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Empirical characterization factors to be used in LCA and assessing the effects of hydropower on fish richness

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ABSTRACT

Hydropower is often presented as a clean, reliable, and renewable energy source, but is also recognized for its potential impacts on aquatic ecosystem biodiversity. We used direct empirical data of change in fish species richness following impoundment to develop ecological indicators to be used in Life Cycle Assessment (LCA), and accounting for hydropower impacts on aquatic ecosystems. Data were collected on 89 sampling stations (63 stations located upstream, and 26 located downstream of a dam) distributed in 26 reservoirs from three biomes (boreal, temperate and tropical). Overall, the impact of hydropower on fish species richness was significant in the tropics, of smaller amplitude in temperate biome and minimal in boreal biome, stressing the need for regionalisation when developing indicators. The impact of hydropower was consistent across scales for a given biome (same directionality and statistical significance across sampling stations and reservoirs). However, the indicators were sensitive to the duration of the study (the period over which data have been collected after impoundment), which can underestimate the impacts. This result highlights the need to account for the duration of the transient dynamics to reach a steady state (rate of change in species richness = 0) before developing ecological indicators. By using the LCA approach, our suggested indicators contribute to fill a major gap in assisting decision-makers when evaluating the potential of alternative energy technologies, such as hydropower, to decarbonize the worldwide economy, while minimizing the impacts on aquatic ecosystems.

1. Introduction

One of the most important challenges we face as a society is the increased demand for energy (SEforALL, 2016, p. 4). In response to this worldwide demand, hydropower is presented as a relatively clean, reliable, and renewable energy source (Tahseen and Karney, 2017; Teodoru et al., 2012), and an interesting option to decarbonize our global economy by reducing greenhouse gas emissions (GHG; Figueres et al., 2017; Potvin et al., 2017). Hydropower supplies <3% of the primary energy worldwide but almost 75% of the world's renewable electricity (International Energy Agency (IEA), 2019; International Hydropower association (IHA), 2020). These numbers will increase in the coming years as many large dams are being constructed worldwide, particularly in developing economies that are mostly located in the tropics (Grill et al., 2015; Winemiller et al., 2016).

Despite its recognized advantages, hydropower can impact aquatic ecosystem functions and biodiversity through the regulation of the river flow, by a drastic change in the hydrological regime, and by the fragmentation of rivers (Bunn and Arthington, 2002; Gracey and Verones, 2016; Renöfalt et al., 2010; Rosenberg et al., 2000). Dams constructed for hydropower transform large rivers (lotic environment) and surrounding lakes into large reservoirs (lentic environment) or series of reservoirs (*sensu* cascade reservoirs; Friedl and Wüest, 2002; Haxton and Findlay, 2009). Upstream of the dam, reservoirs experience variation in water levels far beyond their natural amplitude (*e.g.* drawdown; Kroger, 1973; Zohary and Ostrovsky, 2011). Downstream of the dam, changes in

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seasonal and inter-annual streamflow magnitude and variability are generally reduced (Friedman et al., 1998; Graf, 2006) but can be increased (*e.g.*, hydropeaking; Tonolla et al., 2017), and fish migration can be altered (Pelicice et al., 2015). These modifications can impact the abundance, distribution and population structure of many taxa in the aquatic community at all trophic levels and can ultimately affect the structure and functions of aquatic ecosystems (Furey et al., 2006; Nilsson and Berggren, 2000; Vörösmarty et al., 2010).

Life Cycle Assessment (LCA) is a multidisciplinary framework used to assess the environmental impacts of products, processes and services throughout their whole life cycle (from cradle-to-grave; Finnveden et al., 2009; ISO, 2006). LCA informs about sustainable and sound choices in the context of decision-making and is based on strong scientific evidence. In LCA, inventory flows related to all activities involved in the life cycle (*e.g.*, emissions, resource extraction, change in land use) are first inventoried and then characterized into potential environmental impacts called Characterization Factors (CF; Curran et al., 2011). When compared to other energy production technologies, hydropower scored favorably in LCA studies regarding GHG emissions, air pollution, health risk, acidification and eutrophication of ecosystems (CIRAIG, 2014; Hertwich, 2013; Sathaye et al., 2011). However, the impacts of hydropower on ecosystem quality and biodiversity are still not successfully integrated into LCA.

Including the impacts of hydropower on aquatic ecosystems quality in LCA has been proven to be challenging. It is challenging because of large data requirement, unclear causational effects (*i.e.*, incapacity to clearly describe physical, chemical or biological mechanisms of an impact pathway linking a specific inventory flow with an impact endpoint), incomplete coverage of ecological impacts, and spatial and temporal scaling issues that can hinder its application and validity (Gracey and Verones, 2016; McManamay et al., 2015; Milà i Canals et al., 2009; Teixeira et al., 2016). Different metrics have been proposed to measure the impacts of a product, process or service on ecosystems quality in LCA such as the difference in the number of species (*i.e.*, richness), ecosystem scarcity and vulnerability, functional diversity and habitat suitability curve (Curran et al., 2011; Damiani et al., 2019; Souza et al., 2013). Experts agreed – without a clear consensus – that change in species richness, which is represented as a Potentially Disappeared Fraction of species (PDF) in LCA, was a good and simple starting point to assess biodiversity impacts (Teixeira et al., 2016).

When a change in species richness is used in LCA, it is essential to consider the adequate spatial and temporal scale of impacts. Patterns observed locally (*e.g.*, in a reservoir) cannot always be extrapolated within or across regions. It is also important to evaluate the impact at the steady state (*i.e.*, the time at which change in biodiversity stabilize after impoundment). In LCA, very little studies examined the impacts of hydropower on ecosystems quality, or examined if patterns can be extrapolated across spatial and temporal scales (but see de Baan et al., (2013) for a multiple spatial scales LCA study). Yet, no study uses empirical data to derive CF and Impact Score (IS) to quantify the impacts of hydropower on aquatic ecosystems.

Here, we used empirically derived rate of change in fish species richness over time after impoundment, across 89 sampling stations, belonging to 26 storage reservoirs from boreal, temperate and tropical biomes. The focus of this study was on storage reservoirs because longitudinal data (*i.e.*, data collected over time, before and after damming) were lacking from the other technologies (*e.g.*, run of the river and pumping stations). Our goals were to 1) develop robust empirical CFs (based on PDF) across three spatial scales (sampling station, reservoir and biome), 2) calculate the impact score of impoundment, *i.e.*, transforming a river into a reservoir and its subsequent occupation by the reservoir (ISR; PDF·m^2·year of affected area, accounting for the affected surface and time of occupation), and relate the Impact Score to hydropower generation (IS; PDF·m^2·year of affected area/kWh), and 3) to test the need for regionalisation by examining if the observed patterns were consistent across the three biomes.

