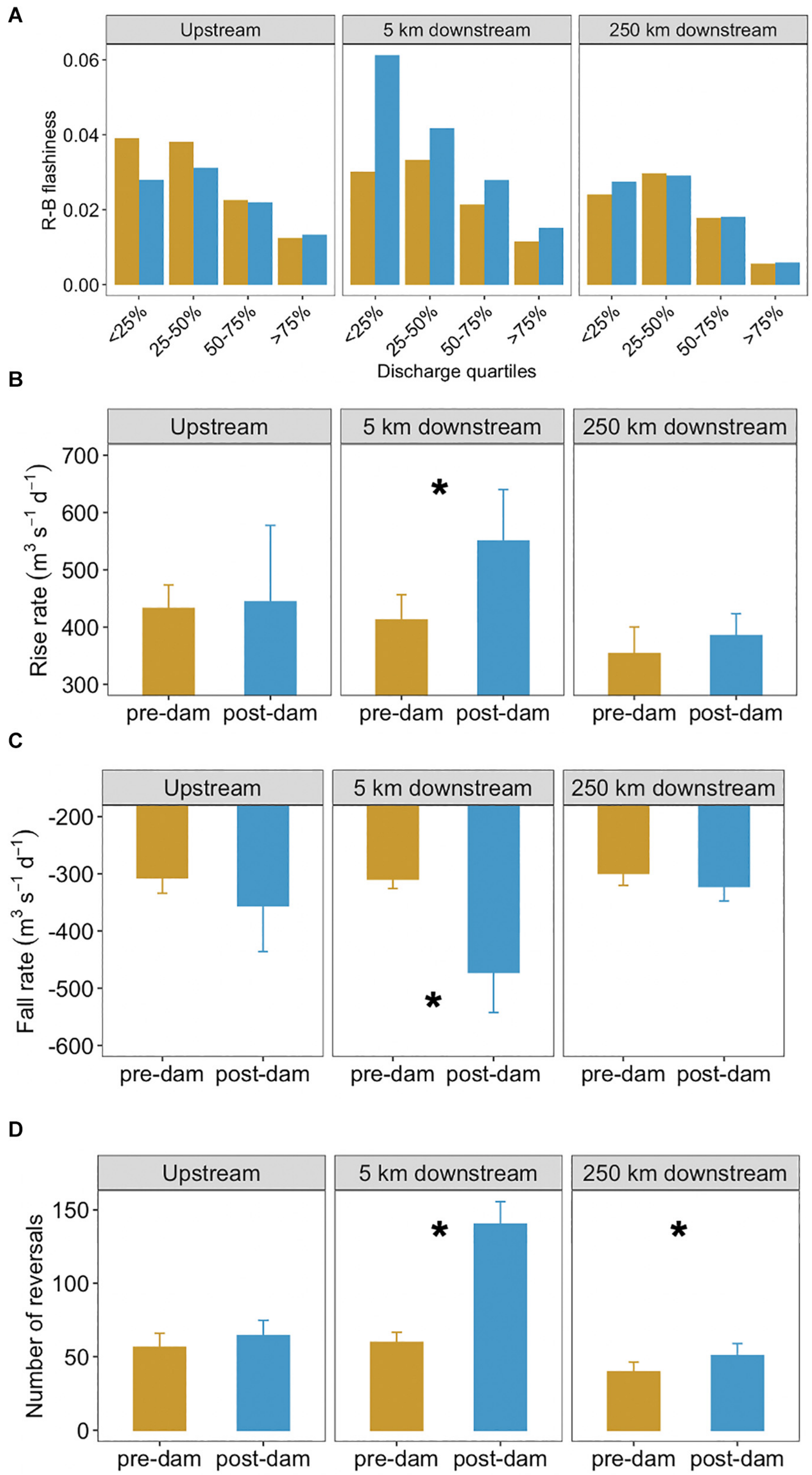

FIGURE 5 | Indicators of short-term variability in flow based on daily discharge between 2006 and 2018. **(A)** Pre- (2006–2011) and post-dam (2012–2017) Richards–Baker (R–B) flashiness index per discharge quartiles at Abunã (upstream of both dams), Porto Velho (5 km downstream) and Humaitá (250 km downstream). Pre- versus post-dam averages (± standard deviation) of annual **(B)** discharge rise rates, **(C)** discharge fall rates, and **(D)** number of discharge reversals at Abunã, Porto Velho and Humaitá. Stars indicate significant pre- versus post-dam differences (two-tailed *t*-test, $p < 0.05$).

the daily mean (Carolli et al., 2015). The hydrologic parameters used here and their ecological implications are described in detail elsewhere (Richter et al., 1996; Baker et al., 2004; Meile et al., 2011; Carolli et al., 2015; Timpe and Kaplan, 2017). Our hydrological analyses compare six years of pre-dam flow data with six years of post-dam flow data, which has been shown to be a satisfactory length of record for flow analysis in low-elevation, high-discharge Amazonian rivers (Timpe and Kaplan, 2017).

RESULTS

Effects on Downstream Seasonal Flood Pulses

The markedly unimodal nature of the Madeira River's seasonal flood pulse was preserved downstream of the Jirau and Santo Antônio dams after their construction (**Figures 3A,C**), which becomes especially clear when comparing pre- and post-dam monthly Pardé coefficients (**Figure 4C**). The post-dam years spanned a wide range of discharge: 2014 had the largest annual average discharge on record, and 2016 had the second lowest annual average discharge on record, with records extending back to 1968. Still, the magnitude and duration of annual discharge maxima and minima were not significantly affected by dam closure (two-tailed t-test, $p > 0.05$), as indicated by annual 3, 7, and 30-day minima and maxima below the dams (**Figures 4A,B**).

Short-Term Effects on Downstream Flows

Although we could not detect signs of disruption in seasonal flood pulses, the dams increased the short-term variability in discharge (**Figures 5**, **6**). We found significant post-dam increases in the R-B flashiness index (**Figure 5A**), daily discharge rise and fall rates (**Figures 5B,C**), and number of reversals at Porto Velho (**Figure 5D**). The mean absolute day-to-day change in discharge nearly doubled after dam closure, increasing from 2.3 to 3.9% (**Figures 3E,F**). The pre- vs. post-dam difference in discharge flashiness increased as discharge decreased (**Figures 3C,E**, **5A**), coinciding with periods of higher water residence time within the reservoir (**Figure 1C**). The day-to-day hydrological alterations observed at Porto Velho (∼5 km downstream of the Santo Antônio dam) are considerably attenuated a few hundred km downstream, as suggested by a lack of significant difference between pre- and post-dam discharge rise and fall rates at the next gaging station (Humaitá, ∼250 km downstream of the Santo Antônio dam) (**Figures 5A–C**). The difference in the number of reversals 250 km downstream of the dam was still significant, but much less pronounced than at Porto Velho (**Figure 5D**). In addition, the low-flow flashiness index at Humaitá was only 13% higher in the post-dam period, as compared to 94% higher at Porto Velho (**Figure 5A**).

Dam operations have also altered downstream flashiness on a sub-daily basis (**Figure 6**). Hourly discharge data available for Porto Velho (5 km downstream of the dams) and Abunã (reference station, upstream of both dams) between 2015 and 2018 (post-dam) indicates that the dams have doubled the magnitude of hydropeaks (**Figure 6A**). In addition, the hourly rate of change in discharge and water level is more variable in response to dam operations, as indicated by interquartile ranges that are three times larger downstream compared to upstream of the dams (**Figures 6B,C**). The hourly rate of discharge and water level changes downstream of the dams has a clear diel pattern, being positively correlated with the median hourly electricity demand from Brazil's North subsystem for the same time period ($r = 0.58$, $p < 0.05$) (**Figure 7**).

DISCUSSION

Our results suggest that the two large run-of-river dams recently built on the Madeira River have not altered downstream seasonal flood pulses, which was anticipated in pre-dam environmental impact studies. However, the operation of these dams significantly increased short-term (daily and sub-daily) flow variability. The observed increase in short-term flashiness downstream of the dams is in part associated with the satisfaction of peak electricity demand, as indicated by a positive correlation

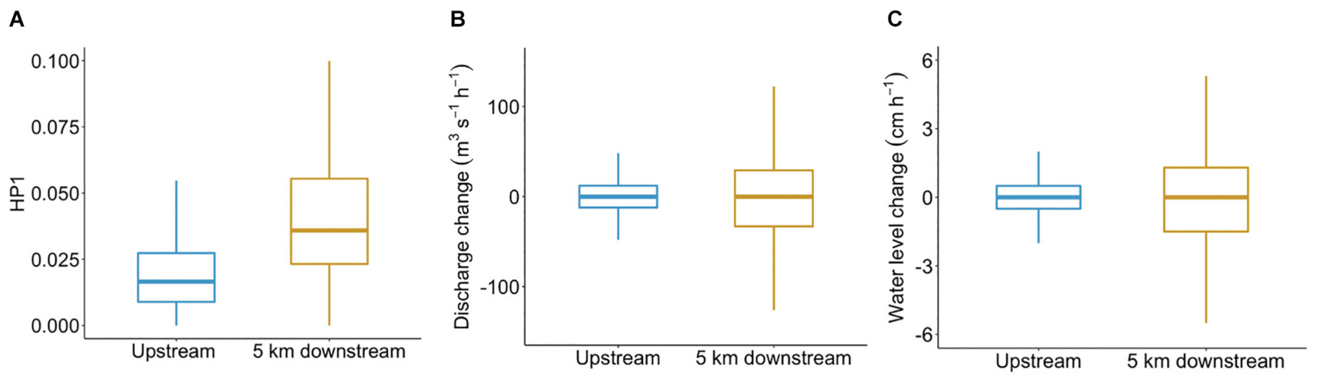

FIGURE 6 | Sub-daily rate of change of flow downstream of the dams. **(A)** Dimensionless indicator of the magnitude of hydropeaking (HP1) on the Madeira River based on hourly discharge data at Abunã (upstream of both dams) and Porto Velho (5 km downstream of Santo Antônio dam). Hourly rate of change in **(B)** discharge and **(C)** water level upstream and downstream of the dams.

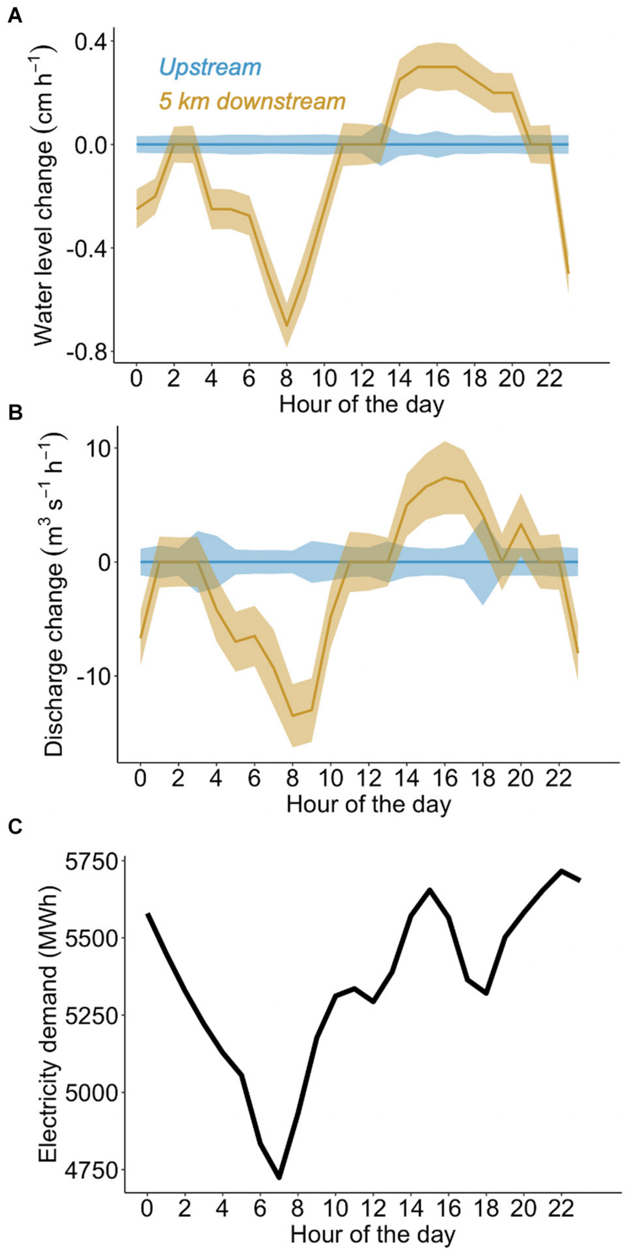

FIGURE 7 | Hydropeaking operations. Median **(A)** water level and **(B)** discharge change at Abunã (upstream of both dams) and Porto Velho (5 km downstream of Santo Antônio dam) based on hourly data between 2015 and 2018. **(C)** Median electricity demand from Brazil's North subsystem based on hourly data for the same time period as in panels **(A,B)**. Electricity demand data are publicly available on the website of Brazil's Operator of the National Electricity System ("ONS," http://www.ons.org.br).

between median hourly discharge change rates and median hourly electricity demand (r = 0.58, $p < 0.05$; **Figure 7**). There is downstream attenuation of the short-term fluctuations, likely explained by mitigating effects of water inflow from tributaries such as the Ji-Paraná and Jamari rivers, as well as channel and floodplain effects (De Paiva et al., 2013; Lininger and Latrubesse, 2016).

Studies on the environmental consequences of hydropeaking in the Amazon Basin are lacking. But effects of hydropeaking have been studied in many other smaller river systems (Zimmerman et al., 2010; Bevelhimer et al., 2015; Kennedy et al., 2016; Bejarano et al., 2018) and include destabilization of sediment accumulations along river bars, disruption of plant and animal life cycles in nearshore zones, and stranding of fishes. Human use of riparian zones and floodplains for growing crops can also be negatively affected in regions with flood-recession agriculture (Richter et al., 2010). Studies in Northern Hemisphere rivers show that hydropeaking causes substantial stranding and entrapment of early life stages of various salmonid fish species, with downramping rates as low as 2.4 cm h^{-1} potentially leading to significant stranding. The median sub-daily rate of water level change (1.5 cm h^{-1}) downstream of the Madeira dams is three times higher than upstream, with rates staying above 2.5 cm h^{-1} during 30% of the time (**Figure 6C**).

Our findings do not allow us to directly link hydropeaking to social and ecological impacts downstream of the dams. But results from a recent ethnobiological study suggest that it is possible that the increased hydropeaking reported here is leading to important socio-ecological consequences downstream of the Madeira dams. Local fishers perceive changes to natural flow regimes as the most negative impact of the Madeira dams (Santos et al., 2020). More specifically, fishers contend that sudden variations in river levels (locally known as "repiquetes") are the most relevant hydrologic impact of the Madeira dams. Some fishers argue that the increased irregularity and unpredictability of the flow regime caused by hydropeaking negatively affect fish catches; according to them, catches increase when river levels begin to fall, and then decrease when the dam releases water (Santos et al., 2020).

Indeed, recent studies report considerable declines in fishery yields downstream of the Madeira dams (Santos et al., 2018; Lima et al., 2020). One of these studies attributed the declines to a combination of blockage of migratory routes by the Madeira dams as well as dam operations that increased downstream water levels (making fishing more difficult) and caused greater water level variability (which could affect fish behavior) (Santos et al., 2018). Our results suggest that the higher post-dam discharges are unlikely to be dam-related, particularly considering that the two dams are not capable of increasing water levels and discharge over annual time scales because their reservoirs do not vary much in volume. Although the average post-dam discharge (19,451 m^3 s^{-1}) was about 6% higher than the average pre-dam discharge (18,396 m^3 s^{-1}), when we exclude the year 2014, characterized by the largest flood on record, the average post-dam discharge (18,082 m^3 s^{-1}) becomes very similar to the average pre-dam discharge. Still, the reported decline in downstream fisheries is consistent with the increased short-term variability in downstream flows that we report, especially considering that the rate of change in discharge has been demonstrated to be an important regulator of fishery yields in the Madeira River (Lima et al., 2017). In fact, local fishers claim that the abrupt daily changes in downstream water levels caused by dam operations disrupt cues that trigger the reproductive migration of fish from nutrient-poor, clear-water tributaries to the nutrient-rich mainstem of the Madeira River (C. Doria, personal observation). Whether hydropeaking effectively disrupts fish migrations remains unknown and merits further investigation. Also, species-specific studies are needed to identify whether observed hydropeaking rates can affect early life stages of fish species inhabiting the Madeira River downstream of the dams.

In summary, our study shows that hydropeaking occurs downstream of the dam. Understanding if and how this hydrological effect translates into social and ecological impacts will be critical to assess the need for mitigation and control strategies. Given the existence of a downstream gage with high-frequency flow data, early warning systems could be developed by dam operators in conjunction with local authorities to alert downstream human populations about abrupt water level changes. Operational protocols that reduce hydropeaking could mitigate its undesirable hydrological, geomorphological, ecological, and social effects on downstream reaches (Greimel et al., 2018; Moreira et al., 2019).

PROSPECTS

Brazilian energy planners and policy makers often advocate the prioritization of storage dams over run-of-river projects for energy security purposes (Abbud and Tancredi, 2010; Cerqueira, 2015), especially considering that the limited energy storage of Amazonian run-of-river dams is likely to get worse in light of climate variability (Stickler et al., 2013; Hunt et al., 2014; Lima et al., 2014). The push for storage dams in the Amazon River system could be facilitated by the current trend toward relaxation of environmental regulations that would make project selection less restrictive in Brazil (Almeida et al., 2016; Fearnside, 2016). Run-of-river dams are likely preferable designs in terms of downstream impacts because storage dams not only lead to hydropeaking, but also cause large-scale changes in seasonal flood pulses (Timpe and Kaplan, 2017). However, the electricity generation by run-of-river facilities is more susceptible to droughts, which could become more common with future increases in climate variability and deforestation in the Amazon basin (Marengo et al., 2009; Stickler et al., 2013; Arias et al., 2020). The effects of ongoing environmental changes on future Amazon hydroelectricity generation and the associated environmental impacts must be critically understood before new dams are constructed.

In conclusion, despite the potential for dams to alter downstream hydrology throughout the world (Lehner et al., 2011; Grill et al., 2019), a suite of other impacts must be factored into decisions about the optimal locations and design of new facilities (World Commission on Dams, 2000). Dam proposals must be evaluated in the context of their impacts on the overall river system, extending across national boundaries and including deltas and coastal waters into which rivers flow (Latrubesse et al., 2017). Uncoordinated construction of dams throughout the world has resulted in environmental impacts that could have been minimized through strategic basin-wide dam planning (Schmitt et al., 2018; Almeida et al., 2019b). New frameworks for watershed-wide, multi-objective optimization of dam planning have recently been proposed for major river basins of the world (Ziv et al., 2012; Schmitt et al., 2019), including the Amazon (Almeida et al., 2019b). It is imperative to consider potential hydrological effects along with other social and environmental impacts related to future Amazon dams.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation, to any qualified researcher.

AUTHOR CONTRIBUTIONS

RA, SH, ER, and FR conceived this study. RA, SH, and AyF analyzed the data. RA wrote the manuscript in close collaboration with SH and with substantive revision by all authors. All authors worked on the interpretation of the data.

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Hydropower Development and the Loss of Fisheries in the Mekong River Basin

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Development of large scale hydropower is proceeding rapidly in the Mekong basin without adequate consideration of the severe and cumulative impacts the dams and reservoirs will, and are already beginning to, have on biodiversity, livelihoods and the economies of the lower Mekong countries. Migratory aquatic species will be particularly affected, and global experience indicates that fishways proposed for large mainstream and tributary dams will not provide effective amelioration. An offset strategy of remediating small weirs, flood control devices, regulators and irrigation works on tributaries and flood plains is more likely to be an effective and economically efficient means of supplementing fisheries to compensate for the negative impact of mainstream dams. Mainstream hydropower developments may result in future stranded assets, high electricity costs and even threaten the sovereignty of lower Mekong countries.

Keywords: Mekong River, hydropower, fisheries, impacts, environmental offsets

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INTRODUCTION

Although the Mekong River system is known to support an extraordinary diversity of freshwater species and a globally significant fishery (Hortle, 2009a), development of hydropower in the basin proceeds apace (Geheb, 2018), with only scant consideration given to the biological resources being lost (Intralawan et al., 2018). In this commentary we outline key management issues and environmental consequences arising from the present trajectory for hydropower development in the Mekong basin. There are other environmental issues confronting the people of the Mekong basin which will also impact the river, including climate change, but at least over the next couple of decades the impact of dams already constructed and under construction is expected to far outweigh impacts arising from changing climate (Lauri et al., 2012; Ngo et al., 2016).

CHARACTERISTICS OF THE MEKONG SYSTEM

The Mekong is one of the most globally significant rivers (Campbell, 2009a). It is significant because of the large human population living in the basin and dependant on the river for their livelihoods both directly (e.g., fisheries, navigation, and water supply) and indirectly (e.g., the annual flood pulse around which farming is based, soil fertility, cultural values). It supports a remarkable diversity of fish (Valbo-Jørgensen et al., 2009), and freshwater gastropods. The Mekong River is one of the 20 largest rivers in the world in terms of discharge. It arises in the Himalayas, and flows through six countries in a politically sensitive region.

The total number of fish species present in the Mekong system will likely never been known. New species are being discovered, taxonomy is being revised, and estimates for species numbers vary depending on the inclusion or otherwise of estuarine fishes. Hortle (2009b) estimated there are at least 850 freshwater fishes in the Mekong. More recently, the Mekong River Commission's Fish Species Database lists 1,144 species, which includes marine visitors and estuarine fishes, in the river (MRC, 2020). This makes the Mekong second to only the Amazon for the variety of fish species present. Included in that fauna are several unusual and charismatic species such as the giant species of catfish, barb and stingray (Hogan et al., 2004; Valbo-Jørgensen et al., 2009). In addition to the fish, the Mekong is known to have at least 285 species of freshwater snails (Attwood, 2009), which constitute over 7% of the globally described species of freshwater gastropods (Strong et al., 2008), as well as other high profile aquatic species (e.g., freshwater dolphins).

The riverine ecosystem has supported what is believed to be the largest riverine fishery in the world. The fishery is estimated to yield approximately two million tons per year (Hortle, 2007; Hortle and Bamrungrach, 2015) with an annual value of US$11 billion (Nam et al., 2015). The annual fish harvest is equivalent to 17% of the annual global inland fisheries harvest of 12 million tons, and 2.4% of the global marine fish harvest of approximately 84 million tons (FAO, 2020).

Fisheries such as that of the Mekong have been difficult to describe in terms of economic importance, and are usually undervalued (Neiland and Béné, 2006; Baran et al., 2007). Most of the harvest is taken by subsistence fishers, who consume much of their own catch. When fish are traded it is partly through direct bartering with local people and partly through thousands of small local markets (Coates et al., 2003). Therefore, although large marine fisheries are conducted primarily by large fishing vessels operating through a relatively small number of well-established fishing ports allowing the catch to be comparatively easily documented and quantified, the Mekong is largely composed of artisanal fisheries. The full extent and importance of the fishery has only become evident as a result of extensive surveys and analyses conducted for the Mekong River Commission since 1995 (e.g., Hortle, 2007; Hortle and Bamrungrach, 2015) and meta-analyses of fish consumption data revealing the global "hidden harvest" from inland fisheries (Fluet-Chouinard et al., 2018).

IMPACTS OF HYDROPOWER DAMS ON THE MEKONG

The major impacts of dams, including hydropower dams, on riverine ecosystems have been well known for decades (Petts, 1984; Nilsson and Berggren, 2000; World Commission on Dams, 2000; Anderson et al., 2015). Generally, the two most important sets of impacts arise from the dam acting as a barrier to the movement of sediment and aquatic organisms such as fish and crustaceans, and the alteration to downstream flows. However, dams may also affect water quality and, through inundation, eliminate flowing water habitats.

In a large river such as the Mekong, which carries a substantial sediment load, the trapping of sediment has two important consequences (Kondolf et al., 2014). Firstly, a reduction in the downstream sediment supply has serious consequences for systems downstream that may depend on that supply. That may include riparian and floodplain systems that derive part of their nutrient supply from deposited silt, and deltaic systems that may change from depositional to erosional systems when sediment supply is reduced or even eliminated completely. Within the Mekong system it has been estimated that, should the full suite of proposed dams be constructed at the proposed locations, the sediment load currently transported to the delta would be reduced by 96% (Kondolf et al., 2014). This would result in both increased erosion and reduction in the area of the delta (Schmitt et al., 2017).

The consequences of a dam as a barrier to fish and aquatic life depend in part on the extent to which the species present undergo obligatory migrations. Concerns about dams as barriers to fish first became prominent in relation to salmonid fisheries in North America when populations of anadromous salmon were devastated when their migratory pathways were blocked by dams (Ferguson et al., 2011; Brown et al., 2013). Similarly, in the Mekong, it is known that many species undergo annual long-distance migrations as part of their breeding cycles, so the potential for dams to disrupt the fishery is high (Halls and Kshatriya, 2009).

It should be noted that dams act as a barrier to both upstream and downstream movement. Upstream movement is blocked because the organisms such as fish may be unable to pass over the spillway or through the power station turbines because the velocity of the current is too high. Downstream movement is impaired because drifting larvae, and even adult fish, are unable to find their way through the standing water of the impoundment to locate the outlet (Pelicice et al., 2015). Moreover, adult and juvenile fish which attempt to pass through hydropower infrastructure experience severe mortality from shear forces, physical strikes from turbine blades, and barotrauma or changes in barometric pressure (Algera et al., 2020).

The second pathway by which dams can impact riverine ecosystems is by alteration of downstream flow patterns (Petts, 1984). Although single-use hydropower dams do not divert water from the river channel, the flow pattern is altered with water usually being retained during the wet season and released during the dry – so wet season flows are reduced and dry season flows increased. When newly constructed dams are filling downstream flows may be reduced in both dry and wet seasons. Both of these patterns have been encountered in the lower Mekong (Hecht et al., 2018; Eyler et al., 2020). Many riverine species of fish and invertebrates have life cycles synchronized to river flow regimes. For instance, eggs and larvae (which are unable to swim or are poor swimmers) of many species are present during low flow periods when they are less likely to be washed downstream, so their life cycles and population recruitment are adversely affected when dry season flows are increased (Campbell, 2009b).

Thirdly, dams may alter the physical and chemical characteristics of the water downstream (Petts, 1984). Water released from the bottom of dams may be colder or contain lower concentrations of dissolved oxygen than normal river water. In addition, the impoundment of the dam, with still or slowly flowing warm clear water, may act as an incubator for algae which are then released downstream.

A final impact is the loss of habitat that occurs when a section of river is inundated and replaced by the standing water of the impoundment. In the case of the Mekong system a large number of dams have been constructed, and many more are under construction or planned (Greater Mekong Dams Observatory, 2020; **Figure 1**). Most of the hydropower systems proposed are "cascades" in which the pondage of each dam backs up virtually to the foot of the wall of the next upstream dam, eliminating riverine habitat entirely. The dam cascades under construction on the mainstream in Lao PDR, for example, will eliminate about 40% of the mainstream

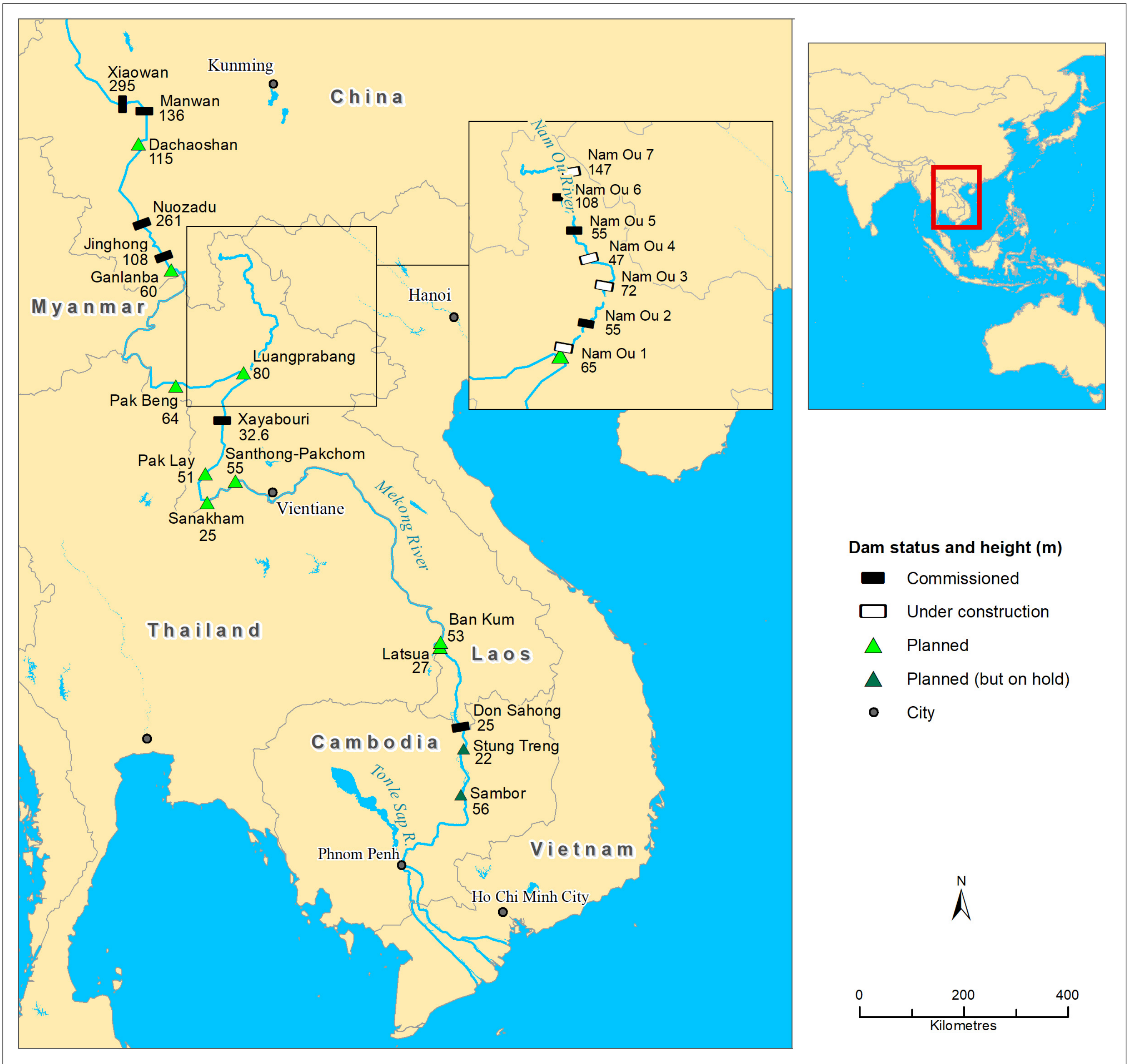

FIGURE 1 | Location of hydropower dams on the mainstream of the Mekong River and on the Nam Ou River. The dam height data are from the Greater Mekong Dams Observatory.

riverine habitat in Lao, which is about 25% of the mainstream habitat downstream of China. A similar cascade presently under construction is eliminating most of the mainstream habitat in China (Wei et al., 2017), and cascades are also flooding the habitats of the Nam Ou, the largest tributary in Lao PDR, and many other tributaries. The loss of habitat alone threatens many riverine species in the system.

The impoundments will support populations of fish and other aquatic species, but the species composition will be changed, and the size of the fisheries will be diminished. While newly created reservoirs often have productive fisheries in their first few years from excess nutrients resulting from decaying vegetation, the fisheries invariably decline to production levels less than the river fisheries which they replaced (Jackson and Marmulla, 2001).

VALUING HYDROPOWER AND FISHERIES

Analyses of the benefits of hydropower and the associated loss of ecosystem services have indicated negative economic consequences for the lower Mekong countries, especially Cambodia and Vietnam (Orr et al., 2012; Pittock et al., 2016; Intralawan et al., 2018; Yoshida et al., 2020), contrary to assessments undertaken by the Mekong River Commission (MRC, 2011). Intralawan et al. (2018) recommended that a new Lower Mekong Basin energy strategy be developed taking into account less hydropower income than previously anticipated, updated forecasts for LMB energy demand, and improved energy efficiency and renewable energy technologies (especially solar power). In light of these analyses, it is instructive to retrospectively examine the background to the relative importance given to hydropower development and fisheries by the governments of the lower Mekong countries.

The international consultants who were first involved in development proposals for the Mekong came predominantly from Europe and North America, where rivers and inland fisheries differed starkly from the Mekong. In those regions there were very few subsistence users of rivers, and the rivers primarily supported recreational fisheries. The only analogous situation in a developed region would be the rivers of north-western America that supported large salmon fisheries, and there the impacts of hydropower development had become highly controversial (Williams, 2008).

The Mekong fishery is diffuse, lacking large scale fishing ports, markets or processing plants, so consultants from developed countries making short term visits to the region largely failed to appreciate the importance of aquatic resources. However, since 1995, mainly because of the work of the fisheries program operated by the Mekong River Commission, more information has become available. For example, the value and importance of the fishery has been progressively identified, and that information has been published and passed to politicians and decision makers within governments in the region and international development agencies. Recent estimates by the MRC put the value of the capture fishery at US$11 billion per year per year (Nam et al., 2015).

One approach to considering the economic impact of hydropower development in the lower Mekong is to consider the fisheries value of the river per unit of river length. Although it is not possible to measure the lengths of rivers precisely because they are fractal values (see Campbell, 2009a), an estimate of the length of the Mekong main channel length from the China-Lao border to the sea is about 2500 km, measured from Google Earth. The fishery is not constrained to the main channel, so including the large tributaries we estimate that there is probably in the order of 7500 km of large river channel in the lower Mekong.

While migratory species would be those most impacted by hydropower dams, not all of the value of the Mekong fishery accrues from migratory fish. Halls and Kshatriya (2009) estimated that 38.5% of the total weight of fish species caught in fisher catch surveys was migratory. This equates to a total value for the migratory fish resource of $4.2 billion, or $565,000 per kilometer. The relative value is higher in the downstream flood-plain reaches where there is more fish production than in the upper, mountainous reaches. Nevertheless, as a generalization, for a dam that inundates say 100 km of river (which is approximately the length of the Xayaburi Dam pondage), the economic benefit from the dam would need to exceed $56 million per year before the dam produces a net positive economic value for the country.

Friend et al. (2009) discussed the reasons why the Mekong capture fisheries are undervalued in relation to hydropower. They identified four aspects: the fisheries were believed to be in an inevitable decline; that it was a marginal activity; that aquaculture could replace capture fisheries; and that there was a trade-off between fisheries and development.

We propose another issue to be considered is whether hydropower is overvalued in the Mekong. There are several reasons that suggest to us that this is a possibility. The first is the relentless sales campaigns that hydropower agencies have waged in the Mekong, and in other less developed regions with large rivers. Second, increasingly commonly proposed large hydropower projects in developed countries have been blocked as their environmental impacts were identified by local people resulting in intense political battles. Blocked in their own countries those organizations sought business elsewhere – often where people were not as aware of the issues and politicians were more tractable. Thirdly, independent analyses have found "overwhelming evidence that budgets are systematically biased below actual costs of large hydropower dams" (Ansar et al., 2014).

In the Mekong there has been a series of proposals dating back to organizations like the Tennessee Valley Authority from the United States, the Compagnie Nationale du Rhône from France (Mekong, 1994), the Snowy Mountains Engineering Corporation and Hydro Tasmania from Australia, and agencies from Canada, Norway, Thailand, and China. Hydro Tasmania started to seek other geographical regions where they could build hydropower dams following bitter political fights over Lake Pedder and the Franklin River in Australia. Similarly dam building companies in Thailand turned their sights to neighboring Mekong countries following political backlashes to proposed dams in southern Thailand, and the economic and public relations disaster of Pak Mun dam on a Thai tributary of the Mekong (Roberts, 1993). In China, there is an increasing resistance to large dams since the completion of the Three Gorges Dam, which many regard negatively. Politicians in Lao PDR in particular have been the recipients of decades of "hard sell" about the benefits of hydropower projects from both private and semi-government corporations seeking future large projects.

In turn, governments in the Mekong countries perceive at least three benefits of hydropower projects. The first is that hydropower projects are considered an important step on a pathway of large-scale industrialization as the primary mechanism to raise incomes and living standards. Second, large-scale development projects are used as evidence of modernity and thus improved status of the country in international discourse. Finally, large infrastructure projects provide abundant opportunities for corruption (e.g., Wells, 2015; Locatelli et al., 2017), and senior decision makers, typically the elites within countries, are well positioned for personal benefit (Andersen et al., 2020).

THE FAILURE OF PLANNING AND DESIGN

The current hydropower developments on the Mekong River are, in part, the outcome of two failures in planning and design. The first is a failure to conduct a basin-wide strategic environmental impact assessment. The second is the application of the cascade model of hydropower development.

The Mekong River Commission instigated a strategic environmental impact assessment of hydropower in the Mekong in 2009 (ICEM, 2010). However, the investigation was constrained by the insistence by member countries that only mainstream dams could be considered, and by the mandate of the Commission being restricted to the river downstream of China. More importantly, the two main recommendations arising from the assessment were not implemented. These were: that decisions on mainstream dams be deferred for 10 years to allow for rigorous and broad assessment of benefits and costs; and that the Mekong mainstream should never be used as a test case for proving and improving full dam hydropower technologies. Nevertheless, we note that basin-wide planning in multi-jurisdictional contexts anywhere in the world present intractable issues, with a history of contested or failed experiences (Campbell, 2007; Campbell et al., 2013).

The model of hydropower development being pursued in the Mekong is cascades of dams, where each dam spills directly into the next downstream pondage. This model, widely applied in China, maximizes the potential electricity yield, but it also maximizes the negative environmental and social impact of dams. Inclusion of environmental and social impacts in assessments often results in a negative net benefit.

The cascade model of dam building also brings into question the basic argument for fishways (including any structure built to assist upstream or downstream movement of fishes past a barrier), namely that they are to maintain migratory routes along the river continuum to enable natural recruitment dynamics (Lira et al., 2017). If connectivity and habitat availability is severely limited by multiple dams on a river, then the basic argument for fishways is devalued – fishways in these circumstances can become "ecological traps" (Pelicice and Agostinho, 2008). The rationale or need for fishways should be assessed on a systematic, basin wide scale, rather than on an individual dam basis (Lira et al., 2017; Birnie-Gauvin et al., 2018).

The overall impact of hydropower dams in the Mekong region needs to include consideration of both mainstream projects and those proposed for tributary streams. Tributary streams contribute a large proportion of the flow and sediment to the mainstream. They also contribute most of the habitat of fish and other aquatic organisms. One option for reducing the overall impact of hydropower developments would be to identify particular tributary streams to be maintained in an unregulated condition as refuges; and conversely, for other tributaries to be designated for hydropower development (Barlow, 2016). This approach was inherent in Ziv et al. (2012) analyses of the trade-offs between power generation and fisheries production for numerous combinations of dams on the mainstream and tributaries. They quantitatively demonstrated options for achieving specified power outputs while minimizing the loss of fisheries production, and that tributary dams in Lao and Cambodia would have graver impacts on fish biodiversity than the combined effects of the six mainstream dams above Vientiane.

Another approach for minimizing the impacts of hydropower development is to evaluate siting, design and operational features of proposed dams in conjunction with power generation, effectively giving equal footing to power output and ecological concerns. This approach was detailed for the proposed Sambor Dam in Cambodia, and could theoretically be coupled with basin wide planning to investigate acceptable boundaries around hydropower development and maintenance of riverine ecosystems (Wild et al., 2019). Presumably at least partly in response to the Wild et al. (2019) report, the Cambodian government has recently suspended plans for hydropower development on the mainstream of the Mekong in Cambodia (WWF, 2020).

WILL FISHWAYS HELP?

Fishways were first developed in the northern hemisphere as a means of reducing the impact of dams on commercially significant salmonid fishes (salmon and trout) (Katopodis and Williams, 2012; Birnie-Gauvin et al., 2018). All the anadromous species of Atlantic and Pacific salmon are strong swimmers, programmed to swim *en masse* to headwaters to spawn, with the progeny returning downstream as smolts or yearlings. Salmonids exhibit biological characteristics (predictable timing of runs upstream and downstream, powerful swimming ability, large size) which when coupled with supportive dam management (well-designed fishways, appropriate timing and quantity of water release, trapping and land transport of smolts past dams) have enabled them to successfully pass upstream and downstream of high dams. The fish passage technology has been developed through decades of research, especially since the 1940s, and billions of dollars in funding. Yet while most populations have survived, they are invariably at a fraction of the size of pre-dam populations (Williams, 2008; Brown et al., 2013; Birnie-Gauvin et al., 2018).

The situation is far more complicated in tropical rivers where multiple fish species are involved, varying in size at maturity from a few centimeters to well in excess of 1 m, migrating at different times of the year in response to various but often unknown environmental cues, and none of which have upstream swimming abilities comparable with salmonids. Downstream migration is usually by passive drift of larvae and actively swimming small juveniles, which are vulnerable to physical damage while moving over or through dam infrastructure.

The problems compound where there are multiple dams along a river. In Lao PDR there is a cascade of six dams planned between Vientiane in central Lao PDR and the northern border with China. If, say, 50% of a fish population ascends each fishway, then only 1.6% of the initial population will pass the most upstream dam. The cumulative effect for downstream migration is similar, and the impact magnifies for each successive generation. Modeling of fish passage scenarios for selected Mekong fishes indicated that to maintain viable populations, small bodied fish would need at least 60–87% success rate for a single dam, rising to 80–95% for two or three dams (Halls and Kshatriya, 2009). Populations of large bodied fishes would be extirpated, as adults are particularly susceptible to mortality when migrating downstream through turbines or over spillways (Halls and Kshatriya, 2009).

Fishways have been built on many hydropower dams outside of salmonid ranges, but we cannot find any reports documenting their successful application in securing long-term viability of populations of target species. Reports of lack of success abound (see reviews by Bunt et al., 2012; Noonan et al., 2012). The numerous inter-related biological, engineering and governance issues constraining successful fish passage development for high dams have been reviewed by Silva et al. (2018). Given the complexity of these issues, the need for case by case testing and application, and the limited financial resources available for fishway experimentation, it is not realistic to expect that they will be resolved in the foreseeable future.

Nevertheless, we note the considerable effort that the two most advanced hydropower development projects on the Lower Mekong (Xayaburi and Don Sahong) are currently devoting to fish passage investigations (Baumgartner et al., 2017). Xayaburi Power Company in northern Lao has utilized European consultancies to design and construct a complex fish passage facility at the cost of more than US$300 million. The company has now entered a public-private partnership with fish passage experts from Australia and the United States to monitor fish movements in the fishway and adapt operational measures to improve upstream and downstream passage.

The Don Sahong Power Company (DSPC – a Lao company) has contracted Sinohydro (a Chinese company) to build the hydropower dam on the Khone Falls in southern Lao. DSPC has published on its website many reports on social and biological aspects of the traditional fisheries in the area (DSHPP, 2020). One of the main areas of investigation has been the opening of alternative fish migration routes through the 11 km wide Khone Falls, as the dam is built on, and thus blocks, the historically important route through the Hou Sahong channel (Baird, 2011). The company has improved fish passage at several sites in five other channels, and has supported a government of Lao committee to remove more than 500 large illegal fishing gears to reduce obstruction of fish passage through those channels (Hortle and Phommanivong, 2019).

Despite the lack of evidence for the efficacy of fishways at high dams (especially for non-salmonids), fishways continue to be proposed by dam proponents and governments in the Mekong region as the preferred means to ameliorate the impacts of dams on fish and fisheries resources. Fishways, particularly for upstream passage, seem to be the social license necessary for dam approval. They enable a counterargument to those outlining the impacts of dams on migratory fishes. Moreover, they indicate good-will and social conscience on the part of developers, and they are physically impressive structures that engender support from management agencies and influential observers.

INVESTING IN ECOLOGICAL OFFSETS AS AN ALTERNATIVE TO FISHWAYS ON HYDROPOWER DAMS

Hydropower dams are the focus of water management impacts on aquatic ecosystems in the Mekong basin, including fisheries. But for fish ecology and production, arguably the collective impact of the multitude of small weirs, flood control devices, regulators and irrigation works on tributaries and flood plains in the basin (see details in Hortle and Nam, 2017) is similarly deleterious to that of hydropower dams. All these water management structures constrain fish migrations to varying degrees, and thus restrict the ability of fishes to access spawning, nursery or feeding habitats (Baumgartner et al., 2012). Because the structures are not physically obvious (they are usually less than 6 m head height) and, we proffer, not implemented by multi-national corporations, they have received comparatively little attention in the public discourse on human-induced impacts on the ecological functioning of the Mekong River.

Since the mid-2000s, a multi-disciplinary research and development program in the Mekong Basin (summarized in Baumgartner et al., 2019a) has: documented the ubiquity of water management structures (Marsden et al., 2014); demonstrated methods for enabling fish to pass low-head barriers (Baumgartner et al., 2019b, 2020); outlined socio-economic benefits of improved fish passage (Millar et al., 2019); developed a decision making tool for assessing benefits-costs of fishways on low-head barriers (Cooper et al., 2019); and demonstrated scale-out of results through irrigation infrastructure rehabilitation projects in southern and northern Lao, funded by the World Bank and the Asian Development Bank, respectively (Baumgartner et al., 2019a). The program continues to generate data on the scale-out and cost-benefit of fishways to reconnect aquatic ecosystems in the Mekong, and more recently has initiated similar work in Myanmar and Indonesia.

The demonstrated success of such interventions for aquatic ecosystem restoration at low-head barriers, coupled with a history of poor results for fishways at hydropower dams, brings to the fore the alternative of offsets (ecological remediation at sites remote from the hydropower dams) or compensation (payments to downstream affected communities) for the impacts of hydropower dams. Offsets or compensation could take many forms, but all would have the specific aim of a better environmental outcome for the equivalent amount of expenditure that would be required to build a fishway at a hydropower dam. Hortle and Nam (2017) reviewed possible mitigation measures for the impacts of dams on fisheries, and suggested that offsets should be considered as part of the cost of dam development and/or funded through the income from hydropower production. By way of example, the relative expenditures for fishways at high- and low-head barriers are stark – US$300 million for a fish passage facility at Xayaburi Dam, versus an outlay of US$800,000 for refurbishment of 10 weirs and road culvert barriers, including incorporation of fishways, in southern Lao (World Bank, 2015).

THE LONGER TERM OUTCOMES FOR THE LOWER MEKONG COUNTRIES

Over the next few decades there are a number of predictable consequences that will arise as a result of the development of hydropower in the Mekong. First, hydropower is expensive power. Costs of generating electricity from wind and, especially, solar are falling, with recent generation costs of US$ 0.03/kWh; while hydro generation is becoming more expensive with recent costs of US$ 0.04/kWh (IRENA, 2018). Within the Mekong, if the lost fisheries value is also factored in, hydropower will be substantially more expensive. Pumped hydro as a means of energy supply smoothing is widely advocated (ARENA, 2020), but the dams in the Mekong are, for the most part, too low to be suitable for that purpose. In a region like the Mekong, where the population is dispersed and local energy requirements are not usually intense, solar generation, which can produce energy on site and therefore does not require massive investment in poles and wires, also has an advantage in lower distribution costs. Countries relying on hydro as their source of power for industry will be at a decided economic disadvantage compared with those using solar power, and under pressure to provide subsidies to support the industry.

Many of the hydro projects in Lao PDR are public private partnership projects. They are constructed by private companies which will then operate them for a fixed period before handing them over to the national government. In the case of the Nam Ou dams and others in Lao the period of private operation is 30 years (Xaypaseuth Phomsoupha, 2015). By the time these projects are handed to the Lao Government many of the dams will contain large amounts of sediment that will at best reduce their efficiency of operation and at worst render them non-functional (e.g., Rădoane and Rădoane, 2005). The costs of refurbishments will be borne by the government.

The impact on sovereignty is frequently a concern for countries sharing international river basins. Tensions between upstream and downstream countries are the most common. Negotiations between Ethiopia and Egypt over dams on the Nile River (Mbaku, 2020), and India with both Pakistan and Bangladesh over the Indus and Brahmaputra River systems, are well known examples which continue to cause international tensions (e.g., see Beach et al., 2000).

In the context of the Mekong sovereignty issues have also been of concern. The MRC member countries have not allowed the MRC to consider issues relating to tributary streams because of potential impacts on national sovereignty. It is also widely believed that part of the reason that China has declined to join the MRC, and is reluctant to share data on flows from the upper Mekong, the Lancang, is because of sovereignty concerns.

In any international river basin countries sharing the basin are influenced by the actions of the others. The most obvious pathway of influence is through upstream countries impacting downstream flows, but in the Mekong, for example, navigation from the sea to Cambodia must pass through Vietnam. Developing and facilitating navigation agreements has been an important role of the MRC.

The construction and/or operation of multiple large dams within or by external jurisdictions continues to be a concern of the Mekong countries. In the upper Mekong, the storage capacity of the six most downstream dams in China is sufficient to store over 40% of the annual Mekong discharge at Chiang Saen in northern Thailand. China therefore has the capacity, if it wishes, to cut off the entire dry season flow at Chiang Saen. Flows would continue from tributaries downstream, but because Chinese government-owned firms control all the dams on Nam Ou and have controlling interests in many of the large dams on other tributaries, even those have the potential to be compromised. Such an outcome has been viewed as fanciful, but recent data indicate that China is prepared to exercise its control at the expense of downstream countries (Eyler, 2020), a most disturbing development.

DATA AVAILABILITY STATEMENT

The original contributions presented in the study are included in the article/supplementary material, further inquiries can be directed to the corresponding author.

AUTHOR CONTRIBUTIONS

Both authors made a substantial, direct and intellectual contribution to the work, and approved it for publication.

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A Modeling Assessment of Large-Scale Hydrologic Alteration in South American Pantanal Due to Upstream Dam Operation

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Natural river flow provides the conditions required to sustain freshwater ecosystems, and the greater the departure from the natural regime, the greater the loss of those ecosystems. In South America, new hydropower dams are continuously being constructed and planned in regions within and around the Amazon basin and in the Upper Paraguay river basin, a region notable for the Pantanal, a huge wetland ecosystem that is largely dependent on the flow regime of the Paraguay river and its tributaries. In this context, it is meaningful to examine the hydrological changes caused by the major Manso dam, that is operating since 2001 at the headwaters of one of the major tributaries of the Paraguay river. This was done for the same case study by other authors in previous studies using only gauging stations data. However, those previous assessments were limited due to the confounding effects of climate variability and the necessity of relatively long term observed time series. Here, we applied a modeling approach to evaluate the changes in hydrological regime caused by Manso dam operation. Our modeling approach was based on the combination of the MGB large-scale hydrologic model with the SIRIPLAN large-scale wetland model. The models were applied, using river reaches from 2 to 10 km, in two scenarios during the period from 2003 to 2015. In the first scenario we used naturalized streamflow at the dam site as input to the hydrological model. In the second scenario we used observed reservoir outflow time series as input to the hydrological model. Our results show that Manso dam has a regulation effect that decreases high flows, increases low flows and reduces lateral connectivity. The decrease in high flows is more pronounced in the region upstream of the Pantanal floodplain, but not limited to, while increase in low flows extends into Pantanal. Timing of maximum and minimum flows is less affected, except for the river reach immediately downstream of the dam. Our results improve the assessment of spatial patterns of hydrologic alteration, giving more confidence in the assessment of magnitude and spatial extension of the effects of Manso dam in the Pantanal region.

Keywords: Manso dam, indicators of hydrologic alteration, hydrological modeling, MGB, SIRIPLAN, Pantanal

INTRODUCTION

An underpinning assumption is that natural river flow provides the baseline for determining what is necessary for ecosystem maintenance, because ecosystems evolved under those conditions (Poff et al., 1997). In other words, a naturally variable regime of flow is required to sustain freshwater ecosystems, and the greater the departure from the natural regime, by water abstractions and flow regulation to maximize obtaining river goods, the greater the loss of those ecosystems (Poff et al., 2010).

River goods are defined as products that are of societal use when extracted or diverted from the river system (Brismar, 2002). Direct benefits or products include drinking, growing food, navigate, supporting industry and producing hydroelectric power. Besides, through river services, which are defined by Brismar (2002) as naturally generated and maintained processes by rivers that are of societal value, people can also collect indirect benefits, including recreation, soil wetting and fertilization of floodplains and deltas, cultural identity and ecosystem maintenance. Nevertheless, increases in water uses to maximize river goods normally result in decreased indirect benefits from river services. This occurs because river flow and water quality are major determinants of river ecosystem condition (Acreman, 2016).

Obtaining direct benefits from water by producing electric power in dams frequently leads to reduced indirect benefits (Ziv et al., 2012). Dams and reservoirs used for hydropower production impact ecosystems by river habitat fragmentation, habitat transformation from lotic to lentic, retention of sediments and nutrients and river flow alteration (FitzHugh and Vogel, 2011; Schmutz and Moog, 2018). Expanding hydropower have been considered one of the main emerging threats to freshwater biodiversity (Reid et al., 2019) and the alteration of river flow regimes for hydropower production is a critical factor responsible for decline in freshwater communities (Poff and Zimmerman, 2010). To put in a global perspective, according to Grill et al. (2019) only 37 per cent of rivers longer than 1,000 kilometers remain free-flowing over their entire length and 23 per cent flow uninterrupted to the ocean.

While in North America and Europe most dams have been constructed before the second half of the XX century, in countries with emerging economies the pace of dam construction is still high (Zarfl et al., 2015; Winemiller et al., 2016). In South America, for example, new hydropower dams are being constructed and planned in regions within and around the Amazon basin, leading to considerable concerns about the possible environmental consequences (Finer and Jenkins, 2012; Tundisi et al., 2014; Pavanato et al., 2016; Forsberg et al., 2017; Latrubesse et al., 2017; Timpe and Kaplan, 2017; Anderson et al., 2018; Fraser, 2018; Santos et al., 2018). Only in the Brazilian Amazon, over 200 new hydropower dams are predicted to be constructed in the next 30 years (Timpe and Kaplan, 2017). When considering small hydropower plants, Couto and Olden (2018) estimated that 82,891 small hydropower plants were operating or under construction in 150 countries and that another 181,976 new plants may be installed if all potential capacity were developed in the next decades, most of them in countries such as Russia, China, India, Brazil, and the United States. Despite being considered small, these dams also may have the capability to significantly alter the hydrological cycle, depending on its characteristic and the river they are located, especially considering hydrologic impact per megawatt (Timpe and Kaplan, 2017), to the point of affecting fauna and flora (Casas-Mulet et al., 2015; Bejarano et al., 2018; Mihalicz et al., 2019; Vehanen et al., 2020).

Regarding the expansion of the hydroelectric matrix in Brazil and its potential damages, besides the Amazon basin, another location where a large number of dams are planned is the Upper Paraguay river basin, a region that is notable for the Pantanal, a huge wetland ecosystem that is largely dependent on the flow regime of the Paraguay river and its tributaries. In this region, there is a debate about the potential impacts of new dam construction in the highlands (Planalto) on the ecosystems of the lower lying Pantanal (Bergier, 2013; Coelho da Silva et al., 2019; Medinas de Campos et al., 2020). In this context, it is worthwhile to carefully examine the hydrological changes that followed the construction of the major Manso dam, that started operation in 2000 at the headwaters of one of the major tributaries of the Paraguay river.

This task is not a completely original effort, since previous studies by Souza et al. (2009); Zeilhofer and de Moura (2009), and Timpe and Kaplan (2017) already analyzed hydrological changes downstream of the Manso dam. Zeilhofer and de Moura (2009) used streamflow data up to 2005, meaning that they analyzed only 4 years of data after the dam started to operate, while Souza et al. (2009) used slightly longer streamflow time series after the dam started to operate, by including data up to 2007. Both 4 and 6 years of data are normally considered to be unsatisfactory for the assessment of hydrological changes. Richter et al. (1997), for instance, propose a minimum of 20 years of pre and post-impact data. In the work by Souza et al. (2009), the authors tried to account for climatic variability by using benchmark gauging stations at rivers not influenced by Manso dam, but with limited success, since climatic variability plays an important role in the hydrological functioning of the region (Collischonn et al., 2001; Barros et al., 2004).

More recently, Timpe and Kaplan (2017) assessed hydrological regime changes due to the operation of several dams in Central and Northern Brazil, and showed that Manso dam has severe impacts on river regime, only second to the Balbina dam, on the Uatumã river, in the Amazon basin. However, their assessment was based on data from a single gauging station (Fazenda Raizama – ANA gauge 66231000) located just 15 km downstream of the dam. Therefore, the studies carried out on the impacts of the Manso dam have so far been restricted either in time, due to the low number of years used for the analysis of hydrological impacts, or in space, as they used few river gauges whose capacities to assess impacts are restricted to the locations where they are installed.

In order to overcome the limitations of previous studies, an alternative way to assess hydrologic alteration could be achieved by applying a distributed hydrological model with the ability of generating discharge time series at a multitude of points within the basin, and capable of representing scenarios with and without the dam operation in a realistic way. After proper parameter calibration, this model could then be applied using observed time series of rainfall, generating results of the two scenarios (with and without the dam) for the same period of time, thus taking complete control of the influence of climatic variability. A further advantage of this method is that it generates results not only at places where observation gauges exist, but at several ungauged sites too, allowing an assessment of spatial patterns of hydrologic alteration.

This model-based methodology was previously adopted by Ryo et al. (2015) who used a distributed hydrological model to quantify the spatial patterns of flow regime alterations along the Sagami River basin network under natural and altered flow conditions. The use of hydrological models was also suggested by Poff et al. (2010) as a part of the ELOHA framework for the definition of environmental flows and advocated by Richter et al. (1997) and Kennen et al. (2008). However, adopting the model-based assessment of hydrological change approach in the Upper Paraguay river basin is particularly challenging, due to the complexities of the physical system, with large floodplains and lakes, and very mild slopes, as discussed by Paz et al. (2010).

In the present paper, we took advantage of previously developed hydrological models in the Upper Paraguay river basin (Paz et al., 2011, 2014; Bravo et al., 2012), and in other environments with significant floodplains (Pontes et al., 2017; Fleischmann et al., 2018) and applied them to assess the magnitude and extension of fluvial regime changes imposed by the operation of Manso dam on rivers within the Pantanal, therefore supplementing previous assessments made by Souza et al. (2009) and Zeilhofer and de Moura (2009). Using computational modeling by combining the MGB large-scale hydrologic model with the SIRIPLAN large-scale wetland model, the effects of Manso dam on the hydrology of the Upper Paraguay River Basin could be identified continuously along the drainage network downstream. With this approach, we could assess hydrologic alteration in any segment of the river network, not only at gauges like on previous studies on the same dam. In addition, confounding climatic variability effects could be isolated since the only difference between the two simulated scenarios is the Manso dam operation. This differentiates the present work from previous studies about the influence of Manso dam operation on the hydrological regime of the Cuiabá river and also the fact that the present research is based on a larger discharge time series than the previous ones. Also, by representing the interaction between river and floodplain through modeling, we were able to better comprehend the impacts that a dam can have on the flow exchange between them and how far those impacts can go, which may be of interest for other locations upstream or in floodplains, not only in the Pantanal or Amazon region, where dams are operating or are intended to be installed.

Therefore, there are two main contributions of our paper. First, we define the magnitude, extent and spatial distribution of changes in the hydrological regime caused by a large dam in the Pantanal, which is a region of globally recognized ecological importance. This definition was made less influenced by other factors than previous studies, such as climate variability, and, therefore, makes it possible to analyze the influence of the dam's operation more clearly. Second, we present an innovative way to analyze changes in hydrological regime by using hydrologic and hydrodynamic models of rivers with extensive floodplains, and with complex drainage patterns. Although the use of distributed hydrological models to assess and quantify the spatial patterns of flow regime alterations along the drainage network of a river basin have been presented before by Ryo et al. (2015), our study extends this approach to the wider and much more complex environment of the Pantanal wetland.

MATERIALS AND METHODS

The South American Pantanal and the Upper Paraguay River basin

The Paraguay river is the major tributary of the Paraná river basin within the Paraná – La Plata river basin, draining an area of about 1.2 million km^2 (Collischonn et al., 2016). The Upper Paraguay region is the 600,000 km^2 basin upstream of the confluence of the Paraguay river with the Apa river, which marks the limits between Brazil and Paraguay (**Figure 1**). The region is within a seasonally dry tropical region, roughly defined by latitudes 14 and 33 South, and longitudes 53 and 60 West, and encompasses parts of three countries (Brazil, Bolivia, and Paraguay).

The landscape in the region is normally divided in two parts: the Planalto (Uplands) above nearly 200 m altitude; and the Pantanal (Wetland) below 200 m altitude. Rivers in the Planalto region are normally incised in bedrock, and flow into the Pantanal where they become alluvial (Assine et al., 2015). The Pantanal is an extensive low-lying region that was probably formed by subsidence followed by infilling with sediments from the Planalto, with 500 m of sediment deposits measured in some places (Assine et al., 2015). Deposition is still occurring, therefore, the Pantanal is a place of changing rivers with active and abandoned alluvial fans, which form a varied system (Assine et al., 2015). Due to this the Pantanal is a mosaic of variable flooded environments, including permanent lakes, water-filled depressions and small lakes that dry out seasonally, temporary flood drainages with sand or short grass and relatively permanent channels that connect flooded areas (Evans et al., 2010; Girard et al., 2010).

In terms of biodiversity, Pantanal is known for its numerous species and a rich diversity of aquatic and terrestrial ecosystems (Agência Nacional das Águas , 2004), being recognized by UNESCO as a World Biosphere Reserve and as a National Heritage by the National Constitution of Brazil. Pantanal presents a unique variety of fishes, although a considerable number have likely not been identified yet (Shimabukuro-Dias et al., 2017). Barletta et al. (2010) state that about 270 fish species have been reported in the Pantanal, while Petrere et al. (2002) cite estimates of 400 fish species. Both artisanal and recreational fisheries are mostly supported by large migratory species, and tourism fishing is gaining more importance in the last decades (Mateus et al., 2004; Barletta et al., 2010). Migratory movements comprise ascending displacements for spawning that can reach 400 km or even more for some species (Hahn et al., 2011; Barzotto and Mateus, 2017). These migrations are in phase with the rainy season, and the high fish productivity in the region is normally related to the seasonal inundation of floodplains (Barletta et al., 2010).

Over most of the region, rainfall in the six wettest months from October to March accounts for more than 80% of the annual total. Annual average precipitation varies from above 2000 mm in the Northeast of the basin to below 1000 mm in the West, along the border between Brazil and Bolivia (CPRM, 2011). Regularity of climate, with well-defined wet and dry seasons, together with the damping effect of floodplain inundation over the progress of the flood wave, results in a highly predicable flood pulse every year, with a single flood peak occurring in more or less the same time every year. This regular flooding and drying behavior, known as flood pulse, has strong influences on the regions ecological processes (Junk et al., 1989, 2006) and, also, strong consequences on human occupation of the Pantanal. This, together with access difficulties and isolation from major demographic centers hindered economic development and human occupation of the Pantanal. As a consequence, the region still has a very low demographic density and features unique natural landscapes, relatively untouched ecosystems, and traditional cultural practices and is known for its outstanding biological resources making it a priority for Brazilian and international conservation efforts (Junk and de Cunha, 2005; Schulz et al., 2019).

Flood waves formed in the Planalto rivers move relatively quickly to the Pantanal, where large portions of the water spread over the floodplain and deviate from the main channels through divergent drainage networks formed over alluvial fans (Paz et al., 2010; Assine et al., 2015). The overflow of the river channel and consequent floodplain inundation reduces peak discharges to more than one-half along the main rivers that flow into the Pantanal. These effects strongly modify the shape of hydrographs from upstream to downstream, as exemplified by Paz et al. (2010). Peak flows typically occur in February in the northeastern Planalto rivers, concomitantly with rainfall maximums. In the floodplains, on the other hand, flood peaks are delayed due to its slow movement, and typically occur in June, which corresponds to the dry season, in the middle of the Pantanal (Paz et al., 2010). The maximum area subject to inundation in the Pantanal, which include permanent open waters of river channels and lakes, is above 130,000 km^2, while the long-term mean inundation area is around 35,000 km^2, according to Hamilton et al. (2002).

Several studies relate the ecological functioning of the region to the fluvial regime, including floods, low water, timing of floods, and other abiotic variables related to hydrology. For instance, Bailly et al. (2008) stated that floods of the Cuiabá river, one of the most important tributaries of the Paraguay river, have an important role in the recruitment of species and influence spawning success as well as juvenile survival, and that floods are the principal trigger for the reproduction of many species of fishes. The overarching influence of the fluvial regime on the ecosystem was further highlighted in several other studies (Catella and Petrere, 1996; Petrere et al., 2002; Marchese et al., 2005; Costa and Mateus, 2009; Lourenço et al., 2012; Pinho and Marini, 2012; Ziober et al., 2012; Penha et al., 2015, 2017; Súarez and Scanferla, 2016; Wantzen et al., 2016; Barzotto and Mateus, 2017; Pereira and Súarez, 2018; Tondato et al., 2018; Santana et al., 2019).

Considering this strong dependency of the ecosystem on the natural hydrological regime in the Pantanal, it is expected that changes to the hydrological regime can severely impact ecosystem functioning, as reported in several other places in the world (Poff et al., 2010), including rivers in South America. River regime changes by dam operation have already impacted the ecosystem of the nearly located Paraná river basin, where flow regulation reduced the extent and duration of flood events, limiting or frustrating the reproductive processes of several fish species (Agostinho et al., 2004) and impacting fish populations (Agostinho et al., 2004, 2007, 2016).

According to the Brazilian Water Agency (ANA), there are currently 47 hydropower plants operating at the Upper Paraguay river basin, and more than 100 other hydroelectric facilities are currently in the proposed or construction stages (Medinas de Campos et al., 2020). Among the currently operating facilities, Manso dam is the one with the greatest potential of impacts on the hydrological regime of the Pantanal, due to its large reservoir, even though some regime changes have been reported for the relatively smaller Ponte de Pedra (Fantin-Cruz et al., 2015) dam.

The Manso Dam

Manso dam is a multipurpose dam that was conceived mainly for hydropower generation and flood control (Zeilhofer and de Moura, 2009; Paes and Brandão, 2013). The dam is located on the Manso River (**Figure 1**), one of the major rivers that form the Cuiabá River, at a point where drainage area is 9365 km^2, and average discharge is estimated at 170 m^3.s^{-1}. The reservoir has a total volume of 7.3×10^9 m^3 with an active storage capacity of 2.95 km^3 in the elevation range between 278 m (minimum operational level) and 287 m (maximum normal operation level), and additional 0.45 km^3 for flood control in the elevation range between 287 and 288.15 m (Paes and Brandão, 2013). The reservoir regulates the flow of the Manso river and of the Cuiabá river downstream. The dam is located almost 300 km upstream of the cities of Cuiabá and Várzea Grande, which are the main targets for flood control. Downstream from the city of Cuiabá, the Cuiabá river drains into the Pantanal, where it connects to lakes and the floodplain through side channels. The confluence of the Cuiabá and Paraguay river is located well within the Pantanal, at about 900 km downstream of the Manso dam.

The reservoir of Manso dam begun to be filled in November 1999, and the dam started operation in November 2000 (Furnas Centrais Elétricas, 2002). The reservoir was scheduled to be filled by the end of 2000 but due to lower than expected precipitation the complete filling only occurred in February 2002, when it reached the maximum level of 287.5 m (Shirashi, 2003).

Hydrological Modeling

We assessed hydrologic alteration caused by Manso dam operation in the Pantanal region using synthetic streamflow time series obtained by a combination of two hydrologic models and the Indicators of Hydrologic Alteration (Richter et al., 1996) metrics.

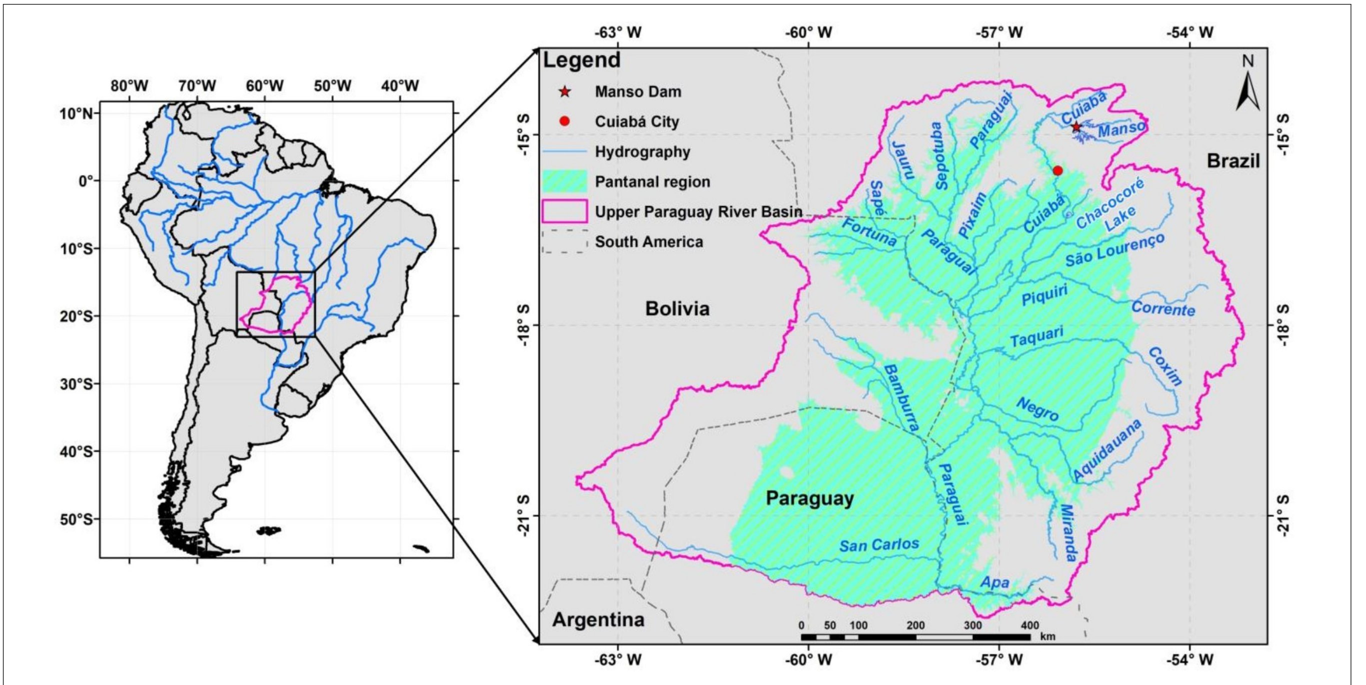

FIGURE 1 | The Upper Paraguay river basin and the Pantanal wetland.

We divided the Upper Paraguay river basin in two regions, following a threshold altitude of 200 m (Assine et al., 2015). In regions above 200 m we applied the MGB large-scale hydrological model (Collischonn et al., 2007), further described in Section "Hydrologic Model of the Upper Basin (Planalto), Input Data, and Scenarios." In regions below 200 m, where the drainage network is not dendritic, and where there are several interconnected lakes, temporary inundated areas and secondary channels, we applied the coupled 1D/2D SIRIPLAN model (Paz et al., 2011), further described in Section "Hydrologic Model of the Lower Basin (Pantanal)." The two modeling sub-regions of the basin are shown in **Figure 2** along with the main river gauges used during the calibration of one model or another.

The MGB model of the upper basin was applied first, and output of this model was introduced as boundary conditions for the SIRIPLAN model, which was applied in sequence, at the nine most important inflow points to the Pantanal, as shown by the gauges in red in **Figure 2**. The blue gauges inside the SIRIPLAN modeling region were used for calibration of this model.

MGB was applied in two scenarios: with and without Manso dam operation, applying methods described in Section "Hydrologic Model of the Upper Basin (Planalto), Input Data, and Scenarios." Results of the MGB model in the two scenarios were then used as boundary conditions (input variables) to the SIRIPLAN model. **Table 1** presents the characteristics of the main river gauges used for calibration and the performance measures Nash-Sutcliffe, used for calibration of maximum flows, Nash-Sutcliffe of the logarithm of the flows, used for calibration of minimum flows, and the volumetric error (BIAS). Further details on the calibration can be found in the work by Paz et al. (2011).

Both MGB and SIRIPLAN are distributed models and provide results of daily streamflow and water stage at river segments of 2 to 10 km covering the whole drainage network of the Upper Paraguay river basin, at least for rivers with drainage area above 20 km^2. Time series covering the period from 01/01/2003 to 31/12/2015, considering the scenarios with and without Manso dam, were then analyzed using the Indicators of Hydrologic Alteration (Richter et al., 1996). These metrics include mean monthly flows, minimum and maximum river discharge values for 1, 3, 7, 30 and 90 days, Julian day of the maximum and minimum flows, as further described in Section "Assessment of Hydrological Change."

Therefore, for each river segment, two different IHA metric values were obtained, one for each scenario. These two values were subsequently compared and the magnitude of change was estimated as a relative departure of the original value for the majority of the metrics, or as an absolute deviation in the case of the Julian day of minimum and maximum flows. This methodology allowed us to obtain results of the spatial variability of dam impact.

Hydrologic Model of the Upper Basin (Planalto), Input Data, and Scenarios

MGB is a semi-distributed, process-based model developed for large-scale to continental regions. It was first presented by Collischonn et al. (2007), and subsequently improved by Paiva et al. (2011); Pontes et al. (2017), and Fleischmann et al. (2018). The model was used before in several large basins of South America, including assessments of climate change impacts in the Amazon (Sorribas et al., 2016); potential impacts of dams on fluvial ecosystems (Forsberg et al., 2017); hydrological reanalysis

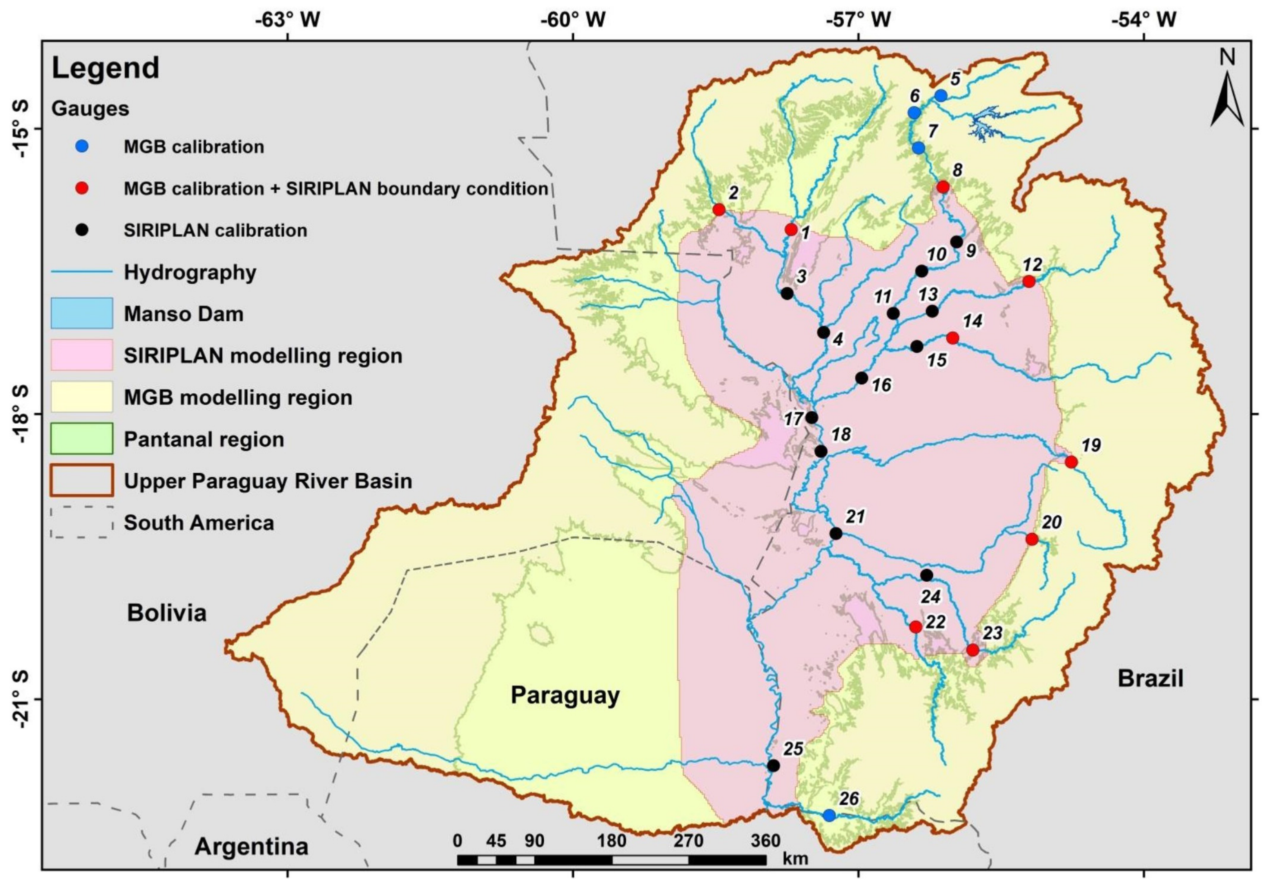

FIGURE 2 | Modeling sub-regions of the Upper Paraguay basin (yellow = MGB model; pink = SIRIPLAN model) and gauges used for calibration (red points are boundary condition locations where streamflow time series calculated with MGB were transferred as input data to SIRIPLAN).

(Wongchuig et al., 2019); water management scenarios in transboundary river basins (Gorgoglione et al., 2019); streamflow forecasting (Fan et al., 2015); and continental hydrological modeling for South America (Siqueira et al., 2018).

In its most recent version, MGB divides the river basin in relatively small unit-catchments. This unit-catchments are defined based on the contribution areas that drains to river segments, which are divided with constant length (Siqueira et al., 2016). Within each unit-catchment, Hydrological Response Units (HRU's) are defined based on soil type and land use, and for each one the water and energy budget is computed through the soil-vegetation system, as described by Collischonn et al. (2007). Surface, subsurface and groundwater outflows from water balance are routed to the main river of the unit catchment using linear reservoirs, while flow propagation through drainage networks is computed using either the Muskingum-Cunge method or 1D hydrodynamic equations (Pontes et al., 2017). To access the influence of dams and reservoir operation within MGB it can be done by introducing internal boundary conditions at the dam location, and forcing the model with observed reservoir outflow data, or by specifying reservoir operation rules (Fleischmann et al., 2019).

In order to do so for the Manso Dam case the model was calibrated using precipitation and river discharge data obtained mainly from the National Water Agency (ANA) in Brazil. We used data from 153 precipitation gauges and a total of 42 streamflow gauges distributed over the whole Upper Paraguay river basin to calibrate the model, focusing on the Planalto region. The main gauges are presented in **Figure 2**. Rainfall information outside Brazil was obtained from the Multi-Source Weighted-Ensemble Precipitation (MSWEP) product, that is based mainly on satellite precipitation estimates.

Operational data of the Manso dam were also obtained from ANA, including observed outflow time series and naturalized time series from 01/01/2001 to 31/12/2015. Naturalized streamflow is an information routinely estimated for the main hydropower reservoirs in Brazil by ANA and ONS (Operador Nacional do Sistema) (Agência Nacional da Água, 2011) and corresponds to streamflow that would be observed at the dam site if there was no flow regulation by reservoirs and no water abstractions upstream.

After model calibration and verification MGB was applied in two scenarios during the period from 01/01/1985 to 31/12/2015. In the first scenario, intended to represent the natural condition, we used naturalized streamflow at the dam site as input to the

TABLE 1 | Characteristics of the main river gauges presented in **Figure 2**, used during calibration of the MGB and SIRIPLAN models, and the performance measures obtained.

Gaude ID	Gauge Code	Gauge name	Drainage area (km^2)	River	Nash	Nash-log	BIAS (%)
1	66070004	Cáceres	32400	Paraguai	0.816	0.851	2.0
2	66072000	Porto Esperidião	5660	Jauru	0.402	0.502	−13.6
3	66090000	Descalvados	47100	Paraguai	0.910	0.920	−5.0
4	66120000	Porto Conceição	64000	Paraguai	0.630	0.620	7.6
5	66160000	Quebó	4260	Cuiabá	0.601	0.789	−25.1
6	66250001	Rosário Oeste	16000	Cuiabá	0.709	0.784	−6.5
7	66255000	Acorizal	19700	Cuiabá	0.758	0.825	−3.8
8	66260001	Cuiabá	23500	Cuiabá	0.748	0.821	−5.5
9	66280000	Barão de Melgaço	28900	Cuiabá	0.940	0.970	−5.8
10	66340000	Porto Cercado	36900	Cuiabá	0.910	0.920	−4.6
11	66360000	São João	38500	Cuiabá	0.820	0.840	−8.8
12	66460000	Acima do Córrego Grande	23000	São Lourenço	0.588	0.823	−1.0
13	66470000	São José do Boriréu	24100	São Lourenço	0.920	0.940	4.9
14	66600000	São Jerônimo	23300	Piquiri	0.581	0.665	−6.8
15	66650000	São José do Piquiri	30000	Piquiri	0.750	0.820	8.9
16	66750000	Porto Alegre	103000	Cuiabá	0.820	0.850	8.3
17	66800000	Amolar	234000	Paraguai	0.670	0.720	6.3
18	66810000	São Francisco	243000	Paraguai	0.700	0.730	−2.0
19	66870000	Coxim	27600	Taquari	0.326	0.531	−7.3
20	66886000	Perto da Bocaina	2840	Negro	−1.034	−0.124	32.9
21	66895000	Porto da Manga	327000	Paraguai	0.820	0.760	2.5
22	66910000	Miranda	15000	Miranda	0.273	0.579	15.7
23	66945000	Aquidauana	15700	Aquidauana	−0.403	0.491	19.4
24	66950000	Porto Ciriaco	17200	Aquidauana	0.760	0.830	−3.5
25	67100000	Porto Murtinho	576000	Paraguai	0.610	0.650	−6.1
26	67170000	São Carlos	10200	Apa	0.484	0.620	11.0

hydrological model. In the second scenario, intended to represent the impact condition, we used observed reservoir outflow time series as input to the hydrological model. In those two scenarios the rainfall-runoff processes upstream of the Manso dam have been turned off, since the dam was represented as a boundary condition. However, in the remaining area of the basin, the rainfall-runoff and flood routing processes have been ordinarily represented by the model.

