

1 Research Article

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3 **Empirical characterization factors assessing the effects of**
4 **hydroelectricity on fish richness across three large biomes**

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19 **Highlights**

20 ▪ This paper is the first to develop global and empirically based characterization factors of the
21 impact of hydroelectricity production on aquatic ecosystems biodiversity, to be used in LCA;

22 ▪ The impact of hydroelectricity production on fish species richness was significant in the
23 tropics, of smaller amplitude in temperate and minimal in boreal biome;

24 ▪ The impact of hydroelectricity production on fish richness was consistent across scales -
25 same directionality and statistical significance across sampling stations, reservoirs and
26 biomes;

27 ▪ The impact of hydroelectricity production on fish richness was sensitive to the duration of
28 the study, highlighting the need for a clear understanding of transient situations before
29 reaching steady states in LCA.

30 **Abstract**

31 Hydroelectricity is often presented as a clean, reliable, and renewable energy source, but is also
32 recognized for its potential impacts on aquatic ecosystem biodiversity. We used empirical data
33 on change in fish species richness following impoundment to develop Characterisation Factors
34 (CF) and Impact Scores (IS) for hydroelectricity production for use in Life Cycle Assessment
35 (LCA). We used data collected on 89 sampling stations (63 upstream and 26 downstream of a
36 dam) belonging to 27 reservoirs from three biomes (boreal, temperate and tropical). Overall, the
37 impact of hydroelectricity production on fish species richness was significant in the tropics, of
38 smaller amplitude in temperate and minimal in boreal biome, stressing for the need of
39 regionalisation. The impact of hydroelectricity production was also quite consistent across scales
40 (*i.e.*, same directionality and statistical significance across sampling stations, reservoirs and
41 biomes) but was sensitive to the duration of the study (*i.e.*, the period over which data have been
42 collected after impoundment), highlighting the need for a clear understanding of transient
43 situations before reaching steady states. Our CFs and ISs contribute to fill a gap to assist decision
44 makers using LCA to evaluate alternative technologies, such as hydropower, to decarbonize the
45 worldwide economy.

46

47 **Keywords:**

48 Life cycle assessment; Hydro-electricity; Biodiversity; Fish; Richness; Biomes

49 **1. Introduction**

50 One of the most important challenge we face as a society is the increased demand for
51 energy (SEforALL, 2016, p. 4). In response to this worldwide demand, hydroelectricity is
52 presented as a relatively clean, reliable, and renewable energy source (Tahseen and Karney,
53 2017; Teodoru et al., 2012), and an interesting option to decarbonise our global economy
54 (Figueres et al., 2017; Potvin et al., 2017) by reducing greenhouse gas emissions (GHGs).
55 Hydroelectricity supplies less than 3% of the primary energy worldwide but more than 70% of
56 the world's renewable electricity (International Energy Agency, 2017; World Energy Council,
57 2016). These numbers will increase in the coming years as many large dams are being
58 constructed around the world, particularly in developing economies that are mostly located in the
59 tropics (Grill et al., 2015; Winemiller et al., 2016).

60 Despite its recognized advantages, the production of hydroelectricity can impact aquatic
61 ecosystem functions and biodiversity through the regulation of the river flow, by a drastic
62 changes in the hydrological regime, and by the fragmentation of rivers (Gracey and Verones,
63 2016; Renöfalt et al., 2010; Rosenberg et al., 2000). Dams constructed for hydroelectricity
64 production, transform large rivers (*i.e.*, lotic environment) and surrounding lakes into larges
65 reservoirs (*i.e.*, lentic environment), or a series of reservoirs (*sensu* cascade reservoirs; (Friedl
66 and Wüest, 2002; Haxton and Findlay, 2009). Upstream of the dam, reservoirs can experience
67 variation in water levels outside of their natural amplitudes (Kroger, 1973; Zohary and
68 Ostrovsky, 2011). Downstream of the dam, changes in seasonal and inter-annual streamflow
69 magnitude and variability are generally reduced (Friedman et al., 1998; Graf, 2006) and fish
70 movement can be altered by the dam. These modifications can impact the biodiversity,

71 abundance, distribution and community structure of many taxa of the aquatic food web (Furey et
72 al., 2006; Nilsson and Berggren, 2000; Vörösmarty et al., 2010).

73 Life cycle assessment (LCA) is used to assess the environmental impacts of products and
74 services throughout their whole life cycle (*i.e.*, cradle-to-grave; Finnveden et al., 2009; ISO,
75 2006). LCA informs about environmentally sound choices in the context of decision-making and
76 is based on scientific evidence. When compared to other electricity production technologies,
77 hydroelectricity scored favorably in LCA studies regarding GHG emissions, air pollution, health
78 risk, acidification and eutrophication of ecosystems (CIRAIG, 2014; Hertwich, 2013; Sathaye et
79 al., 2011). However, some of the impacts of hydroelectricity production on ecosystems and
80 biodiversity are still not successfully integrated into LCA and are underrepresented due to some
81 methodological challenges (de Baan et al., 2013; Gracey and Verones, 2016).

82 Evaluating and including the impacts of hydroelectricity production on aquatic
83 ecosystems quality in LCA has been proven to be challenging because of the large data
84 requirement, unclear causational effects, incomplete coverage of biodiversity impacts, and
85 spatial and temporal scaling issues that can hinder its global application and validity (Gracey and
86 Verones, 2016; McManamay et al., 2015; Milà i Canals et al., 2009; Teixeira et al., 2016).

87 Different indicators have been proposed to measure the impacts on ecosystems and biodiversity
88 in LCA (*e.g.*, difference in species richness, *i.e.*, the number of species, ecosystem scarcity and
89 vulnerability, functional diversity; (Curran et al., 2011; Souza et al., 2013). But experts
90 concluded – without a clear consensus – that change in species richness is a good and simple
91 starting point to assess biodiversity impacts (Teixeira et al., 2016).

92 When change in species richness is used in LCA, it is essential to adequately consider the
93 right spatial and temporal scale of impacts. Patterns observed locally (*e.g.*, in a reservoir) cannot

94 always be extrapolated within or across regions. It is also important to evaluate the impact at the
95 steady state, *i.e.*, at the time at which change in biodiversity stabilize after impoundment. Very
96 few studies examined global impacts of hydroelectricity on ecosystems quality, or examined if
97 patterns can be extrapolated across scales (but see (de Baan et al., 2013) for a multiple spatial
98 scale study), and no study yet use empirically derived Characterization Factor (CF) and Impact
99 Score (IS).

