

Effects of a large run-of-river dam closure on the functional diversity of subtropical freshwater fish assemblages

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Abstract:

Run-of-river hydropower plants are increasingly promoted as a low-impact alternative for energy generation because of their reduced water storage capacity. However, their effects on fish functional structure remain poorly understood. We evaluated changes in fish functional diversity following dam closure at the Foz do Chapecó run-of-river hydropower plant. We quantified 23 functional traits and calculated four indices: functional richness (FRic), functional evenness (FEve), functional divergence (FDiv), and functional dispersion (FDis). These indices were weighted by fish abundance and biomass using catch per unit effort (CPUE) data and compared between pre- and post-damming periods using bias-corrected and accelerated bootstrap confidence intervals (BCa). Functional diversity declined across most river stretches following dam closure. At impoundment sites, FRic and FDis decreased by 69% and 38%, respectively, while downstream tributaries exhibited a 39% reduction in FRic. Low-order upstream

tributaries showed reductions greater than 20% in FEve and FDis, indicating loss of functional complementarity and simplification of ecological strategies. In contrast, upstream tributaries displayed increases in FRic (32%) and FDis (19%), suggesting functional reorganization after reservoir formation. Community-weighted means revealed declines in traits associated with detritivory and parental care, likely reflecting reductions in benthic habitats and reproductive environments following impoundment, whereas traits related to piscivory, herbivory, and migratory behavior increased in upstream sections of the reservoir. Our findings indicate that the closure and operation of a large run-of-river hydropower plant was associated with substantial changes in fish functional structure, likely driven by habitat and ecohydrological alterations. Responses differed among reservoir, tributary, and downstream environments, revealing contrasting functional trajectories.

Keywords: Functional ecology, hydropower dams, impoundment, fish communities, renewable energy.

1. Introduction

The impacts of dams are primarily related to modifications of the water flow and the creation of artificial impoundments (Agostinho et al., 2008). Reservoirs and their associated backwater effects eliminate or reduce stretches of rapids and alter water level fluctuations both upstream and downstream of the impoundment (Poff et al., 1997). These hydrological changes can also impact thermal regimes and sediment transport (Olden and Naiman, 2010) and may facilitate the spread of exotic species by altering habitats and connecting historically isolated aquatic ecosystems (Júlio Júnior et al., 2009). These changes in habitat characteristics and hydrological connectivity have been reported to alter the diversity and structure of biological communities, promoting the spread of

certain species while threatening native freshwater biodiversity (Reid et al., 2019; Song et al., 2018; Turgeon et al., 2019).

Among the various taxonomic groups affected by the construction and operation of hydropower dams, fish assemblages are of particular concern due to their high diversity and socio-economic importance (McIntyre et al., 2016; Sabo et al., 2017). Dams can block migratory routes that are critical for the life cycle of several fish species, as well as restrict dispersal and gene flow (Barbarossa et al., 2020; Carvajal-Quintero et al., 2019). Furthermore, dams alter hydrological and limnological characteristics, leading to functional changes in fish assemblages (Agostinho et al., 2016; Arantes et al., 2019). For instance, the accumulation of organic matter following reservoir filling tends to benefit omnivorous and piscivorous species over more specialized trophic groups (Dias et al., 2020; Ferrareze et al., 2015). Similarly, rheophilic species (specialized in fast-moving water environments) are particularly vulnerable to damming, often being replaced by species with more generalist habits and reproductive strategies (Arantes et al., 2019; Liermann et al., 2012).

Most studies on the impacts of dams on fish assemblages to date have focused on hydropower plants that operate with large reservoirs (Fráguas and Pompeu, 2021; Philippsen et al., 2019). However, run-of-river hydropower plants are becoming increasingly common worldwide (Kuriqi et al., 2021). These plants are characterized by having little to no reservoir storage, which, in theory, would result in milder hydrological impacts (Csiki and Rhoads, 2014; Punys et al., 2015). Nevertheless, there is growing evidence that even run-of-river hydropower dams have noticeable effects on hydrology (Almeida et al., 2020; Csiki and Rhoads, 2014; Kuriqi et al., 2021). For example, run-of-river hydropower plants often produce frequent hydrological changes, known as "hydropеaking" or "repiquets", resulting from short-term (daily and weekly) fluctuations

in river flow to meet peak electricity demands (Almeida et al., 2020; Ashraf et al., 2018). These fluctuations alter the hydrological and geomorphological structure of downstream river stretches, affecting habitat characteristics and fish communities (Simonović et al., 2021). Fish assemblages in these downstream regulated stretches can also be impacted by reduced nutrient input, decreasing resource availability, and increased water transparency, which may ultimately benefit visual predators and alter food webs (Roberto et al., 2009; Zaniboni-Filho et al., 2008).

Studies examining the impact of run-of-river plants on fish assemblages are still limited in the scientific literature, and those available typically focus on assessments of taxonomic composition (e.g., Gibeau et al., 2017; Simonović et al., 2021). However, the functional structure and diversity of fish assemblages have been suggested as suitable indicators for detecting changes in freshwater communities and ecosystems (Hitt and Chambers, 2014; Lin et al., 2021). This is because metrics of functional structure and diversity represent not only species diversity but also their relationships and interactions with abiotic and biotic factors (Díaz and Cabido, 2001). By selecting relevant traits that describe organism functionality, it is possible to infer their role in the ecosystem, independent of taxonomic group or evolutionary relationships (Maire et al., 2015; Violle et al., 2007). Typically, these attributes include traits related to food acquisition, locomotion, nutrient processing, reproduction, and defense against predation (Winemiller et al., 2015).

Given the need for a better understanding of the effects of large run-of-river dams on freshwater ecosystems through the lens of functional ecology, we evaluated the impact of the first five years of operation of the Foz do Chapecó Hydropower Power Plant (HPP) on the functional structure and diversity of fish assemblages. Because different functional diversity indices describe complementary aspects of community organization, we

expected that dam-induced changes in habitat characteristics and hydrological connectivity would modify multiple dimensions of functional structure. Specifically, i) we predicted reductions in functional richness and dispersion, reflecting losses in the breadth and distribution of ecological strategies; ii) we expected stronger responses in mainstem sites directly affected by dam operation than in tributaries; and iii) we anticipated shifts in the relative importance of functional traits, with opportunistic feeding strategies becoming more prevalent and traits associated with specialized habitats and migratory behavior becoming less represented following dam closure.

2. Material and methods

2.1. Study area

The Uruguay River Basin covers an area of approximately 365,000 km², with 176,000 km² located in Brazilian territory (Zaniboni-Filho et al., 2004). The Foz do Chapecó Hydropower Plant (HPP) began operations in October 2010 on the mainstem Uruguay River in the upper portion of the basin (Figure 1; (Eletrobras Furnas, 2022). Classified as a large run-of-river plant, it has an installed generation capacity of 855 MW provided by four Francis turbines and was designed primarily for electricity generation within the Brazilian Interconnected System (Foz do Chapecó Energia S.A., 2002). The dam structure measures 598 m in length and 48 m in height, and the drainage area at the dam site is approximately 53,000 km² (Foz do Chapecó Energia S.A., 2022). Construction started in March 2007, and the reservoir was formed in September 2010, covering an area of 79.2 km², of which 40.0 km² overlap with the original channel of the Uruguay River (Foz do Chapecó Energia S.A., 2022). The reservoir has a normal maximum operating level of 265 m, a minimum operating level of 264 m, and a useful storage volume of 74 × 10⁶ m³ (Eletrobras Furnas, 2022). The plant operates as a run-of-river diversion scheme

with a dewatering section of 19 km in length and a maximum spillway flow of 62,190 m³/s (Foz do Chapecó Energia S.A., 2022).