2. Materials and methods

2.1. General approach

Our novel approach to generate Characterization Factors (CF) and Impact Scores (IS) in LCA was based on the use of direct empirical patterns of change in fish richness in response to river impoundment from an extensive literature search. To calculate CF, we used the Potentially Disappeared Fraction of species (PDF) as the unit to express a temporary change in fish richness in response to reservoir occupation and hydropower in time and space (*i.e.*, the transformed state and steady-state; PDF·year·m^2; Fig. 1; Jolliet et al., 2003). Theoretically, the PDF is the difference between the observed richness before impoundment (R_0) and the richness observed when the fish community reached a steady-state (*i.e.*, change in richness due to impoundment = 0) in the reservoirs or downstream of the dams (R_2 in Fig. 1). Fish can disappear due to altered flow regimes, change in habitat quality, barrier to migration or change in trophic interactions upstream and downstream of the dam. Mitigation measures (*e.g.*, fish ladder, creating and restoring spawning and rearing habitats, minimization of hydropeaking effects; Person et al., 2014; Tonolla et al., 2017) can reduce PDF (smaller ΔR; Fig. 1).

PDF has the advantage to be compatible with other damage oriented impact assessment methods addressing ecosystems quality in LCA such as IMPACT 2002+ (Jolliet et al., 2003) and Impact World + (Bulle et al., 2019), and has been recommended by the UNEP-SETAC Life Cycle Initiative as an adequate and consistent biodiversity attribute (Verones et al., 2017). In this study, we adapted the framework for change in land occupation proposed by de Baan et al. (2013) and Chaudhary et al. (2015) to assess the occupation of a water body (occupation of a riverbed measured in surface-time units, m^2·year). In our study, the CF is expressed in PDF units, or implicitly PDF·m^2·year/m^2·year of water body occupied.

Because no reliable data were available to evaluate the biodiversity recovery when powerplants are decommissioned and dam are removed, we were not able to address the recovery phase in the LCA and therefore focused our efforts on the impacts during the occupation phase (*i.e.*, period covering the construction of the dam until decommission; Fig. 1).

For each reservoir, we calculated two impact scores: one for the reservoir creation and occupation due to the construction of the dam (impact score of the Reservoir (ISR); where CFs were multiplied by the inventory flow which is the area-time (m^2·year) of the rivers and lakes before impoundment and then occupied by the reservoir (hereafter called the annually affected area; see Fig. A) and one for the hydropower generation (IS; where ISR was divided by the annual kWh produced for a given powerplant). We also took a multi-scale approach to examine if patterns observed at the sampling station, reservoir and biome scale were consistent.

2.2. Richness data extraction and literature search

The literature search for this paper has been performed previously for another companion *meta*-analysis examining the global effect of dams on fish biodiversity (Turgeon et al., 2019b). In a nutshell, the search for the *meta*-analysis resulted in 67 references that met our research criteria. See Turgeon et al. (2019b) for a detailed methodology about the literature search, and data extraction.

For this paper, we refined our selection criteria to include only references that had unbiased quantitative data on the fish community before and after impoundment (*i.e.,* longitudinal data), and where the main purpose of the dam was to produce hydropower. A total of 30 references met our selection criteria (Databases A and B). The raw data used to calculate the rate of change in richness are presented in Database B, as well as some potential bias and confounding factors (artisanal fisheries, stocking, water quality) that can affect the estimated richness in the different studies. The sampling effort and fishing gears varied

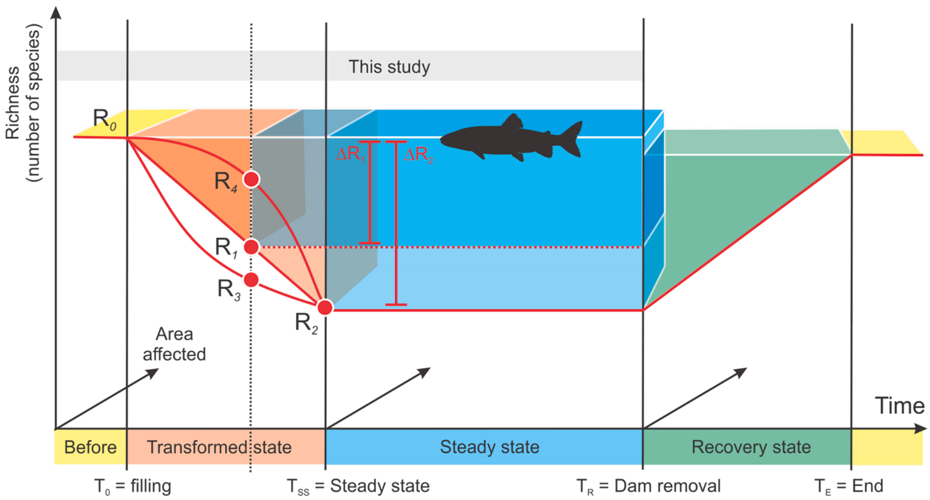

Fig. 1. Schematic representation of an area-time framework representing the rate of change in richness experienced in a reservoir. R_0 represents the richness before impoundment, R_i represents different richness during the transformed state of the reservoir and where the fish community respond to environmental change following impoundment. The ΔRs represent the steady state where fish community should have reached a new equilibrium and where the rate of change in fish species should stabilize. The recovery state should start when the reservoir and dam will be decommissioned. This study addresses the period between the before impoundment to the reach of the steady state.

across references, and sometimes varied among years in some references. The sampling effort was provided in 48% of the sampling station and varied from 1 to 137 gillnets per year (mean ± SD = 7.8 ± 10.0 gillnets per year). About 40% of the references did not have similar efforts across years. Most of the studies used gill nets, which can underestimate small littoral and pelagic species. The duration of the study also varied among references (mean ± SD = 14.2 ± 9.2 years) and was much shorter in the tropics. Rarefied richness (*i.e.*, controlling for the number of samples) was reported in 13% of the references.

2.3. Extracting the area affected by the dam and reservoir

To extract the area affected by the construction of the dam and reservoir, we extracted the area occupied by the rivers and lakes before impoundment both upstream and downstream of the dam (Fig. A). Change in land use from terrestrial to reservoir (inundated land area) is out of the scope of this study, but see Dorber et al. (2018) for a suggested approach to model the net land occupation of hydropower reservoirs in LCA. To get the affected area of each reservoir, we used various sources. For recent reservoirs, we used Google Earth Pro with the historical satellite imagery tool (Landsat/Copernicus images). Other sources of historical maps consisted in the USGS historical topographic maps for most of the United States reservoirs (https://viewer.nationalmap.gov/basic/), the Old Maps Online website for old reservoirs in Africa and South America (http://www.oldmapsonline.org/). A representative image of the riverbed before impoundment was exported as a raster image in QGIS (v.2.18.16; http://www.qgis.org). The affected area was hand drawn as a polygon in a vector layer, and the area of the polygon was extracted. Two polygons of the riverbed before impoundment were extracted per reservoir, one upstream and one downstream of the dam (Fig. A; the dark blue area represent the affected area). Upstream, we assumed that the impacts of the reservoir and the dam on fish community did not go beyond the impounded area and thus, used the upper end of the reservoir as the upper limit of the affected area. For downstream stations, we extracted the area of the polygon for five different distances from the dam (5, 10, 15, 20, 25 km), as well as the distance at which the fish communities were sampled (Database A; mean ± SD; 75.9 km ± 125.3 km; median; 9.4 km). The affected areas calculated with the five different distances and the distance at which fish were sampled all strongly correlate (matrix of Pearson correlation r greater than 0.80; *unpublished analysis*), suggesting that the impact is strongly dependent of the river width. We used 10 km downstream of the dam to set the lower limit of the polygon (based on the median value of 9.4 km).