Hydrologic Model of the Lower Basin (Pantanal)

Outputs from the hydrological model described in Section "Hydrologic Model of the Upper Basin (Planalto), Input Data, and Scenarios" were introduced as inputs for the hydrologic hydrodynamic model of the Pantanal wetland, called SIRIPLAN (Paz et al., 2011, 2014). SIRIPLAN is composed by a 1D hydrodynamic model based on a solution of the full Saint Venant equations for river networks (Tucci, 1978) coupled to a 2D raster-based inundation model, similar to the LISFLOOD-FP model (Bates et al., 2010).

The 1D model simulates the flow routing along the river drainage system, considering cross sections restricted to the main channels. The raster-based model simulates the water accumulation and the 2D propagation of inundation over the floodplains. A water exchange scheme is used to simulate channel outflows to the floodplain and from the floodplain back into the channel (Paz et al., 2011). Additionally, the vertical processes of precipitation, evapotranspiration and infiltration in the floodplain are simulated in the 2D part of SIRIPLAN, following the methods described by Paz et al. (2014).

SIRIPLAN was previously applied in the same region by Paz et al. (2011, 2014). A total of 3965 km of main river channels and 219,514 km^2 of floodplains were represented by the model. Rivers were discretized in 2 km long computational reaches, and floodplains were divided in 46,741 square-grid elements of 0.02 × 0.02 degrees (approximately 2 × 2 km). The model was calibrated by comparing discharge hydrographs and water level time series at several gauging stations on the rivers Paraguay, Cuiabá, and other tributaries, and satisfactorily reproduced the hydrological regime in most of the basin. SIRIPLAN is arguably the most detailed and accurate hydrologic-hydrodynamic model that has been applied to the whole Pantanal.

In addition to daily river discharge and water level time series along the main rivers and over the floodplain, SIRIPLAN's outputs include the amount of water that is delivered from the rivers to the floodplains, and backward. This feature is particularly interesting in the present context of evaluating the hydrological changes due to the operation of an upstream located dam, since hydrological changes may be not limited to river discharges, but possibly include river-floodplain interaction.

TABLE 2 | Indicators of Hydrologic Alteration (IHA) flow regime descriptors (from Richter et al., 1996).

Main IHA groups	Descriptors of river regime
Group 1. Magnitude of monthly water conditions	Mean discharge for each calendar month (12 descriptors)
Group 2. Magnitude and duration of annual extremes	1-day-minimum flow
	1-day-maximum flow
	3-day-minimum flow
	3-day-maximum flow
	7-day-minimum flow
	7-day-maximum flow
	30-day-minimum flow
	30-day-maximum flow
	90-day-minimum flow
	90-day-maximum flow
Group 3. Timing of annual extremes	Date of 1-day maximum flow
	Date of 1-day-minimum flow
Group 4. Frequency and duration of high and low pulses	Annual number of high pulses
	Annual number of low pulses
	Mean duration of high pulses (days)
	Mean duration of low pulses (days)
Group 5. Rate and frequency of change in conditions	Mean daily flow increase
	Mean daily flow decrease
	Number of reversals

Assessment of Hydrological Change

For the assessment of hydrological change, we examined results from the natural and altered hydrological scenarios using the 28 out of the 32 descriptors of river regime proposed by Richter et al. (1996) in their Indicators of Hydrologic Alteration (IHA) methodology. The IHA descriptors are biologically relevant hydrological indices that cover flow magnitude, frequency, duration, timing, and rate of change believed to be essential to ecosystem health (Poff et al., 1997), and are given in **Table 2**. Among the 32 IHA descriptors, we observed that the four metrics related to high and low flow pulses were hypersensitive to small changes in the hydrographs due to the way the low flow and high flow thresholds are defined, so we skipped all those four descriptors of group 4 (**Table 2**).

All the 28 metrics of groups 1, 2, 3, and 5 were obtained for every year along the simulation period (from 2003 to 2015) for the two scenarios (with and without dam operation), and its average value (mean of the 12 simulation years) in the two scenarios were compared to assess the magnitude of change, following equations 1 (for groups 1, 2, and 5) or Eq. 2 (for group 3).

$$A = \frac{X_D - X_0}{X_0} \tag{1}$$

$$A = \frac{J_D - J_0}{\left(\frac{365}{2}\right)} \tag{2}$$

were X_0 is the value of the descriptor in the reference scenario (no dam operation); X_D is the value of the descriptor in the altered scenario (with dam operation); J_0 is the Julian day of the maximum or minimum in the reference scenario; J_D is the Julian day of the maximum or minimum in the altered scenario.

We understand that applying Eq. 1 to changes in timing would lead to the wrong conclusion that a 20 days change in events typically occurring earlier during the hydrological year (lower Julian day values) represents a larger relative change than a 20 days change (delay or anticipation) occurring later in the hydrological year. Therefore, we used Eq. 2 to calculate relative change in minimum and maximum dates, as a substitute of Eq. 1, because we understand that a 20 day change should represent the same relative change, independently of the date in the reference scenario (J_0), and that the worst change in timing would be a delay or anticipation of half year (or 365/2 = 182.5 days), and this should be equivalent to 100% change.

Besides, to calculate average Julian dates we used circular statistics. Circular statistics is a subfield of statistics, devoted to the development of statistical techniques for data on an angular scale, for which there is no designated zero, and the designation of high and low values is arbitrary, such as the direction of wind, the time of the day, or the day of the year, as in statistics of group 3 (Berens, 2009).

Finally, we compared results of the 28 descriptors at all river computational elements of the model downstream of the dam, resulting in 857 points where the IHA descriptors were compared. In order to present the results in a synthesized way and also in the form of maps, the averages of the alteration of all the descriptors within the same group were performed for each simulated model element.

RESULTS

River discharge time series were calculated at river reaches 2 to 10 km long over the whole Upper Paraguay river basin during the period from 2003 to 2015, considering two scenarios: with and without dam operation. In the first scenario, intended to represent the natural or reference condition, we used naturalized streamflow at the dam site as input to the hydrological model. In the second scenario, intended to represent the impact condition, we used observed reservoir outflow time series as input to the hydrological model.

Hydrologic Alteration Along the Main River

In order to evaluate the changes between the two scenarios, hydrographs where plotted at six locations downstream of the dam (**Figure 3**). The results show that just downstream of the dam, the regulation effect of Manso dam operation clearly reduced high flows and increased low flows (**Figure 3B**). Low flows are increased from about 60 $m^3.s^{-1}$ to circa 130 $m^3.s^{-1}$, while maximum flows are reduced from up to 2,100 $m^3.s^{-1}$ to below 900 $m^3.s^{-1}$ at all times. The timing of maximum flows was also altered, with maximum flows occurring 3 to 17 days later in the scenario with dam operation.

By Cuiabá city, nearly 300 km downstream of the dam, and also the point where the Cuiabá river leaves the upper region

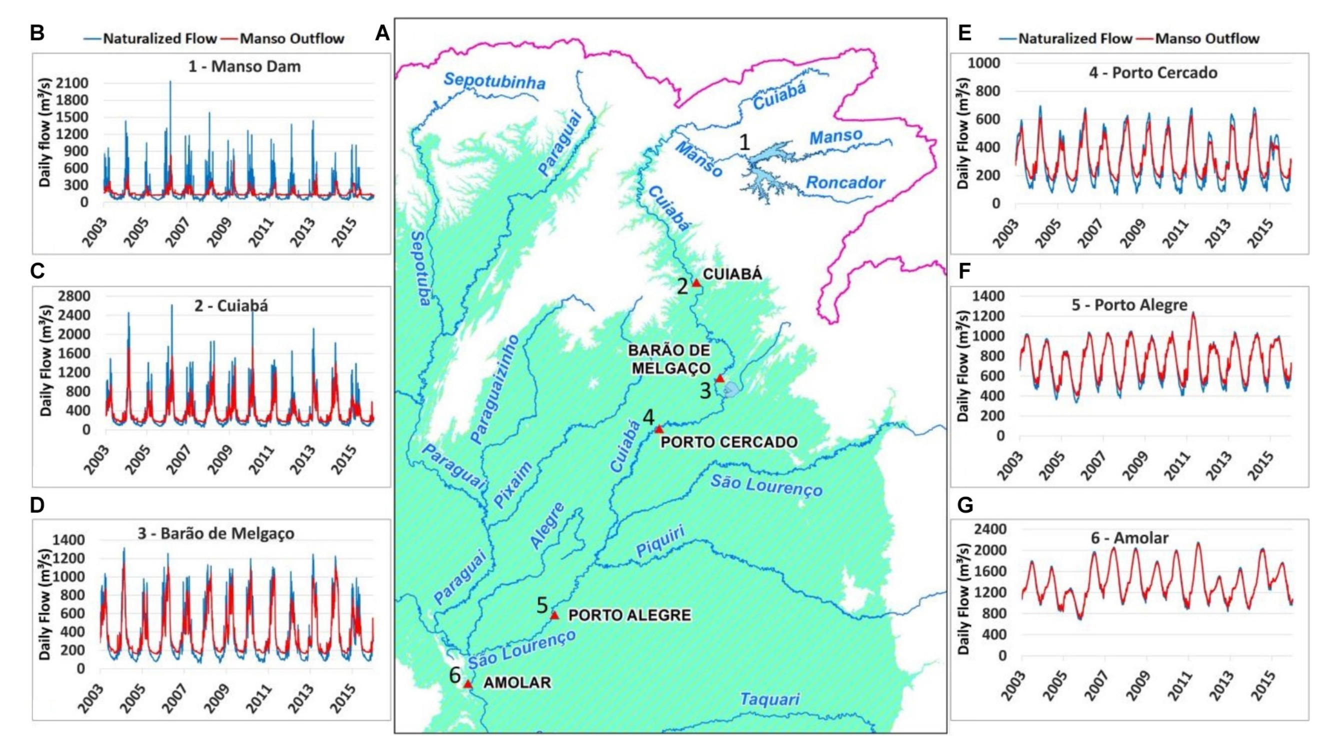

FIGURE 3 | Location **(A)** of the hydrographs produced after simulation with naturalized flow (blue) and Manso outflow (red) as input condition at: Manso dam **(B)**, Cuiabá gauge **(C)**, Barão de Melgaço gauge **(D)**, Porto Cercado gauge **(E)**, Porto Alegre gauge **(F)**, and Amolar gauge **(G)**.

and flows into the Pantanal, the same pattern of alteration is still clearly visible (**Figure 3C**). Low flows are increased from about 93 to 170 $m^3.s^{-1}$, while maximum flows are reduced from the range 1,390–2,617 $m^3.s^{-1}$ to below 1,723 $m^3.s^{-1}$ at all times and can be as low as 650 $m^3.s^{-1}$. The timing of maximum flows was also altered, with maximum flows occurring 2 to 5 days later in the scenario with dam operation.

At Barão de Melgaço, approximately 420 km downstream of the dam, the hydrograph shows that changes in low flows are still clearly visible with an increase from about 90 $m^3.s^{-1}$ in the scenario without dam operation to about 165 $m^3.s^{-1}$ in the scenario with dam operation (**Figure 3D**). Changes in maximum flows, on the other hand, are less marked. This occurs because maximum flows are naturally decreased while the flood wave moves from Cuiabá to Barão de Melgaço, due to river outflowing to the floodplains, mainly through distributary channels. Maximum river discharges at Barão de Melgaço are in the range of 980 to 1,315 $m^3.s^{-1}$ in the scenario without dam operation and drop to about 750 and 1,140 $m^3.s^{-1}$ at the same events in the scenario with dam operation. Changes in the timing of maximum flows at Barão de Melgaço due to dam operation are lower than 5 days.

Circa 550 km downstream of the dam, at Porto Cercado, and already well within the Pantanal, changes in low flows are still clearly visible, while changes in maximum flows are slightly perceived. At this location, changes in timing of maximum flows are practically absent (**Figure 3E**). The same behavior is observed at Porto Alegre (**Figure 3F**), located 825 km downstream of the dam. At this point, dissemblance between the hydrographs are only clearly perceptible at low flows but, even so, with small differences. Finally, at Amolar, after the confluence with the Paraguay river, there is practically no difference between the hydrographs for both scenarios (**Figure 3G**).

It is clear in all IHA groups that changes are higher just downstream of the dam, and progressively decrease downstream, as the Cuiabá river flows into the Pantanal (**Figure 4**). This is due probably to both the entry of large tributaries into the impacted network, as is the case of the São Lourenço river already within the Pantanal, and due to river-floodplain interaction. It is noticeable at **Table 3**, which summarizes the results found regarding IHA statistics for the locations previously analyzed (**Figure 3**), that in January (wet season) average flows decrease more than 44% just downstream of the dam (point 1), more than 22% at Cuiabá (2) and more than 13% at Barão do Melgaço (3). August/September (dry season) average flows increase more than 140% at the dam (1), and more than 70% as far as Porto Cercado (point 4), located more than 500 km downstream of the dam.

Minimum flows increase substantially due to dam operation as far as Porto Cercado (point 4), and are still present as far as Amolar (point 6), on the Paraguay river. Changes in timing of both minimum and maximum flows are large just downstream of the dam (point 1), but decrease rapidly with distance downstream. Rates of change decrease at all six points downstream of the dam, with greater values close to the dam, and lower values as the Cuiabá river enters the Pantanal. Nevertheless, decreases of the rates of change larger than 5% are still present at

TABLE 3 | Summary of relative changes in average IHA descriptors at reference points.

Changes in parameters of the indicators of hydrologic alteration		Location					
		Manso dam	**Cuiabá**	**Barão do Melgaço**	**Porto Cercado**	**Porto Alegre**	**Amolar**
Group 1. Magnitude of monthly water conditions	Mean January flow	−44.6%	−22.7%	−13.7%	−6.5%	0.9%	0.5%
	Mean February flow	−43.3%	−19.9%	−13.5%	−8.3%	−0.7%	−0.2%
	Mean March flow	−26.8%	−12.1%	−10.4%	−8.8%	−1.1%	−0.2%
	Mean April flow	−0.6%	−1.4%	−6.7%	−8.2%	−1.7%	−0.6%
	Mean May flow	47.2%	23.1%	7.5%	−1.7%	−1.9%	−1.5%
	Mean June flow	73.8%	39.2%	26.4%	10.7%	−0.7%	−1.7%
	Mean July flow	115.7%	61.5%	47.0%	29.9%	2.0%	−1.1%
	Mean August flow	146.0%	79.9%	71.4%	56.7%	7.7%	0.8%
	Mean September flow	124.2%	75.2%	78.0%	71.2%	13.8%	3.3%
	Mean October flow	69.5%	42.1%	52.7%	53.0%	12.4%	4.0%
	Mean November flow	16.1%	9.7%	20.8%	21.4%	7.7%	3.2%
	Mean December flow	−27.1%	−13.8%	−8.1%	1.8%	4.7%	2.3%
Group 2. Magnitude and duration of annual extremes	1-day-minimum flow	196.5%	111.4%	108.0%	100.2%	16.8%	3.5%
	1-day-maximum flow	−66.1%	−34.3%	−13.7%	−9.0%	−1.7%	−1.8%
	3-day-minimum flow	209.4%	111.4%	107.0%	99.2%	16.7%	3.5%
	3-day-maximum flow	−59.5%	−31.8%	−13.3%	−9.0%	−1.7%	−1.8%
	7-day-minimum flow	196.5%	106.8%	103.2%	95.5%	16.4%	3.4%
	7-day-maximum flow	−48.9%	−26.1%	−12.3%	−8.9%	−1.7%	−1.8%
	30-day-minimum flow	155.4%	85.5%	83.1%	76.3%	14.4%	3.1%
	30-day-maximum flow	−37.1%	−18.9%	−11.4%	−8.8%	−1.8%	−1.8%
	90-day-minimum flow	125.8%	73.0%	70.6%	62.8%	12.0%	3.0%
	90-day-maximum flow	−34.6%	−16.8%	−11.3%	−8.5%	−1.5%	−1.5%
Group 3. Timing of annual extremes	Date of 1-day maximum flow	−65.8%	−1.6%	−0.5%	1.1%	3.3%	1.1%
	Date of 1-day-minimum flow	18.6%	−1.6%	2.7%	−1.6%	−2.7%	0.0%
Group 5. Rate and frequency of change in conditions	Mean daily flow increase	−78.4%	−45.0%	−32.2%	−33.2%	−4.7%	−5.8%
	Mean daily flow decrease	−81.8%	−51.4%	−34.6%	−34.7%	−11.8%	−7.6%
	Number of reversals	18.1%	14.8%	8.9%	10.1%	0.2%	0.1%

Amolar (point 6), which is on the river Paraguay, and is nearly 900 km downstream of the dam. Finally, the number of flow reversal increases as far as Porto Cercado (point 4).

Figure 4 shows how Manso dam operation changes the average descriptors from IHA along the rivers of the Upper Paraguay basin. These maps suggest that the effects of Manso dam operation are practically dissipated downstream of the points where the Cuiabá river meets the São Lourenço and Piquiri rivers.

Changes in River-Floodplain Interaction

To investigate how Manso dam operation affects the river-floodplain interaction we investigated hydrographs of the outflows from the river to the floodplain. The river-floodplain exchange flow rates simulated in both scenarios were obtained from SIRIPLAN model, with and without the operation of the dam. We selected the approximately 140 km reach between Cuiabá (point 2) and Barão do Melgaço (point 3), which was described by Paz et al. (2014) as a reach where the river mainly loses water to the floodplain, and summed all outflows from the river to the floodplain, resulting in hydrographs of river-floodplain interaction over the entire 140 km reach. River-floodplain interaction is highest during the wet season, and to illustrate this interaction we selected two outflow events that occurred in January/February: one for a relatively wet year (2010) and one for a relatively dry year (2012).

Along the reach between Cuiabá and Barão do Melgaço, from 22 February to 07 March 2010 in both scenarios (with and without Manso dam operation) the peak outflow from the river to the floodplain is above 1200 $m^3.s^{-1}$ in the reference scenario (no dam operation) while it is lower than 700 $m^3.s^{-1}$ in the altered scenario (**Figure 5A**). Hydrograph volume is also lower in the altered scenario by nearly 40%, meaning that river-floodplain interaction is weaker. Meanwhile, from 25 January to February 1st 2012 the outflow peak in the scenario without Manso is close to 600 $m^3.s^{-1}$, while in the altered scenario (with Manso dam operation) there is no outflow from river to floodplain at all (**Figure 5B**).

The same pattern shown in the two described events occur every year, with outflow hydrographs being reduced

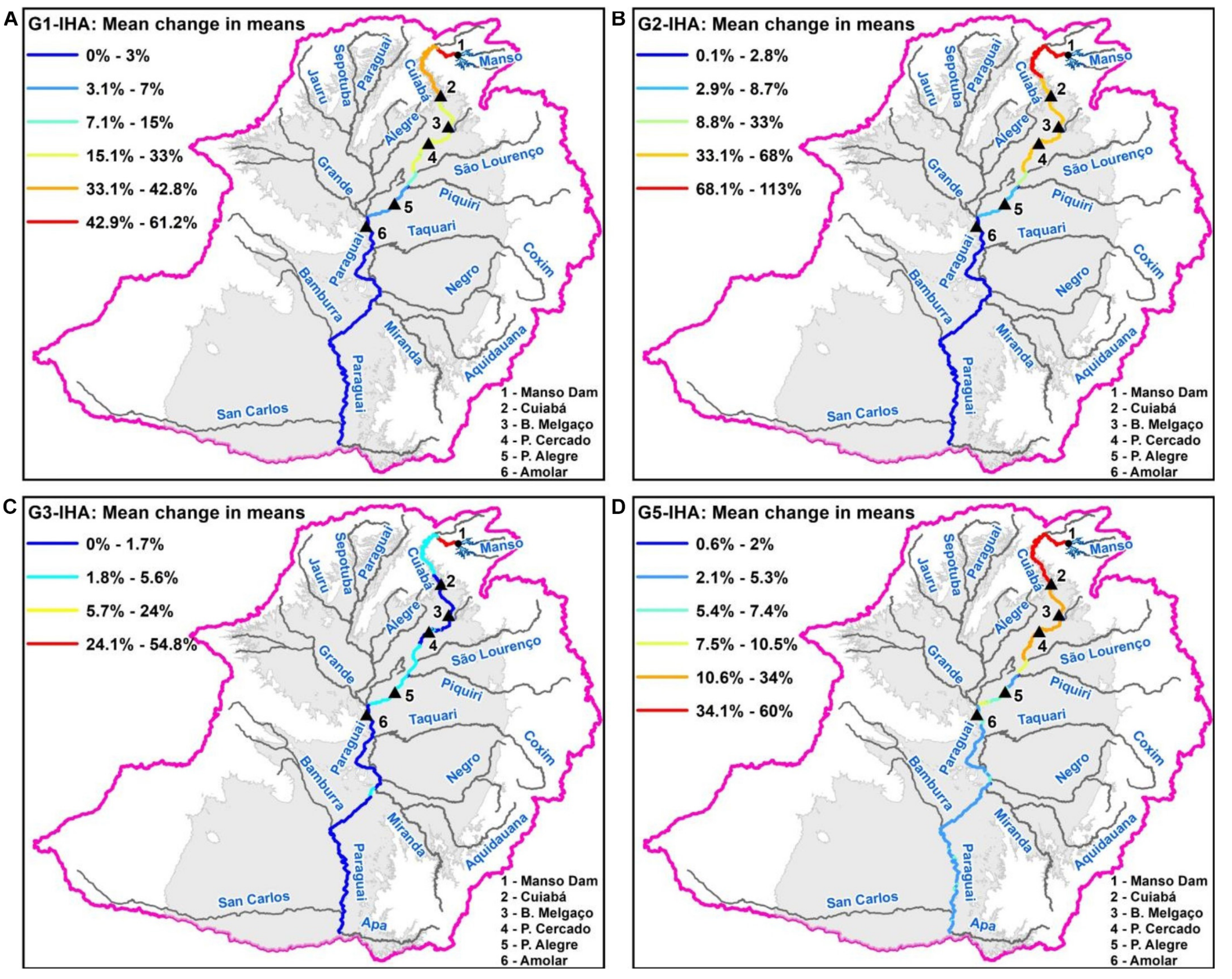

FIGURE 4 | Mean changes in means from Groups 1 **(A)**, 2 **(B)**, 3 **(C)**, and 5 **(D)** of IHA statistics due to Manso operation.

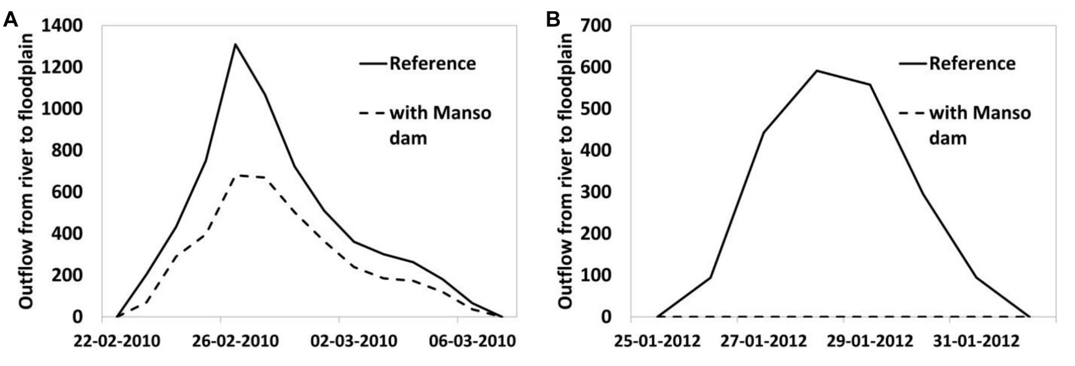

FIGURE 5 | Outflow hydrographs from river to floodplain along the reach of the Cuiabá river between Cuiabá (point 2) and Barão de Melgaço (point 3) in the reference (no dam) and altered (with dam) scenarios: **(A)** wet year (2010); **(B)** dry year (2012).

by the dam operation. Average peak outflows during the years 2003 to 2015 are 64% lower in the altered scenario than in the reference scenario. This result suggests that despite high flows along the main river within the Pantanal (downstream of Cuiabá) are relatively less altered by Manso dam operation, the river-floodplain interaction seems to be severely impacted, at least along the reach between Cuiabá and Barão de Melgaço. The intensity of river-floodplain interaction in the form of outflows from the river to the floodplain seems to be lessened along an important 140 km reach, and, therefore, river-floodplain connectivity, including the flux of sediments, nutrients, fish eggs and larvae may be weakened.

DISCUSSION

Our results show that, as expected, Manso dam has a regulation effect and decreases high flows and increases low flows. The decrease in high flows is somewhat limited to the region upstream of the Pantanal floodplain, while the increase in low flows extends well into the Pantanal. Despite the attenuation of the effects from Manso dam operation while discharge dislocates from the dam toward the Pantanal, noticeable changes can extend more than 500 km downstream of the dam. The Cuiabá river discharge within the Pantanal is more impacted by dam operation during low flows than during high flows, suggesting that the outflow from the Cuiabá to the Pantanal during floods may also be strongly affected by dam operation.

Timing of maximum and minimum flows is less affected by dam operation, except for the river reach immediately downstream of the dam. In comparison to the work by Souza et al. (2009) the present work indicates that changes in 1-day-minimum flow and 1-day-maximum flow go further than those authors indicated. Also, differently from Zeilhofer and de Moura (2009), who noticed an average flow decreases at Cuiabá gauge on the months of November (10%) and December (30%) in the period from 2002 to 2005 due to Manso operation, our findings, which englobes a different and wider period, indicate at the same location an increase of 9.7% of average flow in November and a decrease of 13.8% in December. Overall, our results are in accordance with the analysis provided by Timpe and Kaplan (2017), who analyzed the impact of Manso dam on the Manso river itself, at the Fazenda Raizama gauge. However, we found a much higher impact of Manso dam operation on minimum and maximum flows (112% in average) that those authors (40%), which may be related to the methodology and the time period used.

Regarding river-floodplain interaction in the region where the Cuiabá river flows into the Pantanal, it was strongly affected by Manso dam operation. By lowering maximum flows of the Cuiabá river, dam operation decreases the magnitude of overflows from the river to the floodplain, reducing the lateral connectivity of water. Similar observations were pointed out by Graf (2006) in the Marias River (United States) where floodplain areas were deactivated due to the operation of the Tiber Dam. To avoid this type of impact, an operating rule could be studied for the Manso dam that would mutually optimize urban flood reduction, power generation and maintaining connectivity between the river and the floodplain, especially in times where these connections are most important for fauna. At the same time, other measures to contain floods could be evaluated in the urban centers most affected by floods in the Cuiabá River in order to allow the powerplant to release more flow. These kinds of approaches have already start to be done in locations such the São Francisco river, in Brazil, where changes in river regime caused by flow regulation by dams since the 70s resulted in impacts on fishes (Pompeu and Godinho, 2006; Santos et al., 2012). Current initiatives are trying to minimize these impacts by changing operation rules of the main reservoir to give way to supplementary water releases during downstream floods (Godinho et al., 2007).

Our findings improve the assessment of spatial patterns of hydrologic alteration, giving more confidence in the assessment of magnitude and spatial extension of the effects of Manso dam in the Pantanal region. By using a model-based approach we avoided the confounding effects of climatic variability, extended the period of analysis, and improved the analysis of the spatial variability of hydrologic alteration over the river network, because the assessment of results was not restricted to river gauges. This approach and the models used could improve the assessment of hydrological changes due to dams operation in other locations, specially complex river systems with wetlands where there are planned dams or where impacts have already been noticed such as the ones caused by the Ponte de Pedra Dam (Fantin-Cruz et al., 2015), also located in the Pantanal, or the Balbina Dam (Fearnside, 1989; da Rocha et al., 2019), located in the Amazon Basin.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. This data can be found here: https://www.ana.gov.br/sar0/MedicaoSin, http://www.snirh.gov.br/hidroweb/, and https://drive.google.com/drive/folders/1Z-PmTJ9MYNlzkkfhQkYaMjJ8MQdb1nMQ?usp=sharing.

AUTHOR CONTRIBUTIONS

PJ: assembling and modeling the MGB model, results generation, statistics calculation, results interpretation, and generating figures and tables, writing. MM: assembling and modeling the SIRIPLAN model, results generation, and results interpretation. LR: generating IHA statistics, writing and interpretation of the results. AP: assembling and modeling the SIRIPLAN model, results generation, results interpretation. WC: assembling and modeling the MGB model, results generation and interpretation, and writing. All authors contributed to the article and approved the submitted version.

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Relationship of Freshwater Fish Recruitment With Distinct Reproductive Strategies and Flood Attributes: A Long-Term View in the Upper Paraná River Floodplain

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The flood pulse is the main driving force for communities' structure and functioning in river-floodplain systems. High synchrony exists between the hydrological cycle and reproductive cycle events for several fish species. However, species with different reproductive strategies can respond in different ways to the flood regime. Thus, this study intends to evaluate the relationship between the recruitment of different reproductive guilds of freshwater fish and flood attributes (flood duration, maximum annual water level, and delay of flood) from a time series of 20 years in the Upper Paraná River floodplain, Brazil. The abundance of four guilds was evaluated: (i) long-distance migratory with external fertilization and without parental care (LMEF); (ii) non-migratory or short-distance migratory with external fertilization and without parental care (NEFW); (iii) non-migratory or short-distance migratory with external fertilization and parental care (NEFP); and (iv) non-migratory or short-distance migratory with internal fertilization and without parental care (NIF). Multiple regression analyses were applied between flood attributes and abundance of young-of-the-year or juveniles for each reproductive guild. This study observed a consistent pattern of long-lasting flooding positively influencing the recruitment of all reproductive guilds, while water level intensity and the time of the onset of flooding also influenced some non-migratory strategies. We can conclude that the conservation of fish populations and the maintenance of ecosystem functions and services associated with them need to be considered in the operating protocols of upstream hydroelectric plants, since they are dependent on the flooding controlled by them.

Keywords: ichthyofauna, migratory fish, parental care, internal fertilization, flow control, dams

INTRODUCTION

Dams for hydropower generation have been considered among the most impactful anthropogenic activities for freshwater ecosystems, due to flow modification, invasive species facilitation, and habitat fragmentation (Agostinho et al., 2007; Abell et al., 2008; Timpe and Kaplan, 2017). In the Southern Hemisphere, where the most biodiverse river basins are located (e.g., Amazon, Congo, Mekong), there are an unprecedented number of dam projects (Winemiller et al., 2016). Downstream, the impacts mainly include the alteration of seasonal flood cycles. These effects are exacerbated in stretches of floodplain where dams are cascaded (Agostinho et al., 2008). For fish, these impacts can be on the structure and functions of the assemblages, with changes in the availability of shelter and food, reproductive processes, and rates of growth, mortality, competition, predation, and parasitism (Agostinho et al., 2004). Dams can also alter ecosystem processes and services in which fish are involved, such as feeding supply for aquatic and terrestrial consumers, fisheries, nutrient cycling and transportation, food web regulation, and their roles as environmental engineers (Mormul et al., 2012; Humphries and Walker, 2013).

The hydrologic regime is the selective force behind the diverse livelihood strategies of species, including life-history traits (Wootton, 1990; Zeug and Winemiller, 2007; Abrial et al., 2019; Humphries et al., 2020). The reproductive dynamics and flood regime are closely related (Gomes and Agostinho, 1997; Humphries et al., 1999; Bailly et al., 2008). The occurrence of the natural and periodic flood pulses in tropical floodplains led the biota, especially fish, to develop strategies that maximize reproductive success (Welcomme, 1979; Vazzoler, 1996) and diminish the predation risk on their offspring (Suzuki et al., 2009). A gradient of morphological, physiological, and ecological attributes characterizes the diversity of methods by which fish of freshwater systems reproduce (Winemiller, 1989). Consequently, the flood regime differentially affects the reproduction and recruitment of species [i.e., the addition of new individuals to populations; Gaillard et al. (2008)] with different life histories (Agostinho et al., 2004; Winemiller, 2005; Bailly et al., 2008).

Inundations tend to favor the reproduction of long-distance migratory species because there is a high degree of synchronization between the hydrological regime variation and essential events in the reproductive cycle (Junk et al., 1989; Agostinho et al., 2004). Increases in photoperiod and temperature lead to gonadal development and maturation. The first rains act as a cue for the formation of schools and migrations upstream to search for the best environmental conditions for spawning (Vazzoler and Menezes, 1992; Cowx et al., 1998; Suzuki et al., 2004). The migratory fish usually spawn in the uppermost regions of the basin and use flooded area, dozens of kilometers downstream, as nurseries for their early stages of development (Nakatani et al., 2001; Wantzen and Junk, 2006; Silva et al., 2017; Rosa et al., 2018). For these species, long-lasting floods in the warmer season can maximize recruitment (Gomes and Agostinho, 1997; Oliveira et al., 2015), providing shelter and food for more extended periods.

The influence of the hydrological regime on fish recruitment becomes relevant when we consider the threats imposed by the rapid expansion of impoundments in most basins in Brazil. The results obtained from studies on fish recruitment responses to the flood regime have been divergent, especially if considered fish that do not perform long reproductive migrations for spawning. In the upper Paraná River basin, the reproduction of sedentary species with parental care, internal fertilization, and short-distance migration seems to be less dependent on flooding. Still, the abundance of young-of-the-year (for example migratory species), can be low in years with no or incipient flooding (Agostinho et al., 2004). In the opposite, for the Cuiabá River, floods also appear to have a relevant role in reproduction and recruitment for species with parental care and internal fertilization. In contrast, reproduction of short-distance migrants (SM) appears to be less dependent on flooding (Bailly et al., 2008). Other studies considering adult species members besides the juveniles, report that short-distance migrants (SM) show moderated flood dependence (Fernandes et al., 2009; Vasconcelos et al., 2013).

Migratory fish are even more vulnerable to hydropower plants. In addition to controlling seasonal flooding, dam operation intercepts migratory routes (Agostinho et al., 2005; Pelicice et al., 2015; Lima et al., 2017). The impacts resulting from changes in the hydrological regime and barrier interposed by barrages to fish movements are quite conspicuous in the upper Paraná River basin (Agostinho et al., 2008). The stretch concentrates the largest number of large reservoirs in South America, and the impacts from these reservoirs can be cumulative (Agostinho et al., 2007; dos Santos et al., 2017; Pelicice et al., 2018).

Data obtained from long-term ecological studies (LTES) of the last remaining stretch of a natural river-floodplain system in the upper Paraná River, downstream from a cascade of reservoirs, can facilitate a better understanding of the relationship between the flood and reproductive aspects of fish that are mediated by dam regulation. These studies provide data on a wider variety of hydrological cycles and environmental conditions, allowing a more detailed analysis of how fish assemblages respond to these conditions. Therefore, the data allows researchers to identify a more consistent pattern of recruitment responses over time, including crucial information about how different reproductive guilds can be affected by hydrological variations. In addition, LTES may support the operational management of hydro plants, providing information about the quantity of the water that needs to pass through the dams while maintaining adequate hydrological regimes to ensure acceptable levels of fish reproduction and recruitment.

Thus, this study aims to evaluate the relationship between recruitment of different reproductive guilds and flood attributes (flood duration, maximum annual water level, and flood delay) based on a time series of 20 years in the upper Paraná River floodplain, Brazil. We expect a stronger correlation between the flood regimes and the annual recruitment of long-distance migratory fish than of sedentary fish (i.e., non-migratory). Specifically, this research expects a positive relationship between the recruitment of large migratory fish and the maximum annual water level and duration of flood, and a negative relationship between the recruitment of large migratory fish and the delay in the onset of flooding.

MATERIALS AND METHODS

Study Area

The upper Paraná River floodplain is located on the west bank of the Paraná River (23°43′ – 25°33′S; 54°35′ −53°10′ W) between

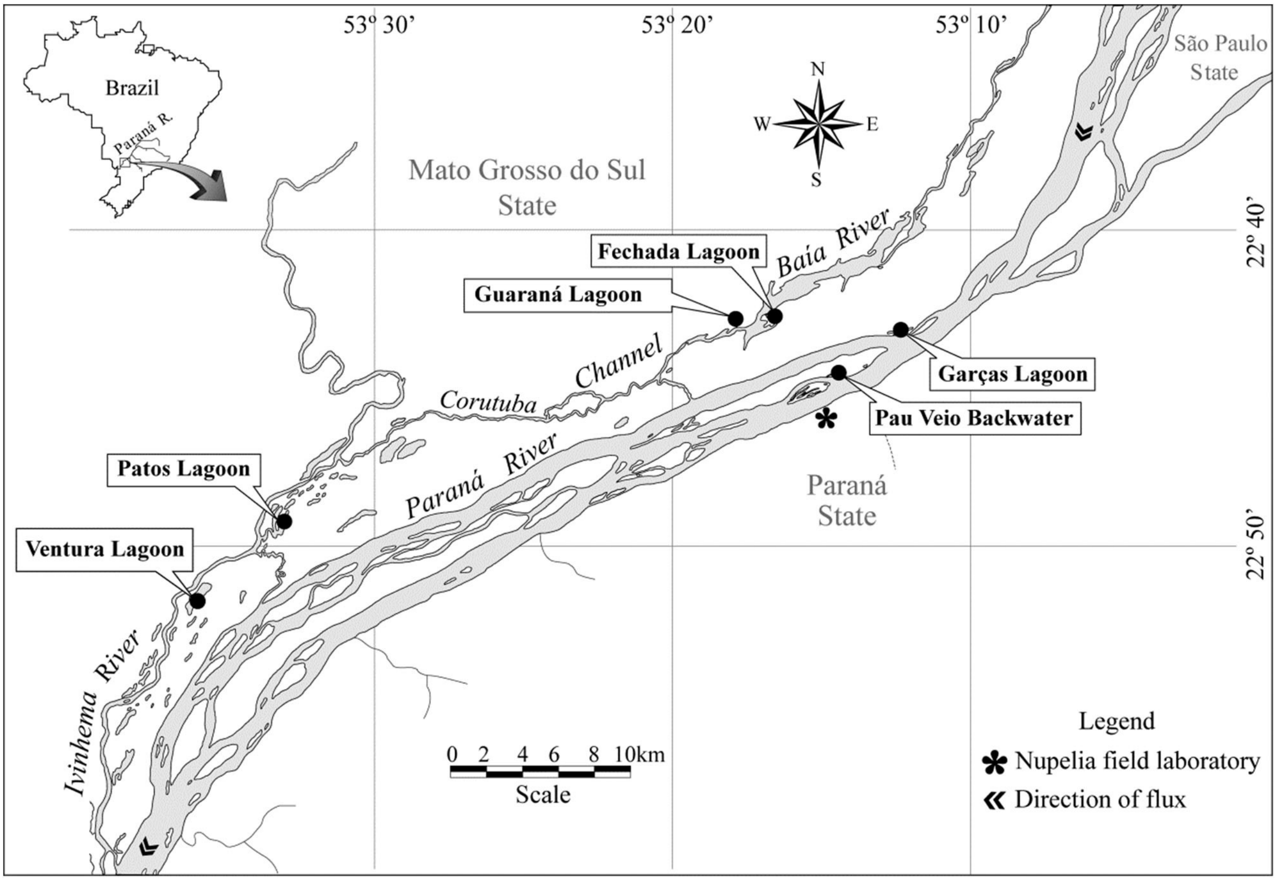

FIGURE 1 | Upper Paraná River floodplain, and location of sampling sites.

the states of Mato Grosso do Sul and Paraná. This region is subjected to variations in the level of the Paraná River and the two tributaries of the west bank, the Baía, and Ivinhema Rivers (Thomaz et al., 2007). The stretch of 230 km with lotic characteristics is the remaining of a 480 km long floodplain, before the Engenheiro Sérgio Motta "Porto Primavera" dam, upstream. Downstream, currently, the floodplain extends on to the Itaipu Reservoir (Souza-Filho and Stevaux, 2004). Porto Primavera is the first of dozens of upstream hydroelectric dams, composing reservoir cascades in the main channel of the Paraná River, as well in the tributaries like Paranapanema River, Tietê River, Grande River, and Paranaíba River, besides the dams are widespread all over the upper Paraná basin (Agostinho et al., 2008).

Sampling

Sampling was conducted quarterly from March 2000 until December 2019 in three rivers and six lakes within the floodplain (**Figure 1**). Fishes were collected using gillnets of different mesh sizes (2.4, 3, 4, 5, 6, 7, 8, 10, 12, 14, and 16 cm between opposite knots) that were exposed for 24 h and checked every 8 h (at 8, 16, and 24 h). Captured fish were anesthetized with 5% benzocaine and euthanized. All captured fish were identified according to Graça and Pavanelli (2007) and Ota et al. (2018). The abundance of fish caught in gillnets was indexed by the catch per unit effort (CPUE; individuals/1,000 m^2 of gillnets exposed during 24 h) for each year. The data for Paraná and Baía rivers were used together, as the latter follows the same hydrological cycle as Paraná River.

The reproductive guilds of fish were divided according to the classification proposed by Suzuki et al. (2004) based on their migratory behaviors for spawning, type of fertilization, and parental care: (i) long-distance migratory with external fertilization and without parental care (LMEF); (ii) non-migratory or short-distance migratory with external fertilization and without parental care (NEFW); (iii) non-migratory or short-distance migratory with external fertilization and parental care (NEFP); and (iv) non-migratory or short-distance migratory with internal fertilization and without parental care (NIF). Species that were not previously classified into the four referenced guilds were categorized based on the existing literature by considering their life histories or following the pattern of their genus according to Oliveira et al. (2018).

To infer about recruitment from each flood cycle, we used only individual young-of-the-year (YOY) for the long-distance migratory fish guild. The selection was made by species and individuals based on maximum length at an age

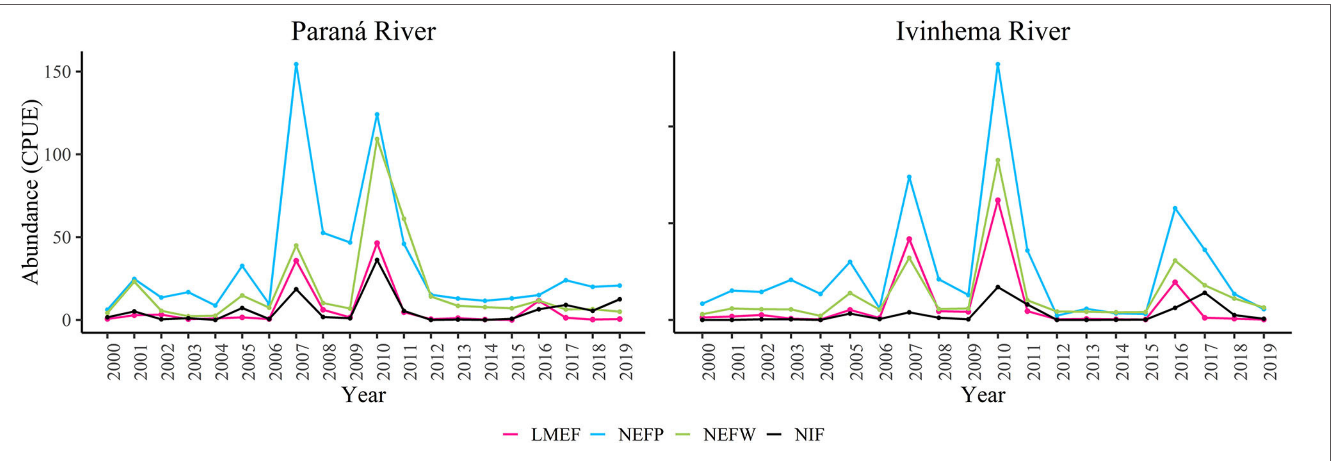

FIGURE 2 | Recruitment of each guild over time (CPUE = catch per unit of effort).

of 1 year, since individuals can remain immature for up to three 3 years. For other guilds that reach lower lengths, the selection of immature individuals was based on the degree of gonadal maturation.

River levels were provided by the National Water Agency (Agência Nacional das Águas—ANA—Sistema Nacional de Informações sobre Recursos Hídricos—SNIRH), which obtains the daily water level (WL; cm in relation to the operation of the Hydrometric Station at 231.8 meters above sea level) from a gauging station in the Paraná River (Porto São José Hydrometric Station; registration number 64575000). A threshold level of 450 cm had previously been established as the level at which the Paraná River overflows onto the floodplain (Comunello et al., 2003). This threshold level (450 cm) corresponds to a discharge of 12,370 m^3/s and a flooded area of 103.5 km^2 out of the 359 km^2 of the floodplain (Rocha et al., 2001). We used only the Paraná River water level, because the nurseries area and recruitment in the Ivinhema floodplain are strongly affected by high levels in the Paraná River.

The examined flood period lasted from October until May of each year, a period when floods and the spawning of long-distance migratory species have historically occurred in the region (Agostinho et al., 2005). Floods were characterized according to the following attributes: (i) flood duration (number of days when the river level remained above 450 cm in Paraná River); (ii) maximum annual water level (intensity; the highest recorded annual river level); and (iii) delay in flooding (the number of 15-day periods between October 1 and the start of flooding) (Suzuki et al., 2009; Oliveira et al., 2015).

Data Analysis

We used the abundance of YOY for LMEF, and juveniles for each other reproductive guild, as an estimate for fish recruitment. The relationships between the annual recruitment and flood attributes for each reproductive guild and river were assessed using a multiple linear regression analysis with model selection (forward-backward), with flood duration, maximum annual water level, and flood delay as predictors of the abundance (dependent variable). We tested the variables for multicollinearity using the variance inflation factor (VIF); we did not use values above 10. The residuals of the models were homoscedastics and the distribution was normal. We chose the best predictor model according to the lowest AIC value and adopted a significance level of 5%. These analyses were performed in the R environment (R Core Team, 2019) using the package car, function "vif;" package MASS, function "stepAIC;" and package stats, function "lm."

RESULTS

Long-term recruitment data for the Upper Paraná River floodplain were available for 103 species of fish (90 in the Paraná-Baía basin with 20 exclusive species and 83 in the Ivinhema basin with 13 exclusive species). The reproductive guild, the occurrence of each species in the basins, the abundance rank by the guild, the length of young-of-the-year of long-distance migratory fish, and the maximum size of juveniles for the other guilds, can be found in the **Table S1**. The recruitment of each guild showed a great difference between the years (**Figure 2**), with the highest values for all guilds in the years 2005, 2007, 2010, 2011, in the Paraná River, and also in 2016 after a local flood in the Ivinhema River, except for NIF, with a later peak in 2017.

Hydrological Cycle

In the Paraná River, during the period between 2000 and 2019, there were two lasting floods (>50 days; longest in 2009–2010 followed by 2006–2007; see **Table 1** and **Figure 3**). The longer-lasting flood was also the largest, with the water levels reaching 717 cm. Moderate floods (between 25 and 50 days) occurred in the cycles 2010–2011, 2004–2005, and 2015–2016, each of which reached high water level (>635 cm). In the other years, the floods were null or incipient. The majority of the floods were onset in January (with seven and eight delays), and just three cycles of flooding began earlier. Two floods (2005–2006 and 2015–2016) were not very long-lasting.

TABLE 1 | Flood attributes for the Paraná River in the studied period.

Flood attributes	Hydrological cycles of Paraná River																			
	99–00	**00–01**	**01–02**	**02–03**	**03–04**	**04–05**	**05–06**	**06–07**	**07–08**	**08–09**	**09–10**	**10–11**	**11–12**	**12–13**	**13–14**	**14–15**	**15–16**	**16–17**	**17–18**	**18–19**
Duration of flood	4	0	12	10	0	33	21	57	10	5	104	40	6	0	0	0	26	0	4	0
Intensity (cm)	507	414	532	504	434	676	516	645	498	506	717	677	498	408	408	341	637	402	473	348
Delay of flood	12	24	10	8	24	7	6	7	12	10	2	8	8	24	24	24	6	24	8	24

Relationship Between Fish Annual Recruitment and Flood Attributes

The reproductive guilds were related to different hydrological attributes, mainly duration and maximum annual river level (**Table 2**). For LMEF (**Figure 4**) the relationship between the annual recruitment and flood duration was positive and significant for both sub-basins (**Figures 4A,B**). Sedentary species with parental care (NEFP; **Figure 5**) also showed a positive and significant relationship between recruitment abundance and flood duration for both sub-basins (**Figures 5A,B**).

Sedentary fish without parental care (NEFW) showed a positive and significant relationship with flood duration in Paraná River (**Figure 6A**) and maximum annual water level and delay of the flood in the Ivinhema River (**Table 2**, **Figures 6B,C**). Those with internal fertilization (NIF) presented a positive and significant relationship with maximum annual water level and negative relationship with the delay of the flood in the Paraná River (**Figures 7A,B**), while for the Ivinhema River, the relationship was positive and significant only for the duration of the flood (**Figure 7C**). All generated models can be found in the **Tables S2**, **S3**.

DISCUSSION

This study evaluated data from a time series of 20 years and found a consistent pattern in which long-lasting floods positively influenced the recruitment of all reproductive guilds, the maximum annual water levels positively influenced some non-migratory guilds, and the delay of flood negatively influenced some non-migratory guilds.

The relationship between flood and migratory fish is well-known. Several studies have described the effects that floods can have on this guild (Agostinho et al., 2004; Bailly et al., 2008; Suzuki et al., 2009; Oliveira et al., 2015). Here, based on a long data series, we confirm this pattern of dependence on fish recruitment in relation to floods, which is null in drought years. However, Mallen-Cooper and Stuart (2003), studying fish in semi-arid and temperate regions in Australia, found two potamodromous species recruiting in years without a flood. The authors, even accepting the flood-pulse concept, discuss a possible plasticity in this reproductive strategy in response to the impacts of hundreds of dams, which regulate water flow.

Unlike migratory fish, the recruitment responses to flood attributes by species of other reproductive strategies are less studied, especially in long-term studies (Agostinho et al., 2004, 2007). In the upper Paraná River, previous studies have shown that flooding could have less effect on the recruitment of short-distance migratory assemblages (Fernandes et al., 2009; Vasconcelos et al., 2013, 2014). However, our results revealed that, at least partially, the recruitment of non-migratory fish responds to flood duration and/or maximum annual water level as well as the delay of flooding. These results corroborate with Agostinho et al. (2004), who verified that the abundance of juveniles for species of all reproductive strategies was low in the floodless year, which was attributed

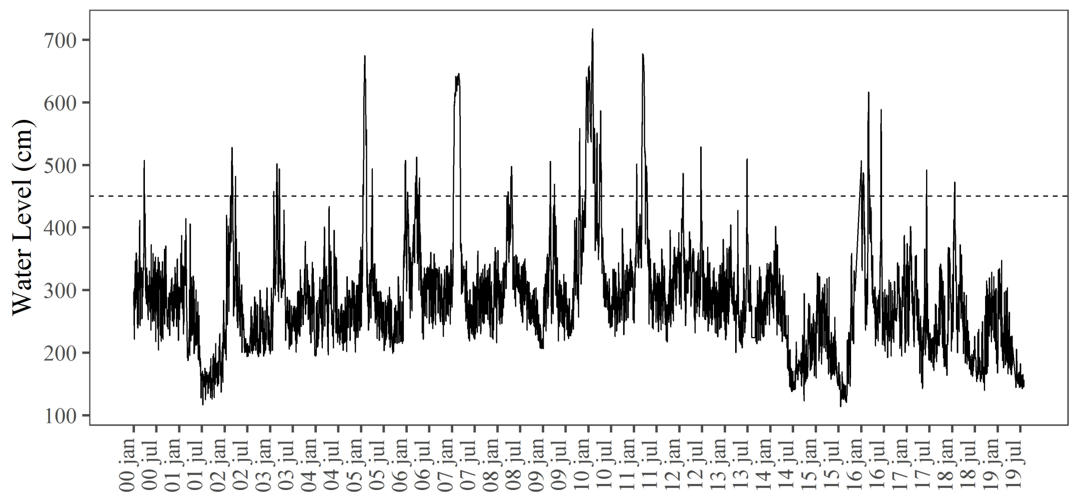

FIGURE 3 | Daily mean river levels in the Paraná River between 2000 and 2019. The dashed line represents the level (450 cm) at which the river overflowed onto the floodplain.

TABLE 2 | Result of the multiple linear regression analysis using the flood attributes as a predictor of the abundance of different reproductive guilds.

	Reproductive guild	X1	X2	X3	Adjusted R^2	F-statistic, (degree of freedom)	Model
Paraná	LMEF	***			0.46	18.06 (1,19)***	Y = 9.008X1
	NEFW	***			0.55	25.96 (1,19)***	Y = 22.55X1
	NEFP	***			0.62	34.14 (1,19)***	Y = 38.62X1
	NIF		**	*	0.43	8.76 (2,18)***	Y = 7.88X2–14.42X3
Ivinhema	LMEF	***			0.47	19.29 (1,19)***	Y = 11.75X1
	NEFW		***	**	0.69	24.22 (2,18)***	Y = 23.88X2–40.85X3
	NEFP	***			0.70	47.78 (1,19)***	Y = 46.13X1
	NIF	***			0.48	20.2 (1,19)***	Y = 3.82X1

*LMEF, Long-distance migratory with external fertilization; NEFW, Non-migratory with external fertilization, without parental care; NEFP, Non-migratory with external fertilization, with parental care; NIF, Non-migratory with internal fertilization; X1, Duration of the flood, X2, Maximum annual water level, X3, Delay of the flood. Significant variables are identified with *(0 '***'0.001, '**'0.01, '*'0.05, ''1).*

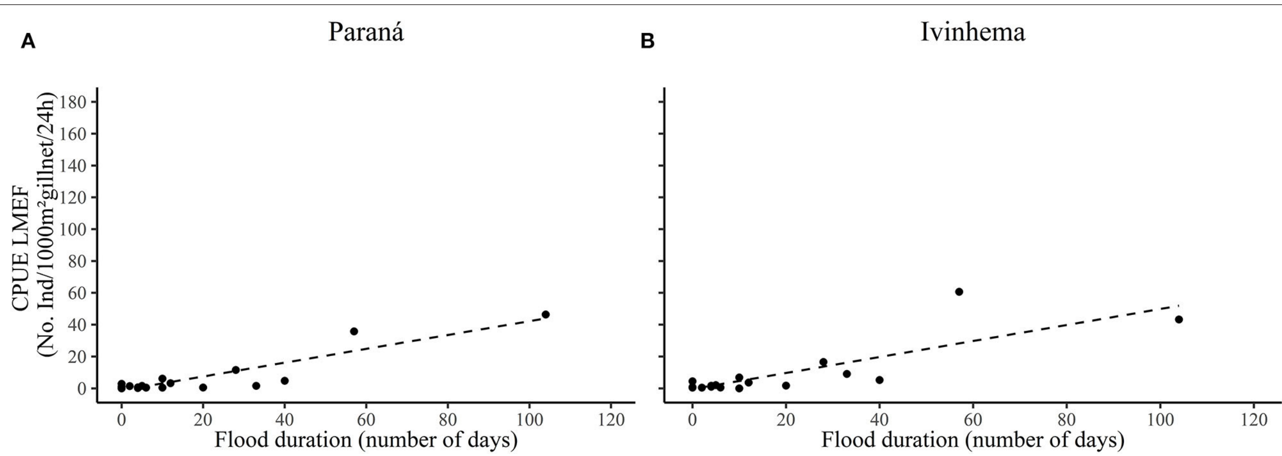

FIGURE 4 | Relationship between the long-distance migratory fish abundance (LMEF; CPUE = catch per unit of effort) and duration of flood in the Paraná River **(A)** and the Ivinhema River **(B)**.

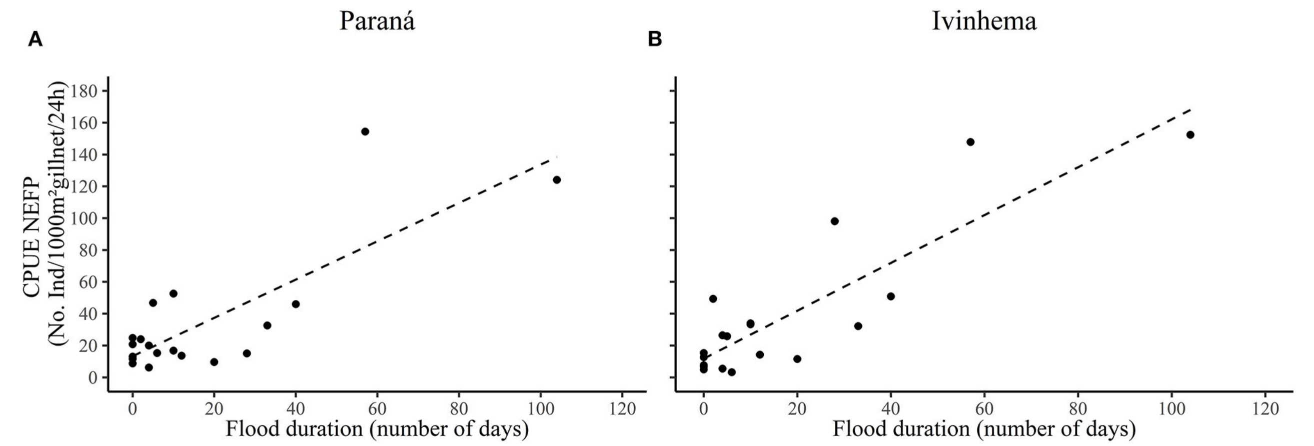

FIGURE 5 | Relationship between non-migratory fish with external fertilization and parental care (NEFP; CPUE = catch per unit effort) and duration of flood in Paraná River **(A)** and Ivinhema River **(B)**.

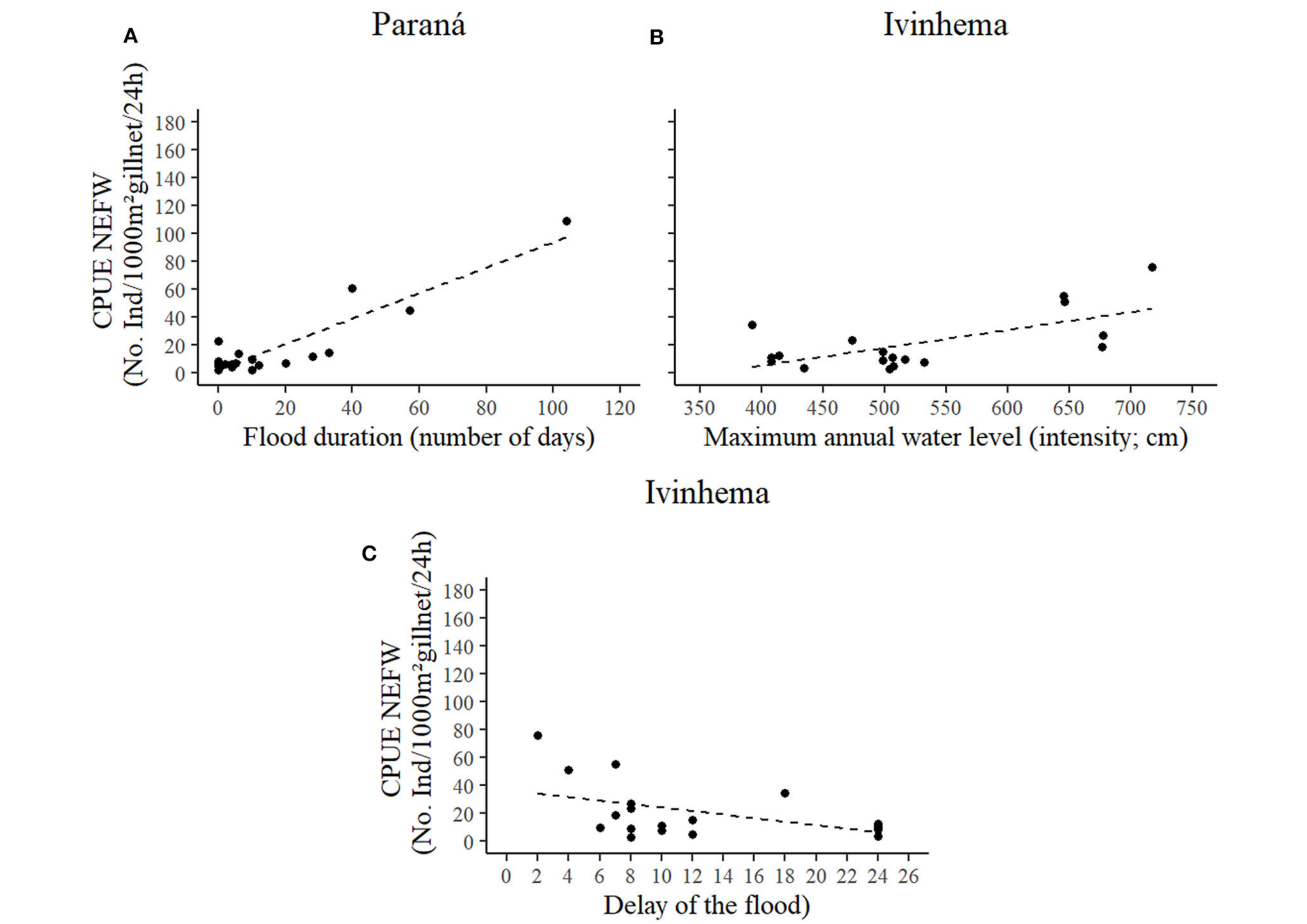

FIGURE 6 | Relationship between non-migratory fish with external fertilization and without parental care (NEFW; CPUE = catch per unit effort) and flood duration in the Paraná River **(A)**, and maximum annual water level **(B)**, and delay of flood **(C)** in the Ivinhema River.

to the pressure of predation, higher in more restricted aquatic environments, like those of the floodplain while the water recedes.

Long-lasting floods, those longer than 50 days, favor the survival of juveniles by providing higher availability of shelter and food for more extended periods before the waterline recedes

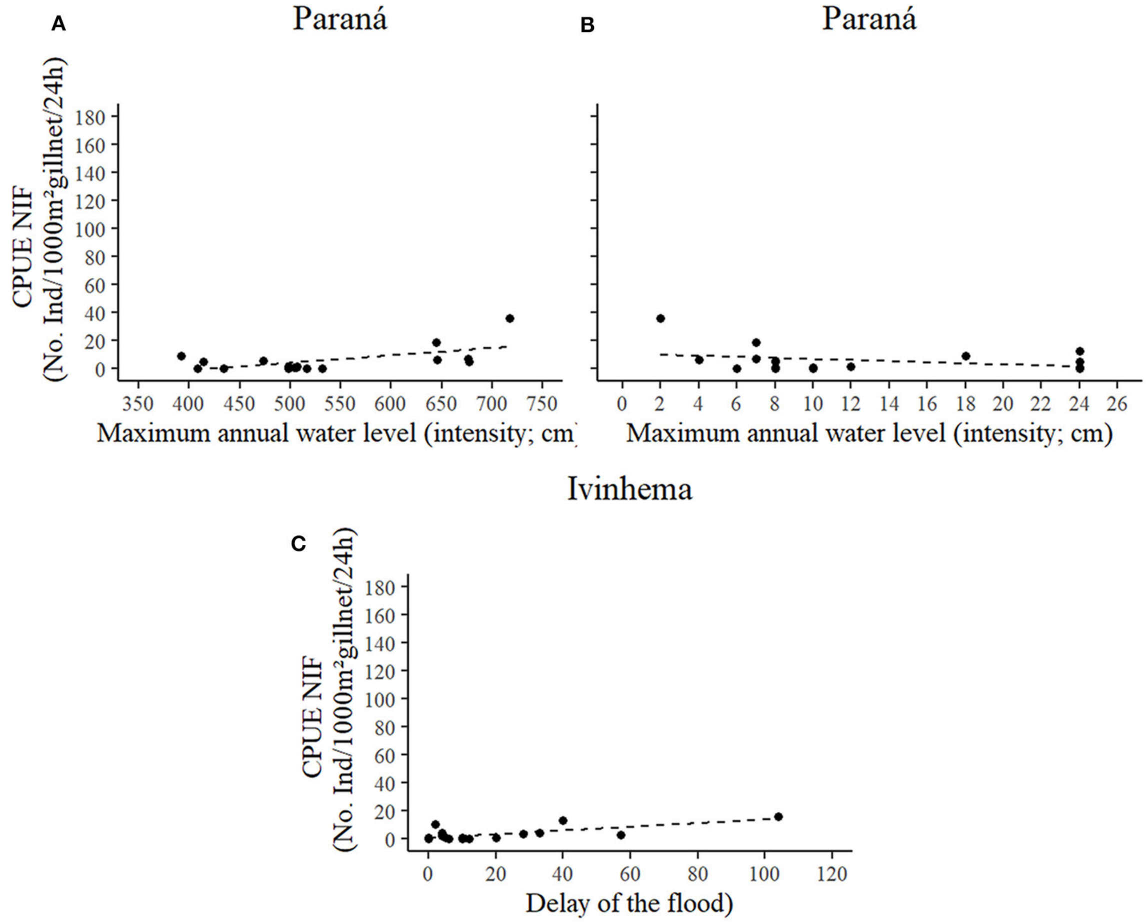

FIGURE 7 | Relationship between non-migratory fish with internal fertilization (NIF; CPUE = catch per unit effort) and maximum annual water level **(A)** and delay of the flood **(B)** in the Paraná River and duration of flood in the Ivinhema River **(C)**.

(Suzuki et al., 2009; Oliveira et al., 2015; Humphries et al., 2020). With lasting floods, the individuals leaving the flooded areas in the ebb period are larger in body size and consequently less susceptible to predation (Agostinho et al., 2004). Thus, this direct relationship between flood duration and resource availability influences a large number of species classified into the four reproductive guilds (LMEF, NEFW, NEFP, and NFI). For long-distance migratory fishes, recruitment can be null or incipient in years when flood is absent, short in duration, or delayed (Agostinho et al., 2004; Suzuki et al., 2009; Oliveira et al., 2015). In this study, migratory fish recruitment was significantly related to flood duration in both dammed (Paraná River) and undammed (Ivinhema River) sub-basins. In fact, recruitment success among these fish is the result of previous processes linked to migration, spawning, egg drift, and initial development, all of them in some way affected by one or more attributes of the flood regime (Vazzoler, 1996; Agostinho et al., 2003, 2007).

Previous studies state that the abundance of non-migratory species, whatever the internal fertilization strategy or attention to offspring, is more independent of the flood regime than the abundance of large, migratory fish. Non-migratory species can take advantage of the environment in other ways when flood is not relevant (Agostinho et al., 2004). Increasing density due to drought shrinks the wet surface, which affects the catchability and can influence the results (Agostinho et al., 2004). When evaluating the recruitment of these groups in a long time-series, the flood response pattern is consistent. Bailly et al. (2008), also using the young-of-the-year as an indicator of the reproductive success of each guild in the Cuiabá River floodplain, found no relationship of non-migratory guild NEFW with any hydrological attribute, despite the abundance of LMEF, NEFP, and NIF related to the flood duration and river levels. As mentioned previously, floods increase the resource inflow in the river-plain system and sustain individuals in the early stages of life for all reproductive guilds (King et al., 2003; Górski et al., 2011). The dilutive effects of the flooding increase the inundated area, relaxing predators' pressure on juveniles. Also, the flood duration protects eggs from exposure to air and desiccation, especially for species that adhere their eggs to substrates or plants in shallow areas or deposit them in nests built close or into the bank, reducing mortality and increasing reproductive success (Agostinho et al., 2007).

Although the absence of floods does not affect the reproduction process of fishes without parental care (NEFW) and with internal fertilization (NIF; Agostinho et al., 2004; Bailly et al., 2008), our results indicate that these species could also be favored by lasting flood conditions and moderate values of maximum annual water level. This trend is especially evident in the Paraná River, where the water is transparent, due to the sediment retention in dozens of upstream dams. Thus, long-lasting floods can benefit NEFW since the recruitment of these species are more dependent of shelter availability (e.g., flooded vegetation and higher turbidity) for a longer time. Even though internal fertilization reduces the time of exposure of gametes and eggs to predation, the higher availability of shelter and food provided by seasonal and high water level floods benefits the survival of larvae and juveniles because of protection for the parent fish. High water levels represent overflow of a larger area, with intermediate elevations also flooded (Souza Filho, 2009) and a large input of different autochthonous resources, benefiting the nutritional condition of the species (Gomes and Agostinho, 1997; Abujanra et al., 2009). On the other hand, the Ivinhema River has more pristine conditions and high turbidity (Roberto et al., 2009), which provide shelter against predation and increased survival, independent from the floods. As the relationship with high levels was positive and significant for NEFW, the flooding of new environments can benefit this guild. Lower water levels are associated with high transparency and less availability of shelter, which increases the predation rate for distinct guilds and reduces juvenile survival (Agostinho et al., 2007). In the same way, NIF can take advantage of the better conditions provided for the long-lasting floods, because of the extended higher shelter availability, food, and the weakening of the interspecific relationships, like predation over the offspring.

The guilds NEFW and NIF showed a negative and significant relationship with delayed flooding. Despite no significant relationship with the annual abundance of juveniles for LMEF and NEFP guilds in this regard, the delay in onset of floods may be important for all reproductive guilds. The species can be affected by delayed flooding because they lose the time of synchronization between the gonadal development and the climatic and pluviometric conditions of the hydrological cycle. The more delayed the flood, the more likely it is to be shorter; therefore, the individuals leaving the shelters will be smaller in size, and the predation risk increases. Among the hydrological cycles evaluated, the flood delay was inversely proportional to the flood duration, since longer floods corresponded to flooding periods with shorter delays. The absence of flood delay associated with lasting floods allows fish get an advantage under suitable conditions of temperature and photoperiod to ensure recruitment success (Górski et al., 2011). When evaluated individually, flood delay can relate significantly to the recruitment of some migratory fish species [*Megaleporinus* spp., *Pseudopimelodus corruscans* and *Prochilodus lineatus*—Oliveira et al. (2015)]. Earlier migrations are a strategy for the fish to avoid expending a higher amount of energy due to a lower swimming capacity to reach the upper stretches of the basin where spawning occurs (Lucas and Baras, 2008). If these species do not migrate earlier, the eggs and larvae fail to reach the nurseries while the floodplain is flooded (Agostinho et al., 2008).

We emphasize that grouping several fish species in the same reproductive guilds is a useful procedure for ecosystems management. Although grouping solves the difficulty of examining several species simultaneously (Winemiller, 1989), each group can hold species with marked differences in life-history strategies and tactics, which can present a wide gradient within the guild. Indeed, this must be considered when proposing management actions since species can exhibit different responses to the same environmental conditions (Oliveira et al., 2015). While species respond in different ways to the floods, other assemblage components such as functional diversity, can exhibit immediate and short responses (e.g., functional richness) or delayed and long-lasting responses for those indices which include abundance and are dependent on the recruitment success (Baumgartner et al., 2018).

As these results demonstrate, long-term ecological studies are necessary and useful for identifying consistent patterns in fundamental aspects of species biology and thus could be applied in conservation measures. Data from this long-term study allowed us to identify the importance of water level, flood duration, and the time of the onset of flooding for the recruitment of species with different reproductive strategies, as well as the different responses of these species to a wide variety of environmental conditions in hydrological cycles with different attributes. Since the flood regime is affected by the operation of hydropower dams, the maintenance of the ecosystem functions and services provided by fish fauna implies that managers and policy-makers understand these relationships. It is necessary to balance the human demands of hydroelectricity with the long-term conservation of biodiversity and stock. An important step and scientific challenge for the future is to calculate the trade-offs between different demands for water, including environmental ones, which could lead to a reservoir operation optimization model.

DATA AVAILABILITY STATEMENT

Requests to access the datasets should be directed to harumi@nupelia.uem.br.

ETHICS STATEMENT

The animal study was reviewed and approved by Ethical conduct in the use of experimental animals of Universidade Estadual de Maringá (CEUA; Technical Advice n° 1420221018 (ID001974) 06/11/18).

AUTHOR CONTRIBUTIONS

AO have conceived the idea and performed the formal analysis. AO, TL, RD, and AA discussed the idea. AO, TL, and HS curated the data. AO, TL, MA-V, RD, HS, and IC organized the data. AO and AA did the project administration. AA supervised the manuscript. AO and TL wrote the original draft. All authors reviewed and edited the writing.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2020.577181/full#supplementary-material

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Gambusia holbrooki Survive Shear Stress, Pressurization and Avoid Blade Strike in a Simulated Pumped Hydroelectric Scheme

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Pumped hydroelectric energy storage (PHES) projects are being considered worldwide to achieve renewable energy targets and to stabilize baseload energy supply from intermittent renewable energy sources. Unlike conventional hydroelectric systems that only pass water downstream, a feature of PHES schemes is that they rely on bi-directional water flow. In some cases, this flow can be across different waterbodies or catchments, posing a risk of inadvertently expanding the range of aquatic biota such as fish. The risk of this happening depends on the likelihood of survival of individuals, which remains poorly understood for turbines that are pumping rather than generating. This study quantified the survival of a globally widespread and invasive poeciliid fish, Eastern gambusia (*Gambusia holbrooki*), when exposed to three hydraulic stresses characteristic of those experienced through a PHES during the pumping phase. A shear flume and hyperbaric chamber were used to expose fish to different strain rates and rapid and sustained pressurization, respectively. Blade strike models were also used to predict fish survival through a Francis dual turbine/pump. Simulated ranges were based on design and operational conditions provided for a PHES scheme proposed in south-eastern Australia. All gambusia tested survived high levels of shear stress (up to 1,853 s^{-1}), extremely high pressurization (up to 7,600 kPa gauge pressure) and the majority (>93%) were unlikely to be struck by a turbine blade. Given their tolerance to these extreme simulated stresses, we conclude that gambusia will likely survive passage through the simulated PHES scheme if they are entrained at the intake. Therefore, where a new PHES project poses the risk of inadvertently expanding the range of gambusia or similar poeciliid species, measures to minimize their spread or mitigate their ecosystem impacts should be considered.

Keywords: pumped hydropower, renewable energy, invasive species, Francis turbine, pressure, shear strain, mosquito fish (*Gambusia* spp.)

INTRODUCTION

Pumped hydroelectric energy storage (PHES) projects are expanding worldwide, driven by the rising global demand for electricity, political renewable energy targets, security of supply, and upgrades to existing water infrastructure (Yang, 2016). Reversible turbines (usually a Francis dual turbine) pump water to a higher elevation reservoir during periods of low electricity demand. When energy demand is high, water is then released back down to a lower elevation reservoir to turn the turbine (Deane et al., 2010). Often, PHES is deemed an economic and sustainable mechanism to provide a large-scale source of energy and firm capacity for other renewables, in particular wind and solar (Harby et al., 2013).

A key feature of PHES is that it requires large differences in geographical altitude between reservoirs for water movement to occur in both an upstream and downstream direction. Therefore, PHES schemes can facilitate a bi-directional connection, sometimes across different waterbodies and even across different catchments. Discussions relating to the biological implications of such water transfers have occurred for some time (Hauck and Edson, 1976). Concerns have been raised around inter-basin water transfers and the effects on aquatic biota including the loss of biogeographical integrity, loss of biota, alien species introductions, and water quality implications (Davies et al., 1992). Specifically with regards to fish, there is evidence of species transfers (Lampert, 1976), impacts on migration (van Esch, 2012) and injuries and mortality (Hauck and Edson, 1976; van Esch, 2012). If a PHES facilitates the unintentional transfer of alien species to new areas, there can be flow on effects such as the loss of biodiversity, predation, and alterations of food webs (Strayer, 2010; Gallardo et al., 2016). Whether these impacts are realized will depend on whether a species will be entrained and survives passage through a PHES scheme.

Fish that pass through hydropower schemes can be exposed to potentially lethal hydraulic mechanisms. These have been extensively studied for conventional hydropower turbines and include elevated fluid shear and turbulence, turbine blade strike, and rapid and extreme pressure variations (Cada, 2001; Cada et al., 2006; Fu et al., 2016; Boys et al., 2018). Fish survival following exposure to these mortality mechanisms varies among species and life stages and depends on the severity of hydraulic stress (Neitzel et al., 2004; Deng et al., 2005; Boys et al., 2016a), fish morphology and swimming ability (Coutant and Whitney, 2000; Deng et al., 2007), and their position in the water column (Silva et al., 2018). In general, while blade strike likelihood is lower for small fish (Deng et al., 2007), the opposite occurs for shear stress, with eggs and larvae being more vulnerable to injury than larger fish (Navarro et al., 2019). Additionally, injuries and mortality tend to proportionally increase with higher shear stress levels (Deng et al., 2010; Navarro et al., 2019).

Due to the complex flow patterns near turbine blades, not all fish passing through conventional hydroelectric turbines are exposed to lethal shear stress or blade strike (Deng et al., 2007). Shear and blade strike exposure depends on the route taken by the fish through the turbine system (Fu et al., 2016). In comparison, all fish passing conventional hydroelectric turbines are exposed to rapid decompression, and the magnitude depends largely on turbine design and operation (Brown et al., 2012a,b). This rapid decompression can result in predictable levels of injury and mortality of fish of all life stages; eggs, larvae, juveniles, and adults (Boys et al., 2016a,b; Pflugrath et al., 2018). It is likely that the insights gained from other studies concerning conventional turbine passage can be directly applied to PHES when turbines are in the generation phase (passing water downstream). It is, however, unlikely that this information is transferable to PHES schemes that are operating in the pumping phase (passing water uphill). The hydraulic conditions created by a turbine are very different during the generating phase, compared to when it is pumping. For instance, whilst a rapid and transient exposure to negative pressures is likely at a generating turbine, fish passing a pumping turbine will experience a rapid compression that will be sustained and slowly released as the water body travels toward the upper reservoir. Unlike the effects of rapid decompression on fish, there have been few studies investigating fish survival following exposure to sustained pressurization (see Belaud and Barthelemy, 1973; Lampert, 1976; Sebert and MacDonald, 1993). While mortality has been observed in fish species exposed to high pressures (Sebert and MacDonald, 1993), in some cases fish do survive pressure in excess of 5,000 kPa (Lampert, 1976). The available evidence suggests the effects of compression vary among species and life stage, the rate at which the compression occurs, the time held at pressure and water temperature. Currently, the effects of compression have not been examined across a broad suite of fish families, and compression rates within PHES schemes are considerably greater than what has been examined on any fish species in the available literature.

As more PHES projects are approved for installation globally, it is imperative to resolve whether fish are likely to survive their hydraulic conditions. This is even more critical when the pumping involves inter-basin transfers of water that could result in range expansions of both invasive and non-invasive fish. In this study, the survival of the poeciliid fish, Eastern gambusia (*Gambusia holbrooki;* Girard, 1859, referred to as gambusia herein), was evaluated using simulations of shear stress, extreme pressurization, and blade strike likely to be experienced when passing through a 2,000 MW PHES scheme being proposed for south-eastern Australia.

Gambusia are a globally distributed species and are considered invasive in many areas, including Australia. They are capable of establishing populations when transferred to new locations (García-Berthou et al., 2005), where they have the potential to cause environmental damage and threaten small-bodied native fish (Rowe et al., 2008; MacDonald et al., 2012; Carmona-Catot et al., 2013). To evaluate the survival likelihood of adult gambusia when passing a PHES scheme during the pumping phase, the following tests were simulated: (1) fish were exposed to a range of shear strains representative of those estimated to occur when passing through draft tubes and close to turbine blades to quantify how survival changed; (2) fish were exposed to extreme pressurization simulating a PHES pumping phase; and (3) the likelihood that a gambusia would be struck by a turbine blade when passing through a Francis turbine was predicted using

TABLE 1 | Summary of adult gambusia measurements used for shear, pressure, and blade strike assessments.

	Fish size: mean TL (mm) ± SD	Fish weight: mean weight (g) ± SD
Shear	23.6 ± 5.10, range: 12.0–47.0	0.12 ± 0.11, range 0.01–1.19
Pressure and blade strike*	30.5 ± 6.0, range: 20.0–47.0	0.37 ± 0.27, range 0.08–1.42

*Only fish length was used for blade strike models.

TABLE 2 | Summary of water quality in holding tanks throughout the study.

Parameter	Mean ± SE	Range
Temperature (°C)	9.76 ± 0.091	7.68–12.07
pH	8.04 ± 0.006	7.89–8.14
Conductivity (ms/cm^{-1})	0.33 ± 0.003	0.29–0.46
Turbidity (NTU)	0	0
Dissolved oxygen (mg L^{-1})	9.73 ± 0.071	7.46–11.33
Total dissolved gas saturation (%)	89.90 ± 0.77	68.40–110.10
Ammonia	0.5 ± 0.021	0.25–1
Nitrite	0.02 ± 0.005	0–0.25
Nitrate	0	0

blade strike models. Ranges tested for each stressor were based on design parameters and operational conditions provided by the entity that commissioned the PHES at the time of publishing, and assumed that fish entrainment would occur at the site.