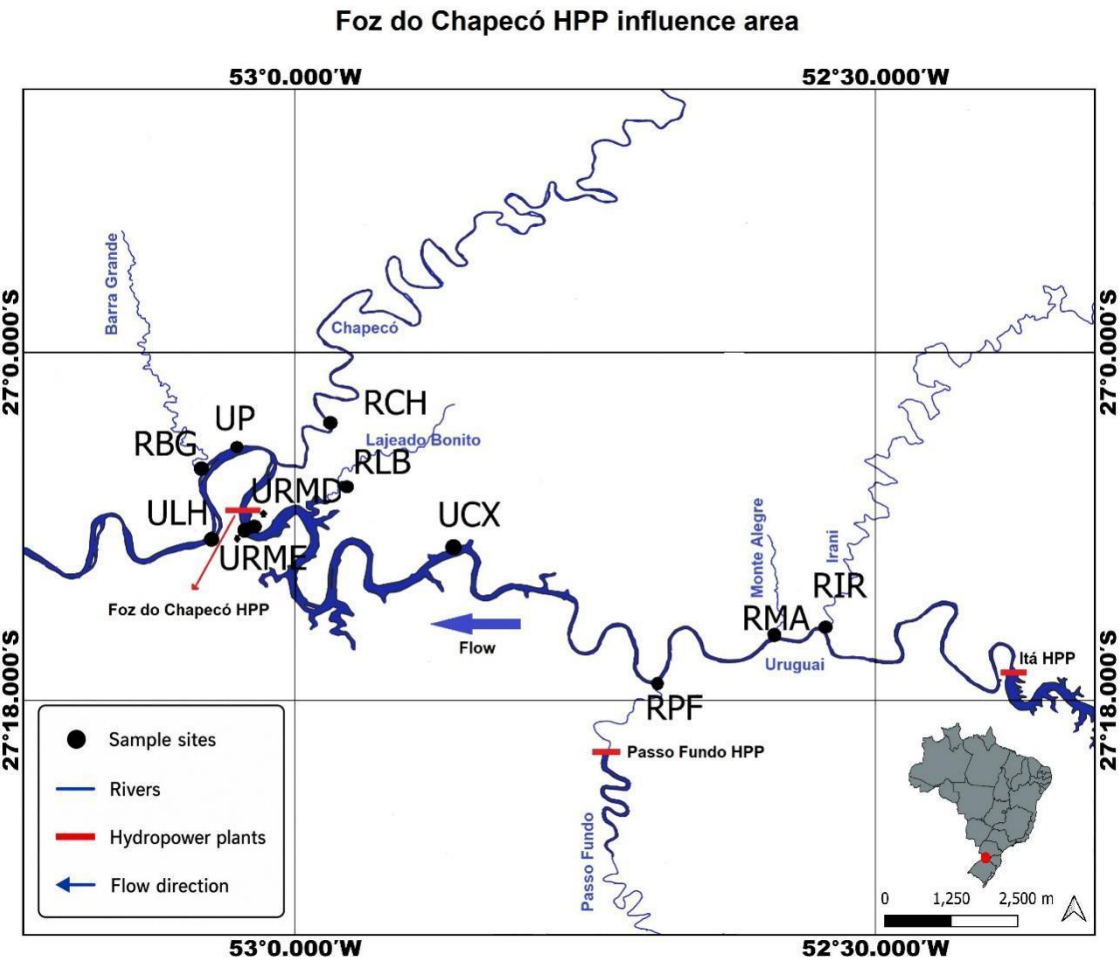


Figure 1. Sampling sites for fish surveys in the area of influence of the Foz do Chapecó hydropower plant, Uruguay River Basin, Brazil. Black dots depict the location of each sampling site: Irani tributary (RIR), Monte Alegre tributary (RMA), Passo Fundo tributary (RPF), Uruguay river mainstem near Caxambu do Sul (UCX); Lajeado Bonito tributary (RLB), Uruguay river mainstem on the left bank of the impoundment (URME), Uruguay river mainstem on the right bank of the impoundment (URMD), Chapecó tributary (RCH), Uruguay river mainstem near Balneário de Pratas (UP), Barra Grande tributary (RBG), Uruguay river mainstem near Balneário de Ilha Redonda (ULH). The location of the Foz do Chapecó hydropower plant and other large dams are represented by red bars.

The Uruguay Basin is highly fragmented by other hydropower plants, including five major dams operating in series on the upper Uruguay mainstem (Couto et al., 2021; Schork and Zaniboni-Filho, 2018). These five large dams form a cascade (Foz do Chapecó 885 MW, Itá 1315 MW, Machadinho 1060 MW, Barra Grande 690 MW, and Campos Novos 880 MW), with the Foz do Chapecó HPP being the furthest downstream among them (Schork and Zaniboni-Filho, 2018).

2.2. Hydrological characterization

To characterize hydrological changes associated with dam closure and operation, river discharge data were obtained for the period from January 1995 to November 2014, during which information was available for all sampling stations. Discharge data were obtained from fluviometric stations available through the HidroWeb v3.2.7 database (Snirh, 2023). Four fluviometric stations were selected to represent the hydrological conditions of the study area (Table S1).

The stations corresponding to the upstream site (UCX) and dam sites (URME and URMD) were located close to the fish sampling locations. For URME and URMD, discharge values were obtained from the Foz do Chapecó station (code 74040080). To estimate discharge at the downstream site (UP), discharge values from the Foz do Chapecó station (code 74040080) and Barra do Chapecó station (code 73960000) were summed. This procedure allowed the combination of upstream discharge information to estimate flow conditions at the downstream sampling site.

For ULH, discharge data were only available after 2013 and therefore historical records from this station were unavailable during the study period. However, this site has a drainage area of approximately 61,600 km², whereas the combined drainage area represented by the Foz do Chapecó and Barra do Chapecó stations is approximately 61,300 km². Therefore, discharge at ULH was estimated through proportional adjustment

based on drainage area. For the post-dam period, turbine discharge from the Foz do Chapecó powerhouse was incorporated into the flow estimates. Detailed information on the fluviometric stations and historical discharge series is publicly available through the HidroWeb platform (Snirh, 2023).

2.3. Sampling.

From June 2007 to September 2012, fish assemblages were sampled quarterly in the area of influence of the Foz do Chapecó HPP (Figure 1). Samples were taken at 11 sites, of which 7 were located upstream of the impoundment (three on the mainstem Uruguay River, two on tributaries, and two on low-order streams), and four sites were located downstream of the impoundment (two on the mainstem Uruguay River and two on tributaries; Figure 1; Table S2).

We employed standardized sets of gillnets (mesh sizes ranging from 1.5 to 10 cm between opposite knots) and trammel gillnets (mesh sizes 4 and 20 cm between opposite knots) to sample fish. The gillnets were installed at dusk (~5 pm) and removed at dawn (~6 am) after a 12-hour exposure. In large rivers, the gillnets were installed along the banks and placed parallel to the shore, while in streams, they were installed in deeper locations to remain fully submerged (Table S3).

All captured fish were measured and identified in the laboratory. Whenever possible, the stomachs of the specimens were removed for analysis of their contents. Fish and stomachs were initially fixed in 10% formalin and then transferred to 70% ethanol. The stomach contents were analyzed to the lowest possible taxonomic level with the assistance of a stereomicroscope. Voucher specimens are deposited in the Ecology Laboratory of Unochapecó and in the ichthyological collection of NUPELIA (State University of Maringá).

2.4. Biological Traits

We compiled a list of 25 functional traits for each fish species, which included 15 morphological, seven trophic, and three behavioral/ecological traits (Table S4-S6). These traits are commonly used in functional diversity analyses of fish assemblages and reflect functional strategies related to survival, dispersal, and habitat use by fish species (Toussaint et al., 2016; Villéger et al., 2012). The morphological traits were derived from morphoanatomical measurements of the captured fish, based on a minimum of five individuals per species whenever possible (Table S4). Three photos of each individual were taken (side, top, and frontal views), and the morphoanatomical measurements were calculated using ImageJ software (Figure S1). Species biomass was estimated by averaging the weight (measured after capture) of all individuals.

Trophic traits were obtained by analyzing the stomach and intestinal contents of 1,473 fish captured across all sampling sites. The stomach and intestinal contents of 53 species were analyzed. The contribution of each food item to the species' diet was determined using the volumetric method (Hyslop, 1980), with volume estimates obtained using gridded Petri dishes. For all assessed individuals, the volume of each food item and the total volume of all food items were quantified. The final trophic traits for each species were calculated as the average volumetric proportion of each food item in its diet. These proportions were combined into seven feeding categories: detritivory, herbivory, insectivory, carcinophagy, molluscivory, invertivory (which includes all other invertebrates not categorized), and piscivory.

The remaining three behavioral/ecological traits were obtained from the literature and online resources like FishBase (Froese and Pauly, 2022); Table S6): habitat use, migratory behavior, and parental care. The habitat use is a categorical trait classifying

fish as pelagic, benthopelagic, or benthic. Both migratory behavior and parental care were represented as binary traits (i.e., present or absent). When specific information for a species was unavailable, data from congeneric species were used (Table S6).

2.5. Data Analysis

The data were standardized based on both the number of individuals captured and the biomass per unit of effort (CPUE; King, 2013). CPUE was calculated separately for abundance and biomass and standardized by net area and exposure time (12 h). Abundance CPUE was calculated as number of individuals/(net area $\times$ 12 h), whereas biomass CPUE was calculated as biomass (kg)/(net area $\times$ 12 h). Initially, 25 functional traits were considered. To avoid overrepresentation of correlated functional dimensions, trait redundancy was evaluated using the variance inflation factor (VIF). In this context, VIF was employed as a trait selection procedure rather than as a diagnostic for predictor multicollinearity in statistical models. Traits exhibiting high collinearity ($VIF \geq 10$) were excluded, resulting in a final set of 23 traits used to construct the functional space and calculate community-weighted means (CWMs) (Marcoulides and Raykov, 2019; Raykov, 2018). After excluding highly collinear traits (body mass and eye position), we estimated a set of indices summarizing different aspects of the functional diversity of fish assemblages: functional richness (FRic; Villéger et al., 2008), functional evenness (FEve; Villéger et al., 2008), functional divergence (FDiv; Villéger et al., 2008), and functional dispersion (FDis; Laliberté and Legendre, 2010). Because these metrics describe different components of functional diversity, they were analyzed separately rather than combined into a single metric. Definitions, ecological interpretations, and references for each index are provided in Table S7.