2.4. Calculation of the rate of change in richness over time

2.4.1. Sampling station scale

For each sampling station *i*, located either upstream or downstream of the dam for reservoir *j*, we calculated the rate of change in richness over time. Because some studies observed non-linear patterns in richness over time following impoundment (Agostinho et al., 1994; Lima et al., 2016), we first tested if a linear function can approximate the rate of change in richness over time with Generalized Additive Models (GAM) with a Gaussian error structure (Wood, 2014, 2006). GAM allow to model non-linear relationships between the response variable (richness) and the explanatory variable (time). All analyses were performed using R v.3.6.3 (R Core Team, 2020). To test for non-linearity of the patterns, we only used the time series with more than four observations (46.2% of the time series; Database A). From those 42 times series, only seven (16.6%) showed a significant non-linear pattern (Database A) with limited curvature (estimated degrees of freedom (edf) < 3). The other time series showed either a linear decreasing, a linear increasing trend or no trend in time (Database A). In light of these results, the rate of change in richness will be approximated by a linear model. See discussion for potential limitations and biased interpretation associated with this assumption.

At the sampling station scale, the rate of change in richness was extracted using the estimated slope of the regression between richness and time (Equation 1) from a general linear model (LM) and we used the standard error of the estimate to calculate the 95% Confidence Interval (CI; Database A). A positive rate of change in richness represented a gain in species, and a negative rate of change represented a loss in species.

2.4.2. Reservoir and biome scales

When more than one station were sampled per reservoir, we used generalized linear mixed effect models (GLMM; lmer function in lme4 package v.1.1–13; Bates et al., 2018). GLMM is an extension to the generalized linear model (GLM) and where the explanatory variables contain random effects in addition to the usual fixed effects. GLMM were used to calculate the rate of change in richness over time at the reservoir, biome and global scales while controlling for the spatial non-independence of the data (pseudoreplication) by using random-effects (random effect = stations at the reservoir scale). We ran separate models for upstream and downstream locations at the reservoir and biome scales. At the reservoir scale, we controlled for spatial non-independence by using sampling stations as a random factor. At the biome scale, we controlled for spatial non-independence by nesting each sampling station *i* into their respective reservoir *j*. Globally, to compare if the rate of change in richness following impoundment differed across biomes (interaction; years*biomes), we also controlled for spatial non-independence by nesting each sampling station i into their respective reservoir j, and we used the boreal biome as the contrast.

2.5. Calculation of Characterization Factors (CF)

2.5.1. Sampling station scale

To calculate CFs, which are calculated as a Potentially Disappeared Fraction of species (PDF), we multiplied the observed rate of change in richness ($\Delta R/\Delta t$; where ΔR stands as the difference in richness and Δt stands for the duration of the study) by the time it takes to reach a defined steady-state t_{ss} (time horizon at which we considered that the rate of change in richness = 0; Fig. 1) as per Equation 1, and divided the result by the average richness observed before impoundment for a given sampling station (R_{0ij}). By definition, in LCA, the CF is positive if we observe a loss in species and negative when we observe a gain in species, so we multiplied the equation by minus 1 (Equation 1). We did this for each sampling station i.

The duration at which fish richness has been sampled for a given study (Δt) varied greatly across studies and biomes (*e.g.*, from less than five years to 40 years; Databases A and B). This can be problematic when comparing short duration studies with longer ones, because the longer the time after impoundment (Δt), the bigger the ΔR will be and the more likely the steady-state will be reached. Short duration studies are, therefore, likely to underestimate PDF (Fig. 1; see R_1 vs. R_2). To make studies comparable, we tested how PDFs extrapolated at different times to reach the steady-state (t_{ss} = 5, 10, 15, 20, 25 and 30 years after impoundment; Equation 1) were plausible, and compared the scenarios with the Observed duration for longer time series with a sensitivity analysis.

To calculate the uncertainty associated with the CFs, we used the standard error (SE) from the estimate of the rate of change in richness (from the GLMM) and multiplied it by the different scenarios of time to reach the steady-state, and then divided it by the average richness observed before impoundment. From this scaled SE, we calculated the 95% CI.

Equation 1: Characterization Factors at the sampling station scale

$$CF_{ij} = -\left(\frac{\left(\frac{\Delta R_{ij}}{\Delta t_{ij}}\right) * t_{ss}}{R_{0ij}}\right)$$

$$CF_{ij} = \left(\frac{R_{0,ij} - R_{ss,ij}}{R_{0ij}}\right)$$

where ($\Delta R_{ij}/\Delta t_{ij}$) is the observed rate of change in richness extracted in sampling station i in reservoir j using the slope of the regression between the observed change in richness (ΔR) for a given period of time (Δt), and t_{ss} are the different scenarios of time until reaching the steady state (5, 10, 15, 20, 25 and 30 years after impoundment).

2.5.2. Reservoir and biome scales

To test if the CFs were valid and robust across scales (sampling station, reservoir and biome), we also computed CFs at the reservoir and biome scales. At the reservoir scale, we calculated a mean upstream CF (if more than one sampling station was available), a mean downstream CF, as well as a Total CF (upstream CF + downstream CF when both upstream and downstream stations were available). For the upstream CF, we averaged the CFs calculated for each upstream station in reservoir j. We then squared the SE associated with the coefficient of regression (slope of the observed change in richness for a given period) for each upstream sampling station of reservoir j added them together to get the total variance for reservoir j. We then divided this variance by the number of sampling stations in reservoir j raised to the power of 2, and square rooted that variance to get the SE of the CF, and then calculated the 95% CI. We did the same procedure for downstream stations. A total CF for a given reservoir j was calculated only if data from upstream and downstream stations were available. To do so, we added the mean upstream CF to the mean downstream CF. To calculate the CF at the biome scale, we used CF_j as units (calculated at the reservoir scale) instead of CF_i (calculated at the sampling station scale). We only used the reservoirs for which we had both upstream and downstream stations.