MATERIALS AND METHODS

Study Species and Husbandry

Adult gambusia (see **Table 1** for size ranges) were collected from water storage dams at Charles Sturt University, Albury (New South Wales, Australia), between September and October 2018 with a 4 mm mesh seine net. Fish were transported a few 100 m in 100 L bins filled with dam water to Charles Sturt University Fish Laboratory (CSUFL) where experiments were conducted. On arrival, gambusia were placed in a 1,000 L quarantine tank and monitored for a week prior to release into additional 1,000 L holding tanks. To minimize stress-related diseases following transport, fish were given a prophylactic salt treatment (5 ppt for 1 week) and maintained at 2.5 ppt throughout the study. Chlorinated town water supply, treated with Safe™ (1 g/1,000 L) to remove chlorine, was used in the holding tanks and recirculated and filtered with a Polygeyser Bead Filter (AST Endurance Model 4000, Aquaculture Systems Technologies, New Orleans). Gambusia were maintained on a diet of *Artemia* spp. nauplii and held for 5–7 days prior to experimentation. Water quality was monitored daily (**Table 2**).

PHES Simulation

The tests conducted in the present study were based on specifications related to a PHES scheme under development in south-eastern Australia. The simulations were based on the assumption that fish were entrained into the PHES from the lower elevation storage reservoir and pumped to the upper, higher elevation storage reservoir while being exposed to a head differential of 7,600 kPa. During a passage event, a fish passes through a Francis reversible turbine (simulated in the blade strike models) and through a series of structures before ascending to the upper reservoir (**Figure 1**). The profile and total passage time simulated for the pressure tests reflected all six turbines operating to represent the most extreme scenario for a passing fish (**Table 3**; **Figure 2**). The shear strain rates tested (up to 1,853 s^{-1}) were designed to incorporate the range of levels known to be generated by PHES schemes, where rapidly flowing water passes near internal structures of the turbines, such as trash racks, draft tubes, wicket gates, and stay vanes.

Shear Experiments

Gambusia were exposed to one of five shear strain rates using a flume consisting of a cylindrical Plexiglas chamber connected to a submerged jet at one end and a fiberglass reservoir tank at the other (**Figures 3**, **4**; for a detailed description see Boys et al., 2014; Navarro et al., 2019). On entry to the chamber, a conical nozzle reduced the flow diameter from 15 to 5 cm over a distance of 26 cm (**Figure 4**). This effectively accelerated the flow to allow a shear environment to be generated and quantified at the position where water within the flume was entrained in the jet stream (based on Neitzel et al., 2000). Nozzle flow rates were manually adjusted to create different jet velocities which produced various strain rates. The maximum strain rate was achieved at the maximum velocity that could be generated in the shear flume. Velocity testing established in Navarro et al. (2019) found that the highest area of strain across all flow rates was between 20 and 30 mm from the jet centerline. For this reason, fish were introduced into the jet through a polycarbonate tube positioned 30 mm above the jet centerline and at an angle of ∼30° to prevent contact of fish with the tube on exit. A small flow of water was used in the delivery tube to transition the fish into the jet.

Strain rates were defined as the maximum strain rate that we can assume a fish was exposed to within the zone of flow establishment (as per Neitzel et al., 2004). Mean jet velocities for the flow rates applied (**Table 4**) were obtained using a total tube connected to a pressure gauge ranging from 0 to 45 psi (with 10 psi increments) positioned 90 mm from the nozzle in the center of the jet. Pressure gauge readings (psi) were converted to meters of head and jet velocities calculated using Bernoulli's equation

$$H = \frac{v^2}{2g} \tag{1}$$

where H is the total head (m), v is the velocity (m s^{-1}) and g is the gravitational constant (m^2 s^{-1}). Once mean jet velocities were obtained, shear strain rates were calculated using the equation suggested by Neitzel et al. (2004)

$$e = \frac{\partial v}{\partial y} \tag{2}$$

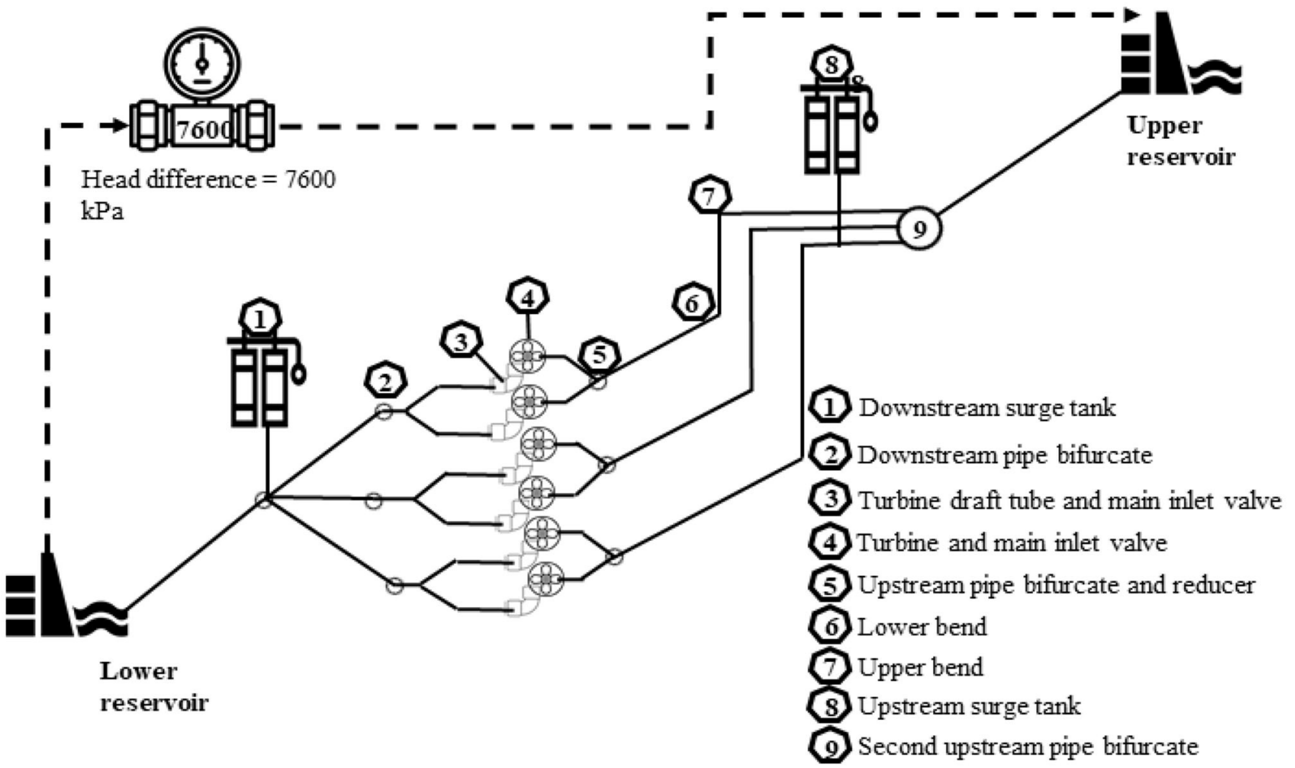

FIGURE 1 | Schematic of the pumped hydroelectric storage informing the head difference between lower reservoir and upper reservoir during the pumping phase and the main structures (numbers 1–9) where pressure changes will occur through the scheme. Note that the head difference was estimated for the scenario where the lower reservoir was operating at full supply level and upper reservoir was at the minimum operating level.

TABLE 3 | Expected pressure changes when pumping from the lower reservoir to the upper reservoir with all six turbines operating (T_6), ranging from 100 kPa to a maximum of 7,600 kPa.

Location in the PHES scheme	Full pumping mode (all turbines operating—maximum flow; T_6)		
	Gauge pressure (kPa)	Duration (s)	Elapsed time (s)
Intake (I)	200	1	1
Intake to surge tank (ST)	1,000	1,951	1,952
Surge tank to bifurcate (B1)	1,100	21	1,973
Bifurcate to draft tube (DT)	1,100	24	1,997
Draft tube to turbine (T)	7,600	0.019	1,997
Turbine to main inlet valve (MIV)	7,600	1	1,998
MIV to bifurcate (B2)	7,500	11	2,009
Bifurcate to reducer (R)	7,500	21	2,030
Reducer to lower bend (LB)	6,600	230	2,260
Lower bend to Upper bend (UB), UB to ST and to bifurcate 3 (B3)	1,100	194	2,454
Bifurcate to outlet (O)	100	4,678	7,132

where v is mean water velocity and y is the distance perpendicular to the force. To provide a fine scale measurement of the shear strain rate at the width of the fish (Neitzel et al., 2004), distance (y) was defined as 10 mm for adult gambusia.

Shear strain rates tested ranged from 505 to 1,853 s^{-1}, with five replicates for each strain rate treatment (**Table 4**). Each replicate consisted of a group of 10 individual gambusia (see **Table 1** for fish sizes) that were dip-netted from holding tanks and inserted into the delivery tube for exposure to the shear environment. A "0" shear strain was applied as a control to consider any potential handling effects, and involved delivering fish via a duplicated deployment tube that was directed into a fish retrieval net. Following shear exposure, gambusia were collected from the fish retrieval net for each test group and immediately assessed for survival. Each fish was then placed with others from its replicate test group in a 16 cm by 16 cm mesh basket floating within the holding tank and survival was reassessed 24 h later. Fish that swam and fed freely during the post-experimental monitoring period were deemed to have survived. After the experiment, all fish still alive were euthanized in 100 mg L^{-1} benzocaine and then measured [total length (TL) in mm] and weighed (to the nearest 0.01 of a gram).

Pressure Experiments

A purposely-built hyperbaric chamber (**Figure 5**) capable of generating extremely fast positive pressure transients (>200 kPa per ms) and able to achieve up to 9,000 kPa (with 0.1% accuracy in pressure regulation) was

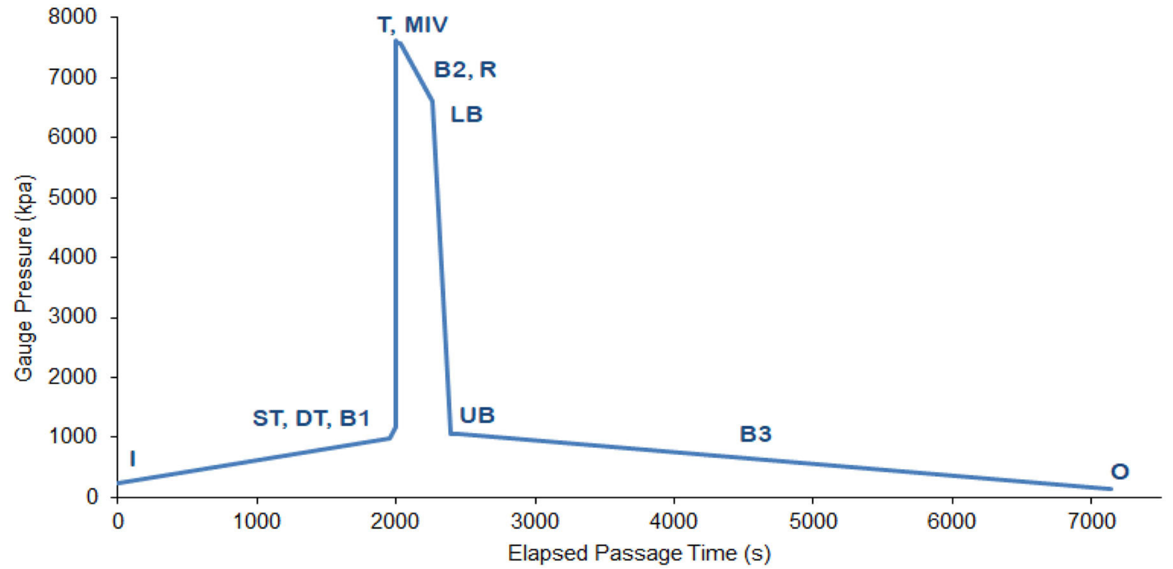

FIGURE 2 | Pressure profile simulated in the hyperbaric chamber at full pumping capacity (all six turbines operating). See **Table 3** for location labels and a full description of pressures and transient times.

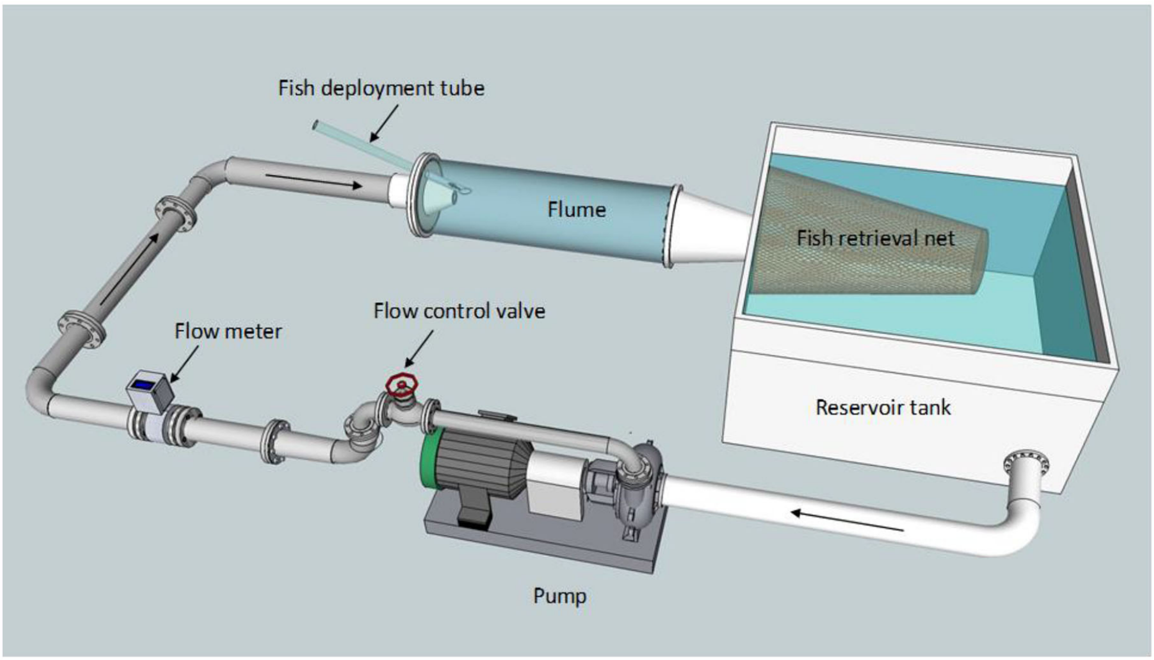

FIGURE 3 | Schematic of the shear flume that was used to expose gambusia to simulated strain rates. Fish are released in the fish deployment tube, exposed to a jet in the flume then collected in the retrieval net before being transferred to holding facilities. Adapted from Boys et al. (2014), used with author permission.

used to simulate the pressure profile experienced by a fish as it would travel through the PHES, from the intake to outlet (**Table 3** and **Figure 1**). The chamber was manufactured to meet the compression spike of the simulation while maintaining a stable environment (oxygen and temperature levels).

The chamber consisted of a 100 L pressure vessel. Fish were inserted in one end through a removable ASME B16.1 Class 600 15 cm flange that could then be sealed by bolting the flange to the chamber. Within this removable flange was a hydraulic ram for water displacement. At the other end of the chamber was a thick acrylic viewing port (**Figure 5**). The chamber was filled

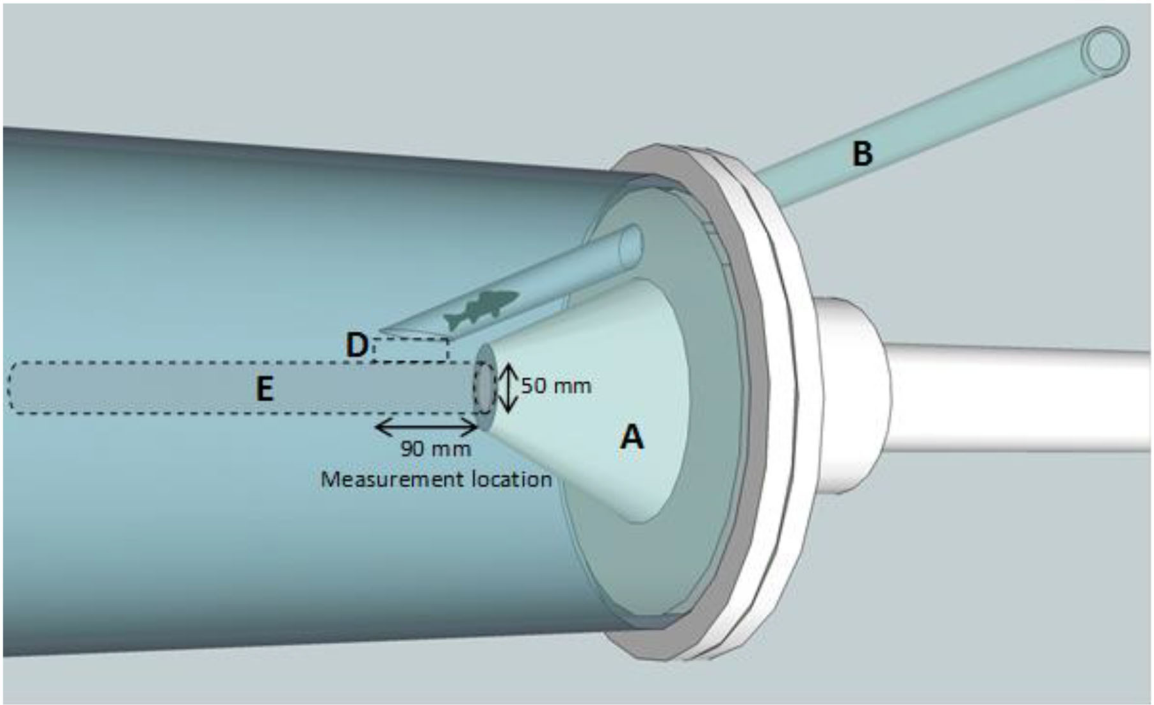

FIGURE 4 | Schematic of the conical nozzle of the shear flume used to reduce the diameter of the flow, effectively accelerating the flow to generate a shear environment. Adapted from Boys et al. (2014), used with author permission. **(A)** Conical nozzle. **(B)** Fish delivery tube. **(D)** Fish exposure to edge of jet and shear forces and **(E)** Flow-establishment zone.

TABLE 4 | Shear chamber flow rates, mean jet velocities and calculated shear strain rates with a summary of experimental treatments and fish size.

Chamber flow rate (L/s^{-1})	Mean jet velocity at nozzle (m s^{-1})	Shear strain (s^{-1})	No. replicates	Fish per replicate
0	0	0	5	10
12	5.05	505	5	
25	12.61	1,261	5	
34	16.87	1,687	5	
40	18.53	1,853	5	

with water using a submersible pump (Ozito Model: PSDW-750 Submersible Pump). Once a pressure profile was initiated, no replacement water entered the chamber. Therefore, the chamber included a bleed off valve where water could be periodically removed to test dissolved oxygen levels and a 10,000 kPa dosing pump to move the sampled water back into the chamber or to oxygenate water before being returned to the chamber. A 23,000 kPa 20 L hydrogen charged accumulator was used to store the large amount of energy required to move the hydraulic ram to rapidly create a desired pre-programmed pressure change in the system. Rapid pressure changes were implemented by a high-end embedded Moog Motion Controller (Moog Inc., USA, http://www.moog.com) moving a high bandwidth Moog hydraulic ram/valve combination to accurately and rapidly displace water in the test chamber. Measurement of all signals (e.g., dissolved oxygen, temperature, and pressure) was achieved using Beckhoff Digitization Hardware (Beckhoff Automation GmbH & Co. KG, Germany, https://www.beckhoff.com/). Thermo-couple temperature channels were calibrated using a Fluke 725 Multifunction Process Calibrator (Fluke Corporation, Washington, https://us.flukecal.com/) to simulate the sensor characteristics. The dissolved oxygen sensor (RDP-Pro X, *In-Situ*, Fort Collins, Colorado USA, https://in-situ.com/us/) was calibrated with the calibration tool and simulation solution provided by the supplier.

The pressure profile simulated in this study was that expected to be experienced by a fish moving from the intake of the lower reservoir, through the PHES scheme and to the upper reservoir (**Figure 1**). The scenario was based on full pumping capacity with all six turbines operating simultaneously at full speed resulting in a total passage time of 2 h (~7,133 seconds) (**Figure 2**). Under these conditions, the highest pressure achieved would be 7,600 kPa at the turbines and the lowest 100 kPa (**Table 3**). Pressure changes and travel times tested were characteristic of the lower reservoir being at full supply level (FSL) and the upper reservoir at minimum operating level (MOL) (**Table 3**). By testing this scenario, a conservative approach was taken to assessing the risk of gambusia survival, because the pressure profiles would be the most extreme. That is, if gambusia survived this scenario, they would also likely survive all other operating scenarios.

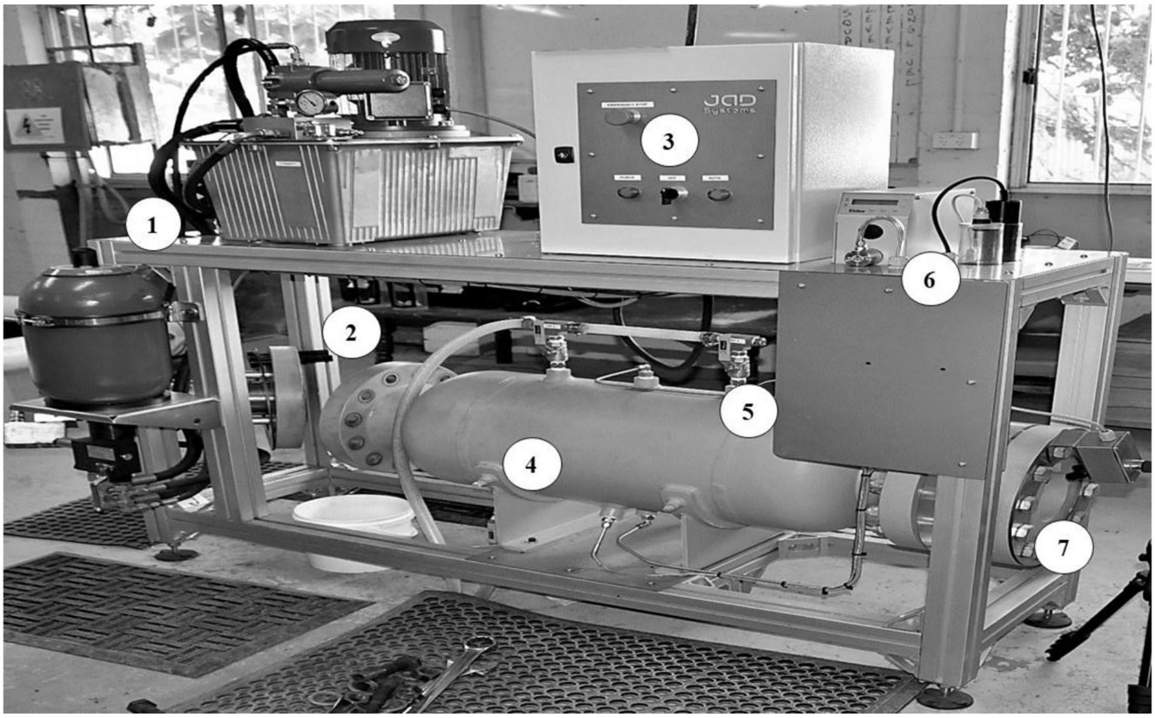

FIGURE 5 | Diagram of the pressure chamber and labeled components: (1) hydraulic pump and cylinder, (2) removable flange where fish capsules were inserted, (3) control panel, (4) pressure vessel, (5) pressure sensor, (6) oxygen and temperature sensor and (7) fixed viewing window.

Each pressure trial involved adding 10 gambusia (see **Table 1** for fish sizes) to each of the two cylindrical Perspex capsules (length 350 mm, diameter 100 mm) that contained water from the holding tanks. The 20 fish were sealed within the capsules using Velcro-attached mesh (1.5 mm^2 mesh) "lids" so that they could not escape but oxygenated water could exchange between the capsules and chamber. Both capsules were inserted into the partially filled chamber, ensuring that no air bubbles were trapped in or around each cylinder. The chamber was then sealed and filled with the same water the gambusia were housed in using a submersible pump (Ozito Model: PSDW-750 Submersible Pump). Once full, the chamber was purged to remove any air bubbles and a computer program with graphical user interface (GUI; LabView) was used to control the chamber hardware to run the pre-programmed pressure profile.

Five replicate test groups of 10 gambusia per capsule (20 per replicate) were exposed to the pressure scenario. Five control replicates were also performed that consisted of the identical handling protocol but without exposing fish to any pressure change. At the end of the experiment, the chamber was drained into an intermediary holding tank using an in-line pump (Ozito Model: TRP-650 Transfer Water Pump), the end flange unbolted and the Perspex cylinders removed. The computer program saved pressure, dissolved oxygen and temperature data for each trial, and these data were examined to confirm the pressure profile that fish were exposed to and to verify oxygen and temperature stability in the chamber throughout the experiment. Gambusia were collected from the capsules and immediately assessed for survival, then placed into a holding tank that contained baskets for each experimental replicate and held for 24 h to reassess survival as a percentage for each test group replicate. All fish still alive after 24 h were euthanized in 100 mg L^{-1} benzocaine and measured following the same procedure as for the shear experiments.

Statistical Analysis

Immediate and 24 h mean survival was pooled and presented as the probability of survival of adult gambusia after 24 h for the shear and pressure experiments. For the shear experiments, a binary logistic regression analysis (logit link) was used to test whether survival probability (after 24 h) was influenced by the shear strain rate with specimens classed as dead (0) or alive (1). Predicted survival rates and Wald confidence intervals were generated and the survival rates presented graphically. For the pressure experiments, the mean survival of gambusia was calculated as a percentage immediately after exposure and after 24 h for each experimental group (control and pressure). All analyses were performed using the Statistical Analysis Software (SAS) package (SAS Institute, Cary, NC, USA).

Blade Strike Model

Based on the assumption that fish must pass through a turbine runner leading edge plane after the sweep of one blade and before the sweep of the next one to avoid the strike, Von Raben (1957) proposed the first deterministic blade strike model. Turnpenney et al. (2000) later defined the "water length" as the distance

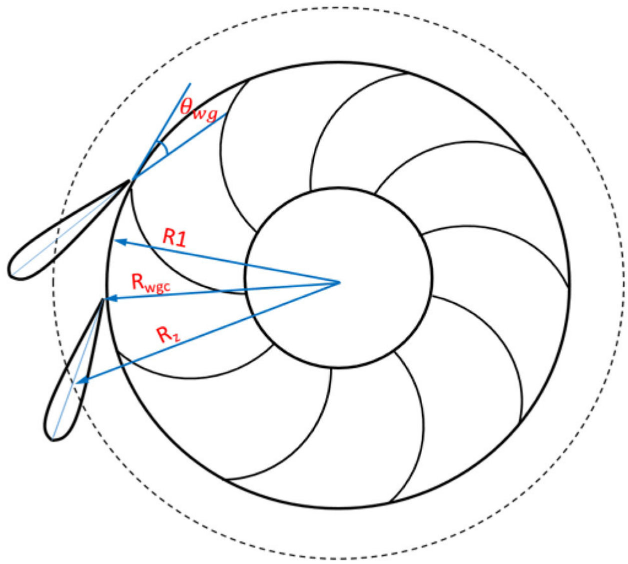

FIGURE 6 | Schematic showing the wicket gate angle and radius of the imaginary cylinder of the wicket gate exit.

between two successive blades along the flow line and derived the blade strike model for Kaplan turbines. Fish aligned with the flow lines and longer than the water length would be struck by the runner blades. Since fish can enter the turbine in random orientations rather than aligning with flow lines, which can shorten the apparent fish length, the deterministic model would maximize the blade strike risk.

Strike probability was given by:

$$P_{Blade\ Strike} = \frac{Fish\ Length}{Water\ Length} \tag{3}$$

Deng et al. (2007) defined "critical passage time" as the time between sweeps of two successive blades. Considering the time a fish needs to pass safely through the plane of the leading edges of the runner blades, Deng et al. (2007) introduced a stochastic blade strike model evaluating the validity of using blade strike modeling as an estimate of the biological performance of several large Kaplan turbines. The blade strike model used here for Francis turbines is derived following Deng et al. (2007), to approximate strike probabilities for the PHES. First, the surface of the imaginary cylinder of a Francis turbine wicket gate exit was calculated (**Figure 6**):

$$A_{wgc} = 2\pi R_{wgc} h_{wg} \tag{4}$$

where R_{wgc} is the radius of the imaginary cylinder of the wicket gate exit and h_{wg} is the wicket gate height. For the proposed PHES turbine, the turbine runner leading edge cylinder is adjacent to the imaginary cylinder of the wicket gate exit due to its high head (>600 m) and low specific speed, so

$$R_{wgc} = R_1 \tag{5}$$

where R_1 is the turbine runner leading edge radius. Radial velocity at the wicket exit is

$$V_r = \frac{Q}{A_{wgc}} = \frac{Q}{2\pi R_{wgc} h_{wg}} = \frac{Q}{2\pi R_1 h_{wg}} \tag{6}$$

Assuming that the angle of a fish at the wicket exit is the same as the wicket gate opening angle θ, then the time a fish needs to safely pass through the imaginary cylinder of the turbine runner leading edges is

$$t = \frac{l \cdot sin\theta}{V_r} \tag{7}$$

where l is the fish length.

The "critical passage time" t_{cr}, is the time between sweeps of two successive blades expressed as:

$$t_{cr} = \frac{1}{n \cdot (\frac{N}{60})} \tag{8}$$

where n is the number of runner blades and N is the runner speed in revolutions per minute (RPM).

A fish will be struck by a turbine blade if it does not pass through the imaginary cylinder of the turbine runner leading edges within the "critical passage time" t_{cr}, so the probability of blade strike can be expressed as:

$$P = \frac{t}{t_{cr}} = \frac{l \cdot sin\theta \cdot n \cdot (\frac{N}{60})}{V_r} \tag{9}$$

For the current simulation, because of the high rotation speed of the Francis turbine and lack of experimental data on blade strike injury for gambusia, the approach taken was to consider the worst case scenario and it was assumed that all fish struck by a blade will be mortally injured.

Deterministic Model

A deterministic model was applied to gambusia at three given turbine discharges: minimum, mid-point and maximum flow. The corresponding wicket gate opening angles were 13°, 20°, and 21°, respectively. A deterministic model lacks probability, that is, it predicts a single unique estimate for each combination of input values. Since fish orientation relative to the flow direction ranges from 0° to 90°, the arithmetic mean of 45° was used as the relative orientation of a fish to the flow direction, and the apparent length of a fish (see **Table 1** for fish sizes) was calculated as fish length∗cos (45°). In this deterministic model, blade strike predictions were a function of fish length and radial injection location, runner geometry, runner rotation rate, and axial flow.

Stochastic Model

Stochastic models use randomized data based on the specific distribution for each independent variable used as parameter inputs (Deng et al., 2007). The stochastic version of the model was implemented using @RISK software (the Palisade Corporation, Ithaca, New York). The software allows users to define distributions for any or all independent variables so

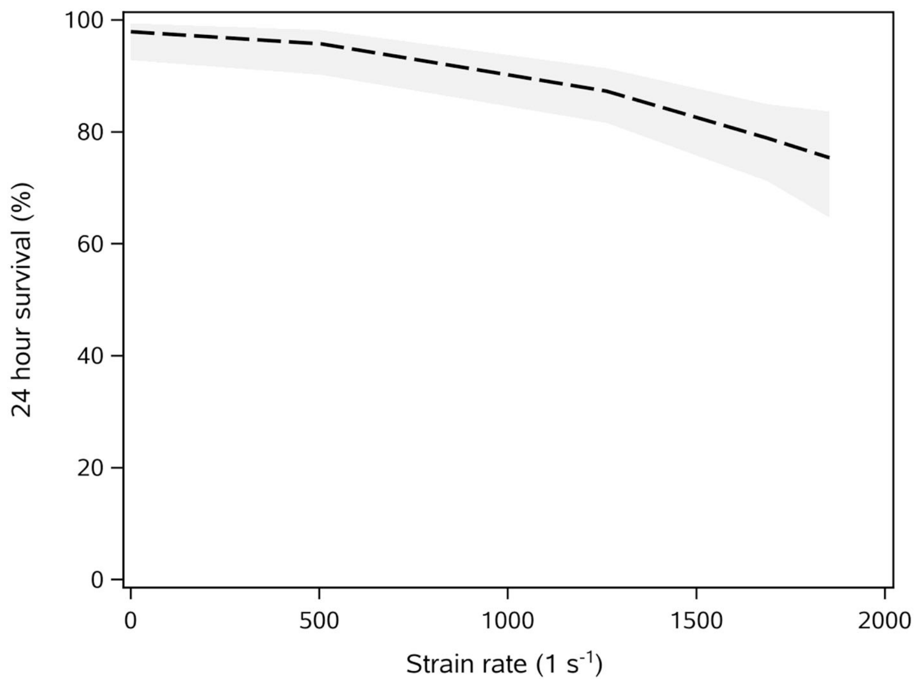

FIGURE 7 | Relationship between shear strain rate and the probability of survival of adult gambusia (after 24 h) determined from a binary logistic regression analysis (logit link). See **Table 4** for a summary of the experimental replicates for each shear strain tested. The line is predicted survival ±95% confidence interval (gray shading). Note the maximum strain included on the model was 1,853 (s^{-1}).

that the variation in variable values is propagated through calculations and is reflected in model predictions. The Monte Carlo simulation method and 10,000 realizations were used for all analyses. In this analysis, three variables were assigned distributions of possible values in simulations: wicket gate angle, fish size, and fish orientation relative to the flow direction. Wicket gate angle was assigned a uniform distribution ranging from the minimum flow angle to the maximum flow angle. The discharge was linear interpolated using the given flow information. Fish size was assigned a normal distribution. Fish orientation relative to flow direction was assigned a uniform distribution from 0° to 90°. Sensitivity and scenario analysis reports were performed using the @RISK software to identify the input distributions most critical to the predicted results. The higher the regression coefficient between the input and the output, the more significant the input is in determining the value of the output.

TABLE 5 | Predictions of the likelihood of a blade strike for gambusia and associated survival (%) through a Francis turbine using a deterministic model for a range of discharges [minimum, mid-point and maximum flow] and fish lengths (TL; minimum 20.0 mm, mean 30.5 mm, maximum 47.0 mm).

	Fish length	Blade strike (%)	Survival (%)
Min flow	Min	2.4	**97.6**
	Mean	3.7	96.3
	Max	5.7	94.3
Mid flow	Min	2.6	97.4
	Mean	4.0	96.0
	Max	6.1	**93.9**
Max flow	Min	2.4	97.6
	Mean	3.7	96.3
	Max	5.7	94.3

Numbers in bold font are the minimum and maximum probabilities predicted for gambusia survival.

RESULTS

Shear Experiments

There was a significant relationship between strain rate and survival of gambusia ($\chi^2 = 21.7$, $df = 1$, $p < 0.0001$). At strain rates of about 1,000 s^{-1}, the survival of gambusia decreased by 16% (Odds Ratio = 0.998, **Figure 7**). At the maximum strain rate tested (1,853 s^{-1}), there was still an estimated 80% survival.

Pressure Experiments

Results for adult gambusia exposed to the pressure change profile with travel time equivalent to all six turbines operating at full capacity showed a 100% survival rate after 24 h in both the control and pressure exposure test groups.

Blade Strike Modeling

Using a deterministic model that included turbine discharge and fish length, the probability of blade strike for gambusia ranged between 2.4 and 6.1%, with overall survival estimates ranging from 93.9 to 97.6% (**Table 5**). Smaller gambusia (20 mm) had similar blade strike probability under all modeled flow scenarios (2.4–2.6%). Larger gambusia (47 mm) had the highest probability of being struck by a blade with 5.7% at both minimum and maximum flow, and 6.1% at mid-flow. The blade strike probability from the stochastic model ranged from 0.2 to 8.3%, with the mean value at 2.9%. The sensitivity analysis for the stochastic model showed that the standardized regression coefficient (r) for fish size was 1.00, indicating that fish size contributed more significantly to the probability of blade strike than wicket gate angle ($r = 0.02$).

DISCUSSION

Most adult gambusia survived elevated fluid shear, extremely high pressurization, and most were predicted unlikely to be struck by a turbine blade. Therefore, there is a high possibility that the majority of adult gambusia will survive passage through the 2,000 MW PHES scheme during the pumping phase if entrained in the turbines.

Shear Stress

The majority of gambusia survived exposure to all shear strain treatments tested, even at the maximum rates tested. Elevated levels of shear are known to be harmful to other fish species (Neitzel et al., 2000) and mortality thresholds have been established under similar laboratory conditions for several species in the USA, such as salmonids [e.g., juvenile rainbow trout and steelhead (*Oncorhynchus mykiss*), juvenile Spring and Fall Chinook salmon (*O. tshawytscha*)], and American shad (*Alosa sapidissima*) (Johnson, 1970; Neitzel et al., 2004); and for Mekong species [e.g., blue gourami (*Trichopodus trichopterus*) and iridescent shark (*Pangasianodon hypophthalmus*) Colotelo et al., 2018]. Mortality thresholds for these species generally ranged between 800 and 1,200 s^{-1}, with blue gourami being the least tolerant (852 s^{-1}) (Colotelo et al., 2018) and steelhead being the most tolerant [i.e., having no reported mortality at the highest strain tested ($\geq$1,008 s^{-1})] (Neitzel et al., 2004). In comparison to these species, our results indicate that gambusia are a relatively tolerant species to shear strain, owing perhaps to their small size and body morphology (Neitzel et al., 2000). Our results from the shear experiments and other similar laboratory studies (e.g., Neitzel et al., 2004; Deng et al., 2005, 2010; Colotelo et al., 2018) have broadened our understanding of the effects of shear forces on a variety of fishes and indicate that there are species-specific responses, which generally relate to fish body morphology sensitivities. What is consistent from these other studies and the present study is, however, the probability of survival decreases with the severity of shear forces.

Naturally occurring fluid shear in freshwater ecosystems is generally below 200 s^{-1}, and aquatic organisms have developed adaptations to these levels (Vogel, 1994; Neitzel et al., 2000). PHES schemes create situations of extremely elevated and potentially lethal levels of shear that can occur in areas such as in a boundary layer near turbine structures including the stay vane, wicket gate, and runner blade leading edge (Neitzel et al., 2004). However, a large proportion of the total number of fish passing through the PHES may not be exposed to lethal or injurious strain rates (Neitzel et al., 2004; Cada et al., 2006). Computational fluid dynamics (CFD) models can be applied to fully understand the shear environment that a fish will encounter during passage and to quantify the proportion of fish survival through the PHES scheme. While our laboratory study looks at a simplistic single exposure, in reality, fish may be exposed to multiple events that could be more likely to kill them. Thus, our simulation may overestimate gambusia survival following shear exposure in the PHES scheme. However, computational modeling for conventional hydroturbines indicates that levels exceeding 1,000 s^{-1} occur in <10% of the overall draft tube area (McEwan and Scobie, 1992). Therefore, the maximum shear values tested in the present study were "extreme" and many fish may not be exposed to levels this high, depending on the path they take through the turbine.

Pressurization

Most adult gambusia survived the extreme and rapid pressurization to be expected during passage through the pumping phase of the PHES scheme, and at levels not tested on fish before. This was necessary to simulate the conditions that are likely to be experienced in PHES schemes. The results were similar to those observed by others previously for lower pressure exposures. Redfin perch (*Perca fluviatilis*), grayling (*Thymallus thymallus*), carp (*Cyprinus carpio*), roach (*Leuciscus rutilus*), whitefish (*Coregoma* sp.), and rainbow trout all had high survival to a rapid increase in pressure up to ∼5,070 kPa (Lampert, 1976). The direct effects of high pressures on fish have been observed to cause immobilization during pressurization, with recovery immediately after pressure is released (Rowley, 1955). Such immobilization was observed in the gambusia tested, along with erratic swimming. Similar to previous studies, normal swimming behavior resumed shortly after exposure. Thus, rapid compression is unlikely to contribute to high mortality in gambusia.

Blade Strike

Considering the operating scenario tested in the present study, and based on deterministic modeling of blade strike effects, the probability of a fish being struck by a blade was low and the associated survival rate of fish was expected to be high (>93%). It is also believed that this may be an underestimate of the probability of survival since the model predictions were based on the conservative assumption that all blade strikes would lead to mortality (i.e., the empirical factor "mutilation ratio" was set at 100%) (Von Raben, 1957).

Other studies have similarly estimated blade strikes from models, but generally under different turbine geometries. Nonetheless, with careful comparisons, they do identify a similar trend of low mortality (high survival). For example, the findings from the present study are consistent with strike mortality estimated from a Francis turbine at the Stornorrfors Dam on the Ume River (Sweden), ranging from 5.3 to 9.7% for juvenile salmon and trout but less than that for adult salmon and trout (ranging between 25.2 and 45.3%) (Ferguson et al., 2008). Similarly the mortality from blade strike at a large Kaplan turbine at Bonneville Dam on the Columbia River (USA) ranged from 3.4 to 4.3% for juvenile salmon (Deng et al., 2007).

The present study is unique in that it has modeled the survival of one of the smallest fish to date. Our results are consistent with the general notion that blade strike is related to overall fish size; the smaller the fish, the less likely it will be struck by a rotating blade (van Esch, 2012; Romero-Gomez and Richmond, 2014; van Esch and Spierts, 2014). The quantitative survival predictions for small fish are likely to be further underestimated owing to drag-vs-inertia effects where their large surface area to mass ratio may help them to be pulled around the blade rather than colliding with it. While it is acknowledged that turbine blade strike impacts should be empirically assessed to validate the modeling predictions presented in this study, this requires a functioning pumped hydroelectric simulator, at a scale close to a PHES turbine. This work could be extended to assess the percentage of struck fish that end up dying (i.e., the mutilation rate).

Although the shear, blade strike, and/or pressure stressors related to turbine passage may not be immediately lethal, fish may die from secondary effects such as physical injuries (e.g., an abrasion or hemorrhage) or later succumb to the effects of the injury or disease (Cada, 2001; Neitzel et al., 2004). Swimming impairments such as disorientation, loss of equilibrium and erratic swimming can leave fish susceptible to predation immediately following hydroelectric passage (Walker et al., 2016). For example, (Groves, 1972) reported that fish exposed to a shear environment were disoriented but regained "normal capacities" within 5–30 min. Similarly, Neitzel et al. (2000) subjected rainbow trout to predators after they were exposed to strain rates of 688 s^{-1} and found they were more susceptible to predation. Collectively these data suggest that exposure to strain rates causing disorientation could result in indirect mortality. During compression experiments, we observed that fish quickly became negatively buoyant (i.e., sank) once under pressure and therefore these fish would generally be unable to maintain their position in the water column whilst being transported to the turbine. They would likely contact continuously through the tunnel wall but the stress and impacts of such passage could not be tested within the current study.

Our independent assessments were designed to simulate some of the physical conditions encountered by gambusia as they pass through a PHES, allowing us to isolate specific factors that may influence fish survival. The specifications and combination of experiments provided here are one step toward assessing the risk of species transfer through the PHES scheme prior to its construction. We acknowledge our study is limited by the assumption that a fish subjected to one of these stressors (e.g., shear) does not become more susceptible to being negatively impacted by other hydroelectric-related stressors (e.g., blade strike and/or pressure changes or even multiple sources of shear during passage). For example, a fish that has become disorientated following exposure to shear stress could become more vulnerable to blade strike. It may well be that a fish exposed to shear, blade strike or pressure could be injured or stressed (Coutant and Whitney, 2000) and the cumulative and combined exposure to multiple stressors could result in reduced survival. One study investigated the component sources of entrainment mortality for juvenile *Gambusia affinis* using a nuclear power plant simulator, by exposing the fish to different combinations of pump speed and water temperatures (Cada et al., 1980). While the stressors were different to what we tested, their study showed that although a single factor (e.g., thermal shock or shear forces) may not have a major effect on entrained fishes, its influence may still be exerted through interaction with other stress factors that are a part of the entrainment experience. In any case, *G. affinis* suffered relatively low mortalities, even when combined with thermal shocks of 10°C above ambient temperatures (Cada et al., 1980). Gambusia survival may be lower due to interacting hydraulic variables in a PHES scheme, and these warrant empirical testing. Nonetheless, gambusia did have high survival for the individual stressors tested in the present study (shear strain, extreme pressurization and blade strike) and can potentially survive multiple stressors (Cada et al., 1980), thus a precautionary approach with regard to management measures should be applied.

The present study focuses on passage survival and was conducted based on the assumption that adult gambusia will become entrained in the PHES scheme. For a more complete assessment of the likelihood that a fish species will establish a new population with the facilitation of a PHES scheme, there are three key components: (1) entrainment risk, including the location of the intake relative to the target species' habitat (Huang et al., 2015; Langford et al., 2016), (2) passage survival through the PHES scheme, and (3) suitable habitat in the new environment (in the present case, the upper reservoir). The susceptibility of a particular species to entrainment can be examined using a range of approaches, including using existing knowledge of species ecology and habitat requirements, obtaining empirical data on the spatio-temporal distribution and swimming ability/rheotactic response of the target species, using computational fluid dynamics (CFD) modeling around the intake (Coutant and Whitney, 2000; Przybilla et al., 2010), laboratory and field experiments (Mussen et al., 2013, 2014), and acoustic telemetry (Stuart et al., 2010; Martins et al., 2014) or underwater cameras (Silva et al., 2018). To predict whether the fish species will establish a population in the new environment following survival through a PHES, screening tools that incorporate species life history and environmental tolerances can be applied (Kolar, 2004). Furthermore, the present study could be repeated to include a survival assessment of the early life stages of gambusia, as they are also a potential mechanism of dispersal.

CONCLUSION

Gambusia had high survival when exposed to individual stressors tested in the present study (shear strain, extreme pressurization and blade strike). Thus, there is more than a marginal risk that gambusia will survive passage to potentially colonize the upper reservoir of a PHES scheme, should they be entrained at the intake. Where a new PHES scheme poses the risk of inadvertently expanding the range of gambusia or similar poeciliid species, measures to minimize their spread or mitigate their ecosystem impacts should be considered. Prior to the consideration of any mitigation or management measures, however, an assessment of their applicability to the site and species of interest is required, followed by experimentation using both laboratory and field conditions, if warranted. Ideally, these measures should be a key consideration during the design and environmental planning phase of a PHES development.

DATA AVAILABILITY STATEMENT

The datasets presented in this article are not readily available. Certain datasets generated for this study may be available on request to the corresponding author. Data associated with confidential information relating to specific design of the turbines has not been made public and may not be released. Requests to access the datasets should be directed to Katherine Doyle, kadoyle@csu.edu.au.

ETHICS STATEMENT

The animal study was reviewed and approved by Charles Sturt University Animal Care and Ethics.

AUTHOR CONTRIBUTIONS

LS, LB, and CB conceived the concept of the study. LB and CB were responsible for the design of the shear testing facilities. JdP designed and built the pressure chamber. LS, LB, CB, NN, and KD designed the experimental procedures. KD, NN, LS, EB, and LB conducted the experiments. KD, NN, LS, EB, LB, and CB wrote and reviewed the manuscript. WR conducted the statistical analysis. ZD and TF developed and generated the blade strike models. All authors contributed to the article and approved the submitted version.

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Further Development of Small Hydropower Facilities Will Significantly Reduce Sediment Transport to the Pantanal Wetland of Brazil

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Small hydropower (SHP) facilities, which are defined by installed capacities <10–50 MW, are increasingly being built around the world. SHPs are viewed as less environmentally harmful than larger dams, although there has been little research to support that assertion. Numerous SHPs have been built, and many more are in development or proposed, in rivers that drain into the Pantanal, a world-renowned floodplain wetland. Three river systems with the largest contributions of sediments to the Pantanal—the Cuiabá, upper Taquari, and Coxim rivers—remain largely undammed. The upland tributaries transport sediments into the Pantanal, thereby affecting geomorphological dynamics and biological productivity of downstream floodplains. This study presents measurements from upstream and downstream of current hydropower facilities, most of which are SHPs, throughout the upland watersheds of the Upper Paraguay River basin to reveal how these facilities may affect the transport of suspended sediments and of bedload sediments. In addition, a predictive model using artificial neural networks (ANNs) estimates the impact of building 80 future SHPs on sediment transport based on observations at current facilities as well as the spatial distribution of future facilities. More than half of current facilities retained suspended sediments: 14 of the 29 facilities showed >20% net retention of suspended sediments, two others retained between 10 and 20%, seven were within 10%, and six showed >10% net release. Bedload sediment transport was a small component of total sediment transport in rivers with high total sediment loads. Multiyear series of satellite images confirm sediment accumulation in several cases. Model predictions of the impacts of future hydropower facilities on suspended sediment concentrations and transport show retention of a large fraction (often much >20%) of sediment inputs. Summing riverine transport rates for inflows into the Pantanal indicates that currently envisioned future hydropower development would reduce the suspended sediment transport by ∼62% from the current rate. This study shows that if SHPs are built on sediment-rich rivers, this may prove problematic for the facilities as well as for downstream ecosystems. These results support recommendations that several river systems presently lacking dams in their lower reaches should be excluded from future hydropower development to maintain the sediment supply to the Pantanal.

Keywords: hydroelectricity, dams, tropical, sediments, bedload

INTRODUCTION

One of the most important effects of the construction of dams on rivers is the retention (trapping) of sediment (Syvitski et al., 2005). Conventional storage dams with reservoirs that are large relative to the river channel (i.e., high reservoir capacity to inflow ratios) often have sediment trapping efficiencies exceeding 90% when newly constructed (e.g., Kondolf et al., 2014b), though the efficiency may decline over time if sediment infilling reduces reservoir capacity. As reservoir capacity decreases relative to inflow discharge, sediment trapping will decline and ultimately reach a point where it is negligible because flow and turbulence maintain sediment transport through the reservoir. Toward that end of the continuum, the design of the hydropower facility, the particle size distribution of transported sediments, and the seasonal patterns of discharge and sediment transport become important variables, although relatively few studies have considered the effects of smaller dams on sediment transport (Csiki and Rhoads, 2010).

The possible retention of sediments by small hydropower (SHP) facilities is important to understand because most hydroelectric dams being built around the world today are SHPs. Given their small size, they would seem to be less environmentally harmful than larger dams, but the effects of modern SHP designs on riverine transport of sediments have seldom been investigated, and almost no research exists outside of developed regions in the North America and Europe (Mbaka and Mwaniki, 2015; Couto and Olden, 2018). Despite the paucity of information, many countries have enacted policies that promote SHPs, including minimal environmental review. Where multiple SHPs are or will be located in series along river systems, their cumulative effects on sediment transport to downstream ecosystems deserve attention (Kibler and Tullos, 2013; Athayde et al., 2019).

The Amazon, Paraná, and Paraguay river watersheds of South America have a large number of existing and proposed SHPs (Couto and Olden, 2018), including in the watershed of the Upper Paraguay River basin in Brazil that drains to the Pantanal floodplains (**Figure 1A**). The Pantanal, which is internationally recognized as a globally important wetland ecosystem, is a 140,000-km^2 complex of coalesced alluvial fans, much of which is subject to seasonal inundation by riverine overflow that commonly lasts for months (Hamilton et al., 1996). The upland tributaries transport sediments into the Pantanal, which affect the geomorphological dynamics of channel and floodplain features, aquatic-terrestrial connectivity, and soil fertility of downstream floodplains. Floodplain lands subject to inundation by sediment-rich river water tend to be more productive (Hamilton, 2002; Junk et al., 2011), and sediment-rich rivers maintain dynamic changes in channel and floodplain geomorphology, which in turn increase floodplain ecosystem diversity (e.g., the Taquari River fan in the Pantanal: Jongman, 2005). In addition to impacts on the transport of sediments and associated nutrients, many species of migratory fishes important to fisheries in the region ascend the upland tributaries to spawn and dams present migration barriers (Campos et al., 2020).

In the Pantanal watershed, as of 2018 there were 47 hydropower facilities in operation (hereafter "current hydropower facilities"), all but four of which are SHPs, with an additional 138 projects under construction, planned, proposed, or identified by the government as prospective sites (hereafter "future hydropower facilities") [Agência Nacional de Águas [ANA], 2018; **Figure 1A**]. Many of the current and future projects are closely situated along river reaches, creating "cascades" where one project begins a short distance below the end of an upstream one.

In light of the ongoing construction and planning of future SHPs in the Pantanal watershed, there is an urgent need to understand how numerous SHPs on the tributaries may, in aggregate, alter the transport of sediments and nutrients from the uplands into the Pantanal. In recognition of these needs, the present study is part of a multidisciplinary research program that has examined many dimensions of the issues surrounding hydroelectric facilities in the tributaries of the Pantanal, including hydrology (Collischonn et al., 2019; Figueiredo et al., in review), sediment transport (this study), nutrient transport (Oliveira et al., in press), and fish and fisheries. Here we present measurements from above and below a number of current hydropower facilities throughout the Pantanal watershed to reveal how these facilities may trap sediments and thereby affect downstream sediment transport to the Pantanal. In addition, we develop a predictive model using artificial neural networks (ANNs) to estimate the impact of future hydropower development on sediment transport into the Pantanal, based on observations at current facilities as well as the distribution of future facilities. A companion paper in this journal (Oliveira et al., in press) presents a complementary examination of how SHPs affect nutrient transport to the Pantanal, including particulate forms associated with sediment transport; the two studies are parts of the same project but are presented separately because of their distinct methodologies and the different ecosystem implications of changes in sediment and nutrient transport. We conclude both

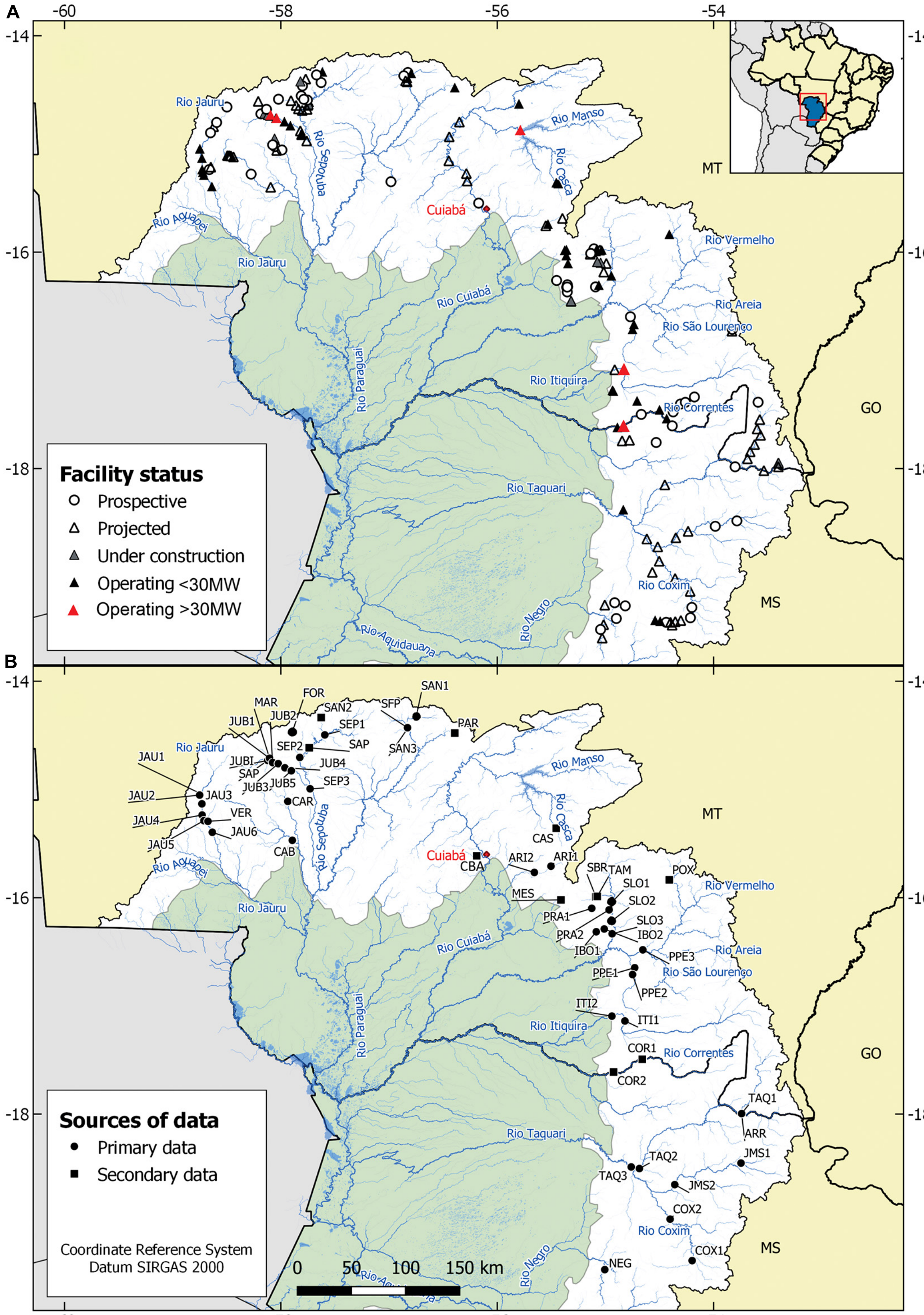

FIGURE 1 | (A) Hydropower facilities in the upland watershed of the Pantanal that are currently in operation as well as future projects that are under construction, planned, or identified as potential sites for hydropower development by either the Brazilian National Electric Energy Agency (ANEEL) or the state environmental agencies (depending on location). Red triangles indicate the four studied facilities with installed capacities >30 MW. **(B)** Sampling sites for sediment transport, including sampling conducted by the authors (primary data) as well as secondary data derived from environmental compliance reports submitted to state agencies (SEMA-MT and IMASUL) and from previous scientific studies. The Pantanal floodplains are shaded in green and rivers (*rios*) and other water bodies are shown in blue. Sampling site codes are identified in **Table 1**.

papers with recommendations developed from consideration of how both sediment and nutrient transport are impacted by future SHP development.

MATERIALS AND METHODS

Study Site

This study examines rivers in the Brazilian portion of the Upper Paraguay River basin that drain to the Pantanal wetland, which in turn drains to the Paraguay River. The upland watershed (150 to 1,400 m a.s.l.) represents 59% of the basin area and lies mainly within Brazil east and north of the Pantanal. Sloping terrain in much of the uplands favors rapid runoff and high sediment production. The Pantanal floodplains lie between 80 and 150 m a.s.l. The Köppen-Geiger climate classification is tropical savanna, with mean annual precipitation in the uplands ranging from 1,200 to 1,600 mm, with ~80% of the annual rainfall in the rainy season from October to April (**Supplementary Figure S1**). The native vegetation in the uplands is Cerrado savanna. Soil erosion has been increased by conversion of extensive areas to cropland (29% of the upland watershed area analyzed in this study) or pasture (22%) (Zeilhofer et al., 2006). Cattle ranching, subsistence and recreational fishing, and ecotourism are major economic activities within the Pantanal, and the floodplains are globally recognized by conservation organizations because they harbor important populations of several endangered mammals and birds (Tomás et al., 2019).

Hydropower Facilities

The characteristics of the current hydropower facilities studied here, as well as the river reaches in which they occur, are given in **Supplementary Table S1**. All but six of these facilities are SHPs, with installed capacities <30 MW. Three exceed 100 MW. However, installed capacity is not always directly related to the degree to which the passage of river water is slowed, and thus to potential effects of these facilities on sediment transport. For example, two of the facilities that exceed 30 MW (Juba I and II, each 42 MW) have dams and reservoirs similar in size to the SHPs, and one of the SHPs (São Lourenço, 29 MW) creates a reservoir comparable in size to larger facilities such as the largest one studied here, Ponte de Pedra (176 MW). Therefore, we analyze the SHPs and larger facilities together in this study.

Many of the SHPs are located on lower-order rivers with low elevational gradients, and most are diversion designs, where a low dam with a small or non-existent reservoir diverts most of the river discharge into an artificial channel (headrace) that carries the water for up to several km to a powerhouse farther down the river valley (**Supplementary Table S1**), leaving the natural channel with as little as 10% of the discharge until the water is returned below the powerhouse. Most of these facilities are "run-of-river," meaning that they cannot alter discharge except on short time scales (Csiki and Rhoads, 2010; Kaunda et al., 2012; Figueiredo et al., in review).

Data on discharge and suspended sediment concentrations (SSCs) from upstream and downstream of current SHPs and several larger hydropower facilities, as well as in reaches where such facilities may be built in the future, were obtained from our own sampling and measurements (primary data) as well as from reports submitted by hydropower companies to the state environmental agencies as required for environmental compliance (secondary data). The SSC and discharge measurements conducted for environmental compliance followed the same field and laboratory methods we used, and analyses were conducted only by certified laboratories with appropriate quality assurance protocols. Secondary data were only included for reaches that we did not sample and the two data sources were never combined for a particular reach. Bedload sediment transport was estimated only where we sampled. The distribution of sampling sites with primary or secondary data is shown in **Figure 1B**, and sampling site codes used in figures are listed in **Table 1**.

Sample Collection and Analysis

Primary data on discharge, SSCs, and bedload sediment transport above and below current hydropower facilities were collected on 13 dates spanning the wet and dry seasons from October 2018 to May 2019 (some locations had fewer dates). The primary data set contains data for 17 hydropower facilities. In addition, on 6–13 dates we sampled a number of rivers at locations close to where SHPs may be constructed in the future. More detail on the sampling sites shown in **Figure 1** is in **Supplementary Table S2**.

At each sampling location we recorded the bathymetric profile of the channel cross-section and installed a staff gage unless one already existed there (a number of gages are maintained by hydropower companies). Discharge was measured across the channel profile on each sampling date using an acoustic Doppler current profiler (*SonTek RiverSurveyor-M9*) following the methods outlined in Agência Nacional de Águas [ANA] (2012). For nine rivers where discharge could not be measured—the Paraguai, Casca, Mestre, Saia Branca, Tenente Amaral, Caeté, Gloria, and Poxoréo rivers—a hydrological model provided estimates (Collischonn et al., 2019).

We collected depth- and flow-integrated water samples by the equal-discharge-increment method. Depth-integrated samples of the water column at each point were obtained with either a DH48 or DH59 integrating sampler depending on hydraulic conditions. Samples from each point were composited in a mixing bucket in volumetric proportion to the discharge contribution of each point, as determined from the profiler data using custom software from the Brazilian National Water Agency [Empresa de Pesquisa Agropecuária e Extensão Rural de Santa Catarina [EPAGRI], 2013]. We analyzed total SSC in water samples gravimetrically after collection of the sediment on filters (0.6 μm pore size). Laboratory analyses of SSC were conducted at Department of Sanitary Engineering at the Federal University of Mato Grosso.

Transported bedload material was collected using a Helley-Smith sampler, and sediment samples from river beds for granulometric analysis were collected using a BMH-60 rotary-bucket bed material sampler (Carvalho, 2008). The Helley-Smith sampler was deployed at three points across the channel. All sediment samples were dried at 105°C for 24 h before

TABLE 1 | List of sampling sites and hydropower facilities they pertain to, with codes for figures and tables.

Water-shed	Tributary	Sampling site	Code	Hydropower facility names	
				Current	Future
Paraguay	Paraguay	Paraguay River, upstream and downstream of SHP Alto Paraguay	PAR	Alto Paraguai	
	Santana	Santana River, upstream SHP Diamante	SAN1	Santana I	
		Santana River at mouth, upstream of SHP Santana I	SAN2		Santana II
		Santana River, downstream of SHP Santana I	SAN3	Santana I	
	S. F. Paula	São Francisco de Paula River, downstream of proposed SHPs	SFP		Saíra, Jaçanã Alta, Biguá
Sepotuba	Maracanã	Maracanã River at mouth	MAR		Taquarinha, Medianeira
	Sapo	Sapo River, ∼13 km upstream of SHP rio Sapo	SAP		Lagoa Grande, Ponte Estreita
		Sapo River, upstream and downstream of SHP rio Sapo	SAP	Rio do Sapo	
	Formoso	Formoso River, ∼250 m upstream of mouth	FOR		Formoso I, II e III
	Jubinha	Jubinha River, upstream of SHP Juba I	JUBI	Juba I	Jubinha I, II e III
	Juba	Juba River, upstream of SHP Juba I	JUB1	Juba I	Juba III e IV
		Juba River, downstream of Juba I Hydroelectric Facility	JUB2	Juba I	
		Juba River, downstream of Juba II Hydroelectric Facility	JUB3	Juba II	
		Juba River, downstream of SHP Graça Brennand	JUB4	Graça Brennand	
		Juba River, downstream of SHP Pampeana	JUB5	Pampeana	Corredeira, Tapirapuã
	Sepotuba	Sepotuba River, downstream of Maracanã River	SEP1		Salto das Nuvens, Sepotuba
		Sepotuba River, downstream of Formoso River	SEP2		Paiaguás, Salto Maciel
		Sepotuba River lower mainstem, downstream Juba River	SEP3		
Cabaçal	Cabaçal	Cabaçal River lower mainstem, downstream of proposed SHPs	CAB		Cabaçal 1,2,3,4,5, and 6
	Caramujo	Caramujo River, downstream of proposed SHPs	CAR		Salto do Céu, Salto Cacau, Salto Vermelho I, Salto Caramujo
Jauru	Jauru	Jauru River, upstream of SHP Antonio Brennand	JAU1	Antonio Brennand	Estivadinho III, Alagados III, Trairão III
		Jauru River, downstream of SHP Antonio Brennand	JAU2	Antonio Brennand	
		Jauru River, downstream of SHP Ombreiras	JAU3	Ombreiras	
		Jauru River, downstream of Jauru Hydroelectric Facility	JAU5	Jauru	
		Jauru River, downstream of SHP Salto	JAU4	Indiavai + Salto	
		Jauru River, downstream of SHP Figueirópolis	JAU6	Figueirópolis	
	Vermelho	Vermelho River at mouth, downstream of proposed SHPs	VER		Rancho Grande, Progresso
Cuiabá	Casca	Casca River, upstream and downstream of SHP Casca II e III	CAS	Casca II e III	
	Mestre	Mestre River, upstream SHP Mestre and downstream SHP Santa Cecilia	MES	Mestre + Santa Cecília	
	Cuiabá	Cuiabá River lower mainstem at Passagem da Conceição hydrological station	CBA		Perudá, Angatu II, Angatu I, Iratambé I, Iratambé II, Guapira

(Continued)

TABLE 1 | Continued

Water-shed	Tributary	Sampling site	Code	Hydropower facility names	
				Current	**Future**
	Aricá	Aricá River, upstream of SHP São Tadeu I	ARI1	São Tadeu I	Aricá-Mirim I
		Aricá River at mouth, downstream of SHP São Tadeu I	ARI2		São Tadeu II
São Lourenço	Tenente Amaral	Saia Branca River upstream and downstream SHP Sucupira	SBR	Pequi	
		Tenente Amaral River, ~10 km above mouth	TAM	Sucupia	Ipê, Mangaba
	Prata	Prata River, upstream of SHP Água Prata	PRA1	Água Prata	
		Prata River, downstream of SHP Água Prata	PRA2		Água Clara, Água Branca, Água Brava
	São Lourenço	São Lourenço River, upstream of São Lourenço Hydroelectric Facility	SLO1	São Lourenço	
		São Lourenço River, downstream of São Lourenço Hydroelectric Facility	SLO2		
		São Lourenço River lower mainstem, downstream of São Lourenço facility	SLO3		
	Ibo	Ibo River, upstream of SHP Sete Quedas Altas	IBO1	Sete Quedas Altas	Europa
		Ibo River, downstream of SHP Sete Quedas Altas	IBO2		
	Poxoréu	Poxoréu River, upstream and downstream of SHP Poxoréu	POX	Poxoréu	
	Ponte de Pedra	Ponte de Pedra River, upstream of SHP Eng. José Gelázio	PPE1	Eng. José Gelázio	
		Ponte de Pedra River, downstream of SHP Eng. José Gelázio	PPE2		
		Ponte de Pedra River, downstream of SHP Rondonópolis	PPE3	Rondonópolis	João Basso
Piquiri	Itiquira	Itiquira River, upstream of Itiquira hydropower facility	ITI1	UHE Itiquira	
		Itiquira River lower mainstem, downstream of Itiquira hydropower facility	ITI2		Itiquira III
	Correntes	Correntes River, upstream of Ponte de Pedra hydropower facility	COR1	UHE Ponte de Pedra	Água Enterrada, Santa Paula
		Correntes River lower mainstem, downstream of Ponte de Pedra facility	COR2		
Taquari	Ariranha	Ariranha River at mouth, downstream of proposed SHPs	ARR		Girassol, Dália, Lírio, Violeta, Orquídea, Primavera, Hortência
	Jauru	Jauru River at BR 359 bridge, upstream of proposed SHPs	JMS1		Jauruzinho, Barra do Piraputanga, Água Fria
		Jauru River, upstream of Coxim River and downstream of proposed SHPs	JMS2		Figueirão,Vila Jauru, Mundo Novo
	Coxim	Coxim River at Fazenda São José, upstream of Camapuã River	COX1		Entre Rios, Lagoa Alta, Ponte Vermelha, Calcutá, Maringá, Fazenda Caranda, Peralta, Água Vermelha
		Coxim River at MS-142 bridge, downstream of Jauru River	COX2		São Domingos, Sucuri
	Taquari	Upper Taquari River, upstream of Ariranha River	TAQ1		Taquarizinho, Barra do Ariranha
		Upper Taquari River at Silviolandia city	TAQ2		Pedro Gomes
		Taquari River below confluence with Coxim River	TAQ3		
Negro	Negro	Negro River, at Negro city	NEG		Rio Negro, Ouro Negro, São Francisco de Assis

In most cases "lower mainstem" refers to the river before its entry into the Pantanal, SHP, small hydropower facility.

weighing. Sediment granulometry was determined using dry sieving (Carvalho, 2008).

Suspended Sediment Data Compilation From Secondary Sources

The secondary data set contains SSC data for an additional 12 hydropower facilities, including water sampling both before and during their operation. In addition, we incorporated data on rivers not sampled during the present study that were available from the environmental agency of the State of Mato Grosso (SEMA-MT). Out of 980 sediment samplings in the secondary data set, we selected 401 that could be used to analyze the effects of current and future hydropower facilities. Selection criteria included correspondence in timing of the upstream and downstream samplings, and at least five dates of sampling. For cases where more than one location was sampled to represent upstream or downstream water quality, the mean was taken, weighted by the relative discharge in the case of more than one tributary coming together above a facility. Reported values that were below the detection limit of the SSC analyses (1 mg L^{-1} for primary data and 10 mg L^{-1} for secondary data) were substituted with the detection limit concentrations.

To estimate the total impacts of all current dams on sediment transport, the impact of the Manso dam in the Cuiabá River system was also considered (**Figure 1A**). The 212-MW Manso hydropower facility was constructed on the Manso River in the 1990s and creates an extensive reservoir of 427 km^2. We analyzed suspended sediment data for the Cuiabá River at the city of Cuiabá, downstream of the Manso River as well as additional tributary inputs that contribute suspended sediments. For the pre-reservoir period, 52 SSC measurements were made by the National Department of Sanitary Works (DNOS) between April 1977 and November 1981 (Barbedo, 2003). For the period after filling the reservoir, 79 SSC measurements were available between September 1999 and October 2016 [station 66260001: Agência Nacional de Águas [ANA], 2020]. These SSC measurements coincided with discharge measurements, allowing comparison of the SSC-discharge relationships.

Data Analysis

We assessed the effects of current hydropower facilities on SSCs and transport by comparing the median SSCs upstream and downstream of each facility, based on a combination of primary data (N = 13 dates in most cases) and secondary data (variable numbers of sampling dates). Transport was calculated as the median concentration times discharge, averaging the discharge estimates above and below each SHP location to avoid potential spurious results caused by the uncertainty inherent in discharge measurements as well as by short-term (sub-daily) fluctuations imposed by the hydropower facilities (hydropeaking: Figueiredo et al., in review).

Bedload sediment transport for cross sections represented by each sampling point was calculated from the Helley-Smith samples following Carvalho (2008):

$$Q_b = \left[\sum \frac{p(q_{i+1} - d_{i-1})}{E_{\mathrm{am}} \times l \times t} \right] \times 86.4 \qquad (1)$$

Where,

Q_b = total bedload sediment transport (load) in the channel (t d^{-1});
E_{am} = hydraulic efficiency of the sampler;
p = dry weight of the sediment sample (kg);
$(d_{i+1} - d_{i-1})$ = channel width of the cross-section represented by the sample (m);
l = width of the sampler opening (m);
t = time sampler was deployed (s).

Observed ratios of upstream to downstream SSCs and transport and bedload sediment transport were grouped into classes based on the percentage change in either direction (i.e., retention or release), similar to the sustainability boundary approach suggested by Richter et al. (2012) for analysis of flow regime alterations in river systems that lack detailed knowledge of the impacts of altered flows. Ratios of <10% were defined as undetectable changes, 10–20% as moderate, and >20% as high alterations.

In addition to comparing the median concentrations and transport rates for all sampling dates combined for each study reach, we conducted statistical analyses of the changes in suspended sediment and bedload transport observed across all individual sampling dates in each reach using a one-sample t-test for which the null hypothesis was zero change. Analyses were conducted after $\log_{10}$ transformation of the concentration changes to improve normality.

Prediction of Impacts of Future Hydropower Facilities

Artificial neural networks were developed to predict the impacts of new hydropower facilities on total suspended solid concentrations and bedload sediment transport. As one of the most commonly used artificial intelligence tools, ANNs are well suited to model phenomena subject to controls that are complex, incompletely understood, and likely non-linear [American Society of Civil Engineers [ASCE], 2000]. The ANN model architecture was a three-layer feedforward network with a non-linear (unipolar sigmoid) activation function (**Supplementary Figure S2**), similar to ANNs that have been applied to study floodplain inundation in the Pantanal (Fantin-Cruz et al., 2011) as well as elsewhere (Dawson et al., 2006).

The ANN models were trained with a data set representing 32 locations (including primary and secondary data). The back-propagative algorithm (Rumelhart et al., 1986), along with training acceleration techniques (Vogl et al., 1988) as well as other needed functions, were custom programmed in the MATLAB R2012b environment. Overfitting was avoided using the cross-validation technique (Hecht-Nielsen, 1989).

For cross-validation training, the data were divided into three samples (53% for training, 26% for validation, and 20% for verification), using a systematic sampling method to provide a representative distribution of the 32 locations for all samples. The extreme values (maximum and minimum) of all variables were included in the training samples and all input data of the future hydropower facilities were within the domain of the trained

ANN, ensuring that model predictions were within the ranges of the training data.

The complexity of the ANNs (number of neurons in the intermediate layer) was defined by the architecture search with the lowest possible complexity that still had the same approximation and generalization capacity of a purposely oversized ANN that was trained without overfitting. These approximation and generalization capacities were verified by the performance of the application to the validation sample, since the verification sample, by definition, cannot participate, neither in training nor in the definition of the ANN architecture (Hecht-Nielsen, 1989).

In the present study, input variables that were considered for the ANNs included contributing watershed area, reservoir area and volume, soil classification [11 classes in the contributing watersheds: Empresa Brasileira de Pesquisa Agropecuária [EMBRAPA], 2018], land use and cover [8 classes: Empresa Brasileira de Pesquisa Agropecuária [EMBRAPA], 2015], and upstream concentrations of suspended sediments. Sufficient information was available for 80 of the 138 potential future hydropower facilities.

The potential watershed yields of sediment to river systems were estimated from the Soil and water assessment tool (SWAT), the full results of which are posted online (Mingoti et al., 2020). Details of this SWAT model are presented by Neitsch et al. (2011) and Arnold et al. (2012). The SWAT model was run for each watershed for the year 2017 using spatial data on climate, topography, soils, and land use and cover. Annual and monthly inputs were estimated for the contributing watersheds above each current and future hydropower facility.

The ANN model was trained with measured concentrations of suspended sediments (n = 571 measurements) above and below 34 current hydropower facilities, as well as the contributing watershed area and reservoir area and volume for each facility (**Supplementary Table S1**). Pearson linear correlations between input and output variables indicated the best predictive variables for each model. Overall performance of the ANNs was evaluated by the Nash-Sutcliffe model efficiency coefficient.

Impacts of Damming on Total Sediment Transport to the Pantanal

We estimated the total sediment transport to the Pantanal from the most downstream sampling points on each major tributary (**Figure 1**). In addition, the aggregate impact of current hydropower facilities was estimated by comparing current sediment transport with the total sediment retention by all current facilities, including the Manso dam. These estimates of total sediment transport and retention at present were then compared with the ANN model predictions of the impacts of future hydropower development on sediment transport.

Remote Sensing of Sediment Accumulation

Time series of optical satellite imagery were visually examined for evidence of sediment accumulation in reservoirs among the study sites. Publicly available imagery was obtained from the Landsat 5 Thematic Mapper, Landsat 7 Enhanced Thematic Mapper Plus, Landsat 8 Operational Terra Imager, China–Brazil Earth Resources Satellite 4, and Sentinel 2 satellites. Two cases are presented here as instructive examples—the Itiquira and São Lourenço hydropower reservoirs.

RESULTS

Sediment Concentrations and Transport

Rivers flowing from the upland watershed into the Pantanal, sampled at points below any hydropower facilities, showed a wide range of SSCs and transport (**Figure 2** and **Supplementary Tables S3, S4**). The sites in **Figure 2** are ordered by increasing discharge, which shows that with the exception of the Coxim River (COX2), rivers below the median discharge had relatively low SSCs and transport. Most of the rivers with the highest SSCs and transport are as yet largely undammed (**Figure 1A**), including the Vermelho, Coxim, upper Taquari, Miranda, and Aquidauana rivers, although the Cuiabá River still carries relatively high SSC loads in spite of a large upstream reservoir (Manso) on one of its principal tributaries (Manso River) that traps nearly all suspended sediment inputs (Carvalho and Lôu, 1990).

Bedload sediment transport in rivers flowing from the upland watershed into the Pantanal was small compared to the

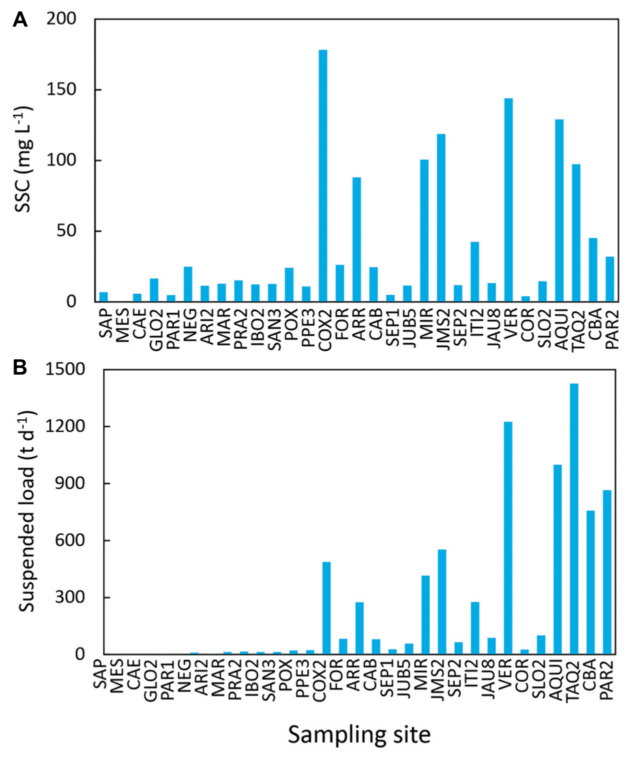

FIGURE 2 | (A) Suspended sediment concentrations (SSC) and **(B)** loads (transport) in rivers that drain into the Pantanal, for sampling sites below any hydropower facilities. Sites are ordered by increasing discharge and their codes are identified in **Table 1**.

suspended sediment load in the rivers with high total sediment loads (**Figure 3**). Bedload comprised the largest percentage of total sediment transport in the Negro and Formoso rivers, which carried low SSCs.

Upstream-Downstream Comparisons to Show Dam Effects

Concentrations and transport of suspended sediments upstream and downstream of 29 current hydropower facilities are compared in **Figure 4** and **Supplementary Table S3**. The changes between upstream and downstream are presented as both concentrations, which bear on the nutrient availability and aquatic ecology of downstream waters, and rates of transport (i.e., t d^{-1}), which bear on the overall sediment supply from the upland watersheds to the Pantanal. For a given hydropower facility, the ratios between concentrations and transport covary because we assumed that discharge did not change and therefore we used the mean of upstream and downstream discharge measurements to calculate transport. We do not consider bedload here because it was a small proportion of total sediment transport except in some rivers with low total sediment transport (**Figure 3**).