100 Here, we used empirically derived rate of change in fish species richness over time,
101 across 89 sampling stations, belonging to 27 storage reservoirs from boreal, temperate and
102 tropical biomes. The focus of this study is on storage reservoirs because of a lack of adequate
103 longitudinal data (data before and after damming) from the other technologies (*e.g.*, run of the
104 river and pumping stations). Our goals were to: 1) develop robust empirical CFs across three
105 spatial scales (sampling station, reservoir and biome), 2) calculate the impact score of the
106 creation of a reservoir (ISR) and of hydroelectricity production (IS) across scales, and 3) to test
107 the need for regionalisation by examining if the observed patterns were consistent across biomes.

108 **2. Materials and Methods**

109 **2.1 General approach**

110 The approach to generate Characterization factors (CF) and Impact Scores (IS) was based
111 on the examination of empirical patterns of changes in fish richness in response to river
112 impoundment across three large biomes (boreal, temperate and tropical) from an extensive
113 literature search. To calculate CF, we used the Potentially Disappeared Fraction of species (PDF)
114 as the unit to express change in richness in response to hydroelectricity production. This unit has
115 the advantage to be compatible with other damage oriented impact assessment methods
116 addressing ecosystems quality such as IMPACT 2002+ (Jolliet et al., 2003) and Impact World +

117 (Bulle et al., 2019) has been recommended by the UNEP-SETAC Life Cycle Initiative as an
118 adequate and consistent biodiversity attribute (Verones et al., 2017). Because no reliable data
119 were available to evaluate the biodiversity recovery when powerplants are decommissioned and
120 dam removed, we were not able to address the recovery phase in the LCA and therefore focused
121 our effort on the impacts during the occupation phase (*i.e.*, time span covering the construction
122 of the dam until complete decommission; Fig. 1). For each reservoir, we calculated two impact
123 scores: one for the reservoir creation and construction of the dam (ISR; where CFs were
124 multiplied by the affected area) and one for the hydroelectricity production (IS; where ISRs were
125 divided by the annual kWh produced for a given powerplant). We also took a multi-scale
126 approach to examine if patterns observed at the sampling station, reservoir and biome scale were
127 comparable.

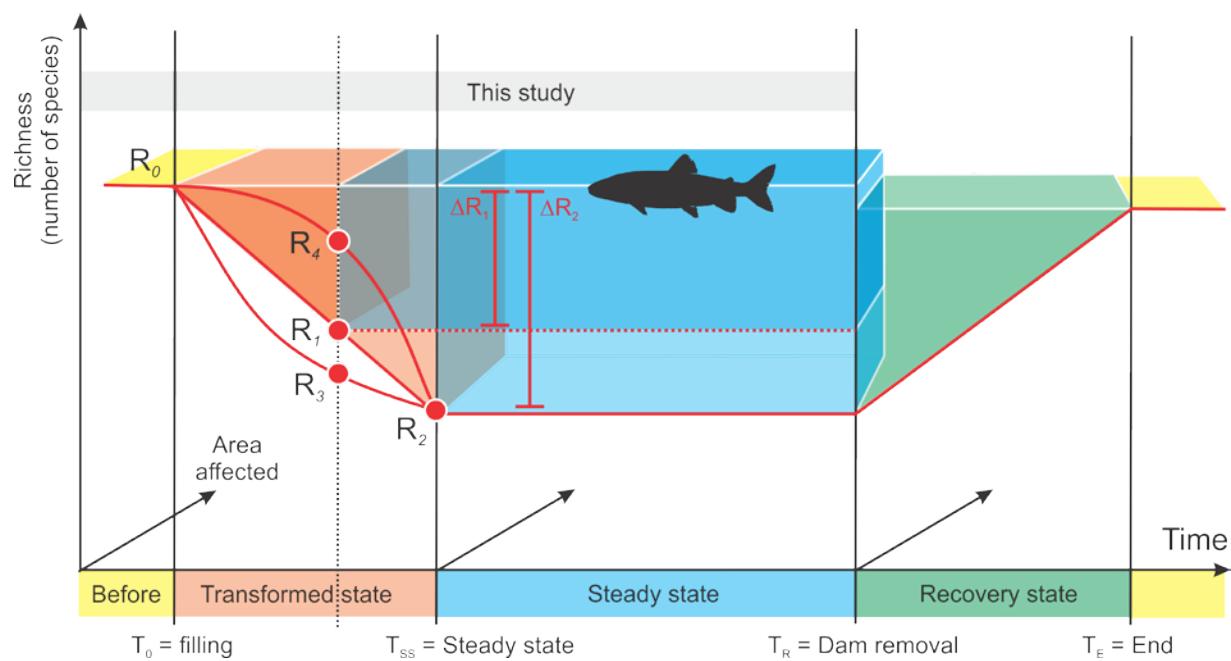


Figure 1. Schematic representation of an area-time framework representing the rate of change in richness experienced in a given reservoir. R_0 represents the richness before impoundment, R_{is} represents different richness during the transformed state of the reservoir and where the fish community respond to environmental change following impoundment. The ΔQs 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.

128 **2.2 Richness data extraction and literature search**

129 The literature search for this paper has been performed previously for another companion
130 meta-analysis examining the global effect of dam on fish biodiversity (Turgeon et al., 2019b). In
131 a nutshell, the search resulted in 668 publications (mostly peer-reviewed articles). For this paper,
132 we excluded modelling and simulation exercises, and we refined our selection criteria to include
133 only references that had unbiased quantitative data on the fish community before and after
134 impoundment, and where the main purpose of the dam was to produce hydroelectricity. Data are
135 limited to storage reservoirs technology and thus does not include run of the river and pumping
136 station technologies due to a lack of longitudinal data. A total of 30 references met our selection
137 criteria (Database A). See Turgeon et al. (2019b) for a detailed methodology about the literature
138 search, and data extraction.