2.6. Calculation of impact scores (ISR and IS)

We were also interested to evaluate the annual impact of creating and operating a specific reservoir (ISR; *i.e.*, evaluating PDF of the area affected upstream and downstream of the dam during one year), and to relate it to the annual hydropower production or generation (IS; elementary flow = kWh produced for a given reservoir). To do so, we multiplied the CF by the area affected (*i.e.*, area occupied by the river and lakes before impoundment; Fig. A). Because we expect different impacts upstream and downstream of the dam, we calculated the annual impact score of creating and operating a reservoir (ISR; $PDF \cdot m^2 \cdot year$) as the sum of downstream (ISR_{dj}) and upstream impacts (ISR_{uj}; Fig. A). Impact scores were calculated at the reservoir and biome scales.

Equation 2: Annual impact score (ISR) of a reservoir j

$$ISR_j = (ISR_{dj} + ISR_{uj}) = \left((CF_{dj} * A_{dj}) + (CF_{uj} * A_{uj})\right) * 1 \text{ year}$$

where A stands for the area affected upstream (uj) or downstream (dj) of the dam for reservoir j (Fig. A).

To calculate the impact score per unit of hydropower produced by a powerplant, the annual impact score of a dam ISR_j was divided by the annual electricity production, Pj (kWh/year).

Equation 3: Impact score per kWh of an operating power plant associated with reservoir j

$$IS_j = \frac{ISR_j}{P_j} = \frac{(CF_{dj} * A_{dj}) + (CF_{uj} * A_{uj}) * 1 \text{ year}}{P_j}$$

3. Results

3.1. Rate of change in fish richness across scales and biomes

Upstream and downstream of the dam, the rate of change in fish richness over time varied across sampling stations, reservoirs and biomes (Fig. 2). Overall, when biomes, reservoirs and sampling stations were combined in a GLMM, richness significantly decreased over time at a rate of 0.29 species per year upstream of the dam (estimate ± SD = −0.293 ± 0.074; 95% CI = −0.439 to −0.148) and at a comparable rate downstream of the dam (0.26 species per year; estimate ± SD = −0.264 ± 0.082; 95% CI = −0.424 to −0.104). The rate of change in richness decreased much faster in the tropic than in temperate and boreal reservoirs (estimate ± SD of the interaction terms between the rate of change and biomes (boreal vs. tropical) = −1.380 ± 0.202; 95% CI = −1.777 to −0.984).

In the boreal biome, there was no significant change in richness over time at all scales (sampling station, reservoir and biome; Fig. 2a, b), and locations (upstream and downstream; GLMM; 95% CI overlapped with zero; Fig. 2a, b). In temperate and tropical regions, the patterns were less consistent, and more variation was observed than in the boreal region. Some sampling stations and reservoirs showed either a significant decrease or an increase in richness over time following impoundment (Fig. 2c–f). In these two biomes, we observed a significant decrease in richness over time at the biome scale for upstream stations (glmm; loss of 0.26 in temperate and 1.6 species per year in tropical reservoirs; Fig. 2c, e). Downstream of the dam, we observed a significant decrease in richness in the temperate region (loss of 0.34 species per year) but not in the tropics (Fig. 2d, f).

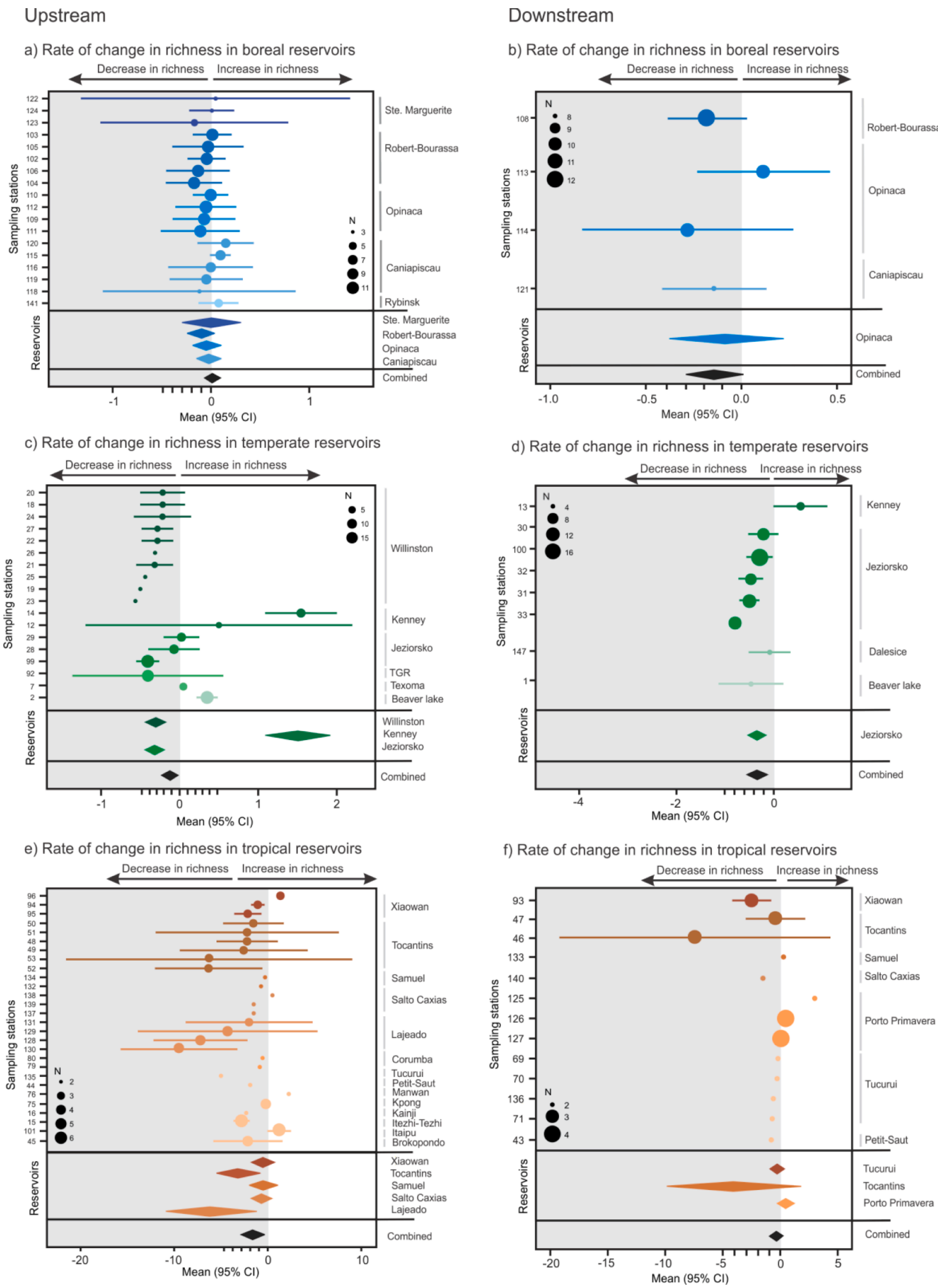

Fig. 2. Empirically derived rates of change in richness and their 95% CI in upstream (left panels) and downstream sampling stations (right panels), and reservoirs from three biomes: boreal (23 sampling stations, 5 reservoirs), temperate (26 sampling stations, 7 reservoirs) and tropical (41 sampling stations, 15 reservoirs). Each circle represents a sampling stations (sampling station ID on the y-axis) that belongs to a reservoir (the name of the reservoirs are provided on the left of each panel). The size of the circles represents the number of observations in the time series used to derive the rate of change in richness with a linear regression. The size of the circles and x-axes differ between the six panels. A positive value means a gain in species, a negative value means a loss in species. Information about the sampling station (number on the y-axis) can be found in Database A.