The matrix of functional traits contained both quantitative and categorical variables, so distances were estimated using Gower dissimilarities with the SYNCSA package, along with the Cailliez correction for the eigenvalues (Debastiani and Pillar, 2012; Legendre and Anderson, 1999). The functional space was represented by Principal Coordinates Analysis (PCoA) based on Gower dissimilarities. To determine the number of dimensions retained for the representation of species in the functional space, a broken-stick randomization model was applied to the ordination axes (Jackson, 1993). Based on this criterion, the first three PCoA axes were retained for subsequent analyses and graphical representation of the functional space. All functional diversity indices were calculated using the 'dbFD' function available in the FD package in R (Laliberté and Legendre, 2010). The 'dbFD' function was also used to obtain the community-level weighted means (CWM) of each functional traits (Lavorel et al., 2008). To ensure reliable estimates, only samples with at least three recorded species were included in the calculations of the indices and CWMs.

Because the post-damming period comprised only two years of observations, our primary objective was to quantify the magnitude and direction of biological changes relative to baseline pre-damming conditions rather than fit more complex temporal models. Therefore, we adopted an effect-size approach based on response ratios, which is commonly employed in meta-analyses and before-after studies to assess ecological responses while minimizing the influence of site-specific variation. This approach is suitable given the limited temporal replication after dam closure and the high spatial heterogeneity among sampling sites. In addition, baseline values were calculated separately for each season and sampling site, thereby accounting for seasonal variation in the comparisons between pre- and post-damming conditions.

The effect size of dam closure on functional indices and taxonomic richness was estimated for groups of sites according to their location in the river network (Uruguay mainstem or tributaries) and their position relative to the dam (upstream, impoundment, or downstream). Effect sizes were evaluated using the response ratio, which quantified the magnitude of change between pre- and post-damming conditions (Cohen, 1988). Samples from the pre-damming period (June 2007–June 2010) were used as baseline conditions, and an average of each index was calculated for each sampling site and season, which were used as replicates. For example, the average of all RBG summer samples from the pre-damming period served as the baseline for comparisons with post-damming summer samples at the same site.

A ratio was calculated for each sample in the post-damming period (2010–2012) by comparing it to the respective baseline condition of the same site or group of sites. All ratios from the post-damming period were transformed using the natural logarithm before analysis. To test whether the effect size (positive or negative) of each post-damming sample was significantly different from the pre-damming sample, the BCa method was used with non-parametric bootstrapped statistical analysis (1000 replicates) and 95% confidence intervals (Davison and Hinkley, 1997). This analysis was conducted using the 'Boot' function of the 'Boot' R package (Canty and Ripley, 2016). In contrast, changes in community-weighted means (CWMs) were evaluated using generalized linear models (GLMs), because the objective was to test for differences in individual trait values between pre- and post-damming periods rather than quantify overall effect sizes. A separate GLM was fitted for each functional trait, with pre- and post-damming periods as explanatory variables. Because multiple traits were evaluated independently within each site category and metric (abundance and biomass), p-values were adjusted using the Benjamini-Hochberg false discovery rate (FDR) procedure to control for Type I error

associated with multiple testing. All data processing steps and statistical analyses were performed using R version 4.4.1 (R Core Team, 2024).

Generalized Additive Models (GAMs) were used to evaluate temporal patterns in river discharge before and after dam closure. The purpose of these analyses was to verify whether dam closure was accompanied by measurable changes in the hydrological regime that could plausibly explain the observed responses in fish functional diversity. Discharge was analyzed for the period between January 1995 and September 2012, corresponding to the temporal extent of fish sampling. An additional analysis covering January 1995 to December 2014 was conducted to evaluate whether hydrological patterns persisted beyond the biological sampling period. GAMs were fitted separately for pre-dam and post-dam periods to identify temporal and seasonal changes in river discharge. These models allow flexible estimation of non-linear relationships between predictors and response variables (Beck and Jackman, 1998). Model complexity was assessed using effective degrees of freedom (eDF), where values close to one indicate approximately linear patterns and higher values indicate increasingly complex non-linear temporal dynamics. Following the approach proposed by (Wood, 2004), smoothing functions were used to describe temporal variation in discharge and to evaluate changes in flow seasonality associated with reservoir filling and hydropower operation.

3. Results

3.1. Composition of fish assemblages

A total of 3,900 individuals from 55 species were recorded in our study, with 52 species documented in the pre-damming period and 48 species in the post-damming period. The orders Characiformes (27 species) and Siluriformes (19 species) comprised the majority of species, with the most representative families being Characidae (14

species) and Loricariidae (12 species), which together accounted for almost half of the recorded species (Table 1). In terms of the number of individuals sampled, the families Loricariidae and Pimelodidae represented more than two-thirds of the total fish abundance (n=2774) and nearly two-thirds of the fish biomass (245 kg) over the entire study period. About 78% of loricariids and 71% of pimelodids were recorded in the pre-damming period. Despite being one of the families with the highest species richness, the Characidae family was less representative in terms of relative abundance, accounting for less than 10% of the total fish sampled.

The most abundant species in both periods was the armored catfish *Hypotomus isbrueckeri* Reis, Weber & Malabarba, 1990, with 570 and 236 individuals captured in the pre- and post-damming periods, respectively (Table 1). In the pre-damming period, the whiptail catfish *Loricariichthys melanocheilus* Reis & Pereira, 2000 (n=332), and the mandi catfish *Iheringichthys labrosus* (Lütken, 1874) (n=317) were the second and third most captured species. The post-damming period was characterized by lower captures of most loricariid and pimelodid species, with the exceptions of the golden sailfin pleco *Hypostomus luteus* (Godoy, 1980) and the bagarito catfish *Parapimelodus valenciennis* (Lütken, 1874). *H. luteus* maintained relatively stable numbers between the two periods, with 98 individuals captured in the pre-damming period and 93 in the post-damming period, while captures of *P. valenciennis* increased from zero in the pre-damming period to 99 individuals in the post-damming period (Table 1).

After the closure of the dam, the reduction in overall captures also resulted in a significant drop in the sampled biomass. The total biomass recorded in the pre-dam period was 270.4 kg, whereas in the post-dam period it decreased to 132.5 kg. The standardized catch per unit effort (CPUE) showed an abundance density of 6,036.8 ind./1000 m² gillnet/12 h in the pre-dam period and 3,157.5 ind./1000 m² gillnet/12 h in the post-dam

period. The biomass density recorded in the pre-dam period was 561.3 kg/1000 m² gillnet/12 h, compared to 348.0 kg/1000 m² gillnet/12 h in the post-dam period (Table 1). *H. luteus* was the species with the highest recorded biomass (53.6 kg) throughout the entire study period, and despite the overall reduction in total biomass after damming, the CPUE for this species increased (pre: 0.046 kg/1000 m² gillnet/12 h; post: 0.067 kg/1000 m² gillnet/12 h). The abundant *H. isbrueckeri* also contributed to the highest biomass (50.5 kg) among all sampled species, with approximately 75% of this biomass being recorded in the pre-damming period (37.9 kg) (Table 1). As the most abundant species, the lower capture of *H. isbrueckeri* after dam closure contributed substantially to the overall reduction in biomass observed during the post-damming period.

3.2. Functional space and functional groups

The first three PCoA axes explained 46.6% of the total variation in species functional traits. Species were distributed into five major functional groups in the functional space (Figure 2, Table 1). The first group is composed of migratory species with larger body sizes, primarily represented by anostomids, curimatids, pimelodids, along with the dourado (*Salminus brasiliensis* Cuvier, 1816) and curimba (*Prochilodus lineatus* Valenciennes, 1837) (Figure 2, Table 1). The second functional group consists mainly of representatives from the family Loricariidae, characterized by their detritivorous feeding habits, inferior mouths, parental care, and preference for benthic habitats (Figure 2, Table 1).