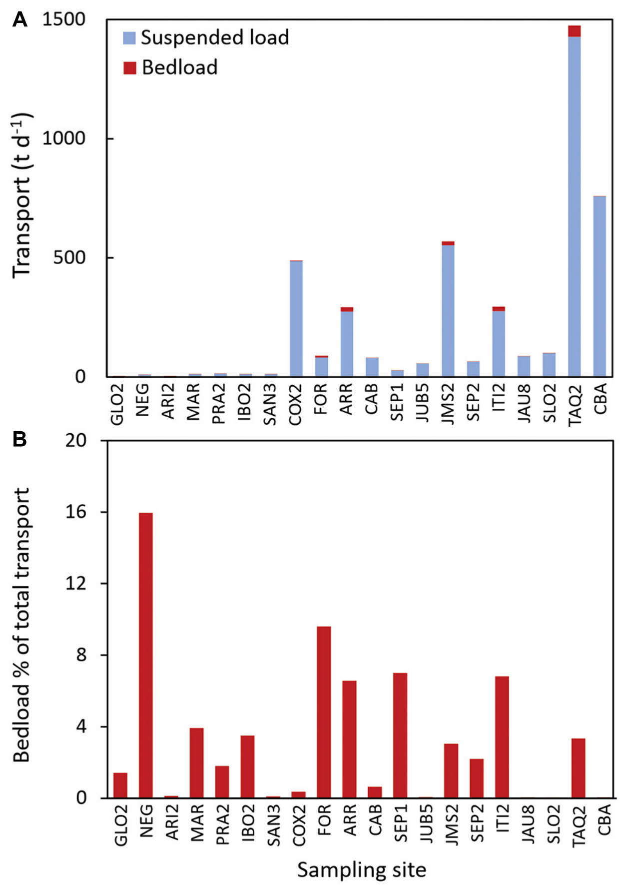

FIGURE 3 | (A) Suspended and bedload sediment transport and **(B)** bedload as percent of total sediment transport (i.e., bedload plus suspended) in rivers that drain into the Pantanal, for sampling sites below any hydropower facilities. All sites shown here have bedload data but in some cases the bars are too small to be visible in **(B)**. Sites are ordered by increasing discharge and their codes are identified in **Table 1**.

Fourteen of the 29 current hydropower facilities showed upstream:downstream SSC ratios >20% above the 1:1 line, indicating net retention of suspended sediments (**Figure 4**). Two others fell between 10 and 20% retention, seven were within 10%, and six showed >10% release. Two of the reaches with the highest SSCs and transport—ITI2 on the Itiquira River and SLO2 on the São Lourenço River—also showed large absolute values of net retention. That is not the case for the next three reaches with relatively high SSCs (POX, CAS, and PPE3), which also retained a large fraction of the suspended sediment input but had much lower rates of transport. Summing all of the riverine transport rates for above and below the 29 facilities studied here shows a total net retention of suspended sediments of 140 kt y^{-1} (i.e., from 518 to 378 kt y^{-1}).

The nearly complete retention of sediments by the large reservoir on the Manso River can be added to the above estimate. Comparison of the SSC-discharge relationships for the Cuiabá River before and after construction shows that damming the Manso River reduced suspended sediment transport downstream in the Cuiabá River at the city of Cuiabá by ~60% (**Supplementary Figure S3**). The total retention by all dams in the upland watershed draining to the Pantanal is thus estimated to be 865 kt y^{-1}, amounting to a 32% reduction in aggregate sediment transport by current hydropower facilities (**Figure 5**).

Net sediment retention by individual hydropower facilities (not including Manso) was best predicted by the upstream SSC concentration and the water residence time in the reservoir, which together explained 54% of the variation using a regression tree (**Supplementary Figure S4**). Facilities where the upstream SSC exceeded 18 mg L^{-1} showed a mean SSC retention of 59%. Residence time was a significant predictor for reaches with SSC < 18 mg L^{-1}, with net retention (mean, 31%) only when the residence time exceeded 1.8 days.

A similar upstream:downstream comparison for the 16 sites that have bedload sediment transport measurements shows that two reaches with the largest absolute rates of bedload sediment transport–SLO2 and ITI2–showed large deviations from the 1:1 line, but in opposite directions (**Figure 6**). The SLO2 reach retained virtually all of its bedload, whereas the ITI2 reach showed net release (the Itiquira reservoir was largely infilled with sediment). Among the reaches with much lower rates of bedload sediment transport, most showed net retention of most of the bedload, although JAU2 on the Jauru River in Mato Grosso (Jauru-MT) showed net release (**Figure 6B**). Several reaches showed bedload retention rates that were so small that any net changes may be inconsequential to downstream ecosystems. Summing all of the riverine transport rates for above and below the 17 facilities with measurements of bedload shows a total net retention of bedload sediments of 2.01 kt y^{-1} (i.e., from 8.64 to 6.64 kt y^{-1}). Actual total retention is considerably larger because bedload was only measured on a subset of the reaches with hydropower facilities.

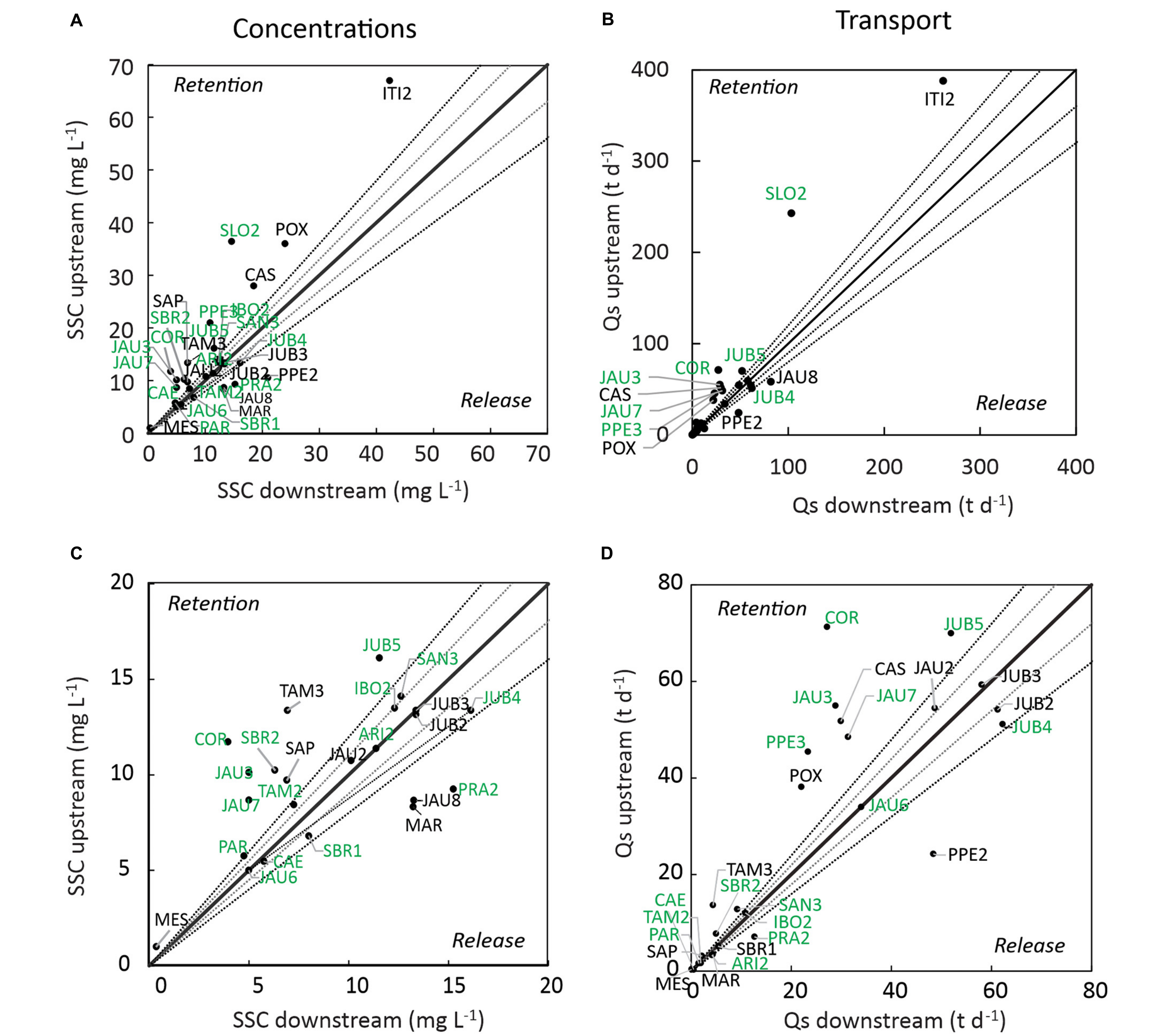

FIGURE 4 | Comparisons of changes in water-column concentrations (SSC) and transport (Qs) of suspended sediments between upstream and downstream of current hydroelectric facilities based mainly on primary data collected in this study as well as some secondary data. **(A,B)** All data on concentrations and transport, respectively; **(C,D)** the same data with high values excluded. Solid line shows the line of parity and dashed lines show bounds of ±10 and ±20% around that line; points above the line indicate net retention and those below indicate net release between the upstream and downstream sampling points. Upstream:downstream ratios that deviate considerably from 1:1 are identified with the codes shown in **Table 1**; codes in green font indicate cases where the statistical analysis of the changes across all individual sampling dates showed significant ($p < 0.05$) differences from zero.

Dividing sediment transport rates for rivers at their points of entry into the Pantanal by watershed areas gives specific sediment yields, which range from 3.38 to 45.5 t km^{-2} y^{-1}. These are compared against potential sediment production estimated by Mingoti et al. (2020) using the SWAT model in **Figure 7** and **Supplementary Table S5**. The ratio of transport to potential production, known as the sediment delivery ratio, ranged from 0.0015 to 0.0098 with the highest ratios in the Cuiabá (0.0098), upper Taquari (0.0092), and Coxim (0.0081) rivers. The sum of sediment transport by upland rivers into the Pantanal was 1.91 Mt y^{-1}, or 23 t km^{-2} y^{-1}, and the overall sediment delivery ratio was 0.0074.

Predicting the Impacts of Future Hydropower Development

Among the potential input variables that were considered for the ANNs, the most significant predictor of sediment retention was the measured upstream SSC concentration, accounting for nearly half of the predictive capability of the model

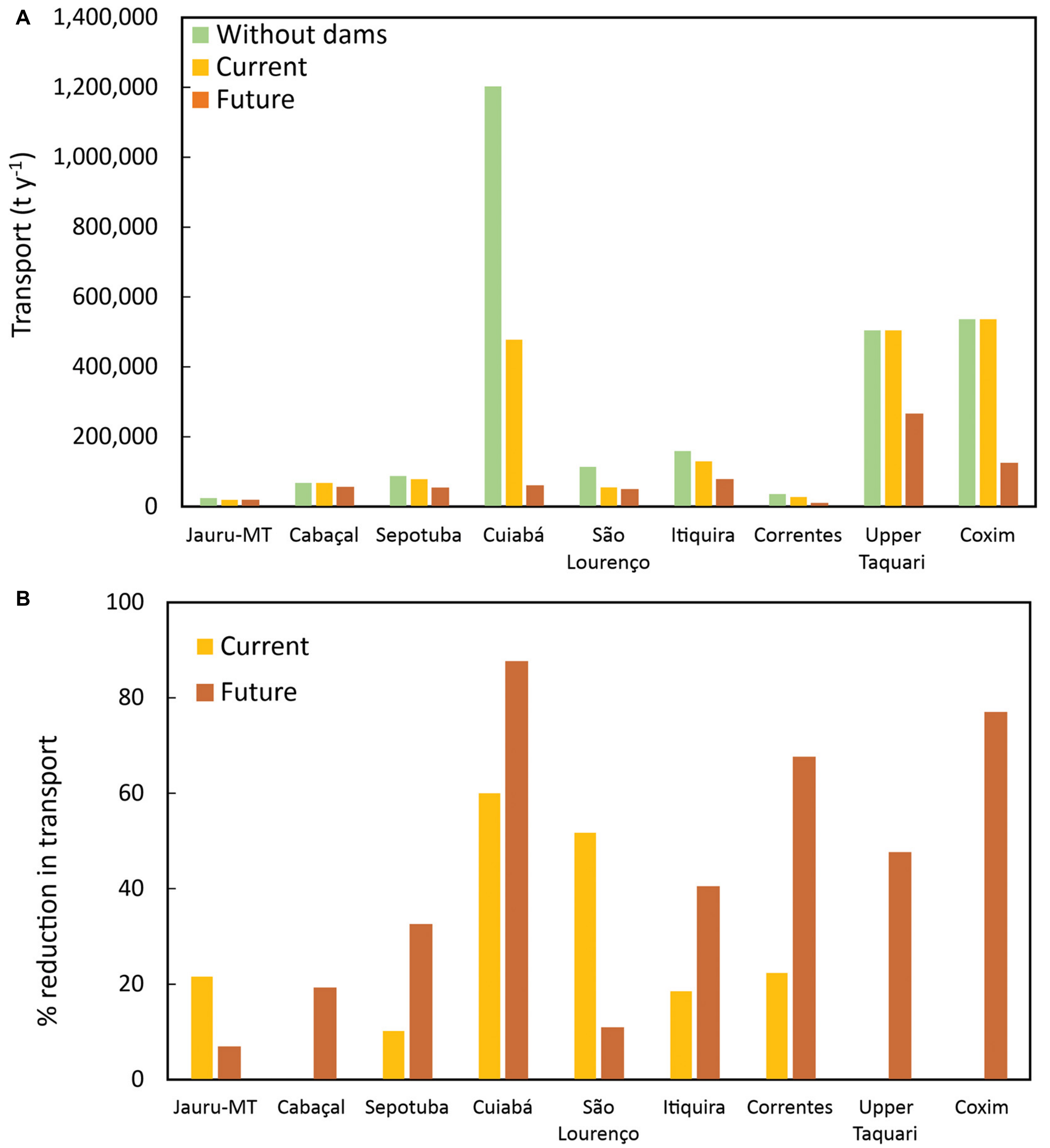

FIGURE 5 | Impacts of damming on total sediment transport to the Pantanal. **(A)** Annual suspended sediment transport to the Pantanal at present with and without current dams, as well as rates predicted by the neural network modeling were all future hydropower facilities to be built in each river system; **(B)** percent change between current rates and predicted future rates.

(**Supplementary Figure S5**). Also important were soil and land use classes (n = 11 and 10 classes, respectively) and watershed sediment yields from the SWAT model, together accounting for about 33% of the predictive capability. The contributing watershed area and reservoir area and volume were also significant, though less important, predictors of SSC. The performance of the ANNs was satisfactory as indicated by the Nash-Sutcliffe model efficiency coefficients, which were 0.889 for training and 0.826 for verification.

Model predictions of the impacts of future hydropower facilities on SSCs and transport show retention of a large fraction (often much >20%) of sediment inputs at most of the reaches across the range of concentrations and transport (**Figure 8**). The measured current and modeled future rates of suspended sediment transport by the major rivers flowing from the upland watershed into the Pantanal, based on the most downstream sampling points, are summarized in **Figure 5**.

Very large decreases in sediment transport are predicted for the three river systems with the largest contributions of sediments to the Pantanal—the Cuiabá, upper Taquari, and Coxim rivers. A number of rivers with lower rates of sediment transport are still predicted to show significant percent reductions if future SHPs are built (**Figure 5B**). Available data for a few smaller rivers with relatively low rates of transport are not shown (Negro-MS,

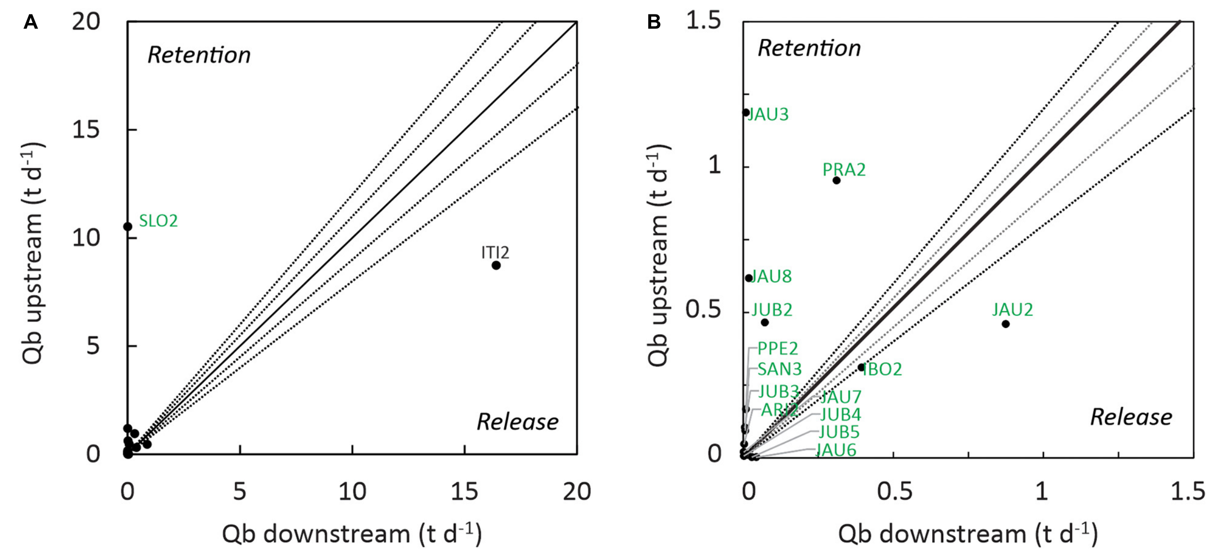

FIGURE 6 | Comparisons of changes in bedload sediment transport between upstream and downstream of current hydroelectric facilities based on primary data collected in this study. **(A)** All data; **(B)** the same data with high values excluded. Solid line shows the line of parity and dashed lines show bounds of ±10 and ±20% around that line; points above the line indicate net retention and those below indicate net release between the upstream and downstream sampling points. Upstream:downstream ratios that deviate considerably from 1:1 are identified with the codes shown in **Table 1**; codes in green font indicate cases where the statistical analysis of the changes across all individual sampling dates showed significant ($p < 0.05$) differences from zero.

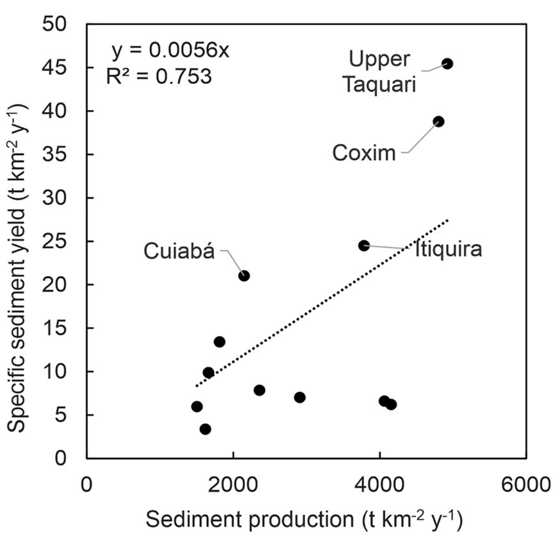

FIGURE 7 | Comparison of specific sediment yields measured in rivers near their points of entry into the Pantanal to potential sediment production in their watersheds as estimated using the SWAT model (Mingoti et al., 2020).

Aricá Mirim, and Ribeirão Ponte de Pedra rivers), and the Taquari River is divided into its two major tributaries (upper Taquari and Coxim rivers) whose confluence is a short distance upstream of the border of the Pantanal. Summing all of the riverine transport rates for inflows into the Pantanal indicates that future hydropower development would result in reductions of 62% of the suspended sediment transport from the uplands to the Pantanal (i.e., from 1.93 to 0.73 Mt y^{-1}).

Remote Sensing of Sediment Accumulation

Two examples of reservoirs with readily visible sediment accumulation over time are shown in **Figure 9**. The São Lourenço dam created a long, relatively narrow reservoir, and the visible sediment infilling has occurred in its uppermost reach, effectively forming a delta in which shallow or periodically flooded areas have become colonized by floating and emergent wetland vegetation. The Itiquira dam created a wide reservoir along a short reach and has filled in considerably in spite of persistent efforts to recover the sand over the years, an operation that is visible at the end of the road on the southeastern edge of the reservoir.

DISCUSSION

This study presents a comprehensive analysis of SHP impacts on sediment concentrations and transport that is unprecedented in the literature and informs decisions about future hydropower development in the Upper Paraguay River basin and in similar settings elsewhere in Brazil and worldwide. In particular, our results are applicable to some of the major frontiers of SHP development that are in similar landscapes, including elsewhere in Latin America, east Africa, and Southern Asia (Couto and Olden, 2018).

Sediment Retention by Current and Future Hydropower Facilities

This study shows that sediment retention, while well known for larger dams and reservoirs, also is characteristic of SHP facilities,

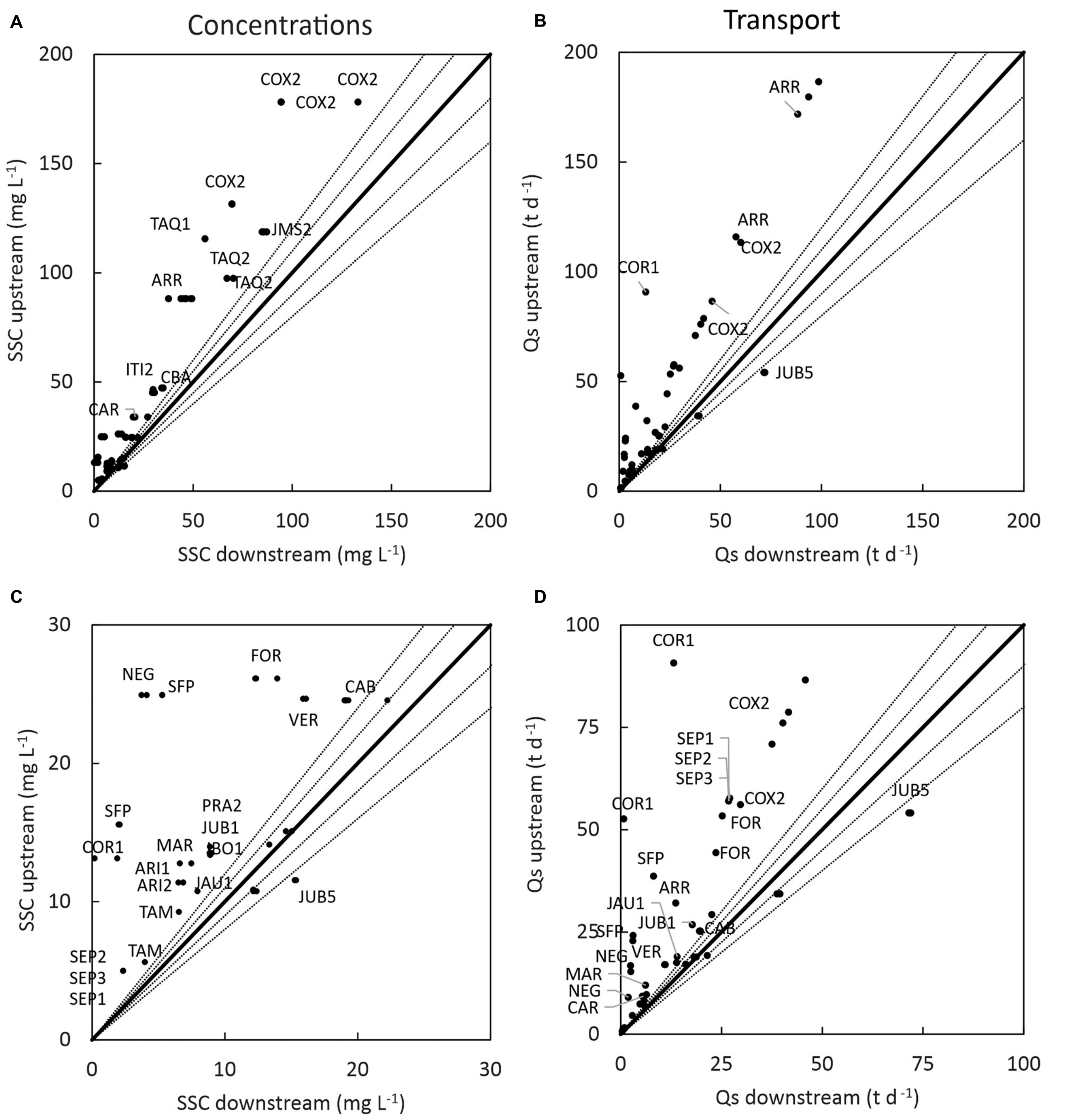

FIGURE 8 | Comparisons of neural network model predictions of changes in suspended sediment concentrations (SSC) and transport (Qs) between upstream and downstream of future hydroelectric facilities, with river systems labeled by codes in **Table 1**. **(A,B)** All data on concentrations and transport, respectively; **(C,D)** the same data with high values excluded. Solid line shows the line of parity and dashed lines show bounds of ±10 and ±20% around that line; points above the line indicate net retention and those below indicate net release between the upstream and downstream sampling points. Upstream:downstream ratios that deviate considerably from 1:1 are identified with the codes shown in **Table 1**.

and that without mitigation measures future construction of SHPs in the remaining undammed river systems with high sediment transport rates will reduce the total sediment transport from the upland watershed to the Pantanal by about 62% from the current rate. Many of the current hydropower facilities are on rivers with relatively low sediment concentrations and transport, and thus their impact on sediment transport from the uplands to the Pantanal has been modest. Nevertheless, certain tributaries with hydropower facilities are experiencing significant retention of suspended sediments, including the large effect of the dam

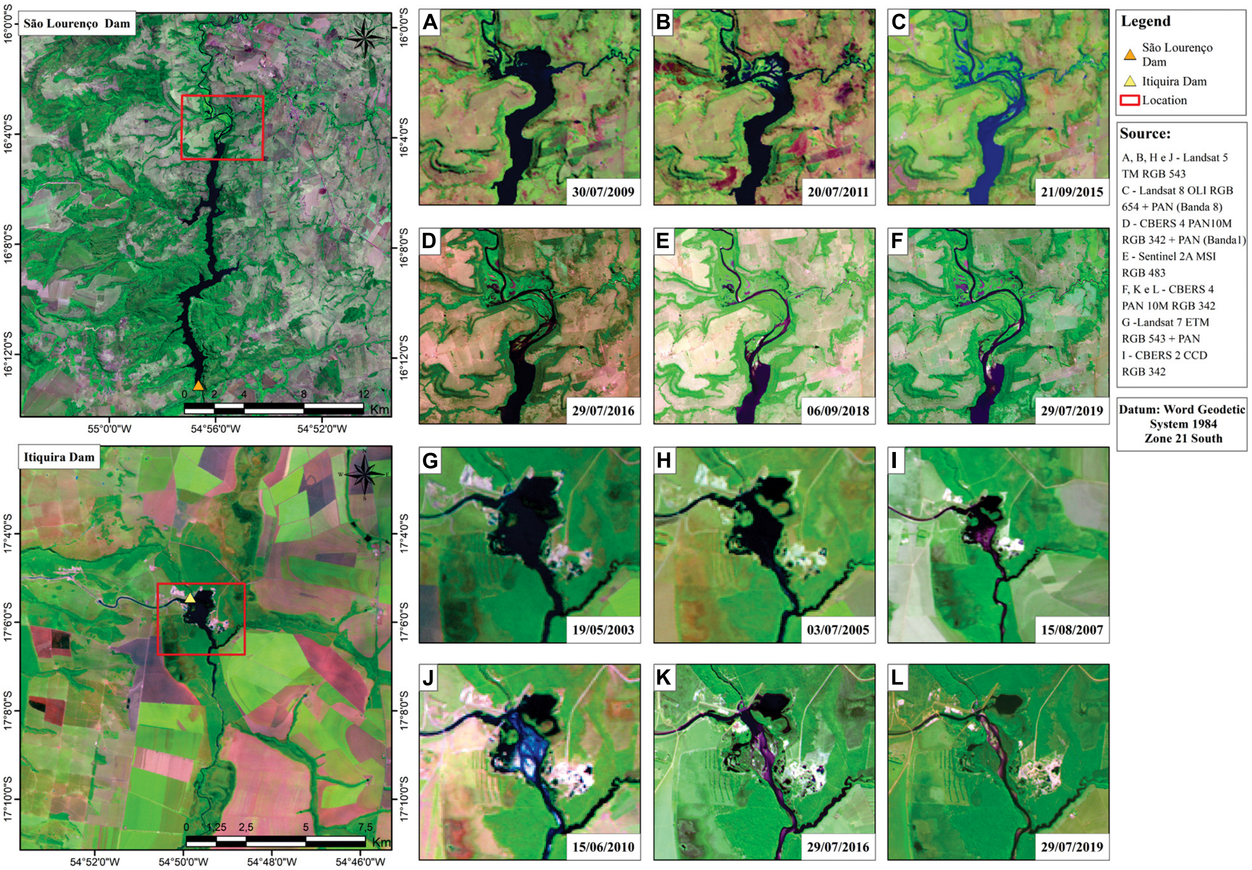

FIGURE 9 | Satellite images showing progressive sedimentation and infilling of reservoirs created by the **(A–F)** Itiquira dam (completed in 2002) and **(G–L)** São Lourenço dam (completed in 2009).

on the Manso River, and satellite images provide evidence of long-term accumulations of sediments above some dams.

The undammed river systems with high sediment loads reflect the combination of erodible soils and agricultural land use, including livestock grazing, in their watersheds, and our results show that damming those river systems may prove problematic for sediment management. The Coxim, upper Taquari, and Cuiabá rivers are particularly vulnerable to sediment trapping by future SHPs, which will likely result in serious problems of infilling above the dams. The sediment retention would also negatively impact downstream river channels and floodplains by altering the geomorphological dynamics, potentially creating a sediment starvation situation where river channels incise, banks become destabilized, and floodplain inundation and accompanying sediment and nutrient deposition are reduced, with negative consequences for fisheries, wildlife habitat, and agriculture and human settlements in the riparian zones (Kondolf et al., 2014a; Wohl et al., 2015).

The direct measurement of sediment retention reported in our study is superior to estimation using models developed to apply to a wider variety of rivers and hydropower facilities (reviewed in Tan et al., 2019). The Churchill (1948) model is considered appropriate for small reservoirs (Morris and Fan, 1998; Carvalho, 2008). Comparison of mean trap efficiencies based on our observations of sediment retention to those estimated with the Churchill model showed divergent results, however, with the Churchill model greatly overestimating sediment trapping at 11 of 16 facilities (data not shown).

A few previous studies in the region have predicted sediment trapping in spite of a lack of direct measurements upstream and downstream of the hydropower facilities. Based on considerably less available data on sediment transport and considering locations of current and future hydropower facilities as of 2010, and assuming 100% sediment trap efficiency by dams, Souza Filho (2013) estimated the potential future reduction in sediment loading to the Pantanal to total 52%, less than the 62.4% estimated in our forecasts. However, in the case of the Taquari River system, both the discharge and SSCs and therefore the rates of transport used by Souza Filho were considerably higher than our measurements (this could reflect a real change over time).

Sediment trapping behind individual dams has been estimated using standard approaches (including empirical observations of sediment transport and estimates of trap efficiency) for some projects prior to their construction. For the Manso dam, Carvalho and Lôu (1990) estimated a >95% trap efficiency, with time to complete infilling of ∼1,000 y. For the Itiquira dam, which has a much smaller and shallower reservoir, Carvalho et al. (2000), using Churchill's trap efficiency model, estimated a lifetime of only 8–12 years (the shorter estimate accounted for a likely increase in soil erosion over time). In fact, since its construction in 2000, the Itiquira reservoir has largely filled in with sediment, as is evident in the satellite imagery.

Although bedload is a small component of total sediment transport in rivers flowing to the Pantanal, it is particularly susceptible to retention behind dams. Over long time periods, and particularly in rivers that are presently largely undammed and carry higher bedload transport (e.g., the Taquari and Cuiabá), the accumulation of bedload can result in significant infilling above these dams. While periodic removal may alleviate infilling problems, if removal entails downstream discharge of high sediment loads there may be ecological risks that require careful consideration (Kondolf et al., 2014a).

Recommendations for Future Hydropower Development

Decisions about whether and where to construct dams in the Upper Paraguay River basin should consider not only sediment transport but also other environmental and social impacts. Two other kinds of impacts that are particularly important are nutrient transport (Oliveira et al., in press) and river system connectivity for migratory fish (Campos et al., 2020).

Based on these results as well as our parallel study on nutrient transport (Oliveira et al., in press) and a recent analysis of migratory fish routes in the region (Campos et al., 2020), development of new SHPs on the remaining undammed tributaries to the Pantanal may present serious risks to downstream river and floodplain ecosystems. Although the numerous dams in the northern part of the watershed have only modestly reduced sediment and nutrient transport because many are on rivers with low rates of transport, our modeling indicates that future construction of dams on rivers with higher sediment loads is likely to substantially reduce sediment transport from the uplands to the Pantanal. Substantial reductions in river sediment transport will likely lead to geomorphic changes that harm river and floodplain ecosystems and compromise the services they provide to people. Retention of sediments also reduces nutrient transport because particulate forms comprise major fractions of total nitrogen and phosphorus transport. Oliveira et al. (in press) showed that the new dams could reduce total phosphorus transport to the Pantanal by 29% from the current rate, with retention mainly by sediment trapping.

Considering potential reductions in transport of sediments as well as nutrients (Oliveira et al., in press), we recommend that new hydropower facilities should not be built on undammed rivers entering the Pantanal that have particularly low sediment and nutrient concentrations and transport, as well as on those that have the highest absolute rates of sediment and nutrient transport to the Pantanal.

Much of the nutrient load of these rivers is in particulate form and is thus associated with suspended sediments (Oliveira et al., in press). River systems with low nutrient concentrations are likely to be most sensitive to reductions in nutrient availability. In response to trapping of sediments and associated nutrients by damming, these rivers and their floodplains likely would experience oligotrophication, with negative consequences for fisheries yields and overall river and floodplain ecosystem productivity, as has been observed with dammed rivers elsewhere (Stockner et al., 2000). This dam-induced oligotrophication may eventually affect the fertility of pastures used for cattle (Forsberg et al., 2017). In the Upper Paraguay River basin, particularly low nutrient concentrations occur mainly in the Sepotuba, Correntes, and São Lourenço river systems, though the latter is already dammed in its lower reach.

River systems that carry the largest quantities of sediments to the Pantanal also deserve protection because their high sediment loads affect river channels and floodplains not only within the Pantanal but also downstream along the Paraguay River beyond the Pantanal (Oliveira et al., 2019). River systems of particular importance to the sediment and nutrient budget of the Pantanal that remain undammed, at least in their lower reaches, include the Cuiabá and Taquari (including its tributary the Coxim River).

CONCLUSION

This study, together with one on nutrient transport (Oliveira et al., in press), presents a comprehensive analysis of SHP impacts that is unprecedented in the literature and informs decisions about future hydropower development in the Upper Paraguay River basin and in similar settings elsewhere in Brazil and worldwide. Current facilities retain suspended and bedload sediment, and model predictions for hydropower facilities that may be built in the future on rivers flowing into the Pantanal point to large reductions in sediment transport, with potential negative consequences for downstream river and floodplain productivity. Negative impacts may be either because the rivers carry low sediment and nutrient concentrations, thereby being sensitive to oligotrophication, or are particularly important overall sediment and nutrient sources to the Pantanal, thereby supporting ecosystem productivity in downstream rivers and floodplains including particularly the Paraguay River axis within the Pantanal. Considering current and potential future effects on both sediment and nutrient transport, we recommend no additional hydropower development on the Cabaçal, Sepotuba, Cuiabá, and Taquari/Coxim rivers. Maintenance of the natural transport of sediments and nutrients from the uplands to downstream rivers and floodplains, as well as the connectivity between the floodplains and upland rivers for migratory fishes, would protect the productivity and biodiversity of the Pantanal.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

IF-C, MO, MS, and SH conceived and carried out the study. IF-C, MO, LS, and MC conducted the field work and data analysis. JC, OP, and RM developed the modeling. IF-C, MO, and SH wrote the manuscript. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2020.577748/full#supplementary-material

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Further Development of Small Hydropower Facilities May Alter Nutrient Transport to the Pantanal Wetland of Brazil

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Small hydropower (SHP) facilities, defined variably but usually by installed capacities of <10–50 MW, are proliferating around the world, particularly in tropical and subtropical regions. Compared to larger dams, SHPs are generally viewed as having less environmental impact, although there has been little research to support that assertion. Numerous SHPs have been built, and many more are in development or proposed, in rivers that drain into the Pantanal, a world-renowned floodplain wetland system located mostly in Brazil. The upland tributaries are important sources of nutrients to the Pantanal, affecting the biological productivity of downstream floodplains. This study presents measurements from upstream and downstream of 25 current hydropower facilities, most of which are SHPs, throughout the upland watersheds of the Upper Paraguay River basin to reveal how these facilities may affect the concentrations and transport of nutrients in rivers flowing to the Pantanal. Artificial neural network models estimated the impact of building 80 future SHPs on nutrient transport into the Pantanal, based on observations at current facilities as well as the spatial distribution of future facilities. Overall impacts of current hydropower facilities were not large, and in most cases were indistinguishable based on comparisons between upstream and downstream. The short water residence times of reservoirs associated with SHPs likely explain their tendency to have little or no effect on nutrient transport. However, model predictions for hydropower facilities that may be built in the future, many on rivers with higher discharge and sediment loads, point to significant reductions in overall TN (8%) and TP (29%) transport, with potential negative consequences for river and floodplain productivity. Negative impacts may be either because the rivers carry low nutrient concentrations and are thereby sensitive to oligotrophication, or they are particularly important overall nutrient sources supporting ecosystem productivity in downstream rivers and floodplains. Together with a parallel study of sediment transport, these results support recommendations that several river systems presently lacking dams in their lower reaches should be excluded from future hydropower development to maintain the nutrient and sediment supply to the Pantanal.

Keywords: hydroelectricity, dams, tropical, water quality, river transport

INTRODUCTION

Small hydropower (SHP) facilities are the most common kind of hydroelectric dams being built around the world, and although they are generally viewed as less environmentally harmful than larger dams, there has been little research to support that assertion, particularly in tropical and subtropical regions where the most new SHPs are being constructed (Mbaka and Mwaniki, 2015; Couto and Olden, 2018). Reflecting the widespread assumption that SHPs have lower environmental and social impacts than larger dams, many countries have enacted policies that promote SHPs, including less stringent environmental review. Brazil is an example, defining SHPs as facilities with installed electrical generation capacities between 5 and 30 MW (ANEEL, 2016). Multiple SHPs may be located in series along river systems, raising concerns about their cumulative effects on rivers and downstream ecosystems, as has been noted in China (Kibler and Tullos, 2013) and in the Amazon Basin (Athayde et al., 2019).

A large number of SHPs have recently been built in the watersheds of the Amazon, Paraná and Paraguay rivers of Brazil (Couto and Olden, 2018). Many more are in development or proposed, including in rivers that drain into the Pantanal, a world-renowned floodplain wetland system in the Upper Paraguay River basin, located mostly in Brazil (**Figure 1A**). The Pantanal occupies 140,000-km^2, most of which is subject to seasonal inundation for up to several months per year by either riverine overflow or delayed drainage of local rainfall or both (Hamilton et al., 1996). The upland tributaries transport nutrients into the Pantanal (Oliveira et al., 2019), thereby affecting the biological productivity of downstream floodplains. Aquatic primary productivity is often limited by either the availability of nitrogen or phosphorus, or co-limited by both nutrients (Guildford and Hecky, 2000), and in floodplains external nutrient supply determines overall ecosystem productivity. As a result, floodplain lands subject to flooding only by relatively nutrient-poor local rainfall tend to be markedly less productive than those inundated by nutrient-rich river water from the upland tributaries (Junk et al., 1989, 2011; Lewis et al., 2000; Hamilton, 2002; Güntzel et al., 2020).

Existing and proposed hydropower facilities in tributaries to the Pantanal are depicted in **Figure 1A**. As of 2018 there were 47 hydropower facilities in operation (hereafter "current hydropower facilities"), the majority of which are SHPs, with an additional 138 projects under construction, planned, proposed, or identified by the government as prospective sites (hereafter "future hydropower facilities") (Agência National de Águas [ANA], 2018). Many of the current and future projects are closely situated along river reaches, creating "cascades" where one project begins a short distance below the end of an upstream one.

Given the numerous SHPs planned or envisioned for development in the Upper Paraguay River basin, decision-makers urgently need to understand how these facilities on the tributaries may alter the transport of nutrients from the uplands into the Pantanal. The current study examines nutrient transport as one component of the basin-level environmental impacts of SHPs, and was carried out in conjunction with related studies on hydrology (Figueiredo et al., in review), sediment transport (Fantin-Cruz et al., 2020), and fish and fisheries.

Here we present measurements from above and below a number of current hydropower facilities throughout the Upper Paraguay River basin to reveal how these facilities may affect downstream water quality and, in turn, the transport of dissolved and particulate nutrients from the uplands to the Pantanal. In addition, we develop predictive models using artificial neural networks to estimate the impact of future hydropower development on nutrient transport into the Pantanal, based on observations at current facilities as well as the spatial distribution of future facilities. A companion paper in this journal (Fantin-Cruz et al., 2020) from the same project analyzes SHP effects on sediment transport to the Pantanal, and both papers conclude with recommendations developed from joint consideration of SHP impacts on nutrient and sediment transport. Our study design was based on the hypothesis that nutrient retention would be a function of water residence time above the dams, and that the sedimentation of particulate forms of nutrients would be the most readily observable effect, recognizing that many facilities may not slow the water enough to show these effects. Dams that produce longer water residence times would also be most likely to show biological retention (i.e., assimilation) or removal (e.g., denitrification) of nutrients.

STUDY SITE

This study examines rivers of the uplands in the Upper Paraguay River basin in Brazil that drain to the Pantanal wetland. The Pantanal lies mostly within Brazil, and drains southward via the Paraguay River. The uplands (150–1,400 m a.s.l.), which represent 59% of the basin area and lie mainly to the east and north of the Pantanal, include a lot of sloping terrain favoring rapid runoff and high sediment production. The Pantanal floodplains lie between 80 and 150 m a.s.l. According to the Köppen-Geiger climate classification, the climate of the region

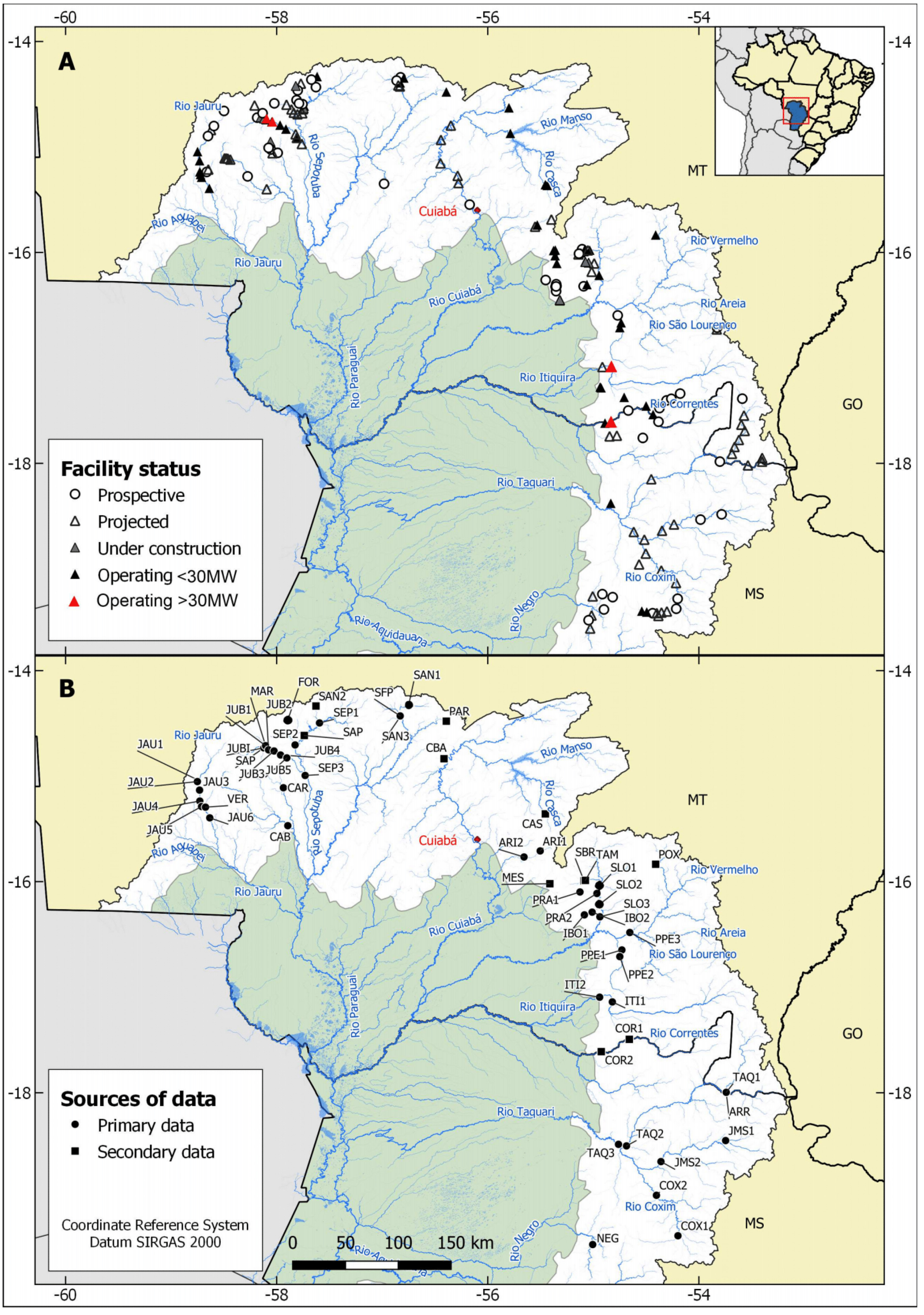

FIGURE 1 | (A) Hydropower facilities in the upland watershed of the Pantanal that are currently in operation as well as future projects that are under construction, planned, or identified as potential sites for hydropower development by either the Brazilian National Electric Energy Agency (ANEEL) or the state environmental agencies (depending on location). Red triangles indicate the four studied facilities with installed capacities > 30 MW. **(B)** Sampling sites for nutrient transport, including sampling conducted by the authors (primary data) as well as secondary data derived from environmental compliance reports submitted to state agencies (SEMA-MT and IMASUL) and from previous scientific studies. The Pantanal floodplains are shaded in green and rivers (*rios*) and other waterbodies are shown in blue. Sampling site acronyms are identified in **Table 1**.

is tropical savanna, with average annual precipitation in the uplands ranging from 1600 to 2100 mm. About 80% of the annual rainfall occurs in the rainy season from October to April (Gonçalves et al., 2011).

The native vegetation in the uplands is Cerrado savanna, but extensive areas are now converted to cropland (29% of the upland watershed area analyzed in this study) or pasture (22%). Human population density is low and Cuiabá city and its environs, situated along the Cuiabá River not far upstream of the Pantanal, is the largest urban area, which together with three other medium-sized cities located in the uplands has about 1,260,000 inhabitants. Water quality concerns in the region involve mostly diffuse pollution by soil erosion and agrochemicals from agricultural activities (Zeilhofer et al., 2006), as well as localized pollution by wastewater effluent from urban areas (Figueiredo et al., 2018).

The Pantanal is internationally recognized as a globally important wetland ecosystem that contains a rich mosaic of terrestrial, seasonally flooded, and aquatic habitats and landscapes. It is a Ramsar Site of International Importance under the Ramsar Convention and a UNESCO Biosphere Reserve. The region supports populations of several endangered mammals and birds including the hyacinth macaw (*Anodorhynchus hyacinthinus*), giant otter (*Pteronura brasiliensis*), jaguar (*Panthera onca*), pampas deer (*Ozotoceros bezoarticus)*, and marsh deer (*Blastocerus dichotomus*) (Tomás et al., 2019). Cattle ranching, subsistence and recreational fishing, and ecotourism are major economic activities within the Pantanal.

MATERIALS AND METHODS

Hydropower Facilities

The characteristics of the current hydropower facilities studied here, as well as the river reaches in which they occur, are given in **Supplementary Table S1**. The information includes facility name, river system, geographic coordinates, mean annual discharge, watershed area, reservoir area and volume, dam height, hydraulic residence time, installed potential, facility design (run-of-river vs. conventional and length of diverted river reach where applicable), and time since construction.

Most of these facilities can be considered small, although six have installed capacities above the Brazilian government's regulatory definition of SHP as <30 MW installed capacity, and three of those exceed 100 MW. Two of those that exceed 30 MW (Juba I and II, each 42 MW) have dams and reservoirs similar in size to the SHPs, and one of the SHPs (São Lourenço, 29 MW) creates a reservoir comparable in size to larger facilities such as the largest one studied here, Ponte de Pedra (176 MW). Thus, because installed capacity is an imperfect indicator of the degree to which the passage of river water is slowed, which in turn determines the potential effects of these facilities on nutrient transport, we analyze the SHPs and larger facilities together in this study.

Most of the hydropower facilities studied here are diversion designs, where a low dam with a small or non-existent reservoir diverts river water into an artificial channel (headrace) for as much as several km to a powerhouse farther down the river valley (**Supplementary Table S1**). Most of the river discharge is normally diverted, leaving the natural channel with as little as 10% of the discharge, then returned to the river below the powerhouse with no net loss or gain. The designs that lack a large reservoir are "run-of-river" facilities inasmuch as they cannot alter discharge except on short time scales (Csiki and Rhoades, 2010; Kaunda et al., 2012; Figueiredo et al., in review). Many of the SHPs are located on lower-order rivers but some are on larger rivers with low elevational gradients.

Nutrient concentrations and other water quality data (described in "Sample Collection and Analysis" for upstream and downstream of current SHPs and several larger hydropower facilities), as well as in reaches where such facilities may be built in the future, were obtained from our own sampling and measurements (primary data) as well as from reports submitted by hydropower companies to the state environmental agencies as required for environmental compliance (secondary data). The nutrient and discharge measurements conducted for environmental compliance followed the same field and laboratory methods we used, and analyses were conducted only by certified laboratories with appropriate quality assurance protocols.

From the 30 current hydropower facilities with available data (**Supplementary Table S1**), 25 had sufficient numbers of sampling dates to compare upstream to downstream nutrient concentrations and transport (17 based on primary data and eight based on secondary data; criteria for selection are described in section "Water Quality Data Compilation From Secondary Sources"). Data for the other five SHPs were included in the model development (section "Prediction of Impacts of Future Hydropower Facilities"). The spatial distribution of sampling sites with primary or secondary data is shown in **Figure 1B**, and sampling site codes used in figures are listed in **Table 1**. Spatial coordinates of sampling sites as well as their locations in river networks appear in **Supplementary Table S2**.

Sample Collection and Analysis

Primary data on discharge, water quality including dissolved and suspended matter, and particulate nutrients in bedload transport above and below current hydropower facilities were collected on 13 dates spanning the wet and dry seasons from October 2018-May 2019 (some locations had fewer collection dates). The primary data set contains water quality data for upstream and downstream of 17 hydropower facilities (**Supplementary Table S2**). In addition, on 6–13 dates we sampled a number of rivers at locations close to where SHPs may be constructed in the future. At each sampling location we recorded the bathymetric profile of the channel cross-section and installed a staff gage unless one already existed there (a number of gages are maintained by hydropower companies). Discharge was measured across the channel profile on each sampling date using an acoustic Doppler current profiler (*SonTek RiverSurveyor-M9*) following the methods outlined in Agência National de Águas [ANA] (2019a,b). For nine rivers where discharge could not be measured—the Paraguai, Casca, Mestre, Saia Branca, Tenente Amaral, Caeté, Gloria, and Poxoréo rivers—a hydrological model provided estimates (Collischonn et al., 2019).

TABLE 1 | List of sampling sites and hydropower facilities they pertain to, with codes for figures and tables.

Watershed	Tributary	Sampling site	Code	Hydropower facility names	
				Current	Future
Paraguay	Paraguay	Paraguay River, upstream and downstream of SHP Alto Paraguay	PAR	Alto Paraguai	
	Santana	Santana River, upstream SHP Diamante	SAN1	Santana I	
		Santana River at mouth	SAN2		Santana II
		Santana River, downstream of SHP Santana I	SAN3	Santana I	
	S. F. Paula	São Francisco de Paula River, downstream of proposed SHPs	SFP		Salra, Jaçanã Alta, Biguá
Sepotuba	Maracanã	Maracanã River at mouth	MAR		Taquarinha, Medianeira
	Sapo	Sapo River, ∼13 km upstream of SHP rio Sapo	SAP		Lagoa Grande, Ponte Estreita
		Sapo River, upstream and downstream of SHP rio Sapo	SAP	Rio do Sapo	
	Formoso	Formoso River, ∼250 m upstream of mouth	FOR		Formoso I, II e III
	Jubinha	Jubinha River, upstream of UHE Juba I	JUBI	Juba I	Jubinha I, II e III
	Juba	Juba River, upstream of UHE Juba I	JUB1	Juba I	Juba III e IV
		Juba River, downstream of UHE Juba I	JUB2	Juba I	
		Juba River, downstream of UHE Juba II	JUB3	Juba II	
		Juba River, downstream of SHP Graça Brennand	JUB4	Graça Brennand	
		Juba River, downstream of SHP Pampeana	JUB5	Pampeana	Corredeira, Tapirapuã
	Sepotuba	Sepotuba River, downstream of Maracanã River	SEP1		Salto das Nuvens, Sepotuba
		Sepotuba River, downstream of Formoso River	SEP2		Paiaguás, Salto Maciel
		Sepotuba River lower mainstem, downstream Juba River	SEP3		
Cabaçal	Cabaçal	Cabaçal River lower mainstem, downstream of proposed SHPs	CAB		Cabaçal 1,2,3,4,5 and 6
	Caramujo	Caramujo River, downstream of proposed SHPs	CAR		Salto do Céu, Salto Cacau, Salto Vermelho I, Salto Caramujo
Jauru	Jauru	Jauru River, upstream of SHP Antonio Brennand	JAU1	Antonio Brennand	Estivadinho III, Alagados III, Tra irão III
		Jauru River, downstream of SHP Antonio Brennand	JAU2	Antonio Brennand	
		Jauru River, downstream of SHP Ombreiras	JAU3	Ombreiras	
		Jauru River, downstream of UHE Jauru	JAU5	Jauru	
		Jauru River, downstream of SHP Salto	JAU4	Indiavai + Salto	
		Jauru River, downstream of SHP Figueirópolis	JAU6	Figueirópolis	
	Vermelho	Vermelho River at mouth, downstream of proposed SHPs	VER		Rancho Grande, Progresso
Cuiaba	Casca	Casca River, upstream and downstream of UHE Casca II and SHP Casca III	CAS	Casca II and III	
	Mestre	Mestre River, upstream SHP Mestre and downstream SHP Santa Cecilia	MES	Mestre + Santa Cecília	
	Cuiabá	Cuiabá River lower mainstem at Passagem da Conceição hydrological station	CBA		Perudá, Angatu II, Angatu I, Iratambé I, Iratambé II, Guapira
	Aricá	Aricá River, upstream of SHP São Tadeu I	ARI1	São Tadeu I	Aricá-Mirim I
		Aricá River at mouth, downstream of SHP São Tadeu I	ARI2		São Tadeu II
São Lourenço	Tenente Amaral	Saia Branca River upstream and douwnstream SHP Sucupira	SBR	Pequi	
		Tenente Amaral River, ∼10 km above mouth	TAM	Sucupia	Ipê, Mangaba
	Prata	Prata River, upstream of SHP Água Prata	PRA1	Água Prata	
		Prata River, downstream of SHP Água Prata	PRA2		Água Clara, Água Branca, Água Brava
	São Lourenço	São Lourenço River, upstream of SHP São Lourenço	SLOI	São Lourenço	

(Continued)

TABLE 1 | Continued

Watershed	Tributary	Sampling site	Code	Hydropower facility names	
				Current	Future
		São Lourenço River, downstream of SHP São Lourenço	SL02		
		São Lourenço River lower mainstem, down stream of SHP São Lourenço	SL03		
	Ibo	Ibo River, upstream of SHP Sete Quedas Altas	IBOI	Sete Quedas Altas	Europa
		Ibo River, downstream of SHP Sete Quedas Altas	IB02		
	Poxoréu	Poxoréu River, upstream and downstream of SHP Poxoréu	POX	Poxoréu	
	Ponte de Pedra	Ponte de Pedra River, upstream of SHP Eng. José Gelázio	PPE1	Eng. José Gelázio	
		Ponte de Pedra River, downstream of SHP Eng. José Gelázio	PPE2		
		Ponte de Pedra River, downstream of SHP Rondonópolis	PPE3	Rondonópolis	João Basso
Piquiri	Itiquira	Itiquira River, upstream of UHE Itiquira	ITU	UHE Itiquira	
		Itiquira River lower mainstem, downstream of UHE Itiquira	ITI2		Itiquira III
	Correntes	Correntes River, upstream of UHE Ponte de Pedra	COR1	UHE Ponte de Pedra	Água Enterrada, Santa Paula
		Correntes River lower mainstem, downstream of UHE Ponte de Pedra	COR2		
Taquari	Ariranha	Ariranha River at mouth, downstream of proposed SHPs	ARR		Girassol, Dália, Lírio, Violeta, Orquídea, Primavera, Hortência
	Jauru	Jauru River at BR 359 bridge, upstream of proposed SHPs	JMS1		Jauruzinho, Barra do Piraputanga, Água Fria
		Jauru River, upstream of Coxim River and downstream of proposed SHPs	JMS2		Figueirão, Vila Jauru, Mundo Novo
	Coxim	Coxim River at Fazenda São José, upstream of Camapuã River	COX1		Entre Rios, Lagoa Alta, Ponte Vermelha, Calcutá, Maringá, Fazenda Caranda, Peralta, Água Vermelha
		Coxim River at MS-142 bridge, downstream of Jauru River	COX 2		São Domingos, Sucuri
	Taquari	Upper Taquari River, upstream of Ariranha River	TAQ1		Taquarizinho, Barra do Ariranha
		Upper Taquari River at Silviolandia city	TAQ2		Pedro Gomes
		Taquari River below confluence with Coxim River	TAQ3		
Negro	Negro	Negro River, at Negro city	NEG		Rio Negro, Ouro Negro, São Francisco de Assis

In most cases "lower mainstem" refers to the river before its entry into the Pantanal. SHP, small hydropower facility.

We collected depth- and flow-integrated water samples by the equal-discharge-increment method. Depth-integrated samples of the water column at each point were obtained with either a DH48 or DH50 integrating sampler depending on hydraulic conditions. Samples from each point were composited in a mixing bucket in volumetric proportion to the discharge contribution of each point, as determined from the profiler data using custom software from the Brazilian National Water Agency (Hidrosedimentos 2.0: Agência National de Águas [ANA], 2019a,b).

We analyzed the water samples for particulate organic carbon (POC), total phosphorus (TP), total nitrogen (TN), soluble reactive phosphorus (SRP), nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+), and specific conductance (corrected to 25°C), pH and turbidity. The sum of nitrate, nitrite, and ammonium is presented here as dissolved inorganic N (DIN). Chemical analyses were conducted at EMBRAPA Pantanal following (1) membrane-suppression ion chromatography for the major ions listed above, NO_3^-, NO_2^- and NH_4^+; (2) flow injection analysis with standard colorimetric methods for SRP, total N and total P (Wetzel and Likens, 2000); and (3) high-temperature combustion in an Elemental Analyzer for C in particulate matter collected on filters. Specific conductance, pH and turbidity were measured in the field during the sampling using a YSI sonde and a turbidimeter, both calibrated following manufacturers' instructions.

Transported bedload material was collected using a Helley-Smith sampler (Carvalho, 2008). In this paper the bedload transport and its nutrient content are used in the calculation of nutrient fluxes. Bedload samples consisted of heterogeneous mixtures of fine inorganic sediment and coarse particulate organic matter. After drying at 105°C for 24 h, the latter fraction was removed by sieving, ground, and mixed back into the samples before subsampling for analysis of C and N content by high-temperature combustion in an Elemental Analyzer and of P content by extraction in hot acid (Andersen, 1975) followed by colorimetric analysis of SRP. More detail on sediment sampling and how hydropower facilities affect the balance between suspended and bedload sediment inflows and outflows is given in Fantin-Cruz et al. (2020).

Water Quality Data Compilation From Secondary Sources

The secondary data set contains water quality data for an additional 22 hydropower facilities, including 14 SHPs that we also sampled, and 8 that have only secondary data. Secondary data were used mainly in the modeling described later, although secondary data from upstream and downstream of 8 SHPs not sampled in this study are included in the graphical analyses together with a larger number of reaches with primary data. To assemble the secondary data set, we incorporated data on rivers that were available from reports submitted by hydropower companies to the environmental agency of the State of Mato Grosso (SEMA-MT). Out of about 3,000 water quality samplings in the secondary data set, we selected 401 and 452 samplings with TN and TP measurements that could be used mainly for the predictive model. Selection criteria included measurements of multiple water quality variables, close correspondence in timing of the upstream and downstream samplings, and at least five dates of sampling. For cases where more than one location was sampled to represent upstream or downstream water quality, the mean was taken, weighted by the relative discharge in the case of more than one tributary coming together above a facility. Reported values that were below the detection limit of the analyses were substituted with the detection limit concentrations. Further screening of the selected secondary data for quality assurance included:

1. Deletion of extreme outliers that became apparent based on comparison with existing published data of the range of chemical characteristics of rivers in the region (Hamilton et al., 1997; Oliveira et al., 2019) and state environmental agency reports.
2. Deletion of ion concentrations that were not commensurate with conductance measurements (this was the case for a few very high concentrations of nitrate and ammonium).
3. Deletion of nutrient concentrations in cases where total dissolved concentrations exceeded total (dissolved plus particulate) concentrations.

Data Analysis

The effects of current hydropower facilities on TN, TP, and POC concentrations and transport were evaluated by comparing median concentrations upstream and downstream of each facility, based on a combination of primary data (N = 13 dates in most cases) and secondary data (variable numbers of sampling dates). Transport was calculated as the median concentration times discharge, averaging the discharge estimates above and below each SHP location to avoid potential spurious results caused by the uncertainty inherent in discharge measurements as well as by short-term (sub-daily) fluctuations imposed by the hydropower facilities (hydropeaking: Figueiredo et al., in review). Observed ratios of upstream to downstream concentrations were grouped into classes based on the percentage change in either direction (i.e., retention or release), similar to the sustainability boundary approach suggested by Richter et al. (2012) for analysis of flow regime alterations in river systems that lack detailed knowledge of the impacts of altered flows. Ratios of $<10\%$ were defined as undetectable changes, 10–20% as moderate, and $>20\%$ as high alterations. In addition to comparing the median concentrations and transport rates for all sampling dates combined for each study reach, we conducted statistical analyses of the changes in concentrations observed across all individual sampling dates in each reach using a one-sample t-test for which the null hypothesis was zero change. Analyses were conducted after $\log_{10}$ transformation of the concentration changes to improve normality.

Prediction of Impacts of Future Hydropower Facilities

Artificial neural networks (ANNs) were developed to predict the impacts of future hydropower facilities on TN and TP (this study) as well as suspended sediment concentrations (Fantin-Cruz et al., 2020). As one of the most commonly used artificial intelligence tools, ANNs are well suited to model phenomena subject to controls that are complex, incompletely understood, and likely non-linear (American Society of Civil Engineers [ASCE], 2000). The ANN model architecture was a three-layer feedforward network with a non-linear (unipolar sigmoid) activation function (**Supplementary Figure S1**), similar to ANNs that have been applied to study floodplain inundation in the Pantanal (Fantin-Cruz et al., 2011) as well as elsewhere (Dawson et al., 2006). The ANN models were trained with a data set representing 30 reaches containing SHPs (including primary and secondary data). The back-propagative algorithm (Rumelhart et al., 1986), along with training acceleration techniques (Vogl et al., 1988) as well as other needed functions, were custom programmed in the Matlab R2012b environment. Overfitting was avoided using the cross-validation technique (Hecht-Nielsen, 1989).

For cross-validation training, the data were divided into three samples (approximately 2/3 for training, 1/3 for validation, and 1/3 for verification), using a systematic sampling method to provide a representative distribution of the 30 locations for all samples. The extreme values (maximum and minimum) of all variables were included in the training samples and all input data of the future hydropower facilities were within the domain of the trained ANN, ensuring that model predictions were within the ranges of the training data.

The complexity of the ANNs (number of neurons in the intermediate layer) was defined by the architecture search with the lowest possible complexity that still had the same approximation and generalization capacity of a purposely oversized ANN that was trained without overfitting. These approximation and generalization capacities were verified by the performance of the application to the validation sample, since the verification sample, by definition, cannot participate, neither in training nor in the definition of the ANN architecture (Hecht-Nielsen, 1989).

In the present study, input variables that were considered for the ANNs included contributing watershed area, reservoir area and volume, soil classification (11 classes in the contributing watersheds: Empresa Brasileira de Pesquisa Agropecuária [EMBRAPA], 2018), land use and cover (8 classes: Empresa Brasileira de Pesquisa Agropecuária [EMBRAPA], 2015), upstream concentrations of TN or TP, and the potential of contributing watersheds to yield nitrogen and phosphorus to rivers. Sufficient information was available for 80 of the 138 potential future hydropower facilities.

The potential watershed yields of organic N and P to river systems were estimated from the Soil and Water Assessment Tool (SWAT), the full results of which are posted online (Mingoti et al., 2020). Details of this SWAT model are presented by Neitsch et al. (2011) and Arnold et al. (2012). The SWAT model was run for each watershed for the year 2017 using spatial data on climate, topography, soils, and land use and cover. Annual and monthly inputs were estimated for the contributing watersheds above each current and future hydropower facility. Watershed yields of organic N and P at each hydropower facility showed considerable spatial variability, ranging from 0.02–5.6 and 0.02–0.93 t ha^{-1} y^{-1}, respectively. Watershed yields of N and P were highest in the Taquari, Coxim, and Jauru-MS watersheds, followed by the Correntes, and were relatively low in the northern watersheds.

We developed independent ANN models to predict changes in the concentrations of TN, TP and suspended sediments (the latter presented in Fantin-Cruz et al., 2020). The ANN models were trained with concentrations of TN (577 records) and TP (622 records) measured upstream and downstream of 30 current hydropower facilities. Pearson linear correlations between input and output variables indicated the best predictive variables for each model. Overall performance of the ANNs was evaluated by the Nash-Sutcliffe model efficiency coefficient.

RESULTS

River Discharge and Chemistry

Discharge and water chemistry were variable among the river systems. The rivers with current and future hydropower facilities that are analyzed in this study range in discharge, based on means for the available periods of record, from 0.7 to 230 m^3 s^{-1}, with the majority (~50%) lower than 25 m^3 s^{-1} (**Supplementary Tables S3**, **S4**). The rivers with the highest discharges include the lower mainstems of the Taquari, Cuiabá and Sepotuba rivers, close to their entry into the Pantanal. The tributaries with the lowest discharges include the Upper Paraguay, Santana, Maracanã and Sapo rivers in the Paraguay River system, the Mestre river in the Cuiabá River system, the Tenente Amaral, Prata and Ibó rivers in the São Lourenço River system, and the Negro River.

Among the 25 current hydropower facilities that we studied, 14 are on rivers with discharges below 50 m^3 s^{-1}, and the remaining 11 are on rivers with discharges between 50 and 100 m^3 s^{-1} (**Supplementary Table S4**). Among the 80 future hydropower facilities that we modeled in this study, all of which would be SHPs (**Figure 1A** and **Supplementary Table S5**), the majority (52 SHPs, or 65%) would be on rivers with discharges below 25 m^3 s^{-1}, with 17 on rivers between 25 to 100 m^3 s^{-1}, and 11 on rivers > 100 m^3 s^{-1}. The reaches with proposed SHPs on the Taquari, Cuiabá, and Sepotuba rivers include the highest discharges among the rivers studied here.

Most of the rivers in the Upper Paraguay River basin carry water that is slightly to moderately acidic and low in dissolved ions, as indicated by pH values ranging from <5–7 and specific conductance values < 20 μS cm^{-1} (**Supplementary Table S3**). Rivers of higher ionic strength include the Jauru in Mato Grosso State (Jauru-MT), Cabaçal, Cuiabá, Miranda, and Aquidauana rivers, with pH values in the range of 7–8 and conductance in the range of 50–200 μS cm^{-1}.

Nutrient Concentrations and Ratios

Concentrations of TN and TP were correlated in rivers draining to the Pantanal (**Figure 2** and **Supplementary Table S3**). TN concentrations were similar between the dry and wet seasons, but TP concentrations tended to be higher in the wet season (**Supplementary Figure S2**). The Taquari River system, including the Taquari, Jauru-MS, and Coxim rivers, tended to have the highest concentrations of both TN and TP in both the wet and dry seasons. The ratios of TN:TP indicate that aquatic primary production in downstream waters can variably be either N or P limited or co-limited by N and P. Molar ratios of TN:TP in the range of 20–50 (equivalent to mass ratios of 9.0–22.6 and depicted as dotted lines in **Figure 2**) are indicative of potential N and P co-limitation (Guildford and Hecky, 2000). The TN:TP ratios for some rivers were higher in the dry season, causing them to shift from likely P limitation or N and P co-limitation in the wet season toward likely co-limitation or limitation by N in the dry season.

Riverine Transport of Nutrients to the Pantanal

Transport of both nitrogen and phosphorus by rivers from the upland watersheds to the Pantanal is dominated by suspended particulate forms, which compose nearly all of the TP and often the majority of the TN (**Figure 3**). Rivers with high rates of transport of TN and TP tend to also have high rates of POC transport, reflecting their higher loads of suspended particulate organic matter. Nutrient transport in the water column is a function of both discharge rates and concentrations,

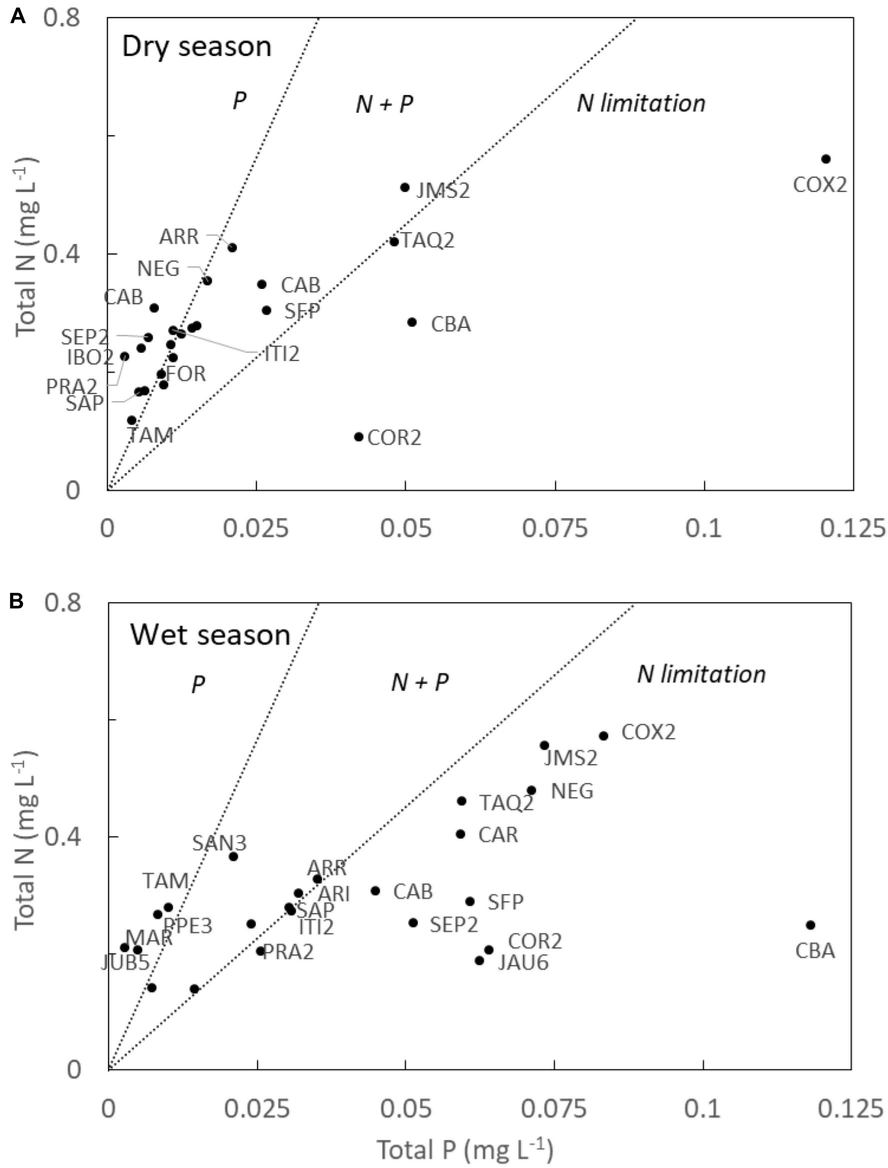

FIGURE 2 | Concentrations of TN and TP in rivers draining to the Pantanal with dashed lines delimiting the zone of likely co-limitation by N and P during the dry season **(A)** and wet season **(B)**. In reaches containing hydropower facilities (13 of the 24 locations), the data shown here are downstream of the facilities. Observations were divided into seasons based on mean monthly rainfall, with May-October and November-April as the dry and wet seasons, respectively. Codes indicate the sampling points detailed in **Table 1**. The figure includes both primary and secondary data and reaches with current hydropower facilities as well as reaches targeted for future facilities.

with discharge exerting primary control, most strongly for TN (**Figure 4**). The most important rivers bringing N and P to the Pantanal among those studied here are thus the largest ones in terms of discharge: the Cuiabá, Taquari, Sepotuba, São Lourenço, and Correntes rivers. Bedload transport of particulate N, P, and C was small (<1% in most cases) relative to transport in the water column (**Figure 5**). Rivers with the highest absolute rates of bedload transport of TN, TP and POC include the Taquari, Itiquira, São Lourenço, and Ariranha, as well as the Formoso in the case of TN and POC but not TP.

Effects of Current Hydropower Facilities on Nutrient Transport

Concentrations and transport of TN, DIN, TP, and POC in the water column upstream and downstream of 25 current hydropower facilities are compared in **Figure 6** and **Supplementary Table S4**. The changes between upstream and downstream are presented as both concentrations, which bear on the productivity of downstream waters, and rates of transport (i.e., kg d^{-1}), which bear on the overall nutrient supply from the upland watersheds to the Pantanal. For a given project, the ratios between concentrations and

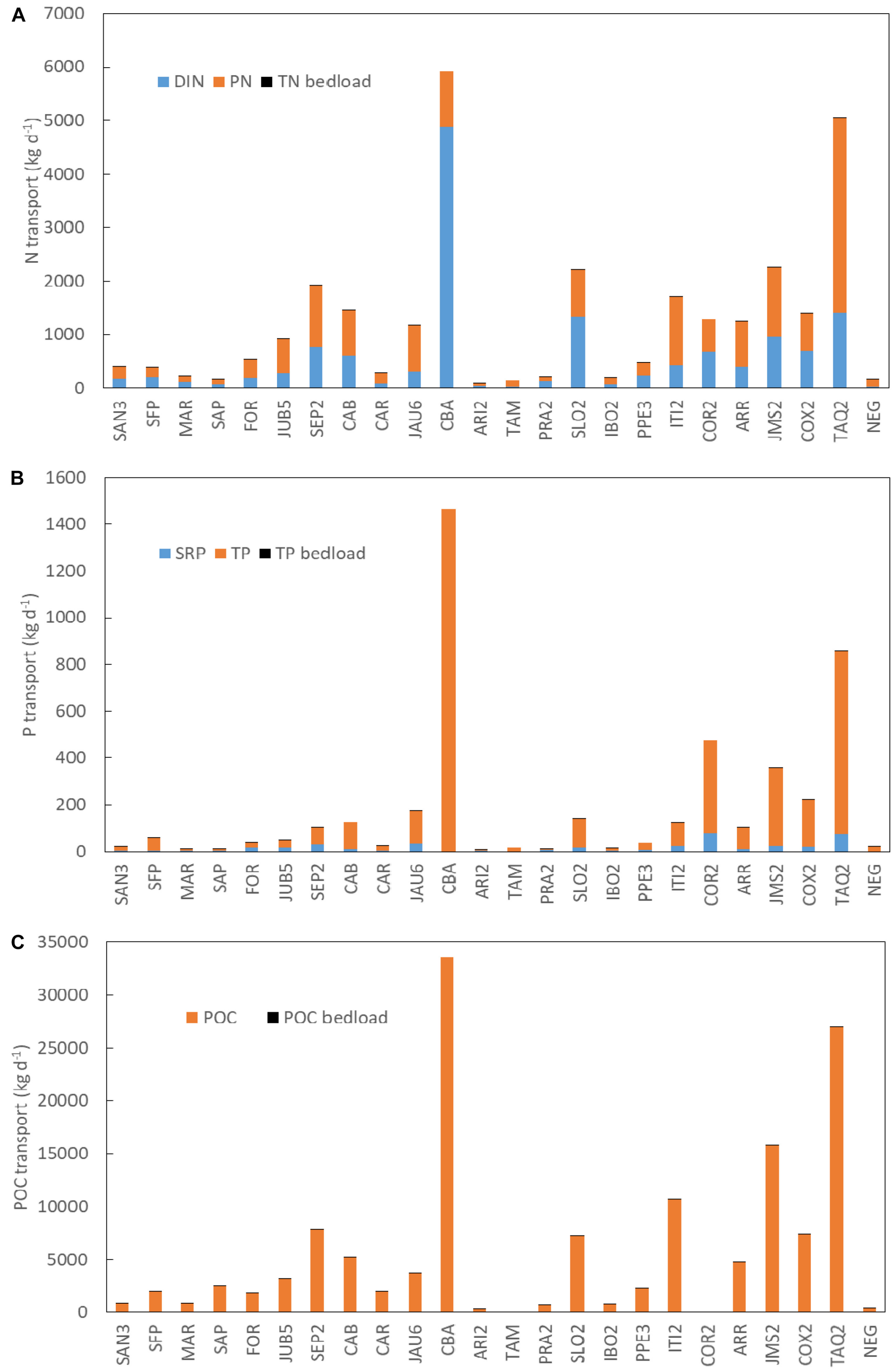

FIGURE 3 | Riverine transport of TN **(A)**, TP **(B)**, and POC **(C)**, with bars divided into dissolved, suspended, and bedload fractions, and of suspended and bedload POC. DIN is the sum of nitrate and ammonium. Bedload estimates, which are hardly large enough to be visible where they exist, were not made for sites CAB, TAM, PPE3, and COR2. Codes indicate the sampling points detailed in **Table 1**. The figure includes both primary and secondary data and reaches with current hydropower facilities as well as reaches targeted for future facilities.

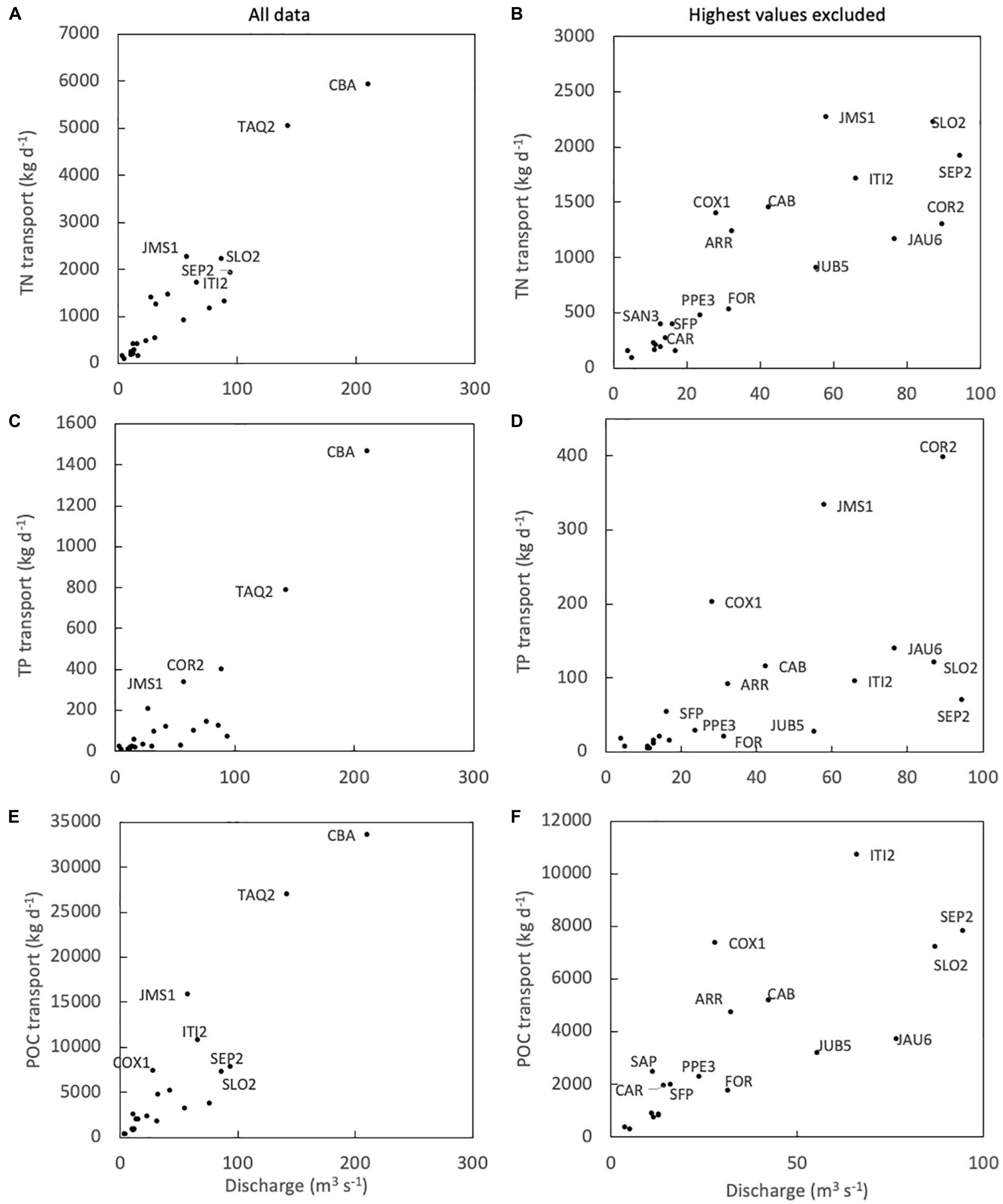

FIGURE 4 | Water-column transport of total nitrogen (TN), total phosphorus (TP), and particulate organic carbon (POC) in relation to river discharge. **(A)** TN, all data; **(B)** TN, rivers with discharge < 100 m^3 s^{-1}; **(C)** TP, all data; **(D)** TP, rivers with discharge < 100 m^3 s^{-1}; **(E)** POC, all data; **(F)** POC, rivers with discharge < 100 m^3 s^{-1}. Codes indicate the sampling points detailed in **Table 1**. The figure includes both primary and secondary data and reaches with current hydropower facilities as well as reaches targeted for future facilities.

transport covary because we assumed that discharge did not change, and hence we used the mean of upstream and downstream discharge measurements to calculate transport. We do not consider bedload nutrient transport here because it was almost always a small proportion of total transport (**Figure 3**).