3.2. Characterization Factors (CF)

The magnitude of CFs was sensitive to the assumption of reaching the steady-state (t_{SS}), and differed across biomes and scales (Fig. 3 and Table 1; see also Figs. B.1, B.2 and B.3 in Supplemental Information). At the sampling station scale, when all sampling stations were combined, CFs (PDF) were higher with higher richness before impoundment (estimate ± SE = 0.17 ± 0.05; p = 0.005). Richness before impoundment was strongly correlated with biome, where tropical reservoirs had a much higher richness before impoundment than boreal reservoirs ($p < 0.001$). In boreal reservoirs, there was no significant CF upstream and downstream of the dam (Fig. B.1) for all steady-state scenarios. When data were combined at the reservoir scale (Fig. 3a), no significant CFs were observed (upstream, downstream and total CF; Fig. 3b and Table 1) and the Total CF at the biome scale was also not significant (Table 1).

At the sampling station scale, in temperate and tropical ecosystems,

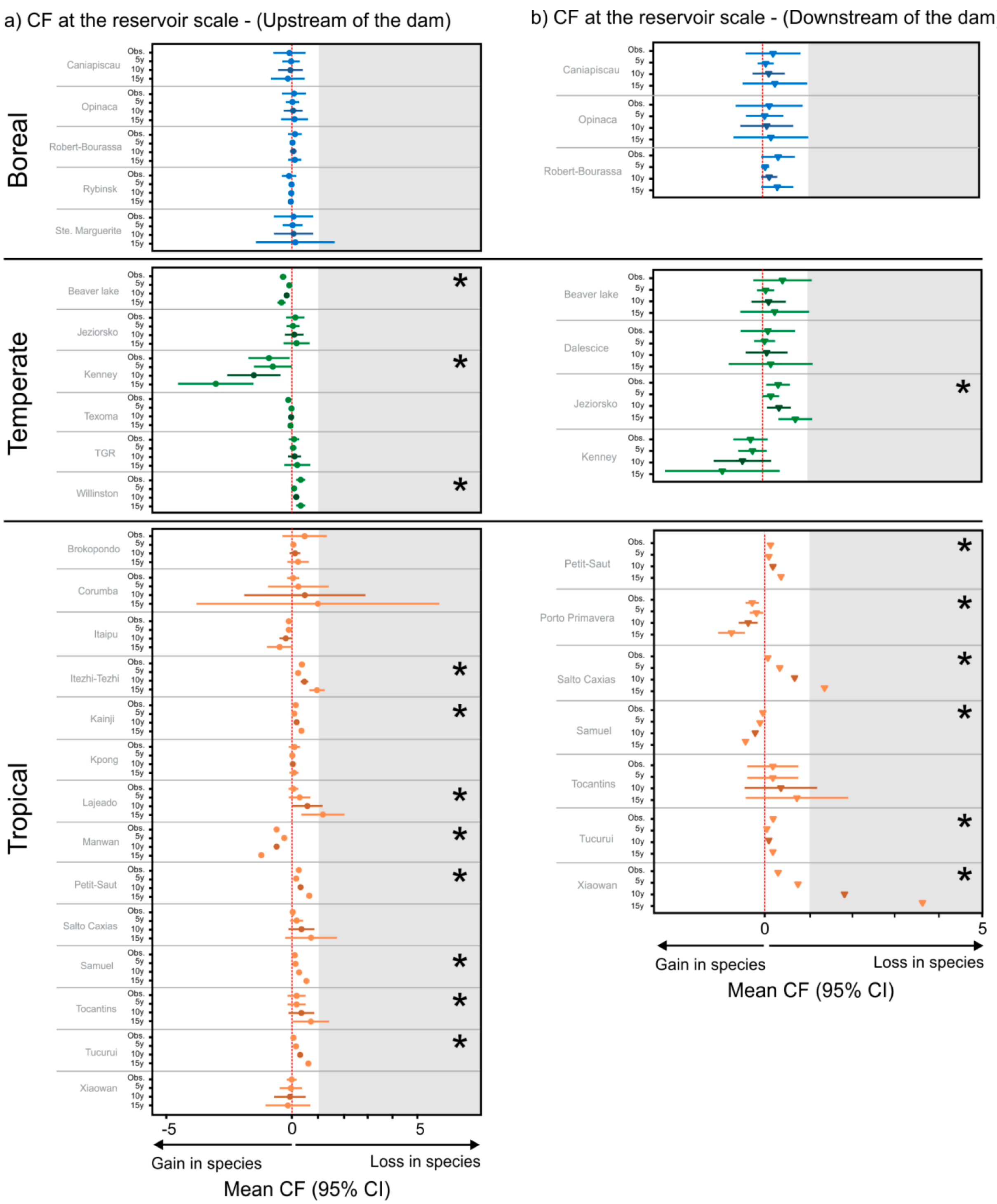

Fig. 3. Characterization factor estimates (CFs) and their 95% CI at the reservoir scale for three biomes, a) upstream and b) downstream of the dam, for the observed duration of the study (Obs.), and the three extrapolated most plausible scenarios of time until reaching the steady state (5y, 10y, 15y). Extrapolated CF values in the grey area means that 100% (or more) of the species were lost, which is not possible. Stars beside the CF values indicate a statistically significant CF (where the 95% CI does not overlap with zero). A positive value means a loss in species, a negative value means a gain in species.

Table 1
Estimates ± Standard Error (SE) for Characterization Factors (CF), Impact Scores for the creation of the reservoir (ISR) and Impact Scores to produce hydropower (IS) at the reservoir and biome scales using the steady-state scenario of 10 years. Values in italics represent significant CF, ISR and IS. *SE of 0.000 are presented when there were only two data points in the time series (before and after impoundment).