A third group is characterized by species with larger fin areas relative to body size, primarily composed of predatory species from the genera *Hoplias* (trahiras) and *Crenicichla* (pike cichlids) (Figure 2, Table 1). The fourth group is predominantly composed of species from the family Characidae, with notable representatives including

3.3. Taxonomic and functional diversity

The presence of the plant led to a significant reduction in functional richness (FRic) of fish assemblages at impoundment sites and downstream tributaries (Figure 3A). The impoundment sites were the most severely impacted, experiencing a 69% reduction in FRic (Figure 3A). This decline is likely associated with a loss in taxonomic species richness at this location, where a 20% reduction in species richness was also observed (Figure 3B). In contrast, only the upstream tributaries showed a 32% increase in FRic after dam construction (Figure 3A), which was attributed to an increase in the presence of migratory species. The only section where a positive effect on taxonomic richness was observed was in the upstream low-order streams, which exhibited a 15% increase (Figure 3B). This increase in taxonomic richness was not reflected in a significant increase in FRic (Figure 3A).

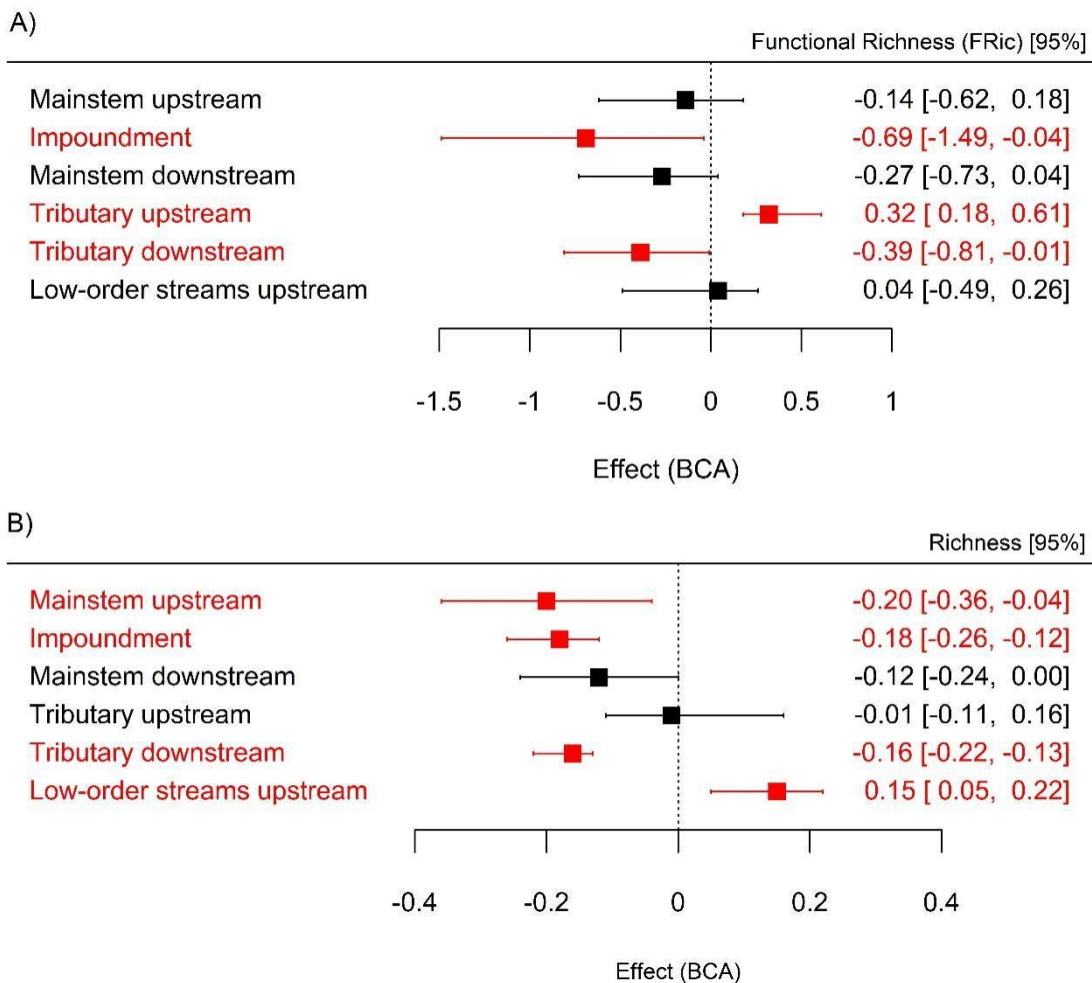


Figure 3. Forestplot with effect-sizes of the dam closure on (A) functional richness and (B) taxonomic richness of fish assemblages in the area of influence of the Foz do Chapecó hydropower plant. Squares symbols represent the average effect-size and lines represent the upper and lower 95% Bootstrapped confidence intervals (CIs). Statistically significant coefficients are depicted in red (i.e., CI ranges that do not cross zero).

The distribution of functional traits (FEve) based on abundance became more equitable at the impoundment sites (7% increase) and in the sites located on tributaries downstream of the dam (3% increase) (Figure 4A). In contrast, FEve remained unchanged between the pre- and post-damming periods for most sites when quantified in terms of biomass (Figure 4B). The exception was the upstream low-order streams, which experienced the greatest decreases in FEve, both in terms of abundance (-20%) and biomass (-28%) (Figure 4A and 4B). Together with the patterns observed for FRic and taxonomic richness, this loss of functional evenness in the low-order streams indicates a post-damming reduction in functional diversity, with key functional roles being dominated by one or a few species, such as the pike cichlid *Crenicichla minuano* and the trahira *Hoplias malabaricus*. These species were replaced by others that share similar functional characteristics, such as the pimelodids *P. valenciennis* and *I. labrosus*.

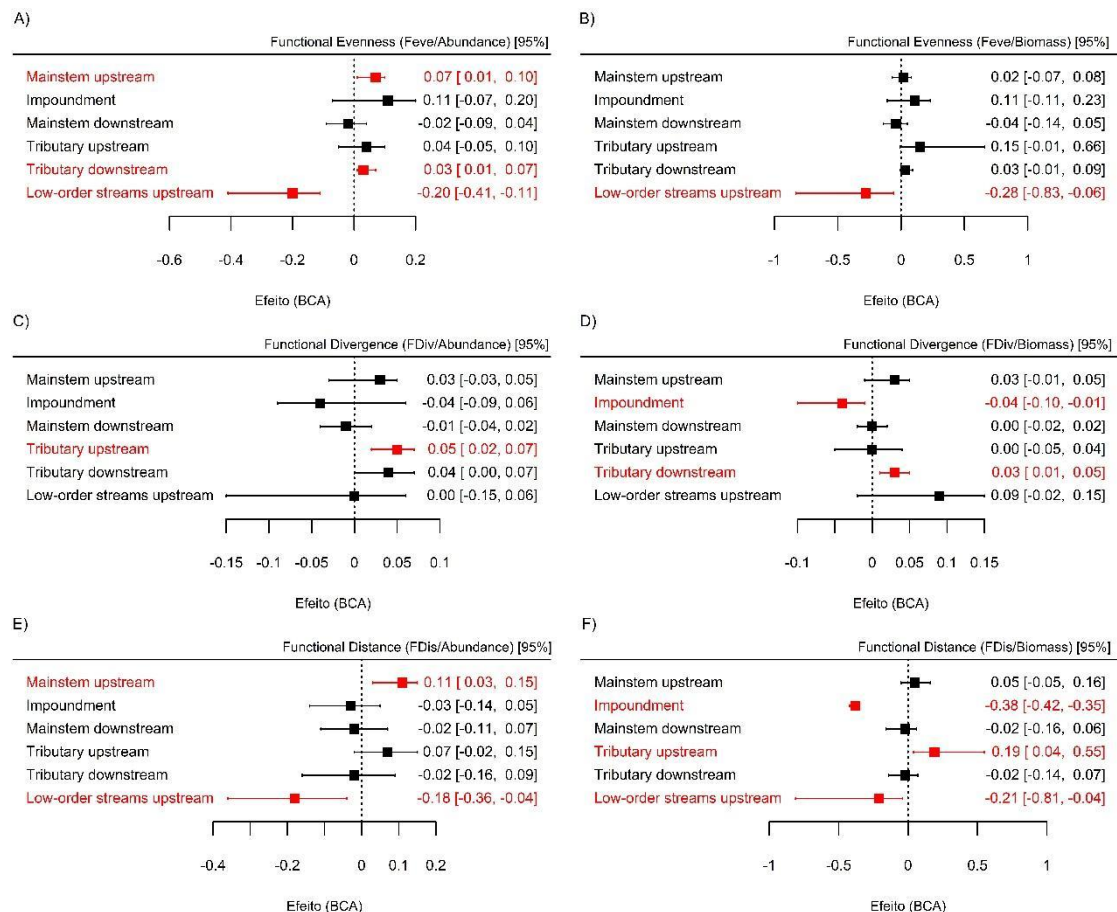


Figure 4: Forestplot with effect-sizes of the dam closure on functional evenness (Feve) (A and B), functional divergence (Fdiv) (C and D) and functional dispersion (Fdis) (E and F) in the area of influence of the Foz do Chapecó hydroelectric plant. Each functional index was fitted with both abundance (A, C, E) and biomass (B, D, F). Square symbols represent the average effect-size and lines represent the upper and lower 95% Bootstrapped confidence intervals (CIs). Statistically significant coefficients are depicted in red (i.e., CI ranges that do not cross zero).