139 **2.3 Extracting the area affected by the dam and reservoir**

140 To extract the area affected by the construction of the dam and reservoir, we extracted the
141 area occupied by the rivers and lakes prior to impoundment (hereafter called the affected area)
142 both upstream and downstream of the dam (Fig. A). Change in land use from terrestrial to
143 reservoir (inundated land area; ILA) is out of the scope of this paper, but see (Dorber et al.,
144 2018) for a proposal to model net land occupation of hydropower reservoirs in LCA. To get the
145 affected area information, we used various sources. For recent reservoirs, we used Google Earth
146 Pro with the historical satellite imagery tool (Landsat/Copernicus images). Other sources of
147 historical maps consisted in the USGS historical topographic maps for most of the United States
148 reservoirs (<https://viewer.nationalmap.gov/basic/>), the Old Maps Online website for old
149 reservoirs in Africa and South America (<http://www.oldmapsonline.org/>). The image of the river
150 bed before impoundment was exported as a raster image in QGIS (v.2.18.16;
151 <http://www.qgis.org>). The affected area was hand drawn as a polygon in a vector layer, and the

152 total area of the polygon was extracted. Two polygons were extracted per reservoir, one
153 upstream and one downstream of the dam. Upstream, we assumed that the impacts of the
154 reservoir and the dam on fish community did not go beyond the impounded area and thus, used
155 the upper end of the reservoir as the upper limit of the affected area. For downstream stations, we
156 used 10 km downstream of the dam to set the lower limit of the polygon. We tested for the effect
157 of different distances downstream of the dam (5, 10, 15, 25 km), in addition to the distance at
158 which data were collected (mean \pm SD; 13 km \pm 45 km; median; 0.35 km) and they were all
159 strongly correlated (Pearson $r > 0.80$; *unpublished analysis*).

160 **2.4 Calculation of change in richness**

161 **2.4.1 Sampling station scale:** For each sampling station i , located either upstream or
162 downstream of the dam in reservoir j , we calculated the rate of change in richness over time with
163 a linear regression. The rate of change in richness was extracted using the estimated slope of the
164 regression between richness and time (Equation 1) and we used the standard error of the estimate
165 to calculate the 95% confidence interval (CI; see Database A). In this study, we assumed a linear
166 relationship between richness and time, but some studies empirically observed a rise and fall of
167 richness over time (Agostinho et al., 1994; Lima et al., 2016). See discussion for potential
168 limitations and biased interpretation associated with this assumption.

169 **2.4.2 Reservoir and biome scales:** When more than one station were sampled per reservoir,
170 we used general linear mixed effect model (glmm; lmer function in lme4 package v.1.1-13;
171 (Bates et al., 2018) to calculate the rate of change in richness over time, separately for upstream
172 and downstream stations. At the reservoir scale, we controlled for temporal non-independence of
173 the data by using sampling station as a random factor. At the biome scale, we controlled for

174 spatial and temporal non-independence of the data by nesting each sampling station i into their
175 respective reservoir j . All analyses were performed using R v. 3.3.2 (R Core Team, 2017).

176 **2.5 Calculation of Characterization Factors (CF)**

177 **2.5.1 Sampling station scale:** To calculate CFs, we multiplied the observed rate of change in
178 richness ($\Delta R/\Delta t$; where ΔR stands the difference in richness and Δt stands for the duration of the
179 study) by the time it take to reach a defined steady state t_{ss} (time horizon at which we considered
180 that the rate of change in richness = 0; see Fig. 1) as per Equation 1, and divided the result by the
181 average richness observed before impoundment for a given sampling station (R_{0ij}). We did this
182 for each sampling station i in reservoir j . The duration at which fish richness has been sampled
183 for a given study (Δt) varied greatly across studies and biomes (e.g., from less than five years to
184 40 years, see Database A). This can be problematic when comparing short duration studies with
185 longer ones, because the longer the time after impoundment (Δt), the bigger the ΔR will be,
186 which can result in an underestimation of PDFs (Fig. 1; see R_1 vs. R_2). To make studies
187 comparable in their steady state, we tested with a sensitivity analysis, different scenarios of time
188 to reach the steady state ($t_{ss} = 5, 10, 20, 25$ and 30 years after impoundment; Equation 1). To
189 calculate the uncertainty associated with the CFs, we used the standard error (SE) from the
190 estimate of the rate of change in richness (from the glmm) and multiplied it by the different
191 scenario of time to reach the steady state and then divided it by the average richness observed
192 before impoundment. From this scaled SE, we calculated the 95% CI.

193 **Equation 1: Characterisation factors at the sampling station scale**

194
$$CF_{ij} = \frac{\left(\frac{\Delta R_{ij}}{\Delta t_{ij}} \right) * t_{ss}}{R_{0ij}}$$

195
$$= \frac{R_{0,ij} - R_{ss,ij}}{R_{0,ij}}$$

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

200 **2.5.2 Reservoir and biome scales:** To test if the CFs are valid and robust across scales
201 (sampling station, reservoir and biome), we also computed CFs at the reservoir and biomes
202 scales. At the reservoir scale, we calculated a mean upstream CF, a mean downstream CF, as
203 well as a mean CF (upstream CF + downstream CF when upstream and downstream stations
204 were available). To do so, we averaged the CFs calculated for upstream sampling stations in
205 reservoir j . We then squared the SE associated with the coefficient the regression (slope of the
206 observed change in richness for a given period) for each upstream sampling station of reservoir j
207 added them together to get the total variance for reservoir j . We then divided this variance by the
208 number of sampling stations in reservoir j raised to the power of 2, and square rooted that
209 variance to get the SE of the mean CF, and we calculated the 95% CI. We did the same
210 procedure for downstream stations and for the biome scale. At the biome scale, we used CF_j as
211 units (calculated at the reservoir scale) instead of CF_i (calculated at the sampling station scale).

212 **2.6 Calculation of impact scores (ISR and IS)**

213 We were also interested to evaluate the potential environment impact of creating a
214 reservoir (ISR; elementary flow = area affected upstream and downstream of the dam) and of
215 producing hydroelectricity (IS; elementary flow = kWh produced for a given reservoir). To do
216 so, we multiplied the CF by the area affected by the reservoir and the dam (*i.e.*, area occupied by

217 the rivers and lakes prior to impoundment, see Fig. A). Because we expect different impacts
218 upstream and downstream of the dam, we calculated the impact score of creating and operating a
219 reservoir (ISR; CF m² yr) as the sum of downstream (ISR_{uj}) and upstream impacts (ISR_{uj}; Fig.
220 A). Impact scores were calculated at the reservoir and biome scales.