Reservoirs/ biome	Biome	U, D, or U + D	CF (PDF*y) Estimate ± SE	ISR (PDF*km^2*y) × 1.0E + 08 Estimate ± SE	IS (PDF*km^2*y/ kwh) Estimate ± SE
BOREAL	**B**	**U + D**	**0.159 ± 0.204**	**0.604 ± 12.30**	**−0.061 ± 0.012**
Ste-Marguerite	B	U	0.069 ± 0.401	0.240 ± 1.397	0.009 ± 0.051
Rybinsk	B	U	−0.021 ± 0.027	−0.890 ± 1.183	−0.138 ± 0.184
Robert Bourassa	B	U + D	0.243 ± 0.247	3.360 ± 4.758	0.009 ± 0.013
Opinaca	B	U + D	0.148 ± 0.813	3.084 ± 15.77	0.008 ± 0.042
Caniapiscau	B	U + D	−0.085 ± 0.850	−4.630 ± 33.02	−0.201 ± 1.436
TEMPERATE	**T**	**U + D**	**−0.524 ± 0.442**	**−0.028 ± 0.029**	**−0.102 ± 0.350**
Three Gorges	T	U	0.109 ± *0.000	5.152 ± 0.000	0.005 ± 0.000
Texoma	T	U	−0.026 ± 0.007	−0.032 ± 0.008	−0.013 ± 0.003
Kenney	T	U + D	−1.974 ± 2.351	−0.067 ± 0.042	−0.559 ± 0.350
Jeziorsko	T	U + D	0.468 ± 0.575	0.048 ± 0.075	0.288 ± 0.440
Dalesice	T	D	0.093 ± 0.244	−0.064 ± 0.068	–
Beaver Lake	T	U + D	−0.068 ± 0.103	−0.064 ± 0.018	−0.035 ± 0.010
TROPICAL	**TR**	**U + D**	***0.781 ± 0.148***	***2.620 ± 0.721***	***0.056 ± 0.010***
Xiaowan	TR	U + D	1.853 ± 0.987	0.024 ± 1.906	−0.000 ± 0.010
Tucurui	TR	U + D	*0.421 ± 0.164*	*−10.35 ± 2.603*	*0.048 ± 0.012*
Tocantins	TR	U + D	0.747 ± 1.011	2.210 ± 1.552	0.093 ± 0.065
Samuel	TR	U + D	0.069 ± 0.144	0.737 ± 0.194	0.081 ± 0.021
Salto Caxias	TR	U + D	*1.065 ± 0.192*	*2.057 ± 0.403*	*0.038 ± 0.007*
Porto Primavera	TR	D	*−0.381 ± 0.111*	*−0.891 ± 0.259*	*−0.008 ± 0.002*
Petit Saut	TR	U + D	*0.532 ± 0.174*	*0.350 ± 0.085*	*0.075 ± 0.018*
Manwan	TR	U	*−0.608 ± 0.000*	*−0.518 ± 0.000*	*−0.007 ± 0.000*
Lajeado	TR	U	*0.619 ± 0.310*	*5.120 ± 2.604*	*0.116 ± 0.058*
Kpong	TR	U	0.042 ± 0.046	0.008 ± 0.009	0.001 ± 0.001
Kainji	TR	U	*0.192 ± 0.000*	*2.298 ± 0.000*	*0.165 ± 0.000*
Itezhi-Tezhi	TR	U	*0.501 ± 0.078*	*0.481 ± 0.076*	*0.076 ± 0.012*
Itaipu	TR	U	−0.243 ± 0.129	−1.872 ± 0.996	−0.002 ± 0.001
Corumba	TR	U	0.518 ± 0.000	0.390 ± 0.000	0.027 ± 0.000
Brokopondo	TR	U	0.124 ± 0.109	0.510 ± 0.451	0.057 ± 0.050

patterns were more heterogeneous. There were some significant PDFs, but also gain in species upstream (Fig. B.2a and Fig. B.3a) and downstream of the dam in some stations (Fig. B.2b and Fig. B.3b). When data were combined at the biome scale for temperate reservoirs, the total CF, which is not significant, diverged from the reservoir and sampling station scale patterns (*i.e.*, significant loss or gain in richness; Fig. 3, Fig. B.2 and Table 1). Because we need upstream and downstream stations to generate the total CF at the reservoir and biome scales, we used a subset of reservoirs. Two reservoirs in this subset experienced an increase in richness (Fig. 3 and Table 1). We also observed significant gains and losses of species at the sampling station and reservoir scales (Fig. 3 and Table 1), and a significant PDF at the biome scale in the tropics (Table 1).

Sensitivity analysis (Fig. C) suggested that simulated CFs beyond the 10 years scenario (10SS) to reach the steady-state after impoundment was unlikely because many reservoirs lost 100% of the original richness, which was never observed in any reservoirs (Fig. 3). The 5y time scenario until reaching the steady-state scenario underestimated species loss compared to the observed duration (Fig. 3 and Fig. C). For these reasons, we considered a 10y steady-state scenario as being the most plausible to compare the impact of dams and impoundment across biomes and reservoirs and to compute ISR and IS.

3.3. Impact Scores for the creation of the reservoir (ISR) and hydroelectricity production (IS)

Annual Impact Scores of reservoirs (ISR) and for the corresponding hydropower production (IS) differed across reservoirs and biomes (Fig. 4, Table 1). In boreal and temperate reservoirs, ISRs were not significant for the observed duration of the study (O; Fig. 4a, b), nor for the steady-state scenario of 10 y (SS10; Fig. 4a, b). Four tropical reservoirs showed significant ISRs (Fig. 4c). When the ISR and IS were calculated at the biome scale, we observed small and non-significant ISR for boreal and temperate biomes, and a significant ISR for the tropics (Fig. 4). The directionality and significance of IS were comparable to ISR for both the reservoirs and biome scales (Table 1).

4. Discussion

4.1. Regionalisation is needed

Based on available empirical data, we demonstrated that regionalisation is needed to evaluate the impacts of hydropower in LCA because the observed rate of change in fish richness in reservoirs varied across biomes, being minimal in boreal, marginal in temperate ecosystems, and significant in the tropics. This result suggests that hydropower in northern countries (*e.g.*, Canada, Russia, Norway, Sweden, Finland, Iceland), accounting for 14% of the worldwide installed hydropower capacity in 2019 (IHA, 2020), may have limited impacts on fish richness. On the other hand, our dataset demonstrated that hydropower in the tropics has significant impacts on fish richness at all scales. Recently, rivers of species-rich tropical regions located in Brazil (installed capacity of 100.2 GW, 64% of the generated electricity in Brazil, 8% globally; IHA, 2020) and China (installed capacity of 352.3 GW, 17% of the generated energy in China, 27% globally; IHA, 2020), have been extensively harnessed for hydropower (Stickler et al., 2013; Winemiller et al., 2016; Ziv et al., 2012). Future hydropower development (planned and currently in construction) is concentrated in China, the Mekong region, Latin America and Africa, and the largest potential for future development is in Asia (IHA, 2020). These regions have a high fish richness and endemic species, some of these regions are recognized as aquatic biodiversity hotspots, and they will be particularly impacted by climate change due to a loss in water availability and increased temperature (Xenopoulos and Lodge, 2006). In a collective effort to decarbonize the worldwide economy and reduce GHG emissions, we urgently need appropriate supporting decision tools that consider long-term