The upstream tributaries showed a 5% increase in functional divergence (FDiv) after damming (Figure 4C). When assessing biomass metrics, a 3% rise in FDiv was observed in the downstream tributaries (Figure 4C). At the impoundment site, there was a 4% decrease in FDiv for biomass, indicating a loss of more specialized functions within the communities that leads to a simplification and concentration of functional traits. Conversely, the increase in FDiv for abundance in the upstream tributaries and for

biomass in the downstream tributaries indicates that the distribution of functional traits shifted towards the extremes of the functional space after the dam construction.

After damming, there was an 11% increase in the average functional dispersion (FDis) of individuals within the functional space in the mainstem river upstream of the dam (Figure 4E). This pattern suggests an increase in the abundance of species with more marginal representation in the functional space at upstream sites, likely driven by the increased abundance of herbivores, piscivores, insectivores, and larger predators (Table 2). Conversely, FDis decreased at the impoundment sites — significant only for biomass, with a 38% reduction — due to declines in the biomass of loricariids (group 2) and migratory fish like *Schizodon nasutus* and *Pimelodus absconditus* (group 2). Significant reductions in FDis after damming were also detected in low-order streams located upstream of the impoundment for both abundance (-18%) and biomass (-21%) (Figure 4F).

3.6. Community-level weighted means

The main changes in CWM values were concentrated in the upstream main channel and involved lower values of detritivory (adjusted $p = 0.004$ and 0.030 for abundance and biomass, respectively), parental care (adjusted $p = 0.017$ and 0.033), and benthic habitat use (adjusted $p = 0.004$ and 0.006), which were traits largely associated with the Loricariidae family (Table 2). In contrast, higher values of piscivory (adjusted $p = 0.004$ and 0.028) and traits related to body and mouth morphology were observed in this section (Table 2). An evaluation of each site category revealed a decrease in CWM values for the benthic habitat trait upstream of the dam, primarily associated with lower abundances of benthic species, particularly loricariids. In contrast, an increase in benthopelagic species from the Characidae family was observed in this area (Table 2).

Moreover, sites upstream of the dam experienced an increase in herbivores, mainly *S. nasutus* (adjusted $p = 0.004$), as well as an increase in piscivorous species, mostly due to higher abundance and biomass of Pimelodidae (Table 2).

The increase in species richness (Figure 3B) was accompanied by higher invertivory in low-order streams (adjusted $p = 0.024$), indicating a greater contribution of species exploiting aquatic invertebrates (Table 2). In tributaries upstream of the dam, piscivory also increased significantly (adjusted $p = 0.014$ for both abundance and biomass). In tributaries located downstream of the dam, CWM values changed little after dam closure, with the exception of an increase in caudal fin ratio (adjusted $p = 0.032$), mainly associated with higher abundance and biomass of *Hypostomus regani*, *Paraloricaria commersonoides*, and *Pimelodus maculatus* (Table 2). Complete results for all trait comparisons, including non-significant interactions, are provided in Table S8.

3.7. Hydrological changes

Generalized additive models revealed significant temporal patterns in river discharge before dam closure in the mainstem sections UCX, URME/URMD, UP and ULH, with effective degrees of freedom ranging from 3.20 to 5.16 and adjusted R^2 values between 0.041 and 0.086 (Table S9; Figure S2). The relatively low adjusted R^2 values likely reflect the high natural variability in river discharge, which is influenced by multiple climatic and hydrological drivers beyond the temporal component modeled here. During this period, discharge exhibited relatively smooth long-term trends, characterized by moderate fluctuations and gradual increases toward the end of the pre-damming period (Figure S2).

Following dam closure, temporal patterns differed among sections. Significant non-linear relationships persisted at UCX (edf = 4.55, $F = 3.04$, $p = 0.016$), UP (edf = 4.67, $F = 3.02$, $p = 0.016$), and ULH (edf = 6.33, $F = 4.87$, $p < 0.001$), whereas post-damming smooth terms were not significant at URME/URMD and RCH (Table S9). The explanatory power of the models increased after dam closure, with adjusted R^2 values ranging from 0.040 to 0.442 and deviance explained varying from 8.1 to 51.6%.

The strongest post-damming response was detected at ULH, where the GAM explained 51.6% of the variation in discharge (adjusted $R^2 = 0.442$). In both UP and ULH, discharge trajectories became more complex after dam closure, showing marked oscillations during the first years of operation followed by increasing trends toward the end of the study period (Figure S2). In contrast, URME/URMD and RCH exhibited comparatively stable flow patterns with no significant temporal trends after impoundment.

4. Discussion

This research highlights that the construction and operation of large run-of-river hydropower plants are associated with changes in the functional structure of fish assemblages that resemble those observed in rivers regulated by dams with larger reservoirs (Arantes et al., 2019; Lima et al., 2018; Oliveira et al., 2018). Run-of-river hydropower plants are generally perceived as less harmful to fish biodiversity due to their smaller or even absent reservoirs (Kuriqi et al., 2021; Nasir, 2021). However, even without the formation of a typically large reservoir, the upstream impoundment and downstream flow regulation associated with the Foz do Chapecó HPP coincided with significant changes in the functional structure of fish assemblages. These patterns were not restricted to migratory species, as sedentary species also exhibited marked changes

following dam closure and subsequent hydropower operation. The observed shifts in fish functional structure are consistent with the changes in river discharge detected after dam closure. The reduction in the natural variability of the flow regime and the increased hydrological regulation likely altered habitat availability and environmental heterogeneity, which may have favored species with functional traits better suited to these modified conditions (Chen et al., 2023; Oliveira et al., 2018).

The spatial position relative to the dam played a key role in the patterns observed in the fish assemblages, as previously anticipated. Marked changes in the functional structure of the fish assemblages were observed following dam construction, particularly upstream of the dam, where the effects of lentification (i.e., the conversion of lotic habitats into lentic environments) were most pronounced. Over time, there was a noticeable shift in the functional composition of these assemblages, characterized by a decreased representation of species with detritivorous feeding habits, parental care and preference for benthic habitats. Declines in these functional traits are directly linked to the reduction in Loricariidae populations at the impoundment and upstream sections of the river. Loricariidae species from the La Plata system are usually associated with fast-flowing waters, being highly susceptible to changes in habitat structure and frequently reported as a group vulnerable to damming in Neotropical rivers (Agostinho et al., 2008; Lima et al., 2018; Vidotto-Magnoni et al., 2015). This interpretation is consistent with the observed changes in river discharge following dam closure, indicating that alterations in flow regime may have contributed to the observed shifts in habitat availability. The reduction in availability of pool habitats after the dam construction — habitats characterized by low flow and high temperatures that typically form during the summer — may also have contributed to the declines in Loricariidae populations as these pools

are considered critical for the reproduction of Loricariidae in the upper Uruguay River (Ávila-Simas et al., 2014).

Important changes in the trophic traits of the fish assemblages were detected in sites located upstream of the dam. Piscivory and herbivory increased mainly in the upstream main channel, following patterns similar to those observed in other large dams constructed in Brazil.(Perônico et al., 2020; Schork and Zaniboni-Filho, 2020). This increase can be linked to the release of dissolved nutrients from submerged organic matter shortly after the dam closure, which has been reported to induce major structural changes in aquatic food webs immediately after damming (Dias et al., 2020; Fráguas and Pompeu, 2021). The new sources of food and shelter associated with flooded vegetation attract a wide range of fish species, often followed by a rise in piscivorous fish such as pike characins (*Acestrorhynchus pantaneiro* Menezes, 1992), saicangas (*Oligosarcus* spp.), and trahiras (*Hoplias* spp.). Indeed, other studies have reported a significant surge in *A. pantaneiro* populations in the Uruguay Basin following the implementation of large hydropower projects (Schork and Zaniboni-Filho, 2020; Zaniboni-Filho et al., 2008). The increase in catches of this and other piscivorous Characiformes can be attributed to several factors, including a rise in smaller forage species (Dos Santos et al., 2017), decreased water turbidity caused by lentification (Schork and Zaniboni-Filho, 2020), and their high feeding adaptability (Cantanhêde et al., 2008).