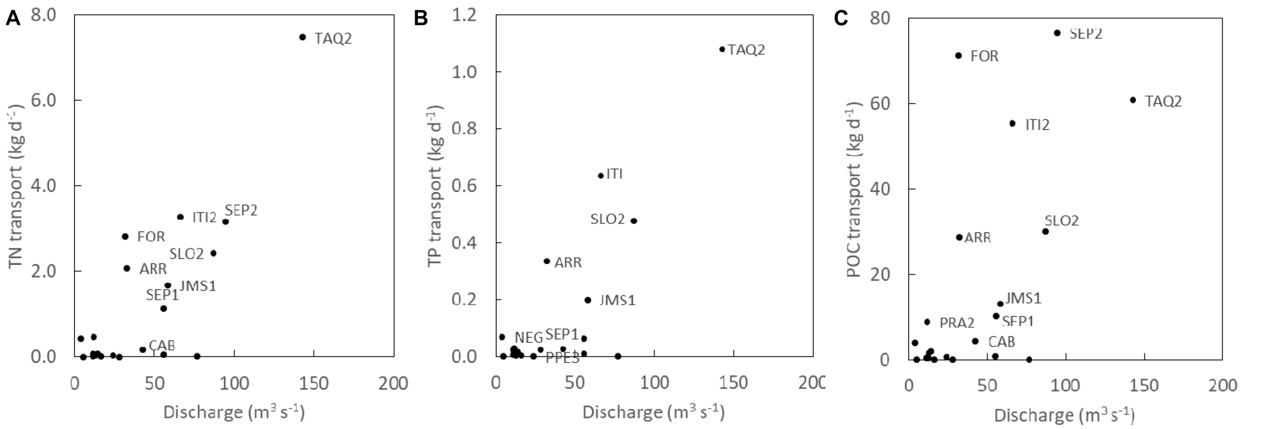

FIGURE 5 | Bedload transport of **(A)** total nitrogen (TN), **(B)** total phosphorus (TP), and **(C)** total carbon (TC) in relation to river discharge. Codes indicate the sampling points detailed in **Table 1**. The figure includes primary data for reaches with current hydropower facilities as well as reaches targeted for future facilities.

Comparison of median values shows that there is no consistent trend for either nutrient release or retention in reaches containing current hydropower facilities, with ratios falling approximately equally on either side of the 1:1 line for TN, DIN, TP, and POC (**Figure 6**). Some reaches show upstream: downstream ratios for TN and TP that deviate considerably (>20%) from the 1:1 line, but with no correspondence in which facilities deviate between the two variables. Seven of 25 hydropower facilities deviated from 1:1 by >20% for TN, 9 of 17 deviated by >20% for DIN, 17 of 25 deviated by >20% for TP, and 3 of 17 cases deviated by >20% for POC. The reaches with the largest deviations in concentrations of TN and TP from the 1:1 line are mainly facilities on smaller river systems (Santana, Sapo, Casca, Saia Branca and Mestre rivers). The greatest reduction in TP concentration and transport was observed where the Correntes River flows through the Ponte de Pedra reservoir (COR2), as well as the PCH São Lourenço on the river of the same name; these are two of the largest reservoirs included in this study.

Statistical analysis of the concentration changes across all individual sampling dates for each hydropower facility using one-sample t-tests showed significant ($p < 0.05$) differences from zero for a minority of reaches (marked with green in **Figure 6**), reflecting the high variability in the results. Differences were significant for 5 of 25 reaches for TN (20%), 2 of 17 for DIN (12%), 5 of 25 for TP (20%), and 1 of 17 (6%) for POC. The percentage of significant results for TN and TP is much larger than could be expected by chance alone. The reaches with significant changes in TN and/or TP across all sampling dates are on the Jauru, Juba, Correntes, Santana, and Mestre river systems.

Predicting the Impacts of Future Hydropower Development

Among the potential ANN input variables, by far the most significant predictor of TN and TP retention were the measured upstream concentrations, accounting for 74 and 57% of the predictive capability of the models, respectively (**Supplementary Figure S3**). Less important but still significant were watershed nutrient yields from the SWAT model, land use (n = 10 classes), watershed area, and reservoir area and volume. Soil classes were not significant predictors for TN or TP. The performance of the ANNs was satisfactory, as indicated by the Nash-Sutcliffe model efficiency coefficients. The coefficients for the TN model were 0.847 for training and 0.823 for verification, and for the TP model they were 0.712 and 0.606, respectively.

Predicted impacts of future hydropower facilities on TN concentrations and transport showed a diversity of effects ranging from considerable retention to little effect to considerable release (**Figure 7**). The greatest TN retention in terms of both concentrations and transport is predicted for the multiple future SHPs on the Coxim and Taquari rivers, which are presently undammed and carry relatively high TN concentrations and loads. In contrast, future SHPs on the Jauru-MS (coded as JMS2), Juba, and Itiquira rivers, which carry TN concentrations on the low end of the range, are predicted to release more TN than they retain.

In contrast to TN, for which future SHPs may either cause net retention or release, model predictions for TP show either net retention, which is often considerable, or neutral effects of SHPs (**Figure 7**). The greatest decreases in TP concentrations are predicted for SHPs on the Taquari river system, which carries relatively high TP concentrations and loads of suspended material. The predicted retention of TP by the multiple future SHPs on the Cuiabá River above the city of Cuiabá is particularly large.

The measured current and modeled future rates of transport of TN and TP by the major rivers flowing from the upland watershed into the Pantanal, based on the most downstream sampling points, are summarized in **Figure 8**. Available data for a few smaller rivers are not shown (Negro, Aricá Mirim, and Ribeirão Ponte de Pedra rivers), and the Taquari River is divided into its two major tributaries (Upper Taquari and Coxim rivers) whose confluence is a short distance upstream of the border of the Pantanal. As is apparent in **Figure 7**, TN is predicted to decrease in some rivers and increase in others as a result of future hydropower development, whereas TP tends to decrease,

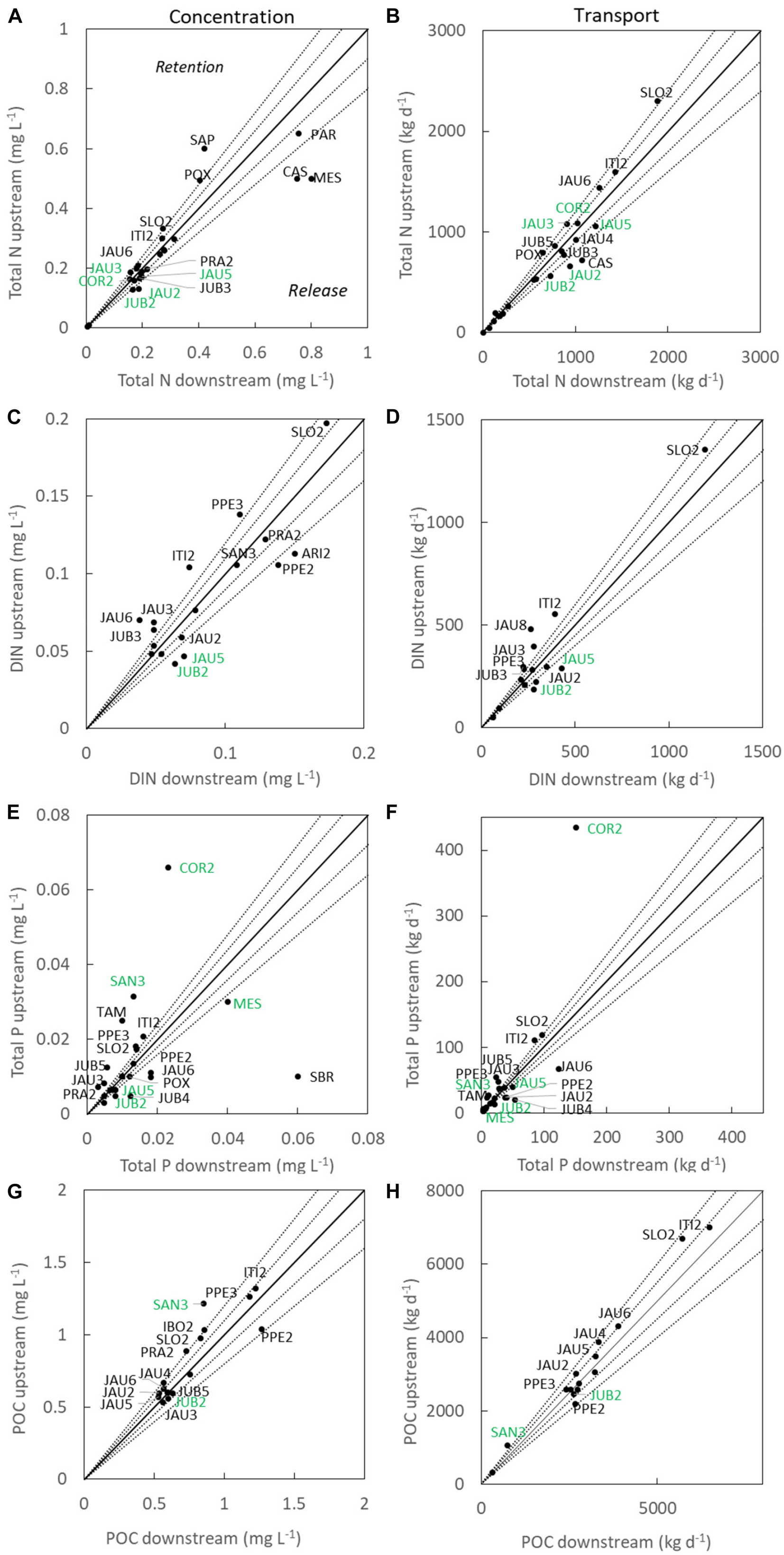

FIGURE 6 | Continued

FIGURE 6 | Comparisons of changes in concentrations and transport between upstream and downstream of current hydroelectric facilities based on primary data collected in this study as well as secondary data. **(A,B)** total nitrogen (TN) concentrations and transport, **(C,D)** dissolved inorganic N concentrations and transport, **(E,F)** total phosphorus (TP) concentrations and transport, and **(G,H)** particulate organic carbon (POC) concentrations and transport. Solid line shows the line of parity and dashed lines show bounds of ±10 and ±20% around that line; points above the line indicate net retention and those below indicate net release between the upstream and downstream sampling points. Upstream:downstream ratios that deviate considerably from 1:1 are identified with the codes shown in **Table 1**; codes in green font indicate cases where the statistical analysis of the concentration changes across all individual sampling dates showed significant ($p < 0.05$) differences from zero. Both primary and secondary data are included in the figure, and all data are in **Supplementary Table S4**.

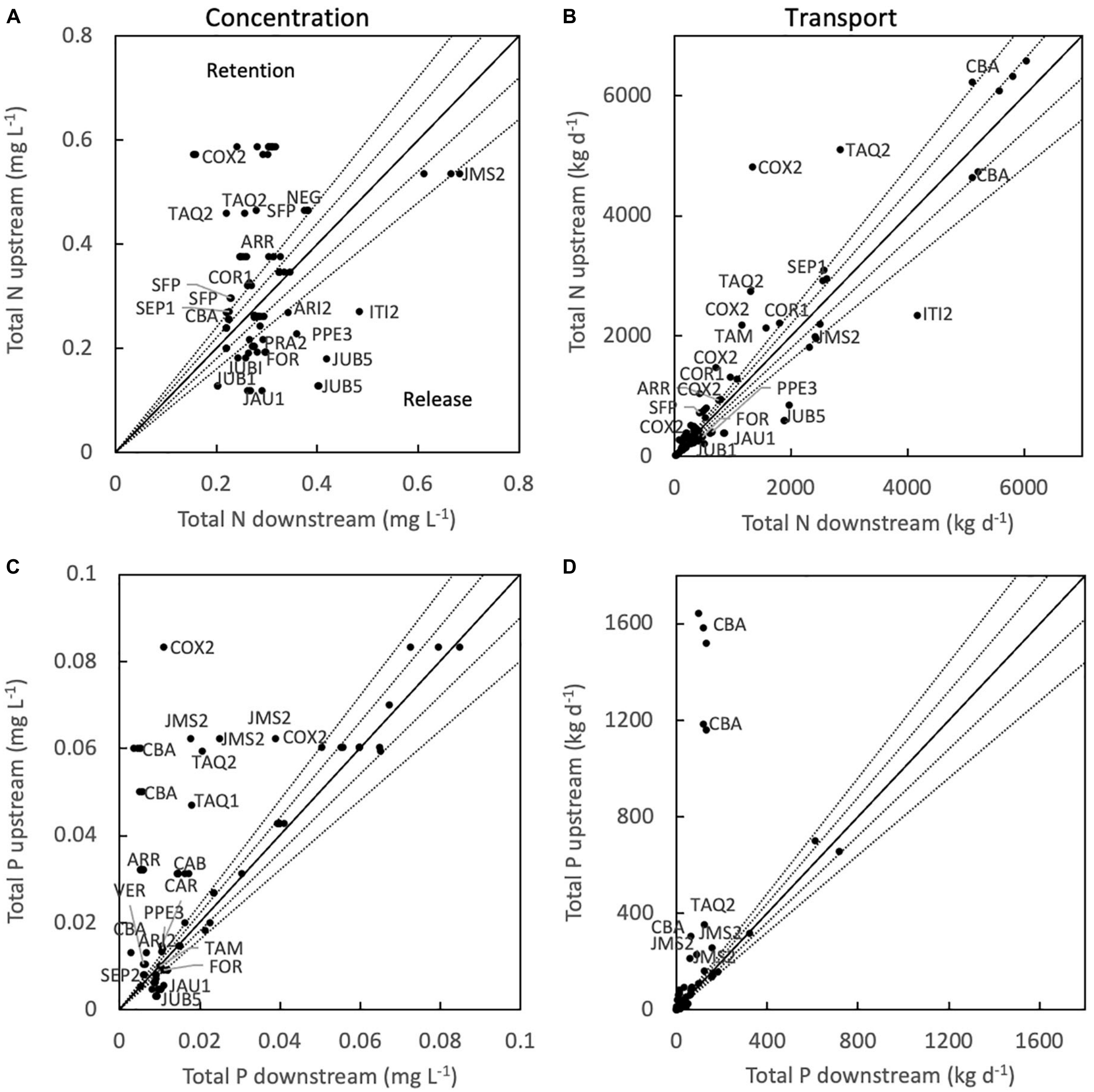

FIGURE 7 | Comparisons of neural network model predictions of changes in total nitrogen and phosphorus between upstream and downstream of future hydroelectric facilities, with river systems labeled by codes in **Table 1**. **(A)** Total N concentrations, **(B)** total N transport, **(C)** total P concentrations, and **(D)** total P transport. Solid line shows the line of parity and dashed lines show bounds of ±10 and ±20% around that line; points above the line indicate net retention and those below indicate net release between the upstream and downstream sampling points. Upstream:downstream ratios that deviate considerably from 1:1 are identified with the codes shown in **Table 1**. Both primary and secondary data are included in the figure and all data are in the **Supplementary Table S5**.

especially in the three rivers with highest rates of transport, or not change much. Summing all of the riverine transport rates (including the aforementioned smaller rivers) indicates that future hydropower development would result in net reductions of 8% of the TN transport and 29% of the TP transport from the uplands to the Pantanal.

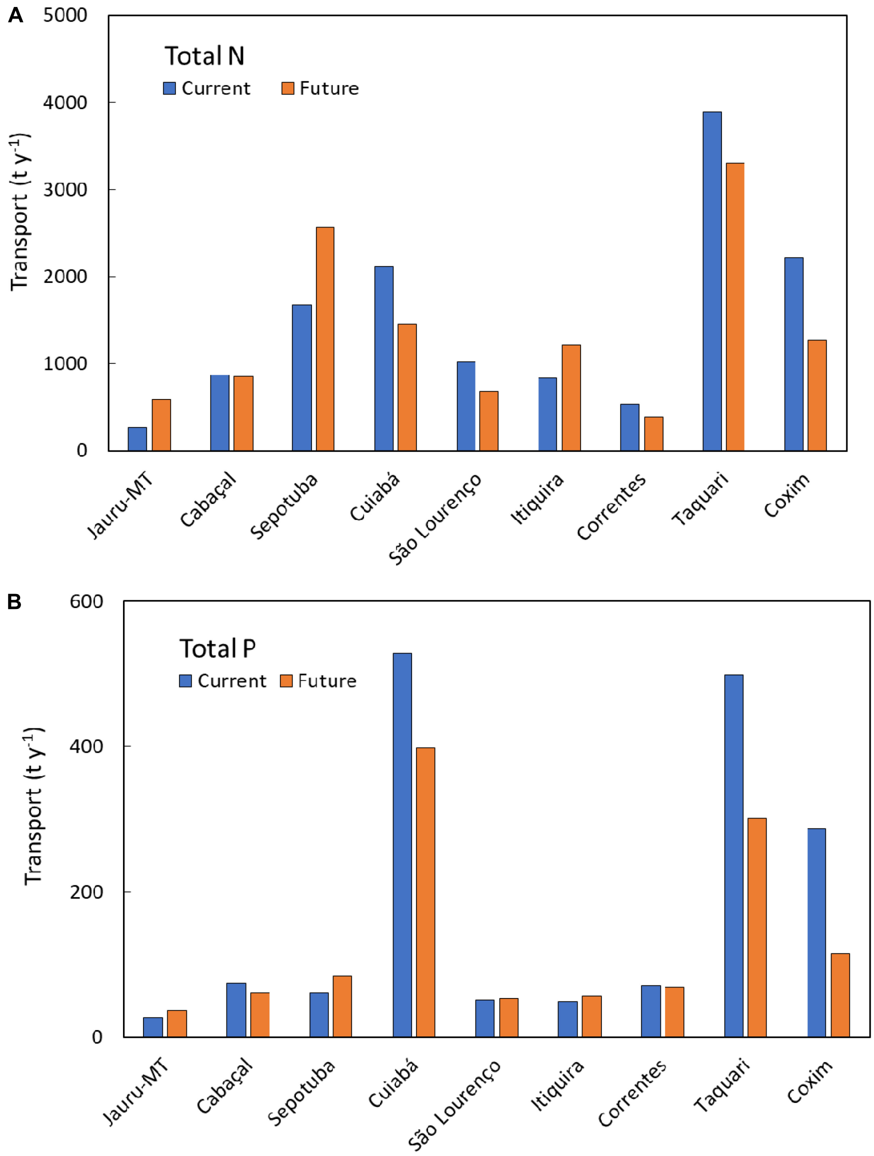

FIGURE 8 | Summary of transport of **(A)** total nitrogen and **(B)** total phosphorus by rivers from the upland watershed into the Pantanal. Bars show current rates and rates predicted by the neural network modeling were all future hydropower facilities to be built in each river system.

DISCUSSION

Our comprehensive analysis of how SHPs affect nutrient concentrations and transport, which is unprecedented in the literature, applies not only to future hydropower development in the Upper Paraguay River basin and elsewhere in Brazil, but also worldwide. The results are especially pertinent to decision-making regarding further SHP development in similar landscapes and climates elsewhere in Latin America, East Africa, and Southern Asia (Couto and Olden, 2018). Landscape and climate are relevant because they determine the nature and quantity of riverine nutrient and sediment loads, and climate is also relevant because warm climates can support high rates of biological activity all year.

Overall the impacts of current hydropower facilities on nutrient transport were not large, and in most cases were not distinguishable based on comparisons between samples taken upstream and downstream of the facilities. This contrasts with the well-documented retention of N and P by larger reservoirs around the world (Maavara et al., 2020). As we hypothesized at the outset, the short water residence times of most of the reservoirs associated with SHPs likely explain their tendency to have little or no detectable effect on nutrient transport. However, model predictions for future hydropower facilities project significant reductions in TN and TP concentrations and/or transport, with potential negative consequences for downstream river and floodplain productivity. This difference between the conclusions for current and future hydropower facilities is explained by the larger discharge and sediment loads of a number of the river systems where future facilities are planned (e.g., the Taquari and Cuiabá rivers) (Fantin-Cruz et al., 2020); the modeling accounted for the greater potential trapping of sediments and associated nutrients by dams on more sediment-rich rivers.

Nutrient Concentrations in Rivers Draining to the Pantanal

The rivers in the upland portion of the Upper Paraguay River basin generally show concentrations of TN and TP that would be associated with oligotrophic to mesotrophic states in recipient water bodies (Wurtsbaugh et al., 2019). Total P is carried predominantly in particulate form, whereas dissolved inorganic N tends to be a substantial fraction of TN concentrations and transport. These results are consistent with data reported by Oliveira et al. (2019) for the major rivers at their points of entry into the Pantanal. Ratios of TN: TP suggest that aquatic primary production would potentially be more N-limited in the wet season and P-limited in the dry season, likely reflecting the lower concentrations of suspended particulate P at lower flows (Guildford and Hecky, 2000). Bedload transport of TN, TP, and POC is consistently small compared to transport in the water column; no studies of bedload nutrient transport in these kinds of rivers are available for comparison.

Effects of Current Hydropower Facilities on Nutrient Transport

Comparison of upstream to downstream concentrations for 25 reaches containing current hydropower facilities showed that most facilities did not markedly alter TN, DIN, TP, and POC concentrations and transport. Overall the majority of reaches showed no consistent changes in nutrient concentrations, although TN and TP showed changes in 16 and 15% of the reaches, respectively.

The relatively large reservoirs with longer water residence times would be expected to show the most nutrient retention. As river water passes through the PCH São Lourenço, with a water residence time of up to 18 days, Fantin da Cruz et al. (in review) showed water quality changes including reductions in pH, dissolved oxygen, suspended solids, and turbidity, and Fantin da Cruz et al. (in review) showed that the majority (62%) of sediment inputs was retained, yet we found little change in TN, TP or POC. Another large reservoir (mean water residence time of 16 days) is Ponte de Pedra, a 176-MW hydropower facility on the Correntes River. Our data showed significant reductions in TP concentrations, and a previous study documented reductions in turbidity and concentrations of total suspended solids, TP, and nitrate (Fantin-Cruz et al., 2016). Smaller reservoirs in the present study showed few changes, although median concentrations suggest that TN was retained by the Itiquira dam and POC was retained by some dams on the Jauru, Correntes, and São Lourenço rivers. However, those changes were not consistently observed across all sampling dates, as indicated by the lack of statistical significance.

There are a few studies of SHPs from this region or comparable settings to compare with our observation that most current SHPs did not markedly alter concentrations and transport of TN and TP. Fantin da Cruz et al. (in review) found that SHPs on tributaries in the São Lourenço River watershed with short water residence times (1–2 days) may have been associated with longitudinal increases in pH and dissolved oxygen, but had no detectable effect on temperature, total dissolved solids, suspended solids, turbidity, or chemical oxygen demand. In contrast, Coelho da Silva et al. (2019) reported changes in water quality along the Jauru River where a series of six hydropower facilities produce cumulative total water residence times of ~17 days. Comparisons of samples collected over the years before and after hydropower facility construction in the Jauru River showed changes in suspended solids, TN, and TP, although in variable directions, and the authors noted the difficulty of ascribing the cumulative changes to the facilities given their variable designs and the limited pre-dam sampling. Timpe and Kaplan (2017) analyzed hydrological alterations at multiannual time scales by a large number of dams in the Amazon and Upper Paraguay basins, including eight facilities we also studied on the Santana, Juba, Jauru, Casca and Aricá rivers (**Supplementary Table S1**). Those authors found a tendency for dams to significantly alter flow regimes, with larger effects in lowland facilities with large dams and reservoirs, although the magnitude of alteration was comparable between large dams and SHPs when the alterations were scaled to the facilities' installed capacities. Hydrological alterations at sub-daily scales by many of the hydropower facilities studied here are analyzed by Figueiredo et al. (in review).

Reviews of the impacts of SHP dams elsewhere in the world have either reported a lack of measurements of nutrients (Anderson et al., 2015; Kelly-Richards et al., 2017; Athayde et al., 2019) or a tendency for only small effects of low-head dams (Mbaka and Mwaniki, 2015). A number of studies have documented sediment retention to the point of complete infilling of reservoirs behind old run-of-river dams in the U.S. that were candidates for removal (Csiki and Rhoades, 2010), but how that long-term sediment accumulation relates to annual retention of sediments and associated nutrients above today's dams is unclear.

Predicting the Impacts of Future Hydropower Development

The ANN model predictions show further reductions in TN and TP concentrations and transport with the construction of all future hydroelectric facilities This is attributable to the expansion of future SHP construction into presently undammed river systems, such as the Cuiabá and Taquari (including its tributaries, the Coxim and Jauru-MS rivers), which carry higher loads of suspended particulate material that is prone to retention by sedimentation (Fantin-Cruz et al., 2020). Many of the current SHPs are located on smaller rivers with lower concentrations of suspended material and nutrients, and their smaller dams produce reservoirs with short water residence times. As expected, the ANN model predicted larger effects on rivers with higher particulate nutrient concentrations and transport, such as Taquari River system, which is the watershed with highest yields of N and P predicted by the SWAT model, greatly exceeding those of most of the northern watersheds (Mingoti et al., 2020).

Decreases in TP concentrations and transport are particularly likely to affect downstream primary production of both aquatic ecosystems and, where sediments are deposited on floodplains within the Pantanal, terrestrial plant growth during the dry season. Preferential retention of TP relative to TN would increase the likelihood of P limitation downstream. On an annual basis, most of the suspended sediments carried by rivers into the Pantanal become deposited on the floodplains (Oliveira et al., 2019). Fluvial inputs provide the only significant P inputs to these ecosystems, and TN:TP ratios show that decreases in TP concentrations will push these waters from likely N limitation toward N + P co-limitation or P limitation. From a regional standpoint, the most consequential reductions in TP transport into the Pantanal would be caused by future hydropower development in the Cuiabá and Taquari/Coxim river systems.

In some cases the model predictions indicated net increases (release) of TN or TP (**Figures 7,8**), and the Sepotuba River system showed the largest predicted net releases of TN and TP with future hydropower development. Either net TN or TP retention or release is conceivable as water passes through a hydropower facility, with retention likely attributable to sedimentation of particulate matter as well as biological uptake, whereas release could reflect remineralization from sedimented organic matter. Inputs of organic matter could be episodic and/or occur as coarse material, and in either case they could have been missed by our sampling. Large accumulations of coarse particulate matter were sometimes visible above SHP dams. Another possibility is that there are local sources of nutrient input from adjacent uplands.

Recommendations for Future Hydropower Development

Effects of new hydropower facilities on downstream nutrient concentrations and transport are one of a number of environmental and social considerations for decisions about whether and where to construct dams in the Upper Paraguay River basin. Effects on sediment transport and fish migrations are two other environmental impacts of paramount importance (Fantin-Cruz et al., 2020; Campos et al., 2020). Considering concentrations and transport of nutrients together with the parallel study of sediments presented by Fantin-Cruz et al. (2020), we argue that new hydropower facilities should not be built on undammed rivers entering the Pantanal that have particularly low nutrient concentrations and transport, as well as on those that have the highest absolute rates of transport to the Pantanal.

River systems with low nutrient concentrations are likely to be the least productive, and therefore the most sensitive to reductions in either TN or TP or both. The lowest nutrient concentrations reported in this study were found in the Sepotuba, Correntes and São Lourenço river systems. In response to present and future damming and the consequent retention of nutrients, these rivers and their floodplains may experience oligotrophication, with negative consequences for fisheries yields and overall river and floodplain ecosystem productivity (Stockner et al., 2000). Reduced riverine sediment and nutrient loads may eventually reduce the productivity of pastures used for cattle (Forsberg et al., 2017) and of fisheries within the Pantanal.

River systems that carry the largest quantities of nutrients to the Pantanal also deserve protection because their high nutrient loads support river and floodplain ecosystem productivity not only as they flow through the Pantanal but also downstream along the Paraguay River axis (Oliveira et al., 2019). River systems of particular importance to the nutrient budget of the Pantanal that remain undammed in their lower reaches include the Cuiaba and Taquari/Coxim.

Effects on sediment transport to the Pantanal are larger and are also a key consideration for these recommendations (Fantin-Cruz et al., 2020). Based on the results of these two studies, we recommend no future hydropower development on four river systems presently lacking dams in their mainstem reaches within the uplands – the Cabaçal, Sepotuba, Cuiabá, and Taquari/Coxim rivers. Without dams, these rivers would maintain the natural export of nutrients and sediments from the uplands to downstream rivers and floodplains that is essential to support the productivity and biodiversity of the Pantanal Wetland. Additionally, important fish migration corridors would be preserved (Campos et al., 2020).

DATA AVAILABILITY STATEMENT

All datasets generated for this study are included in the article/**Supplementary Material**.

AUTHOR CONTRIBUTIONS

MO, IF-C, MS, DF, ED, and SH conceived and carried out the study. IF-C, MO, and MC conducted the field work and data analysis. JC, OP, and RM developed the modeling. MO, IF-C, and SH wrote the manuscript. All the authors contributed to the article and approved the submitted version.

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Dam-Induced Hydrologic Alterations in the Rivers Feeding the Pantanal

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Tropical river basins have experienced dramatically increased hydropower development over the last 20 years. These alterations have the potential to cause changes in hydrologic and ecologic systems. One heavily impacted system is the Upper Paraguay River Basin, which feeds the Pantanal wetland. The Pantanal is a Ramsar Heritage site and is one of the world's largest freshwater wetlands. Over the past 20 years, the number of hydropower facilities in the Upper Paraguay River Basin has more than doubled. This paper uses the Indicators of Hydrologic Alteration (IHA) method to assess the impact of 24 of these dams on the hydrologic regime over 20 years (10 years before and 10 years after dam installation) and proposes a method to disentangle the effects of dams from other drivers of hydrologic change using undammed "control" rivers. While most of these dams are small, run-of-the-river systems, each dam significantly altered at least one of the 33 hydrologic indicators assessed. Across all studied dams, 88 of the 256 calculated indicators changed significantly, causing changes of 5–40%, compared to undammed reaches. These changes were most common in indicators that quantify the frequency and duration of high and low pulses, along with those for the rate and frequency of hydrologic changes. Importantly, the flow regime in several undammed reaches also showed significant alterations, likely due to climate and land-use changes, supporting the need for measurements in representative control systems when attributing causes to observed change. Basin-wide hydrologic changes (in both dammed and undammed rivers) have the potential to fundamentally alter the hydrology, sediment patterns, and ecosystem of the Pantanal wetland. The proposed refinement of the IHA methods reveals crucial differences between dam-induced alteration and those assigned to other drivers of change; these need to be better understood for more efficient management of current hydropower plants or the implementation of future dams.

Keywords: hydrologic alteration, IHA, Upper Paraguay Basin, hydropower plants, dams, rivers, Pantanal, Brazil

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INTRODUCTION

Growing electricity demand is a major global challenge as the need to find efficient and sustainable energy sources increases (International Energy Agency – IEA, 2019). Globally, total renewable power capacity more than doubled in the decade 2007–2017 (REN21, 2018), with a recent resurgence in focus on hydropower, especially in the developing tropics

(Zarfl et al., 2015; Winemiller et al., 2016). Although hydroelectric energy is considered a source of "clean" energy (Newell et al., 2019), there are myriad environmental and social impacts related to dam operations (e.g., Fearnside, 2016; Lima et al., 2016; Latrubesse et al., 2017; Athayde et al., 2019) that must be evaluated when considering the true costs and benefits of hydropower. One critical factor is how dam operation and location alter the hydrological dynamics in the basins where impoundments are located (e.g., Magilligan and Nislow, 2005; Poff and Zimmerman, 2010).

Brazil, in particular, has an energy matrix strongly concentrated on hydroelectricity, with ∼65% of its energy generated by hydropower plants[1]. Country-wide, there are 639 hydropower plants, including 422 small hydroelectric power plants referred to as PCH in Brazil (defined as having production potential between 5 and 30 MW and reservoirs with a surface area smaller than 3 km^2) and 217 hydroelectric facilities referred to as UHE (defined as having a capacity >30 MW in operation). Over 40% of the operating hydropower plants are localized in the Brazilian Southeast and West Central regions, while planned facilities, mainly PCH (>90%) will be built in the West Central and North regions. In the Upper Paraguay Basin, where the Pantanal is found, there are 57 operating hydropower plants, of which 6 are UHE and 51 PCH (only 28 PCH with available data). Eight more hydropower plants are in construction or soon to be constructed[1]. An additional 80 hydropower plants are projected or planned. In 2008, there were 39 operating hydropower plants in the Upper Paraguay River Basin (Girard, 2011).

In the Upper Paraguay basin, dams are not usually built to divert water for irrigation purposes. Dams are constructed to provide electrical energy. A notable exception was the Manso dam, which was built for hydropower generation but as well to regulate droughts and flood downstream in the city of Cuiabá (Zeilhofer and Moura, 2009). This does not mean that reservoirs are not used by farmers. They do indeed use the reservoir for irrigation and also to provide drinking water for cattle, which is one of the priorities usage of water recognized by Brazilian water legislation. Consumption uses in the northern part of the basin remove 7.0% (27.4 m^3/s) of the minimum flow of seven consecutive days, of which 44.0% is used for irrigation and 19.7% by animals (Agência Nacional de Águas - ANA, 2018). Withdrawals directly from reservoirs correspond to <0.01% of the available flow. Thus, despite the scenario of increased water withdrawal in the basin, these uses can be considered insignificant for changes in the flow regime on a daily scale.

Currently, many dams are constructed on the same river, one downstream of another, in cascade. Dams on the Jauru or the Juba rivers provide a good example (**Figure 1**). These cascades of dams can be all small hydroelectrical facilities (PCH), like in the Santana case, or include one or more large dams (UHE) like in the Juba case. Unlike the larger dams (UHE), small dams cannot control the seasonal riverine flow regime and are considered as run-of-the-river as they canalize a portion of a river, usually with little or no impoundment.

If hydropower development plans are fully implemented, about 40% of the flow to the Pantanal would go through one or more hydropower plants in the future (Souza Filho, 2013), which could significantly change the ecological dynamics of the Pantanal, altering not only the hydrology but also the thermal, nutrient, sediment and water chemistry regimes, geomorphology, and ecology (Olden and Naiman, 2010; Gao et al., 2012). Specific impacts resulting from impoundments include the reduction of aquatic and terrestrial productivity within the flooded areas (Burford et al., 2011; Schindler and Smits, 2017); instability of river channels (Brooks et al., 2012); reduction of available habitats and landscape alteration (Miranda et al., 2015; Aguiar et al., 2016); reduced connectivity between the river and the riparian zone; disturbance of hydrological regime (Stevaux et al., 2012); and changes in water quality (Fantin-Cruz et al., 2016; Silva et al., 2019).

Most of the recent facilities in the Upper Paraguay basin are small hydropower plants due to current government incentive policies. However, according to the World Comission on Dams - WCD (2000), small and medium-sized hydropower plants have the potential to modify vital ecosystem functions and affect water security in similar ways as large projects. The operation of small hydropower was considered to have minimal environmental impacts because they do not typically involve water storage and diversion (Small Hydro Energy Efficient Promotion Campaign Action – SHERPA, 2010; Werthessen, 2014). However, small dams can have large impacts, especially relative to their production capacity (Timpe and Kaplan, 2017); this is particularly troubling given their accelerating, widespread, and largely unregulated expansion both globally (Couto and Olden, 2018) and in Brazil (Athayde et al., 2019; Campos et al., 2020).

Understanding dam-induced changes to the flow regime in the Upper Paraguay River Basin is of regional and international relevance, as the Pantanal, a vast floodplain wetland, is considered to be one of the most diverse terrestrial biomes in the world. The Pantanal is a UNESCO World Heritage Site (Junk and Cunha, 2005), and the ecosystem produces environmental services of great economic value to the region, including maintenance of regional microclimates, regulation of river discharge, fishing, native pasture, habitat for threatened species, and wintering ground for migratory species (Wantzen et al., 2008; Tomas et al., 2019; Campos et al., 2020). The provision of these services depends on the flood pulse from upstream regions which delivers nutrients, sediments, fauna, and plant propagules to the floodplain and maintains large areas flooded for long periods (Girard et al., 2010). This natural behavior of flood-pulsed rivers conditions multiple ecological processes in the river-floodplain system, including the richness, abundance, and distribution of fauna, flora, and human activities in the region (Junk et al., 1989; Oliveira et al., 2019; Silveira et al., 2020). Constructed and planned dams in the Upper Paraguay Basin have the potential to alter nutrient cycling, sedimentation processes downstream, and the flood pulse (Ivory et al., 2019). Other potential dam impacts in the

[1] Agência Nacional de Energia Elétrica – ANEEL (2020a) Sistema de Informações de Geração da ANELL SIGA – Empreendimentos por Sub-bacia. https://clck.ru/NpGrX [Accessed June 6, 2020].

FIGURE 1 | Dams on the river feeding the Pantanal. The upper left inset shows the Pantanal extent (yellow area) in the Upper Paraguay Basin (UPB—green area) in South America. The lower left inset shows the Brazilian Pantanal (yellow area) in the Brazilian portion of the UPB (green area) in the Brazilian States of Mato Grosso and Mato Grosso do Sul. The main figure shows the study area: the Paraguay river (red), the dammed (blue) and undammed or control rivers (yellow), of the Brazilian UPB (green shaded area) flowing into the Brazilian Pantanal wetland (yellow shaded area) and hydroelectric power plants (<30 MW: triangles; > 30 MW: stars), control stations (squares), and fluviometric stations (circles).

Pantanal floodplain are blocking fish migration routes (Campos et al., 2020) and thus reducing fisheries production; progressive water loss, especially in permanent water bodies, which is likely to provoke changes in the composition and structure of their biological assemblages (Silio-Calzada et al., 2017); and repercussions for the livelihoods of local communities who depend on fishing and the collection of other natural resources (Schulz et al., 2019).

Many studies have been carried out to evaluate the ecohydrological consequences of dam construction and operation (Costigan and Daniels, 2012). More than 170 hydrological indices have been developed to identify the various components of the flow regime and assess their contributions to the ecology of rivers (Olden and Poff, 2003). To assess hydrologic alterations in aquatic ecosystems, Richter et al. (1996) proposed a method for evaluating 33 different indicators of the flow regime, called Indicators of Hydrologic Alteration (IHA). This method has been widely used all over the world (Richter et al., 1996, 1997; Magilligan and Nislow, 2005; Rocha, 2010; Li and Qiu, 2016; Zhang et al., 2016; Sabino, 2017; Timpe and Kaplan, 2017). The type and magnitude of dam-induced hydrologic change found across these studies vary greatly as a function of climate, geography, dam type, and reservoir operation protocols, suggesting that regional analyses are needed to characterize relevant aspects of flow alteration in specific areas.

Environmental impact assessments commonly compare post-impact outcomes with what "would have happened" if the impact had not occurred (referred to as the "counterfactual") (Valle and Kaplan, 2019). A common approach for testing the counterfactual is to compare variables of interest before and after the impact of a dam under the assumption that the counterfactual would be similar to the pre-impact observations. However, this assumes stationarity of all other relevant drivers. Multiple studies have used his before/after approach to quantify how dams alter riverine hydrology (e.g., Forsberg et al., 2017; Sabo et al., 2017; Gierszewski et al., 2020) despite this limitation. Stationarity is assumed by hydrologic indicator methods, including IHA, but this assumption may not be valid when other variables (e.g., climate, land use, and management regimes) are dynamic or directional, which can be the case when assessing long-term hydrological trends (Milly et al., 2008; Salas and Obeysekera, 2014).

Given this limitation, we sought to explicitly adjust for other drivers of change to quantify impacts that are related to the dam, rather than other changes such as land use or climate (Fantin-Cruz et al., 2015). This was accomplished by comparing flow variation in rivers impacted by hydropower plants to flow variation in undammed rivers, which were used as controls (see Singer, 2007; Rheinheimer and Viers, 2015; Meitzen, 2016).

The overarching goal of our study was to quantify the effects of hydropower plants on the hydrological regimes of rivers feeding the Pantanal. We used the IHA (Richter et al., 1996) to assess how the construction of dams changed the hydrological parameters in the dammed rivers. We then identified which IHA parameters were most affected by hydropower plant operation in the Upper Paraguay River Basin.

MATERIALS AND METHODS

Study Area

The study area encompasses the portion of the Upper Paraguay River Basin within the State of Mato Grosso (**Figure 1**). The Upper Paraguay River Basin has a humid tropical climate, with a well-defined rainy season between October and March and a dry season between April and September. Mean annual precipitation varies between 800 and 2,000 mm, with an average of 1,368 mm/year (Agência Nacional de Águas - ANA, 2018). The Upper Paraguay River Basin comprises a drainage area of ∼600,000 km² (Bravo et al., 2012), with the Brazilian portion totaling 362,380 km². Forty-eight percent of the Brazilian basin is within the State of Mato Grosso with the remainder in the State of Mato Grosso do Sul (Instituto Brasileiro de Geografia e Estatística – IBGE, 2016). About 150,000 km² of the Brazilian portion of the Upper Paraguay River Basin is classified as floodplains, where the Pantanal is located, and the rest is described as the upper plateau (Bergier et al., 2018) (**Figure 1**). The plateau region has an average altitude above 200 m and a maximum elevation of 1,400 m in the eastern portion of the basin with well-defined and convergent drainage (Agência Nacional de Águas - ANA, 2018). The Mato Grosso upper plateau is considered an important region for the Pantanal supply since the rivers that arise in this portion of the Upper Paraguay River Basin contribute more than two-thirds of the average annual flow into the Pantanal (Girard, 2011). Dams are built in the plateau region, and most of the dams in this study are located in the upper half of the plateau region closer to river headwaters where gradients are generally steeper. Notable exceptions are the Itiquira and Ponte de Pedra large hydropower plants built, respectively, on the Itiquira and Correntes rivers which are located in the lower half of the upper plateau region (**Figure 1**). Note that The Pantanal is fed by the upper plateau rivers. It has an average altitude of 80–150 m and is characterized by an intricate drainage system of large lakes, divergent watercourses, and areas of seasonal drainage and flooding.

Hydropower Plants, Dammed, and Undammed Rivers Gage Stations

The initial goal of the study was to use all hydropower plants in the study area, but many did not have sufficient flow data for IHA analysis. The hydropower plants retained for analysis were selected according to the extent of the available fluviometric record before and after dam construction in monitoring stations located down-river of the dam or dam cascade. Ideally, monitoring flow stations downstream of the hydropower plants should have at least 10 years of data available before and after the operational start dates (Richter et al., 1997). The Cuiabá (6628000) and São Lourenço (66400000) monitoring stations did not meet this requirement for the post-operation time series, as the dams were recently built (**Table 1**). However, the length of record required to support IHA analysis in the tropics vary widely, as low as 2–7 years for rivers with strongly repeated seasonality (Timpe and Kaplan, 2017). These dams were thus included as they had 7 and 9 years of post-operation fluviometric data, respectively.

Sufficient hydrologic information was available for 24 operational hydropower plants in the Mato Grosso State Upper Paraguay River Basin (**Table 1**). Several of these are in cascades of dams on the same river, such as dams on the Juba (4 hydropower plants), Jauru (6 hydropower plants), and Santana (2 hydropower plants) rivers (**Table 1**, **Figure 1**). One gage station downstream from each one of these cascades of hydropower plants was available for IHA analysis. In other cases, the only available monitoring fluviometric station was downstream from a group of dams in the same drainage basin, such as the arrangement of 5 hydropower plants in the São Lourenço River Basin and of 4 hydropower plants built in the Piquiri basin. In the São Lourenço River arrangement, the São Lourenço hydropower plant is downstream from the other four on the São Lourenço main channel. It is also the only one with a significant reservoir (5 km^2). The Sucupira and Pequi hydropower plants are installed on the Saia Branca River, the two other hydropower plants are installed on the Ibó stream and the Tenente Amaral River. The Saia Branca, Tenente Amaral, and Ibó are all tributaries of the São Lourenço.

In the Piquiri River Basin, the arrangement is composed of the Itiquira hydropower plant, built on the Itiquira River, and the Ponte de Pedra, Aquárius, and São Gabriela hydropower plants are built on the Correntes River. Both the Correntes and the Itiquira River are tributaries of the Piquiri River, where the monitoring station is located. As the Itiquira hydropower plant is the largest in this arrangement, it will be referred to as Itiquira hydropower plants. In the cases of the Manso, São Tadeu, and Poxoréo hydropower plants, there is only one hydropower plant on each river with a downstream fluviometric station (**Figure 1**, **Table 1**). For each hydropower plant, cascade, or arrangement of hydropower plants, the operation period starts when the first dam was built. We did not consider the construction period as only the operation start date was known with certainty. All data before the operation start date were taken as the pre-operation period. All data concerning hydropower plant characteristics (installed power, reservoir area, dam locations, start of operation dates) were obtained from the Brazilian Agency for Electrical Energy—ANEEL[2]. All data regarding monitoring fluviometric stations (location, contributing area, daily discharges, length of record) were obtained from the National Water Agency - ANA web site[3] Hidroweb – Portal Hidroweb. http://www.snirh.gov.br/hidroweb/ [Accessed June 6, 2020]. Distances between downstream fluviometric stations and the dam, or the closest dam in a cascade or arrangement were obtained from Google Earth and correspond to the river path distances.

All hydropower plants with installed power $\leq$ 30 MW are operating as run-of-the-river systems; even if the dam elevates the water line to produce an impoundment (generally lower than

[2]Agência Nacional de Energia Elétrica – ANEEL (2020b) SIGEL - Sistema de Informações Georreferenciadas do Setor Elétrico SIGEL– Download. https://sigel.aneel.gov.br/Down/ [Accessed June 6, 2020].
[3](Agência Nacional de Águas - ANA, 2020)

TABLE 1 | Characteristics of the hydropower plants used to study hydrological alterations in the Upper Paraguay Basin.

Hydropower plant (dam, cascade, arrangement)	Names of the hydropower plant	Operation starting date	River	Installed Power (MW)	Reservoir Area (km^2)	Number of gage station (River name)	Station dates	
							Start	End
1. Jauru cascade	Antonio Brennand	Sep., 2002	Jauru	22.0	0	66071400 (Jauru)	June 22nd, 1979	June 31st, 2017
	Jauru	Jun., 2003		121.5	2.62			
	Indiavaí	Aug., 2003		28.0	0.27			
	Ombreiras	Jul., 2005		26.0	3.47			
	Salto	Dec., 2007		19.0	0.79			
	Figueirópolis	Sep., 2010		19.4	7.44			
2. Juba cascade	Juba I	Nov., 1995	Juba	42.0	0.92	66055000 (Sepotuba)	September 9th, 1969	April 4th, 2018
	Juba II	Aug., 1995		42.0	2.79			
	Graça Brennand	Jun., 2008		27.4	5.34			
	Pampeana	May., 2009		28.0	4.17			
3. Santana cascadde	Diamante	Dec., 2005	Santana	4.2	0.49	66006000 (Santana)	November 10th, 1967	October 31st, 2017
	Santana I	Apr., 2012		14.8	1.19			
4. Manso dam	Manso	Nov., 2000	Manso	210.0	427	66231000 (Manso)	July 12th, 1981	December 31st, 2017
5. São Tadeu I dam	São Tadeu I	Dec., 2010	Aricá-Mirim	18.0	0.46	66280000 (Cuiabá)	June 01st, 1966	December 31st, 2017
6. São Lourenço arrangement	Sucupira	Oct., 2008	Saia Branca	4.5	0.071	66400000 (São Lourenço)	April 11th, 1965	June 16th, 2017
	Pequi	Dec., 2008	Saia Branca	6.0	0.04			
	São Lourenço	Apr., 2009	São Lourenço	29.1	5			
	Sete Quedas Alta	Dec., 2010	Cór. Ibó	22.0	0.18			
	Cambará	Dec., 2012	Ten. Amaral	3.5	0.057			
7. Poxoréo dam	Poxoréo	Jan., 1998	Poxoréo	1.2	0.18	66430000 (Vermelho)	October 24th, 1987	November 31st, 2017
8. Itiquira arrangement	Itiquira	Nov., 2002	Itiquira	110.5	1	66600000 (Piquiri)	December 25th, 1967	January 31st, 2018
	Ponte de Pedra	Jul., 2005	Correntes	176.1	17			
	Aquárius	Sep., 2006		4.2	0			
	Santa Gabriela	Sep., 2009		24.0	0.71			

The numbering of the hydropower plants is the same as in ***Figure 1****.*

3 km^2). Most of these hydropower plants do not have floodgates and consequently, the dam is not equipped to actively change the seasonal flow regime of the river. The hydropower plants over 30 MW (Juba I and II, Jauru, Manso, Itiquira, and Ponte de Pedra), have floodgates which allow for the active alteration of the flow regime downriver. However, most of these have small reservoirs (< 8 km^2) which limit this capacity. Only Manso and Ponte de Pedra hydropower plants have large reservoirs. The Manso hydropower plant has a 427 km^2 reservoir, roughly equivalent to 3 years of mean discharge (Zeilhofer and Moura, 2009). The Pontes de Pedra hydropower plant reservoir area is 17 km^2, storing a volume equivalent to about 1 month of discharge (Fantin-Cruz et al., 2016).

In addition to pre- post-dam analysis, each dammed river was evaluated against one station in an undammed river (**Figure 1**, **Table 2**). Each station on an undammed river was chosen so that it was on a reach with no dams upstream and no downstream dams nearby. Ideally, several additional criteria to help select undammed river stations were followed whenever possible: (1) it should be in the same drainage basin as the hydropower plant; (2) the river reach should be geographically close to the station monitoring the hydropower plants; (3) and the station should be in the same river section position (e.g., if the monitoring station on a dammed river was mid-basin, the undammed river one should be too), so that river regimes would be similar. By doing so, it is also likely to reduce hydrologic changes due to the geographical variations in geology, geomorphology, and topography. Due to data availability, several of the ideal criteria could not be met for each river assessed here. Specifically, the São Tadeu undammed station is far from the dammed station; the São Lourenço undammed station is not in the same drainage basin as the undammed river monitoring gage; the Itiquira

TABLE 2 | Characteristics of the undammed rivers gage stations used as controls in this study.

Controlled hydropower plant	Operation starting date	Undammed river gage station number	Undammed river	Station dates	
				Start	End
Jauru cascade	Sep., 2002	66074000	Aguapeí	December 14th, 1965	January 31st, 2018
Juba cascade	Nov., 1995	66050000	Sepotuba	July 24th, 1971	December 31st, 2007
Santana cascade	Dec., 2005	66008000	Jaquara	November 12th, 1967	December 31st, 2017
Manso dam	Nov., 2000	66173000	Da Casca	August 13th, 1982	December 31st, 2016
São Tadeu I dam	Dec., 2010	66140000	Cuiabá	July 09th, 1979	April 30th, 2018
São Lourenço arrangement	Oct., 2008	66440000	Jorigue	June 26th, 1979	February 28th, 2018
Poxoréo dam	Jan., 1998	66440000	Jorigue	June 26th, 1979	February 28th, 2018
Itiquira arrangement	Nov., 2002	66520000	Itiquira	June 01st, 1971	December 31st, 2017

The date of operation (equal to the start of operation for the hydropower plants) was used to run the Indicator of Hydrologic Alteration analysis.

undammed station is up basin while the dammed river station is down basin (**Figure 1**). Note, however, that all gage stations, either for dammed or undammed rivers are located in the upper plateau region.

Analysis Framework

The Indicators of Hydrologic Alteration (IHA) is based on the analysis of hydrologic data available either from existing measurement points, such as stream gauges or wells (Richter et al., 1996). It uses 33 indicators to statistically characterize hydrologic variation within each year. These indicators inform on ecologically significant features of surface and groundwater regimes influencing aquatic, wetland, and riparian ecosystems. The 33 hydrologic alteration indicators were evaluated by comparing the hydrological regime before and after the start of operation of each hydropower plant using IHA 7.1 software (The Nature Conservancy - TNC, 2009) which produces measures of central tendency and dispersion for each parameter between defined "pre-dam" and "post-dam" time frames, allowing to quantify dam impacts assuming stationarity of other drivers. The IHA organizes these 33 indicators into 5 groups: (i) magnitude of flows (median discharge) in each month; (ii) magnitude and duration of annual extreme flow conditions (medians minima and maxima discharge of 1, 3, 7, 30, and 90 days); (iii) timing of annual extreme flow conditions (date of the 1-day minima and maxima); (iv) frequency and duration of high and low pulses (thresholds were set as the median flow plus or minus 25%), and; (v) rate and frequency of hydrologic changes (obtained by dividing the hydrologic record into "rising" and "falling" periods, which correspond to periods in which daily changes in flows are either positive or negative, respectively). Each group of parameters is associated with ecosystem functions listed in **Table 3**; a thorough description of each indicator is provided by Richter et al. (1997).

One further step was included before the calculation of the hydrological alteration. To adjust for hydrological alterations due to other drivers of change (land use and climate) and estimate only alterations provoked by hydropower plants, hydrological changes in dammed rivers were compared with those found in undammed rivers. The hydrological alteration in the undammed rivers was calculated using the same procedure as for dammed rivers. Impacts in dammed rivers that were not deemed different from those encountered in undammed rivers were not included in the calculation of the hydrologic alteration.

The IHA software (The Nature Conservancy - TNC, 2009) calculates a deviation factor by comparing post to pre-impact periods for 33 hydrologic indicators. For each dammed river, an undammed river was chosen to monitor the extent of parameter deviation in the absence of hydropower plants. In this study, the deviation factor was called the hydrologic alteration factor (*HA*) (Timpe and Kaplan, 2017). The HA_i for each IHA parameter i was the relative change of the median of the indicator of the post-impact period ($M_{i,post}$) to the pre-impact period ($M_{i,pre}$) in percent:

$$\boldsymbol{HA_i} = \left(\frac{\boldsymbol{M_{i,\,post}} - \boldsymbol{M_{i,\,pre}}}{\boldsymbol{M_{i,\,pre}}}\right) \times 100 \tag{1}$$

HA_i was calculated for dammed and undammed rivers. In any undammed river, the pre and post-impact periods were the same as in the dammed river. The IHA software provided its test of significance, called significance count (SC). The SC was calculated by randomly shuffling data across the entire period of record and regenerating pre- and post-impact medians 1,000 times. The SC was the fraction of those 1,000 iterations for which calculated *HAi* (Equation 1) values were greater than those for the unshuffled data and could be likened to a *p*-value in parametric statistics (The Nature Conservancy - TNC, 2009). Whenever SC was > 0.05 in dammed or undammed rivers, HA_i was considered not significant.

Next, a step-wise screening algorithm was used to assess whether a given HA_i was different between dammed (d) (HA_{id})

TABLE 3 | Summary of hydrological parameters used in the Indicator of Hydrologic Alteration Software (IHA) and their characteristics.

IHA Groups	Hydrological parameter	Ecosystem functions
Group 1: Magnitude of monthly flow conditions	Median discharge for each calendar month	• Habitat availability for aquatic organisms • Soil moisture availability for plants • Availability/Reliability of water for terrestrial animals • Access by predators to nesting sites • Influences water temperature, oxygen levels, photosynthesis in water column
Group 2: Magnitude and duration of annual extreme flows conditions	Annual minima (1-, 3-, 7-, 30-, 90-days medians discharge) Annual maxima (1-, 3-, 7-, 30-, 90-days medians discharge) Number of days with Zero flow Base Flow Index (7-day minimum flow/mean flow for year)	• Balance of competitive, ruderal, and stress-tolerant organisms • Structuring of river channel morphology physical habitat and ecosystem conditions • Soil moisture stress in plants • Dehydration in animals • Anaerobic stress in plants • Volume of nutrient exchanges between rivers and floodplains • Duration of stressful conditions in aquatic environments
Group 3: Timing of annual extreme flow conditions	Julian dates of each annual 1-day maximum Julian dates of each annual 1-day minimum	• Compatibility with life cycles of organisms • Predictability/availability of stress for organisms • Access to special habitats during reproduction or to avoid predation • Spawning cues for migratory fish
Group 4: Frequency and duration of high and low pulses	Number of high pulses each year Number of low pulses each year Median duration of high pulses within each year, in days Median duration of low pulses within each year, in days	• Frequency and magnitude of soil moisture stress for plants • Frequency and duration of anaerobic stress for plants • Availability of floodplain habitats for aquatic organisms • Access for waterbirds to feeding, resting, reproduction sites • Influences bedload transport, channel sediment textures, and duration of substrate disturbance (high pulses)
Group 5: Rate and frequency of hydrologic changes	Rise rates: median of all positive differences between consecutive daily values Fall rates: median of all negative differences between consecutive daily values Number of hydrologic reversals	• Drought stress on plants (falling levels) • Entrapment of organisms on islands, floodplains (rising levels) • Desiccation stress on low-mobility stream edge organisms

Adapted from Richter et al. (1996) and The Nature Conservancy - TNC (2009).

and its undammed control (c) rivers (HA_{ic}). This algorithm removed any changes that were statistically similar between dammed and undammed rivers. First, the HA_{id} was tested for significance from zero. If it was not significant, its value was set to zero. Otherwise, HA_{id} was compared to the deviation of the same IHA parameter on the undammed river (HA_{ic}). If HA_{ic} was not significant, then HA_{id} was conserved for calculation of overall alteration. Otherwise, HA_{id} was further screened as follows: when HA_{ic} was significant, but its direction was opposite of HA_{id} ($HA_{id}/HA_{ic} < 0$), then HA_{id} was conserved for calculation. Otherwise, HA_{id} was further screened as follows: if HA_{ic} was significant and its direction the same as HA_{id} ($HA_{id}/HA_{ic} > 0$), then HA_{id} was conserved for further calculation only if $HA_{id}/HA_{ic} \geq 1.25$ (i.e., the proportional change in the IHA parameter in the dammed river was more than 25% greater than that in the undammed river); otherwise HA_{id} was set to 0 (zero).

Equation 1 yielded both positive and negative values of HA, corresponding to an increase or decrease in IHA parameters between the pre- and post-operation periods, respectively. IHA results for dammed and undammed rivers were summarized by calculating the arithmetic average (HA_{mean}) of the absolute value of HA_i, following Timpe and Kaplan (2017). HA_{mean} was calculated either using unadjusted HA_i provided by the IHA software (unadjusted HA_{mean}) or using only the ones adjusted by the previously described step-wise screening algorithm (adjusted HA_{mean}). All other parameters not retained by the step-wise screening algorithm were set to zero (0).

RESULTS

Flow Regimes of Dammed and Undammed Rivers

Discharge time series at fluviometric stations on dammed and undammed rivers indicated that dams were installed on seasonal river reaches, some highly variable. The fluvial regime also varied markedly between rivers (**Figure 2**). The dammed and undammed Jauru river stations (**Figure 2B**) are more typical of mid-reach regimes with amplitude to median ratios around 5. In contrast, the dammed Poxoréo River station and its control on the undammed Jorigue River, (**Supplementary Figure 1A**) illustrated flow regimes of the upper basin reaches. The amplitude (maximum-minimum discharge) of the Poxoréo river was more than 60 times its median flow, while for the Jorigue River, the same ratio is over a hundred. In these two cases, the regimes of the dammed and undammed river stations were well-paired. Other such well-paired stations included the Manso hydropower plant (**Supplementary Figure 1B**, ratios of 11 and 8.5, for dammed and undammed stations, respectively), the Juba hydropower plant (**Supplementary Figure 1E**, both ratios around 5), and

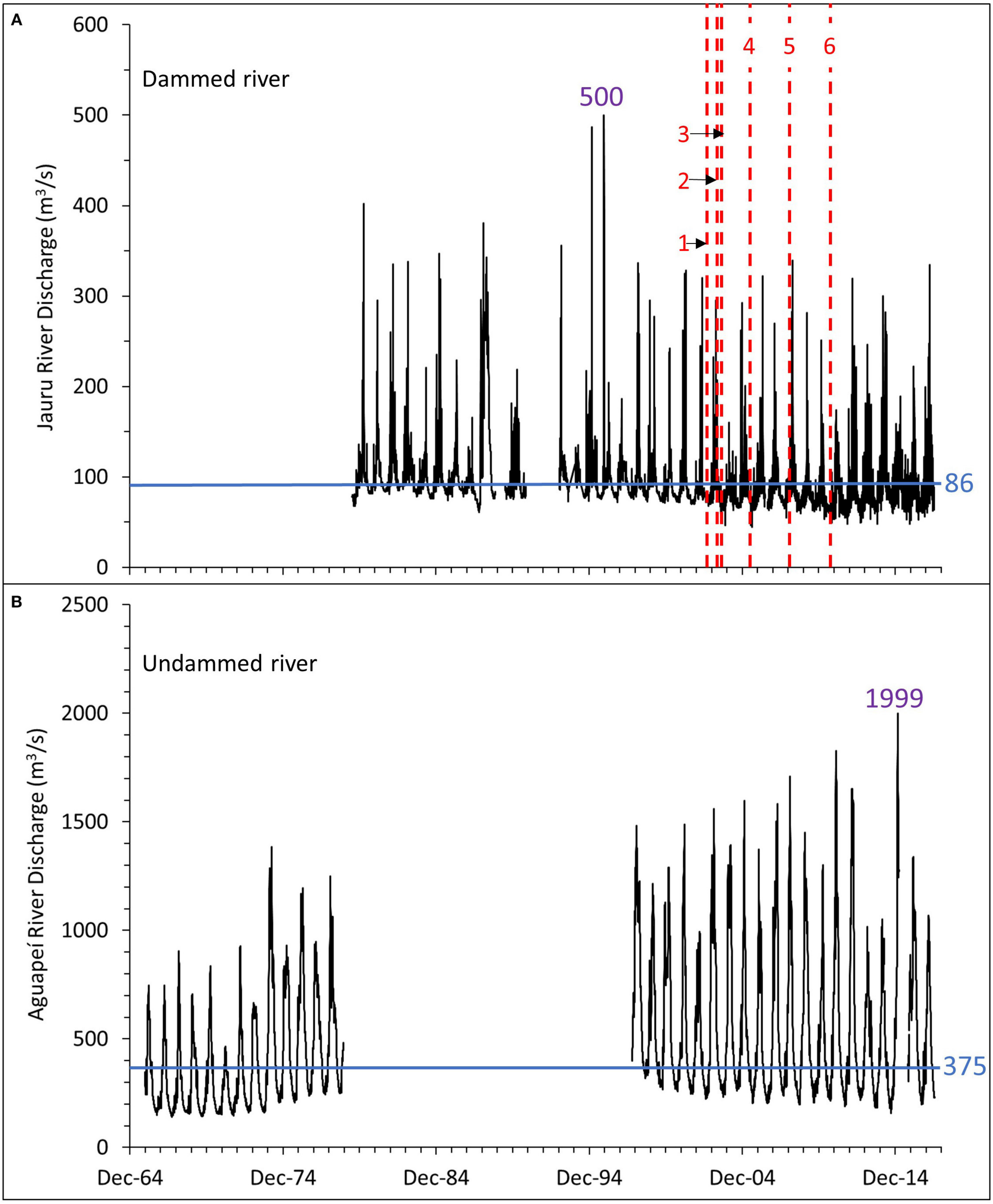

FIGURE 2 | Discharge hydrographs for the Jauru cascade of hydropower plants. Median flow (horizontal blue lines) for dammed and undammed river gages are also shown. Maximum flows are also shown in purple. **(A)** Hydrograph of the dammed Jauru River. The starting operation date for each hydropower plant is marked by a vertical red dash line and the name of the hydropower plant is written or numbered (see **Table 1** for details): 1 Antonio Brennand; 2 Jauru; 3 Indiavaí; 4 Ombreiras; 5 Salto; 6 Figueirópolis. **(B)** The undammed Jauru control river (Aguapei River).

Santana hydropower plant (**Supplementary Figure 1C**, ratios of 16 and 31).

Other dammed and undammed gage stations were less well-matched. At the Itiquira hydropower plant, the dammed river station (**Supplementary Figure 1F**), located in the down-basin reach (**Figure 1**), had a ratio of the amplitude to the median flow of 3 compared to 24 at its undammed control station in the upper mid-basin reach. At the São Tadeu hydropower plant (**Supplementary Figure 1D**) stations, these ratios were 5,3, and 28. The most extreme case was the São Lourenço hydropower plant (**Supplementary Figure 1G**) where these ratios were ∼7 (dammed river) and 143 (undammed river).

It was sometimes possible to detect clear temporal variations in the flow regime associated with the installation of the hydropower plants. For example, for the Jauru hydropower plants cascade (**Figure 2A**) the number of low pulses increased, and the corresponding HA parameter reached 537.5% after dam operation (**Table 4**). Low-flow pulses were also observed at the undammed river control station, but less frequently, and the corresponding HA was also lower (**Supplementary Table 1**). The progressive diminution in base flow, after the start of the Jauru hydropower plant operation, was observed in both dammed and undammed river hydrographs (**Figures 2A,B**). The decrease in the undammed river was steeper than in the dammed river. The base flow index on the undammed river was more than 5 times higher than on the dammed river (**Table 4**, **Supplementary Table 1**), indicating that the hydropower plant operation may not have been the cause of this decline.

For the Manso hydropower plant (**Supplementary Figure 1B**), there was an increase in base flow, as well as the diminution in the number of maximum annual flows, but no similar changes occurred on the undammed control river where there was a progressive decrease in baseflow. Changes in base flow were common on dammed and undammed river stations, both before and after hydropower plant installation. In the São Tadeu (**Supplementary Figure 1D**), there was an increase in baseflow around 2001, which preceded the operation of the São Tadeu I dam in 2009. Increasing and decreasing base flows were also observed at both the Juba dammed and undammed stations, as well as at the Itiquira and São Lourenço dammed stations, before and after the beginning of the hydropower plant operation.

Changes in Flow Regime in Dammed and Undammed Rivers

The unadjusted HA_{mean} and the contribution from each IHA parameter group for dammed and undammed river stations varied (**Figure 3**). Unadjusted HA_{mean} of dammed rivers varied from 61.6% for the Poxoréo hydropower plant to 17.6% for the São Lourenço basin hydropower plants. Hydrological alteration for the undammed control rivers was generally lower than in the regulated rivers, varying from 32.5% at the São Lourenço arrangement to 12% at the Juba cascade. For example, HA_{mean} on the dammed Poxoréo station was more than three times larger than at its undammed control station. The alteration was about 2 times higher for the Manso hydropower plant, 1.5 times higher for the Jauru and Juba hydropower plants, and 1.3 times higher for the São Tadeu hydropower plant. For the Itiquira hydropower plants, dammed, and undammed HA_{mean} were roughly equal. Unexpectedly for the São Lourenço and Santana hydropower plants, HA_{mean} of the undammed rivers was larger than for dammed rivers. Overall, hydrological alterations in the frequency and duration of high and low pulses (group 4 parameters) and the rate and frequency of hydrologic changes (group 5 parameters) constituted more than half of the observed HA_{mean} in both dammed and undammed rivers, with the exception for the Manso hydropower plant, where group 4 and 5 hydrological alterations were about 40% of the total dammed river HA_{mean}.

Changes in Flow Regime in Dammed Relative to Undammed Rivers

Widespread and significant alterations of the flow regime occurred across many rivers in the Upper Paraguay River Basin before and after the initiation of hydropower operations (**Table 4**). On dammed rivers, the IHA parameters that most consistently differed from their undammed counterparts were August median discharge, the 1-day minimum flow, and the low pulse count, each with 5 occurrences. Conversely, the April median discharge, the 30-day maximum, and the low pulse duration were never found to be different in dammed and their undammed river controls (**Table 4**).

The Poxoréo hydropower plant, Jauru River hydroelectric cascade, and Manso hydropower plant had the largest changes, with adjusted HA_{mean} of 43.8, 34.9, and 26.1% respectively (**Figure 4**). In these systems 16, 19 and 14 parameters were different from their undammed controls, respectively. For the other hydropower plants, the number of parameters different from their undammed controls varied from 1 to 10 (**Table 4**). The smallest adjusted HA_{mean} values were for the Itiquira and São Lourenço River complexes: 8.0 and 4.9%, respectively, and only five and one of 33 IHA indicators differed from their undammed rivers controls (**Table 4**).

The adjusted mean hydrologic alteration of the frequency and duration of high and low pulses (group 4 parameters) and the rate and frequency of hydrologic changes (group 5) were generally highest across all systems (**Figure 4**). However, dramatic changes were also observed in the timing of annual extreme flow conditions (group 3) at the Manso hydropower plant and for the magnitude of monthly flow conditions (group 1) at Manso and Poxoréo. The average adjusted HA_{mean} for frequency and duration of high and low pulses was 55.8%, while the average adjusted HA_{mean} rate and frequency of hydrologic changes was 22.5%. In both cases, the highest values of HA_{mean} for groups 4 and 5 were in the Jauru system, with values of 152.1 and 95.7%, respectively.

For group 4 (frequency and duration of high and low pulses), changes to the low and high pulse counts drove the largest variation in all but two hydropower plants (**Table 4**). Differences in low pulse count were highest for the Jauru (537.5%) and Juba (166.7%) series of hydropower plants. In Poxoréo and São Lourenço hydropower plants, the high pulse durations increased by 322.2 and 158.3%, respectively. In group 5 (rate and frequency

TABLE 4 | Hydrologic alteration in % of the IHA parameters of the hydropower plants as calculated from equation 1.

	Parameters	HA Poxoreu	HA Jauru	HA Manso	HA Santana	HA São Tadeu	HA Juba	HA Itiquira	HA São Lourenço
Magnitude of monthly flow conditions	October	***39.3***	***−8.9***	***61.8***	−36.1	16.2	−8.6	−4.1	−7.6
	November	***98.6***	***−17.1***	***46.5***	***−31.8***	−1.1	−5.0	−9.4	−13.9
	December	***89.4***	***−14.2***	−12.7	−23.3	−23.9	−9.3	−14.8	−26.3
	January	***101.9***	***−14.1***	−26.4	−19.4	−21.5	***−22.4***	−24.2	−19.0
	February	***119.7***	−4.3	−19.1	−2.3	4.1	−10.6	−9.0	−9.1
	March	***155.5***	***−11.7***	−33.4	10.9	−1.3	−2.7	−21.4	−6.3
	April	79.5	−8.9	−18.3	3.2	19.1	−0.2	−10.5	6.9
	May	***70.5***	***−16.2***	10.5	−10.5	2.3	***−13.3***	−10.3	−12.0
	June	***68.6***	***−13.5***	***41.4***	−12.6	12.1	***−15.7***	−6.7	−13.6
	July	36.9	***−12.2***	***62.3***	−22.5	21.8	−7.0	−5.9	−7.5
	August	***54.4***	***−14.4***	***78.0***	***−44.8***	***32.4***	−6.9	−2.4	4.1
	September	29.9	***−11.6***	***80.0***	***−44.2***	***37.4***	−5.8	−10.1	10.2
	Average Group 1	66.5	11.2	30.8	10.1	5.8	1.1	0.0	0.0
Magnitude and duration of annual extreme flows conditions	1-day minimum	23.9	***−28.9***	***66.5***	***−44.2***	***55.7***	***−11.0***	−11.2	−16.8
	3-day minimum	15.4	***−20.0***	***63.8***	***−45.0***	***56.3***	−9.1	−11.0	−18.7
	7-day minimum	45.9	***−13.8***	***68.2***	***−47.3***	***56.7***	***−9.2***	−10.8	−16.5
	30-day minimum	48.5	***−12.5***	***68.3***	***−49.4***	***39.3***	−8.2	−8.5	−13.6
	90-day minimum	39.2	***−11.9***	***67.4***	−33.6	***30.0***	−7.6	−6.8	−5.2
	1-day maximum	***−52.5***	−6.6	−50.5	15.4	0.0	5.2	−11.6	12.6
	3-day maximum	***−45.7***	−15.2	−44.2	6.1	0.1	12.2	−11.6	16.0
	7-day maximum	***−33.2***	−7.3	−32.5	2.0	1.2	14.0	−11.3	−0.6
	30-day maximum	18.3	−7.4	−12.7	−4.3	2.3	3.1	−13.9	6.0
	90–day maximum	***45.0***	−2.6	−21.4	0.4	8.0	−6.5	−15.5	0.9
	Numb. Zero days	0	0	0	0	0	0	0	0
	Base flow index	−7.5	***−4.4***	***70.0***	***−38.7***	***53.5***	−2.4	7.9	−9.9
	Average Group 2	16.0	7.9	36.7	16.9	26.5	1.8	0.0	0.0
Timing of annual extreme flow conditions	Date of minimum	−27.3	5.5	***−61.8***	−3.0	2.5	−1.6	2.2	10.1
	Date of maximum	7.7	−3.6	9.8	***21.6***	−3.3	***13.4***	−9.6	3.6
	Average Group 3	0.0	0.0	30.9	10.8	0.0	6.7	0.0	0.0
Frequency and duration of high and low pulses	Low pulse count	−87.5	***537.5***	−100.0	***100.0***	***−100.0***	***166.7***	***100.0***	−66.7
	L. pulse duration	33.3	−16.7	−25.0	−14.3	0.0	−25.0	−35.0	145.0
	High pulse count	***−52.4***	***70.8***	−44.4	28.6	25.0	***77.8***	***100.0***	0.0
	H. pulse duration	***322.2***	−30.0	0.0	−42.9	1.4	0.0	−37.5	***158.3***
	Average Group 4	93.6	152.1	0.0	25.0	25.0	61.1	50.0	39.6
Rate and frequency of hydrologic changes	Rise rate	***−54.1***	***119.9***	−72.4	−25.7	−13.4	15.9	***−16.5***	−51.1
	Fall rate	−19.4	***−91.2***	47.1	−11.0	***16.3***	***−27.8***	***20.7***	21.4
	N. of reversals	−47.4	***75.9***	***−19.2***	***62.5***	23.0	***38.0***	***17.1***	−0.4
	Average Group 5	18.0	95.7	0.0	20.8	5.4	21.9	18.1	0.0
	Adjusted HA_{mean}[a]	43.8	34.9	26.1	15.3	14.9	11.2	8.0	4.9
	Unadjusted HA_{mean}[b]	61.6	38.4	44.9	26.8	21.3	17.6	18.4	22.2

Bold values: statistically significant, but did not differ from undammed rivers. Bold and italic values: statistically significant and different from undammed rivers. The averages for each group of parameters were calculated using HA_i in the step-wise screening algorithm: all those that were not in bold and italics were treated as 0. [a]*HA_{mean} calculated with adjusted HA_i.* [b]*HA_{mean} calculated with unadjusted HA_i.*

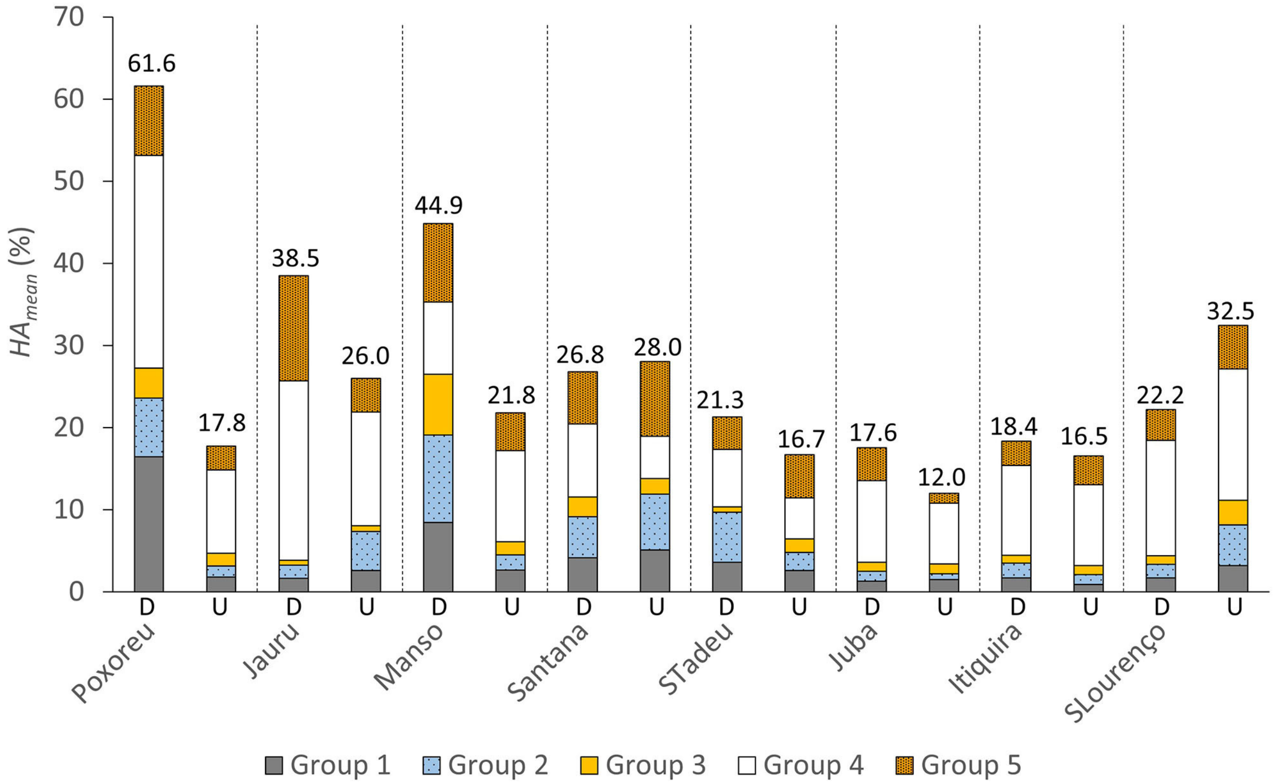

FIGURE 3 | Comparative mean hydrological alteration (HA_{mean}) for dammed and undammed rivers. HA_{mean} was calculated using unadjusted HA_i. The contribution of each parameter group to the overall HA is also illustrated. Dash lines separate each pair of dammed and undammed rivers. In each pair, the left bar column is HA_{mean} for the hydropower plants dammed river and the right one is the HA_{mean} at the control undammed river. The values of HA_{mean} are indicated. Group 1: magnitude of monthly flow conditions; group 2: Magnitude and duration of annual extreme flow conditions; group 3: Timing of annual extreme flow conditions; group 4: Frequency and duration of high and low pulses; group 5: Rate and frequency of hydrologic changes.

of water condition changes), rise and fall rates were the largest observed change between pre- and post-operation.

Since some HA_i were set to zero to perform the calculation of HA_{mean} with the step-wise screening algorithm, it is by definition lower than the mean hydrological alteration calculated without it, as can be seen by comparing the adjusted HA_{mean} of **Figure 4** to the unadjusted HA_{mean} of **Figure 3** (for dammed rivers). Adjusted HA varied from 4.9 to 43.8%. In some cases, as for the Jauru hydropower plants, the adjusted HA_{mean} (34.9%) was similar to the unadjusted HA_{mean} (38.4%), while in others like the Manso hydropower plant (44.9 vs. 26.1%) or the São Lourenço arrangement (22.9 vs. 4.9%), the differences were quite large (**Table 4**).

In dammed rivers, of the 256 calculated HA_i, 88 (34.4%) were found to be significant (**Supplementary Table 2**). However, when HA_i was significant, it was almost always (83 times out of 88) found to be different than in undammed rivers (**Table 4**). The number of significant HA_i on undammed control rivers was 57 (22.3%), but only 27 (10.5%) of the HA_i were significant at both the dammed and undammed stations (**Supplementary Tables 1**, **2**). Of these 27, 16 changes occurred in opposite directions, indicating that these were different. Of the remaining, 11 occurrences where the same parameter was significant and varied in the same direction, only in 6 cases was the ratio of the HA of the dammed river to the undammed river >1.25. These results indicate that the vast majority of significantly altered parameters in dammed rivers differed from those of undammed rivers. The HA_{mean} value adjusted by the step-wise algorithm devised in this study was able to capture these changes and is therefore useful for identifying flow-alteration effects from hydropower plants, even where the HA of the undammed control river was equally (or more) altered compared to the dammed river (e.g., São Lourenço, Santana, Itiquira; 3 out of the 8 studied hydropower plants).

The adjusted HA provided a different view on the severity of impacts than with the unadjusted HA (**Figures 3**, **4**). Using the unadjusted HA, the impact ranking from highest to lowest was: Poxoreu, Manso, Jauru, Santana, São Lourenço, São Tadeu, Itiquira, and Juba. Using the adjusted HA, the impact ranking from highest to lowest was: Poxoreu, Jauru, Manso, Santana, São Tadeu, Juba, Itiquira, and São Lourenço.