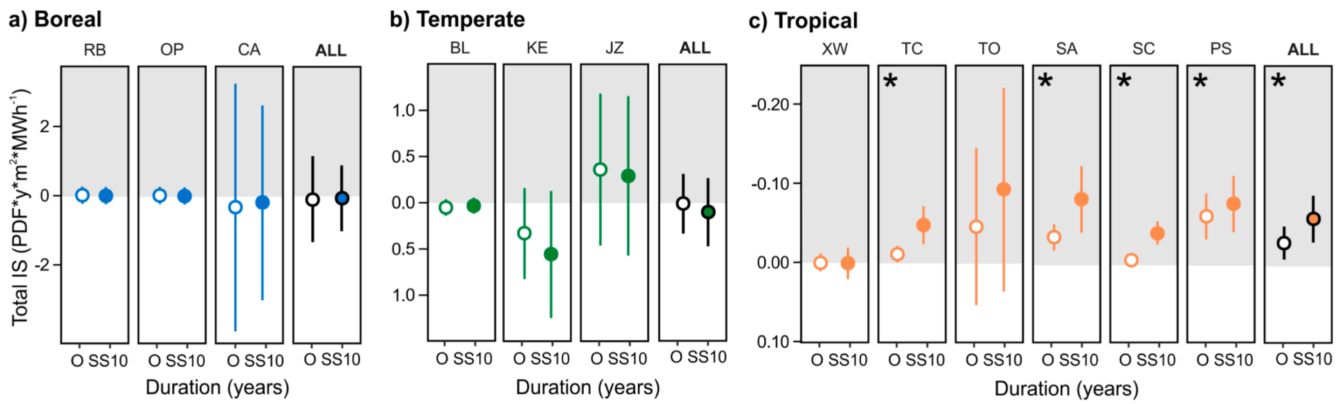

Fig. 4. Total Impact Score for hydropower a) boreal, b) temperate and c) tropical reservoirs for the Observed duration of the study (O) and for the steady state scenario of 10 y (SS10) at the reservoir and biome (ALL) scales. RB = Robert-Bourassa, OP = Opinaca, CA = Caniapiscau, BL = Beaver Lake, KE = Kenney, JZ = Jeziorsko, XW = Xiaowan, TC = Tucurui, TO = Tocantins, SA = Samuel, SC = Salto Caxias, PS = Petit-Saut. The star in each panel indicates a statistically significant IS.

economic, environmental and social costs (Fearnside, 2016; Kahn et al., 2014). Using our developed indicators in the LCA framework, accounting for potential impacts of hydropower on aquatic ecosystems biodiversity, could help in this respect.

4.2. First empirically derived indicators in LCA

Apart from few unpublished attempts (Humbert and Maendly, 2008), this contribution is the first to empirically and directly address the impacts of reservoir creation (change in land use) and hydropower generation on one aspect of biodiversity in LCA. Recent methods and contributions in LCA indirectly addressed the impact of water shortages or water consumption (sometimes used as a surrogate for hydropower) on biodiversity using Species-Discharge Relationships (SDR; Dorber et al., 2019b; Hanafiah et al., 2011; Tendall et al., 2014) or Species-Area Relationships (SAR; de Baan et al., 2013; Verones et al., 2013). It is quite risky to relate potential change in water discharge attributed to water consumption to change in species richness using SDR because these curves reflect evolutionary and ecological outcomes roughly in equilibrium with natural discharge (Rosenberg et al., 2000; Xenopoulos and Lodge, 2006). Data limitations to build SDR curves are severe, especially for change in biodiversity. Species richness numbers are not readily available for most rivers of the world, and temporal sequences spanning changes in discharge are extremely rare. Data limitations thus make difficult any rigorous tests of SDR or SAR models (Xenopoulos and Lodge, 2006). Moreover, we still do not know the impact pathways and the main drivers of potential changes in biodiversity in reservoirs and regulated rivers. The impacts of damming a river go well beyond changes in water discharge. Dams and reservoirs drastically change the hydrological regime and the riverscape connectivity and may change the strength of trophic interactions upstream and downstream of the dam (Gracey and Verones, 2016; Renöfalt et al., 2010; Turgeon et al., 2019b, 2019a). These alterations may be more important than a change in discharge in affecting change in richness. Unless the impact pathway is convincingly understood, or SDR strongly validated with empirical data of true impacts, we must be extremely careful in choosing fate and effects factors in LCA.

4.3. Importance of temporal and spatial scaling in LCA

Great insights are achieved when multiple spatial and temporal scales are considered and/or compared because patterns observed at one scale are often not transferable to another scale (upscaling, downscaling issues; Levin, 1992). In this study, the calculation of CF was strongly sensitive to the duration of the study but not so much to the spatial scale examined (*i.e.*, sampling station, reservoir and biome). We assumed a linear rate of change in richness over time since impoundment because of data limitation to test for non-linear patterns (see the GAM analysis results; Database A). This assumption would not be too problematic if the duration of the study was long enough to reach the steady-state convincingly (*i.e.*, new species assemblage equilibrium, where the change in richness stabilizes after impoundment; Fig. 1) or if the duration of study was comparable across studies. However, the observed duration of the studies varied greatly (from only one year after impoundment, to 54 years after impoundment; Database A) and the steady-state was likely not reached in many reservoirs, especially in the tropics where time series were shorter. This implies that CFs and ISs developed from short-duration studies will be underestimated (Fig. 1; R_1 vs. R_2 resulting in two ΔRs). This pattern will be exacerbated if the relationship is non-linear (sigmoid, a rise and fall, or a non-linear accelerating decreasing rate; Fig. 1; R_4 vs. R_2; Agostinho et al., 1994; Lima et al., 2016). CFs and ISs could also be overestimated if the relationship is non-linear at a decelerating decreasing rate (Fig. 1; R_3 vs. R_1). We also do not have the data to test if the time it takes to reach the steady-state is similar across latitudes (*e.g.*, processes and patterns are suggested to be faster in the tropics than in boreal regions; Monaghan et al., 2020; Turgeon et al., 2016). To circumvent these problems, and to compare CFs and ISs across studies, we tested the sensitivity of different scenarios of time until reaching the steady-state and concluded that using 10y after impoundment for all studies was the most plausible scenario. We demonstrated that the impacts changed in magnitude depending on the duration of the studies and a standardization must be considered in LCA.