The observed changes following dam closure and operation were not restricted to the impoundment and mainstem sites, as important shifts in fish assemblages were also detected in tributaries (i.e., rivers and low-order streams). Contrary to our initial expectation, reductions in functional richness within downstream sections were more pronounced in tributary rivers than in the mainstem sections. Flow regulation downstream of the dam have been previously reported to trap fish schools in pools in the mainstem,

which can limit their access to the tributaries and inhibit migrations (Agostinho et al., 2021; Calderon and An, 2016). This pattern is in agreement with the observed hydrological changes after impoundment, which were more evident in the mainstem, while tributaries exhibited comparatively stable flow conditions. Declines in functional richness downstream of the dam were less evident as mainstem fish assemblages tend to have a high functional redundancy, not necessarily reflecting the loss in taxonomic richness observed. Conversely, a slight increase in functional diversity was detected in tributaries upstream of the dam. With the closure of the dam, a notable shift occurred in the fish composition of the upstream tributaries where species from the mainstem river that were previously absent – like the migratory piscivorous *S. brasiliensis* – were first detected. This change in composition may be a consequence of the increased movements of fish into these tributaries as they seek more favorable environments for foraging and reproducing (Marques et al., 2018). Functional diversity reflected these changes in taxonomic composition, with upstream tributaries exhibiting increased representation of piscivorous species and the expansion of functional space.

Fitting functional indices with abundance or biomass data can produce very different outcomes as we observed for FEve, FDiv and FDis. These differences can be attributed to the high diversity in body shapes and sizes in freshwater fish, and the general negative relationships between species size and abundance in the wild – i.e. smaller fish tend to be more abundant than large-bodied species (Cohen, 1988; Fontrodona-Eslava et al., 2021). Such distinction is particularly evident in certain functional groups like the piscivorous species that can range from small (*Oligosarcus* spp. and *A. pantaneiro*) to large (*S. brasiliensis*) even within the same order (Characiformes). Our results highlight the importance of evaluating both abundance and biomass metrics when assessing the effects of environmental change on functional diversity, similar to what other studies

recommend for ecological indicators based on taxonomy (Scharf et al., 2000; Woodward et al., 2005). Biomass metrics, for example, can provide insights about ecosystem functioning by reflecting how resources are allocated across different trophic levels or functional groups, offering a more direct measure of energy transfer and productivity compared to abundance, which only quantifies the number of individuals without accounting for their size or contribution to ecosystem processes (Dornelas et al., 2011; Fontrodona-Eslava et al., 2021). Both data types should be considered complementary rather than substitutes in impact assessments examining functional diversity.

The reduction in functional diversity detected after the construction of Foz do Chapecó HPP raises concerns about the stability and resilience of the aquatic ecosystems as species with unique functions often play crucial roles in maintaining ecological stability (Teichert et al., 2017; Toussaint et al., 2016). Our results suggest that tributaries of the Uruguay River are important refugees for migratory and non-migratory fish that rely on free-flowing habitats to persist, yet the mainstem Uruguay and its tributaries have been heavily fragmented by hydropower dams of different sizes (Couto et al., 2021; Hahn et al., 2011). Upstream from the Foz do Chapecó HPP, there are two large hydropower plants operating in the mainstem Uruguay River – Itá (first operation: 2000) and Machadinho (operation: 2002) HPPs. Further upstream, two additional large plants operate on the mainstem Pelotas and Canoas Rivers that form the Uruguay – Barra Grande HPP (operation: 2005) and Campos Novos HPP (operation: 2007) (Hüffner and Engel, 2011). In this context, migratory species such as the dourado (*S. brasiliensis*), which depend on extensive stretches of free-flowing rivers for reproduction and migration, are particularly vulnerable (Hahn et al., 2011). The fragmentation of river networks severely restricts their access to critical spawning grounds, leading to declines in population size and genetic diversity (Ribolli et al., 2021). Because Foz do Chapecó HPP was the most

recently commissioned large hydropower plant in the Upper Uruguay cascade, fish assemblages in the study area were already subjected to previous fragmentation effects caused by upstream reservoirs, potentially increasing the cumulative impacts on migratory species. This highlights the urgent need to prioritize conservation strategies that maintain connectivity in river systems, particularly in tributaries that still offer suitable habitats for these ecologically and economically important species. Addressing these challenges is essential to ensuring the persistence of migratory fish and the ecosystem services they provide. Previous research have described decades of instability in aquatic ecosystems and fish assemblages after the construction of large dams in Neotropical rivers (Agostinho et al., 2008; Moreira et al., 2023; Pelicice et al., 2021), which suggests that the functional structure of fish assemblages in the Upper Uruguay system should continue under change after the implementation of all these dams.

Despite the consistent patterns observed in functional diversity and community-weighted means, some limitations should be acknowledged. Fish assemblages were monitored for approximately two years after dam closure, representing only the initial stage of ecosystem reorganization. Previous studies have shown that ecological responses to large dams may continue for decades in Neotropical rivers, suggesting that the patterns reported here may represent only the early phase of a longer adjustment process. Furthermore, fish assemblages are influenced by multiple interacting factors acting across temporal and spatial scales (Agostinho et al., 2016; Costa et al., 2024; Keppeler et al., 2025; Schork and Zaniboni-Filho, 2020). Consequently, climatic variability associated with phenomena such as El Niño events, as well as anthropogenic pressures including fishing activities, may also have contributed to the observed patterns (Mol et al., 2000; Pelicice et al., 2017). Although the patterns described here are consistent with expectations from the literature (Arantes et al. 2019; Keppeler et al. 2025), a continuous

monitoring is necessary to fully understand the ecosystem response to run-of-river hydropower operations.

The patterns observed in this study have important implications for the ecological management of large run-of-river hydropower plants. Although the most pronounced changes in fish functional structure were observed in the regulated mainstem, tributaries exhibited comparatively fewer alterations, highlighting their potential role in maintaining ecological integrity within regulated river networks. These findings reinforce the importance of conserving free-flowing tributaries as part of river management strategies, including the "mainstream development, tributary protection" approach (Couto et al., 2021; Jager et al., 2015). Although our study was not designed to evaluate restoration measures such as dam removal, the observed associations between flow alteration and changes in fish functional structure provide ecological information that may help inform future restoration planning and environmental decision-making aimed at conserving fish diversity and ecosystem functioning.

5. Conclusions

Our results indicate that the implementation and operation of a large run-of-river hydropower plant were associated with significant changes in the functional structure and diversity of fish assemblages throughout the area influenced by the Foz do Chapecó HPP, with the strongest responses occurring in the upstream main channel and impoundment sections. The main findings of this study can be summarized as follows:

- **Functional richness (FRic) decreased by 69% at impoundment sites,** accompanied by a 20% reduction in taxonomic richness, indicating substantial functional simplification of fish assemblages.

- **Upstream tributaries exhibited a 32% increase in functional richness,** whereas low-order streams showed a 15% increase in species richness without a corresponding increase in functional richness.
- **Functional dispersion (FDis) increased by 11% in the upstream main channel,** but declined by 38% at impoundment sites and by 18–21% in upstream low-order streams, indicating contrasting responses among sections of the river network.
- **Traits associated with loricariids declined significantly upstream of the dam,** including detritivory, parental care, and benthic habitat use, whereas herbivory and piscivory increased in the upstream main channel.
- **Secondary responses extended to tributaries,** where changes in species composition altered functional richness and trait distributions.

Overall, these results challenge the widespread perception that run-of-river hydropower plants have limited ecological impacts and reinforce the importance of preserving tributaries and maintaining river connectivity to mitigate the long-term consequences of damming.

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7. Data Availability Statement

The datasets generated during the current study are available from the corresponding author on reasonable request.

8. Author contribution statement

Conceptualisation, Data analysis, Preparation of figures and tables: CI, RSR, TBAC. Developing methods, Conducting the research, Data interpretation, writing: CI, TBAC, JDM, GMSF, RSR.

9. Conflicts of interest/Competing interests

The authors declare no conflicts of interest associated with this manuscript.

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1013 **Table 1.** Fish species sampled in the influence area of Foz do Chapecó hydropower plant between June 2007 and September 2010. Fish captures
1014 are summarized by species for both pre-damming (Pre) and post-damming (Post) as relative abundance, biomass in g, and captures per unit of
1015 effort based on relative abundance (CPUEAbu; ind./1000m²/12h) and biomass (CPUEMass; g/1000m²/12h) data.