As with the unadjusted HA, adjusted HA of parameters related to the frequency and duration of high and low pulses (group 4) and rate and frequency of hydrologic changes (group 5) were usually the largest, except in the Manso case where they were absent. Furthermore, with the adjusted HA, alterations of the

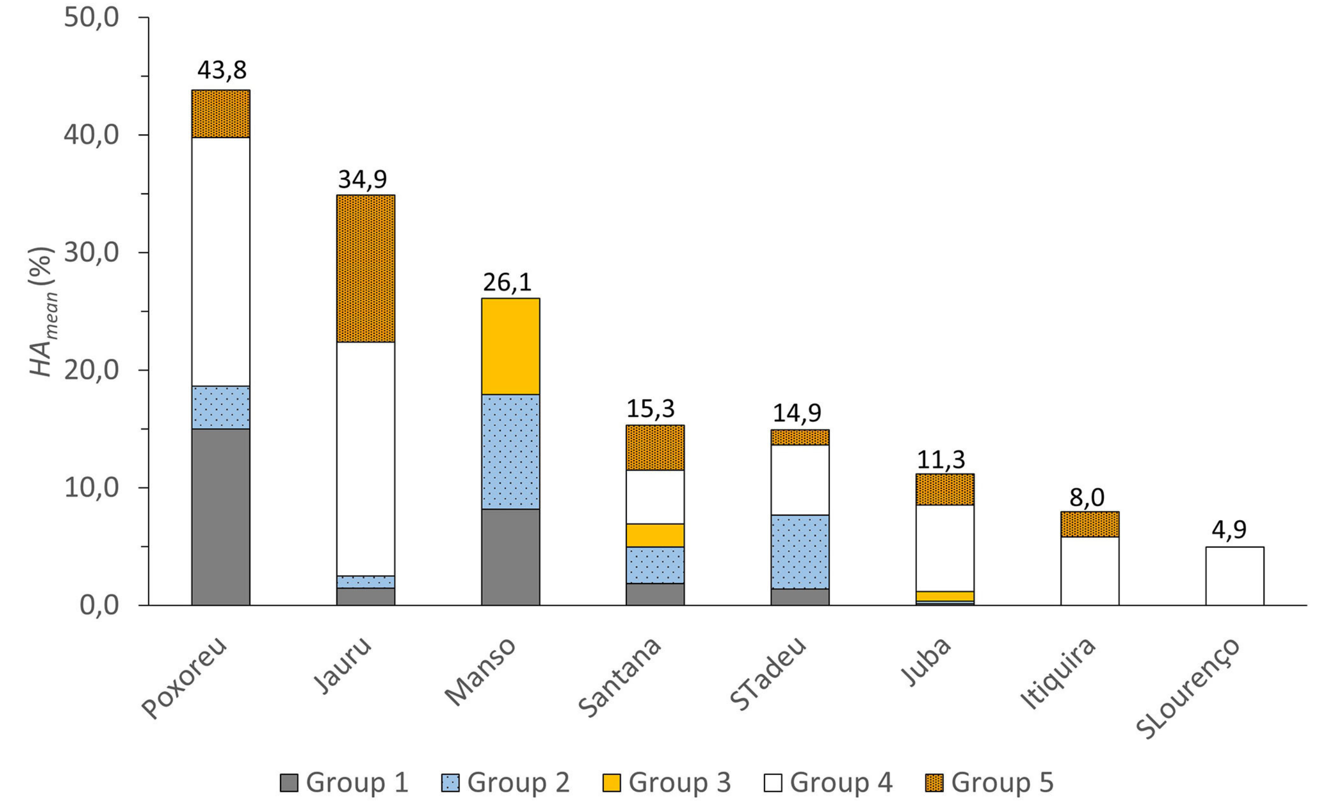

FIGURE 4 | Adjusted mean hydrologic alteration (*HA_{mean}*) of the studied hydropower systems. *HA_{mean}* was calculated using adjusted *HA_i* obtained from the step-wise screening algorithm described in section Hydropower Plants, Dammed, and Undammed Rivers Gage Stations. The contribution of each parameter group to the overall HA is also illustrated. Values of *HA_{mean}* are indicated. Group 1: magnitude of monthly flow conditions; group 2: Magnitude and duration of annual extreme flow conditions; group 3: Timing of annual extreme flow conditions; group 4: Frequency and duration of high and low pulses; group 5: Rate and frequency of hydrologic changes.

timing of annual extreme flow conditions (group 3) are only seen in Manso, Santana, and Juba. For the Itiquira hydropower plants, the adjusted HA showed that these dams would only influence parameters of groups 4 and 5, while in the São Lourenço case, only parameters of group 4 would be affected.

At the parameter level, substantial differences also existed between unadjusted and adjusted HA. For example, in the Poxoreu group 4, low pulse count did not contribute to the adjusted HA even when its magnitude was comparable to high pulse count. The same can be said of the 1- and 3-day maximum in Manso group 2, the number of reversals in São Tadeu group 5, the 7-day maximum in Juba group 2, and the low pulse duration of São Lourenço group 5 (**Supplementary Table 1**).

DISCUSSION

Hydrological Alterations in Dammed and Undammed Rivers

From what precedes it is clear that many hydrological alterations in this study were not only dam-induced but that other drivers of change modified the regimes of the studied rivers. However, the analysis revealed that hydrologic alterations likely due to changes in land use, irrigation, and climate in the Upper Paraguay River Basin were different than those provoked by hydropower plants, as significantly altered parameters in undammed rivers were different from those in dammed rivers.

The hydrologic alterations found in this paper were consistent with those found in other studies that used IHA to evaluate the variation in the river flow regime caused by the operation of hydropower plants. There was an HA of 56.3% for the Porto Primavera hydropower plants in the adjacent Paraná River basin (Rocha, 2010). In the Brazilian Amazon region, mean HA varied between 8 and 108% (Timpe and Kaplan, 2017), including a mean HA of 29% for the Jauru hydropower plants cascade (38.4% in this study), 62% for the Manso Dam (44.9% in this study), 21% for the Juba hydropower plants cascade (17.6%), and 18% for the Itiquira hydropower plants arrangement (18.4%). The different time windows for pre- and post-dam operation, and sometimes different monitoring stations probably explains these differences.

As in other studies, the frequency and duration of high and low pulses (group 4 parameters) and the rate and frequency of hydrologic changes (group 5) were often the most affected elements of the hydrologic regime. For example, Zhang et al. (2016) reported greater variations for the IHA parameters of groups 4 and 5, especially for the number and duration of low pulses, in studies in southwest China. Richter et al. (1996) also observed the greatest alteration to elements of the frequency and duration elements of the flow regime captured by IHA groups 4 and 5. Timpe and Kaplan (2017) observed that for all of the 33 hydropower plants they evaluated in Amazonian rivers, the hydrological alteration was also generally highest for groups 4 and 5. This was related to the capacity of flow regulation by hydropower plants which reduces extreme flows (maximum and minimum) to maximize the energy generation; significantly altering the rates and durations of the maximum and minimum peaks. Richter et al. (1996) reinforce this thesis, showing that the operation of hydropower plants severely affects the behavior of river flow pulses, increasing the number of occurrences and reducing the duration of these pulses (i.e., group 4) and increasing the rates of rise and fall in the flow records (i.e., group 5).

Hydrological alterations were observed in all undammed rivers, with HA_{mean} ranging from 12.0 to 32.5%, and 22.3% of the parameters significantly different between post and pre-operation periods. However, HA_{mean} for the undammed rivers controlling the Santana and the São Lourenço arrangement was larger than HA_{mean} of their corresponding dammed rivers (**Figure 3**). Furthermore, in the undammed controls for the Manso and Itiquira, more hydrological parameters were significantly altered than their respective dammed rivers (**Supplementary Table 2**). In the Upper Paraguay River Basin, the replacement of native vegetation with pastures and crops is ubiquitous in the basin. It is estimated that deforestation of 15% of the native vegetation in the floodplain area and 60% of the upper plateau has occurred in the Upper Paraguay River Basin (World Wide Fund for Nature – WWF, 2015). Existing studies on the conversion of natural soils to agriculture indicate resulting modifications to the hydrological regime. These modifications include changes in many important hydrologic parameters, including mean runoff and sediment concentration (Tucci, 2002; Rocha, 2010; Nobrega, 2014), siltation of river beds (Galdino et al., 2002), and interannual variability of minimum flows (Rocha, 2010; Rocha and Tommaselli, 2012).

Elsewhere in the world, changes in land use, irrigation, and climate have altered hydrology. For example, on the Cimarron River in Oklahoma, a variety of land use and cover changes changed a historically flashy river to a more stable river. There, HA parameters related to the magnitude of monthly flow conditions (group 1 parameters), the timing of annual extreme flow conditions (group 3), and frequency and duration of high and low pulses (group 4) were more pronounced (Dale et al., 2015). In the Mediterranean, water abstraction for irrigation purposes strongly affected the flow regime in irrigated catchments. The parameters related to the timing of annual extreme flow conditions (group 3), frequency and duration of high and low pulses (group 4), and rate and frequency of hydrologic changes (group 5) were strongly impacted (Stefanidis et al., 2016). In the Geba catchment, Ethiopia, the expansion of agricultural and grazing land at the expense of natural vegetation increased almost all hydrological parameters from 1972 to 2014 (Gebremicael et al., 2019).

In the Upper Paraguay River Basin, the largest hydrological alterations at control stations were observed for parameters related to frequency and duration of high and low pulses (group 4) and rate and frequency of hydrologic changes (group 5), whose combined relative importance varies from 50 to 84% of the total HA_{mean} (**Figure 3**). As for the magnitude of monthly flow conditions (group 1), the magnitude and duration of annual extreme flow conditions (group 2), and the timing of annual extreme flow conditions (group 3), HA_{mean} of undammed rivers were either larger (Jauru, Santana, São Lourenço) or quite similar (Juba and Itiquira) compared to those of dammed rivers (**Figure 3**).

Impacts of Hydrological Alterations in the Pantanal Ecosystems Functions

For most of the studied dammed (**Figure 4**) and undammed rivers (**Figure 3**), except for the case of Manso, the larger hydrological impacts were related mainly to the frequency and duration of high and low pulses (group 4 parameters) and to a lesser extent to the rate and frequency of hydrologic changes (group 5). According to Richter (1996 – **Table 3**), these hydrological impacts would generally change the stress level for terrestrial plants, the availability of habitat for aquatic organisms, water birds and other terrestrial animals, the desiccation stress for stream edge organisms, amongst others (**Table 3**). For the pantaneiros, these may result in harsher conditions for the pasture sustaining cattle farming and lower fish catches (Schulz et al., 2019). In the case of the Manso dam, the alteration of the magnitude of monthly flow conditions (group 1), the magnitude and duration of annual extreme flows conditions (group 2), and the timing of annual extreme flow conditions (group 3) of the Manso River would result in changes in habitat availability for aquatic organisms, changes on oxygen levels in the water column, dehydration in animals, change the duration of stressful conditions in aquatic environments, change the access to special habitats during reproduction or to avoid predation and disturb spawning cues for migratory fishes.

Many studies relating the floodplain ecosystem functions to the hydrological regime link flood pulse and aquatic organisms (Petrere et al., 2002; Bailly et al., 2008; Costa and Mateus, 2009; Lourenço et al., 2012; Pinho and Marini, 2012; Ziober et al., 2012; Scanferla and Súarez, 2016; Barzotto and Mateus, 2017; Penha et al., 2017; Tondato et al., 2018; Pereira and Súarez, 2019; Santana et al., 2019). However, in most of these studies, the hydrological conditions were not sufficiently detailed to make direct correspondence with the observed HA in the upper plateau rivers feeding the Pantanal. In some cases, however, it is possible to make inferences between the upper plateau river regimes and the ecological functions in the floodplain. For example, Wantzen et al. (2016) viewed the most severe environmental problems currently threatening floodplain invertebrates as alteration of the natural rhythm of the flood pulse. This change affects all flood pulse-adapted species, not only invertebrates, which lose their habitats when the frequency and duration of high or low pulse (group 4 parameters) are altered as we observed in all dammed rivers except for the Manso case (**Table 3, Figure 4**).

The intensity, frequency, and amplitude of the flood and drought phases also can modify connectivity and affect the biodiversity of the benthic assemblages in the Pantanal (Marchese et al., 2005). The long duration of hydrological connectivity acts to unify effects on the physical and biological characteristics of neighboring water bodies in river-floodplain systems. Isolating or drying water bodies, which occurs with long droughts, reduce benthic diversity. Similarly, Catella and Petrere (1996) found that the floodplain lakes function as a dry season feeding ground for small-sized species of fish, which are potential prey for the more highly valued larger species of fish. The number and connectivity of these lakes can be linked to the magnitude and duration of annual extreme flow conditions as well as to the frequency and duration of high low pulses. With the operation of the Manso dam, the peak of the drought (timing of annual extreme conditions) not only arrived sooner than before the dam but all parameters related to the magnitude and duration of annual minima increased (magnitude and duration of annual extreme flow conditions, **Table 4**).

Even though the dam impacts on fish and fisheries in the Pantanal are not still clearly visible or demonstrated, in the nearly located Paraná River basin, such impacts are now well-established. There, dam construction started in the 1980s, and large dams are more numerous than in the Upper Paraguay River Basin. Their impacts on river regimes have reduced the extent and duration of flood events, limiting the reproductive processes of several fish species (Agostinho et al., 2004) and impacting fish populations (Agostinho et al., 2004, 2007, 2016).

Drawbacks and Usefulness

The step-wise screening algorithm presented here retains a specific HA parameter based on several conditions. The "removal" of land use and climate-induced changes in hydrology is, in the end, not a complete removal. The hydrologic alteration factor (HA) of the dammed river is only conserved if significant. The HA of the dammed river is further considered if, in comparison, the HA in the undammed river (1) is not significant, (2) is of the opposite direction as in the dammed river, or (3) if the HA of the dammed river is at least 25% larger than in the undammed river. In the latter case, the HA due to land uses or climate change could still contribute to the "overall" HA in the dammed river. However, in this specific study, this is unlikely to substantially change our findings. Of the 83 HA retained for the calculation of the adjusted HA_{mean}, only 6 (7.2%) were conserved because of this last criterion.

The definition of pre and post-impact period to assess the HA of other drivers of changes at the undammed control river is one main weakness of the proposed method. For example, the undammed Jorigue River, due to its location, was used as a control for both the Poxoréo and São Lourenço hydropower plants (**Table 2**). When the Jorigue River station was used as the Poxoréo control, the year defining the pre/post-operation period was 1998, yielding HA_{mean} of 17.8%. When it was used as São Lourenço control, the defining year was 2008, and HA_{mean} was 32.5% (**Figure 3**). Not only was the overall HA larger, but the relative contributions of the parameter groups also varied. When the undammed Jorigue River was used as a control for the Poxoreu dammed river, HA related to the magnitude and duration of annual extreme flow conditions (group 2) was the least important. When used as a control for the São Lourenço hydropower plants, the HA of group 2 was third in importance, almost as much as the rate and frequency of hydrologic changes (group 5) (**Figure 3**, **Supplementary Table 1**).

Dams are built relatively quickly unlike increasing irrigation, land use, and climate change which occur progressively over much longer periods that could be equivalent to the temporal length of the fluviometric time series used to assess the HA. For the Jorigue River, the larger HA found when using it as the São Lourenço control might have reflected the fact that the length of record of the post-impact period was relatively short (10 years from 2008 to 2018) and prone to the influence of stochastic events such as large floods that were infrequent during this time span (2008–2018), while relatively frequent before (**Figure 2A**). Alternatively, it might capture better the fact that irrigation in the Cerrado region increased dramatically after 2000 (Martins and Santos, 2017). There is no simple alternative to choosing the start of operation date to assess HA from other drivers of change. An alternative would be to analyze independent of stationarity, such as proposed by Valle and Kaplan (2019), who suggested using a Gaussian Copula model to predict the counterfactual in the presence of substantial data gaps through the integration of data from multiple sources. These models have been widely applied in hydrology to quantify the association between multiple hydrological variables, such as drought duration, affected area, and severity, annual maxima of stream flows or rainfalls, and to predict associations among climate and flows.

Finally, the choice of the undammed river gage station is linked to the availability of data. As noted in section Hydropower Plants, Dammed and Undammed Rivers Gage Stations, these gages were selected so that dammed and dammed river regimes would be similar, and that hydrological variations due to geology, geomorphology, and topography would be reduced. However, hydrologic changes due to these factors, namely gage positions in the drainage basin, hydrological variations in geology, geomorphology, and topography, could not be controlled and were not quantified.

Despite these drawbacks, using the proposed step-wise algorithm to assess HA has several advantages for the management of dams and the conservation of the Pantanal. It selects only dam-induced alteration and reduces the number of parameters that management would have to focus on to diminish HA. For example, in the Manso case, the efforts to reduce impacts to the Cuiabá River flow would have to concentrate on parameters related to the magnitude of monthly flow conditions, the magnitude, and duration of annual extreme flows conditions, and the timing of annual extreme flow conditions since no parameters related to the frequency and duration of high and low pulses nor the rate and frequency of hydrologic changes significantly differed from their undammed river control. Specifically, median flows of June to October significantly differed from their control and these could be targets for restituting a more natural flow to the Cuiabá River and its flooding regime in the Pantanal as already proposed by Zeilhofer and Moura (2009). Regarding flow extremes, operation targets would only be on minima as no maxima significantly differed from their control. Regarding timing, dam operation affected the date of the minimum flow but not the maximum, so operation targets could be set in direction of the pre-operation period minimum date.

Since most of the studied dams (21 out of 24) were part of cascades or arrangements, reducing impacts might not be as straightforward as in the single Manso hydropower plant. However, the HA analysis reveals crucial differences in the most-impacted parameters that ought to be understood to better manage actual hydropower plants or the implementation of future ones. For example, the bulk of the impact in the Jauru, Juba, Itiquira, and São Lourenço was related to the frequency and duration of high and low pulses (**Figure 4**). However, in the São Lourenço case, it was only the high-pulse duration which was affected, while for the other three hydropower plants, the low and high pulse counts were most impacted (**Supplementary Table 1**).

In this paper, we refined the widely used IHA method to adjust for the effects of other hydrologic drivers such as land-use and climate change to estimate the impacts due to damming. These impacts differ from those of other drivers. It is compelling to assert what are effectively the alterations provoked by the operation of hydropower plants on river systems to make the right decisions to diminish these impacts. Further research is needed to assess hydrological alterations relevant to different drivers of change. In the Pantanal region, where the number of hydropower plants is growing fast, this is essential to strike a balance between the benefits of hydropower and its impacts on fluvial ecosystem services.

DATA AVAILABILITY STATEMENT

Publicly available datasets were analyzed in this study. This data can be found at: http://www.snirh.gov.br/hidroweb/.

AUTHOR CONTRIBUTIONS

PE and PG conceived the idea and curated and organized the data. PE, PG, and IF-C discussed the idea. PE ran the IHA software and wrote the original manuscript draft. PG, PE, HT, IF-C, and DK discussed the results. IF-C did the project administration. PG supervised the manuscript. PE, IF-C, HT, PG, and DK reviewed and edited the writing. All authors contributed to the article and approved the submitted version.

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SUPPLEMENTARY MATERIAL

The Supplementary Material for this article can be found online at: https://www.frontiersin.org/articles/10.3389/fenvs.2020.579031/full#supplementary-material

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Hydropeaking by Small Hydropower Facilities Affects Flow Regimes on Tributaries to the Pantanal Wetland of Brazil

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Hydroelectric facilities often release water at variable rates over the day to match electricity demand, resulting in short-term variability in downstream discharge and water levels. This sub-daily variability, known as hydropeaking, has mostly been studied at large facilities. The ongoing global proliferation of small hydropower (SHP) facilities, which in Brazil are defined as having installed capacities between 5 and 30 MW, raises the question of how these facilities may alter downstream flow regimes by hydropeaking. This study examines the individual and cumulative effects of hydropower facilities on tributaries in the upland watershed of the Pantanal, a vast floodplain wetland system located on the upper Paraguay River, mostly in Brazil. Simultaneous hourly discharge measurements from publicly available reference and downstream gage stations were analyzed for 11 reaches containing 24 hydropower facilities. Most of the facilities are SHPs and half are run-of-river designs, often with diversion channels (headraces). Comparison of daily data over an annual period, summarized by indicators of hydrological alteration (HA) that describe the magnitude, frequency, rate of change, and duration of flows, revealed differences at sub-daily scales attributable to hydropeaking by the hydropower facilities. Results showed statistically significant sub-daily HA in all 11 reaches containing hydropower facilities in all months. Discharge indicators that showed the highest percentage of days with increased variability were the mean rates of rise and fall, amplitude, duration of high pulses, maximum discharge, and number of reversals. Those that showed higher percentages of decreased variability included minimum discharge, number of high pulses, duration of stability, and number of low pulses. There was no correlation between HA values and physical characteristics of rivers or hydropower facilities (including installed capacity), and reaches with multiple facilities did not differ in HA from those with single facilities. This study demonstrates that

SHPs as well as larger hydropower facilities cause hydrological alterations attributable to hydropeaking. Considering the rapid expansion of SHPs in tropical river systems, there is an urgent need to understand whether the ecological impacts of hydropeaking documented in temperate biomes also apply to these systems.

Keywords: hydroelectricity, dams, load following, tropical, hydrology, index of hydrological alteration

INTRODUCTION

Small hydropower (SHP) facilities are the most common kind of hydroelectric dams being built around the world, and although they are generally viewed as less environmentally harmful than larger dams, there has been little research to support that assertion, particularly in tropical and subtropical regions where most new SHPs are being constructed (Mbaka and Mwaniki, 2015; Couto and Olden, 2018). Reflecting the widespread assumption that SHPs have lower environmental and social impacts than larger dams, many countries have enacted policies that promote SHPs, including less stringent environmental impact assessments. In Brazil and many other countries, multiple SHPs may be distributed in series along river systems, raising concerns about their cumulative effects on rivers and downstream ecosystems (Kibler and Tullos, 2013; Athayde et al., 2019).

Even dams that are small may affect channel morphology, sediment transport, and deposition (Baker et al., 2011; Olden, 2016; Couto and Olden, 2018). Where significant impoundments exist relative to the size of the stream, artificially warm or cold water released downstream can negatively affect the aquatic biota (Zaidel et al., 2021). 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 (Ovidio and Philippart, 2002; Santucci et al., 2005; Pompeu et al., 2012; Couto et al., 2021), and passage through turbines can harm or kill larval and adult fishes, shrimp, and other aquatic animals (DuBois and Gloss, 1993; Benstead et al., 1999).

A well-known effect of larger hydroelectric dams is the release of water at variable rates over the course of the day (i.e., sub-daily) to accommodate variation in electricity demand, and the resultant short-term variability in downstream velocity, discharge, and water levels is known as load following or hydropeaking (Bejarano et al., 2017). The ecological impacts of hydropeaking have mostly been studied at large facilities in North America and Europe, where mitigation measures have been designed to protect against stranding of fishes and maintain minimum flows to avoid desiccation of fish eggs (Moreira et al., 2019).

The ongoing global proliferation of small hydropower (SHP) facilities, which in Brazil are defined as having installed capacities between 5 and 30 MW [Aneel (Agência Nacional De Energia Elétrica), 2016], raises the question of how downstream flow regimes may be altered. Hydrological effects of SHPs are of particular concern in the upland watershed of the Pantanal, a world-renowned floodplain wetland system located mostly in Brazil. While the effects of hydropeaking are unlikely to extend into the floodplains due to longitudinal attenuation (Collischonn et al., 2019), the unnatural sub-daily variability in river flow regimes in reaches downstream of SHP facilities could affect behavior and reproduction of fishes that migrate upstream from the Pantanal (Campos et al., 2020), in addition to resident fishes and other aquatic and riparian organisms.

Existing and proposed hydropower facilities in tributaries to the Pantanal are depicted in **Figure 1**. As of 2018 there were 47 hydropower facilities in operation (hereafter "current hydropower facilities"), the majority of which are SHPs, with an additional 138 projects under construction, planned, proposed, or identified by the government as prospective sites (hereafter "future hydropower facilities") (Agência Nacional de Águas, 2018). Most of these SHPs present diversion designs, where a low dam with a small or non-existent reservoir diverts river water into an artificial channel for as much as several km to a powerhouse farther down the river valley (Oliveira et al., 2020). The majority of the river discharge is normally diverted, leaving the natural channel with little as 10% of the discharge. The SHP designs that lack a large reservoir are "run-of-river" facilities inasmuch as they cannot alter discharge except on short time scales (Csiki and Rhoads, 2010; Kaunda et al., 2012). Many of the SHPs are located on lower-order rivers but some are on larger rivers with low elevational gradients.

In light of the ongoing construction and planning of future SHPs in the Pantanal watershed, there is an urgent need to understand how numerous SHPs on the tributaries may, in aggregate, alter the transport of water, sediments, and nutrients from the uplands into the Pantanal, and as well produce enough barriers to the upstream migration of fishes from the Pantanal to impede their reproduction and reduce their populations. In recognition of these needs, the present study is part of a multidisciplinary research program that has examined many dimensions of the issues surrounding hydroelectric facilities in the tributaries of the Pantanal, including hydrology (this study), sediment transport (Fantin-Cruz et al., 2020), water quality (Oliveira et al., 2020; Fantin da Cruz et al., 2021), and fish and fisheries (Campos et al., 2020; Ely et al., 2020). In this study, evidence for hydropeaking is evaluated based on discharge patterns in river gages downstream of 11 reaches containing a total of 24 hydropower facilities compared to simultaneous measurements at reference gages not influenced by the facilities. Comparison of hourly data, summarized by indicators of hydrological alteration, reveals differences at sub-daily scales that may be attributable to the hydropower facilities and aspects of their design. Accordingly, relationships between the observed hydrological alterations and the hydraulic and

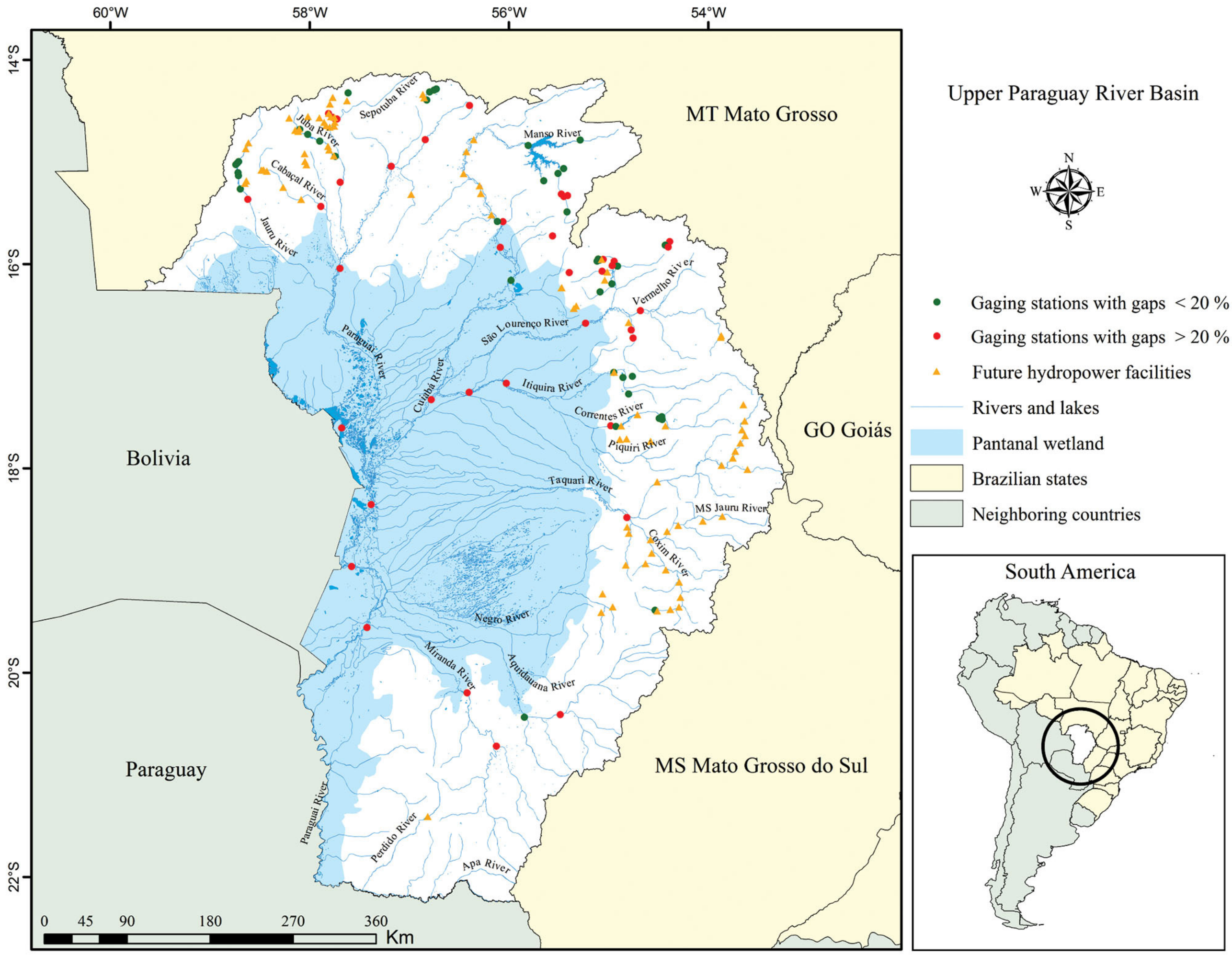

FIGURE 1 | Map of the Upper Paraguay River Basin showing the distribution of river gaging stations upstream and downstream of currently operating hydropower facilities, indicating the stations with data gaps of <20% that allowed upstream-downstream comparisons for 24 hydropower facilities. The map also shows future hydropower projects that are either under construction, planned, or identified as potential sites for hydropower development in the Pantanal watershed by either the Brazilian National Electric Energy Agency (ANEEL) or the state environmental agencies (depending on location).

hydrological characteristics of rivers and hydropower facilities were also examined. These included installed potential, mean discharge, watershed area, reservoir area, hydraulic residence time, diverted natural channel length, and dam design. The paper ends with recommendations on further research to better understand how hydropeaking by small hydropower facilities may affect the aquatic biota of downstream reaches.

MATERIALS AND METHODS

Study Site

This study examines rivers in the Brazilian portion of the uplands in the Upper Paraguay River basin that drain to the Pantanal wetland. The Pantanal lies mostly within Brazil, and drains southward via the Paraguay River. The uplands (150–1,400 m a.s.l.), which represent 59% of the basin area and lie mainly to the east and north of the Pantanal, include sloping terrain favoring rapid runoff and high sediment production. The Pantanal floodplains (80–150 m a.s.l.) are subject to extensive seasonal inundation by overflow of river inflows originating in the uplands as well as delayed drainage of local rainfall (Hamilton et al., 1996). According to the Köppen-Geiger climate classification, the climate of the region is tropical savanna, with average annual precipitation in the uplands ranging from 1,200 to 1,800 mm. About 80% of the annual rainfall occurs in the rainy season from October to April (Gonçalves et al., 2011).

The native vegetation in the uplands is Cerrado savanna, but extensive areas are now converted to cropland (29% of the upland watershed area analyzed in this study) or pasture (22%). Human population density in the rural municipalities is low with mostly <10 inhabitants km^{-2}. Cuiabá city and its environs, situated along the Cuiabá River <50 km upstream of the Pantanal, is the largest urban area, which together with three other medium-sized cities located in the uplands has about 1,260,000 inhabitants.

TABLE 1 | Locations and characteristics of the hydropower facilities and river discharge gage stations.

Multiple (cascade) or isolated facility	Facility	River	Installed potential (MW)	Mean discharge (m^3/s)	Watershed area (km^2)	Reservoir area (km^2)	Hydraulic residence time (days)	Diverted natural channel (km)	Design	Reference gage station	Downstream gage station	Year
Cascade Jauru	Antônio Brennand	Jauru	21.96	46.5	1,590	0.05	0.1	0.99	RoR	66071355	66071470	2018
	Ombreiras	Jauru	26	60	2,207	2.91	9.5	0	RoR			
	Jauru	Jauru	121.5	85.4	2,620	2.62	2.7	0.95	Conv			
	Indiavaí	Jauru	28	70.1	2,320	0.22	0.3	0	RoR			
	Salto	Jauru	19	79.9	2,657	1.06	0.5	0.66	Conv			
	Figueiró-polis	Jauru	19.4	102	2,960	7.44	4	-	Conv			
Cascade Juba	Juba I	Juba	42	55.2	1,550	0.82	1	3.6	n/a	66051000	66052900	2018
	Juba II	Juba	42	61.25	1,808	2.5	1.8	2.4	n/a			
	Graça Brennand	Juba	27.4	77.9	1,974	5.92	9.6	0	RoR			
	Pampeana	Juba	28	80	2,503	4.17	5.8	1.2	RoR			
Cascade Ponte de Pedra	Eng. José Gelásio da Rocha	Ponte de Pedra	24.4	26.9	1,680	0.27	0.9	6.6	Conv	*	*	2018
	Rondonó-polis	Ponte de Pedra	26.6	28.62	1,733	0.02	0.1	2	Conv			
Cascade Santana	Diamante	Rio Santana	4.23	12.92	560	0.49	0.7	0	Conv	66005400	66005960	2018
	Santana I	Rio Santana	14.8	-	804	1.17	-	-	n/a			
Cascade Tenente Amaral	Sucupira	Saia Branca	4.5	11.02	356	0.07	0.3	1.5	Conv	66390090	66386000	2018
	Pequi	Saia Branca	6	10.22	327	0.02	0	2.6	Conv			
	Cambará	Tenente Amaral	3.6	9.97	332	0	0	1.3	Conv			
	Embaúba	Tenente Amaral	4.5	10.02	320	0.09	0.3	1.7	Conv			
Isolated	Maracanã	Córrego Maracanã	10.5	4.49	148.2	0.38	1.4	2.7	Conv	66051000	66025500	2017
Isolated	São Tadeu I	Aricá-Mirim	18	6.31	256	0.46	0.1	2.8	RoR	66162000	66260110	2017
Isolated	São Lourenço	São Lourenço	29.9	108	5,775	1,290	10.8	0	RoR	66450010	66400390	2018
Isolated	Santa Gabriela	Correntes	24	54.2	3,132	0.43	0.1	2.2	RoR	6648360	66484500	2018
Isolated	Itiquira	Itiquira	96.6	72.9	5,137	2.1	0.8	11	Conv	66522000	66525100	2018
Isolated	Ponte de Pedra	Correntes	176.1	80.7	4,000	14.5	15.9	12.7	RoR	66483600	66493000	2018

Design indicates run-of-river (RoR) or conventional (Conv) where conventional indicates capability to regulate discharge (n/a = facilities did not provide this information). Station numbers are from the Sistema Nacional de Informações sobre Recursos Hídricos do Brasil. Year refers to the period of analysis of hydrological data.

[*] *Information provided by the hydropower company.*

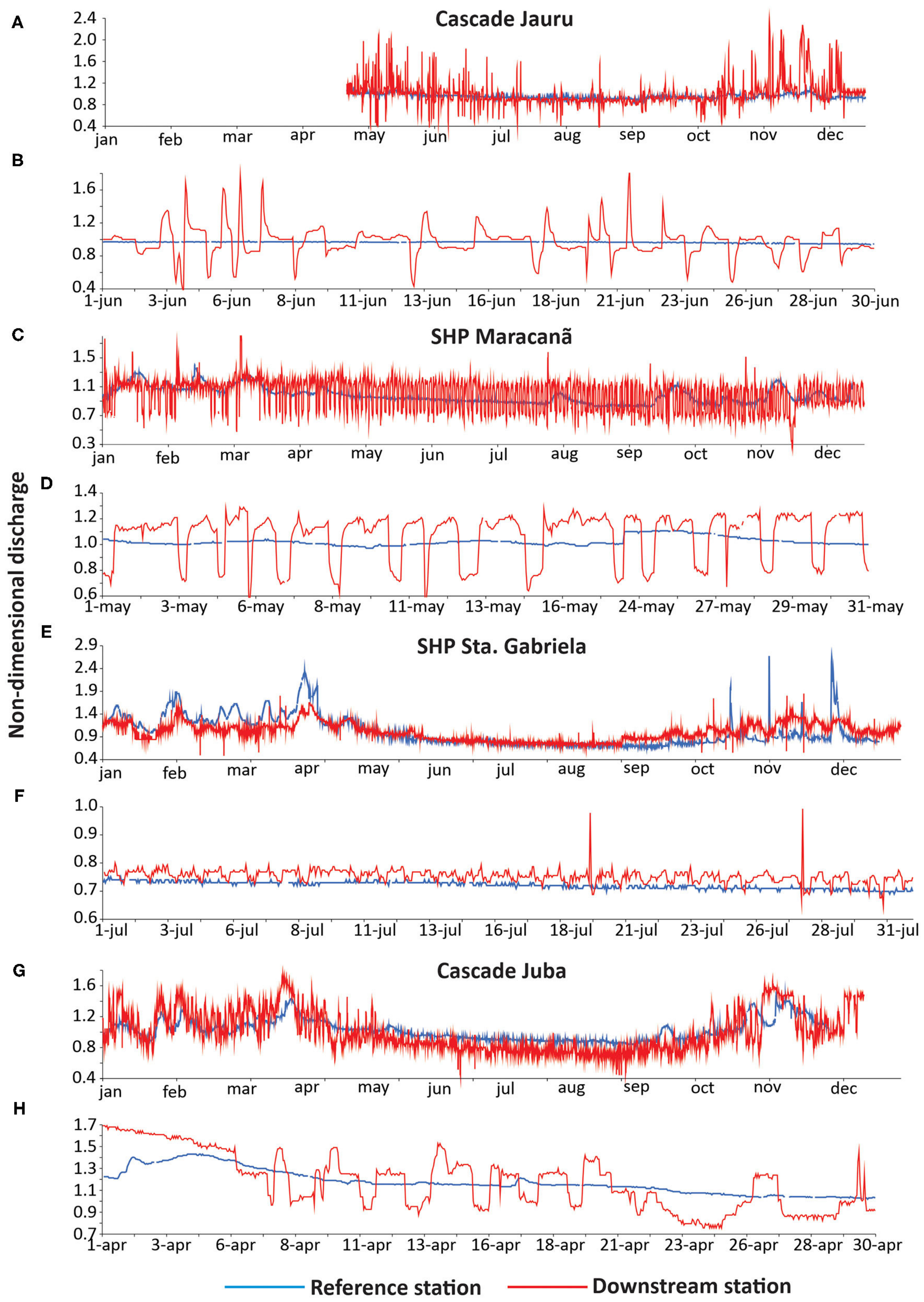

FIGURE 2 | Hydrographs showing examples of hydrological alteration by small hydropower (SHP) facilities during the 2018 calendar year (2017 for Maracanã) and in representative months of that year during the season of lower discharge. Red lines show discharge downstream of the hydropower facilities and blue lines show the reference discharge station. All discharge data are standardized to the mean annual discharge to facilitate comparisons among rivers. **(A,B)** Cascade Jauru on the Jauru River in Mato Grosso State; **(C,D)** SHP Maracanã on the Córrego Maracanã (a tributary of the Sepotuba River); **(E,F)** SHP Santa Gabriela on the Correntes River; and **(G,H)** Cascade Juba on the Juba River.

TABLE 2 | Indicators of hydrological alteration at the sub-daily scale.

Component of the flow regime	Parameter (Abbreviation)	Units	Description
Magnitude	Minimum discharge (Qmin)	–	Daily median discharge standardized to the annual mean of minimum hourly discharge
	Maximum discharge (Qmax)	–	Daily median discharge standardized to the annual mean of maximum hourly discharge
	Amplitude (Qamp)	–	Daily median discharge standardized to the annual mean of the difference between maximum and minimum hourly discharges
Frequency of pulses	Number of high pulses (Nhp)	Nhp/day	Median daily number of times that the discharge is above the 3rd quartile at the reference site
	Number of low pulses (Nlp)	Nlp/day	Median daily number of times that the discharge is below the 1st quartile at the reference site
Rate of change	Mean rate of rise (RrQ)	–	Mean rate of daily rise in discharge
	Mean rate of fall (RfQ)	–	Mean rate of daily fall in discharge
	Number of reversals (Nrev)	Nrev/day	Median daily number of times that the sign of change in discharge reversed over the day
Duration	Duration of stable discharge (Dsta)	Hours/day	Median daily duration of stable discharge
	Duration of high pulses (Dhp)	Hours/day	Median daily duration of high pulses
	Duration of low pulses (Dpb)	Hours/day	Median daily duration of low pulses

Study Reaches, Data Sources, and Processing

The study region has data for 108 gaging stations with sub-daily measurements, of which 80 have rating curves to estimate discharge from stage and the remainder recorded only stage with no discharge measurements, and were installed at dams. Of the 80 stations with discharge data, 40 had sufficiently complete records for our analysis (i.e., gaps amounting to <20% of the year) and met our quality checks (**Figure 1**). These stations permitted upstream-downstream comparisons for the 24 hydropower facilities whose characteristics are shown in **Table 1**. Six of the 24 hydropower facilities were bounded by gaging stations, whereas the other 18 are sequentially arranged within 5 reaches, in which case we call them "cascades," and thus we evaluate 11 reaches in this study. The watersheds above the 24 hydropower facilities range in area from 148 to 5,775 km^2, and the rivers range in long-term mean discharge from 4.5 to 108 $m^3\ s^{-1}$.

Most of these hydropower facilities can be considered small, although five have installed capacities above the Brazilian government's regulatory definition of small hydropower as <30 MW installed capacity, and two of those exceed 100 MW. Two of those that exceed 30 MW (Juba I and II, 42 MW each) have dams and reservoirs similar in size to the SHPs. One of the SHPs (São Lourenço, 29 MW) creates a reservoir comparable in size to larger facilities such as the largest one studied here, Ponte de Pedra (176 MW). Thus, installed capacity is an imperfect indicator of the potential environmental effects of these facilities (Couto and Olden, 2018). Hence, we analyze the SHPs and larger facilities together in this study to examine similarities and differences in their downstream hydrological effects.

All facilities have dams that form reservoirs, which range in area from 0.01 to 14.5 km^2, in volume from 0.035 to 111 hm^3, and in hydraulic residence time from 0.1 to 15.9 days. Twelve of the 24 are run-of-river designs, nine have the capacity to regulate discharge (labeled as conventional in **Table 1**), and information on design was unavailable for the other three. Most (17) of the facilities divert water from the natural channel into headraces for distances ranging from 0.66 to 12.7 km.

Figure 1 shows the distribution of existing and future hydropower facilities as well as available river gage stations considered in this study. Discharge data were downloaded from a public portal called the *Sistema Nacional de Informações sobre Recursos Hídricos do Brasil* (http://www.snirh.gov.br/hidrotelemetria). We used only stations with high-frequency measurements (i.e., every hour or more often). Discharge time series were screened for gaps, defined as either zero discharge (none of these rivers are intermittent) or missing date, time, and/or discharge data within the temporal sequence. For cases with missing discharge data, the sequential dates and times were added to enable us to estimate the percentage of missing data. Each discharge time series was inspected for outliers that were obviously unrealistic, as well as for abrupt changes that might reflect equipment problems, and in these cases the suspect data were replaced with gaps. For further analysis we selected only time series with gaps amounting to <20% of the total times, and gaps were excluded from statistical summaries. We analyzed data from 2018 where possible, though in some cases we had to use 2017 data because data gaps in 2018 amounted to >20%.

We analyzed discharge time series where stations existed both upstream and downstream of one or more hydropower facilities, which in many cases were measurements made by the hydropower companies as required for environmental compliance. Only stations with discharge data, as opposed to just water level as is often measured at the dams, were selected. Hereafter we use the term reference in place of upstream because not all cases presented a gaging station immediately upstream of the hydropower facility. In some cases we had to use a reference station well upstream, but not downstream of other hydroelectric facilities, and in one case we

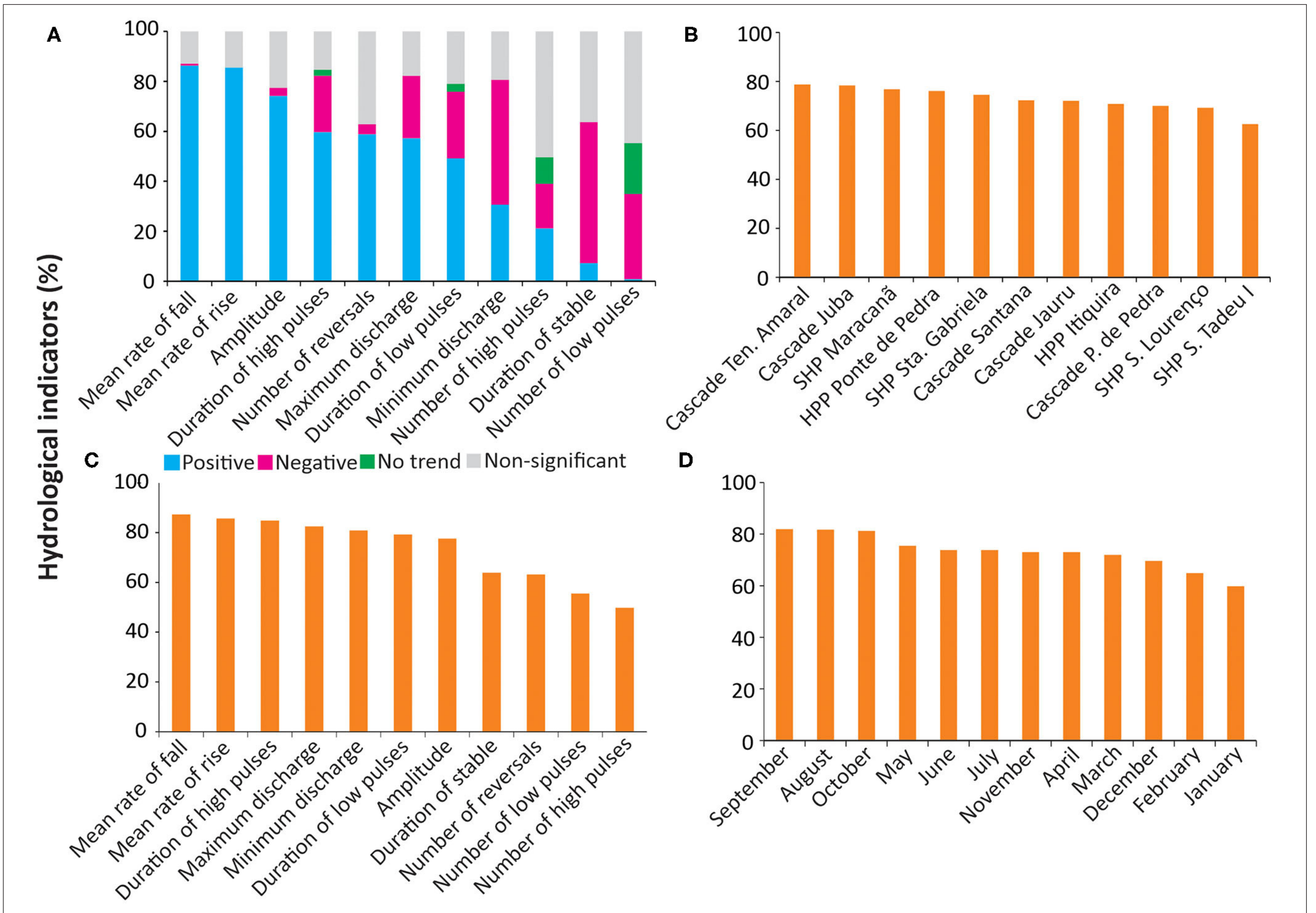

FIGURE 3 | Summary of the sub-daily indicators of hydrological alteration (HA): **(A)** Relative proportions of the monthly median values of HA showing significant positive values (increasing), negative (decreasing), no trend (indistinguishable medians) but with statistically significant differences in data distributions, and non-significant monthly medians; **(B)** Percentages of the hydrological indicators that showed statistically significant differences based on comparisons of daily values of each month, arranged by hydropower facility; **(C)** the same arranged by the indicators in **Table 2**; and **(D)** the same arranged by month.

had to use a station on a downstream tributary with similar watershed features and discharge (the Vermelho River below the SHP São Lourenço). Discharge data were standardized to the mean annual discharge to facilitate comparisons across river reaches of different discharge rates (Bejarano et al., 2017). This standardization assumes that there is a fixed proportionality between flows at the reference site and downstream flows in a particular river reach.

Indicators of Flow Regime Alteration

We calculated sub-daily flow regime metrics from discharge data at 1-h intervals, based on the methods of Greimel et al. (2016), Timpe and Kaplan (2017), and Bejarano et al. (2017). These methods adapt the widely used Indicators of Hydrological Alteration approach (IHA; Richter et al., 1996) to produce sub-daily Indicators of Hydrological Alteration including 11 indicators that describe the magnitude, frequency, rate of change, and duration of flows. The indicators were calculated at daily time scales based on pairwise comparisons of temporally matched data from the reference site and downstream of hydropower facilities (**Table 2**).

For a particular hydrological indicator, the difference between the reference site and downstream of the hydropower facility was considered significant in a particular month only when the monthlong series of daily values showed statistically significant differences based on the non-parametric Mann-Whitney U-test ($\alpha \leq 0.05$).

For each of the indicators in **Table 2** that showed significant differences between the reference and downstream sites in a particular month, we evaluated the hydrological alteration (HA) attributable to the hydropower facilities following the method described by Timpe and Kaplan (2017):

$$HA\ (\%) = \left(\frac{(Mdown - Mref)}{Mdown}\right) \times 100 \tag{1}$$

where HA is the median percent alteration in the indicator, M_{down} is the median daily value of the indicator downstream of the hydropower facility, and M_{ref} is the median daily value of

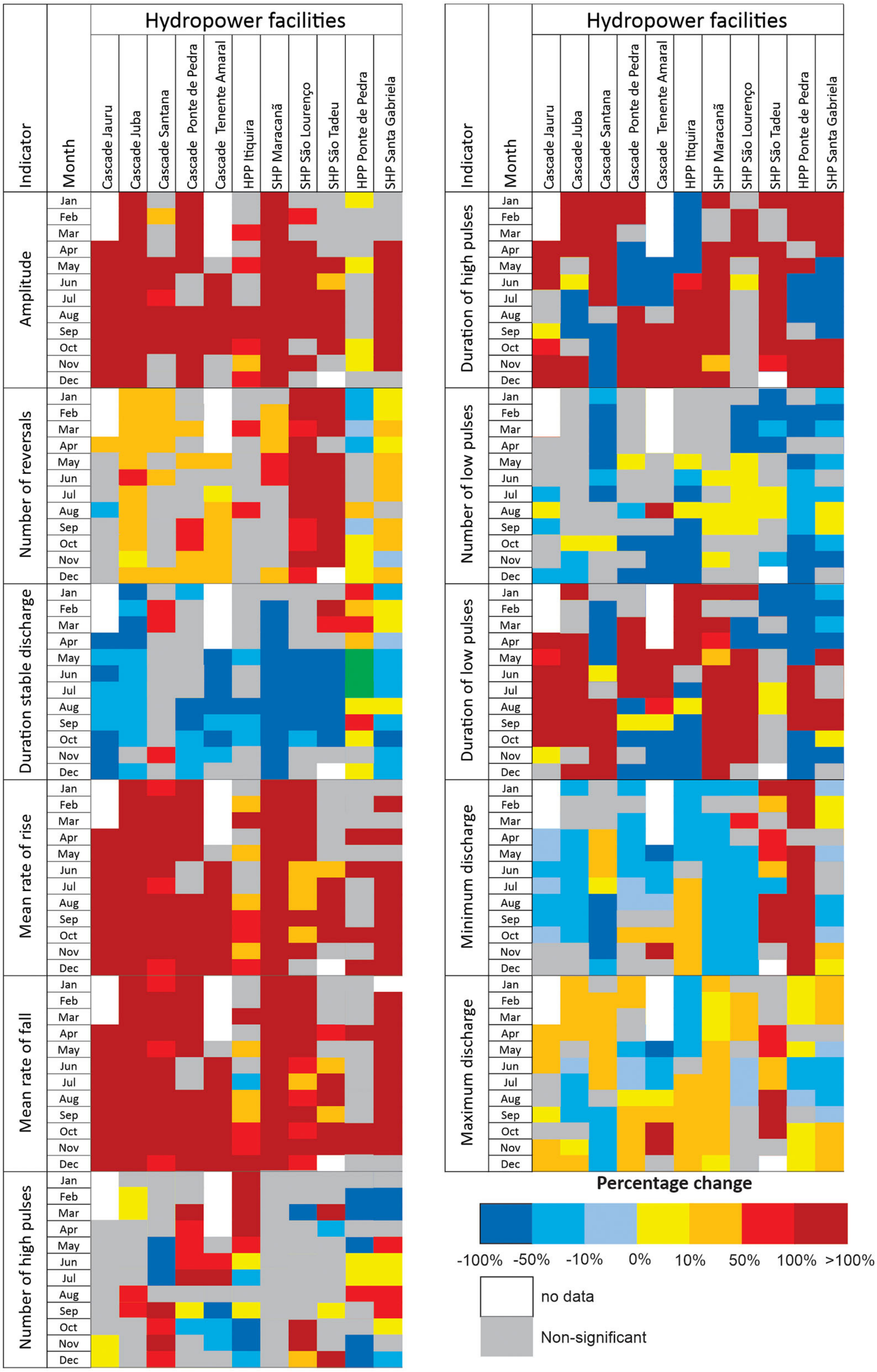

FIGURE 4 | Percentage change in sub-daily indicators of hydrological alteration for each study reach by month, showing only cases where there was a statistically significant difference (Mann-Whitney test) between the upstream reference site and the downstream site.

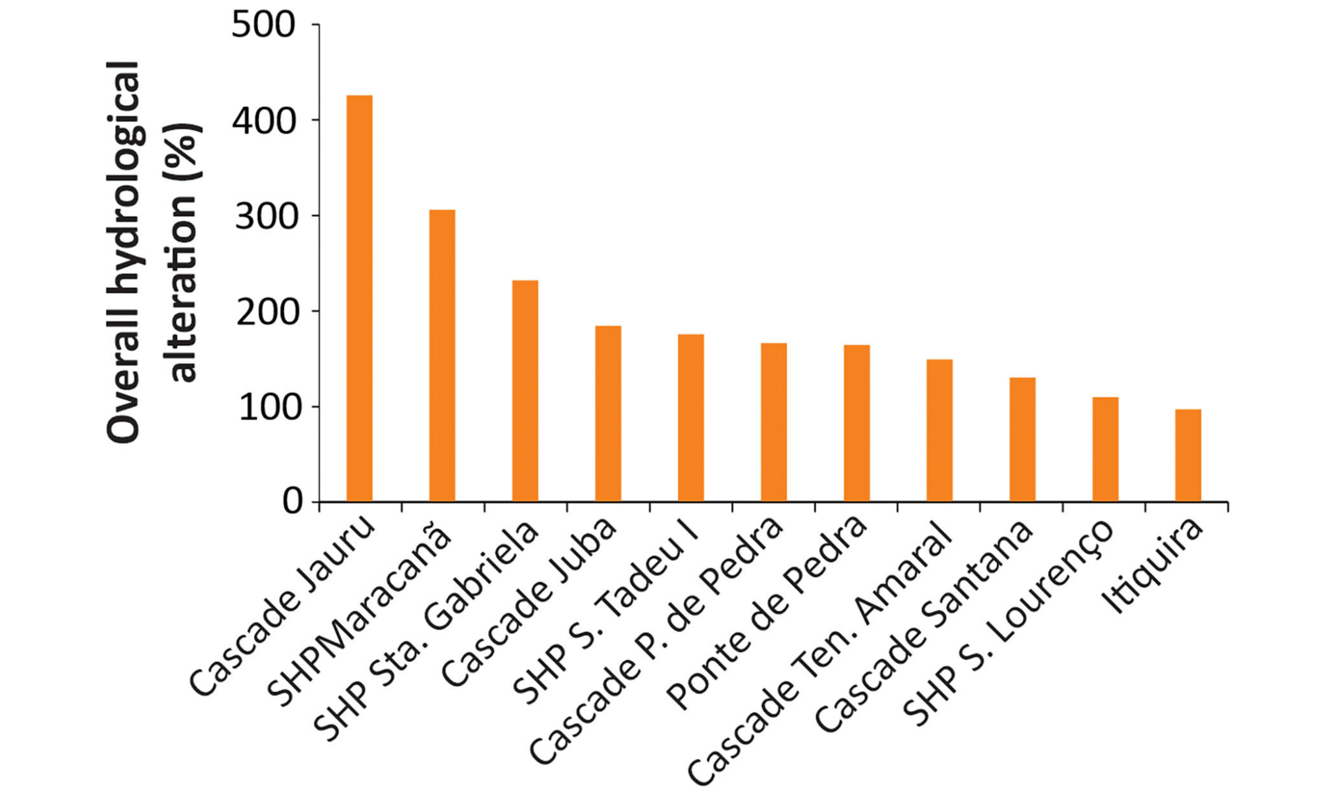

FIGURE 5 | Overall hydrological alteration (HA overall) across all indicators and months for each hydropower facility (in the five reaches containing more than one facility, the labels show the name of the facility that is closest to the downstream gaging station).

the indicator at the reference site. Equation (1) was computed at daily intervals, from which monthly medians of HA were determined. Significant positive values of HA indicate an increase in the indicator from the reference site to downstream, negative values indicate a decrease, and in some cases the medians of the distributions were equal but the Mann-Whitney test indicated significant differences in the distributions of daily values around the median.

To facilitate HA comparisons among reaches with hydropower facilities, we calculated the overall HA across all indicators and months for each hydropower facility (Timpe and Kaplan, 2017). We summed the absolute values of the monthly HA values that were statistically significant, then divided that sum by the total number of months with data (including months that did not show significant differences between the reference and downstream sites in a particular month, effectively counting them as zero HA values).

We evaluated the effects of hydraulic and hydrological characteristics at each hydropower facility (**Table 1**) on the monthly HA values as well as the overall HA using the Spearman's rank correlation coefficient ($\alpha \leq 0.05$). Where more than one hydropower facility existed between the upstream and downstream gage stations, we examined correlations by two alternative approaches—using just the characteristics of the most downstream hydropower facility or using the sum of characteristics of all of the facilities in the reach (except in the case of discharge). The Mann-Whitney test was then employed to determine whether the number of indicators with significant HA values as well as the sub-daily HA values differed between those two approaches ($\alpha \leq 0.05$).

RESULTS

Hydrological Alteration at Sub-daily Time Scales

Example hydrographs comparing reference and downstream stations for reaches with particularly marked HA show clear sub-daily variation imposed by the facilities (**Figure 2**). This variation occurs with a visible diel periodicity below the Santa Gabriela (**Figures 2E,F**) and Maracanã (**Figures 2C,D**) hydropower facilities, but is more irregular below the multiple facilities in the Jauru (**Figures 2A,B**) and Juba cascades (**Figures 2G,H**). These examples comparing hydrographs above and below reaches with the highest overall HA values reveal variable diel patterns of alteration ranging from irregular with high and low pulses of brief duration (Jauru and Juba cascades) to relatively regular with higher and lower periods lasting longer (SHP Maracanã and SHP Santa Gabriela), and these examples provide an indication of the magnitude of sub-daily variation that can be induced by the hydropower facilities (**Figure 2**). The magnitude of discharge variability would likely be accompanied

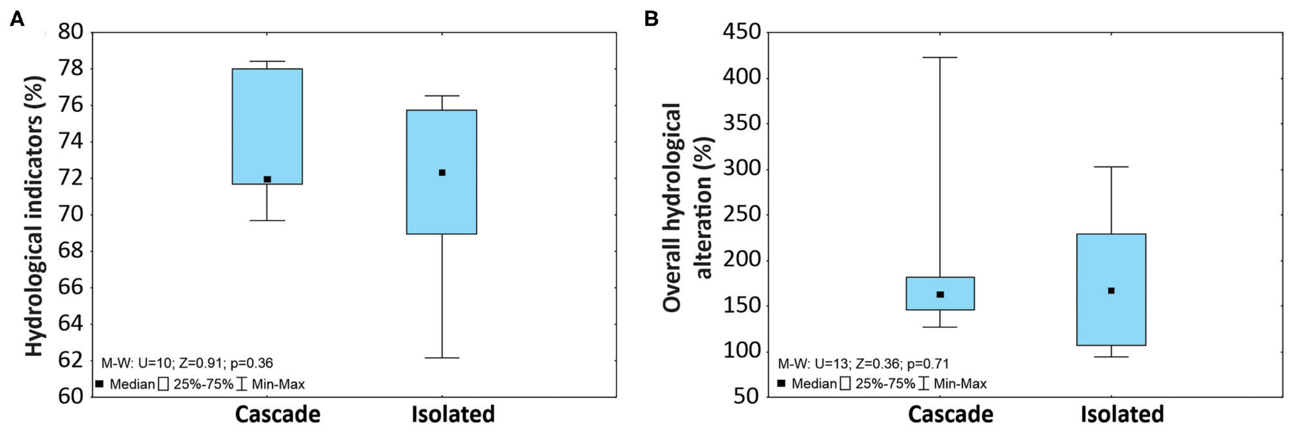

FIGURE 6 | Comparison between reaches containing more than one hydropower facility (cascades) with those containing a single facility (isolated) showing no significant differences in either **(A)** the percentage of significant indicators of hydrological alteration or **(B)** the overall HA.

by significant changes in the wetted area of channels downstream of these facilities, particularly during low discharge periods.

Analysis of 11 indicators of hydrological alteration in 11 reaches containing a total of 24 hydropower facilities, most of which are SHPs, provides clear evidence of sub-daily variability that can be attributed to hydropeaking by dam operations (**Figures 3**, **4**). Almost all of the indicators showed significant differences between gages at reference sites and gages downstream of the reaches in most months over the year of analysis. The greatest alterations involved the rates of change (rises and falls) in discharge and the magnitudes of minimum flows (often lower) and maximum flows (often higher). The duration of stable periods decreased in most cases. The hydrological alteration was most marked at the height of the dry season (Aug–Oct) but was apparent in all months. There were more than twice as many significantly positive values of monthly HA (and thus increased variability) than negative ones (49 vs. 22% overall) (**Figure 3A**). Discharge indicators that showed the highest percentage of increases (positive HA values) were the mean rates of rise and fall, amplitude, duration of high pulses, maximum discharge, and number of reversals. Those that showed higher percentages of decreases than increases include minimum discharge, number of high pulses, duration of stability, and number of low pulses.

The hydropower facilities that most strongly altered downstream hydrology were the Cascade Tenente Amaral (sum, 19.5 MW) and the Cascade Juba (sum, 139.4 MW), each located on rivers with those names. In both of these reaches, 78% of the paired comparisons of monthly indicators showed significant alteration based on the Mann-Whitney test (**Figure 3B**). The indicators that showed the most frequent alterations were the mean rates of rise and fall in discharge, with 91 and 89% showing significant alterations, respectively. Indicators that were least often significant include the numbers of high and low discharge pulses (58 and 51%, respectively) (**Figure 3C**). The highest percentages of statistically significant sub-daily hydrological alterations occurred during months of lower discharge and particularly from August through October, although the percentages exceeded 50% in all months (**Figure 3D**). The full temporal resolution of the data summarized in **Figure 3** is depicted in **Figure 4**.

Among the 11 reaches, the overall HA varied by >4-fold among the reaches analyzed (**Figure 5**). The greatest overall HA occurred in the Jauru River reach containing the six hydropower facilities (Jauru Cascade in **Table 1**: 423%), followed by the Maracanã (302%), Santa Gabriela (229%) and Juba Cascade (181%) reaches. The overall HA did not vary in rank order of installed capacity; the lowest overall HA values were found for two of the larger facilities in terms of installed capacity (Itiquira and SHP São Lourenço), whereas the highest overall HA was found for the Jauru cascade containing six facilities with one particularly large one.

Relationships Between Hydrological Alteration and Characteristics of Rivers and Facilities

The physical characteristics of rivers and facilities in the 11 reaches (**Table 1**) were not significantly correlated with the number of indicators that changed significantly between upstream and downstream (**Figure 3A**), nor with the monthly HA values (**Figures 3C**, **4**) (statistical results not shown). In addition, comparison between reaches containing more than one hydropower facility with those containing a single facility showed no significant differences in either the percentage of significant indicators of hydrological alteration (**Figure 6A**) or the overall HA (**Figure 6B**), and therefore no evidence for cumulative impacts.

DISCUSSION

Our results are consistent with earlier studies that evaluated hydrological alterations by subsets of these hydropower facilities. Timpe and Kaplan (2017) analyzed indicators of hydrological alteration at multiannual time scales below 33 hydropower facilities, 16 of which were SHPs, across the Amazon and Upper

Paraguay River basins, including several of the facilities we analyze here as well as the much larger reservoir created by the Manso Dam (212 MW) on the Cuiabá River. Although lowland hydropower facilities with the largest dams and reservoirs induced the greatest alterations, Timpe and Kaplan noted that for reaches containing single facilities that were either large dams or SHPs, the SHPs induced alterations similar in magnitude to the large dams when scaled to installed capacity (i.e., % HA per MW). Ely et al. (2020), in a multiyear analysis of indicators of hydrological alteration, found that high and low pulse counts as well as the number of reversals were the most frequent dam-induced impacts in the Upper Paraguay River basin.

Other studies have examined individual hydropower facilities in the upper Paraguay River basin. Braun-Cruz et al. (2021) reported evidence of hydropeaking by the Itiquira hydropower facility on the Itiquira River, which was included in the present study. The downstream hydrological effects of the much larger Manso Dam were recently evaluated in detail by Jardim et al. (2020). Fantin-Cruz et al. (2015) analyzed IHA at multiannual time scales attributable to the Ponte de Pedra hydropower facility, also one of our study sites and our largest facility in terms of installed capacity (176 MW). Seven indicators were significantly altered by Ponte de Pedra—magnitude of lowest monthly flow, minimum flows of 1, 3, and 7 days, maximum flow of 90 days, and counts of high and low pulses—and the reservoir released higher flows during the dry season.

Sub-daily hydrological alterations attributable to hydropeaking have been documented below hydropower facilities elsewhere throughout the world (Bejarano et al., 2017), though only in a few studies of modern SHPs (e.g., Lu et al., 2018; Xiao et al., 2019). Hydrological alterations by SHPs tend to occur over short time scales as the number of active turbines is increased during high demand in the day and reduced at night, particularly below run-of-river facilities with relatively small reservoir volumes that depend on managing discharge to meet short-term variation in electricity demand (Bevelhimer et al., 2015). Many of these SHPs are diversion designs in which most of the discharge is directed into a headrace leading to the powerhouse, and thus fluctuations in discharge through the turbines may cause opposite fluctuations in the diverted portion of the natural channel. In contrast, large dams and reservoirs tend to dampen seasonal variation in outflow discharge, releasing more water during the dry season and attenuating short-term discharge peaks resulting from precipitation or snowmelt (Magilligan and Nislow, 2005).

The ecological implications of hydropeaking for downstream ecosystems are little known for tropical rivers (Jumani et al., 2018), but have been studied in temperate zones with negative impacts increasingly demonstrated for riparian and aquatic plants (Bejarano et al., 2018), macroinvertebrates (Kennedy et al., 2016; Leitner et al., 2017; Schulting et al., 2019), and especially for fishes (Vollset et al., 2016; Boavida et al., 2017; Costa et al., 2019; Rocaspana et al., 2019; Vehanen et al., 2019). As a result of increasing awareness of these impacts, abrupt changes in water level and velocity associated with hydropeaking have received increasing regulatory attention (Hauer et al., 2017, Hayes et al., 2019, Moreira et al., 2019), particularly in rivers supporting important fisheries. In the case of the Itiquira hydropower facility in the upper Paraguay River, Braun-Cruz et al. (2021) provided circumstantial evidence that a fish kill involving stranding was linked to hydropeaking by the dam operations.

In conclusion, we have demonstrated that many SHPs, as well as somewhat larger hydropower facilities, cause hydrological alterations on sub-daily time scales attributable to hydropeaking to meet varying energy demand. In our data set, these hydrological alterations were not correlated with characteristics of the river reaches or the facilities. In addition to those variables, prediction of hydrological alterations caused by hydropeaking would likely require information on operating procedures at each facility, which was not available to us.

Considering the rapid expansion of small hydropower development in tropical environments (Couto and Olden, 2018), there is an urgent need to understand whether the ecological impacts of hydropeaking documented in temperate biomes also apply to these systems. This will be challenging because life cycles and behavior of the aquatic biota in tropical rivers in relation to river hydrology are less well-understood, and even migration routes of fishes that support socioeconomically valuable fisheries are incompletely known (Campos et al., 2020). It is obvious that the aquatic biota will likely be harmed by abrupt decreases in water levels causing stranding of fishes and other aquatic animals as well as the temporary emergence of aquatic substrata that would otherwise remain underwater. However, changes in depth and wetted area of the river channel as a result of hydropeaking depend on channel morphology (Moreira et al., 2019), information that is lacking for the rivers we study here, as it is for most other regions of the world where SHPs are proliferating. If negative impacts are revealed, research will be needed on the costs vs. benefits of mitigation of these changes by altering dam operations, as for example those proposed for the SHP Ponte de Pedra by Fantin-Cruz et al. (2015). In addition to mitigating impacts of existing SHPs, planning for new SHP locations and designs needs to consider how the resultant hydrological modifications may negatively affect migratory fishes and other aquatic animals, not only by producing barriers and directing most of the flow through turbines, but also altering downstream hydrology. Such planning should be conducted at the scale of entire river basins to minimize negative impacts on migratory populations (Couto and Olden, 2018; Lange et al., 2018; Couto et al., 2021).

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

AUTHOR CONTRIBUTIONS

This study was conceived and carried out by IF-C, PG, PZ, and SH. IF-C, GS, and RB developed the computational routine for the high frequency time series analysis. Field work and data analysis were conducted by IF-C, JF, EU, EM, and HT. The paper was written mainly by IF-C, JF, and SH. All authors contributed to the article and approved the submitted version.

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Stakeholder Perceptions on the Governance of Fisheries Systems Transformed by Hydroelectric Dam Development in the Madeira River, Brazil

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Hydroelectric dams often have significant impacts on freshwater fisheries. Major impacts are known to be driven by changes in river hydrology and fish ecology, but the role of governance arrangements in mitigating or exacerbating fisheries impacts from hydropower development is less understood. This study presents an analysis of stakeholder perceptions about the effects of hydroelectric dam implementation on fisheries governance arrangements in the Madeira River basin, Brazil. Semi-structured interviews were conducted with 50 stakeholders representing the fishers and fish traders, government, non-governmental organizations, and the private sector. Fishers, non-governmental, and private sector agents perceived hydropower development to be the strongest factor driving fisheries decline or change over the past 10 years, while government staff perceived overfishing to be an equally or even more important factor. Most stakeholders affirmed that fisheries governance arrangements have weakened in the face of hydropower development, and that these arrangements have been insufficient to effectively mitigate or compensate for negative impacts on fisheries. Fishers, non-governmental and private sector agents saw lack of opportunities to participate in fisheries governance as a major contributing factor, while government staff emphasized lack of qualified personnel, lack of trust between agencies, and control over the decision-making process held by hydropower companies. Perspectives on other implications of governance arrangements were shared across stakeholder groups. These included increased conflicts; lack of interaction and coordination between agencies; the fragility of fishers' social organization; lack of trust and reciprocity between organizations; and power imbalances between stakeholders. The results show that hydropower development impairs and changes relationships between diverse players involved in fisheries governance, which can exacerbate existing weaknesses and negatively affect fishery sustainability. Drawing from the perspectives and comments of the various stakeholders who participated in the study, we provide recommendations to improve freshwater fisheries governance in the Madeira River basin and in the Brazilian Amazon.

Keywords: mitigation, inland fisheries, institutional arrangements, hydroelectric dams, social-ecological impacts, freshwater fisheries, Amazon, fisheries governance

INTRODUCTION

Hydropower development is an electric energy supply strategy adopted by many Asian, Latin American, and African countries (Soares-Filho et al., 2006). Government leaders have pursued the implementation of these projects to meet their countries' growing electricity demand. They often highlight positive aspects of hydropower, including energy security, low carbon emissions, increased employment, and economic development (Prado et al., 2016; MME/EPE, 2017). The negative aspects of these projects on social, environmental, and economic dimensions are frequently underestimated or ignored by developers and decision-makers worldwide (Araújo and Moret, 2016; Siciliano et al., 2016; Moran et al., 2018; Athayde et al., 2019).

Construction of large hydroelectric projects triggers significant modifications in the physical-chemical dynamics of aquatic ecosystems and in the composition and abundance of the local ichthyofauna (Castello and Macedo, 2016; Winemiller et al., 2016). These changes in turn lead to significant socioeconomic impacts for riverine communities, where fishing represents an important source of animal protein and income (Agostinho et al., 1997; Fearnside, 2014; Doria et al., 2018b). Biophysical impacts of hydroelectric dams on fisheries can be mitigated by improving governance arrangements for managing fisheries, by improving the design and operation of hydropower dams, and by developing or strengthening broader public policies such as resettlement or welfare programs (WCD, 2000). Mitigation measures could include, for example, restrictions on fishing in the vicinity of fish passage facilities, requirements to maintain environmental flows, and compensation payments for lost income from fishing (WCD, 2000). Conversely, failure of governance arrangements to mitigate dam impacts on fisheries, or worse, deterioration of existing fisheries governance arrangements in the face of dam construction, can exacerbate the overall impact on fisheries. Therefore, it is essential to consider both the social and ecological dimensions of the fisheries system when assessing or planning to identify and mitigate the impact of dam projects (Lorenzen et al., 2007).

Previous studies have addressed the impacts of dams on fisheries resources, on fishing activities, and on riverine communities (Lima et al., 2012; Castello and Macedo, 2016; Winemiller et al., 2016; Arantes et al., 2019). However, there is a lack of research on potential social impacts of dams, integrating the perceptions of key stakeholders such as local fishers, private sector (Doria et al., 2018b), dam-builders, and other stakeholders interested in dam developments (Kircherr et al., 2016).

Fisheries system (FS) can be considered a type of social-ecological system (SSE) that includes the natural resources used, its users, the governance and management systems and how these interact and affect the SSE and its components (Ostrom, 2009). Diverse authors have studied SSEs in different settings and common property resources, with a focus on fishing resources (e.g., Imperial and Yandle, 2005; Lorenzen, 2008; Burns and Stöhr, 2011; Basurto et al., 2013; London et al., 2017; Yatim et al., 2018; Doria et al., 2020). A central challenge for understanding fisheries systems is to elucidate how governance arrangements might influence fishery resources and the sustainability of the system as a whole (Ostrom, 1990; Lorenzen, 2008; Burns and Stöhr, 2011). Here, we define fisheries governance as the public and/or private coordinating steering regulatory processes based on different stakeholders' behavior and formal and informal institutional arrangements (Burns and Stohr, 2011).

According to Ostrom (1990) and Berkes and Ross (2013), specific characteristics of governance systems may be typically associated with SSEs resilience and sustainability, such as the design and implementation of rules adapted to local needs and conditions. Where this is the case, stakeholders are able to participate in rule design and modification or have the right to formulate their own rules. Other characteristics of sustainable governance systems include diversified and innovative engaged governance (involving collaborative organizations); equity of participation and power among group's members; the affected group's social capital related to past experiences of cooperation; as well as strong leadership, which allows community action in response to internal and external influences and pressures on the fisheries system (Ostrom 1990; Berkes and Ross 2013).

This study aims to evaluate, from a stakeholder perspective (fishers, researchers, dam-builders, and government), how the implementation of hydroelectric dams may affect tropical fisheries systems through effects on their governance arrangements, and how these arrangements may influence the overall system sustainability. We present a case-study analysis of the Madeira fisheries system, where two large hydroelectric plants were built in 2011, contrasting our findings with other contexts and experiences, in Brazil and internationally.

Conceptual Framework

The current study was guided by a theoretical framework for analyzing fisheries systems governance (**Table 1**). This framework was developed based on the architecture of governance elements proposed by Burns and Stöhr, (2011): Social organizational configuration and cognitive-normative configuration. The social organizational configuration includes descriptors of the main stakeholders (actors) (D1) and their interactions (authority and responsibility; expertise and knowledge) (D2), their perceptions of dialogue among stakeholders (D3), and procedures for legitimate decision making (D4). The cognitive-normative configuration related to informal constraints (norms of behavior, conventions, and self-imposed codes of conduct) includes descriptors of the

TABLE 1 | Theoretical framework used to describe the Governance architecture of the Fisheries System and their sustainability, modified from Agrawal and Ostrom (2001), Burns and Stöhr (2011) and Berkers and Ross (2013).

	Descriptor	Sustainability indicator
Social organizational configuration	Main stakeholders (D.1) Interactions among the entities (D.2) Organizations have the: - Authority and responsibility (D.2.1) - Expertise and knowledge required over the problem (D.2.2) Actor's perception about Dialogue among the stakeholders (D.3) Procedures for legitimate decision-making (formal and informal) (D.4)	Social capital (e.g., experiences of cooperation) (SI.1) Existence of strong leadership (SI.2) Engaged governance (involving collaborative organizations; sharing information about the system or the process) (SI.3) Trust and reciprocity between the stakeholders (SI.4) Equity of participation and power (SI.5) Rules adapted to local rules; fishing rules have not changed or changed in an adaptive manner after the dam (SI.6)
Cognitive-normative configuration	Conceptualization of the situation: Key driver in the system (D.5) Goals and priorities which are expected to be applied in the policy-making and governing processes (D.6) Conflicts occurrence (D.7) Suggestions over the problem or to improve the fisheries systems and future perspectives (D.8)	Stakeholders able to participate (SI.7)

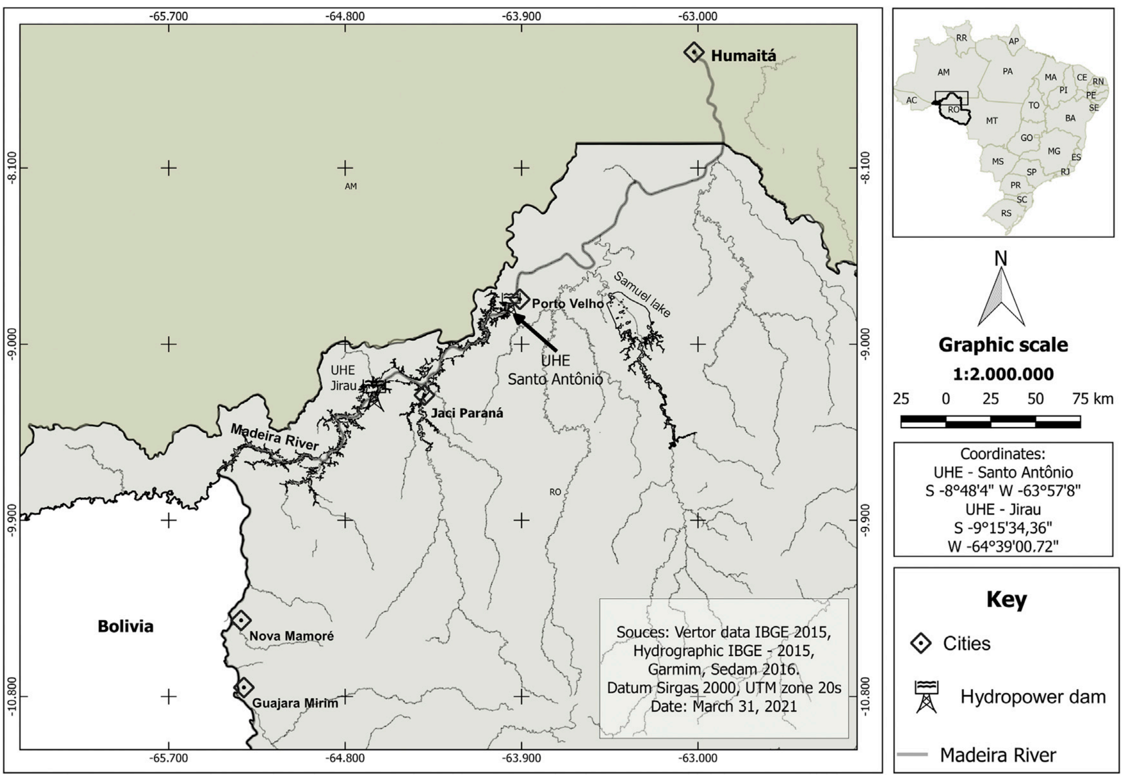

FIGURE 1 | Geographic location of the study area, showing the Madeira River crossing Rondônia State (Brazil), and the Jirau and Santo Antônio dams.

conceptualization of the situation and the main drivers (D5), the goals or priorities expected to be applied in the decision-making process (D6), the occurrence of conflicts (D7) and suggestions for addressing the problem (D8). Descriptors in the social-organizational configuration are related to a set of governance sustainability indicators: characteristics that have been shown in many studies to indicate the capacity of the stakeholders and institutions to respond to change and to maintain the systems' socioeconomic and environmental sustainability (Agrawal and Ostrom, 2001; Berkers and Ross 2013). We use these sustainability indicators to discuss whether changes in the social-organizational configuration identified in our study are likely to enhance or reduce the overall sustainability of the fisheries system.