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

222

$$\begin{aligned} ISR_j &= (ISR_{uj} + ISR_{dj}) \\ 223 &= (CF_{uj} * A_{uj} + CF_{dj} * A_{dj}) * 1 \text{ year} \end{aligned}$$

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

226 To calculate the impact score per unit of hydroelectricity produced by a powerplant
227 associated to a given reservoir, the impact score ISR_j was divided by the annual electricity
228 production, P_j (kWh/year).

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

230

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

231 **2.7 Sensitivity analysis of the Steady State Scenarios (SS)**

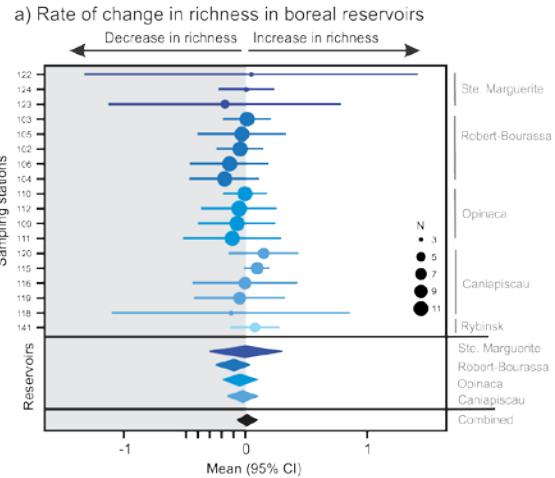
232 Because we simulated CF and IS for different steady state scenarios (5, 10, 20, 25 and 30
233 y after impoundment) we evaluated the sensitivity of CF for each incremental increase in the
234 time to steady-state (tss).

235 **3. Results**

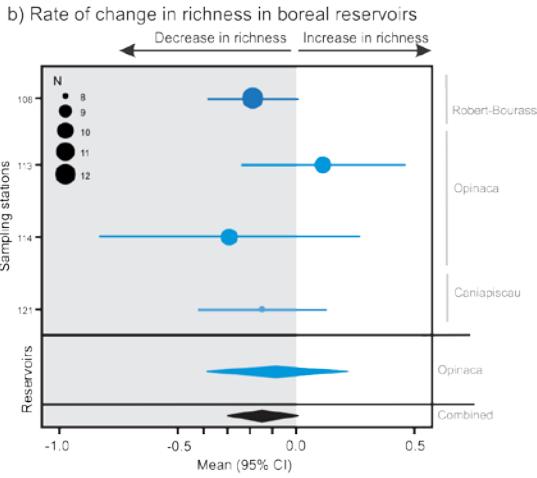
236 ***3.1 Rate of change in fish richness across scales and biomes***

237 Upstream and downstream of the dam, the rate of change in fish richness over time varied
238 strongly across sampling stations, reservoirs and biomes (Fig. 2). When all biomes, reservoirs
239 and sampling stations were combined, richness significantly decreased over time at a rate of 0.29
240 species per year upstream of the dam (estimate \pm SD = -0.293 ± 0.074 , 95% CI = -0.439 to -
241 0.148) and at a comparable rate downstream of the dam (0.26 species per year; estimate \pm SD = -
242 0.264 ± 0.082 , 95% CI = -0.424 to -0.104). In the boreal biome, there was no significant change
243 in richness over time at all scales (sampling station, reservoirs and biome) and for both upstream
244 and downstream stations (95% CI overlapped with zero; Fig. 2 a, b). In temperate and tropical
245 regions, we observed a significant decrease in richness over time at the biome scale for upstream
246 stations (loss of 0.26 and 1.6 species per year respectively; Fig. 2 c, e). Downstream of the dam,
247 we observed a significant decrease in richness in the temperate region (loss of 0.34 species per
248 year) but not in the tropics (Fig. 2 d, f). In these two biomes, some sampling stations and
249 reservoirs showed either a significant decrease or, interestingly, an increase in richness over time
250 following impoundment (Fig. 2 c-f).

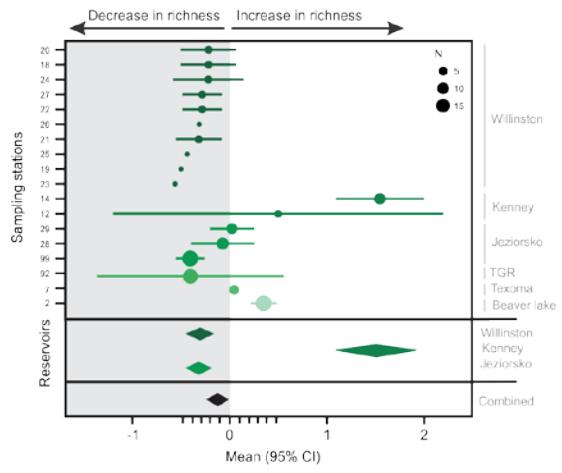
Upstream



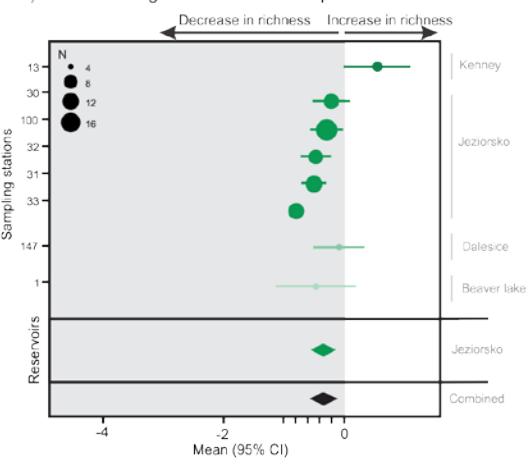
Downstream



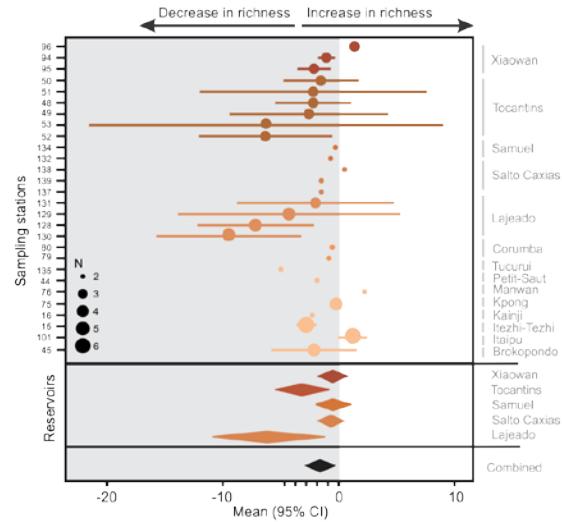
c) Rate of change in richness in temperate reservoirs