		Functional group	Relative abundance		Biomass (g)		CPUEAbu(ind./1000m ² gillnet/12h)		CPUEBio(g./1000m ² gillnet/12h)	
			Pre	Pos	Pre	Pos	Pre	Pos	Pre	Pos
CHARACIFORMES										
Acestrorhynchidae	<i>Acestrorhynchus pantaneiro</i> Menezes, 1992	4	35	22	1643.00	1673.00	93.01	86.58	4169.00	6911.00
Anostomidae	<i>Leporinus amae</i> Godoy, 1980	1	22	0	1209.00	0.00	58.08	0.00	3244.00	0.00
	<i>Leporinus striatus</i> Kner, 1858	1	2	4	83.40	230.30	2.57	14.71	107.20	845.00
	<i>Schizodon nasutus</i> Kner, 1858	1	74	52	34120.0 0	17577.00	153.10	109.30	69481.00	35286.00
Bryconidae	<i>Salminus brasiliensis</i> (Cuvier, 1816)	1 & 5	6	14	7963.00	18247.00	11.02	32.90	12749.00	41510.00
Characidae	<i>Andromakhe saguazu</i> (Casciotta, Almirón & Azpelicueta, 2003)	4	23	19	429.30	337.30	70.39	72.42	1169.00	1076.00
	<i>Astyanax abramis</i> (Jenyns, 1842)	4	5	3	94.77	81.60	6.43	3.86	121.80	104.90
	<i>Astyanax aramburui</i> Protogino, Miquelarena & López, 2006	4	1	2	31.30	50.08	1.29	7.17	40.23	98.67
	<i>Astyanax lacustris</i> (Lütken, 1875)	4	47	10	881.00	210.10	115.80	42.46	1995.00	828.10
	<i>Astyanax procerus</i> Lucena, Castro & Bertaco, 2013	4	22	0	480.60	0.00	92.46	0.00	1755.00	0.00
	<i>Bryconamericus iheringii</i> (Boulenger, 1887)	4	3	1	51.20	6.76	8.82	5.88	150.60	39.77
	<i>Diapoma pyrrhopteryx</i> Menezes & Weitzman, 2011	4	5	5	67.90	169.60	24.82	6.43	350.70	218.00
	<i>Galeocharax humeralis</i> (Valenciennes, 1834)	4	4	7	206.60	991.50	11.76	15.25	607.70	1743.00
	<i>Oligosarcus brevioris</i> Menezes, 1987	4	14	4	642.00	97.28	47.97	20.59	21320	431.10
	<i>Oligosarcus jacuiensis</i> Menezes & Ribeiro, 2010	4	1	1	46.90	49.18	1.29	2.94	60.28	144.70
	<i>Oligosarcus jenynsii</i> (Günther, 1864)	4	6	14	374.40	823.90	14.34	61.39	917.90	3480.00
	<i>Psalidodon dissensus</i> (Lucena & Thofehrn, 2013)	4	0	6	0.00	216.10	0.00	7.71	0.00	277.80
	<i>Psalidodon fasciatus</i> (Cuvier, 1819)	4	61	24	2147.00	488.30	129.70	112.30	4753.00	1989.00
	<i>Psalidodon xiru</i> (Lucena, Castro & Bertaco, 2013)	4	5	3	64.10	135.10	24.82	3.86	309.90	173.70
Curimatidae	<i>Cyphocharax voga</i> (Hensel, 1870)	1	0	6	0.00	346.10	0.00	35.29	0.00	2036.00
	<i>Prochilodus lineatus</i> (Valenciennes, 1837)	1	10	6	10241.0 0	9790.00	22.79	13.97	18312.00	15968.00
	<i>Steindachnerina brevipinna</i> (Eigenmann & Eigenmann, 1889)	1	90	15	2784.00	480.10	203.10	37.49	6398.00	1156.00

Erytrinae	<i>Hoplias australis</i> Oyakawa & Mattox, 2009	3	3	3	1893.00	845.60	7.17	14.71	4733.00	4201.00
	<i>Hoplias lacerdae</i> Miranda Ribeiro, 1908	3 & 5	6	7	2841.00	2929.00	26.47	24.82	10431.00	11269.00
	<i>Hoplias malabaricus</i> (Bloch, 1794)	3 & 5	17	13	3437.00	6394.00	75.73	47.97	14894.00	25061.00
Parodontidae	<i>Apareiodon affinis</i> (Steindachner, 1879)	4	127	3	3399.00	79.41	288.90	13.05	8048.00	285.20
Serrasalminae	<i>Serrasalmus maculatus</i> Kner, 1858		4	3	786.10	33.93	10.11	8.45	2001.00	106.60
CICHLIFORMES										
Cichlidae	<i>Crenicichla celidochilus</i> Casciotta, 1987	3	1	1	62.40	48.00	2.94	1.29	183.50	61.70
	<i>Crenicichla minuano</i> Lucena & Kullander, 1992	3	4	0	88.50	0.00	23.53	0.00	520.60	0.00
	<i>Crenicichla missioneira</i> Lucena & Kullander, 1992	3	18	10	1184.00	813.70	46.86	23.70	2371.00	1766.00
	<i>Crenicichla vittata</i> Heckel, 1840	3	2	1	295.50	76.38	5.88	1.29	869.10	91.18
	<i>Geophagus iporangensis</i> Haseman, 1911	3	4	1	118.30	133.60	11.76	2.94	347.90	392.90
	<i>Gymnogeophagus lipokarenos</i> Malabarba, Malabarba & Reis, 2015	3	42	0	934.00	0.00	97.95	0.00	2566.00	0.00
GYMNOTIFORMES										
Gymnotidae	<i>Gymnotus omarorum</i> Richer-de-Forges, Crampton & Albert, 2009	2	8	0	316.20	0.00	26.10	0.00	1222.00	0.00
Sternopygidae	<i>Eigenmannia virescens</i> (Valenciennes, 1836)	4	10	3	205.60	94.92	39.52	10.11	976.60	306.40
PERCIFORMES										
Scianidae	<i>Pachyurus bonariensis</i> Steindachner, 1879	4	94	44	5894.00	1911.00	177.1	207.70	11559.00	8018.00
SILURIFORMES										
Auchenipteridae	<i>Auchenipterus osteomystax</i> (Miranda Ribeiro, 1918)		4	6	154.90	366.00	11.77	29.41	455.60	1895.00
Heptapteridae	<i>Rhamdia quelen</i> (Quoy & Gaimard, 1824)	1	24	9	5004.00	2318.00	70.76	25.73	14146.00	5068.00
Loricariidae	<i>Hemiancistrus fuliginosus</i> Cardoso & Malabarba, 1999	2	88	8	2218.00	185.00	194.10	31.98	5449.00	682.40

Pimelodidae

<i>Hypostomus isbrueckeri</i> Reis, Weber & Malabarba, 1990	2	570	236	37911.0 0	12598.00	1105.00	526.20	73606.00	32230.00
<i>Hypostomus luteus</i> (Godoy, 1980)	2	98	93	31610.0 0	22000.00	147.50	284.50	46006.00	67044.00
<i>Hemiancistrus meizospilos</i> Cardoso & da Silva, 2004	2	145	6	3918.00	191.90	314.60	9.37	9590.00	379.30
<i>Hypostomus regani</i> (Ihering, 1905)	2	24	5	6188.00	566.00	32.50	8.08	8842.00	796.70
<i>Hypostomus roseopunctatus</i> Reis, Weber & Malabarba, 1990	2	53	17	5937.00	2123.00	90.93	31.42	9664.00	3607.00
<i>Hypostomus spiniger</i> (Hensel, 1870)	2	33	7	13130.0 0	1458.00	100.50	20.22	35293.00	4520.00
<i>Hypostomus uruguayensis</i> Reis, Weber & Malabarba, 1990	2	32	0	5485.00	0.00	41.13	0.00	7051.00	0.00
<i>Loricariichthys melanocheilus</i> Reis & Pereira, 2000	2	332	20	12960.0 0	1103.00	493.00	78.67	19483.00	4382.00
<i>Paraloricaria commersonoides</i> (Devincenzi, 1943)	2	50	10	5497.00	1231.00	69.23	28.67	7592.00	3770.00
<i>Rineloricaria capitonia</i> Ghazzi, 2008	2	3	1	135.10	56.50	7.17	1.29	282.80	72.62
<i>Rineloricaria reisi</i> Ghazzi, 2008	2	5	0	441.40	0.00	13.05	0.00	891.80	0.00
<i>Iheringichthys labrosus</i> (Lütken, 1874)	1	317	60	19757.0 0	4608.00	727.00	181.40	47437.00	12668.00
<i>Pimelodus absconditus</i> Azpelicueta, 1995	1	89	33	8165.00	2677.00	150.80	61.18	13042.00	4558.00
<i>Pimelodus atrobrunneus</i> Vidal & Lucena, 1999	1	31	17	2244.00	995.90	58.06	26.45	4177.00	1461.00
<i>Pimelodus maculatus</i> Lacepède, 1803	1	225	67	24591.0 0	13510.00	475.90	200.30	49530.00	33536.00
<i>Parapimelodus valenciennis</i> (Lütken, 1874)	1	0	99	0.00	1153.00	0.00	564.00	0.00	5449.00