MATERIALS AND METHODS

Study Site: The Madeira Small-Scale Fisheries System

The Madeira River is the main and most important tributary of the right bank of the Amazon River in flow and sediment transport. Its watershed accounts for almost 20% of the Amazon basin across three countries: Brazil, Bolivia, and Peru (Goulding, 1979) (**Figure 1**). The Madeira River basin's ichthyofauna is recognized for its great diversity, with more than 1,057 species described (Ohara et al., 2015). This rich fish fauna is also of great regional socioeconomic importance, generating animal protein and income for subsistence and commercial fishers (Doria et al., 2012).

Driven by the Brazilian National Electric Energy Plan and the government's Accelerated Growth Plans (Fearnside 2014; MME/EPE, 2017), two large hydroelectric dams have been built in the Madeira River basin, which together have an installed capacity of around 7,000 MW: Santo Antônio (operation starting in 2011) and Jirau (operation starting in 2012). The current study focuses on the region of direct and indirect influence of both dams in the municipalities of Guajará Mirim, Nova Mamoré, and Porto Velho (Rondônia State, Brazilian Amazon).

Madeira Fisheries and Fishers' Characteristics

The Madeira River and its tributaries including the Mamoré and Guaporé Rivers, which altogether cover a flooded area of about 2,500 km^2, are the main fishing grounds in the study area (Doria et al., 2017). Prior to the construction of the dams, the three main fish markets of the region had an average production of 619 tons/year for Porto Velho (RO) and 245 tons/year for both Humaitá (AM) and Guajará-Mirim (RO) (Doria and Lima, 2015). Fishing activity is characterized as a small-scale, multi-species artisanal fishery with use of diverse and simple fishing gear, fishing trips of generally short duration and relatively low fisheries yield. The fishing fleet consists mainly of small non-motorized and motorized wooden canoes (~1,000 units, storage capacity of less than 600 kg) and few larger fishing boats (average capacity: 2,500 kg).

The fisheries exploit about 74 different species. Before the construction of the dams, construction, five of these species accounted for 57% of catches: barba-chata (*Pinirampus pirinampu*), pacu-common (*Mylossoma duriventre*), curimatã (*Prochilodus nigricans*), jatuarana (*Brycon amazonicus* and *B. melanopterum*) and Dourada (*Brachyplatystoma rousseauxii*) (Doria and Lima, 2015). Most of these species are migratory, with migrations for reproductive, trophic, or dispersal purposes being strongly influenced by water level and flow (Goulding, 1979; Lima et al., 2017).

Fishing in the Madeira River is an activity of great regional socioeconomic importance, involving about 3,000 commercial fishermen (Doria et al., 2012; Doria and Lima, 2015). Typical local fishing families are composed of two or more fishers. Fishers are typically male, with an average age of 40 years; two-thirds of whom have not completed primary education (Lima et al., 2012). The importance of fishing for these families is emphasized by fish consumption, estimated at 0.5–1.0 kg *per capita* day, and by the average monthly fish landings per family involved in fishing (369 ± 405 kg). Of these landings, 13% is destined to family consumption and 87% for sale. Income obtained from fish sales represents 50–100% of an average riverine family's income (US$ 507 ± 522 in 2009), with the remainder being derived mostly from small-scale agriculture (Doria et al., 2016). Fisheries monitoring data covering two decades prior to dam construction showed relative stability on the catches (Doria et al., 2018a), but catches declined after the Santo Antônio and Jirau dams were built (Lima et al.,2020).

Governance Arrangements

The Madeira fisheries system comprises governmental organizations at different scales (Federal, State, and Municipal); non–governmental organizations; civil society organizations; commercialization chains organizations and consumers; and the private sector (adapted from Doria et al., 2020). In **Table 2** we synthesize the most important organizations and their roles. Artisanal fisheries activity in the Amazonian region is regulated by federal (Ministry of Fisheries and Aquaculture and Ministry of the Environment) and state agencies (State Environmental Agency). These entities are charged with designing and monitoring the implementation of public policies, laws and regulations, as well as with monitoring fishing activity. The Ministry of Fisheries and Aquaculture, along with the Technical Assistance and Rural Extension Company and the Agriculture State Agency state-based agencies, have the mandate to promote fisheries development and sustainability.

The State Environmental Agency (SEDAM) is responsible for overseeing, planning, and managing fisheries. However, this agency works mainly on the supervision and enforcement of fisheries regulations, along with other state environmental policies. Until 2014, issues related to fishing in the State of Rondônia were discussed by the Technical Chamber organized by SEDAM, which was composed of representatives of four entities involved in fishing, and one representative from a fishers colony or association. After this, the Technical Chamber was dissolved. The fishers are organized into local associations (fishers colonies) and syndicates, and at federal scale, into fishers federations. These entities have the mandate to defend fishers' rights (e.g., public policies and legislation compliance, benefits from the government, and compensations) and promote the fishers' class empowerment. Scientific or academic organizations are represented in this study by the Laboratory of Ichthyology and Fisheries of the Federal University of Rondônia (LIP/UNIR), which has been researching fish and fishing dynamics on the Madeira basin since 2000, and which provides information on fishing and fish to fishers colonies and to the government whenever requested.

The licensing and implementation of hydroelectric dams on Brazilian federal rivers is monitored by the Federal Environmental Agency (IBAMA). Santo Antonio and Jirau

TABLE 2 | Organizations by groups of stakeholders and geographic scale (Federal, State, municipal) of the Madeira Fisheries System and their main function.

Government		**Users**	
• Environmental and fisheries management/enforcement		**all level**	Fisher, riverine community, indigenous people
Federal	Ministry of fisheries and aquaculture (MPA) -	**all level**	Middlemen
State	State Environmental and development Agency (SEDAM)	**all level**	Consumers
	Environmental Police (BPA)	**Non-government**	
• Implementation and enforcement of national environmental policies; monitor the dam license process		**• Representation of professional fishers in issues affecting their interests**	
Federal	Brazilian Institute of environment and Renewable resources (IBAMA)	Federal	Federal Fisher association
• Promotion of agricultural, fish farming and fishing production		Municipal	Fisher colony (Colônia de Pescadores Z-1, Z- 2 and Z-13)
State	State Secretary for agriculture, livestock and land regularization (SEAGRI)	**• Support to fishermen/associations; fisheries management**	
	Technical assistance and rural extension company (EMATER)	All level	Non-governmental Organization
Municipal	Municipal Secretariat of agriculture	**• Generate technical subsidies for fisheries management**	
• Maintain the legal order and safety		All level	Science
Federal and state	Public Prosecutions (MPF and MPE)	**• Construction and operation of the dam; impact studies, monitoring, mitigation and compensation of the dam impacts**	
Federal	Navy	Public/Private	Hydropower companies

dams, for instance, should comply with the Technical Instruction n°. 060/2008 of the Environmental Licensing Department (IBAMA, 2008), which states that: "*…the impacts caused on fisheries should be mitigated and/or compensated, to guarantee the environmental sustainability and the improvement of the livelihoods of impacted populations*. Additionally, it is recommended that the implementation of a program should be focused*: 1) on the maintenance of the fisheries activities, 2) on the social compensation for the impacted fisheries activities; 3) on the definition of a new technological pattern, including actions for the reorganization of the (fishing) activity, when necessary.*"

Interviews with Stakeholders

To understand stakeholder perceptions about the fisheries system, interviews were held with key informants from the main stakeholder groups, including representatives of fishers, and staff from governmental and non-governmental organizations, as well as private dam developers (N = 50) (**Table 3**).

The key informants to be interviewed were selected from among the stakeholder groups using the following: 1) technical reports of the Fisheries Monitoring Program published by dam companies, where it was possible to identify the companies' staff and government officers involved, as well as managers and policy-makers (at different levels) for the fishing sector, from 2009 to 2013; and 2) the database of the LIP/UNIR, which was used to identify the most active community leaders and fishers (greater number of landings by locality) (**Table 3**).

Organizations were officially contacted by an invitation letter or email. In the case of dam building companies, responses to the invitation were negative, and in some cases, there was no response at all. Consequently, we chose to make direct contact with the companies' key informants listed in the reports.

Interviews were conducted from December 2014 to February 2015. Informed consent for participation was obtained from all individuals prior to initiating the interviews. The researcher verbally explained the interview procedures, the participants were given the opportunity to ask questions, and then the participants gave their verbal consent to participate. This research was developed under the Amazon Dams International Research Network support and ethical standards, according to the IRB Protocol number #2014-U- 0490. Each semi-structured interview took 45 min on average. Interviews were carried out in the municipalities of Guajará Mirim and Nova Mamoré (upstream of the Jirau dam); in the community of Cachoeira do Teotônio and in the district of Jaci-Paraná (Santo Antônio reservoir area); and in the communities of São Sebastião and São Carlos (downstream area) (**Figure 1**).

TABLE 3 | Description of the sampled number of interviewees per stakeholder group. The number of agencies interviewed is indicated in parentheses.

Group	Subgroup	Respondents
Users	Fishers	26
	Middlemen	3
	Sub-total	29
Managers or employees		
Government	Federal agencies (2)	3
	State agencies (2)	2
	Municipal agency	1
No government	Fisher's colony (3)	3
	Non-government organization (4)	4
	Hydropower companies (2)	4
	Scientists	4
	Sub-total	21
	Total	50

A semi-structured questionnaire (open and closed questions) was developed to collect information regarding the descriptors and indicators summarized in **Table 1** (*see* **Supplementary Material S1**). The interviews were transcribed and analyzed using the Nvivo 10 software to categorize and code the qualitative data, aiming to compare and contrast the answers and interpretations of each theme. The responses were grouped by stakeholder group: users, government staff, or staff from non-governmental entities. Responses were coded according to themes and subthemes, defined by expected answers from the

FIGURE 2 | Pictorial representation of the Madeira fisheries system used in the stakeholders' interviews and cognitive maps. Each of the insert in the map corresponded to a card, on the bottom the fishes, from left to right, from bottom to top: Fisher; Fishers colony or association; Hydropower Company; Non-governmental organization; Science; Middlemen; Consumers and in the middle the Government.

literature review, and by actual answers. New subthemes were created and coded based on the frequency with which they were cited (over three times). In the end, the codes were reviewed, refined, grouped (when possible) and classified into positive or negative analysis, represented by (+) and (−) in the **Supplementary Material** tables. For each descriptor, we considered the most frequent and relevant answers by group or for all respondents, when they corresponded to 20% or more of the answers.

The governance architecture of the Madeira fisheries system after the dam construction was summarized, highlighting the main results for each descriptor. The results were described considering the descriptors numerical sequence and the sustainability indicators proposed in **Table 1**.

Card Games: Cognitive Map of the Fisheries System

During the interviews, the stakeholders' description of the governance network and its interactions were facilitated through the use of a card game developed to help elicit and visualize the stakeholders' understanding, interaction, and participation, and also to aid the visualization of the fisheries system's structure according to each participant's perception (adapted from Pretty et al., 1995). The game consisted of cards with drawings depicting the main fisheries system's components: fish, fishers, government, researchers, class association, hydropower companies; and consumers (**Figure 2**).

After a brief explanation of the game's purpose, the interviewees were asked to identify which organizations had worked with or had a relationship with fisheries or fishers in the region. According to the interviewee's responses, the cards representing the stakeholders were organized on the table. From the selected cards, the participants were asked to draw a picture of the fisheries system indicating the quality of the relationships between the organizations. For "strong" relationships, which are positive for maintaining fishing sustainability, continuous lines were used. For "weak" or negative relationships, dashed lines were used (**Figure 3**). The weak and strong responses were added and expressed as percentage of responses related to all answers. The higher values were considered as more important to designate the connection between two given actors. A cognitive map of the Madeira River fisheries system was produced to enable visualization of the system's main stakeholders (cited in more than 5% of the interviews), as well as the strength and quality (weak or strong) of the relationships between stakeholders in the system. Visualization of the cognitive map model representing the frequency of interactions among the stakeholders indicated the intensity of relationships by the line's thickness.

RESULTS

The main results regarding the governance architecture of the Madeira River fisheries system after the dam construction implementation are synthesized in **Table 4**. In general, most stakeholders stated that fisheries-related institutional arrangements had weakened in the face of hydropower development and that the arrangements had been insufficient to effectively mitigate or compensate for fisheries impacts. Fishers, non-governmental and private sector personnel mentioned the lack of opportunities to participate in fisheries governance as a major weakness, government staff emphasized lack of qualified personnel, inter-agency lack of trust, and the decision-making control

FIGURE 3 | Drawing of the perception of actors about the Madeira Fisheries System and a picture of the interview. The continuous blue link here represents the strong relation and the dashed red link represent the weak relation.

TABLE 4 | The governance architecture of the Madeira Fisheries System after the dam implementation.

Social organizational configuration	
Main Stakeholders	Fisher; Fishers associations, hydropower companies, Federal Environmental Agency staff, State Environmental Agency staff, researchers, ONG personnel
Authority and responsibility	Dam licensing process: Federal environmental agency; fishery management on Rondônia State: State environmental agency
Dialogue among the stakeholders	Negative evaluation; got worse
Interactions among the entities	Negative evaluation; got worse
Expertise and knowledge requirements	Federal University of Rondônia: Status and analysis of fish stocks and fisheries dynamics; hydropower companies: fisheries monitoring and impact assessments; government agencies: no data available
Procedures for legitimate decision-making	Decisions on fishing: autocratic Decisions on mitigating the dam impacts: autocratic (the power of the system is concentrated in the government and hydropower companies)
Cognitive-normative configuration	
Key driver	After the dam: declining fish stocks in the Madeira basin, fisheries impact (income and livelihoods)
Goals and priorities	Keep fish healthy fish for sustainable fisheries
Solutions	Interventions to strengthen the policies: 1) qualified participation of Fishers (affected stakeholders) and their institutions in the decision- making process; and 2) guarantee opportunities for communication and follow-up, with those affected, by government agencies responsible for fisheries management and for evaluating/monitoring the projects' impacts on the activity

held by hydropower companies. Perspectives on other factors were shared across stakeholder groups. These included increased conflicts, lack of interaction and coordination between agencies, fragility of fisher's social organization, lack of trust and reciprocity between organizations, and power imbalance between stakeholders. In the following subsections, we detail the findings synthesized on **Table 4**.

Social Organizational Configuration

The interviewees cited 20 entities as participants of the Madeira fisheries system. Twelve were cited by more than four respondents, and the most-cited were the fishers, followed by the fishers' colony or association and the hydropower companies (D.1; **Figure 4**). Fishers were the focal point of the system, where most of the interactions to or from other stakeholders converge.

About the *Interactions among the entities* (D2) a total of 500 relationships were identified using the card game. Of these, 61% were weak relationships and 39% were strong. The fishers generally had weak ties to government agencies responsible for fisheries management and with the fisher's colonies, entities that should work with fishers to guarantee the fishery sustainability, as well as to defend their rights. On the other hand, a stronger relationship was expressed between fishers and science or academic institutions, with the Federal Public Prosecution Service (MPF), and with Non-governmental Organizations. These entities do not have the role of managing the fishing activity. The agency responsible for environmental licensing and supervision of hydroelectric projects (Federal Environmental Agency) and local government appear to have a strong relationship with the hydropower companies and a weak connection with other entities in the system, a situation that is likely to undermine fisheries resilience and sustainability (**Figure 4**).

Based on interviewees' perceptions, the entities that have authority and responsibility (D2.1) over the main problem are the hydropower companies (54% of all interviewees) and the government (24% of the interviewees) (**Supplementary Table S1**).

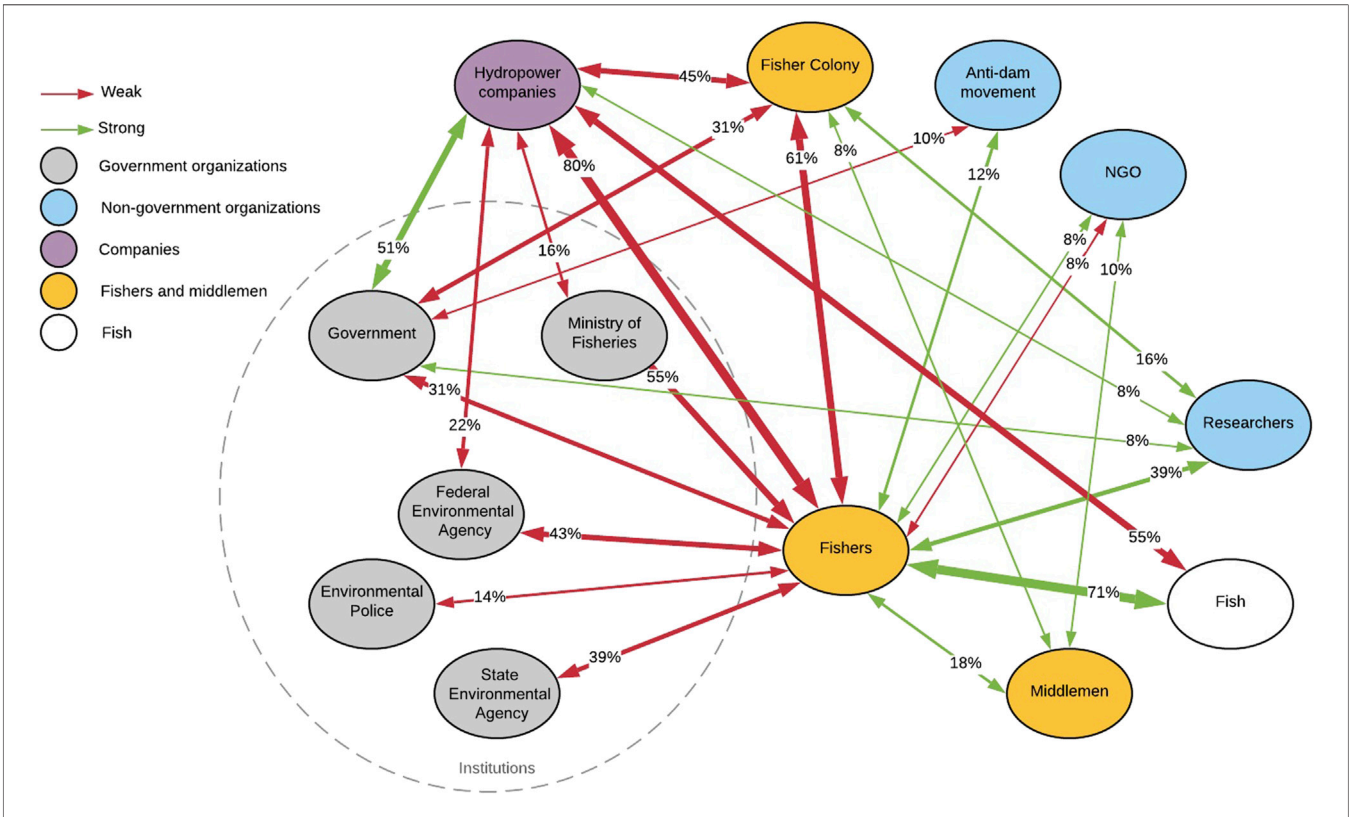

FIGURE 4 | Cognitive map of the Madeira Fisheries System after implementation of the Santo Antônio and Jirau dams. The circles represent the institutions cited by the respondents: the green lines represent strong relationships that contribute to sustainability, while red lines represent weak relationships that do not contribute to the fishing sustainability. Line density represents the percentage of responses related to all answers.

Lack of trust in hydropower companies (92%) stands out among the three stakeholders groups' responses. The same lack of confidence was mentioned in relation to the Ministry of Fisheries and Aquaculture, the Federal Environmental Agency and local government agencies, but with a lower percentage of interviewees (30, 10, and 32% respectively) (Trust and reciprocity between the stakeholders - SI.4). Some testimonies that clearly represent this perspective are:

"[...] there is a very spurious relationship between everybody and the company. The company has a very clear purpose. Because of this, the whole process gets very skewed ... the company keeps insisting and forcing the bar to decrease the cost of it. The discussion of respectability happens at the beginning of the process (e.g.,: Let's do everything and strive) then all of that is lost. After the LO (operating licensing) it gets worse " (Federal Government employee).

"[...] When the turbines get closed, a lot of fish die ... and then they bury everything... Now the fishers will catch a fish if they do not arrest soon.... I have a friend who works there and said it to me. I ask if he can take a photo. He says no because the guy pays his bills - gets fired" (Fisher).

"[...] letting the company hire anyone who wants to do the monitoring or mitigation, is to put the fox to take care of the chicken." (Non-governmental organization member).

In this context, the three groups reported that the hydropower companies own the power or control of the fisheries system at that moment, followed by the local government (50 and 24% of the interviewees, respectively) (**Supplementary Table S1**).

"[...] what exists is an open path to corruption, to neglecting. The fact that the company pays for everything and is the leader of this process only leads to the destruction of the political relations that existed before. The company brings that money and changes all the relationships of interest. It puts money in such a direct way, both in Federal Environmental Agency, in the government and the fishers etc. It ends up being the great dominant. It does it enabled by the money, corrupting the entire system. In the Madeira project, maintaining people's financial conditions and way of life was never considered as a central objective. The main objective is to implement the project. If you can keep the first goal associated with another one, that's okay, if it does not go well, we'll deploy it the same way. " (Federal government employee).

According to the interviewees, the entities that have the required expertise and knowledge over the main problem (D2.2) are the dam-builders (74% of the interviewees) (**Supplementary Table S1**). This is because hydropower companies coordinate fisheries monitoring programs in the affected area, and also control data and information generated by these programs. Respondents argued that this information is not shared by the hydropower companies (SI.4) (84% of the interviewees) (**Supplementary Table S1**). In this sense, 50% of personnel from non-governmental organizations questioned the effectiveness of monitoring program oversight carried out by the Federal Environmental Agency. This situation suggests a lack of equity in power and participation between the organizations (SI.5).

Theoretically, any technical information, as well as information acquired through monitoring programs should be presented and discussed by the hydropower companies with civil society in public hearings, with the working group on the impacts of fisheries and dams, and with fisheries monitoring program participants. However, respondents demonstrated that they have little knowledge about these discussion spaces (less than 20% of interviewees), especially in the fishers group. It is noteworthy that only 36% of the fishers interviewed knew about the fishery compensation plan, of which they are ostensibly the main beneficiaries (**Supplementary Table S1**).

The dialogue (or lack thereof) among stakeholders (D.3) was viewed as negative after the dams construction by the majority of the interviewees (**Supplementary Table S2**). Below, we share some interviewee testimonies that express this finding:

"[...] *my impression is that there is no dialogue. It was a deaf and mute conversation. No one wanted to see or solve anything in meetings ....* " (Hydropower company employee).

"[...] *it seemed to me that even the environmental agencies did not decide. It depended on the interest of local politicians, it always depended on who was in charge ... there was no local communities' participation in the decisions about fishing.* " (Non-governmental organization).

"[...] *IBAMA's relationship with the hydropower companies is complex and varied. There were many initiatives, multiple ways of relating both at the technical level and at the political level. At the technical level, we tried to have a close relationship, we actively participated in the conception and we had a huge responsibility in its execution. The relationship worsened after the LO (operating license) was granted, and I don't see the IBAMA relationship with the dam company as good as it was before. It became more bureaucratic and we were not able to open local spaces for conversation, which is very negative.* " (Hydropower company employee).

"[...] *the problem is how the staff (from dam-builder) dialogue with society. The studies are clear about the environmental impacts, there is no doubt. The problem lies in the connections with society, which is willing to participate and also give an opinion. There are provisions for this to be transparent. But there is a lot of resistance in doing this because there is no fertile environment for this discussion. Our councils have become very weak for these discussions and civil society is not prepared. Besides, the companies are not ready to discuss this. They do not see this as important to legitimize the process.* " (Federal government employee).

Most of the reasons cited for this scenario are related to management problems and conflicts of interest between organizations (D.3; **Supplementary Table S2**). The group of government and users highlighted problems with the fisheries monitoring program, as well as political conflicts and interests between organizations. The lack of trained personnel, in governmental entities, to monitor hydropower companies' proposals for compensation and alternative income generation, was cited for all stakeholders. On the other hand, non-governmental organizations personnel emphasized 1) an increase in the number of fishers; 2) management problems within organizations; and 3) political and interest conflicts between organizations.

Some testimonies that voice this result are:

"... *I see that, unfortunately, economic power always overlaps with social interests, especially considering vulnerable social groups such as riverine communities and fishers. I regret the ignorance of the public agencies regarding maintaining a minimum of condition for these people. I see the environmental agency (IBAMA) completely alienated, always connected to the version given by the hydropower companies* (State public prosecutor).

"... *the MPA* (Ministry of Fisheries and Aquaculture) *is an agency that is very accessible to fishers, but it seems that it does not know how to walk. Chico Mendes Institute for Biodiversity Conservation (ICMBIO) lives in poverty, never has financial resources to do anything, so it can't do anything ... only understands that sustainability is forbidding fishing ... can't open up for discussion; IBAMA is an enforcement body qualified to supervise but has no legs for that, lack of resources, sporadic inspections, and Secretary of State for Environmental Development (SEDAM) is the great problem for fishing within the State.* " (Non-governmental organization member).

Most of fishers interviewed (94%) are affiliated with the municipal association or colony, what could express their social capital (SI.1). However, 38% of them reported that they do not have the power or leadership (SI.2) to help them solve fishing problems and few (28%) recognize the colony presidents as leaders.

About the procedures for decision-making (D.4) in the fisheries system, the entity responsible for fishing management is the State Environmental Agency (cited by 56% of the interviewees), that should consult the Technical Chamber of Fisheries (TCF). According to 84% of the interviewees, they have not had the opportunity to attend meetings or discuss fisheries with the TCF (**Supplementary Table S2**). The responses about the TCF are negative, revealing that this chamber is used to legitimize pre-defined government decisions with little stakeholders' participation, which leads one to believe that autocratic management practices were used.

There were changes in the fishing rules after the dam implementation (SI.6) according to 30% of the interviewees (**Supplementary Table S2**). One example is that fishers were banned from fishing in areas where they fished, which required them to adapt to new fishing methods and to where tehy traditonally fish in more distant locations. Most of them (84%) claimed that they have not participated in discussions about changes caused by hydroelectric dams. This result reinforces gaps in stakeholder's participation (SI.7).

Cognitive-Normative Configuration

All three major groups of interviewed stakeholders perceived changes in fisheries over the past 10 years. The majority of interviewees (78%) considered the construction of the dams to be the most important factor triggering these changes (D.5) (**Supplementary Table S3**). According to three stakeholders groups, dams have caused significant changes in the riverine communities' livelihoods. Many of them are negative and relate to economic losses due to changes in fishing activity, including a decrease in income from fish sales, increased costs of the fishing activity, and a decrease in fish abundance. Changes related to fish and fisheries are also negative. The three groups agreed and reported these changes in decreasing order of importance: decrease in fish abundance, changes in the aquatic system, and changes in fish migration patterns (**Supplementary Table S3**).

As mentioned before, according to Brazilian rules regarding environmental licensing of hydroelectric dams, the goal (D.6) of mitigation/compensation is that: "*the impacts caused on fisheries should be mitigated and/or compensated, to guarantee the environmental sustainability and the improvement of the livelihoods of the impacted populations*" (Technical Instruction n^{o}. 060/2008).

According to 30% of the interviewees, there were no fishing conflicts in the region prior to implementing the dams (D.7; **Supplementary Table S3**). However, 22% reported that there was a conflict between fishers and enforcement agents, and 18% reported conflicts for fishing areas. They mentioned that during the dam implementation, new conflicts appeared, including conflicts between fishers and dam construction companies (34%), increased conflicts between fishers and enforcement agencies (30%), and disagreement related to changing rules regarding fishing areas (42%). The fishers pointed out that they were prohibited from fishing on the rocks near the dam, where they traditionally fished, and that their fishing areas became inaccessible or were damaged by dam construction.

The fragility of the system, especially concerning management, is highlighted by suggestions for improvement (D.8) such as: creating economic alternatives and management of the fishing activity; recognition of fishers' rights; improvement of dialogue between entities; and the implementation of improved and independent monitoring processes (not controlled by the hydropower companies). (**Supplementary Table S3**). A negative perspective for the future of the fisheries system was reported by more than 90% of interviewed people. This perspective is expressed by the following examples: *"[...] fish will not exist in 10 years . . . there won't be any fish in the river, and there won't be anyone to fish in the river* (Fisher).

DISCUSSION

Results from this study strongly suggest that hydroelectric dam construction has caused profound negative effects in the Madeira River region. Impacts extend not only to biophysical elements and processes, but also to social and institutional relationships, fish abundance, and fishers' access to fishing resources. As noted in this and in other studies, these changes have also affected livelihoods and fishing activities, reducing catch and consequently revenue (WCD, 2000; Gutberlet et al., 2007; Santos et al., 2018; Arantes et al., 2019; Figueiredo et al., 2019; Lima et al., 2020). These impacts are attributable in large part to physical, hydrological, and ecological changes, but our results indicate that governance failures have likely contributed to exacerbating some effects (e.g., through changing rules that limit access to traditional fishing areas) and prevented appropriate mitigation or compensation actions. Based on the results presented here, as well as building on previous research, we suggest that analyses of fisheries systems should integrate stakeholders and their interactions, as well as governance processes, in addition to the hydrological and ecological attributes as drivers of dam impacts and mitigation effectiveness (Ostrom 1990; Lorenzen et al., 2007; Lorenzen 2008; Siciliano et al., 2016). In a social-ecological system such as fisheries systems, the governance, including institutional arrangements and their relationships and conflicts, promote understanding of social complexities and how these relationships might affect fisheries sustainability in the long-term (Gutberlet et al., 2007).

Whereas all stakeholders perceived fishers to be at the center of the fisheries system, their relationships with relevant governance agencies were weak even before hydropower development and were further weakened because of it. Weak organization and representation of artisanal fishers are factors Brazil, which contributes to the fisheries sector's invisibility in the context of hydropower development (Doria et al., 2018b). The extent to which local impacts could be mitigated by institutional arrangements and affected communities' ability to withstand impact are partly determined by their social capital and resilience (Siciliano et al., 2016). In this sense, it was observed that although the majority of users (fishers) are members of the Fishers' Associations, these organizations are very fragile, disorganized, and lacking strong leadership. In the case of the Madeira river, organizations are weakened by the abrupt change in the system caused by dam construction. In this already fragile context, organizations are forced to recreate new forms of governance, involving a transformed ecosystem, internal actors and new external actors. The new rules of governance, after dam construction, were not adapted to local fishers' needs and conditions. The affected stakeholders were not able to participate in rule adjustments, there was no governance compromised with the fisheries system sustainability, and no equity of participation and balance of power among group members. Strengthening fishers and their organizations to improve their representation and participation in the decision-making process is therefore crucial, as is the recognition of fishers' rights (Siciliano et al., 2016).

In addition to dialogue, effective impact assessment and mitigation requires understanding and quantifying ecological, fisheries and social impacts, as well as considering trade-offs and synergies. In the Madeira, this was hampered by hydropower companies' control over environmental and fisheries impact studies. There was an evident lack of dialogue and interaction between the agencies responsible for the dam licensing process and for fisheries management. This evident disconnect reinforced negative opinions of the assessments and the feeling of abandonment among fishermen, thus accentuating conflicts between interested parties. Research on dams' social impacts shows that gaps in knowledge and failures to understanding the trade-offs involved are common in the Americas (Kirchher et al., 2016), Africa, and Asia (Legese et al., 2018; Siciliano et al., 2018). Poor data management and communication is a problem that needs to be addressed to improve the planning and mitigation of hydroelectric dams and to support more transparent assessments and communication of trade-offs involved in dam development decisions (Kirchherr et al., 2016; Athayde et al., 2019).

Also, we found that the governance arrangements of the Madeira fisheries system do not display key characteristics associated with resilience and sustainability (Ostrom, 1990; Agrawal and Ostrom, 2001; Berkers and Ross 2013). In this situation, a participatory process may be initiated to help re-evaluate governance arrangements and support interventions aimed at enhancing resilience and sustainability (Legese et al., 2018).

Finally, to address the main problems mentioned by the stakeholders, which directly affect the fisheries system's governance and sustainability, we offer the following recommendations:

- Strengthening fishers' access to information and institutional organization to improve their participation in the decision-making process by: providing clear information (before, during, and after the project's implementation) on all stages of the process, explaining how potential impacts are monitored and/or mitigated, clarifying how they can participate in discussion spaces, and offering leadership training courses;
- Recognition of fishers' rights by creating spaces for participatory and equitable consultation, discussions and decision-making about fishing and the impacts generated by the dams, and offering, when possible, free legal support for fishers and their associations;
- Creating economic alternatives and management strategies for the fishing activity that target place-based needs and opportunities. This could include technically supporting aquaculture practices, local fishers' input to select potential areas for installation and management of such initiatives, promoting community management of reservoir lakes and natural lakes, and supporting the fish production chain from the affected communities, adding value to managed fish;
- Strengthening of government agencies responsible for fisheries management and for evaluating/monitoring the project's impacts on the activity, with increased financial and human resources training to assess, monitor and manage impacts and fisheries, as well as implementing independent monitoring processes with government control and support from public research entities;
- Improving the dialogue between entities to guarantee opportunities for communication and follow-up with those affected. This could be done through previous training programs, empowerment of fishers and representatives, and regular meetings between the fisheries system stakeholders, with equitable participation in discussions and decision-making on impacts, as well as on the results of monitoring and mitigation initiatives.

Our study provides an in-depth analysis of dam-related changes in the governance system of a tropical river fishery. Results indicate that governance failures can contribute to exacerbating dam impacts, and that these failures have prevented appropriate mitigation and/or compensation efforts. While this study was based on a case study focusing on the Madeira River in the Brazilian Amazon, the results point out to specific governance attributes that are likely to affect many other systems. These insights might help practitioners and scientists to identify, examine, and when necessary, to address such attributes in their focal systems.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors, without undue reservation.

ETHICS STATEMENT

Ethical review and approval was not required for the study on human participants in accordance with the local legislation and institutional requirements. The participants provided their written informed consent to participate in this study. This study is supported and in accordance to the Institutional Review Board Protocol # 2014-U-0490, of the University of Florida, for research conducted in connection to the Amazon Dams International Research Network (ADN).

AUTHOR CONTRIBUTIONS

CD research design, data collection and analysis, article writing. JD-G research design, article writing and review. MP article writing. KL article writing and review. SA research design, article writing and review.

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SUPPLEMENTARY MATERIAL

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METHODS
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Quantifying Cooperation Benefits for New Dams in Transboundary Water Systems Without Formal Operating Rules

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New dams impact downstream ecosystems and water infrastructure; without cooperative and adaptive management, negative impacts can manifest. In large complex transboundary river basins without well codified operating rules and extensive historical data, it can be difficult to assess the benefits of cooperating, in particular in relation to new dams. This constitutes a barrier to harmonious development of river basins and could contribute to water conflict. This study proposes a generalised framework to assess the benefits of cooperation on the management of new dams in water resource systems that do not have formal sharing arrangements. Benefits are estimated via multi-criteria comparison of historical reservoir operations (usually relatively uncooperative) vs. adopting new cooperative rules which would achieve the best results for riparian countries as evaluated by a water resources simulator and its performance metrics. The approach is applied to the Pwalugu Multipurpose Dam (PMD), which is being built in Ghana in the Volta river basin. The PMD could impact downstream ecosystems and infrastructure in Ghana and could itself be impacted by how the existing upstream Bagre Dam is managed in Burkina Faso. Results show that with cooperation Ghana and Burkina Faso could both increase energy production although some ecosystem services loss would need to be mitigated. The study confirms that cooperative rules achieve higher overall benefits compared to seeking benefits only for individual dams or countries.

Keywords: hydropower and ecosystem service trade-offs, dam operating policies, many-objective trade-off analysis, cooperation in transboundary water systems, Volta river basin

INTRODUCTION

Growing food, water and energy demand is fuelling the expansion of water resources infrastructure such as dams (Zarfl et al., 2014; Siciliano et al., 2018; Gerlak et al., 2020). New infrastructure can modify a river's flow regime impacting benefits and services provided by existing downstream infrastructure and natural assets like wetlands or floodplains. These often play a central role in the functioning of aquatic ecosystems and of local communities by providing food and drinking water supply services (Mul et al., 2017). Furthermore, new infrastructure can generate or intensify conflicts between different basin stakeholders, especially in transboundary river basins without cooperative water management arrangements (Sadoff and Grey, 2009; Basheer et al., 2018; Wheeler et al., 2018). In this context, water resource systems models can be used to identify and quantify the potential impacts of new infrastructure and different operating policies of existing and new infrastructure (Loucks, 1992; Thiessen and Loucks, 1992).

Motivated by the need to identify the best possible approaches to share transboundary waters, optimisation models have often been used to filter through the nearly unlimited ways water sharing rules could be structured and parametrised. Different methods have been applied including linear optimisation (e.g., Kucukmehmetoglu and Guldmann, 2004, 2010; Porse et al., 2015), stochastic dynamic programming and stochastic dual dynamic programming (e.g., Tilmant and Kinzelbach, 2012; Tilmant et al., 2012; Arjoon et al., 2014; Kahsay et al., 2019). Kucukmehmetoglu and Guldmann (2004) use a linear optimisation model to identify optimal allocation of water resources in the Euphrates and Tigris river basin considering water demands for irrigation, urban consumption and hydropower generation, where the economic consequences of different strategies with cooperative and non-cooperative scenarios were assessed. Arjoon et al. (2014) use a stochastic dual dynamic programming model to assess the hydro-economic risk of the Grand Ethiopian Renaissance Dam (GERD) in the eastern Nile river basin, where they demonstrated that if the riparian countries (Ethiopia, Sudan, and Egypt) agree to cooperative management in the basin, the GERD would increase the basin-wide benefits, increasing water security in Sudan and Egypt during dry years.

Other scholars have sought to more realistically represent institutional realities and motivations within models. Methods used in this vein include for example agent-based models (e.g., Becker and Easter, 1999; Kilgour and Dinar, 2001; Giuliani and Castelletti, 2013; Yoon et al., 2021) and game theory (e.g., Bennett et al., 1998; Bhaduri and Liebe, 2013; Bhagabati et al., 2014). Giuliani and Castelletti (2013) presented a multiagent decision-analytic framework to model and analyse different levels of cooperation and information exchange among multiple decision makers in the Zambezi river basin. Bhagabati et al. (2014) investigated the net benefits of hydropower developments and water resource utilisation in the Mekong transboundary river basin using a game theory approach where different adaptation strategies that benefit all riparian countries were proposed. A potential advantage of game theory or agent-based methods is their ability to represent more realistic scenarios of non-cooperation and non-centralised management in transboundary systems, involving multiple institutional or stakeholder interests. These methods assume each agent or institution can give priority to their own objectives and the interactions between parties can be identified and described (Madani, 2010; Giuliani and Castelletti, 2013). Optimisation techniques typically assume all parties are willing to cooperate and exchange information to achieve better system-wide outcomes; in practice this can be unrealistic given the political and institutional contexts within transboundary river basins (Giuliani and Castelletti, 2013).

Classical optimization, agent-based and game theory methods may however be difficult to apply when representing large-scale complex systems including non-linear physical and institutional processes. In this case rule-based simulation of water resources systems (Loucks, 1992; Loucks and Van Beek, 2005; Matrosov et al., 2011) can be helpful because they do not require assuming optimising behaviours and they can be applied to large-scale complex real-world non-linear water systems, coupling hydrological complexity with the complexity of human water management rules. Simulation models allow to test and refine management strategies through scenario simulation and can incorporate water management impact information from ecology, economics and stakeholder-informed metrics of performance (Wurbs, 1993; Loucks and Van Beek, 2005; Harou et al., 2009; Voinov et al., 2016). Water resource management simulation models can help improve stakeholder understanding of large-scale and complex systems and thereby contribute to their design, management, and operation (Hall et al., 2019).

In the last decade hybrid simulation-optimisation methods have become popular; this involves simulating a variety of behaviours (without necessarily assuming optimising agents), then linking simulators to external independent search algorithms which help filter through the numerous system design and management options whilst considering the motivation and behaviours of simulated agents. Linking water resources system simulators to multi-objective evolutionary algorithms (MOEAs) has proven effective at allowing water manager and planners to assess trade-offs among conflicting objectives and explore strategies under multiple plausible future scenarios (e.g., Kasprzyk et al., 2009; Herman et al., 2014; Zeff et al., 2014; Matrosov et al., 2015; Huskova et al., 2016; Giuliani et al., 2018; Wheeler et al., 2018; Geressu and Harou, 2019; Wild et al., 2019). MOEAs use search techniques to approximate a multidimensional Pareto front identifying a non-dominated ("best achievable") set of intervention strategies (Reed and Kasprzyk, 2009; Maier et al., 2014, 2018). For the different system design and/or management alternatives identified in the Pareto front, an incremental change made to improve one objective will come at the expense of one or more other objectives (i.e., the system has reached a level of system performance where any further improvement will necessarily come at a cost). The resulting multi-objective trade-off analysis can assist stakeholders in managing or planning river system infrastructure and services. Example applications, several of which are in Africa, include (e.g., Hurford and Harou, 2014; Smith et al., 2015; Zeff et al., 2016; Geressu et al., 2020; Hurford et al., 2020a,b).

To date, there have been relatively few multi-objective trade-off studies in transboundary water systems seeking to assess whether a new development in one country may or may not conflict with benefits in others (Geressu and Harou, 2015; Wheeler et al., 2018). For example, Giuliani et al. (2016a) explored the impact of different climate projections on existing infrastructure in the Red river basin, shared by China and Vietnam. They used many-objective Direct Policy Search

(Giuliani et al., 2014) to identify optimal reservoir operations that offset possible negative impacts on hydropower generation in the basin. Wheeler et al. (2018) presented a framework to evaluate the impact of the GERD in Ethiopia on the downstream infrastructure in the eastern Nile basin. They coupled the Borg MOEA (Hadka and Reed, 2013) with a hydro policy model of the basin to explore different management infrastructure strategies under different levels of cooperation between the riparian countries. Schmitt et al. (2019) proposed a multi-objective optimisation-based framework for planning dams in the transboundary Mekong basin considering trade-offs between sediment and hydropower.

This study builds on the multi-objective water system design literature contributing a multi-objective framework for evaluating impacts of new dams on the performance of existing natural and built transboundary basin systems with poorly codified operating rules. The proposed methodology is divided into three stages. Stage one uses a policy identification approach (Giuliani et al., 2014), based on many-objective direct policy search, to identify historical operating policies for existing infrastructure using observed operational data (observed flow and storage volume); stage two, based on two scenarios, cooperative and historical non-cooperative, identifies optimal operational strategies for new infrastructure; finally, in stage three, a stochastic evaluation of the operating policies identified in stages one and two is conducted. The cooperative scenario assumes transparent coordination between the riparian countries and infrastructure operators through a central water management institution in the basin. The historical non-cooperative scenario omits coordination of historical operations with new infrastructure. The framework is applied to the new Pwalugu Multipurpose Dam (PMD), in Ghana, within the Volta river basin. The new dam is designed to produce hydropower, provide flood protection, food production and employment via a new irrigation scheme. Our application aims to demonstrate how formal synergistic operation of new and existing infrastructure in the transboundary, multi-reservoir Volta basin could improve water resource system ecosystem services for multiple actors.

The paper is organised as follows: section "The Volta River Basin Context" presents a description of the Volta river basin; section "Methods and Tools" introduces the Volta river basin simulation model and describes the methods. Section "Results and Discussion" presents and discusses results and section "Conclusion" concludes.

THE VOLTA RIVER BASIN CONTEXT

Basin Description

The Volta river basin (**Figure 1**) is located in West Africa and is shared by six riparian countries where Ghana and Burkina Faso make up the largest area of 42 and 43%, respectively, while the remaining 15% is distributed between Benin, Côte d'Ivoire, Mali and Togo (Mul et al., 2015). The Volta's total average annual flow is approximately 40,400 [Mm^3 $year^{-1}$] and making it one of the major rivers of Africa (McCartney et al., 2012). The Volta's water resources are used for irrigation, hydropower generation, fisheries, domestic, industrial, mining, livestock, and water transport. According to Mul et al. (2015), and Baah-Kumi and Ward (2020) different factors such as high population growth, climate change, floods in wet seasons, and increased economic activity have motivated new infrastructure to regulate flow in the basin. Proposed infrastructure includes single and multi-purpose dams for hydropower, large-scale irrigation schemes and public water supply.

Figure 1 shows the most relevant built and natural assets (existing and planned) in the basin. Existing built infrastructure includes the Kanozoe, Ziga, Loumbila, Bagre Dams in the White Volta, Kompienga dam in the Oti, Bui dam in the Black Volta, and Akosombo and Kpong dams in the Lower Volta, and associated irrigation schemes. Existing natural assets includes the floodplain downstream of Pwalugu, which floods annually when the White Volta river overflows, distributing water and sediments beyond the riverbanks. The floodplain supports local communities' livelihoods enabling flood recession agriculture (FRA), pond fishing, year-round domestic water supply and grazing grounds for livestock during the dry season (Mul et al., 2017).

The PMD is being implemented in the White Volta sub-basin in the Upper East region of Ghana, downstream of the Bagre Dam in Burkina Faso (Mosello et al., 2017; Baah-Kumi and Ward, 2020). The project is managed by the Volta River Authority (VRA), in collaboration with the Ghana Irrigation Development Authority (GIDA) and funded by the Ghanaian government. The PMD is expected to contribute to the development of Northern Ghana, one of the countries' poorest regions (Darko et al., 2019). According to Volta River Authority (2018) the PMD has the potential to irrigate 20,000 ha and to produce 176 GWh. Another function of the PMD would be flood protection resulting from intense rainfall events in northern Ghana, exacerbated by spills from the Bagre Dam (Darko et al., 2019).

Context for the Development of Pwalugu Dam

The Volta river basin in West Africa is shared between six countries with Ghana and Burkina Faso covering over 80% of the area and where most of the large water infrastructures in the basin are found (Mul et al., 2015). Water resources in the basin provide essential services that support the economies of its riparian communities and wider countries (Mul et al., 2015, 2017). Ecosystem services and water resources infrastructure in the basin facilitate food production, access to renewable and cheap energy via hydropower, flood protection and provision of water for industrial and domestic use (Baah-Kumi and Ward, 2020). In the last two decades Ghana and Burkina Faso have experienced significant economic and population growth putting increased pressure on their water resources. This led the two governments and the transboundary basin agency, the Volta Basin Authority (VBA), to consider developing some new infrastructure to control and increase water availability in the basin as well as provide other benefits such as increased food production via formal irrigation schemes and hydropower. The VBA is a basin-wide institution that, among

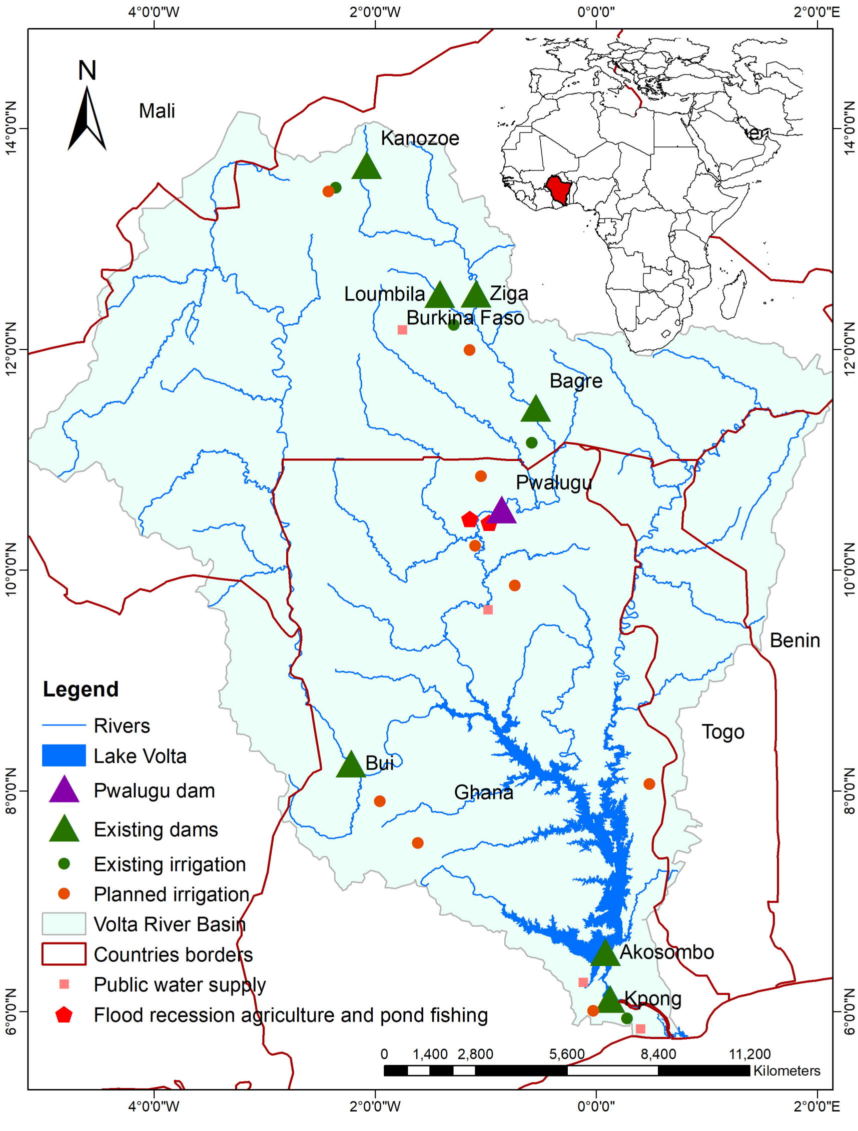

FIGURE 1 | Volta river basin. Location of infrastructure in the six-country river basin.

other mandates, is developing a transboundary cooperative institutional framework to guide future water use in the multi-country basin (Baah-Kumi and Ward, 2020).

The PMD is located on the White Volta sub-basin close to the Pwalugu village in the Upper East region of Ghana, downstream of the Bagre Dam in Burkina Faso (Mosello et al., 2017; Baah-Kumi and Ward, 2020). The main benefit of the dam is considered to be hydropower production, which is aimed at increasing energy connectivity in the northern regions of the country. The dam will also, provide flood protection, food production and employment via a new irrigation scheme. As an energy exporting country, Ghana would benefit from increasing overall hydropower production, which also supplies Burkina Faso. However, while providing these benefits, the PMD could disrupt the current flow regime which allows FRA and pond fishing (Mul et al., 2017). Also, reduced downstream water availability because of increased water storage, evaporation and irrigation water use may reduce hydropower production downstream at Akosombo (Baah-Kumi and Ward, 2020). Furthermore, the PMD itself could be impacted by the operation of the existing Bagre Dam or expansion of the irrigated area upstream in Burkina Faso. Ideally these dams would be operated jointly to maximise the benefits to both countries and to limit negative impacts. However, despite the existence of the transboundary VBA basin-wide agency, current management and development initiatives amongst the riparian countries are relatively uncoordinated (Baah-Kumi and Ward, 2020). For example, historically, more than 80% of the annual inflow into the Bagre Dam is spilt during peaks flow periods (August/September) (Mul et al., 2017; Darko et al., 2019) contributing to flooding in Northern Ghana. Floods in Northern Ghana have led to the deaths of more than 63 people and displaced over 100,000 in addition to causing widespread damage to farmlands, roads, houses, and bridges (Davies, 2018; IFRC, 2019). According to Baah-Kumi and Ward (2020), Ghana and Burkina Faso have agreed that Burkina Faso will provide 2 weeks' notice before the start of annual Bagre Dam spillage in late August or early September, allowing Ghana take mitigation measures to reduce flood damage.

METHODS AND TOOLS

The framework presented in this study (**Figure 2**) is composed of three stages. Stage 1 formulates the planning and management problem and identifies historical operating rules when information about them is not available. The policy identification step calibrates a river basin simulation model representing existing infrastructure to available historical data via many-objective optimisation; in the literature this has also been referred to as "direct policy search" (Giuliani et al., 2014). This identifies operating policies for existing infrastructure that reproduce the system's historical operation under the observed inflow, release, and storage volume time series. In Stage 2, two scenarios, cooperative and historical non-cooperative, are defined and a robust many-objective direct policy search is implemented to design new infrastructure operating policies for both existing and planned assets. In Stage 3 evaluation of the operational policies over an ensemble of plausible futures and sensitivity analysis are conducted.

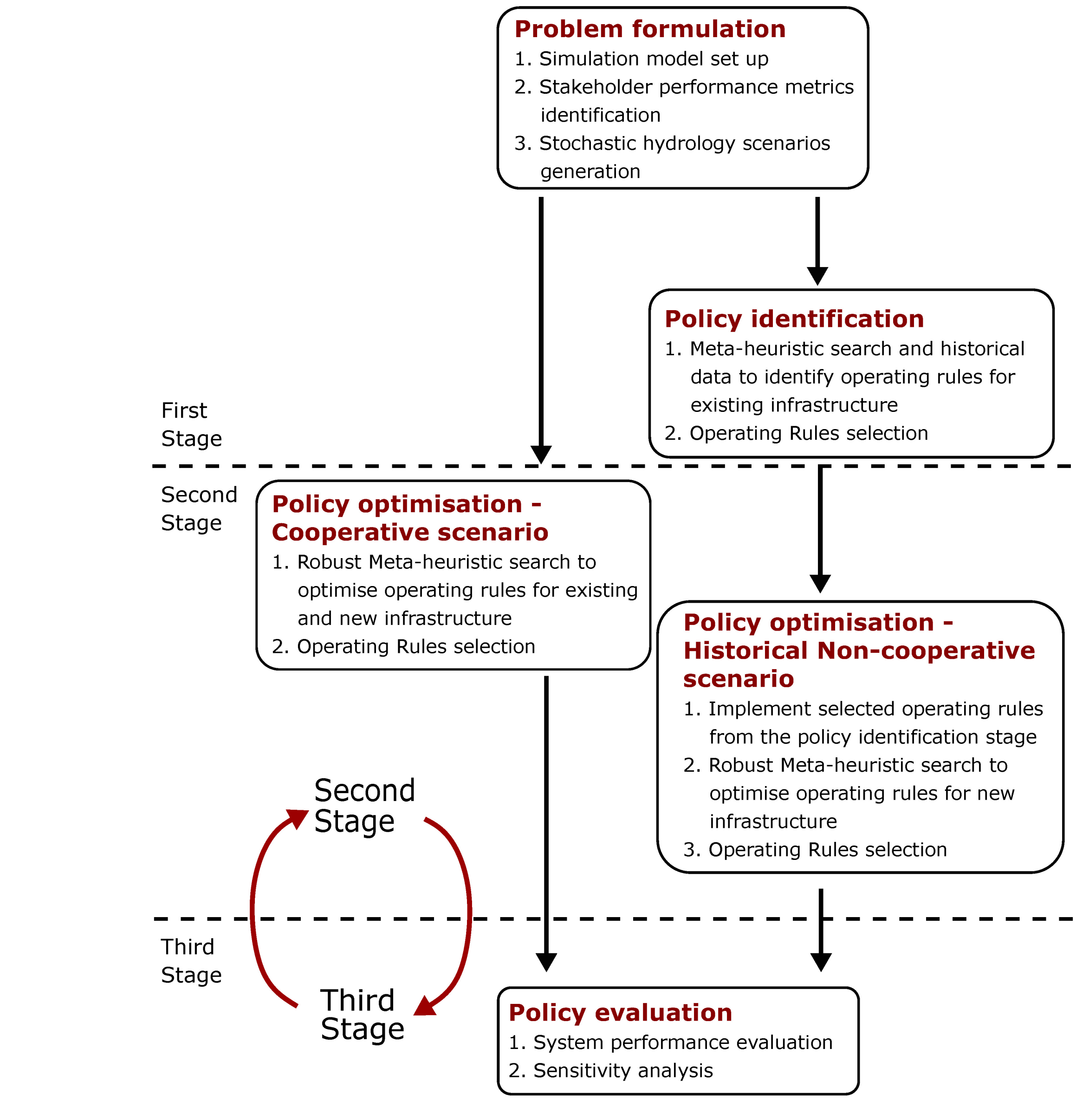

FIGURE 2 | Schematic of the proposed framework.

The cooperative scenario represents a situation where there is a central water management institution in the basin (as is the case of the Volta river basin) and where the countries and the infrastructure operators agree management strategies of existing and future infrastructure. This scenario also assumes all data is shared (inflows, water demands, water withdrawals, and future developments) across the basin. This assumption is not always met in practice given the political and institutional settings in many transboundary basins (Giuliani and Castelletti, 2013). However, according to Sadoff and Grey (2009) an integration scenario that recognises the opportunities for distribution of benefits and costs, and the different alternatives of cooperation, is a worthy goal of integrated water resources management. The historical non-cooperative scenario assumes there is no cooperation between the riparian countries and infrastructure operators with regards to reservoir operation. Here, the upstream countries maximise their benefits without consideration of the impacts on the downstream countries built or natural assets.

Volta River Basin Simulation Model

The river basin simulation uses the Python Water Resources model (Pywr), a Python-based water resources system simulator freely available as an open-source Python library (Tomlinson et al., 2020). Pywr is an optimisation-driven simulator and solves a linear program at every simulation time-step to allocate water to different nodes by minimising allocation penalties. The model can be summarised by a mass balance equation Eq. (1) solved at each node in the network representing incremental catchment inflow and water demands at ecosystem service delivery and infrastructure locations:

$$S_{t+1,n} = S_{t,n} + q_{t,n} - e_{t,n}\left(h_{t,n}\right) \mathrm{C}^{\mathrm{R}} \left(\sum_{\forall i} u_{t,n}^{i} - sp_{t,n} \right) \quad \forall\, t, n \tag{1}$$

where, $S_{t,n}$ is the volume of water stored in the reservoirs at node n, in time-step t; $u^i_{t,n}$ represents the water allocation for the water uses (i) in the system, namely public water supply (pws), hydropower (hp), and irrigation schemes (is). Irrigation demand is defined by the water demand of each crop (ct) integrating an irrigation scheme whilst $sp_{t,n}$ denotes spill flows from reservoirs; $q_{t,n}$ represents the inflows to the nodes and $e(\cdot)_{t,n}$ represent evaporation, which uses on the water level $h_{t,n}$ in the reservoir to determine its real-time surface area using a bathymetry curve. C^{R} is the network connectivity matrix in the system [$\mathrm{C}^{\mathrm{R}}_{\mathrm{j,k}} = 1$ (-1) where node j receives water from (to) node k]. For releases to consumptive water uses (mainly irrigation uses), the network connectivity matrix tracks possible flows that return to the network as a fraction of the release.

Figure 3 shows the network schematic of the Volta river basin simulation model, and a screenshot of an online model deployment (Knox et al., 2019) which allows its access by collaborating parties. The model includes the existing infrastructure in the basin and the proposed PMD project (**Table 1**). Existing built infrastructure in Burkina Faso includes the Kanozoe dam and attached Yako irrigation scheme; the Loumbila dam that provides water to an irrigation scheme and public water supply to Ouagadougou the capital of Burkina Faso; the Ziga dam provides most of the public water supply to Ouagadougou and the Bagre Dam that produces hydropower and provides water to an existing irrigation scheme. Modelled existing infrastructure in Ghana includes the Bui dam and attached irrigation scheme; the Akosombo dam and the Kpong run-of-river hydropower plant and an irrigation scheme associated to Kpong. Also, aggregated public water supply for the Ghana Water Company Limited is modelled downstream of the PMD near the city of Tamale as well as aggregated abstractions from Lake Volta and an abstraction downstream of Kpong for Accra.

Modelled natural assets include FRA and pond fishing, downstream of the PMD. These benefits are made possible by the annual flood that occurs during the rainy season. Floods deposit nutrients that replenish the soil, enabling the practice of FRA

TABLE 1 | Summary of the existing and future built and natural infrastructure.

Name	Type	Size	Use	Status
Kanozoe	Built	Storage: 75 Mm^3 Irrigation: 5,319 ha	Irrigation, Fishing	Existing
Loumbila	Built	Storage: 50.9 Mm^3 Irrigation: 700 ha	Irrigation, Fishing, Public water supply	Existing
Ziga	Built	Storage: 470 Mm^3	Fishing, Public water supply	Existing
Kompienga	Built	Storage: 2,020 Mm^3 Hydropower: 14 MW	Hydropower	Existing
Bagre	Built	Storage: 2,320 Mm^3 Irrigation: 4,695 ha Hydropower: 16 MW	Irrigation, Fishing, Hydropower	Existing
Akosombo	Built	Storage: 155,500 Mm^3 Hydropower: 1,020 MW	Fishing, Public water supply, Hydropower	Existing
Bui	Built	Storage: 12,700 Mm^3 Planned irrigation: 30,000 ha Hydropower: 400 MW	Irrigation, Hydropower	Existing
Kpong	Built	Storage: Run-off-river Hydropower: 168 MW	Irrigation, Public water supply, Hydropower	Existing
PMD	Built	Storage: 2,622 Mm^3 Irrigation: 20,000 ha Hydropower: 59 MW	Irrigation, Fishing, Public water supply, Hydropower	Planned
Bagre irrigation expansion	Built	Up to 50,000 ha	-	Planned
Flood plain	Natural	-	Flood recession agriculture and pond fishing	Existing

by the local inhabitants. As the flood recedes, depressions in the banks are left with residual floodwater containing fish, allowing residents to perform pond fishing during an annual ceremony (Mul et al., 2017).

Performance Metrics

Performance metrics quantify the benefits generated by the built and natural assets in the Volta river basin. Metrics include annual and firm hydropower production, irrigation yield, prescribed environmental flows downstream of the PMD, and FRA benefits. The flood recession pond fishing benefit was considered in the simulation model, however, it was not incorporated in the optimisation process. The impact analysis on natural assets was focused on FRA because this activity represented greater financial benefits compared to the flood recession and pond fishing benefits.

Hydropower Generation

The Volta water model quantifies the average Annual Energy production [GWh year^{-1}], and the Firm Power at 90% exceedance of each hydropower dam:

$$HP_{t,n} = \eta g \gamma_w h_{t,n} u_{t,n}^{hp} \tag{2}$$

$$f_n^{AE} = max\left[\underset{\Xi}{mean}\left(\frac{1}{T}\sum_{t=1}^{T} HP_{t,n}\right)\right] \tag{3}$$

$$f_n^{FP} = max\left[\underset{\Xi}{mean}\left(HP_{90,n}:P\left(HP_{t,n} = HP_{90,n}\right) = 0.90\right)\right] \tag{4}$$

where, $HP_{t,n}$ is the hydropower generation; η is the turbine efficiency [%] (Akosombo = 0.93, Kompienga = 0.85, Bagre = 0.8, Bui = 0.85, Pwalugu = 0.9 and Kpong = 0.93); g is the gravitational acceleration constant, 9.8 m s^{-2}; γ_w is the water density [kg m^{-3}]; $h_{t,n}$ is the net hydraulic head [m]; T is the simulation time horizon and Ξ represents the hydrological ensemble. Finally, f_n^{AE} and f_n^{FP} are the average Annual Energy and Firm Power metrics, respectively.

Irrigation

Benefits of formal irrigation schemes are represented by irrigation yield. Each scheme's irrigation water demand and yield are estimated by using the Crop Water Requirements method proposed by the FAO (Allen et al., 1998):

$$CWR_{t,(ct\in n)} = max\left(0, Kc_{t,(ct\in n)}(ETo_{t,(ct\in n)} - R_{t,n})A_{(ct\in n)}\right) \tag{5}$$

$$IWR_{t,n} = \sum_{ct\in n} \frac{CWR_{t,(ct\in n)}}{\alpha_{ct}\beta_{ct}} \tag{6}$$

$$CR_{t,n} = \frac{u_{t,n}^{is}}{IWR_{t,n}} \tag{7}$$

$$f_n^{Y} = mean\left[\underset{\Xi}{mean}\left(\frac{1}{T}\sum_{t=1}^{T} CR_{t,n}\left(A_n y_n\right)\right)\right] \tag{8}$$

where, $CWR_{t,n}$ is the crop water requirement per node (irrigation scheme) and per month $(m \in t)$. $Kc_{t,(ct\in n)}$, $ETo_{t,(ct\in n)}$, and $R_{t,n}$ are the monthly crop water coefficient, effective evapotranspiration [mm day^{-1}] per crop and effective precipitation [mm day^{-1}], obtained from Sadick et al. (2015), respectively. $A_{(ct\in n)}$ is the area [ha] reserved for each crop type. $IWR_{t,n}$ is the monthly irrigation water requirement per irrigation scheme, α_{ct} and β_{ct} are the overall irrigation and conveyance efficiencies, assumed to be 0.8 and 0.7, respectively. $CR_{t,n}$ is a curtailment ratio, y_n is the annual yield [ton/ha] per irrigation scheme. Finally, f_n^{Y} is the average irrigation yield metric per irrigation scheme.

Environmental Flow

The environmental flow was calculated as a minimum flow that is exceeded 95% of the time according to PMD feasibility study (Volta River Authority, 2018), Eq. (9).

$$f_n^{EF} = max\left[\underset{\Xi}{mean}\left(q_{95,n}:P\left(q_{t,n} \le q_{95,n}\right) = 0.95\right)\right] \tag{9}$$

where, $q_{t,n}$ is the flow downstream [Mm^3 day^{-1}] of the PMD and f_n^{EF} is the environmental flow metric, maximised in the policy identification process under the hydrological ensemble Ξ.

Flood Recession Agriculture

Flood recession agriculture is dependent on the seasonal flooding of the floodplain during the peak of the rainy season (July–September). The magnitude of the annual peak (August or September) determines the total area sown by locals each year (Balana et al., 2015). Low flood peaks do not result in the river overflowing the banks, preventing FRA activities. Once the flooding threshold is breached, the flooded area increases with increasing flood peak. Extreme floods can negatively affect FRA by removing the fertile topsoil. Therefore, the area suitable for FRA reduces to zero for extreme flows (95% exceedance probability).

$$q_n^{FRA} = \frac{1}{T}\sum_{t} max\left(q_{(Aug)\epsilon t,n}, q_{(Sep)\epsilon t,n}\right) \tag{10}$$

$$Y_n = A_n(q_n^{FRA}) f_{FRA} C_y \tag{11}$$

$$f_n^{FRA} = max\left[\underset{\Xi}{mean}\left(\beta_{FRA} \times Y_n\right)\right] \tag{12}$$

where, q_n^{FRA} is the average annual flow in August or September during the simulation horizon; $q_{t,n}$ is the flow in August and September; $A_n(\cdot)$ is the flooded area [ha]; f_{FRA} is a suitability factor (Balana et al., 2015); C_y is the crop yield [ton ha^{-1}] assuming a typical FRA crop mix of maize, beans, Bambara beans, soya, millet and groundnuts (Sidibé et al., 2016); Y_n is the total FRA yield [ton year^{-1}]; β_{FRA} is the average regional market price of the crops at $1,222 ton^{-1} (Pettinotti, 2017). Finally, f_n^{FRA} is the benefit of the FRA activity.

Policy Identification

To evaluate the impact of new infrastructure operation it is necessary to identify and simulate the historical operational strategy of the existing infrastructure in the basin. Here, we use an explicit policy identification approach (Khadem et al., 2020),

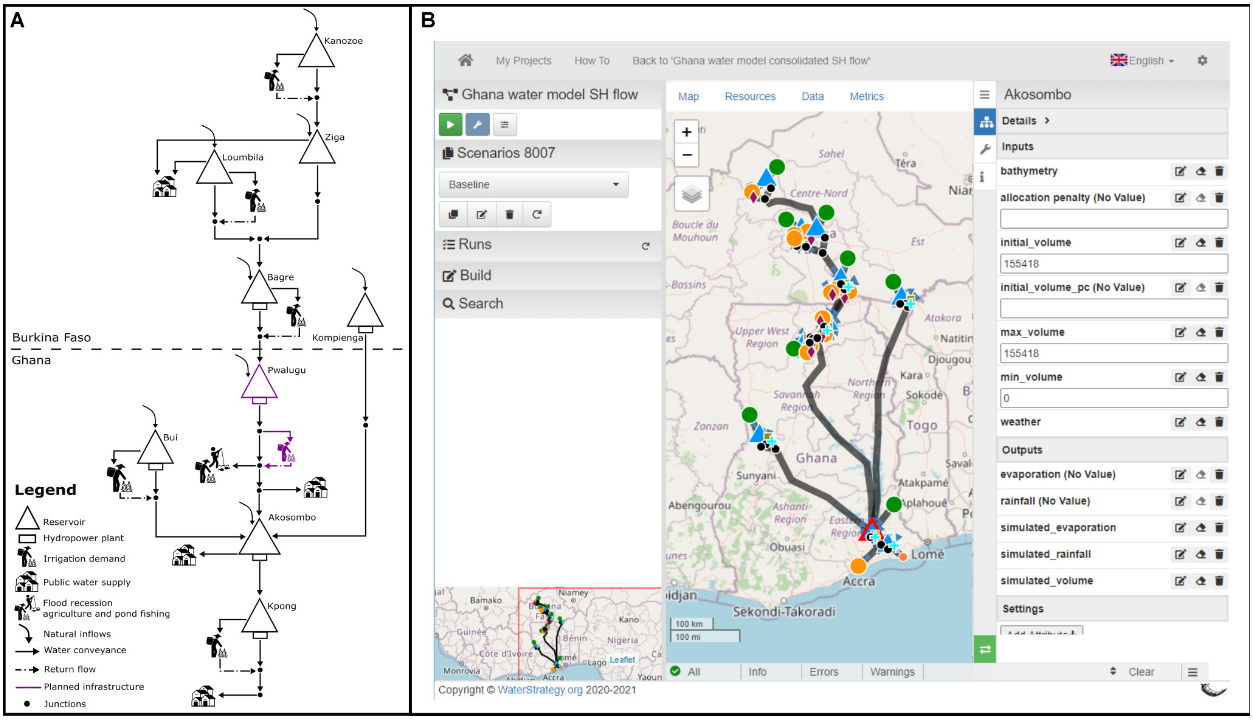

FIGURE 3 | Panel **(A)** shows the Volta river basin simulation model schematic, and panel **(B)** shows a screenshot of the WaterStrategy online interface (Knox et al., 2019) which allows collaborative development and use of the Python water resource simulation model (Pywr).

which requires a historical time series to estimate the operating rules of the existing infrastructure. The policy identification uses many-objective direct policy search to approximate operating rules of existing infrastructure using quantitative statistics metrics as objectives (Moriasi et al., 2007).

Operating Rules

We employed Gaussian radial basis functions (RBF) to model system operating rules. RBFs have shown good performance representing functions for a large class of problems, including reservoir storage and time into release decisions (Giuliani et al., 2014, 2016b; Zatarain Salazar et al., 2017; Geressu and Harou, 2019). The Gaussian RBF is defined by Eq. (13).

$$\varphi(\mathbf{x}) = \sum_{i=1}^{l} w_i \times exp\left[-\sum_{j=1}^{m} \frac{\left(x_j - c_{j,i}\right)^2}{b_{j,i}^2}\right] \quad (13)$$

where $m = 2$ is the number of input variables $\mathbf{x}$ (namely time and reservoir level); l is the number of RBFs ($l = 4$); w_i is the weight of the ith RBF (φ_i); $c_{j,i}$, and $b_{j,i}$ are the m-dimensional centres and radius vectors of the ith RBF, respectively. The centres and radius include: $c_{j,\mathrm{i}} \in [-1, 1]$ $b_{j,\mathrm{i}} \in [0, 1]$ $w_{j,\mathrm{i}} \in [0, 1]$ $\sum_{i=1}^{n} w_i = 1$. The parameter vector $\boldsymbol{\theta}$ is defined as $\boldsymbol{\theta} = \left[c_{j,\mathrm{i}}, b_{j,\mathrm{i}}, \mathrm{w_i}\right]$. For more details we refer the reader to Giuliani et al. (2014).

Historical Many-Objective Optimisation

Similar to Giuliani et al. (2016a) this study applies direct policy search, where the operating policy is parametrised using Gaussian RBF and then the parameters of the RBF are optimised using many-objective optimisation using the system's performance metrics (section "Performance Metrics") as objectives. In this work, in addition to optimising the operating policy parameters to maximise the system's metrics, we used the *Nash-Sutcliffe* efficiency metric to calibrate the operating policy parameters such that they reproduce observed reservoir volume time-series. Using observed historical data, we attempt to identify historical operating policies:

$$\mathbf{F}(\boldsymbol{\theta}_n) = \left(f_n^{AE}, f_n^{FP}, f_n^{NS}\right) n = Bagre,\ Bui,\ Kompienga,\ and\ Akosombo \quad (14)$$

where, f_n^{NS} is the *Nash-Sutcliffe* objective; $\boldsymbol{\theta}_n$ is the vector of decision variables which are the parameters for the reservoir operating rules. The objective function $\mathbf{F}(\cdot)$ vector is obtained by simulating the Volta river basin simulation model over the horizon T and the historical hydrologic scenario under the set of operating rules ($\boldsymbol{\theta}_n$) defined by the search algorithm.

Policy Optimisation

Historical Non-cooperative Scenario

The historical non-cooperative scenario is based on the Policy identification stage, where operating rules for existing dams were identified under historical hydrological conditions. This scenario assumes business as usual with dam operators focussing maximising their own benefits. In this scenario, the PMD maximises its benefits without consideration of the impacts on the downstream built infrastructure. We represent this scenario by a many-objective robust optimisation formulation that maximises the benefits generated by the operation of the new PMD and the downstream natural assets over a hydrological ensemble. In this study, robust optimisation is employed where the performance metric values of the simulations over hydrological scenario ensemble are aggregated into a single percentile value statistically (Herman et al., 2015; Kwakkel et al., 2016; McPhail et al., 2018). In this formulation the operating policies for existing infrastructure identified in stage 1 are static. Eq. (15) present the objective vector to be optimised:

$$\mathbf{F}(\boldsymbol{\theta}_n) = \left(f_n^{AE}, f_n^{FP}, f_n^{Y}, f_n^{EF}, q_n^{FRA}\right) n = PMD \tag{15}$$

where, $\boldsymbol{\theta}_n$ is the vector of decision variables which are the parameters of the PMD operating rules. The objective function $\mathbf{F}(\cdot)$ vector is obtained by simulating the Volta river basin simulation model over the horizon T and the hydrological ensemble Ξ under the set of operating rules $(\boldsymbol{\theta}_n)$ defined by the search algorithm.

Cooperative Scenario

This scenario assumes practical coordination and collaboration amongst the riparian countries and infrastructure operators, where institutional integration allows revising and possibly modifying the operation of existing dams to maximise basin-wide benefits. For the analysis, we only considered the Bagre and Akosombo dams because these two dams are upstream and downstream, respectively, of the PMD. The operation for the Bui and the Kompienga dams is considered static and uses operating rules identified in Stage 1. Eq. (16) present the objective vector to be optimised.

$$\mathbf{F}(\boldsymbol{\theta}_n) = \left(f_n^{AE}, f_n^{FP}, f_{PMD}^{Y}, f_{PMD}^{EF}, q_{PMD}^{FRA}\right) n = Bagre,\ PMD,\ and\ Akosombo \tag{16}$$

where, $\boldsymbol{\theta}_n$ is the vector of decision variables of the operating rules for the Bagre, PMD, and Akosombo reservoirs. Note, that the cooperative scenario formulation only includes the objectives f_n^{Y}, f_n^{EF}, and q_n^{FRA} for the PMD, whilst the f_n^{AE} and f_n^{FP} objectives are calculated for the Bagre, PMD and Akosombo reservoirs. Similar to the historical non-cooperative scenario, the objective function $\mathbf{F}(\cdot)$ vector is obtained by simulating the Volta river basin simulation model over the horizon T and the hydrological ensemble Ξ under the set of operating rules $(\boldsymbol{\theta}_n)$ defined by the search algorithm.

Hydrological Inputs

The Volta river basin simulation model requires historical inflow time-series and possible future hydrological scenarios to perform the Policy identification and the Policy optimisation, respectively. The historical inflow scenario is used to simulate and identify the historical system operation. Meanwhile, the possible future scenarios are used to evaluate the alternative policies over a wide range of hydrological conditions.

For the historical inflow scenario, given the lack of long-term observational flow data at all upstream reservoir locations, we used a reconstruction of the monthly historical naturalised river flows in the basin derived from a hydrological model, following Lin et al. (2019). We used the incremental flows at each dam location to represent the inflows to the reservoirs. Additionally, this historical scenario was used to generate a stochastic ensemble (Ξ) of hydrologic realisations for each reservoir located in the basin. We used the synthetic streamflow generation method proposed by Kirsch et al. (2013), which relies on Cholesky decomposition to maintain autocorrelation, and a bootstrap resampling technique to preserve multisite correlation (Herman et al., 2016).

Computational Experiment

For the Policy identification and the Policy optimisation we linked the Volta river basin simulation model to the Borg MOEA (Hadka and Reed, 2013) algorithm to solve the many-objective optimisation problem. This identifies the Pareto-approximate ("best achievable for any combination of priorities between objectives") set of reservoir system operating rules and quantifies the trade-offs between performance objectives that they imply. Borg has been shown to successfully handle complex, non-linear, and non-concave problems when searching for non-dominated solutions (Hadka and Reed, 2012; Zatarain Salazar et al., 2016). We used the default algorithm parametrisation, with an initial population of 100 individuals. Additionally, we set epsilon values equal to 1.0 for annual energy production, 0.1 for firm power, 0.05 for irrigation yield, 0.05 for environmental flow and 0.05 for FRA.

The Volta basin simulation model runs on a weekly time step (t) over a 20-year time horizon (T = 20 years) for both the historical and 30-member future ensemble (Ξ). The Policy identification and the Policy optimisation were run for 10 random seeds where each seed was run over 400,000 function evaluations (i.e., 4,000,000 simulations for the hydrological ensemble). The final Pareto-approximate curve is obtained as the set of non-dominated solutions from the combined results of all optimisations.

RESULTS AND DISCUSSION

First Stage: Historical Operation Identification

We use historical observed reservoir storage time series to identify the historical policy operating of existing infrastructure. Daily data were available for the Akosombo reservoir (01/01/1991 to 12/31/2010) and monthly data for the Bagre reservoir (01/01/2000 to 12/31/2010). In addition to the *Nash-Sutcliffe* efficiency objectives used to identify the reservoir operation for Akosombo and Bagre, we use system performance metrics (see section "Performance Metrics") to identify the operation for the reservoirs Bui and Kompienga, where historical time series were not available.

Figure 4 shows the historical observed versus the simulated volume for the Akosombo and Bagre reservoirs. The operating policy for the existing reservoirs was selected based on the maximum values for the Nash-Sutcliffe metric obtained in the optimisation process, 0.75 and 0.63 for the Akosombo

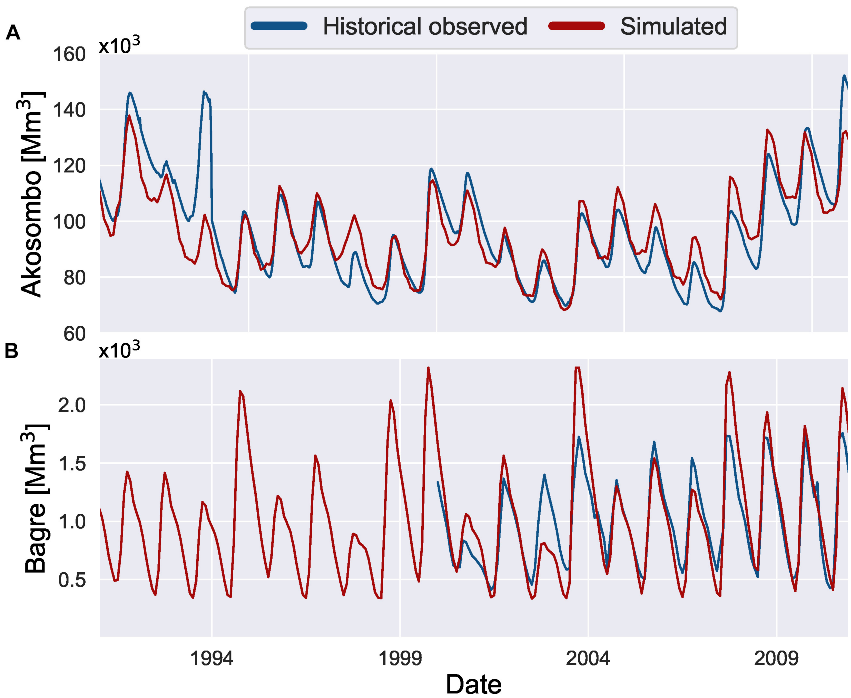

FIGURE 4 | Comparison of observed and simulated reservoir volumes used for policy identification. Panel **(A)** shows the observed and simulated volume for the Akosombo reservoir, and panel **(B)** shows the observed and simulated volume for the Bagre reservoir.

and Bagre Nash-Sutcliffe metrics, respectively. According to Moriasi et al. (2007), these results are classified as good and satisfactory, respectively. In general, the historical operating policy can reproduce the seasonality of both reservoirs. The storage volume is reduced every year almost to the minimum operating level in July and reservoirs fill in the wet season. However, in some years modelled storage is overestimated or underestimated and in the case of the Bagre reservoir, modelled storage reaches lower troughs as compared to the observed data. This is because the inflow time-series used by the simulation model are taken from the hydrological model and are subject to errors in their year-to-year variability.