Some patterns observed in upstream stations were not corroborated by patterns observed in downstream stations, suggesting that potentially different impact pathways affect the fish community upstream and downstream of the dam. The impacts upstream of the dams might be more closely related to the transformation of a lotic (river characteristics) into a lentic environment (lake characteristics) and to water levels fluctuations (*e.g.*, drawdown in the reservoir; Carmignani and Roy, 2017; Paller, 1997). Whereas downstream impacts might be related to variation in water discharge and flow (hydropeaking, residual flow; Holzapfel et al., 2017; Tonolla et al., 2017), limited sediment transport (Ibàñez et al., 1996; Schmitt et al., 2019), and dam acting as a barrier to fish migration (Pelicice et al., 2015; Porto et al., 1999). In this study, we used the area affected (upstream and downstream of the dam), and we assumed that the extent of the impacts of damming the river was limited to the reservoir (upstream of the dam) or to 10 km downstream of the dam. We have very limited information on the spatial extent to which the impacts of impoundment can be detected in fish community. Some studies detected significant changes in fish community and richness after impoundment upstream of the reservoir (Araújo et al., 2013; Lima et al., 2016; Penczak and Kruk, 2005) and as far as 25 km downstream of the dam (de Mérona et al., 2005). However, the impacts on fish community are likely to be site-specific because they will depend on how the dam and reservoir are managed (*e.g.*, hydropeaking, magnitude of the drawdown, mitigation measures such as fish ladder) and the connectivity to tributaries to the impounded river. Some reservoirs were also part of a cascading complex (39% of the reservoirs; Database B). In this situation, some impacts can be additive but will depend on the distance between reservoirs in the complex. More studies are needed to determine the spatial and temporal extents, the impact pathways, and the factors contributing to changes in fish community upstream and downstream of the dam.

In this study, the observed empirical changes in richness from 89 sampling stations (upstream and downstream of the dam) were transferable to the 26 reservoirs studied and were also transferable, but to a lesser extent, to the biomes. Our spatial coverage is global, but the resolution (grain) of the CFs and ISs was coarse given the limited amount of empirical data available. As empirical data and evidences will accumulate, the next steps would be to refine the resolution of our indicators to the scale of major habitat types (MHTs) or freshwater ecoregions of the world (FEOW; Abell et al., 2008), and to consider other taxonomic groups (macroinvertebrates, aquatic and riparian vegetation).

4.4. Limitations of developed CFs using species richness

Even though experts agreed on using species richness as a good starting point to model biodiversity loss in LCA (Teixeira et al., 2016), the use of PDF can be problematic for several reasons. First, it is imprudent to interpret a pattern of increased species richness (or no change in richness) as an indication of no impact of hydropower on biodiversity, if the pattern results from an increase in non-native species (*i.e.*, not from the initial regional pool of species, including exotic). We used the change in fish richness but did not discriminate between native and non-native species because this information was not provided for all studies, but see Kuipers et al., (2019) for converting local PDF into global PDF accounting for threatened and endemic species. In boreal reservoirs, no non-native species have been observed, so the developed CFs are considered robust (Tereshchenko and Strel'nikov, 1997; Turgeon et al., 2019a). In temperate reservoirs, the observed increase in richness after impoundment in Beaver lake, Kenney and Texoma reservoirs (Figs. 2 and 3), was actually due to an increase in non-native species (Gido et al., 2000; Martinez et al., 1994; Rainwater and Houser, 1982). In tropical reservoirs, an increase in non-native species has also been observed in Itaipu, Manwan and Xiaowan reservoirs, all showing an increase in richness over time (Li et al., 2013; Lima et al., 2016; Xiaoyan et al., 2010). A companion study (Turgeon et al., 2019b), looking at a larger dataset and including reservoirs used for other purposes (*e.g.*, irrigation and flood control), found a gradient of impact on fish biodiversity from being minimal in boreal, intermediate in temperate and important in tropical reservoirs. Small CF and IS calculated in this study in temperate reservoirs may be underestimated and should thus be interpreted with caution. Future studies should look separately at the change in richness of native species and non-native species to estimate PDF and thus CFs.

Second, looking at PDF do not account for a potential change in fish assemblages (Potentially Affected Fraction of species; PAF in the LCA nomenclature), nor in species that are more vulnerable (endemic and/or threatened). In addition, many species are rare species, and they are not as easily detected by using the common selective fishing gear, which can underestimate richness if the sampling effort is insufficient (MacKenzie et al., 2005). The use of eDNA could be a promising approach to reduce bias related to species detectability (Rees et al., 2014). Several alternative indicators and models have been suggested and used to account for loss in biodiversity in LCA (*e.g.*, functional diversity and ecosystem scarcity; Souza et al., 2015) but data requirement is tremendous, species have different adaptive capacity in different regions of the world and will respond to impoundment differently. Most importantly we must deal with the incommensurable challenge of developing CFs locally or regionally but apply them globally with the same rigour and criteria.

Finally, our developed CFs and ISs evaluated the impacts of hydropower on fish richness in storage reservoirs on the aquatic affected area (*i.e.*, former riverbed and existing lakes that were transformed into reservoirs upstream of the dam, and the regulated river downstream of the dam) and not on the terrestrial area flooded following impoundment (Fig. A). On the flooded area, a simplistic assumption could consider a loss of 100% of the impounded terrestrial habitat and a gain of 100% aquatic habitat. The biodiversity impact on the flooded area is a relevant and timing issue, and some promising work has been done in this respect to model net land occupation of reservoir and the impact of land inundation on terrestrial biodiversity (Dorber et al., 2019a, 2018).

5. Conclusions

By using empirical data on the rate of change in fish richness over time, this study is the first to propose robust and empirically developed CF and IS (ISR; PDF·m^2·y and ISs; PDF·m^2·y / kWh) of the effects of hydropower on aquatic biodiversity to be used in LCA. Our results suggest that the impact of hydroelectricity production on fish richness is significant in tropical reservoirs, marginal in temperate and not significant in boreal reservoirs, which calls for regionalisation in LCA. A sensitivity analysis also demonstrated that the simulated CFs and ISs were sensitive to the time it takes to reach the steady-state for fish communities after impoundment. A 10 years time frame after impoundment was judged to be the most plausible scenario. Finally, CFs and ISs were relatively robust to upscaling and downscaling issues (*i.e.*, patterns were consistent in their directionality across sampling stations, reservoirs and biomes). Hydropower can be part of the solution to decarbonize our global economy but will come at substantially higher ecological cost to the tropics.

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Katrine Turgeon: Conceptualization, Methodology, Formal analysis, Data curation, Writing - original draft, Visualization, Funding acquisition. **Gabrielle Trottier:** Methodology, Writing - review & editing. **Christian Turpin:** Conceptualization, Methodology, Writing - review & editing, Funding acquisition. **Cécile Bulle:** Conceptualization, Methodology, Writing - review & editing, Supervision. **Manuele Margni:** Conceptualization, Methodology, Writing - review & editing, Supervision.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Reference Data

Data available in data at doi:10.6084/m9.figshare.13085111, doi:10.6084/m9.figshare.13085150.

Appendix A. Supplementary data

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