d) Rate of change in richness in temperate reservoirs



e) Rate of change in richness in tropical reservoirs



f) Rate of change in richness in tropical reservoirs

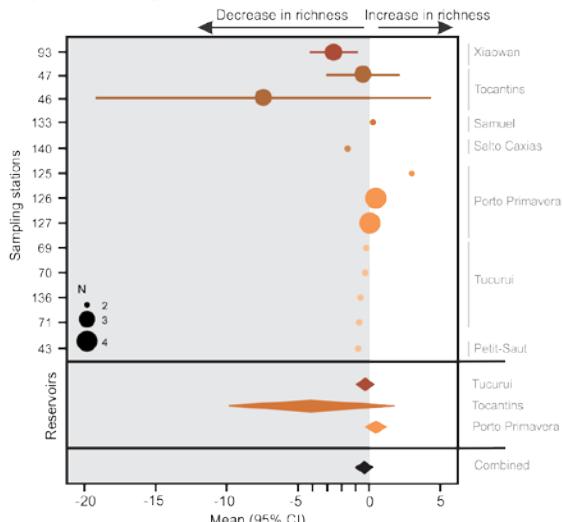


Figure 2. Empirically derived rates of change in richness and their 95%CI in upstream and downstream sampling stations, and reservoirs from three biomes: boreal (23 sampling stations, 5 reservoirs), temperate (26 sampling stations, 7 reservoirs) and tropical reservoirs (41 sampling stations, 15 reservoirs). The size of the circles represents the number of observations in the time series used to derive the rate of change in richness.

251 3.2 *Characterization factors (CF)*

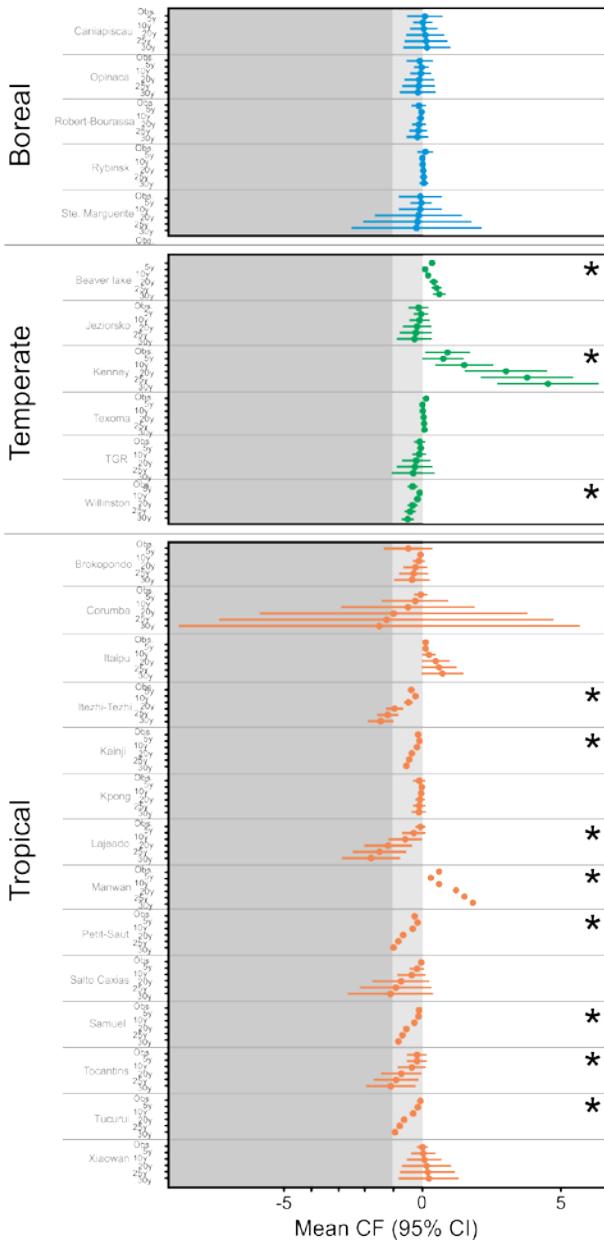
252 The magnitude of the impact and statistical significance of CFs were sensitive to the
253 assumption of reaching the steady-state (tss), differed across biomes, but were consistent across
254 scales and position (downstream or upstream of the dam; Fig. 3, Figs. B.1, B.2 and B.3). In
255 boreal ecosystems, there was no significant loss of species upstream and downstream of the dam
256 at the sampling station scale and for all steady state scenarios (Fig. B.1). When data were
257 combined at the reservoir scale, no loss of species was observed upstream (Fig. 3 a), and a
258 marginal loss of species was observed in one reservoir downstream of the dam (Fig. 3 b, Table
259 1). In temperate and tropical ecosystems, there were some significant gains and losses of species
260 upstream (Fig. B.2a and Fig. B.3a) and downstream of the dam at the sampling station scale (Fig.
261 B.2b and Fig. B.3b). When data were combined at the reservoir scale for temperate and tropical
262 ecosystems, we also observed gains and losses of species (Fig. 3, Table 1).

263 Sensitivity analysis suggested that simulated CFs for steady state scenario reached 15y
264 after impoundment and beyond were unlikely because many reservoirs lost 100% of the original
265 richness which was never been observed in any reservoirs (Fig. 3, Fig. C). Steady state scenario
266 reached at 5y underestimated species loss when compared to the observed duration (Fig. 3, Fig.
267 C). For these reasons, a steady state scenario reached at 10y will be used to calculate impact
268 scores, and to compare the impact of impoundment across biomes and reservoirs.

269 **Table 1.** Estimates \pm Standard Error (SE) for Characterization factors (CF), Impact scores for the
 270 creation of the reservoir (ISR) and impact scores to produce hydroelectricity (IS) at the reservoir
 271 and biome scales.