Second Stage: Design and Adaptation of System Infrastructure Operation Policies

In the second phase of the method, we identify Pareto-approximate PMD reservoir release operating rules under two scenarios, historical non-cooperative and cooperative, where, in the non-cooperative, the Bagre and the Akosombo operating policies are held constant. Whilst in the cooperative we allowed Bagre and Akosombo operating policies to adapt to the new built infrastructure (the PMD) to maximise overall basin benefits. Results of the historical non-cooperative and cooperative scenarios (**Figure 5**) are visualised using a parallel plot (Inselberg, 1997). In **Figure 5**, each vertical axis represents an objective in the optimisation formulation whilst each coloured line represents a non-dominated operating policy portfolio for each new and existing dam. That point at which each coloured line intersects each respective vertical axis represents the performance of that policy in that objective. An ideal solution in the figure would lead to a straight horizontal line that intersects every axis at the top. Crossing lines between axes indicate trade-offs between two adjacent objectives. The “Ghana Average Energy” axis represents a non-optimised metric which is the sum of the Pwalugu and Akosombo average energy objectives. This metric is visualised in the figure to show the impacts of the upstream reservoirs’ operation (Bagre and Pwalugu) on the overall energy production in Ghana.

The full set of non-dominated portfolios for the historical non-cooperative and cooperative scenarios are presented in light blue and light red, respectively. We selected two policy portfolios from the historical non-cooperative scenario (solid and dashed line in blue) to evaluate in detail. The portfolio represented by the solid blue line achieves the highest PMD average energy 197.2 [GWh year^{-1}] in the set of historical non-cooperative portfolios, whilst at the same time the flow (at 95% exceedance) downstream of the PMD is greater than the minimum environmental flow (0.18 Mm^3 day^{-1}) requirement (Volta River Authority, 2018). The dashed line corresponds to an operating policy that achieves the highest PMD firm power 13.5 [MW]. Both operating policies, however, fall in the lower range of FRA benefits generating 9.3 [$M year^{-1}] and 6.3 [$M year^{-1}] for the portfolios represented by the solid and dashed lines, respectively. As under in the historical non-cooperative scenario we employed static operating policies for the Bagre and Akosombo dams (identified in the Policy identification stage), their range of performance is small across all Pareto-approximate policies.

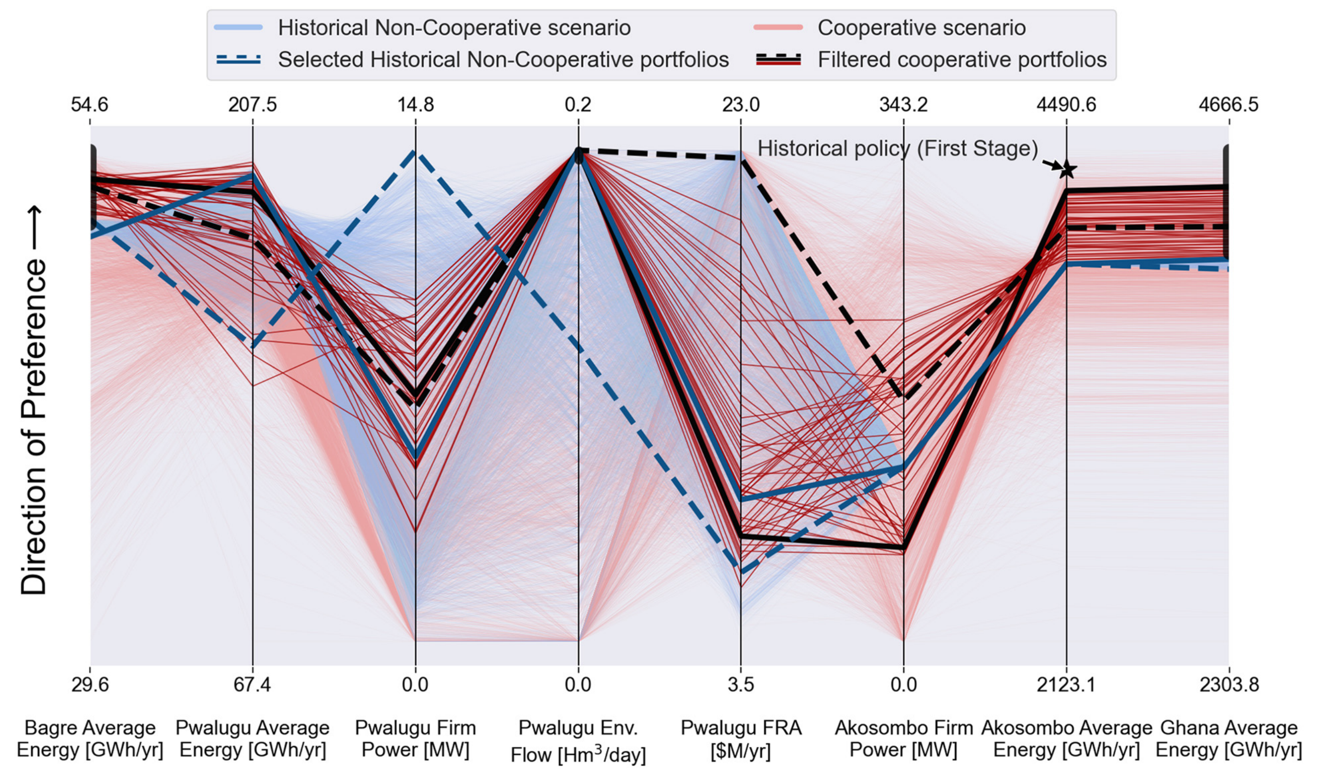

FIGURE 5 | Parallel axis plot comparing the historical non-cooperative and cooperative scenario. Light blue and light red are the full set of non-dominated solutions for the historical non-cooperative and cooperative scenario, respectively. Blue solid and dashed lines are selected portfolios of the historical non-cooperative scenario and the black and solid red lines are filtered portfolios of the cooperative scenario. Vertical thick black bars over parts of the first, fourth and eighth axes are "filters" which allow selecting a subset of the efficient designs in the plot. The Akosombo average energy [4,457.8 GWh year^{-1}] of the historical policy, identified in the First Stage, is shown with a black star.

Cooperation between the two riparian countries is more likely to occur if both countries receive a net benefit as a result of the cooperation (Sadoff and Grey, 2009; Jeuland et al., 2017). The solutions for the cooperative scenario are filtered such that those that only increase energy generation from Bagre in Burkina Faso 51.1 [GWh year^{-1}], and energy generation in Ghana 4151.3 [GWh year^{-1}] as compared to the historical non-cooperative scenario are seen in bold colours. We also filter out all policies that result in flows downstream of the PMD that are greater than 0.18 [Mm^3 day^{-1}], the minimum environmental flow identified in Volta River Authority (2018). Amongst the remaining cooperative solutions, we select two the "Built Infrastructure portfolio" represented by the solid black line maximises Ghana Average energy (producing 4490.7 [GWh year^{-1}]) whilst the "compromise portfolio" represented by the dashed black line maximises ecosystem services in Northern Ghana by maximising FRA benefits (generating 22.5 [$M year^{-1}]).

Figure 5 shows how that energy generation and firm power from the Akosombo dam may be impacted if the historical operating policies of existing dams are not adapted to the new flow regime resulting from PMD development (blue lines). The filtered cooperative portfolios (red lines) show that implementing cooperative operating policies results in Akosombo hydropower benefits (average energy and firm power) being less negatively impacted than if non-cooperative policies were implemented. In addition, hydropower benefits at Bagre can be increased, and environmental flows downstream of the PMD can be maintained.

According to Baah-Kumi and Ward (2020), the PMD development may impact Akosombo operation because the inflows to Lake Volta could be reduced due to increasing water storage, water demands and evaporation upstream of Lake Volta. Our analysis confirms this conclusion; however, **Figure 5** shows that the negative impact of Akosombo could be greater without the coordinated infrastructure management. For example, Akosombo average energy is reduced by 11.5% from the energy generated historically (black star in **Figure 5**), in the selected non-cooperative scenarios. However, if cooperative policies are implemented this impact is reduced to only 3.6% (solid black line in the **Figure 5**). This policy achieves a positive net energy production of 0.74% in Ghana and 6.2% for Bagre in Burkina Faso (see **Figure 5**). The FRA benefits are reduced by 63.5% from the historical 25.2 [$M year^{-1}], in the selected non-cooperative portfolio (solid blue line). Whilst under the cooperative portfolio, which maximises hydropower benefits (solid black line) the FRA benefits are reduced by 69.4%. However, note that under the compromise operating policy (dashed black line), the FRA benefits could be reduced only by 10.7%.

Third Stage: Implications for Infrastructure Operation on the Volta River Basin

To investigate the possible range of benefits generated in the Volta basin with the identified operating policies we performed a sensitivity analysis by simulating the system over 100 hydrological scenarios. Policies considered in this analysis include the selected cooperative and non-cooperative policies as well as those in the entire cooperative filtered set in **Figure 5**.

Figure 6 shows the exceedance curves for the different metrics and different operating policies. **Figures 6A,B** visualize the PMD hydropower metrics while **Figures 6C,D** show metrics related to the new irrigation scheme and the impact to the environmental services – FRA – downstream of the PMD at Pwalugu in the

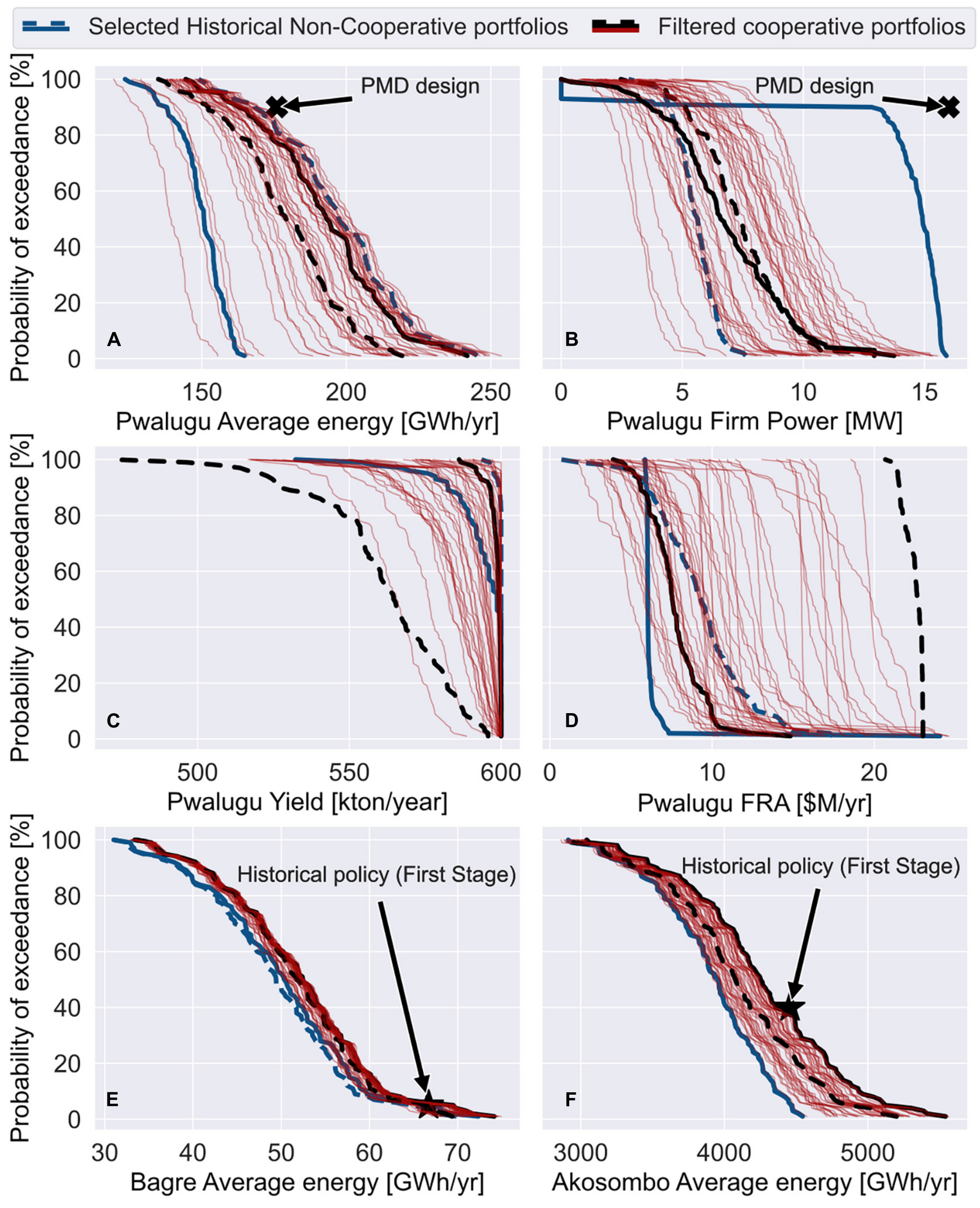

FIGURE 6 | Probability of exceedance curves for different metrics in the system. Panel **(A)** shows the Pwalugu Average energy, panel **(B)** shows the Pwalugu Firm Power, panel **(C)** shows the Pwalugu Irrigation Yield, panel **(D)** shows the Pwalugu Flood Recession Agriculture (FRA), panel **(E)** shows the Bagre Average energy, and panel **(F)** shows the Akosombo Average energy. The PMD average energy and firm power design (Volta River Authority, 2018) is displayed in panels **(A,B)**, respectively.

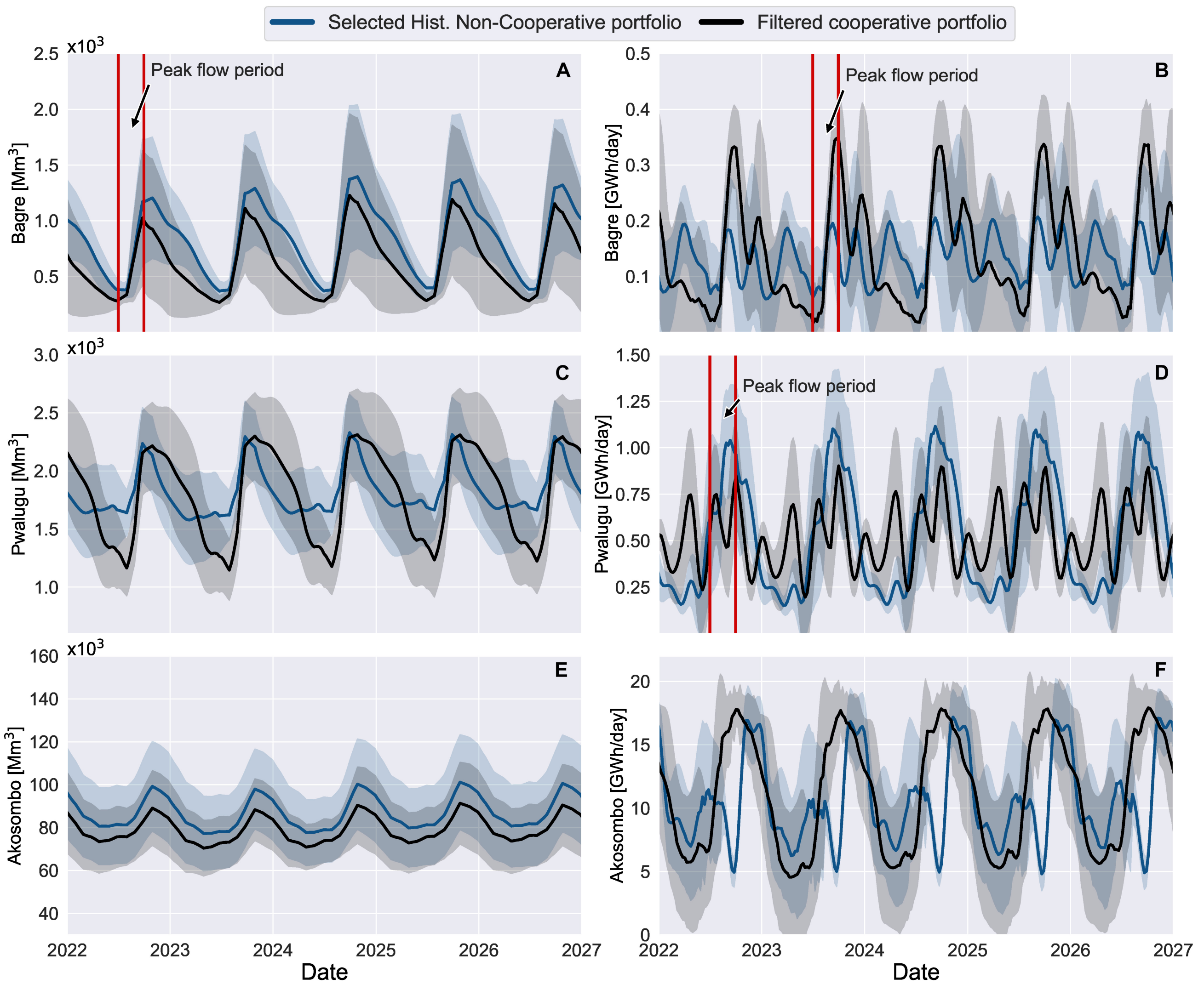

FIGURE 7 | Reservoirs operation. Panel **(A,C,E)** show the mean value under the hydrological scenarios and the standard deviation (light blue and grey area) of the stimulated reservoir volume in Bagre, Pwalugu, and Akosombo, respectively. Panel **(B,D,F)** show the mean value under the hydrological scenarios and the standard deviation of the simulated energy production for the Bagre, Pwalugu, and Akosombo dams, respectively.

Upper East Region of Ghana. Finally, **Figures 6C,F** show the average energy produced at Bagre and Akosombo, respectively. In general, the exceedance curves show that the operating policies identified by the cooperative scenario perform better according to almost all metrics for all hydrological scenarios. The selected cooperative policy that maximises Ghana Average energy (solid black line in **Figure 6**) shows that the Pwalugu average energy and firm power (**Figures 6A,B**) is 162.7 [GWh year^{-1}] and 5.1 [MW] at 90% of exceedance, respectively. Note that the PMD was designed to produce 176 [GWh year^{-1}] with a firm power of 16 [MW]. According to our results the designed annual energy is achieved at only 77% exceedance under the same cooperative policy (**Figure 6A**).

Figures 6C,D, show that a trade-off exists between irrigation yield and FRA downstream the PMD. Policies that result in higher benefits in FRA in Pwalugu result in lower irrigation yields from the formal irrigation scheme associated with the PMD. For example, if the selected cooperative policies (solid and dashed black lines) are compared, the formal irrigation yield under the cooperative policy that maximises Ghana Average energy is 596.0 [kton year^{-1}] at 90% of exceedance whilst the FRA benefit is 8.7 [$M year^{-1}]. However, under the portfolio that maximises the FRA benefits, the formal irrigation yield is reduced to 526.9 [kton year^{-1}] and the FRA benefits increase to 21.6 [$M year^{-1}]. The flood recession ecosystem services benefits occur downstream of the PMD in the floodplains. The irrigation system abstracts water downstream of the PMD hydropower releases. These abstractions result in smaller downstream floods and lower overall flood recession benefits, implying there is direct competition between formal and informal irrigation. **Figures 6E,F** show the Bagre and the Akosombo average energy production in the cooperative scenario outperform the energy production in the historical non-cooperative scenario under all hydrologic scenarios.

Figure 7 shows five-operation-years for the Bagre, Pwalugu, and Akosombo reservoirs, for the operating policy represented by the black and blue solid lines under the hydrological scenarios. **Figures 7A,C,E** show the mean value under the scenarios and the standard deviation (light blue and grey area in **Figure 7**) for the simulated reservoir volume in Bagre, Pwalugu, and Akosombo, respectively. While **Figures 7B,D,F** show the same information but for the simulated energy production for the Bagre, Pwalugu, and Akosombo dams.

According to **Figures 7A,E**, the mean volume for Akosombo and Bagre follow a similar seasonal operating pattern as compared to the historical operation of both reservoirs (see section "First Stage: Historical Operation Identification"), where reservoir storages reduce to lower volumes in the dry season and fill in the wet season. However, building and operating the PMD under the selected cooperative policy (black line) results in lower volumes in Akosombo in all seasons compared to the historical operation without the PMD development (see **Figure 4**). This results from the increase in upstream storage, water consumption, and evaporation and the increased hydropower generation in peak flow periods (see **Figure 7F**) to maintain historical energy generation levels. The Pwalugu reservoir operation follows a similar seasonal pattern to the Bagre reservoir, where the storage volume is reduced every year almost to the minimum operating level in July and reservoirs fill in the wet season. However, this operation for Pwalugu is more pronounced (lower reservoir volumes) in the cooperative scenario. This could potentially reduce flood peaks downstream of the PMD by increasing Pwalugu's capacity to buffer possible spills from Bagre protecting from floods in Northern Ghana. However reducing the flood peak would also reduce the ecosystem-services derived from this flood (i.e., FRA) (see **Figure 7D**) and (**Figure 6D**), as well as increase risks that the dam is not able to refill in some years.

Under the cooperative policy (black line), on average Bagre operates at a lower volume and increases its hydropower generation during peak flow periods, as compared to the historical non-cooperative scenario (see **Figures 7A,B**). This added to steady Pwalugu hydropower releases (see **Figure 7D**) increases the water availability downstream minimising the negative impacts on the Akosombo energy production, as compared to the non-cooperative scenario. However, increasing energy production in Bagre during peak flows reduce its ability to generate hydropower during off-peak flows periods (**Figure 7B**). This could possibly increase energy import dependency in Burkina Faso during these periods despite total annual generation by Bagre being greater in the cooperative scenario. This possible negative impact for Burkina Faso could be offset by increasing energy trades between Ghana and Burkina Faso during those periods.

Future Work and Limitations

The analysis presented in this paper considers hydrologic uncertainty within the robust optimisation. The stochastic ensemble used in the optimisation process is based on the historical flow regime in the basin. Climate change may modify this flow regime and could impact the long-term robustness of the operating policies identified in the study. Future studies could apply the proposed framework under climate change uncertainty to identify cooperative water strategies that are robust to climate change and support long-term sustainable water resources management. Another potential direction for future research could be to explore the impacts of political or economic uncertainty or the consequences of deviation from operating agreements by each party via optimisation or sensitivity analysis.

CONCLUSION

This paper explores the multiple potential impacts of the proposed Pwalugu Multipurpose Dam (PMD) could have on existing infrastructure and water services of the Volta river basin. We presented a novel framework that links river basin simulation to many-objective search to approximate existing operating rules and design new ones in the presence of complex multi-actor trade-offs. The approach identifies benefits of collaborative management in systems where current river management is not codified in formal regulations, as is the case in many multi-region river basins in the global south. We evaluated the possible impacts that new development could impose on ecosystem services and downstream infrastructure and how the PMD itself could be impacted by different operating strategies of the existing upstream Bagre Dam in Burkina Faso under two alternative scenarios assuming cooperation and non-cooperation.

Results indicate that the PMD has the potential to reduce inflows into the Akosombo dam. This would likely have a negative impact on hydropower generation from Akosombo, with up to 11.5% reduction in energy generation compared to the case without the PMD. However, if cooperative infrastructure operation is adopted, the impact could be reduced by only 3.6%, and result in a positive net energy production of 0.74% in Ghana overall and a 6.2% increase for Bagre Dam hydropower production in Burkina Faso. Environmental services in Northern Ghana may also be impacted in a non-cooperative operation scenario decreasing floods downstream of the PMD and possibly reducing benefits to local communities that depend directly on flood recession activities. Under a cooperative scenario, the Bagre reservoir could increase hydropower releases during peak flow periods increasing its total net annual generation. This operative strategy could increase water availability downstream of the PMD minimising the negative impacts on Akosombo energy production, resulting in net positive energy production in Ghana even after constructing the PMD. However, such a strategy could reduce Bagre energy generation during low flow periods increasing Burkina Faso's dependence on energy imports from Ghana during the dry season, despite the higher overall total annual generation. Results showed there is room for riparian countries to negotiate cooperative strategies to offset possible negative impacts generated by the new PMD. The PMD provides an opportunity for the VBA to implement cooperative operational water management strategies in the basin.

DATA AVAILABILITY STATEMENT

The raw data supporting the conclusions of this article will be made available by the authors upon consultation with the relevant national authorities who own the data.

AUTHOR CONTRIBUTIONS

JG: conceptualization, methodology, software, formal analysis, writing – original draft, and visualization. EM: conceptualization, methodology, software, formal analysis, and writing – review and editing. EO, AB-B, and JD: conceptualization, and writing – review and editing. MM, LP, SG, and JS: methodology and writing – review and editing. DS: conceptualization, writing – review and editing, and funding acquisition. JH: conceptualization, supervision, and writing – review and editing, and funding acquisition. All authors contributed to the article and approved the submitted version.

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Towards Good E-Flows Practices in the Small-Scale Hydropower Sector in Uganda

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Stakeholders of the small-scale (<50 MW generation capacity) hydropower sector in Uganda recognise the importance of sustainable development of the resources that have social and ecological importance. Uganda is experiencing a boom in hydropower projects resulting in over generation of electricity and its exportation to neighbouring nations. Limited policies are currently available in Uganda to direct the sustainable development of this sector. Environmental flows (e-flows) practices established for the Nile Basin region and international good e-flows practices can contribute to sustainable management of hydropower developments in Uganda. The paper defines and explains e-flows, identifies water resource attributes of importance for e-flows determination associated with hydropower and threat associated with this activity in Uganda, and provides good e-flows determination and management practices based on regional and international information. The determination and management of e-flows in the hydropower sector in Uganda is largely dependent on the availability of and quality of hydrology, hydraulic and flow-ecosystem and flow-ecosystem service relationship information. This review of good-practice e-flows practice for the small hydropower sector in Uganda provides guidance to support multiple stakeholders of water resources in Uganda for a better future for all of its vulnerable communities and the environments they depend on.

Keywords: hydropower, environmental flows, river ecology, Uganda, sustainability, water resources management and development

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INTRODUCTION

Hydropower is the primary source of electricity generation in Uganda. It accounts for 78% of the total installed capacity of 1182.2 MW, generating 3330 GWh of electricity (ERA, 2019). Large-scale hydropower has an installed capacity of 813 MW, 68% of total installed capacity, characterized by a number of stations on the Victoria Nile, including the Bujagali (250 MW), Kiira (200 MW), Nalubaale (180 MW), the Isimba Falls (180 MW) which became operational in early 2019. Smaller hydropower developments (<50 MW in capacity) currently produce a combined 176 MW, which totals 22% of Uganda's installed power generation capacity (sensu ERA, 2019). The small-scale hydropower plant sector in the Africa is developing rapidly with numerous Independent Power Producers (IPPs) developing plants in more than half of the countries in Africa (Moner-Girona et al., 2016; O'Brien et al., In press). These plants are considered to have better cost-benefit ratios than large hydropower dams and fossil fuel power generation plants due to their relatively low cost of installation, robustness and longevity, and importantly the potential to access remote communities who have a high demand for power (Pang et al., 2015; O'Brien et al., In press). This sector is expected to grow considerably in the near future (O'Brien et al., In press). Hydropower plants almost inevitably have pernicious impacts on the wellbeing of river ecosystems and the livelihoods of people using their ecosystem services (McCarthy et al., 2008; Liechti et al., 2015; Lynch et al., 2019). These impacts could include disruptions in the connectivity of river habitats, and/or changes in the volume, timing, duration and frequency of flows and indirect impacts on other environmental variables such as water quality. While the management of the flows of water in rivers is well established globally, in Africa developments from South Africa in particular has dominated sustainable water resources management (King and Pienaar, 2011; Nile Basin Initiative, 2016; GET FiT, 2018; O'Brien et al., In press). African nations including Kenya and Tanzania have directly included South African water resource management policies into their legislation, and the Nile Basin Initiative has included components of South African policies as good practice into their water resource management (Nile Basin Initiative, 2016; O'Brien et al., 2018; Dickens et al., 2019; O'Brien et al., In press). In Uganda water resource developers, conservationists, scientists and regulators have limited guidance on the local effect of altered flows associated with hydropower development. While the Nile Basin Initiative (NBI) provides regional guidance on e-flows and how to manage multiple stressors (Nile Basin Initiative, 2016), it does not provide good practice guidance that is specific to the water resources in Uganda to allow these Ugandan stakeholders to manage the effects of altered flows associated with small-scale hydropower development. Sustainable water resource management that considers e-flows is essential to ensure that development in Uganda does not negatively impact on vulnerable ecosystems and the human communities who depend on these systems for their livelihoods.

Management of the e-flows of a river is recognized as a possible way of mitigating the impacts of hydropower plants, and indeed understanding the e-flows of all rivers has become a corner-stone of water resources management (Horne et al., 2017). Since the 1990s e-flows are generally now included not only in water resources planning but also as part of the mitigation of individual river developments including hydropower schemes (Poff and Zimmerman, 2010; Pahl-Wostl et al., 2013; Poff and Matthews, 2013; O'Brien et al., 2018). Environmental flows also now form an essential part of the indicator method for the Sustainable Development Goal SDG 6.4.2 indicator of the degree of "water stress" being exerted on a water resource (Vanham et al., 2018; Dickens et al., 2019), and thus should be on the agenda of all water-resource managers.

It was only after the 1990s that the effects of altered flows in the environment began to be considered in a dedicated manner (Pahl-Wostl et al., 2013; Horne et al., 2017) This transpired after extensive dam construction in particular led to large scale obstruction of free-flowing rivers and a noticeable loss of ecosystem services, including fish stock in particular and natural habitats and biodiversity (Poff and Matthews, 2013). The prevailing question became to understand the water flow-ecology relationships, the human impacts on the delivery of ecosystem services and above all the question; "how much water does a river need to sustain itself and the livelihoods of people who depend on them" (Pahl-Wostl et al., 2013).

Over subsequent decades, the concept of e-flows has evolved to encompass river flow variability, river connectivity (longitudinal and lateral), ecosystem services and human wellbeing and a suite of methods have been developed and applied globally to quantify various components of e-flows (Tharme, 2003; Petts, 2009; Adams, 2014). Today various e-flows policies (King and Pienaar, 2011; Nile Basin Initiative, 2016), frameworks and determination methods (Poff et al., 2010; Nile Basin Initiative, 2016) are available to contribute to the development of "good" e-flows practices for Uganda.

Implementing a strategy to provide for e-flows is essential for hydropower development primarily because e-flows provide a boundary for development, providing a measure of river flow that should not be lost as it is needed to sustain the ecosystem and the people who rely on that ecosystem, and also because implementation of e-flows provides a way to mitigate the impact of the hydropower development on a river system. This paper aims to review and recommend e-flows determination and management practices applicable to the small-scale hydropower sector in Uganda and the sustainable management of water resources in Uganda across multiple spatial scales. The paper defines and explains e-flows, identifies water resources of importance for e-flows determination associated with hydropower development in Uganda, and recommends appropriate e-flows determination and management practices based on regional information.

Water Resources in Uganda

Uganda is a landlocked nation of 241550.7 km^2, located within the equatorial regional of Africa (**Figure 1**). The nation receives an annual average rainfall of 1180 mm primarily during two rain seasons (Nsubuga et al., 2014). Uganda is located within the White Nile Basin between Lake Victoria and the Sudd Wetland of the Nile basin and has 16% of its area covered by lakes or wetlands. This nation has abundant water resources and is ideally located within a region of Africa with a high demand for electricity and abundant potential for hydropower. The landscape of Uganda consists of a high altitude (1,050 m.a.s.l.) plain within the rift valley of Africa, with mountains in the east (Elgon Mountain 4,321 m.a.s.l) and in particular along the west of the country (Rwenzori and Virunga mountains, maximum 5,119 m.a.s.l. The average flow (discharge in m^3/s) from the major lakes into the Nile Rivers in Uganda has been variable, particularly when comparing the "dry period" of 1905–1961 with average flows of 840 m^3/s and the wet period from 1962 to 2008 with average flows of > 1200 m^3/s (Nsubuga et al., 2014). In Uganda the population has increased from 12 Million people in 1980 to 44 Million in 2008 (Nsubuga et al., 2014). All of these people depend on the water resources of the nation and the services it provides. Sustainable development and management of these resources for the biodiversity and livelihoods of vulnerable Ugandans is imperative.

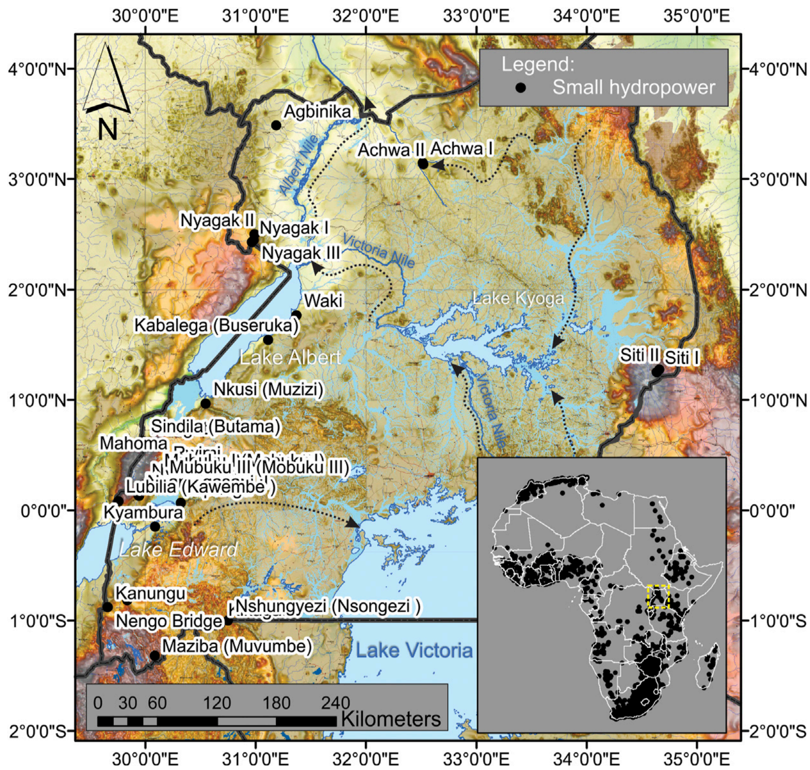

FIGURE 1 | Map of Uganda, with lakes and rivers and direction of flow (dotted lines). Small (<50 MW capacity) hydropower plants included.

Small-Scale Hydropower in Uganda

There are currently > 29 small-scaled hydropower plants in operation or being built in Uganda (**Table 1**; **Figure 1**). With the new developments power production of small plants will increase production in Uganda to at least 332 MW (34% of hydropower generation in Uganda, (**Table 1**). Small-scale hydropower stations are concentrated in the Rwenzori Mountains and Mount Elgon, and also on less turbulent reaches of river in lowland, lower rainfall regions, with varying ecology and land use systems compared to the large-scale stations on the Victoria Nile. Many of the developments (n > 14) have been commissioned through the GET FiT (Global Energy Transfer Feed-in Tariff scheme) program. Launched in 2013 with the intention to leverage private investment for renewable energy generation in Uganda, it has been developed by the Government of Uganda, the Electricity Regulatory Agency (ERA) and KfW, receiving funding from international donors. The program includes 20 small-scale renewable energy generation projects, including a number of small-scale hydropower stations, as well as solar and bagasse (**Table 1**). As of 2020, 139 MW of installed capacity is in operation, generating > 271 GWh, 7% of total electricity supplied to Uganda, including fourteen small-scale hydropower stations (GET FiT, 2018).

In recent years Uganda has experienced periods of electricity over-capacity, allowing export to neighboring Tanzania, Kenya, Rwanda and the Democratic Republic of Congo (ERA, 2019). Peak electricity demand reached 644 MW in 2018 (ERA, 2018), with over-capacity increasing since the Isimba Falls (180 MW) became operational in early 2019. Uganda faces a period of further over capacity in light of the 600 MW Karuma and the 600 MW Ayago stations presently under construction on the Victoria Nile, due to be operational by 2020 and 2026, respectively; the Achwa 1 and 3 stations on the River Achwa (41 and 10 MW); the Muzizi station (48 MW) near the south-east shores of Albert Lake; and the 157 MW GET FiT renewable energy portfolio when fully operational. With slow to moderate growth in industrial demand and slow increase in rural household connections, this in turn may mute prospects for further deployment of private sector-developed smaller scale hydropower projects in the near to mid-term. However, experience from other African countries where electricity became available (e.g., South Africa had 1.3 million new connections over a three year period from 2006–9) has shown very rapid uptake, within the context of a conducive institutional, economic and operational environment (RoSA, 2011).

Improving electricity transmission and distribution would allow greater levels of domestic consumption in Uganda, especially considering that only an estimated 22% of the population and 14% of households have access to electricity (World Bank, 2019). Electricity consumption in Uganda is amongst the lowest levels in the world, half the average of Sub-Saharan African countries (Kamese, 2004). Biomass is a critical source of energy for the majority of the population,

TABLE 1 | Summary of the small (<50 MW capacity) hydroelectric power plants in Uganda with developer, capacity, commissioning date and involvement as a GET FiT project (GET FiT, 2018).

Name	River	Latitude	Longitude	Project special purpose vehicles/Developer	Capacity (MW)	Commission date	GET FiT
Achwa I	Achwa	3.148056	32.514167	Berkeley energy	10.0	WIP	NO
Achwa II	Achwa	3.135000	32.520833	Berkeley energy	41.0	2019	NO
Agbinika	Kochi	3.485416	31.185417	Uganda government	20.0	WIP	NO
Bugoye (mobuku II)	Mubuku	0.309038	30.102083	Bugoye hydro limited	13.0	2012	NO
Kabalega (buseruka)	Wambabya	1.545485	31.111478	Hydromax limited	9.0	2013	NO
Kanungu	Ishasha	-0.878611	29.657500	Eco-power limited	6.6	2011	NO
Kikagati	Kagera	-1.029090	30.679243	Kikgati power company limited	16.0	WIP	Yes
Lubilia (kawembe)	Lubilia	0.083333	29.754444	Lubilia kawembe hydro ltd	5.4	2018	Yes
Mahoma	Mahoma/ Dura	0.478611	30.273056	Mahoma Uganda limited	2.7	2018	NO
Maziba (muvumbe)	Muvumbe	-1.318750	30.082957	Muvumbe hydro Uganda limited	6.5	2017	Yes
Mpanga	Mpanga	0.067388	30.321557	Africa energy management system	18.0	2011	NO
Mubuku I (mobuku I)	Mubuku	0.318611	30.100000	Tibet hima mining Co. ltd	5.0	1956	NO
Mubuku III (mobuku III)	Mubuku	0.260278	30.149444	Kasese cobalt company limited	9.9	2009	NO
Nengo bridge	Mirera	-0.814583	29.833370	Jacobsen elekro	6.5	WIP	NO
Nkusi (muzizi)	Nkusi	0.966206	30.546295	PA technical services Uganda limited	9.6	2018	Yes
Nshungyezi (Nsongezi)	Kagera	-1.000239	30.745595	Nsongezi power company limited	39.0	WIP	NO
Nyagak I	Nyagak	2.430556	30.963889	-	3.5	2012	NO
Nyagak II	Nyagak	2.500028	30.989583	Public private partnership	5.0	2018	NO
Nyagak III	Nyagak	2.449945	30.981250	-	4.4	WIP	NO
Nyamwamba	Nyamwamba	0.230850	29.985817	Africa EMS Nyamwamba limited	9.2	2018	Yes
Rwimi	Rwimi	0.390055	30.181250	Rwimi EP company limited	5.5	2017	Yes
Siti I	Siti	1.250000	34.636944	Elgon hydro siti (PVT) limited	5.0	2017	Yes
Siti II	Siti	1.276389	34.657778	Elgon hydro siti (PVT) limited	16.5	2016	Yes
Sindila (butama)	Sindila	0.630000	29.978056	Butama hydro-electricity company ltd	5.3	2019	Yes
Waki	Waki	1.766116	31.368750	Hydromax Nkusi ltd	4.8	2018	Yes
Kyambura	Kyambura	-0.148889	30.088611	Zibra limited	7.6	2019	Yes
Ndugutu	Ndugutu	0.615556	29.979444	Ndugutu power company Uganda limited	5.9	2019	Yes
Nyamagasani I	Nyamagasani	0.137778	29.934722	Rwenzori hydro private limited	15.0	WIP	Yes
Nyamagasani II	Nyamagasani	0.130000	29.942500	Nyamagasani 2 hydroelectric power project limited	6.0	WIP	Yes

particularly in rural areas, accounting for an estimated 90% of Uganda requirements (UNDP, 2014).

Effects of Hydropower Developments and Operation on River Ecosystems

Hydropower plants impact on river ecosystems in numerous ways; mainly through the alteration of the river water and sediment flow regimes, cause water quality impacts, barriers and disturbance to wildlife stressors (Moog, 1993; Rolls and Bond, 2018; O'Brien et al., In press). Generally, small scale hydropower production is unique in that it can be non-consumptive where water is diverted through hydropower generation infrastructure and then returned to the river system (**Figure 2**). When there is no overall impact on river flows, small hydro developments are termed "run of river" which comes together with the connotation that these types of hydropower plants do not cause any negative ecological impacts (Anderson et al., 2015) (**Figure 2**). However, the synergistic impacts related to the formation of barriers, change in erosion dynamics and water quality of the rivers as well as alterations in the timing duration and frequency of flows on daily and seasonal scales must be considered (Anderson et al., 2015; O'Brien et al., In press).

Small-scale hydropower generally can contribute to the following ecosystem issues:

1. Alterations to the natural flow regime (consider **Figure 2**).
   a. Where there is storage as part of the hydropower development, storage dams can buffer natural flow variability, and may provide opportunity for other users of water to withdraw water from the storage and thus reduce volumes (Anderson et al., 2015; Rolls and Bond, 2018).
   b. The configuration of dam/weir and downstream powerhouse/tailrace impacts particularly on the river section below the dam and above the tailrace, which in some situations may become a dewatering zone/reach (**Figure 2B**). Flows in this section tend to fluctuate widely depending on operation of the system, resulting in a highly stressed ecosystem (**Figures 2B,D**). This is particularly exacerbated in Uganda when the power grid receiving power from hydropower facilities experiences failures (trips) and the hydropower plant must immediately cease supplying power. During these occurrences, normal operation of the plant ceases with water being diverted from the headrace, over the weir into

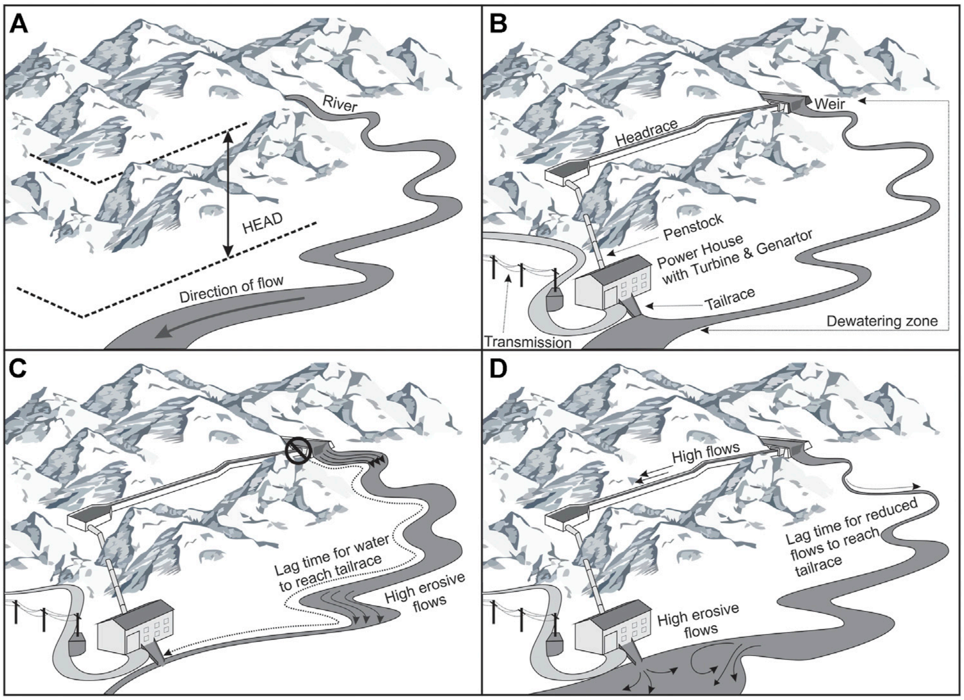

FIGURE 2 | Diagram of a typical small (<50 MW) hydropower installation build on a natural landscape **(A)**. The infrastructure associated with the development is indicated in **(B)** and the effect of rapid flow (volume) alterations on the rivers is demonstrated in Figure **(C,D)**. Rapid flow alterations can be associated with a loss of transmission capacity where the diversion of flows into the headrace of the facility is cut off **(C)**, this results in a decrease (no return) of flows in the river below the tailrace for the duration of the down time. During this period, elevated flows flow over the weir and down the dewatering zone during a lag period until flows reach the tailrace. When power is restored to the grid **(D)** Additional flows in the penstock and headrace are added to flows in the river, and usually elevated to "ram" the grid to energize it. During this period, additional flows are released from the tailrace into the river and the dewatering zone flows are noticeably reduced or cut off until normal operation is resumed **(B)**.

the dewatering zone. During this period flows reduce significantly downstream of the power plant as there is a lag period for the water being diverted from the weir to reach the power plant. An example of this was monitored by the authors on 2–3 October 2018 at the Rwimi plant (**Table 1**), which experiences numerous such transmission restriction events, two of which were observed. During transmission loss events, following the halting of water flow into the headrace, automatic valves in the power house divert water over the weir into the dewatering zone. It took 22 min (**Figure 2C**) for this diversion to reach the powerhouse inundating the dewatering zone. During this lag period, flows below the power house reduced from 4.9 m^3/s (± 0.5 m^3/s) to >1 m^3/s in less than 5 min, which was maintained during a lag period as diverted flows inundated the dewatering zone for 22 min. Flows below the power plant then returned to 4.2 m^3/s and were sustained during the down period for approximately 6 h until power generation resumed. In the power plant when the turbines were activated approximately 6 h after transmission loss (consider that this period is highly variable), the hydropower plant needed to energize the transmission grid, temporarily increasing electricity generation by ramming excess water stored in the penstock, headrace and weir. This resulted in an additional rapid increase in observed flows downstream of the power plant from 4.2 to 7.1 m^3/s that was planned to be maintained from approximately 20 min after which generation would be reduced and maintained. Unfortunately after approximately 15 min of ramming the flows through the power plant to energize the grid another transmission failure occurred. And the flows downstream of the plant again returned to <1 m^3/s. During normal operation flows returned to 4.2 m^3/s. These events were reported to occur consistently (on at least three to five occasions per week) on this single plant at Rwimi (Rwimi EP Company Limited, 2018) (shape of these flows demonstrated graphically in **Figure 3A**). These radical changes in flow will have serious impacts on the ecosystem of the dewatering zone as well as for a distance below the tail-race. It was observed that many invertebrates died in a few minutes, and the local people flocked to the river to pick up stranded fish. Rapid increases in flow do occur in nature, so most river ecosystems are adapted to deal with them. Rapid declines in flow, however, are abnormal and many attributes of ecosystems are unable to retreat in response to the falling water level, often resulting in mass mortalities (Power et al., 1996; McAllister et al., 2001; Anderson et al., 2015; Rolls and Bond, 2018). When this happens on a daily basis, this can

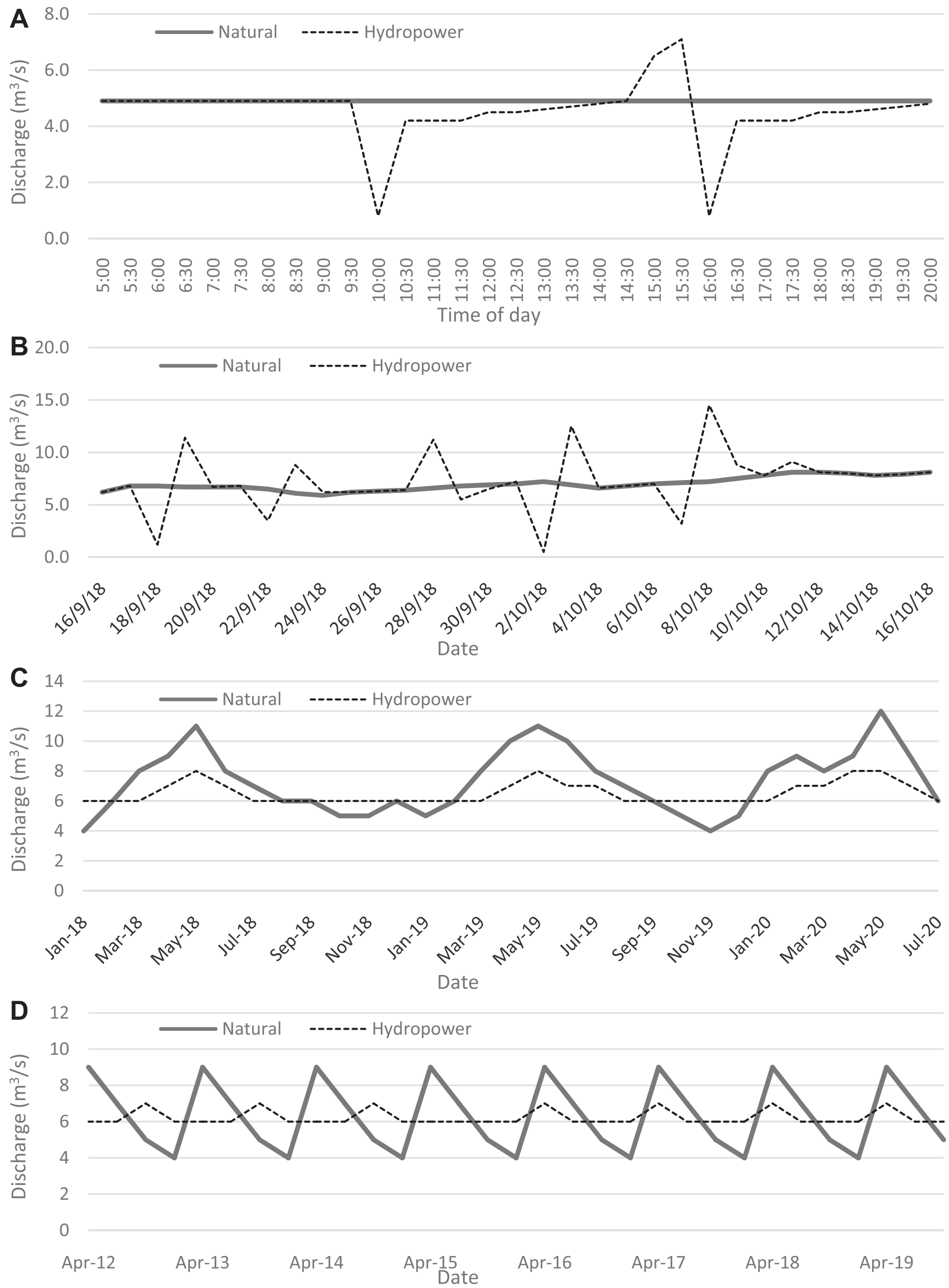

FIGURE 3 | Hourly **(A)**, daily **(B)**, monthly **(C)** and multi-annual **(D)** hydrograph of a river with natural and small-scale hydropower plant flows. Including demonstrations of transmission loss effects **(A)** and associated hydropeaking **(A,B)**, loss of flow variability **(C,D)** with example of reverse hydrograph [**(C)**, April 2012 to Apr 2015].

be devastating to the populations and overall biomass of in particular invertebrates but also fish. During the event observed in the Rwimi River on 2 October 2018, large Cyprinids that preferred deep habitats were forced into shallow pools during the approximately 22 min after the transmission failure event on the Rwimi Power Plant. These fish were harvested by local communities during this vulnerable period.

c. Water releases, either directly from the storage dam or from the tailrace, are generally driven by the need for power generation. Such releases are unlikely to be sympathetic to ecological needs and generally fluctuate rapidly, sometimes on a daily or even hourly basis but also having sustained effects at monthly and multi-annual time-scales (**Figure 3**). **Figure 3** demonstrates the flow scenario observed at the Rwimi Power plant during power transmission outages.

Outflow from below a hydropower plant including hypothetical natural (base) flows and actual hydropower releases to demonstrate the effects of short-term transmission cuts (**Figure 3A**) and associated hydropeaking (**Figures 3A,B**), loss of flow variability (**Figures 3C,D**). Both a constant baseload as well as a peaking power release will have negative impacts on the downstream ecosystem for reasons that will be discussed below. Similar fluctuations may occur where generation alternates on and off in order to provide for fluctuating demand for electricity.

2. The impact of altered flows on the instream and riparian ecosystem.
   a. Change in river morphology; repeated inundation then draining can cause slumping of soil on the river banks, and thus bank erosion (Mcallister et al., 2001; Anderson et al., 2015). Dams and weirs also intercept sediment flows, and can thus lead to scouring of the downstream river and armouring of the habitat. The water can also become less turbid, which will be negative for species evolved to live in turbid waters e.g., clear water exposes them to predators (Anderson et al., 2015).
   b. River connectivity; ecosystems are unique in that they change and develop as the river runs from source to sea, so a mountain stream is a very different ecosystem to a coastal plain river. Constructing a large dam or weir in the path of a river breaks the continuity that many ecosystem functions depend on (Anderson et al., 2015; Zarfl et al., 2015). Key amongst these is that various fauna, particularly fish but also invertebrates, need to migrate upriver in order to complete their life-cycle and also they contribute to ecosystem functioning in both the upstream and downstream portions of the river. Dams and weirs thus can have a substantial impact on ecosystem connectivity. While by-pass structures e.g., fish ladders, can ameliorate these impacts, their benefits are usually only partial (Lynch et al., 2019; O'Brien et al., 2019). Note that many rivers have natural breaks in connectivity e.g., waterfalls or very steep rapids, to which the river ecosystem has evolved. Larger waterfalls may provide a permanent barrier to upstream migration of some species although eels can generally climb these barriers when they are still small. Even the biggest waterfalls do not prevent downstream migration. Smaller waterfalls and rapids may become less of a barrier during flood flows as organisms may be able to scale these obstacles using the still water at the edge.
   c. Signals to biota; altered flows send out confusing signals to biota living downstream, confusing in particular their natural cues to migrate and breed (Lynch et al., 2019). Many species, including both fish and invertebrates, have very specific requirements for increased water flows (e.g., that arrive at the beginning of the wet season) as well as specific water quality (e.g., rising temperatures) to stimulate their need to breed. Regulated flows can thus be detrimental to populations by affecting their ability to complete life-cycles.
   d. Desiccation of the river substrate; following a sudden drop in flow can lead to loss of the periphyton (algae living on the rocks etc.), leading to a loss of primary production and thus a loss of food for the rest of the food chain including people. In addition, death of invertebrates and even fish is commonplace (Anderson et al., 2015).
3. Other impacts of small-scale hydropower plants on biota:
   a. Mechanical damage of turbines that injure and or kill fish and other biota is well documented and should be carefully considered when establishing small-scale hydropower developments, while new turbine designs factor in non-destruction of fish (Charles and Whitney, 2001; Schilt Carl, 2007). Loss of these fauna can affect both ecosystems and the livelihoods of subsistence fishermen.
   b. Disturbance to wildlife impacts are derived from hydropower developments that facilitate water resource development and urbanization of natural areas. The increase of people and their activities along rivers with the maintenance of the hydropower infrastructure results in a disturbance to wildlife where many mobile aquatic (such as fish) and riparian species (such as mammals and aquatic birds) avoid the development area. These impacts are similar to the effect of alien invasive species that compete with and or predate on indigenous animals (Kennard et al., 2005). Indigenous species tend to avoid areas of negative disturbances (Ellender and Weyl, 2014).
   c. Reduced resilience of biota to environmental variability and climate change; the synergist effects of barrier formation, habitat alterations and impacts of activities on the life-cycle ecology of species all affects the resilience of species to natural and anthropogenic changes in environmental variability including climate change. Many aquatic animals that lose resilience have reduced ability to survive droughts and or excessive changes between dry and wet phases of ecosystems (Arias et al., 2014).

Incorporating E-Flows Management for the Hydropower Sector in Uganda

Environmental flows *describe the quantity, timing, and quality of freshwater flows and levels necessary to sustain aquatic ecosystems which, in turn, support human cultures, economies, livelihoods, and well-being* (Arthington et al., 2018), a definition that emerges from the Brisbane Declaration (2007). The definition of e-flows spans the twin responsibilities of management, to balance *the use* and *the protection* of the water resource, i.e., it seeks to provide the flows required to maintain sustainable ecosystems and at the same time, the human use derived from the ecosystems to meet livelihoods (Arthington et al., 2018).

Environmental flows exist and can be determined for all riverine, wetland, estuary, lake and groundwater ecosystems, whether the ecosystem is in a natural or altered state (Arthington et al., 2018). Prior to the development of water resources, flow variability is influenced by natural climatic, hydrological and physical ecosystem processes that are considered to represent the "natural" or "historical" flow

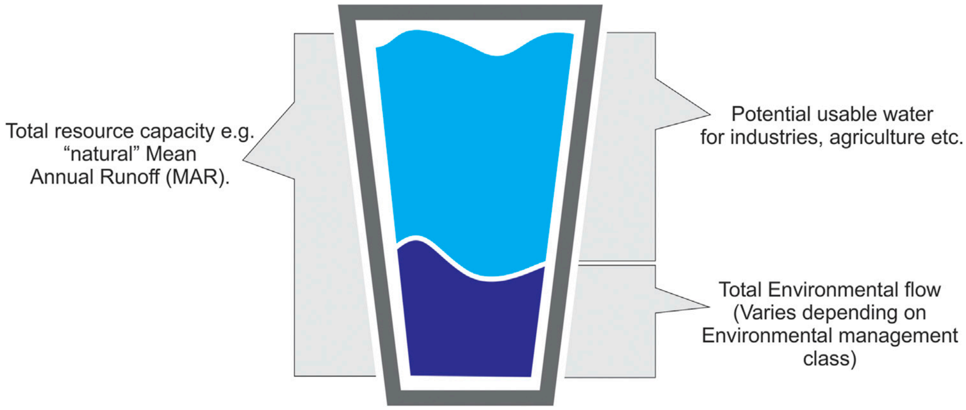

FIGURE 4 | A schematic view showing the total water resource, the e-flows (EF) and the utilizable/allocatable portion (adapted from Dickens, 2018).

regime (Poff et al., 2018). Adoption of e-flows and e-flows management is only important when the development, or use of resources and other anthropogenic activities such as climate change or water pollution, poses a risk as changes in river flow may become excessive and threaten the resource base on which development depends. Indeed, e-flows can be described for all water resources including those in a natural condition, but usually are only characterized and managed when a conflict between the use and protection of water resources threatens the sustainability of the resource (Arthington et al., 2018).

The diagram below (**Figure 4**) offers a simplistic view of the volumes of water required for e-flows (adapted from Dickens et al., 2018). Environmental flows form the foundation for water resource management and in many countries are guaranteed by law. All volumes of water in excess of the e-flows can be considered to be the utilisable or "allocatable" water that resource managers can allocate to hydropower, agriculture, industry or domestic water users. **Figure 4** does not however show an essential component of e-flows, i.e., the timing, frequency and duration of flows designed to represent the natural hydrograph of a river.

An example hydrograph (**Figure 5**) from the Niger River in Mali (Dickens et al., 2018) shows how the duration, timing and frequency of e-flows can match the shape of the natural hydrograph. During the dry months the e-flows in this case take up nearly 100% of the river flow, while in the wet season, only approximately half, this meaning that the allocatable water is mainly available during the wet season (i.e., for the Niger River). Just how much of the water is available for abstraction and allocation, and at what times of the year, is the subject of an e-flows determination, designed to ensure that the river ecosystem continues to provide services to society and at the same time protect the ecosystem resource base.

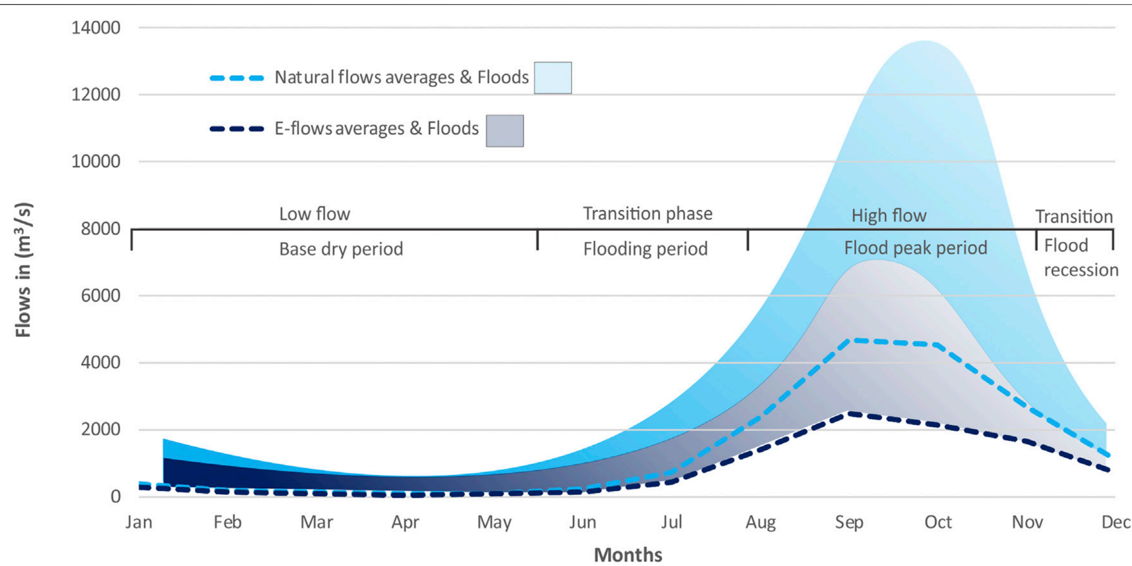

FIGURE 5 | Graphical representation of the monthly average and flood hydrograph, and the environmental flows of the Niger River in Mali. The dotted lines represent the monthly averages, and the shaded areas the flood peaks (reference Dickens et al., 2018).

Hydropower and e-Flows in Uganda

In Uganda, the National Water Policy (GoU, 1999) and the National Water Act (GoU, 1997) that direct the sustainable development of water resources, do not explicitly mention e-flows. However, the national water policy does state that water should be allocated to the environment and particularly water resources should be managed to provide a minimum flow to maintain water quality and aquatic ecosystems, but without providing guidance on how to achieve this or how much may be required (GoU, 1999:30). In an examination of the applicability of e-flows within Uganda (Okori, 2010), it was found that the basis upon which minimum flows of water permits have been developed did not incorporated minimum flow requirements for ecosystem health. One of the first government documents to recognize e-flows was the Environment Impact Assessment Guidelines for Water Resources Related Projects in Uganda (GoU, 2011:77). It advocated environmental awareness training and the evaluation of river flow requirements in relation to planned water projects. Following on from this, the Water Supply Design Manual stressed the residual ecological and Environmental flows of rivers has to be guaranteed in the context of water supply abstraction (GoU, 2013:54). This manual suggests that to determine the e-flows of a river, the model should consider hydraulics, hydrology, meteorological and biological parameters. Environmental flow requirements have also been included in Environmental and Social Impact Assessment (ESIA) for hydropower projects, recommending that weir design should be subjected to unconditional minimum flow requirements at approved flow rates to be determined by the Department of Water Resources Management (GoU, 2018:96). Furthermore, the recent Uganda Catchment Management Planning Guidelines (GoU, 2019) acknowledge the need for e-flows for environmental sustainability, stating that 'the amount of existing water use must be taken into account, as well as the amount of stream flow that is needed to maintain critical seasonal flows for water quality management, environmental and ecological requirements, and to protect water off-takes that depend on river water levels to function (GoU, 2019). However, it also highlights the lack of data and policies needed to establish e-flows requirements in Uganda (GoU, 2019).

In the absence of explicit e-flows guidelines for Uganda, the Nile Basin Initiative strategy on e-flows, of which Uganda is a member, provides suitable direction for e-flows implementation that conforms to best e-flows practice (Nile Basin Initiative, 2016; Nile Basin Initiative, 2017; O'Brien et al., 2018). This strategy provides context of e-flows management at local, regional and basin scales and describes how this can be achieved in consideration of other users of the Nile Basin. The strategy then describes the e-flows framework and guiding principles for managing e-flows in the Nile Basin.

Opportunities for e-flows management in Uganda were workshopped with the Electricity Regulatory Authority of Uganda, national regulators, specialists and stakeholders in Kasese in 2018. This workshop included a series of site visits and formal and informal discussions between stakeholders pertaining to water resources development, hydropower in Uganda and e-flows. Stakeholders represented at the workshop included; government regulators, conservationists, developers, development beneficiaries and impacted and affected parties. During this workshop stakeholders discussed challenges to the implementation of good e-flows policy, including implementation of the Nile e-flows strategy in Uganda, particularly amongst government regulators (Deutsche Gesellschaft für Internationale Zusammenarbeit (GIZ), 2018). The primary concerns included: there is an incomplete understanding of the meaning of e-flows and how to integrate these into strategies and day to day management; a lack of regulations or strategies allied to the policy for the management of e-flows; inappropriate methodologies and/or level of detail in setting e-flows requirements, as well as inefficient procedures, resulting in loss of staff time and supporting resources, loss of power generation and grid stability in critical supply situations, and environmental degradation and social safeguard issues; inadequate capacity for compliance monitoring of e-flows requirements also resulting in loss of staff time and supporting resources, and/or environmental degradation and social safeguard issues. These shortcomings formed the basis of this paper. Resolution is urgently needed to ensure proper management of the water resource. An additional concern is that hydropower developers and operators are also involved in the management of hydropower activities themselves, in that they have to comply with government regulations, including those for e-flows. The same officials are often involved in development and authorisations resulting in a conflict of interest. There appears to be a general lack of guidance from government agencies during project conception and design, resulting in inefficient planning, exacerbated by largely non-transparent government procedures resulting in inconsistent setting of e-flows requirements by concerned government agencies and thus an uneven field of competition between (private) developers. The lack of proper regulations has also resulted in the somewhat arbitrary setting of e-flows requirements by concerned government agencies, leading to projects sometimes becoming economically non-viable. Furthermore, developers have also complained that disproportionate e-flows requirements need to be sustained during operations, thus limiting generation and the resulting viability of the development. There was also a perceived lack of knowledge about the management of dams and power plants to provide for e-flows, thus this paper.

Riparian (and other) stakeholders are vulnerable to threats associated with unsuitable management of e-flows that results in loss of ecosystem services to stakeholders in downstream river basins in particular (Mander et al., 2017; O'Brien et al., 2018). This may be due to deterioration of aquatic and related ecosystems in their immediate vicinity, resulting in lost opportunities and natural capital (Mander et al., 2017).

Given the above situation, there is need for an improvement of the regulation of e-flows by government agencies, together with a need for developers to factor in more confident estimates of e-flows together with an understanding of how hydropower affects the ecosystem and the people who rely on this ecosystem. Consideration thus needs to be given to the existing legal framework and how this may be developed into

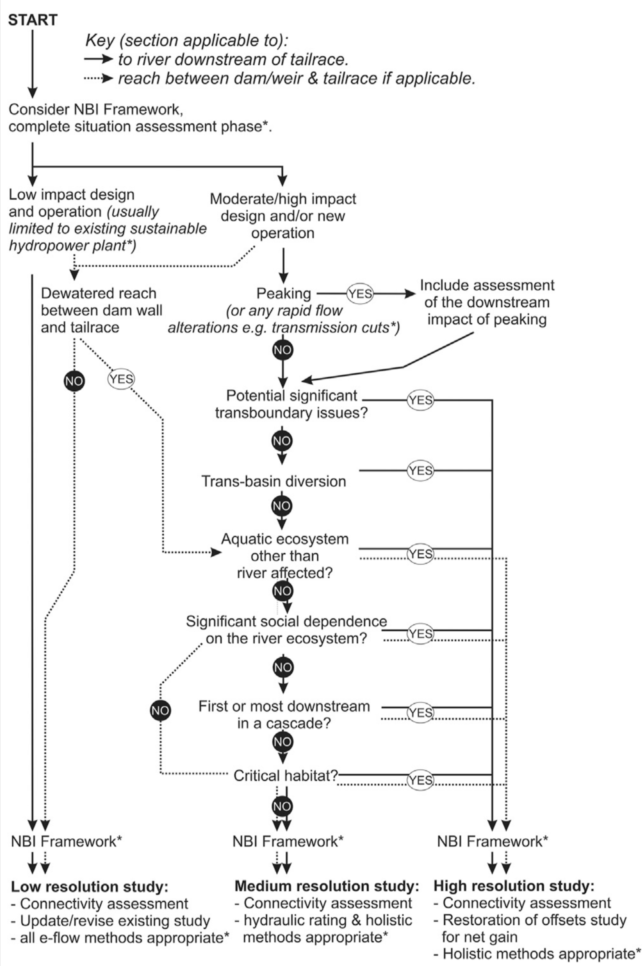

FIGURE 6 | Decision support system (tree) for environmental flow assessments for small scale (<50 MW capacity) Hydropower development in Uganda. Solid line and dotted line represents pathway for e-flow consideration downstream of tailrace and hydropower plant and the dewatered zone or reach between diversion weir and tailrace respectively. If both attributes of a hydropower plant development are appropriate then both lines should be considered in a precautionary manner where preference for high resolution, holistic methods are afforded. Adapted from World Bank (2018) and Nile Basin Initiative (2016).

future policy, strategies and regulations; system scale water resources planning with basin level e-flows determination, supporting the mid-to longer-term pipeline of larger scale hydropower projects including those on the Nile, but also clusters of small schemes in mountainous areas; improving e-flows releases of existing operations, e.g., the ecological optimization of releases and compliance monitoring; and lastly, specific issues, such as sites in national parks, mitigation of impacts of hydropower peaking, design and operation of fish ladders all need consideration.

Regional Integration

Being located within the larger Nile River Basin, the most relevant NBI document for the management of e-flows in Uganda is the "Strategy for the Management of E-flows in the Nile Basin" (Nile Basin Initiative, 2016). The strategy was prepared by the Nile Technical Advisory Committee (NILE-TAC) and Nile Basin Environmental flows Expert Group through the course of the "Preparation of NBI Guidance Document on Environmental flow" (Nile Basin Initiative, 2016). The goal of the strategy is to: "facilitate and develop a culture of incorporation of collaborative, best practice e-flows management into the water resource planning, management and policies of the countries who share the Nile Basin (short term) to ultimately result in the establishment of an integrated, basin scale e-flows management system (long term)". Finally, the strategy advocates the allocation of water resources in a manner that does not jeopardize the functioning of the resource. The strategy also supports international conventions and agreements that consider the sustainable management of water resources and specifically Environmental flows. These include: 1) consideration of the Sustainable Development Goals (SDGs), 2) requirements of the Convention on Biological Diversity, of which Uganda is a member, 3) the Aichi Biodiversity Targets for 2020 and 4) the RAMSAR guidelines for the allocation and management, including e-flows, of water resources in a sustainable manner (Nile Basin Initiative, 2017). These regional policies align to advocate "good" e-flows practices that are summarized in the NBI e-flows strategy and include involvement of stakeholders in a governance system aimed at subsidiarity, keeping e-flows assessments as simple as is necessary, applying adaptive management principles and so to continuously learn from application, sharing experiences and possibly expertize across the basin, and lastly to be willing to manage e-flows at multiple scales (from local to basin).

Recommendations for E-Flows Methods and Approaches

Good practice e-flows management in Uganda has the potential to make a noticeable contribution to the sustainable development of the water resources of Uganda, this includes small-scale hydroelectric power generation. The point of departure for good e-flows management practice is to ensure that management efforts meet the definition of e-flows, defined above (Arthington et al., 2018). Then it is important to identify roles and responsibilities of stakeholders and types of activities that may affect water resources that triggers the need for e-flows management activities (World Bank, 2018). The NBI e-flow strategy (Nile Basin Initiative, 2017) and Nile Basin e-flow framework (Nile Basin Initiative, 2016) and guiding elements for best practice in e-flows (Poff et al., 2018) all provide good-practice direction on the roles and responsibilities of different stakeholders who are responsible for e-flows assessments. Formal national custodians of water resources should be responsible for large-regional and or basin scale e-flows management, which includes sustainable development of water resources and meeting the needs of local communities and the people who depend on these resources for survival. In Uganda, government representatives of the Ministry of Water and Environment and Electricity Regulatory Authority in particular are primarily concerned with hydropower development and water resource management to manage regional and basin scale e-flows in Uganda, and contribute to Nile Basin management. These regulatory stakeholders issue authorization for local and reach scale developments in consideration of the contribution of activities to larger regional management endeavors. Developers are required to obtain authorization for developments that must address local, reach and on occasion regional scale effects of activities/developments. Ultimately the decisions on how to manage the balance between the need to use and protect water resources is socio-political (Dickens et al., 2018), with society deciding what constitutes an acceptable risk to the ecosystem, above the sustainability threshold, in terms of the benefits that are gained from the ecosystem. The more the ecosystem is "used", the greater the risk becomes that it may fail to provide further resources, in which case both the ecosystem and society will have suffered loss (Dickens et al., 2018). By putting e-flows in place, and only using the "allocatable" amounts that do not impinge on the e-flows, society will be ensuring its own future as the ecosystem will continue to produce benefits for society.

The World Bank (2018) has developed a Good Practice Handbook for E-flow (WB-GPH) for Hydropower Projects, especially for the guidance of hydropower activities in the private sector in emerging markets or developing nations. This WB-GPH provides information on the potential effects of hydropower on water resources, e-flows assessments, methods and tools and provides a decision support tree for selecting e-flows methods for individual projects, e-flows and adaptive management and terms of references for e-flows assessments (World Bank, 2018) (**Figure 6**). Consider also that once operational the Nile e-flow framework may provide low confident e-flows requirements for all major rivers and tributaries in the Nile Basin (Nile Basin Initiative, 2016). With this information, and an understanding of the development and operational requirements for new hydropower plants more robust, more confident e-flows can be determined. The World Bank (2018) decision tree for e-flows assessment recommends low, medium or high-resolution e-flows determination for all hydropower activities depending on the potential attributes of proposed developments including: constriction of barriers; existence of a dewatering reach; plan for peaking; vulnerable ecosystems and ecosystem attributes including critical habitats; social dependence on existing resource and transboundary and regional effects. When undertaking e-flows assessments the following good-practice guiding elements obtained from Poff et al. (2018) and principles for e-flows management obtained from Nile Basin Initiative (2017) should be considered:

1) Engage stakeholders in the entire e-flows determination process, particularly in the visions and objectives determination process.
2) Ensure benefits of water resource allocation and or developments are shared between local and regional stakeholders.
3) Environmental flows attempt to achieve a sustainable balance between the protection of water resource and the needs of society to use them. This is a trade-off that needs to be made by society, in the context of regional use and protection scenarios/opportunities, and needs to be informed by evidence that describes the ecosystem. Consider also the downstream vs. upstream effects of flow and non-flow stressors.
4) In e-flows assessments carefully identify what can be attained (and what cannot) from an implementation of e-flows regimes. Apply requisite simplicity concepts to processes and only make the assessments as complicated as necessary.
5) Consider how environmental water goals and applications embed within and interact with other realms of influence that emerge with water governance and management at system scales.
6) Clearly identify at what spatial and temporal scale e-flows applications are appropriate and intended.
7) Environmental flows assessments should be evidence based and flow-ecology and flow-social relationships should be described in a clear and quantitative manner.
8) Use appropriate e-flows determination methods that are transparent and robust. Ensure that uncertainty associated with the methods are explicitly presented.
9) Incorporate nonstationary and process-based understanding into e-flows science and implementation to meet a new future.
10) Make efforts to engage with the proponents and engineers of new water infrastructure developments or proposed relicensing opportunities for existing infrastructure.
11) Embrace adaptability principles of learning while doing and attempt to introduce adaptive management into e-flows practices where new information is integrated into the management processes and outcomes are flexible and can be adjusted as they are implemented and monitored.

From the emergence of e-flows determination procedures in the early 1990s many methods have been established and reviewed (Tharme, 2003; Petts, 2009; Poff et al., 2018; Brown et al., 2020 for example). Available methods can be grouped into four main categories including: 1) hydrological, 2) hydraulic rating, 3) habitat simulation (or rating), and 4) holistic, with some recent developments of holistic methods into frameworks for e-flows assessments of large regional scales (Poff et al., 2018). Methods differ in complexity, uncertainty, cost and time resources to determine e-flows. Consider the Appendix for detailed comparisons between methods and some advantages and disadvantages associated with the use of available e-flows methods. For planning purposes, the hydrological, hydraulic rating and habitat simulation methods are commonly applied. For developments habitat simulation and holistic methods dominate (Tharme, 2003; Poff et al., 2018). Consideration of regional implications and the Nile e-flow framework should then be considered (Nile Basin Initiative, 2016). The methods tend to be applied hierarchically (Tharme, 1996; Poff et al., 2018), often starting from hydrology-based approaches which are more appropriate in a precautionary, low-resolution framing of environmental water requirements at a water resources planning level, to increasingly comprehensive assessments using holistic methods where the importance of certainty in the results is much greater. Although e-flows determination methods are dominated by riverine ecosystem methods, some methods allow for the consideration of estuaries, wetlands, lakes ecosystems and ground water ecosystems for example (King and Louw, 1998; Hughes and Louw, 2010; King, Brown and 2010; O'Brien et al., 2018).

When e-flows assessments are undertaken in Uganda good practice requires consideration of the requirements of the Nile e-flow framework for regional scale application of e-flows (Nile

Basin Initiative, 2016). This will allow evidence collected from site and reach scale e-flows assessments to contribute to regional scale assessments and inferences to other sites with similar socio-ecological characteristics (Nile Basin Initiative, 2016). The Nile e-flow framework consists of seven phases summarized briefly in the context of e-flows for small hydropower developments:

1) Situation Assessment and Alignment Process phase: in this phase review all information pertaining to water resource management and e-flows associated with the proposed development. Use of the Nile e-flow framework checklist is recommended (Nile Basin Initiative, 2016).
2) Resource Quality Objectives Setting phase: targets and or a vision for a sustainable balance between the use and protection of water resources is required. These examples should be considered in site scale e-flows assessment as well as documented to contribute to a understanding of objectives for larger scales (Nile Basin Initiative, 2016).
3) Hydrological Foundation phase: in this phase of the framework hydrological statistics and associated understanding of the volume, duration, frequency and timing of flows is determined. This approach is well defined in the framework and should be considered to direct site scale assessments that can contribute to regional scale assessments (Nile Basin Initiative, 2016).
4) Ecosystem Type Classification phase: the framework and its ability to extrapolate flow-ecosystem and flow-ecosystem service relationships and information pertaining to the effects of flow and non-flow stressors on ecosystems is dependent on knowledge of the ecosystem characteristics. Collecting this data for site scale e-flows assessments is paramount for the application of the framework and good-practice for site scale assessments (Nile Basin Initiative, 2016).
5) Flow Alterations phase: knowledge of how flows will change due to hydropower developments is a fundamental requirement of good-practice e-flows assessments. This information and how accurate e-flows assessments were established to mitigate the effects of altered flows is important information for the Nile e-flow framework (Nile Basin Initiative, 2016).
6) Flow-Ecological-Ecosystem Services Linkages phase: all good-practice e-flows assessments must be based on understanding of flow-ecosystem and flow-ecosystem service relationships. Usually site scale e-flows assessment have opportunities to collect quantitative evidence that supports local e-flows assessments and will contribute to the application of the framework (Nile Basin Initiative, 2016).
7) Environmental flows Setting and Monitoring phase: in this phase of the framework, appropriate holistic e-flows assessments are implemented that benefit from data available in the catchment. This phase also includes an adaptive management component all of which can benefit from site scale applications.

Figure 6 provides a synthesis of the World Bank (2018) decision tree for e-flows assessments for hydropower and where the Nile e-flows framework (Nile Basin Initiative, 2016) in the context of small hydropower developments in Uganda.

CONCLUSION

Stakeholders of the small-scale hydropower sector in Uganda recognize the need to balance resource development that contributes to the livelihoods of vulnerable African communities, and sustainable ecosystems from which vulnerable human communities derive services. Environmental flows principles and practices are available to contribute to sustainable hydropower development and protect socio-ecological systems for present and future generations. Uganda is currently simultaneously in the process of developing hydropower plants and e-flows policies with limited guidance on e-flows management. Environmental flows management concepts have developed from the 1990s into an international good practice that contributes to sustainable water resource developments. We have provided a synthesis of existing good e-flows practices for consideration by the small hydropower development sector in Uganda, including methods and their appropriate use and consideration at multiple spatial scales and for regional policies and frameworks.

The determination and management of e-flows in the hydropower sector in Uganda is largely dependent on the availability of and quality of hydrology, hydraulic and flow-ecosystem and flow-ecosystem service relationship information. Unfortunately major constraints to regional e-flows program developments that may have considerable negative socio-ecological and economic benefits includes data ownership and secrecy, poor data capturing resulting in loss of information, and the lack of transparency of evidence collected. This review of good-practice e-flows practices that is applicable to the small hydropower sector in Uganda, and considers regional developments, can support the sustainable development of water resources in Uganda for a better future for all of its vulnerable communities and the environments they depend on.

AUTHOR CONTRIBUTIONS

GO'B conceived the idea of the research in collaboration with the team, collected and evaluated information and wrote the manuscript. CD also contributed to the idea of the research in collaboration with the team, formatted the content of the manuscript, collected and evaluated information and co-wrote, edited the manuscript. CM also contributed to the idea of the research in collaboration with the team, formatted the content of the manuscript, collected and evaluated information and co-wrote, edited the manuscript. ME also contributed to the idea of the research in collaboration with the team, and co-wrote, edited the manuscript.

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