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1027 **Table 2.** Community-weighted means (CWMs) of functional traits estimated for the pre- and post-damming periods in the area influenced by the
1028 Foz do Chapecó Hydropower Plant. Only traits showing statistically significant differences after false discovery rate (FDR) correction (adjusted p
1029 < 0.05) for either abundance or biomass are presented for each site category. The complete set of CWM comparisons is provided in Supplementary
1030 Table 9.
1031

Trait	Site category	Metric	Pre damming	Post damming	F	Pr(>F)	p (adjusted)
Herbivore	Main Upstream	Abundance	0.046 ± 0.020	0.126 ± 0.069	15.786	0.001	0.004
Detritivore	Main Upstream	Abundance	0.100 ± 0.016	0.041 ± 0.028	38.038	0.001	0.004
	Main Upstream	Biomass	0.101 ± 0.023	0.051 ± 0.046	10.798	0.013	0.03
Insectivorous	Main Upstream	Abundance	0.033 ± 0.020	0.081 ± 0.020	6.388	0.021	0.048
Molluscophagus	Main Upstream	Abundance	0.128 ± 0.105	0.027 ± 0.040	5.860	0.026	0.050
	Main Upstream	Biomass	0.131 ± 0.095	0.030 ± 0.051	6.320	0.022	0.042
Invertivorous	Low Order Stream Upstream	Abundance	0.002 ± 0.005	0.048 ± 0.034	15.386	0.001	0.024
	Low Order Stream Upstream	Biomass	0.002 ± 0.004	0.055 ± 0.040	16.159	0.001	0.024
Piscivorous	Main Upstream	Abundance	0.011 ± 0.011	0.071 ± 0.036	31.505	0.001	0.004
	Main Upstream	Biomass	0.014 ± 0.017	0.070 ± 0.077	8.159	0.010	0.028
	Tributary Upstream	Abundance	0.025 ± 0.023	0.106 ± 0.082	14.671	0.001	0.014
	Tributary Upstream	Biomass	0.016 ± 0.016	0.088 ± 0.079	13.736	0.001	0.014
Habitat _ Benthic	Main Upstream	Abundance	0.764 ± 0.192	0.294 ± 0.284	19.52	0.001	0.004
	Main Upstream	Biomass	0.759 ± 0.213	0.288 ± 0.338	14.76	0.001	0.006
Habitat _ Benthpelagic	Main Upstream	Abundance	0.235 ± 0.191	0.692 ± 0.294	18.000	0.001	0.004
	Main Upstream	Biomass	0.236 ± 0.209	0.712 ± 0.338	15.32	0.001	0.006
Parental care	Main Upstream	Abundance	0.095 ± 0.032	0.046 ± 0.036	9.892	0.006	0.017
	Main Upstream	Biomass	0.095 ± 0.033	0.043 ± 0.045	7.013	0.016	0.033
Lenght	Main Upstream	Biomass	0.071 ± 0.006	0.092 ± 0.012	12.843	0.002	0.010
Open mouth format	Main Upstream	Abundance	0.067 ± 0.009	0.089 ± 0.019	11.749	0.003	0.009
	Main Upstream	Biomass	0.066 ± 0.011	0.085 ± 0.015	10.869	0.004	0.014
Feeding method	Main Upstream	Abundance	0.058 ± 0.023	0.081 ± 0.040	7.157	0.015	0.038

	Main Upstream	Biomass	0.047 ± 0.017	0.090 ± 0.032	15.567	0.001	0.006
Eyes	Main Upstream	Biomass	0.072 ± 0.013	0.090 ± 0.016	7.827	0.012	0.030
Body shape	Main Upstream	Abundance	0.061 ± 0.015	0.103 ± 0.040	11.567	0.003	0.009
	Main Upstream	Biomass	0.063 ± 0.014	0.094 ± 0.027	11.719	0.003	0.013
Pectoral fin rate	Main Upstream	Biomass	0.115 ± 0.044	0.063 ± 0.021	8653	0.009	0.028
Caudal peduncule strangulation	Main Upstream	Abundance	0.092 ± 0.014	0.078 ± 0.008	6.18	0.023	0.048
Caudal Fin Rate	Tributary downstream	Biomass	0.066 ± 0.009	0.083 ± 0.002	8.38	0.008	0.032
Fin surface	Main Upstream	Abundance	0.097 ± 0.017	0.060 ± 0.017	21.974	0.001	0.004
	Main Upstream	Biomass	0.097 ± 0.021	0.052 ± 0.022	19.53	0.001	0.006

11. Supplementary materials

Table S1. Streamflow gauging stations along the Uruguay River used to obtain discharge data within the area of influence of the Foz do Chapecó Hydropower Plant (HPP). Source: HydroWeb v3.2.7 database.

Table S2: Sites sampled in the area of influence of the Foz do Chapecó hydropower plant sampled from June 2007 to September 2012.

Table S3. Fishing gear employed in the area of influence of the Foz do Chapecó hydropower plant from June 2007 to September 2012. Sampling sites include RIR - Irani (tributary upstream); RMA - Monte Alegre (Low-order stream upstream); RPF - Passo Fundo (tributary upstream); UCX - Uruguay (Caxambu do Sul) (Mainstem downstream); RLB - Lajeado Bonito (Low-order stream upstream); URME - Uruguay river (dam - left bank; URMD - Uruguay river (dam - right bank) (Impoundment); RCH - Chapecó river (Tributary downstream); UP - Uruguay river (Balneário de Pratas) (Mainstem downstream); RBG - Barra Grande (Tributary downstream); ULH - Uruguay river (Balneário de Ilha Redonda) (Mainstem downstream).

Table S4. List of morphological functional traits used in this study (adapted from Villéger et al. 2012). BL - Standard body length, BH - Body height, HCP - Minimum height of the caudal peduncle, CFH - Caudal fin height, CFA - Caudal fin area, DPT - Distance between the pectoral fin insertion to the upper body, HUP - Height from the upper body to the pelvic fin insertion, PFL - Pectoral fin length, PFA - Pectoral fin area, HHY - Head height along the vertical axis of the eye, ED - Eye diameter, DCY - Distance from the center of the eye to the bottom of the head, DMB - Distance from the top of the mouth to the bottom of the head, BW - Body width, HTM - Height of the mouth, WTM - Width of the mouth. Acronyms available in supplementary figure 6.

Table S5. Trophic and morphological traits of the ichthyofauna sampled in the area of influence of the Foz do Chapecó hydropower plant, in the period from June 2007 to September 2012.

Table S6. Biological traits of the ichthyofauna obtained from the specialized literature. Where, Habitat: B=Benthopelagic, D=Benthic, P=Pelagic, Migratory behaviour: Y=Yes and N=No, Parental care: Y=Yes and N=No.

Figure S1. Morphological traits that were measured: CP - Standard body length, ALT - Body height, APC - Minimum height of the caudal peduncle, ANC - Caudal fin height, ANA - Caudal fin area, DPT - Distance between the pectoral fin insertion to the upper body, AC - Height from the upper body to the pelvic fin insertion, ANP - Pectoral fin length, ANP - Pectoral fin area, ACA - Head height along the vertical axis of the eye, DO - Eye diameter, OIC - Distance from the center of the eye to the bottom of the head, BIC - Distance from the top of the mouth to the bottom of the head, LC - Body width, AB - Height of the mouth, LB - Width of the mouth. Adapted from Villéger et al. 2012.

Table S7. Definition, ecological significance, and original references of the four functional diversity indices used to characterize fish assemblages in the area influenced by the Foz do Chapecó Hydropower Plant. Together, these indices provide complementary information on the volume, regularity, divergence, and dispersion of functional traits within multidimensional trait space.

Table S8. Community-weighted means (CWMs) of functional traits estimated for the pre- and post-damming periods in the area influenced by the Foz do Chapecó Hydropower Plant. The table includes the complete set of CWM comparisons for all functional traits, site categories, and abundance- and biomass-based metrics.

Table S9. Results of generalized additive models (GAMs) fitted to monthly discharge series before and after dam closure in the area influenced by the Foz do Chapecó Hydropower Plant. Models included a smooth term for time and the El Niño Southern Oscillation (ENSO) index. The table presents the estimated degrees of freedom (edf), F statistics, p-values, percentage of deviance explained, and adjusted R^2 for each site and period.

Figure S2. Generalized additive model (GAM) fits describing temporal variation in monthly river discharge before and after dam closure in the area influenced by the Foz do Chapecó Hydropower Plant. Models included a smooth term for time and the El Niño Southern Oscillation (ENSO) index as explanatory variables. Solid lines represent the fitted values and shaded areas indicate 95% confidence intervals. Vertical dashed lines denote the beginning of power generation at Foz do Chapecó HPP. Site abbreviations are as follows: UCX = upstream control section, URME/URMD = impoundment section, RCH = downstream tributary, UP = downstream mainstem section, and ULH = lower downstream section.