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

a) CF at the reservoir scale (Upstream)



b) CF at the reservoir scale (Downstream)

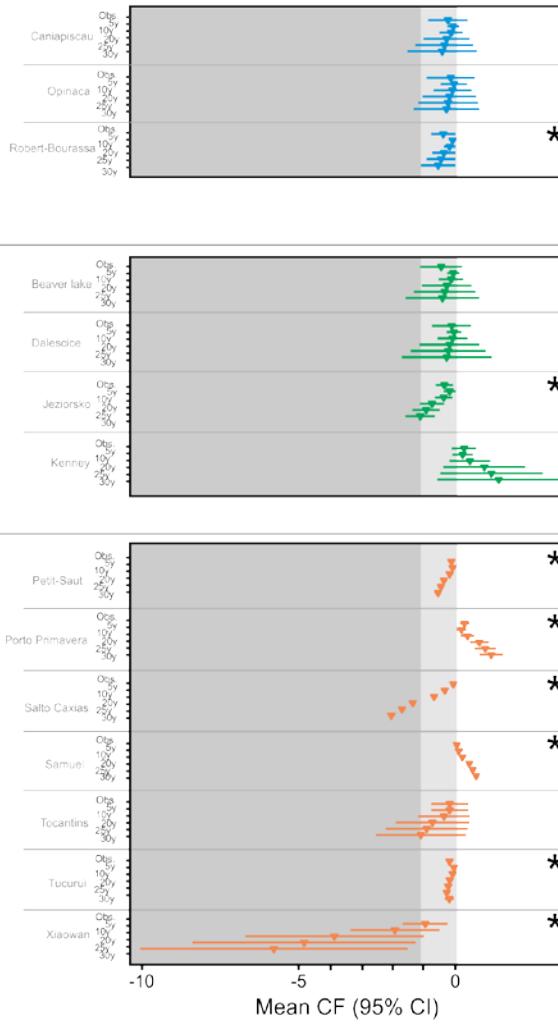


Figure 3. Characterization factor estimates (CFs) and their 95% CI at the reservoir scale for three biomes, a) upstream and b) downstream of the dam and for the observed duration of the study, and the 5 simulated steady state scenarios (5y, 10y, 20y, 25y and 30y). A negative value represents a loss of species and a positive value a gain in species. CF values in the dark grey area means that 100% of the species were loss. Stars beside the CF values indicate a statistically significant CF.

272

273 3.3 Impact scores for the creation of the reservoir and for hydroelectricity production

274 Impact scores for the creation of reservoirs (ISR) and for hydroelectricity production (IS)

275 differed across biomes and reservoirs (Fig. 4, Table 1). ISRs in boreal and temperate regions

276 were not significant for the observed duration of the study (O; Fig. 4 a) and for the steady state

277 scenario of 10y (SS10; Fig. 4 a). However, three tropical reservoirs showed a significant ISR
 278 when using the SS10 (Fig. 4 a). These results translated into an ISR of 0 for boreal and temperate
 279 biomes, and a significant ISR for the tropics (Fig. 4 a, ALL). The directionality and significance
 280 of IS were comparable to ISR for both the reservoirs and biome scales (Fig. 4 b, Table 1).

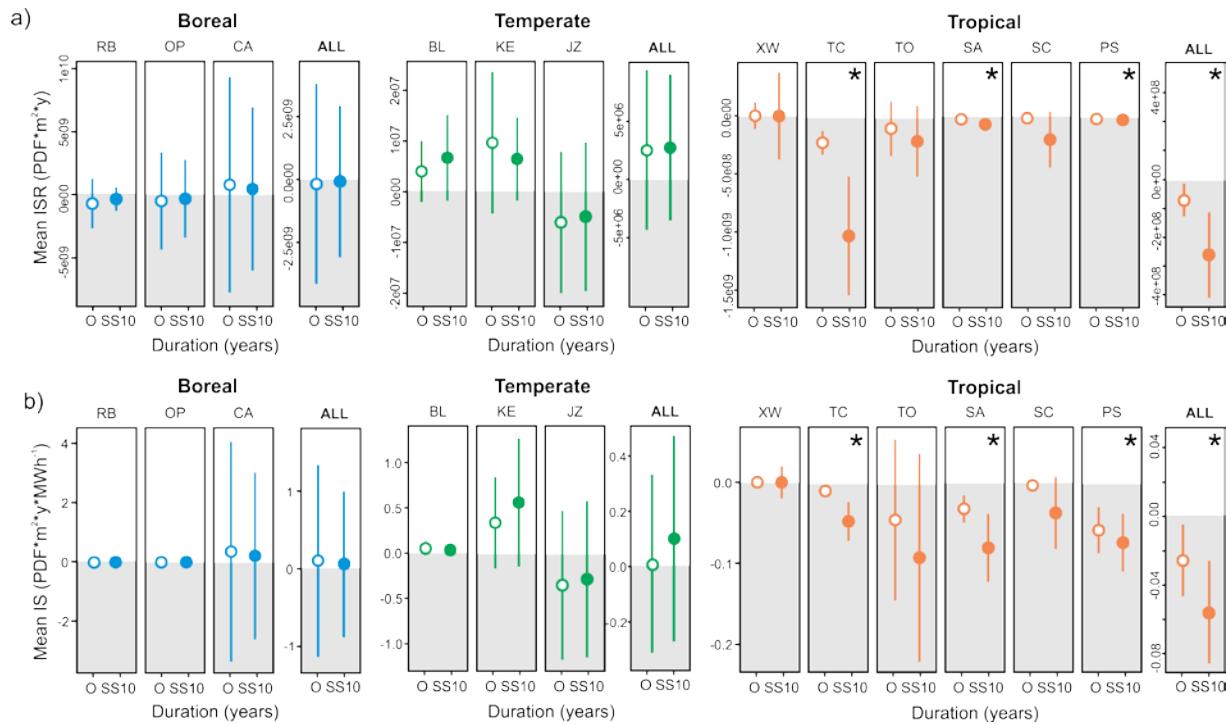


Figure 4. a) Mean reservoir and biome (ALL) impact score for the creation of the reservoir (ISR) and b) mean reservoir and biome (ALL) impact score of hydroelectricity production for the observed duration of the study and for the steady state scenario of 10 y in the three biomes. RB = Robert-Bourassa, 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. Stars beside the CF values indicate a statistically significant CF.

281 **4. Discussion**

282 **4.1 Regionalisation is needed**

283 Based on available empirical data (89 sampling stations located upstream and
 284 downstream of the dam and belonging to 27 reservoirs across three large biomes), we
 285 demonstrated that regionalization is needed for this impact category in LCA because the
 286 observed rate of change in fish richness in hydroelectric reservoirs varied across biomes, being

287 minimal in boreal, marginal in temperate ecosystems, and significant in the tropics. This result
288 suggests that hydroelectricity production in Northern countries located in the boreal biomes (e.g.,
289 Canada, Russia, Norway, Sweden, Finland, Iceland), which account for more than 15% of the
290 installed hydroelectricity production capacity in 2016 (International Energy Agency (IEA),
291 2016), has limited impacts on fish biodiversity. On the other hand, our dataset demonstrated that
292 hydroelectricity production in the tropics has significant impacts on fish biodiversity at all scales.
293 Rivers in species-rich tropical region located in Brazil (installed capacity of 91.7 GW, 85% of
294 the generated energy in Brazil, 8% globally) and China (installed capacity of 319 GW, 17% of
295 the generated energy in China, 27% globally), have been extensively harnessed for
296 hydroelectricity production (Stickler et al., 2013; Winemiller et al., 2016; Ziv et al., 2012).
297 Future hydroelectric development (planned and currently in construction) is concentrated in
298 China, the Mekong region, Latin America and Africa, and the largest potential for future
299 development is in Asia (International Energy Agency (IEA), 2016). All these regions have high
300 fish richness and endemic species, some of these regions are recognized as biodiversity hotspots,
301 and they will be particularly impacted by climate change regarding loss in water availability
302 (Xenopoulos and Lodge, 2006). In a collective effort to decarbonize the worldwide economy and
303 reduce GHG emissions, we urgently need appropriate supporting decision tools that consider
304 long term economic, environmental and social costs (Fearnside, 2016; Kahn et al., 2014). The
305 use of our developed CFs and ISs in LCA, accounting for potential impact of hydropower on
306 aquatic ecosystems biodiversity, could help in this respect.

307 ***4.2 First empirically derived CF and IS***

308 Apart from few unpublished attempts (Humbert and Maendly, 2008), this contribution is the
309 first to empirically address the impact of hydroelectricity production on biodiversity in LCA.
310 Recent methods and contributions in LCA addressed the impact of water shortages or

311 consumption on biodiversity using Species-Discharge relationships (SDR; (Hanafiah et al., 2011;
312 Tendall et al., 2014)) or Species-Area relationships (SAR; (de Baan et al., 2013; Verones et al.,
313 2013)) but none of these contributions addressed the time it takes to reach the steady-state
314 (Souza et al., 2015). It is also quite risky to relate potential change in water discharge to change
315 in species richness using SDR because these curves reflect evolutionary and ecological outcomes
316 roughly in equilibrium with natural discharge (Rosenberg et al., 2000; Xenopoulos and Lodge,
317 2006). Data limitations to build SDR curves are severe, especially for change in biodiversity.
318 Species richness numbers are not readily available for most rivers of the world, and temporal
319 sequences spanning changes in discharge are extremely rare. Data limitations thus make difficult
320 any rigorous tests of species–discharge models (Xenopoulos and Lodge, 2006). Moreover, we
321 still do not know the impact pathways and the main drivers of potential changes in biodiversity
322 in reservoirs and regulated rivers. The impacts of damming a river go well beyond changes in
323 water discharge. Dams and reservoirs drastically change the hydrological regime and the
324 riverscape connectivity and may change the strength of trophic interactions upstream and
325 downstream of the dam (Gracey and Verones, 2016; Renföfalt et al., 2010; Turgeon et al., 2019b,
326 2019a). These alterations may be much more important than change in discharge in affecting
327 change in richness. Unless the impact pathway is convincingly understood, or SDR strongly
328 validated with empirical data, we must be extremely careful in our choice of fate and effects
329 factors in LCA.

330 **4.3 Importance of temporal and spatial scaling in LCA**

331 Great insights are achieved when multiple spatial and temporal scales are considered and/or
332 compared because patterns observed at one scale are often not transferable to another scale
333 (upscaling, downscaling issues; (Levin, 1992). In this study, the calculation of the CFs and ISs
334 was strongly sensitive to the duration of the study but not to the spatial scale examined (*i.e.*,

335 sampling station, reservoir and biome). We assumed a linear rate of change in richness over time
336 since impoundment. This assumption would not be problematic if the duration of the study was
337 long enough to convincingly reach the steady state phase (*i.e.*, new species assemblage
338 equilibrium where the change in richness stabilizes after impoundment; Fig. 1) or if the duration
339 of study was comparable across studies. However, the observed duration of the studies varied
340 greatly (from only one year after impoundment, to 54 years after impoundment; Database S1)
341 and the steady state was likely not reached in many reservoirs, especially in the tropics. This
342 imply that CFs and ISSs developed from short duration study will be underestimated (see Fig. 1;
343 R_1 vs. R_2 resulting in two ΔQs). This pattern will be exacerbated if the relationship is non-linear
344 (sigmoid, a rise and fall, or a non-linear accelerating decreasing rate; Fig. 1; R_4 vs. R_2) which is
345 highly plausible (Agostinho et al., 1994; Lima et al., 2016). Most of the time series do not allow
346 to test for non-linearity because they were too short, or the time steps between sampling events
347 were too long. We also do not have the data to test if the time it takes to reach the steady state is
348 similar across latitudes (*e.g.*, might be faster in the tropics and slower in boreal regions). To
349 circumvent these problems, and to compare CFs and ISSs across studies, we tested the sensitivity
350 of different steady-state scenarios (5, 10, 20, 25 and 30y after impoundment) and assumed that
351 using 10y after impoundment for all studies was a plausible scenario. We demonstrated that the
352 impacts changed in magnitude depending on the duration of the studies and a standardization
353 must be considered in LCA.

354 Some patterns observed in upstream stations were not corroborated by patterns observed in
355 downstream stations suggesting that potentially different impact pathways affect the fish
356 community upstream and downstream of the dam. The impacts upstream of the dams might be
357 more closely related to the transformation of a lotic (river characteristics) into a lentic (lake

358 characteristics) environment and to water levels fluctuations, whereas downstream impacts might
359 be related to variation in water discharge (hydropeaking or not), and the dam acting as a barrier
360 to fish migration/movement. In this study, we assumed that the extent of the impacts of damming
361 the river was limited to the reservoir (upstream of the dam) or to 10 km downstream of the dam.
362 We have very limited information on the extent to which the impacts of impoundment can be
363 detected on fish community. Some studies detected significant changes in fish community and
364 richness after impoundment upstream of the reservoir (Araújo et al., 2013; Lima et al., 2016;
365 Penczak and Kruk, 2005) and as far as 25 km downstream of the dam (de Mérona et al., 2005).
366 However, the impacts on fish community upstream of the reservoirs and downstream of the dam
367 is probably site-specific because they will depend on how the dam is managed (*e.g.*,
368 hydropeaking or not) the and the connectivity to tributaries. More studies are needed to
369 determine the spatial extent, the impact pathways, and the factors contributing to changes in fish
370 community when damming a river, upstream and downstream of the dam.

371 In this study, the observed empirical changes in richness from 89 sampling stations
372 (upstream and downstream of the dam) were transferable to the reservoirs studied and were also
373 transferable, but to a lesser extent, to the biomes. Our spatial coverage is thus global but the
374 resolution (grain) of the CFs and ISs was coarse given the limited amount of empirical data. As
375 empirical data and evidence will accumulate, the next step would be to refine the resolution at
376 the scale of major habitat types (MHTs) or freshwater ecoregions of the world (FEOW; Abell et
377 al., 2008) and to consider other taxa (macroinvertebrates, aquatic and riparian vegetation).

378 ***4.4 Limitations of developed CFs and ISs***

379 Even though experts agreed on using species richness as a good starting point to model
380 biodiversity loss in LCA (Teixeira et al., 2016), the use of Potentially Disappeared Fraction of

381 species (PDF) is problematic for several reasons. First, it is imprudent to interpret a pattern of
382 increased species richness (or no change in richness) as an indication of no impact of
383 hydroelectricity production on biodiversity, if the pattern results from an increase in non-native
384 species (*i.e.*, not from the initial regional pool of species, including exotic). We used change in
385 fish richness but did not discriminate between native and non-native species because this
386 information was not provided for all studies. In boreal reservoirs, no non-native species have
387 been observed so the developed CFs and ISs are considered robust (Tereshchenko and
388 Strel'nikov, 1997; Turgeon et al., 2019a). In temperate reservoirs, the observed increase in
389 richness after impoundment in Beaver lake, Kenney and Texoma reservoirs (Figs 2, 3 and 4),
390 was actually due to an increase in non-native species (Gido et al., 2000; Martinez et al., 1994;
391 Rainwater and Houser, 1982). In tropical reservoirs, an increase in non-native species have also
392 been observed in Itaipu, Manwan and Xiaowan reservoirs, all showing an increase in richness
393 over time (Li et al., 2013; Lima et al., 2016; Xiaoyan et al., 2010). A companion study (Turgeon
394 et al., 2019b), looking at a larger dataset and including reservoirs used for other purposes (*e.g.*,
395 irrigation, flood control), found a gradient of impact on fish biodiversity from being minimal in
396 boreal, intermediate in temperate and important in tropical reservoirs. The small CFs and ISs in
397 temperate reservoirs may be underestimated and should thus be interpreted with caution. Future
398 studies should look at the fate of both native species and non-native species to develop the CFs
399 and ISs.

400 Second, looking at PDF do not account for potential change in fish assemblages
401 (potentially affected fraction of species; PAF) or in species that are more vulnerable (endemic
402 and/or threatened). Several alternatives indices and models have been suggested and used to
403 account for loss in biodiversity in LCA (*e.g.*, functional diversity, ecosystem scarcity) (Souza et
404 al., 2015) but data requirement is tremendous, species have different adaptive capacity in

405 different regions of the world and will respond to impoundment differently, and most
406 importantly we must deal with the incommensurable challenge of developing CFs and ISs locally
407 or regionally but apply them globally with the same rigor and criteria.

408 Finally, our developed CF and ISs evaluated the impacts of hydroelectricity production in
409 storage reservoirs and only on the aquatic ecosystem's biodiversity (affected area; river and lakes
410 transformed into reservoirs) and not on the terrestrial area transformed into a reservoir. A
411 simplistic assumption could consider a loss of 100% of the impounded terrestrial habitat and a
412 gain of 100% aquatic habitat. The biodiversity impact on the flooded area is very relevant issue,
413 and some promising work have been done in this respect to model net land occupation of
414 reservoir in Norway (Dorber et al., 2018).

415 **5. Conclusions**

416 By using empirical data on the rate of change in fish richness over time, with data before
417 and after impoundment, on more than 89 sampling stations located upstream and downstream of
418 the dam, and belonging to 26 reservoirs across three large biomes, this study is the first to
419 propose robust and empirically developed characterization factors and impact scores of the
420 effects of hydroelectricity production on aquatic biodiversity. Our results suggest that the impact
421 of hydroelectricity production on fish richness is significant in tropical reservoirs, marginal in
422 temperate and not significant in boreal reservoirs which calls for regionalization in LCA. Our
423 results also demonstrated that the calculation of PDFs, and consequently ISs, was sensitive to the
424 time it takes to reach the steady state for fish communities. A steady state scenario of 10 years
425 after impoundment was the most plausible scenario based on the examination of PDFs at the
426 sampling station and reservoir scales. Finally, PDFs and ISs were relatively robust to upscaling
427 and downscaling issues (*i.e.*, patterns were consistent in their directionality across sampling

428 stations, reservoirs and biomes), but the statistical significance of the impact changed across
429 scales. Hydropower can be part of the solution to decarbonize our global economy but will come
430 at substantially higher ecological cost to the tropics (Pelicice et al., 2017; Winemiller et al.,
431 2016; Ziv et al., 2012).

432

6. Funding

The authors disclosed receipt of the following financial support for the research: This work was supported by a MITACS elevate grant to KT [Funding Request Ref. FR09321-FR09323].

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